

Forest fire induced impacts on phosphorus, nitrogen, and chlorophyll *a* concentrations in boreal subarctic lakes of northern Alberta

P. McEachern, E.E. Prepas, J.J. Gibson, and W.P. Dinsmore

Abstract: The biogeochemistry of 10 headwater lakes in burnt peatland–conifer catchments and 14 in unburnt catchments was evaluated throughout a summer 2 years following forest fire in a boreal subarctic region of northern Alberta. Cation exchange within burnt catchments resulted in proton flux and a 9% reduction in mean pH. Lakes in burnt catchments contained more than twofold higher ($P \ll 0.01$) mean concentrations of total, total dissolved, and soluble reactive phosphorus, 1.5-fold higher ($P \ll 0.01$) concentrations of dissolved organic carbon, and more than 1.2-fold higher ($P < 0.05$) concentrations of total and total dissolved nitrogen, nitrate + nitrite, and ammonium compared with reference lakes. Total phosphorus concentration explained 86% of the variance in reference lake chlorophyll concentration but was not related to chlorophyll concentration in burnt lakes. Analysis of chlorophyll – total phosphorus residuals suggested that algae in burn-impacted lakes were light limited. With the addition of five lakes burnt between 1961 and 1985, time since disturbance and percent disturbance combined explained 74% of the variance in total phosphorus among burnt lakes. Fire caused increased flux of materials to the study lakes with slow recovery over decades.

Résumé : Nous avons étudié la biogéochimie de 10 lacs de tête situés dans des bassins incendiés de tourbières et de conifères, et de 14 dans des bassins non incendiés, tout au long d'un été deux ans après un feu de forêt survenu dans une région boréale subarctique du nord de l'Alberta. L'échange cationique dans les bassins incendiés occasionnait un flux de protons et une réduction de 9% du pH moyen. Les lacs des bassins incendiés présentaient des concentrations moyennes plus de 2 fois supérieures ($P \ll 0,01$) de phosphore total, de phosphore dissous total et de phosphore réactif soluble, des concentrations 1,5 fois supérieures de carbone organique ($P \ll 0,01$) et des concentrations plus de 1,2 fois supérieures ($P < 0,05$) d'azote total et d'azote dissous total, de nitrate/nitrite et d'ammonium par rapport aux lacs témoins. La concentration de phosphore total (PT) expliquait 86% de la variance de la concentration de chlorophylle (CHL) dans les lacs témoins, mais n'était pas reliée à la CHL dans les lacs des bassins incendiés. L'analyse des concentrations résiduelles de CHL–PT semble indiquer que les algues des lacs touchés par l'incendie subissaient une réduction de l'éclaircissement. Si l'on ajoute les données sur 5 lacs de zones incendiées entre 1961 et 1985, la durée écoulée depuis l'incendie et le pourcentage de perturbation combinés expliquent 74% de la variance du PT entre les lacs des bassins incendiés. Le feu causait une augmentation de l'apport de matériaux dans les lacs étudiés, dont le rétablissement va être lent et s'étaler sur des décennies.

[Traduit par la Rédaction]

Introduction

Interest in fire as a natural disturbance impacting catchment biogeochemistry and surface water quality is rekindling as a relative benchmark for anthropogenic impacts. However, studies on the effects of fire on water quality suggest that impacts are regionally or fire specific due to organic soil characteristics, severity of fire, and hydrologic regime. The magnitude of ion flux from catchments is thought to depend 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 experimental fires where a third or less of the organic soil layer was burnt, no detectable changes in soil water and stream water carbon, nitrogen, phosphorus, their inorganic fractions, or major cations were reported (e.g., Richter et al. 1982). Following more severe fires, increases in nutrient flux were large but usually short-lived (<5 years), and changes in surface water concentrations did not exceed interannual variation (McCull and Grigal 1975; Bayley et al. 1992; Minshall et al. 1997). These studies occurred primarily in regions with thin organic soils, whereas studies in burnt permafrost–peatland dominated systems such as those in northern Alberta are rare.

Water chemistry has been studied in burnt catchments with small proportions of peatlands. A comparison of burnt and unburnt portions of a *Sphagnum fallax* – black spruce (*Picea mariana*) mire demonstrated no detectable long-term (8 years) impact of fire on water chemistry (Vitt and Bayley 1984). In a review of circumpolar studies, MacLean et al. (1983) reported that fire rarely burnt the entire organic layer and changes in soil water chemistry did not result in nutrient

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flux from peat-derived soils. However, wetland catchments had elevated phosphorus flux after severe fires (Bayley et al. 1992). The previous studies give little indication of potential impacts from forest fire in catchments with deep peat soils (0.5 m or more), underlain by glacial till and containing permafrost or seasonal frost lasting a majority of the summer. A lack of data makes it difficult to address the potential impacts of large-scale forest disturbance proposed for catchments of northern Alberta, particularly those containing permafrost where forestry and petroleum activities are increasing.

A unique opportunity to examine extensive damage to peatlands was provided when 129 000 ha of the Caribou Mountains, a subarctic plateau, were razed by fire in 1995. In this single event, one third of the plateau was burnt, equaling 50% of the mean annual area burnt in the province of Alberta between 1994 and 1998 (Alberta Environment, unpublished data). Impacts on lake water chemistry were expected because of high fire severity, a large proportion of catchments burnt (between 60 and 100%), and hydrology dominated by flow through peatlands. We hypothesized that the 1995 fire in the Caribou Mountains would reduce the base cation exchange capacity of peat and increase mineralization of nutrients to produce (i) increased base cation and nutrient concentrations in surface waters, (ii) corresponding increases in phytoplankton biomass, and (iii) increased suspended organic and inorganic seston. We also examined long-term (decades) impacts from fire and relationships between catchment characteristics and lake water chemistry.

Materials and methods

The Caribou Mountains (59°N, 115°W) in Alberta, Canada, are an erosional remnant forming a large, relatively flat plateau 500 m above the Peace River valley. Peatlands cover 56% of the Caribou Mountains and are predominately underlain by poorly drained cryosolic and brunisolic soils (Strong and Leggat 1992). Subsurface geology is shale, feldspathic sandstone, and siltstone of deltaic and marine origin. Open forest of black spruce and an understory of mixed *Sphagnum* spp., feathermosses, brown mosses, and lichens dominate the vegetation. Prior to the 1995 fire, most spruce stands in burnt and reference catchments originated between 1860 and 1910 (Alberta Vegetation Inventory 1 : 20 000 map series, 1983), indicating that there had not been a stand-replacing fire in 90 years. Trembling aspen (*Populus tremuloides*) is concentrated in upland areas. Mean percent upland per catchment is less than 30% but is 83% in one burnt catchment (Table 1). Permafrost is estimated to be between 0.5 and 0.75 m beneath the organic soil surface within continental bogs (Strong and Leggat 1992), which cover an average of 62% of the study catchments. Collapse scars signify degraded permafrost (Vitt et al. 1994) and were present in bogs averaging an additional 19% of the study catchments prior to the fire. In the fire, understory herbaceous, sedge, and lichen cover were incinerated but damage to moss and peat varied. Impact from burning was mostly limited to 0–20 cm of peat. In some locations, burning continued within deeper peat until the following summer. Late August testing in five catchments revealed no permafrost in the peat layer (down to mineral soil) of burnt areas, whereas it was present in five unburnt continental bogs. Mean May–August temperatures were 10.2°C (automated climate station), typically with less than 800 growing degree-days and between 400 and 450 mm of annual precipitation (Strong and Leggat 1992).

Ten headwater lakes were selected with catchments where between 50 and 100% of the tree cover was killed by fire (mean 83%, median 90%, SD 15%). These, plus 14 headwater lakes in unburnt reference catchments and five headwater lakes in catchments burnt between 1961 and 1985, were sampled monthly after ice-out from late June to late August – early September (Fig. 1). The lakes selected from previously burnt catchments were used in time since disturbance and percent disturbance analyses only. Among all combined lakes, differences in surface elevation were less than 90 m across the 120-km distance of this study on the plateau. The study lakes had zero to three neighbours within 20 km contributing the same drainage. Lakes within each of these drainage groups were within 15 m surface elevation of their neighbours. Lake surface areas ranged from 2.6 to 1173 ha and mean depths from 0.3 to 11.6 m. All lakes were either polymictic or, if stratification was observed (late July for deeper lakes), mixed conditions were restored by the next sampling event (August–September). In all lakes, dissolved oxygen concentrations exceeded 5 mg·L⁻¹ 0.5 m above the bottom through the summer. Three stream, four fen, and six bog sites were also sampled during the summer of 1997. Two streams, MS1 and MS2, drained adjacent unburnt watersheds of 40 and 6 km², respectively. MS3 drained a 1-km² watershed that was entirely burnt. Thus, the watershed ratios were 40:6:1 for the three streams. MS2 was gauged for discharge calculations. Samples were collected from MS1, MS2, and MS3 on the same four sample dates, which included bankfull and baseflow conditions. Our limited data show that the ratio of discharge from MS3 relative to the reference streams was 20:1.5:1. Based on this limited data set, the burnt catchment, MS3, released between two- and four-fold more catchment-weighted discharge than the two unburnt catchments.

Ground cover in each catchment was classified as upland–aspen, peatland, or open water. Peatlands were subdivided into veneer bog, peat plateau, poor fen, and rich fen based on vegetation and slope characteristics identified from 1994 1 : 20 000 aerial photographs (Halsey et al. 1997). Percentage of fire disturbance per drainage basin was estimated from 1996 aerial photographs. Bathymetric maps were constructed from depth measurements along 5–15 transects on each lake. Depth measurements were recorded by echosounding at equal time intervals along transects while traveling at a constant velocity and used to interpolate depth contours with ArcInfo. Catchment slope was calculated by computing elevation gain divided by linear distance to lakeshore (CS1: D'Arcy and Carignan 1997) with 10–30 transects from topographic high points and intermediate saddles around the watershed.

At the deepest site in each lake, water temperature and dissolved oxygen (YSI 50B) and light penetration (LI-COR LI-185 and LI-192SB flat sensors) profiles were recorded at 0.5-m intervals. Transparency was estimated with a 20-cm Secchi disc. A 12-L euphotic zone composite water sample was collected with multiple hauls of a polyvinyl chloride 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. pH of the composite sample was measured with a Hanna HI9025 meter calibrated before each use.

Triplicate 25-mL aliquots were pipetted into 30-mL glass tubes used directly in digestion processes for total and dissolved nitrogen (TN, DN) and phosphorus (TP, DP) analysis. Unfiltered water was used for TN, TP, and ammonium (NH₄⁺-N) analyses. Filtrate (Millipore HA, 45 µm) was used for DN, DP, and nitrate + nitrite (NO₃⁻-N) analyses. TN and DN samples were preserved with 10 µL of 40% H₂SO₄ for storage (1–2 weeks) and 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 and Rigler 1982). TN and DN concentrations were determined by second-derivative spectroscopic analysis of persulfate-oxidized samples (Crompton et

Table 1. Physical parameters for the Caribou Mountain study lakes.

Lake	W_o (ha)	A_o (ha)	Volume (10^3 m ³)	Z_m (m)	Slope (%)	Upland (%)	Peatland (%)	Disturbance (%)	τ (days)
Reference lakes									
C1	663.2	52.3	510	1.07	1.5	1	94	0	75
C2	614.8	93.4	1 910	2.17	2.3	2	95	0	217
C5	1068	76.5	568	0.75	2.4	6	79	0	80
C6	260.4	91.1	261	0.30	3.1	16	79	0	55
C7	569.0	90.0	578	0.66	2.9	6	87	0	61
C8	8208	60.8	612	0.99	1.4	5	85	0	49
C9	146.3	86.1	1 563	1.83	2.6	13	86	0	551
C11	204.8	134.4	4 071	3.06	6.1	65	35	0	887
C30	3218	536.1	55 564	11.60	2.0	28	67	0	770
C32	3018	953.4	27 318	2.90	3.0	12	81	0	314
C34	5571	166.5	5 840	3.50	4.0	22	77	0	194
C35W	825.0	175.0	18 240	5.50	6.1	32	66	0	419
C45	4627	1173	93 840	8.00	4.0	14	83	0	660
C47	2732	59.0	697	1.18	6.0	72	25	0	56
Lakes in catchments burnt in 1995									
C17	700.9	164.8	2 300	1.54	1.9	83	16	80	105
C23	1053	65.8	2 436	4.10	1.9	19	80	90	32
C24	1761	159.8	10 332	6.36	3.6	35	62	95	42
C25	820.1	151.9	4 278	2.87	4.2	25	72	90	171
C26	1520	36.6	523	1.68	3.3	25	73	75	36
C27	581.1	28.1	220	0.85	6.0	30	69	60	43
C41	32.6	2.6	22	0.86	2.0	43	57	100	68
C42	34.3	7.4	53	0.72	3.0	51	49	98	96
C43	691.9	8.3	50	0.60	4.0	51	45	95	26
C46	359.4	4.4	25	0.58	6.0	44	51	60	44
Lakes in catchments burnt between 1961 and 1985									
C12	642.9	40.5	440	1.22	2.9	6	88	20	101
C13	483.8	175.5	1 745	1.04	1.3	12	81	80	133
C14	268.9	62.4	1 779	2.93	4.3	27	73	80	298
C15	2257	128.3	2 013	1.67	2.8	6	91	70	77
C16	995.1	53.4	1 056	2.04	4.2	42	55	80	129

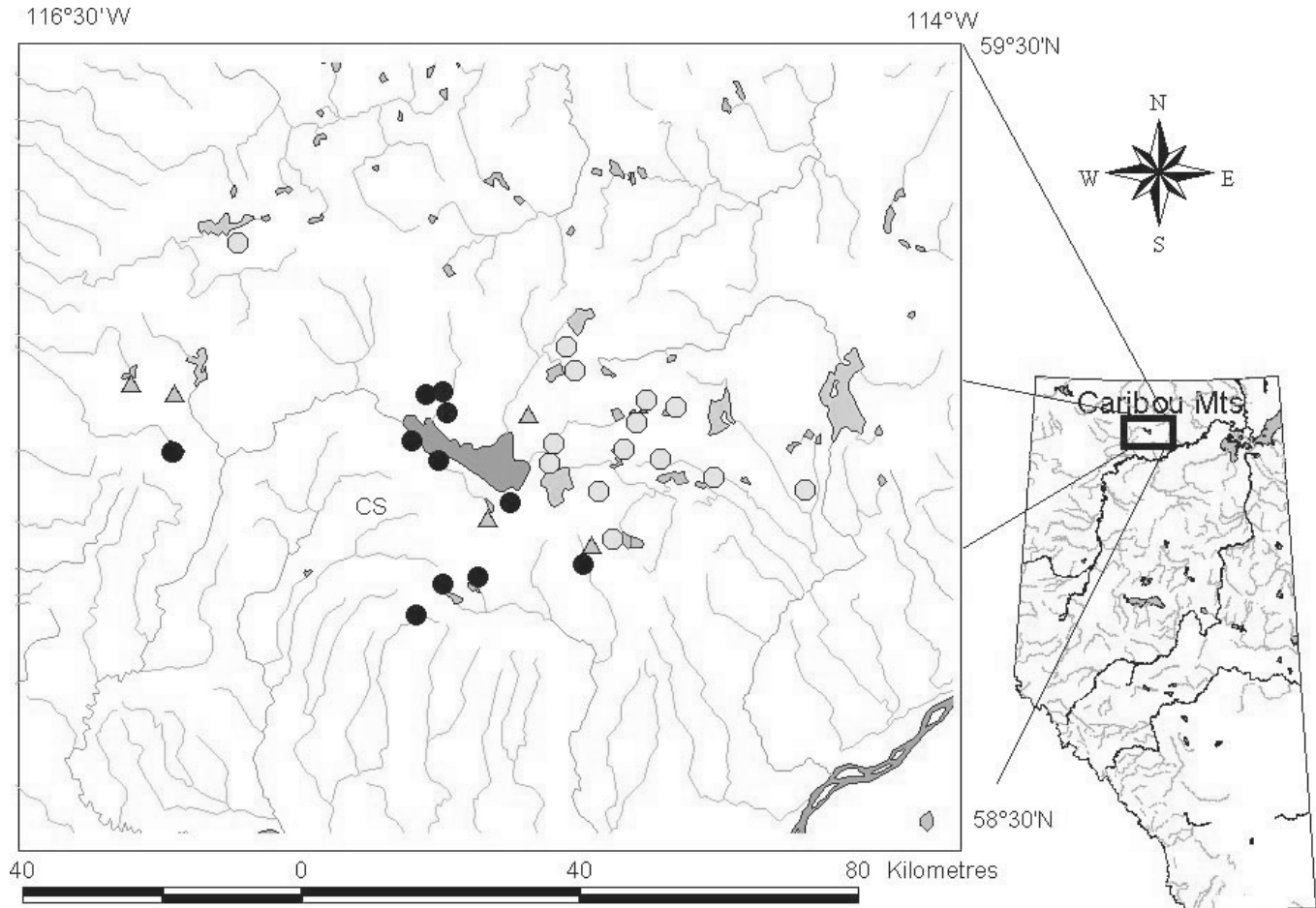
Note: W_o , watershed area; A_o , lake surface area; Z_m , lake mean depth; τ , residence time. Slope, upland, and peatland are percentages for each catchment and disturbance is the percentage impacted by forest fire.

al. 1992). Soluble reactive phosphorus (SRP) concentrations were measured by molybdate blue absorption from membrane filtrate within 24 h of collection. NH_4^+ -N and NO_3^- -N samples were frozen in 125-mL polyethylene bottles for analysis with indophenol blue (NH_4^+ -N) and cadmium reduction (NO_3^- -N) with automated colorimetry (Technicon methods 100-700 W/B and 155-71W). Total suspended solids, nonvolatile suspended solids (NVSS), and volatile suspended solids (VSS) were collected on preashed GF/F filters (mean particle retention 0.7 μm) and analyzed in duplicate after American Public Health Association (1993). Total and bicarbonate (HCO_3^-) alkalinities were estimated with the Hach phenolphthalein/bromocresol method (American Public Health Association 1993). Filtrate from GF/F filters for analysis of cations (Ca^{2+} , Mg^{2+} , Na^+ , K^+), dissolved organic carbon (DOC), and anions (Cl^- , SO_4^{2-}) were stored in 60-mL polyethylene bottles. Cation samples were preserved with 60 μL of 40% H_2SO_4 , while DOC and anion samples were refrigerated. Cations were analyzed by atomic absorption flame spectroscopy (Perkin-Elmer AS90 and AA3300), DOC 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 desiccated with silica-gel and frozen. Total chlorophyll *a* (CHL) was extracted with 90% ethanol (Sartory and Grobbelaar 1986) and concentrations were determined with fluorometric methods (Knowlton 1984).

Summer mean (June – early September) values are used in all analyses. Data are presented as box and whisker plots where the box outlines 25–75% of the data and whiskers denote 10–90% of the data. The line within each box represents the median value. pH values were converted to $[\text{H}^+]$ prior to averaging and analysis and converted back to pH for reporting. Data were tested for deviation from a normal distribution using Kolmogorov–Smirnov at $P = 0.05$. Data were either normally distributed or were \log_{10} transformed to meet assumptions for normality and were analyzed with *t* test, Pearson's correlation, and univariate least-squares regression. Probability values and correlation coefficients are reported for these tests. We specify when nonnormal data were analyzed with nonparametric tests (Mann–Whitney *U*). ANCOVA was used to test differences in linear regressions between burnt and reference systems. We used discriminant analysis to determine if reference and burn-impacted lakes were different in measured catchment and lake physical characteristics. Percent open water could not be in-

Fig. 1. Location of Caribou Mountain study lakes in Alberta, Canada. Reference sites (gray circles) contain lakes with no fire in their catchments since approximately 1910, previously burnt sites (triangles) contain lakes with catchments burnt between 1961 and 1985, and burnt sites (black circles) contain lakes in catchments burnt in 1995. An automated climate station (CS) operated for 2 years.



cluded in the discriminant analysis because of zero values. Differences were deemed significant at $P = 0.05$. All analyses used Statistica 4.1 and StatView 4.5 for Macintosh.

Results

Fire impacts on water chemistry

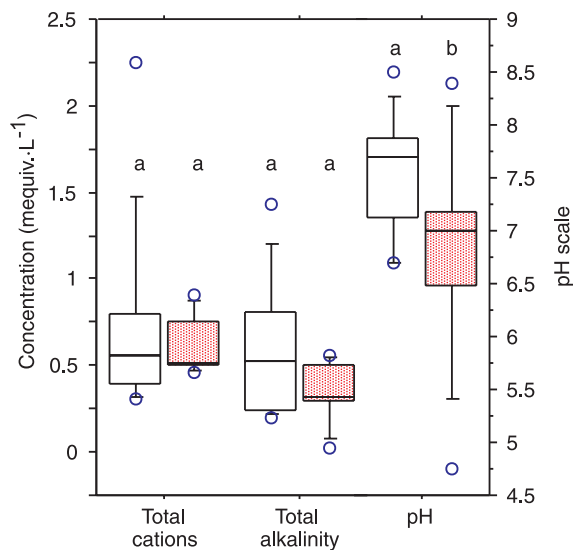
Several physical characteristics of lakes and their catchments differed between burnt and reference systems (Table 1). Reference lakes tended towards larger surface areas (Mann–Whitney, $P = 0.02$), lake volumes (Mann–Whitney, $P = 0.05$), and percent open water cover per catchment (Mann–Whitney, $P = 0.01$). Percent open water averaged 5 and 1% of reference and burnt catchments, respectively. The small proportion of open water and its concentration in collapse scars, areas of degraded permafrost, make it unlikely that it contributed to chemical differences in the lakes. Drainage ratios (watershed area (W_o)/lake surface area (A_o)) and A_o to volume (V) ratios, considered more important in determining land–water linkage than W_o or A_o alone, were not different between burn-impacted and reference lakes ($P = 0.6$ and 0.3 , respectively). Slope was also not different between burn-impacted and reference catchments ($P = 0.3$). Discriminant function analysis, containing drainage ratio,

A_o/V , catchment slope, and percent upland contained one root and was not significant (Wilk's $\lambda = 0.75$, canonical $r = 0.5$, $P = 0.3$). Likewise, substituting A_o , W_o , and V for W_o/V and W_o/A_o did not produce a significant discriminant function ($P = 0.2$). Therefore, it is unlikely that chemical differences observed between reference and burn-impacted lakes resulted from physical characteristics alone.

Total base cation concentrations (Ca^{2+} , Mg^{2+} , Na^+ , K^+) of Caribou Mountain lake water (median $0.52 \text{ mequiv}\cdot\text{L}^{-1}$) were all below the world average for freshwater ($2.33 \text{ mequiv}\cdot\text{L}^{-1}$). There were no detectable differences in mean total base cation concentrations (Fig. 2) between burnt ($0.61 \text{ mequiv}\cdot\text{L}^{-1}$) and reference ($0.72 \text{ mequiv}\cdot\text{L}^{-1}$) lakes ($P = 0.53$). As a percentage of total cations, Ca^{2+} was lower and K^+ was higher in burnt compared with reference lakes ($P < 0.05$).

Fire affected lake acidity and ion balances, possibly through increases in organic anions in burnt lakes. Lake water in burnt catchments was moderately acidic relative to reference lake water (mean pH = 6.9 and 7.6, respectively, $P = 0.04$). Burnt lakes demonstrated low total alkalinity compared with reference lakes; however, differences in means were not detectable (16.7 and $30.3 \text{ mg CaCO}_3\cdot\text{L}^{-1}$, $P = 0.07$) (Fig. 2). Sulfate concentrations were elevated in burnt lakes (Mann–Whitney, $P = 0.01$) compared with reference lakes,

Fig. 2. Summer mean total base cations, total alkalinity, and pH for lake waters in reference (open boxes, $n = 14$) and burnt catchments (shaded boxes, $n = 10$). Letters above the boxes indicate differences between reference and burn-impacted lakes ($P < 0.05$).



with median concentrations of 2.4 and 1.5 $\text{mg}\cdot\text{L}^{-1}$, respectively. Fire did not affect chloride concentrations, which averaged 0.15 and 0.13 $\text{mg}\cdot\text{L}^{-1}$ for burnt and reference lake water, respectively ($P = 0.3$). Mean total anions (SO_4^{2-} , Cl^- , HCO_3^- , NO_3^-) balanced mean total cations in reference lake water (mean = 0.723 $\text{mequiv}\cdot\text{L}^{-1}$, $P = 1.0$). All lakes in burnt catchments had a deficit in total anions (0.46 $\text{mequiv}\cdot\text{L}^{-1}$, $P < 0.01$). The average deficit of 25% below mean total base cations suggested a large pool of organic acids in burnt lakes. Most *Sphagnum*-derived acids contain $-\text{COOH}$ groups, which act as anions in solution while contributing protons (Clymo 1984). Anion deficits were linearly correlated with DOC in burnt ($r^2 = 0.84$, $P < 0.01$, $n = 10$) and less so in reference lakes ($r^2 = 0.26$, $P = 0.06$, $n = 14$). Fire apparently increased organic anion concentrations along a relationship with DOC that also existed in reference lakes.

Lake water in burnt catchments had elevated phosphorus and nitrogen concentrations. In reference lakes, mean TP, DP, and SRP concentrations were 33, 14, and 4 $\mu\text{g}\cdot\text{L}^{-1}$, respectively (Fig. 3). Lake water in burnt catchments had 2.6-, 3.2-, and 6.8-fold higher TP, DP, and SRP concentrations, respectively, than reference lakes ($P < 0.01$). In reference lakes, mean TN, DN, NO_3^- -N, and NH_4^+ -N were 655, 488, 3, and 15 $\mu\text{g}\cdot\text{L}^{-1}$, respectively (Fig. 4). Burn-impacted lake water contained 1.2-fold higher DN ($P = 0.02$), threefold higher NO_3^- -N ($P = 0.04$), and 1.4-fold higher NO_3^- -N (Mann-Whitney, $P = 0.03$) concentrations. Mean TP and DP in the two reference streams (MS1 and MS2) were 48 and 28 $\mu\text{g}\cdot\text{L}^{-1}$, 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 $\mu\text{g}\cdot\text{L}^{-1}$, respectively, while MS3 contained 2.2-fold higher TN and 1.2-fold higher DN concentrations. Increased export of phosphorus and nitrogen from burnt relative to reference watersheds was likely given the two- to four-fold higher catchment-weighted discharge from MS3.

Caribou Mountain water had high concentrations of DOC,

Fig. 3. Summer mean concentrations of TP, DP, and SRP for reference (open boxes, $n = 14$), burnt (shaded boxes, $n = 10$) and previously burn-impacted lakes (hatched boxes, $n = 5$). Letters above the boxes indicate differences between reference and burnt means ($P < 0.05$). Analytical statistics were not applied to previously burnt lakes.

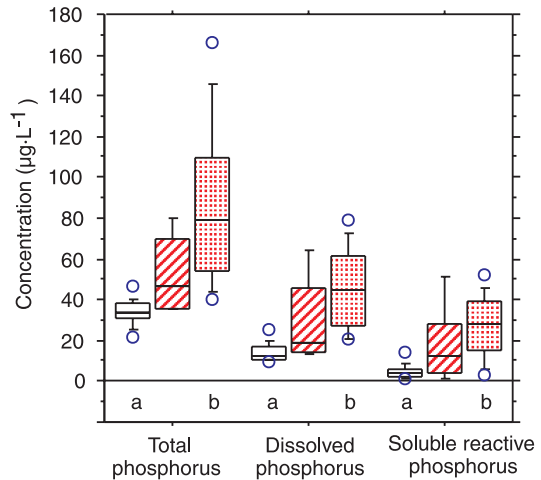
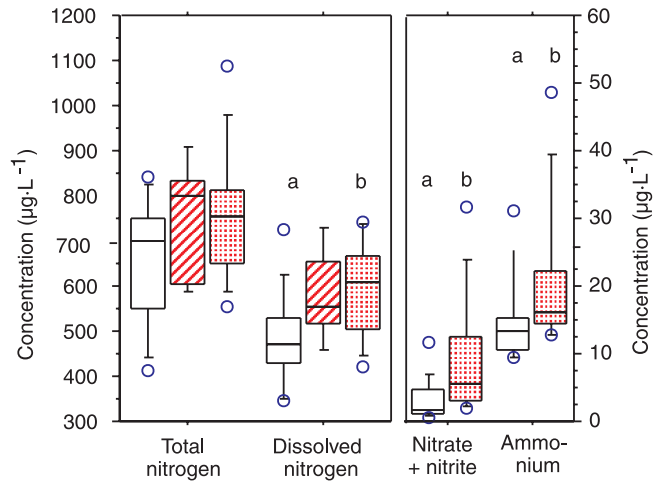


Fig. 4. Summer mean TN and DN for reference (open boxes, $n = 14$), burnt (shaded boxes, $n = 10$), and previously burn-impacted lakes (hatched boxes, $n = 5$). NO_3^- -N and NH_4^+ -N are for reference and burnt lakes only. Letters above the boxes indicate differences in mean concentrations between reference and burnt lakes ($P = 0.05$).



particularly in burnt systems (Table 2). Lake mean DOC concentration in burn-impacted lake water (25 $\text{mg}\cdot\text{L}^{-1}$) was 1.6-fold higher ($P < 0.01$) than in reference lakes (16 $\text{mg}\cdot\text{L}^{-1}$). Further, in the reference streams MS1 and MS2, DOC concentrations averaged 33 and 37 $\text{mg}\cdot\text{L}^{-1}$, respectively. In MS3, which drains a burnt area, DOC averaged 28 $\text{mg}\cdot\text{L}^{-1}$. The three sampled fens contained almost identical DOC concentrations (45, 45, and 48 $\text{mg}\cdot\text{L}^{-1}$) despite receiving drainage from burnt and unburnt areas. MS3 exported three and two times more DOC per unit catchment area than MS2 and MS1, respectively, given the two- to four-fold higher weighted discharge from MS3.

Caribou Mountain water was highly coloured, especially

Table 2. Summary of chemical characteristics for the Caribou Mountain study lakes.

Lake	CHL ($\mu\text{g}\cdot\text{L}^{-1}$)	TP ($\mu\text{g}\cdot\text{L}^{-1}$)	DP ($\mu\text{g}\cdot\text{L}^{-1}$)	SRP ($\mu\text{g}\cdot\text{L}^{-1}$)	TN ($\mu\text{g}\cdot\text{L}^{-1}$)	DN ($\mu\text{g}\cdot\text{L}^{-1}$)	NO_3^- -N ($\mu\text{g}\cdot\text{L}^{-1}$)	NH_4^+ -N ($\mu\text{g}\cdot\text{L}^{-1}$)	DOC ($\text{mg}\cdot\text{L}^{-1}$)	DIC ($\text{mg}\cdot\text{L}^{-1}$)	VSS ($\text{mg}\cdot\text{L}^{-1}$)
C1	9.87	31	16	6	609	514	2.6	17.8	22	6.3	1.35
C2	16.58	38	20	9	744	527	11.6	31.2	18	5.8	3.49
C5	12.17	34	11	6	688	452	1.1	10.1	17	5.4	3.21
C6	14.51	36	10	5	823	474	3.2	13.3	12	4.7	4.63
C7	24.94	46	17	2	756	470	1.4	10.6	17	4.3	4.64
C8	4.68	40	25	14	661	612	5.0	13.6	27	7.6	1.57
C9	9.42	32	13	3	729	528	4.6	9.3	12	7.6	4.28
C11	6.79	31	11	2	750	585	1.2	15.4	12	14.8	1.73
C30	4.74	22	9	3	412	349	7.0	14.4	13	8.3	0.80
C32	7.73	32	12	2	445	345	1.0	10.9	11	3.5	2.20
C34	6.66	26	13	1	550	466	1.7	11.5	18	11.5	1.40
C35W	22.72	38	12	4	710	429	0.6	25.2	16	11.2	3.60
C45	9.22	26	11	1	454	361	4.9	12.2	10	7.7	1.80
C47	12.59	35	17	5	840	723	1.1	13.5	27	10.6	1.76
C17	39.23	125	66	36	1086	561	31.8	48.6	18	6.7	5.92
C23	5.94	79	55	40	552	468	6.1	12.9	22	6.3	1.70
C24	10.67	78	62	40	622	505	16.3	18.7	19	6.4	1.71
C25	24.31	55	21	9	751	421	2.6	12.7	14	6.2	4.16
C26	5.15	109	47	32	718	636	5.7	14.4	30	7.6	1.46
C27	3.89	98	43	24	649	583	12.6	16.1	27	8.4	1.12
C41	11.77	48	28	15	871	741	3.0	16.1	32	9.6	1.78
C42	5.97	40	21	4	757	668	5.6	22.2	21	14.6	1.59
C43	20.05	166	79	52	811	664	1.9	30.0	31	7.1	5.51
C46	7.26	54	33	17	779	734	4.2	15.5	35	8.9	2.02
C12	11.59	36	19	12	809	630	20.3	18.3	23	6.8	2.76
C13	19.74	46	14	2	799	534	3.8	70.9	14	6.5	4.27
C14	9.03	36	15	5	588	456	11.4	91.7	12	12.0	1.42
C15	4.89	80	64	51	607	556	9.0	19.9	27	6.3	1.09
C16	14.29	67	40	21	907	728	42.4	76.2	27	5.4	2.85

Note: Values are summer means. TC – TA is total cations minus total anions.

within burnt catchments. Reference lake water mean colour was $151 \text{ mg Pt}\cdot\text{L}^{-1}$ and ranged from 26 to $388 \text{ mg Pt}\cdot\text{L}^{-1}$, comparable with lake water colour in the boreal mixedwood ecoregion to the south, where colour ranged from 8 to $358 \text{ mg Pt}\cdot\text{L}^{-1}$ (E.E. Prepas, unpublished data). Mean colour in burn-impacted lake waters ($342 \text{ mg Pt}\cdot\text{L}^{-1}$) was 2.3-fold higher ($P < 0.01$) than in reference lakes. Colour was related to DOC in all lake waters (Table 3). Variance in light penetration was closely associated with colour of lake water; the natural logarithm of Secchi transparencies was negatively related to colour, while light extinction coefficients were positively related to colour (Table 3). Mean Secchi transparencies in burn-impacted lakes (0.73 m) were 54% of those in reference lakes ($P < 0.01$), while the mean light extinction coefficient ($0.787\cdot\text{m}^{-1}$) was 1.7-fold higher in burn-impacted lakes ($P < 0.01$) compared with reference lakes. Secchi depth and light extinction were correlated ($r^2 = 0.36$, $P < 0.01$, $n = 24$); when lakes C24 and C45 were removed, the relationship was stronger ($r^2 = 0.85$, $P < 0.01$, $n = 22$). Lake C24 had a high extinction coefficient ($1.28\cdot\text{m}^{-1}$) for its Secchi depth (1.0 m) due to the continuous surface bloom of *Aphanizomenon* sp. as flakes. Lake C45 had a high extinction ($0.54\cdot\text{m}^{-1}$) for its Secchi depth (2.8 m) due to wind conditions during sampling. Elevated DOC and colour

in burn-impacted lakes reduced transparency and likely enhanced the potential for light limitation of phytoplankton growth after fire.

Fire affected inorganic suspended solids concentrations. Mean NVSS in reference lakes was $0.65 \text{ mg}\cdot\text{L}^{-1}$, whereas burnt lakes contained twofold higher NVSS (Mann–Whitney, $P = 0.04$). VSS concentrations were not different between reference and burnt lakes, averaging 2.6 and $2.7 \text{ mg}\cdot\text{L}^{-1}$, respectively ($P = 0.88$). Increased transport of inorganic particles from burnt catchments is inferred.

Fire did not appear to affect lake water CHL concentrations. Among reference lakes, mean CHL was $12 \mu\text{g}\cdot\text{L}^{-1}$ and \log_{10} -transformed values were strongly related to TP ($r^2 = 0.83$, $P < 0.01$, $n = 13$) (Fig. 5). Despite the much higher phosphorus concentrations observed in burn-impacted lake waters, CHL was not elevated, averaging $13 \mu\text{g}\cdot\text{L}^{-1}$. There was no detectable relationship between CHL and TP concentrations for burn-impacted lakes ($r^2 = 0.06$, $P = 0.5$). 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 ($r^2 = 0.44$, $P < 0.01$, $n = 10$) and to the TN:TP ratio ($r^2 = 0.69$, $P < 0.01$, $n = 10$). The lack of response in CHL to elevated phosphorus in burn-impacted lake

NVSS (mg·L ⁻¹)	Ca ²⁺ (mg·L ⁻¹)	Mg ²⁺ (mg·L ⁻¹)	Na ⁺ (mg·L ⁻¹)	K ⁺ (mg·L ⁻¹)	SO ₄ ²⁻ (mg·L ⁻¹)	Alkalinity (mg CaCO ₃ ·L ⁻¹)	Colour (mg Pt·L ⁻¹)	TC – TA (mequiv·L ⁻¹)	pH
0.34	4.40	1.20	0.83	0.08	1.35	9.75	317	-0.1307	6.70
0.37	5.24	1.33	0.28	0.25	1.13	15.00	209	-0.0623	7.70
0.42	4.37	0.95	0.32	0.17	0.70	12.25	172	-0.0505	7.29
0.43	5.65	1.18	0.42	0.31	0.95	19.75	76	0.0128	7.86
1.13	4.12	0.94	0.41	0.22	0.82	11.25	205	-0.0601	7.08
0.91	7.30	1.59	0.37	0.09	1.55	15.25	389	-0.1755	6.69
0.83	8.06	1.91	0.54	0.43	1.05	32.50	34	0.0823	7.88
0.36	18.89	4.17	2.00	0.52	7.69	71.75	26	0.2147	8.49
1.00	10.40	2.00	0.60	0.40	2.30	36.75	105	0.0687	7.47
0.40	6.00	1.40	0.40	0.30	2.10	10.88	77	-0.1759	7.00
1.00	14.60	2.90	0.90	0.40	2.50	54.38	140	0.1287	7.82
0.90	11.00	2.40	0.90	0.50	4.20	40.50	112	0.1053	8.27
0.60	8.50	2.30	0.50	0.30	1.40	35.25	55	0.0965	7.77
0.41	27.01	7.67	6.01	0.49	48.73	59.00	196	-0.0542	7.68
1.84	6.46	1.62	0.85	0.62	2.71	16.50	240	-0.1192	8.38
2.28	7.18	1.35	0.47	0.57	1.63	16.25	331	-0.1395	7.06
0.84	6.04	1.36	0.57	0.82	2.66	15.50	250	-0.0886	6.94
1.09	7.84	2.27	0.53	0.79	2.70	27.75	112	-0.0052	7.96
1.02	12.07	3.06	0.82	0.50	7.04	25.25	480	-0.2479	7.10
2.87	10.24	2.38	0.91	0.34	1.96	26.50	396	-0.1819	7.17
0.51	5.37	1.78	0.69	1.11	9.51	1.13	424	-0.2474	4.75
0.30	5.98	1.85	0.74	0.70	5.43	16.00	212	-0.0638	6.61
1.51	6.54	1.59	1.20	0.61	7.96	6.75	456	-0.2174	6.06
0.82	10.20	2.68	2.76	0.24	16.62	15.00	523	-0.2079	6.48
0.65	7.35	1.58	0.63	0.05	1.23	16.25	312	-0.1728	7.20
0.62	6.71	1.94	1.30	0.25	1.54	26.50	112	0.0070	7.55
0.64	14.28	3.76	1.26	0.63	5.12	54.75	87	0.1132	7.93
0.81	5.67	1.47	0.81	0.33	3.49	10.00	454	-0.1741	6.60
0.89	7.91	1.98	0.51	0.29	5.34	12.75	344	-0.2161	7.26

Table 3. Univariate regressions for independent (X) and dependant (Y) variables.

X	Y	n	Intercept	Slope	r ²	SE of slope	MS error	F	P
DOC (mg·L ⁻¹)	Colour (mg Pt·L ⁻¹)	24	-156	19.3	0.89	1.44	2 595	175	<0.0001
Colour	ln Secchi (m)	24	0.62	-0.003	0.72	0.0004	0.066	53.6	<0.0001
Colour	ε (m ⁻¹)	24	0.32	0.001	0.58	0.0002	0.026	30.9	<0.0001
log ₁₀ W _o /A _o	DOC (mg·L ⁻¹)	24	8.7	10.7	0.62	1.78	21.4	36.3	<0.0001
Burnt		10	10.1	12.2	0.69	2.87	17.2	17.9	0.003
Reference		14	9.04	8.0	0.68	1.60	10.8	25.1	0.0003
log ₁₀ W _o /A _o	Colour (mg Pt·L ⁻¹)	24	9.17	210	0.57	38.9	10 205	29.2	<0.0001
Burnt		10	41.0	248	0.78	45.9	4 377	29.3	0.0006
Reference		14	18.4	141	0.59	34.8	5 131	16.4	0.0016
DI	log ₁₀ TP (mg·L ⁻¹)	15	1.52	0.005	0.74	0.001	0.038	40.58	<0.0001
DI	log ₁₀ DP (mg·L ⁻¹)	15	1.13	0.007	0.76	0.001	0.059	43.74	<0.0001

Note: ε, light extinction coefficient; W_o/A_o, drainage ratio.

waters and the relationship with Secchi depth are consistent with light limitation.

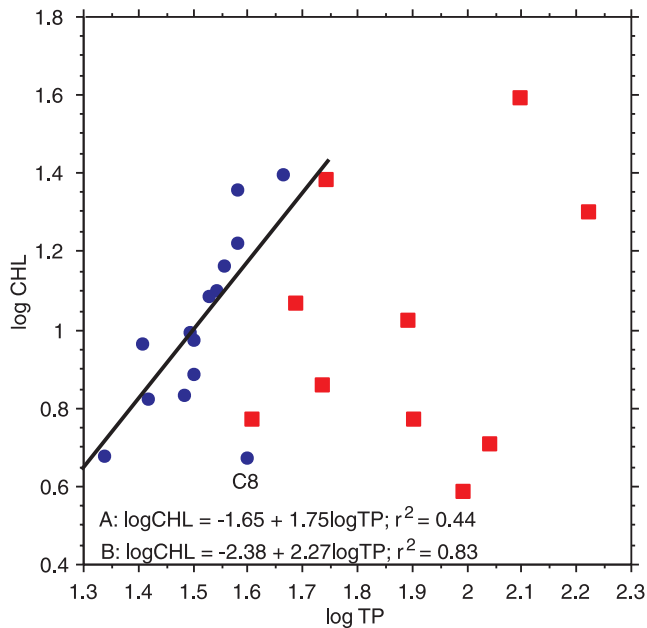
General patterns relating lake chemistry to catchment features were poor. There were no detectable relationships between nitrogen or phosphorus concentrations of lake waters with drainage ratio ($r^2 < 0.25$, $P > 0.3$). However, colour and DOC were linearly related to log drainage ratio (Table 3). Colour and DOC tended to increase at a faster rate with drainage ratio in burnt lakes compared with reference lakes (Table 3);

however, the slopes of these relationships were not distinguishable (ANCOVA, $P = 0.09$ and 0.20 , respectively).

Percent disturbance and time since disturbance

Water in the previously (1961–1985) burnt lakes had phosphorus concentrations intermediate between those of recently (1995) and nonburnt lakes. TP and DP concentrations for burnt lakes increased in a positive-linear relationship with percent disturbance (intercept forced through mean TP

Fig. 5. Relationship between CHL and TP (both $\mu\text{g}\cdot\text{L}^{-1}$) for lakes in reference (circles) and burnt catchments (squares). Values are \log_{10} transformed. A: equation and line for all reference lakes only; B: equation without C8, which deviates from other reference lakes because of its high colour (388 $\text{mg Pt}\cdot\text{L}^{-1}$).



and DP for reference lakes). TP and DP appeared to decline with the natural logarithm of time since disturbance. Data were not normally distributed in time or percent disturbance, so we combined both hypothesized disturbance patterns into a single disturbance index (DI):

$$\text{DI} = \% \text{ disturbance} \times e^{-Kt}$$

where t was time since disturbance (90 years for reference lakes) and K was a decay constant. The value for K was predicted from an exponential decay function $K = 0.693 \cdot T^{-1}$, where T was the half-life, which was estimated from the relationship between $\log_{10}\text{DP}$ and TP and the natural logarithm of time since disturbance. The T was estimated at 11 and 20 years for DP and TP, respectively. The DI explained 74 and 76% ($P < 0.01$) of the variance in TP and DP, respectively, among the 15 impacted lakes (Table 3). We included reference conditions in the analysis by forcing the intercept through mean TP and DP concentrations for reference lakes (intercept value from Table 3) assuming that they represented a zero DI.

Discussion

Catchments in the Caribou Mountains are fundamentally different from those where fire effects have historically been studied. The large proportion of inundated peatlands in the Caribou Mountains created unique nutrient responses following fire. For example, forest fire usually augments NO_3^- -N and, to a lesser degree, phosphorus export from granitic regions (Lewis 1974; McColl and Grigal 1975; Wright 1976). The relatively severe Caribou Mountains fire caused elevated phosphorus and, to a lesser degree, nitrogen concentrations in lakes, which suggested higher phosphorus

compared with nitrogen export following fire. Similar findings are reported from other wetland systems. Burnt fens exported phosphorus and retained nitrogen, while burnt upland catchments exported nitrogen and held phosphorus in the granitic Experimental Lakes Area (Bayley et al. 1992). Nitrogen retention in Alberta peatlands typically exceeds 98% (Li and Vitt 1997), a value comparable with the greater than 85% retention reported by Bayley 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 upland fire studies because of reduced phosphorus retention in peatlands after fire.

Increase in the flux of divalent cations and potassium from organic soils usually occurs after fire (Tiedemann 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 likely remained intact, liberating protons as cations were exchanged (Clymo 1984). The result was a fivefold median increase in $[\text{H}^+]$, decreased pH, and reduced alkalinity in burnt lakes, while base cation concentrations did not change. Organic acids associated with elevated DOC in burn-impacted lakes possibly added to overall acidification while contributing organic anions (presumably R-COO^-).

The relationship between CHL and TP in reference lakes on the plateau indicates a strong association between phytoplankton biomass and TP. The slope for \log_{10} -transformed values (2.31) was more positive and the intercept (-2.46) more negative than values reported for North America (Nurnberg 1996).

A general model for the movement of water to lakes and streams in the discontinuous permafrost region of boreal Alberta may include groundwater systems or flow through organic peat. Surface runoff is minimal, even during intense storms, because the infiltration capacity of surface moss and peat (acrotelm) is about $1 \text{ cm}\cdot\text{s}^{-1}$, declining to 10^{-4} to $10^{-6} \text{ cm}\cdot\text{s}^{-1}$ in anaerobic peat (catotelm) layers (Ingram 1983). Groundwater could discharge through streambeds and lakes themselves, as they appear to be the only places without permafrost, other than the hydrologically disconnected collapse scars. Inputs from the peat layer should be restricted to a small effective contributing area when the water table is limited to the catotelm. Rain events that raise the water table to the acrotelm should cause a substantial increase in effective contributing area and loading of peat-derived water due to higher hydraulic conductivity in the acrotelm. Macropores, typically occurring at the peat-mineral interface (e.g., Hill 1993), may also be important in delivering water high in DOC and protons. However, water discharge from fens likely had a greater influence on surface water chemistry than that from bogs by virtue of their higher discharge rates (Halsey et al. 1997). Groundwater in the Caribou Mountains had base cation concentrations 20-fold higher than found in lakes (P. McEachern, unpublished data). The similarity between fen and lake water chemistry and the high cation concentrations of groundwater suggest that groundwater discharge to lakes was minimal and restricted to discharge through fens.

Time since disturbance and percent disturbance were important factors in nutrient enrichment among burnt lakes.

When combined into a DI, they explained 74 and 76% of the variance in lake TP and DP concentrations, respectively. An exponential decay model for time since disturbance impacts was used because it matched the trend for our limited temporal data. The hypothesis is justifiable because recovery rates should initially be rapid as new growth and microbial communities are reestablished followed by a decrease with increasing time as microbial and plant communities stabilize and nutrients are flushed from the lake and lost to sediments.

Our results suggest that forest fire had a profound impact on surface water quality in the 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 were not retained by peat. Although base cations were likely liberated from burnt material, cation exchange with peat resulted in an increase in the flux of protons. Elevated phosphorus concentrations slowly returned to reference conditions depending on both the magnitude of disturbance (percent disturbance) and the time since disturbance. A single DI combining percent disturbance and time proved effective; however, a larger lake set spanning a range in time is needed to test the hypothesized exponential and linear components of this model. The effects of fire on water chemistry may be larger in peatland-dominated catchments than elsewhere, due to elevated export of phosphorus and long recovery periods.

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