Comparative impacts of fire and forest harvesting on water quality in Boreal Shield lakes

Richard Carignan, Pierre D'Arcy, and Sébastien Lamontagne

Abstract: Water quality was monitored in Boreal Shield lakes for 3 years following their simultaneous impact by clearcut logging or wildfire. Seventeen similar undisturbed lakes served as references. Dissolved organic carbon (DOC) and the light attenuation coefficient (ε_{PAR}) were up to threefold higher in cut lakes than in reference and burnt lakes. Compared with median values for reference lakes, cut and burnt lakes had higher concentrations of total phosphorus (TP) (two- to three-fold), total organic nitrogen (TON) (twofold), and K^+, Cl^-, and Ca^{2+} (up to sixfold). NO_3^- and SO_4^{2-} concentrations were up to 60- and 6-fold higher, respectively, in burnt lakes than in reference and cut lakes. In most cases, impacts were directly proportional to the area harvested or burnt divided by the lake's volume or area. These simple models correctly predicted the changes observed in three lakes harvested during the study. Some of the observed effects occur on different time scales. Mobile ions released by fire (K^+, Cl^-, SO_4^{2-}, NO_3^-) or harvesting (K^+, Cl^-, some DOC) are rapidly flushed out of the watershed (50% decrease in 3 years). Other constituents or properties (TP, TON, DOC, ε_{PAR}, Ca^{2+}, Mg^{2+}) show little change or are still increasing after 3 years and will take a longer time to reach normal levels.

Résumé : Nous avons mesuré la qualité des eaux dans plusieurs lacs du bouclier canadien pendant trois années suivant leur impact par la coupe à blanc ou par des feux de forêt. Dix-sept lacs vierges semblables servirent de témoins. Les concentrations en carbone organique dissous (COD) et le coefficient d'atténuation lumineuse (ε_{PAR}) étaient deux à trois fois plus élevés dans les lacs coupés que dans les lacs brûlés. Par rapport aux valeurs médianes observées dans les lacs témoins, les lacs coupés et brûlés avaient des concentrations de deux à trois fois plus élevées en phosphore total (PT), deux fois plus élevées en azote organique total (NOT) et jusqu'à six fois plus élevées en K^+, Cl^- et Ca^{2+}. Le NO_3^- et le SO_4^{2-} étaient respectivement 60 et six fois plus élevés dans les lacs brûlés que dans les lacs coupés et les témoins. Dans les lacs brûlés et coupés, la majorité des impacts étaient directement proportionnels au rapport entre la superficie déboisée et la superficie ou le volume du lac récepteur. Ces modèles simples ont prédit correctement les effets observés dans les lacs déboisés pendant l'étude. Les effets observés se déroulent sur des échelles temporelles différentes. Les ions mobiles minéralisés par le feu (K^+, Cl^-, SO_4^{2-}, NO_3^-) ou relâchés après coupe (K^+, Cl^-, du COD) ainsi qu'une partie du COD sont rapidement (50% en 3 ans) lessivés du bassin-versant. Les autres substances ou propriétés (PT, NOT, DOC, ε_{PAR}, Ca^{2+}, Mg^{2+}) ont peu changé ou augmentent encore après 3 ans et prendront plus de temps à revenir à des valeurs normales.

Introduction

Aquatic ecosystems of the Canadian boreal forest are influenced by recurring (50–300 years) wildfires that annually cover from less than 0.1 to 1.0% of the land surface (Natural Resources Canada 1996). During the last century, forest harvesting has increased to such an extent that it now exceeds fire as a disturbance agent in large tracts of the boreal forest, with an annual removal rate reaching 1% in some regions. Boreal forest exploitation is currently shifting towards natural disturbance based models (Hunter 1991, 1993; Franklin 1993). These models propose that fire and logging have similar impacts on ecosystems and that forest exploitation patterns emulating fire should preserve ecosystem integrity. These important assumptions remain largely unverified, however, for both terrestrial and aquatic ecosystems.

Received September 2, 1999. Accepted May 11, 2000.
J15345

R. Carignan,[1] P. D'Arcy, and S. Lamontagne. Département de sciences biologiques, Université de Montréal, C.P. 6128, succ. Centre-Ville, Montréal, QC H3C 3J7, Canada.

[1]Author to whom all correspondence should be addressed. e-mail: carignar@ere.umontreal.ca

Given the diversity in vegetation, landforms, and geology occurring throughout the deciduous, mixed, and boreal forest biomes, relatively few studies have documented the effects of these disturbances on the chemistry and biology of surface waters. Most have described the effects of fire or clearcutting on stream chemistry or hydrology (e.g., Likens et al. 1970; Nicolson et al. 1982; Bayley et al. 1992), while a minority of studies have focused on lakes (Wright 1976; Rask et al. 1993, 1998; Garcia and Carignan 1999). Although coherent models have yet to emerge regarding the impact of either disturbance on surface waters, previous studies have established that the loss of forest cover by fire and clearcuts increases, to varying degrees, runoff and the export of particulate matter, nutrients, major ions, and colour. Few studies were designed, however, to compare the effects of forest removal by fire and clear-cutting or to provide quantitative impact models linking the type and extent of watershed disturbance to water quality in lakes.

During the summer of 1995, four major wildfires totalling

1620 km^2 occurred in a portion of Québec's boreal forest undergoing extensive harvesting operations. The large number of lakes (~800) simultaneously impacted by fire or harvest provided a unique opportunity to compare and to model the effects of both disturbances on water quality and on biotic communities in similar lakes (see Patoine et al. 2000; Planas et al. 2000; St-Onge and Magnan 2000). The present report compares some of the chemical and physical changes that occurred in the lakes during the first 3 years after impact. A companion paper (Lamontagne et al. 2000) compares element export from the catchments of these lakes.

Methods and materials

Study region

The lakes are located in a 30 000-km^2 area surrounding Gouin Reservoir in Haute-Mauricie, Québec (48°50′N, 75°00′W) (Fig. 1). The region intersects the Grenville and Superior geological provinces of the Canadian Shield. The three dominant rock formations encountered are a gneissic complex rich in quartz, plagioclase, biotite, and horneblende, a granodioritic gneiss, and a mixture of intrusive granodiorites and granites in the Superior province, which crosses the northeast quadrant of the region. The low (40–150 m net elevation) rolling hills are covered by glacial deposits that are generally thin (0–2 m) but that can locally exceed 10 m thick. The forest is mainly composed of black spruce (*Picea mariana*), balsam fir (*Abies balsamea*), jack pine (*Pinus divaricata*), white birch (*Betula papyrifera*), and trembling aspen (*Populus tremuloides*). Wetlands cover ~10% of the region, which annually receives ~ 1000 mm of precipitation (40% as snow). Runoff was estimated at 562, 694, and 490 mm in 1996, 1997, and 1998, respectively (Lamontagne et al. 2000). Runoff was high in 1997 and low in 1998 compared with its long-term average of 586 mm.

The fire-impacted lakes are located in two of the four large 1995 wildfire areas, Parent (645 km^2) and Belleplage (406 km^2) (Fig. 1), where fire severity ranged from medium to high. Most of the fire-impacted lakes described below are located in catchments where complete loss of foliage occurred, often with more than 40% of the boles fallen and partially consumed. Less severe crown fires leaving some intact vegetation occurred in 50 and 25% of the catchments of lakes FP2 and FP30, respectively. Little (<5%) or no salvage logging occurred in the watersheds of the study lakes. Clearcuts were of the CPRS type (cut with protection of regeneration and soils). Cut block size generally ranged from 5 to 50 ha but occasionally reached 200 ha, with 20-m buffer strips fringing permanent streams, lakes, and wetlands.

Lake selection

Thirty-eight thermally stratified headwater lakes were selected on the basis of comparable size, basin morphometry, and catchment properties. The lakes formed four groups: 16 reference lakes ("N" lakes), nine burnt lakes, which had most of their catchment burnt in 1995 ("FP" and "FBP" lakes), nine cut lakes, which had 9–72% of their catchment logged in 1995 ("C" lakes), and four lakes that had 5–24% of their catchment logged during the study ("P" lakes). The following criteria were used in the selection process: forest age exceeding 50 years, <6% of catchment occupied by wetlands, drainage ratio (DR) between 2 and 15, hydrological order near 1, maximum depth exceeding 5 m, fetch longer than 1 km, perturbation by fire or logging in 1995, and no salvage logging on burnt catchments.

All retained lakes met these conditions, with the following exceptions. Lakes C43 and N120 had some 40- and 45-year-old tree stands in their catchments. Note that stands aged >70 years on Québec's vegetation maps can be considerably older. Lake order is not always exactly 1, as 17 lakes have smaller lakes in their drainage areas. These smaller lakes represent, on average, only 14% of the surface area of the downstream study lakes. Lake FP15 (48 ha) is an exception and has a much larger (154 ha) headwater lake. Logging operations continued after 1995 in the catchment of Lake C2 where the fraction cleared increased from 31 to 72% from 1995 to 1998. One lake (P107) slated for logging during the study had only 5% of its catchment harvested in 1997–1998 and was treated as a reference lake.

Lake morphometry and catchment properties

Bathymetric maps were established from aerial photographs and from 15–25 echosounder transects. Provincial vegetation and soils maps (1 : 20 000) and a digitizing table were used to measure vegetation composition and age and to estimate the type and thickness of superficial deposits. Provincial topographic maps (1 : 20 000) were used to delineate watersheds and to measure wetland cover. The topography of each drainage area was digitized to produce 10-m-resolution digital elevation models and to measure mean catchment slope (Arc/Info, ESRI). Water residence times were not corrected for increased runoff due to loss of forest cover.

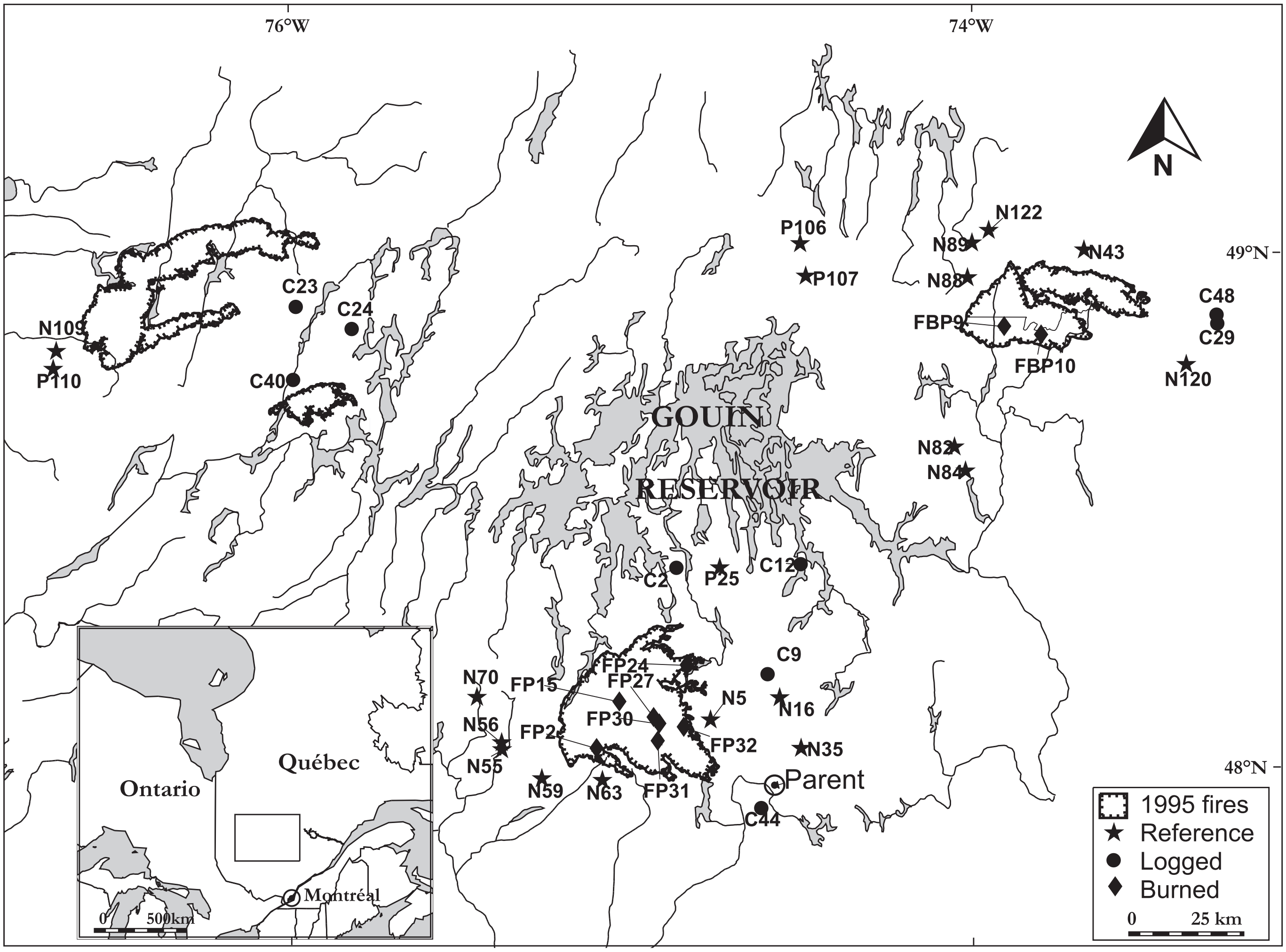
Sampling and chemistry

The lakes were sampled three times each year during the ice-free season (a few days after ice-out in late May or early June, in mid-July, and in mid-September). The penetration of light available to photosynthesis (photosynthetically active radiation (PAR), 400–700 nm) and the PAR extinction coefficient (ε_{PAR}) were measured with Li-Cor LI-190 and LI-192SA sensors. Integrated water samples were taken from the euphotic zone (1% of incident PAR), filtered within 12 h when necessary, and stored at 4°C until analyzed (within 72 h). Samples for dissolved organic carbon (DOC) measurements and occasional colour (absorbance at 440 nm) measurements were filtered on washed Gelman Supor 0.45-μm membranes and kept at 4°C. DOC was measured (Shimadzu TOC-5000) within 72 h by infrared gas analysis after sample acidification and He sparging followed by Pt-catalyzed oxidation at 700°C. In these lakes, DOC concentrations measured with Supor 0.45-μm membranes are identical, within measurement precision, to those obtained with precombusted GF/C filters. Total phosphorus (TP) was measured using the molybdenum blue method (Stainton et al. 1977) after autoclaving 50-mL samples with 0.5 g of potassium persulfate for 1 h at 120°C. NO_3^- and NH_4^+ were measured on filtered samples by automated flow injection analysis (Lachat methods 10-107-04-1-B and 10-107-06-1-F). Total nitrogen (TN) was measured as NO_3^- after alkaline persulfate digestion of 50-mL samples at 120°C (D'Elia et al. 1977). TN was not measured in June and July of 1997. The ions Cl^-, SO_4^{2-}, Ca^{2+}, Mg^{2+}, Na^+, and K^+ were measured by ion chromatography (Dionex DX-500). Alkalinity was measured by Gran titration and pH was measured in the laboratory with a double-junction electrode on air-equilibrated samples.

Data analysis

The significance of between-group (reference, cut, and burnt lakes) differences for a given year was assessed using one-way analysis of variance (ANOVA) with a nonparametric Kruskal–Wallis test when distributions were significantly skewed. Repeated-measures ANOVA (RMA, SAS version 6.12) was used to detect significant differences between seasons and years for individual and paired treatments (reference-cut, reference-burnt). Linear regressions were performed on normalized ($\log_{10}$ transformed) data when the significance of relationships obtained with untransformed data depended on a few extreme observations. A stepwise forward variable selection procedure with an F ratio of 4 was used for multiple regressions. The Kruskal–Wallis test was used to verify that regression residuals were normally distributed. Unless otherwise noted, significance tests and regression parameters are significant

Fig. 1. Map of the study region showing the four 1995 fires and the location of the reference, harvested, and burnt lakes.



at the $p < 0.05$ level. Variable definitions and units are summarized in Table 1.

Results and discussion

Lake morphometry and catchment properties

Several key limnological variables such as DOC, light penetration, and TP are likely to be influenced by morphometric and catchment properties such as the DR, the water residence time, and catchment slope (Schindler 1971; Engstrom 1987; Rasmussen et al. 1989). The four groups of lakes do not differ significantly (ANOVA, $p > 0.05$) with respect to all properties listed in Table 1, with the following exceptions: lakes harvested in 1995 (C lakes) are, on average, located 20 km further north than reference lakes (N lakes), reference lakes tend to be shallower (4.3 versus 5.8 m) than burnt lakes (F lakes), and burnt lakes had less conifers (51.2% of catchment area (CA)) than the three other groups (65.3–74%). The significant difference in mean depth between reference and burnt lakes is due to the inclusion of one particularly deep lake (FBP27, mean depth (ZM) = 10 m) in the group. Note that although the four groups of lakes do not differ significantly with respect to DR, reference lakes have a lower median DR (5.2) than cut (6.8) and burnt (8.7) lakes.

The study lakes are representative of a large number of lakes in the area. A systematic inventory of the 1549 lakes occurring in four quadrants totalling 3350 km^2 located within the study region shows that 30% of the water surface area is composed of lakes having the same range of lake area (LA) as our study lakes (R. Carignan, unpublished data). Our lakes are probably biased towards low DR, however, since lakes occupy 7.5% of the surface area, which implies an average DR of 12.3.

Mean annual chemical properties and ε_{PAR} for reference, cut, and burnt lakes are compared between groups and between years in Fig. 2. In order to increase the homogeneity of the cut lakes group, Lakes C44 and C40, which had only 9 and 11% of their drainage area harvested, respectively, are excluded from these analyses. Of all variables measured, only pH and alkalinity did not show significant differences between groups (Fig. 2, pH and NH_4^+ not shown). In the 38 lakes, pH ranged from 5.7 to 7.2, with a median of 6.6 in 1996. The box plots in Fig. 2 present a useful but somewhat biased overview of the major differences and similarities observed between reference, cut, and burnt lakes. Cut lakes had a smaller average proportion (58%) of their drainage areas cleared compared with burnt lakes (91%). The effects of harvesting and fire shown in Fig. 2 are therefore not directly comparable.

Table 1. Watershed characteristics and morphometry of the study lakes.

	Reference (*n* =17)				Harvested (1995, *n* = 9)				Harvested (1996–1997, *n* = 4)				Burnt (1995, *n* = 9)			
Variable	Median	Mean	Min.	Max.	Median	Mean	Min.	Max.	Median	Mean	Min.	Max.	Median	Mean	Min.	Max.
Latitude (LAT)	48.31	48.40	47.93	49.00	48.71	48.52	47.87	48.85	48.82	48.74	48.34	48.97	48.05	48.22	47.99	48.81
Longitude (LONG)	74.57	74.67	73.41	76.71	74.65	74.8	73.32	76.02	74.66	75.14	74.52	76.73	74.95	74.73	73.83	75.13
Altitude (ALT)	442	453	395	558	422	440	394	556	430	434	391	485	468	460	422	497
Catchment area (CA, km^2)*	1.74	2.01	0.45	4.46	2.58	3.44	0.59	10.28	2.47	2.77	1.32	4.82	3.07	4.93	0.57	19.72
Lake area (LA, km^2)	0.35	0.43	0.15	0.81	0.32	0.56	0.18	2.31	0.43	0.50	0.33	0.81	0.36	0.40	0.17	0.64
Total lake are (ΣLA, km^2)	0.35	0.45	0.15	0.98	0.32	0.62	0.20	2.59	0.48	0.52	0.34	0.81	0.41	0.63	0.18	2.03
Watershed area (WA = CA + ΣLA)	2.20	2.46	0.61	4.81	2.89	4.06	0.90	12.87	2.90	3.30	1.76	5.64	3.42	5.55	0.75	21.75
Drainage ratio (DR = CA/ΣLA)	4.42	5.17	2.33	15.38	6.83	6.75	1.96	13.18	5.22	4.99	2.97	6.54	8.70	7.14	2.51	11.64
Mean depth (ZM, m)	4.1	4.3	2.1	8.5	4.6	5.7	4.6	8.9	4.6	4.6	4.6	4.6	5.3	5.8	4.2	10.0
Max. Depth (ZMAX, m)	11	12	7	21	12	14	5	30	12	14	10	23	15	17	10	34
Residence time (τ, years)	1.40	1.64	0.32	4.00	1.07	1.31	0.44	3.22	1.81	1.75	1.29	2.09	0.84	1.59	0.32	5.58
Catchment slope (SLO, degrees)	5.8	6.4	4.0	9.4	4.9	5.3	3.3	10.9	6.8	6.9	5.2	8.9	6.0	6.7	4.4	10.0
Area burnt (FIR, km^2)	0	0	0	0	0	0	0	0	0	0	0	0	**2.31**	**4.64**	**0.58**	19.51
Area logged (CUT, km^2)	0	0	0	0	**0.99**	**1.70**	**0.11**	**7.52**	**0.30**	**0.46**	**0.20**	**1.04**	0	0	0	0
% of CA burnt	0	0	0	0	0	0	0	0	0	0	0	0	**94.6**	**91.3**	**50.1**	**100.0**
% of CA logged	0	0	0	0	**48.3**	**47.0**	**8.5**	**73.2**	**19.9**	**17.4**	**5.9**	**24**	0	0	0	0
% of CA as wetlands	1.7	1.9	0.0	7.0	1.6	2.3	0.0	6.2	1.7	2.3	0.5	5.1	0.5	0.9	0.0	3.6
% of CA as conifer	69.8	65.3	32.3	79.3	75.8	74.0	52.3	87.5	66.3	67.7	52.2	86.1	**51.2**	**51.7**	**28.7**	**81.9**
% of CA as deciduous	27.1	28.8	11.2	65.6	17.3	21.4	12.5	47.2	31.6	29.8	13.9	42.0	**32.5**	**31.0**	**5.7**	**61.3**
Vegetation age (years)			40	>80			50	>90			50	70			50	70

Note: Attributes with significantly different between-group medians are shown in bold.
*The catchment is defined here as the terrestrial portion of the watershed.

Table 2. Linear regression equations.

No.	Lakes	Year	Model	r^2	SE
1	R	1996–1998	log DOC = – 0.61±0.07 + 0.35±0.10 log DR	0.44	0.09
2	R	1996–1998	log ε_{PAR} = – 0.30±0.06 + 0.42±0.08 log DR	0.65	0.07
3	R + C	1996	DOC = 0.29±0.03 + 0.083±0.017(CA/VOL) + 0.25± 0.03(CUT/VOL)	0.86	0.09
4	R + C	1997	DOC = 0.30±0.03 + 0.073±0.017(CA/VOL) + 0.22±0.03(CUT/VOL)	0.82	0.09
5	R + C	1998	DOC = 0.29±0.03 + 0.076±0.016(CA/VOL) + 0.17±0.03(CUT/VOL)	0.80	0.09
6	R + C	1996	ε_{PAR} = 0.60±0.06 + 0.20±0.03(CA/VOL) + 0.59±0.07(CUT/VOL)	0.90	0.18
7	R + C	1997	ε_{PAR} = 0.72±0.05 + 0.19±0.03(CA/VOL) + 0.49±0.06(CUT/VOL)	0.89	0.17
8	R + C	1998	ε_{PAR} = 0.67±0.06 + 0.21±0.04(CA/VOL) + 0.52±0.07(CUT/VOL)	0.87	0.20
9	R + C	1996	DOC = 0.27±0.05 + 0.027±0.009(DR) + 0.081±0.016(CUT/ΣLA)	0.70	0.13
10	R + C	1997	DOC = 0.29±0.05 + 0.022±0.008(DR) + 0.072±0.015(CUT/ΣLA)	0.66	0.13
11	R + C	1998	DOC = 0.27±0.05 + 0.023±0.008(DR) + 0.057±0.014(CUT/ΣLA)	0.63	0.12
12	R	1996–1998sp	log TP = –0.90±0.05 + 0.25±0.08 log DR – 0.22±0.07 log ΣLA	0.74	0.06
13	R + C	1996–1998sp	TP = 0.18±0.02 + 0.010±0.004(DR) + 0.026±0.007(CUT/ ΣLA)	0.57	0.06
14	R + C	1996–1998sp	TP = 0.19±0.02 + 0.032±0.009(CA/VOL) + 0.074±0.019(CUT/VOL)	0.70	0.05
15	R + F	1996–1998sp	TP = 0.17±0.02 + 0.014±0.003(DR) + 0.028±0.003(FIR/ ΣLA)	0.90	0.05
16	R + F	1996–1998sp	TP = 0.19±0.02 + 0.033±0.010(CA/VOL) + 0.138±0.014(FIR/VOL)	0.86	0.06
17	R + C + F	1996–1998sp	TP = 0.18±0.02 + 0.010±0.003(DR) + 0.031±0.003(CUT + FIR)/ΣLA	0.83	0.06
18	F	1996sp	log NO_3^- = –1.6±0.3 + 1.2±0.2 log(FIR/ΣLA) + 1.6±0.4 log SLO	0.90	0.15
19	R + C	1996	TON = 13.6±0.7 + 1.55±0.39(CA/VOL) + 2.6±0.8(CUT/VOL)	0.68	2.1
20	C	1996	K^+ = 5.5±2.9 + 3.3±1.0(CUT/ΣLA)	0.62	4.5
21	C	1996	Cl^- = 4.0±1.0 + 1.0±0.3(CUT/ΣLA)	0.58	1.6
22	F	1996	K^+ = 12.7±3.7 + 1.4±0.5(FIR/ΣLA)	0.51	4.9
23	F	1996	Cl^- = 6.8±1.7 + 0.61±0.24(FIR/ΣLA)	0.48	2.2
24	C	1996	Ca^{2+} = 31.0±2.2 + 1.9±0.7(CUT/ΣLA)	0.51	3.4
25	R	1996–1998	SO_4^{2-} = –329±90 + 4.7±1.2(LONG)	0.52	4.0
26	R	1996–1998	Alkalinity = 728±236 – 9.2±2.7(LONG)	0.44	9.8

Note: R, reference lakes; C, cut lakes; F, burnt lakes; sp, springtime values only.

DOC and light penetration

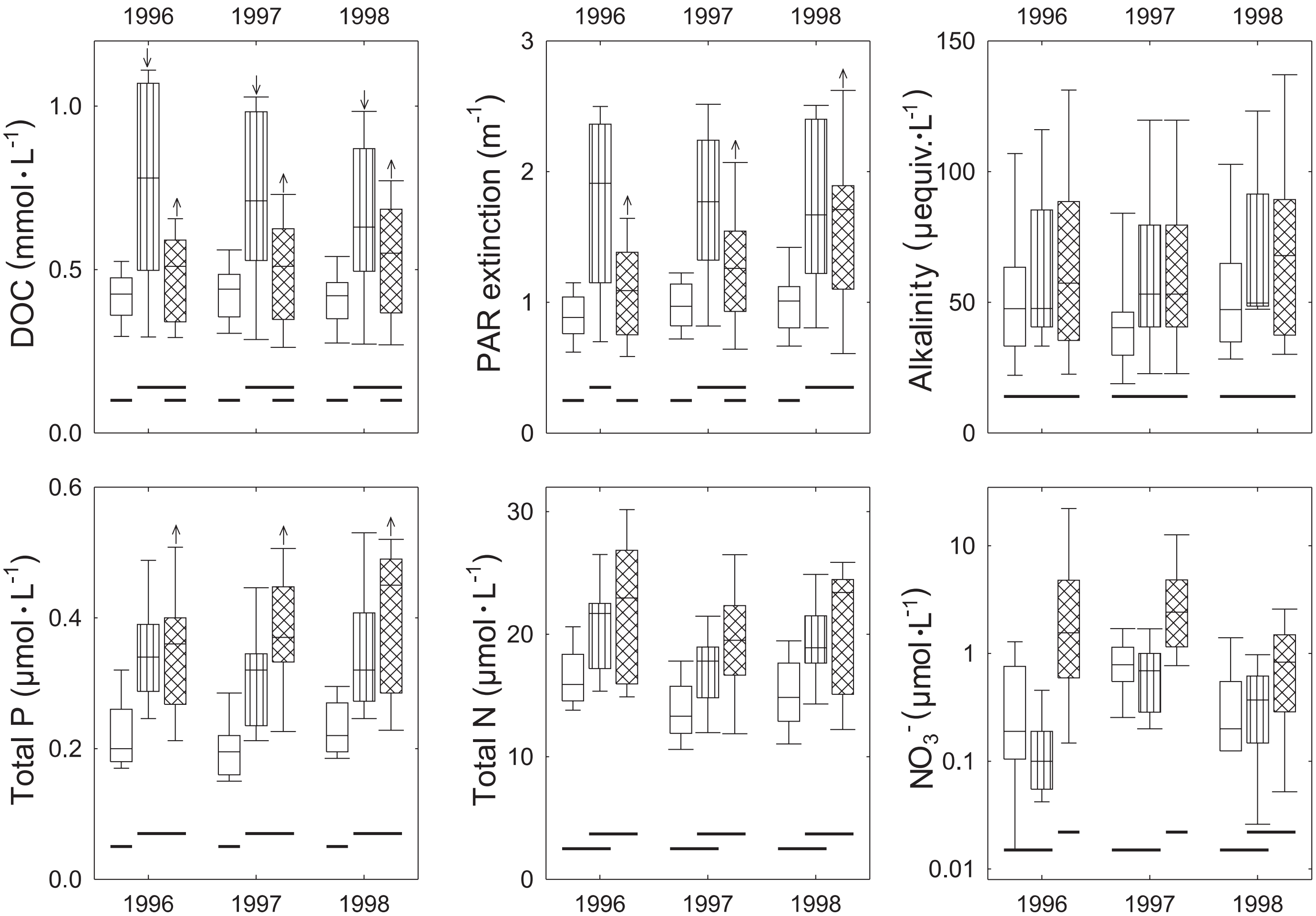
In reference lakes, log DOC (3-year averages) was significantly related (r^2 = 0.44) (Table 2, model 1) to log DR, as usually found (Schindler 1971; Engstrom 1987; Rasmussen et al. 1989). A similar but more accurate relationship was found between the 400- to 700-nm light extinction coefficient (log ε_{PAR}) and log DR (r^2 = 0.65) (model 2).

Compared with reference lakes, several cut lakes had higher DOC concentrations and ε_{PAR}, with DOC showing a significant decreasing trend with time (Figs. 2 and 3). DOC concentrations in burnt lakes were intermediate but did not differ significantly from those of reference and cut lakes. In contrast with cut lakes and compared with reference lakes, DOC and ε_{PAR} increased significantly with time in burnt lakes. DOC concentrations showed a significant (RMA, p < 0.05) seasonal pattern in the three groups of lakes, with midsummer values rising 5–15% above those of spring and late summer (Fig. 3). The ε_{PAR} was closely related to DOC in the 38 lakes (r^2 = 0.93). Small seasonal variations in DOC were not always reflected in ε_{PAR}, however, particularly in cut lakes (Fig. 3). Factors such as the bleaching of coloured DOC and seasonal variations in the light-absorbing properties of DOC entering the lakes may explain the slight uncoupling observed between ε_{PAR} and DOC. In reference lakes, the average ε_{PAR} was lowest (0.91) in 1996, increased significantly by 10% during the wettest year (1997), and decreased slightly in 1998.

The high DOC concentrations and ε_{PAR} values observed in cut lakes (Figs. 2 and 3) cannot be entirely attributed to their higher median DR (Table 1) compared with reference lakes. Inclusion of the cut lakes on a linear regression plot of DOC versus DR for reference lakes (Fig. 4) shows that Lakes C2, C12, C23, and C24 lie above the model's upper prediction limit (1996 values), whereas Lakes C9, C29, C40, C44, and C48 lie within the model's prediction limits. Lakes within the model's prediction limits either have a low DR or had only a small fraction of their drainage area harvested. Figure 4 suggests that DOC in cut lakes will significantly exceed the range of natural variability only when their DR exceeds about 4 and when more than about 30% of their drainage area has been cleared. The only exception to this rule is Lake C9, which has a DR value of 6.8, with 45% of CA harvested; this lake is deeper and has a longer residence time (1.1 years) than Lakes C2, C12, C23, and C24 (0.4–0.7 year). When ε_{PAR} is plotted as in Fig. 4, similar results are obtained except that this time, Lakes C2 and C48 lie well above the model's upper prediction limit.

Simple relationships exist between DOC, ε_{PAR} in lakes, and the extent of forest removal in their catchments. The best regression found between DOC and harvesting (1996 data) uses the CA/VOL ratio (square metres/cubic metres) plus a volumetric impact ratio, defined as the harvested surface area to lake volume ratio (CUT/VOL (square metres/cubic metres)) and explains 86% of the variability in DOC (Fig. 5; Table 2, model 3). The slope of the impact ratio decreased markedly in subsequent years, as DOC loading to these lakes decreased (models 4 and 5). Similar relationships are also found for ε_{PAR} (models 6–8; Fig. 5), with the important difference that the slope of these models does not decrease as rapidly with time. For management purposes, CUT/VOL can

Fig. 2. Box plots showing median values with the 25th and 75th and the 5th and 95th percentiles of annual average concentrations of dissolved constituents and of the light extinction coefficient ε_{PAR} in reference (open boxes, n = 17), cut (hatched boxes, n = 7), and burnt lakes (cross-hatched boxes, n = 9). The thick horizontal lines indicate between-treatment differences: for a given year, boxes with different levels are significantly different. Arrows on top of the boxes indicate significant increasing or decreasing trends with time compared with the reference group (paired RMA). Two lakes (C40 and C44) where only a small portion of the catchment had been harvested are excluded from these analyses.



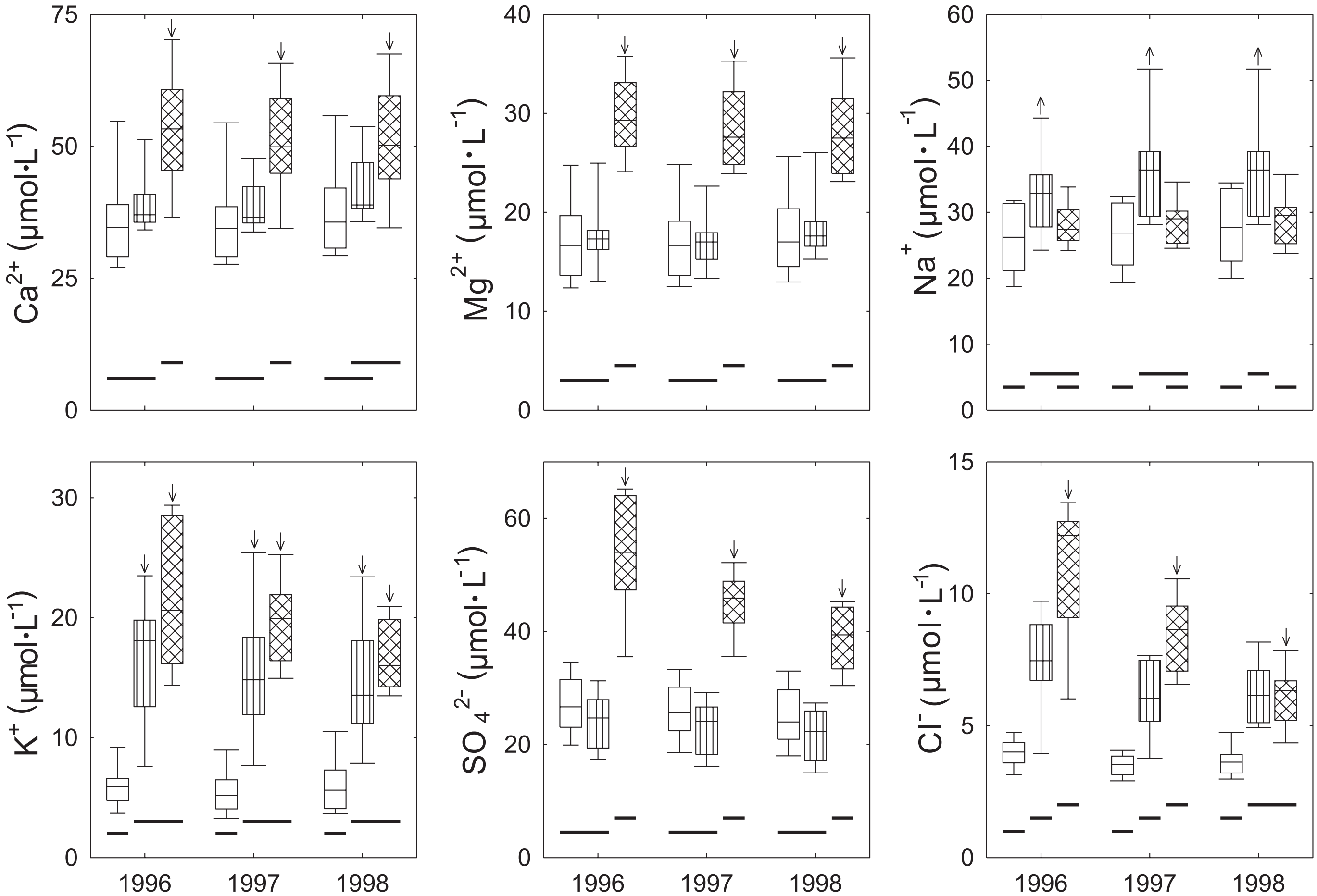
be substituted for a more easily obtained areal impact ratio (CUT/ΣLA) but with a loss of precision (models 9–11).

DOC and ε_{PAR} do not evolve at a similar rate in cut lakes. Between 1996 and 1998, DOC decreased in all cut lakes except Lake C9. Linear regression of DOC versus time (n = 9 data points) for individual lakes shows that the decrease is significant in Lakes C2 (–0.07 mmol·year^{-1}), C12 (–0.11 mmol·year^{-1}), C24 (–0.09 mmol·year^{-1}), and C29 (–0.015 mmol·year^{-1}). The decrease is also marginally significant in Lake C48 (–0.02 mmol·year^{-1}, p = 0.07). These decreases in DOC are not paralleled by decreases in ε_{PAR}. The only significant changes in ε_{PAR} were positive and occurred in Lakes C29 (+0.06 m^{-1}·year^{-1}) and C40 (+0.07 m^{-1}·year^{-1}). During the same interval, DOC changed significantly, by a much smaller amount (–0.006 mmol·year^{-1}), in only two of the 16 reference lakes.

Fire-impacted lakes differ from cut lakes in two ways in terms of DOC and ε_{PAR}. First, DOC concentrations in the fire-impacted group were not significantly different from those of reference lakes 1 year after impact (Figs. 2–4); second, they increased significantly (RMA) during the following years (Fig. 2). When individual lakes were examined by linear regression, DOC increased in all lakes except Lake FBP10, but the changes were significant only in Lakes FP2 and FP32. Likewise, ε_{PAR} was near reference levels in 1996 but increased significantly afterwards in all burnt lakes except Lakes FBP10 and FP27 (Figs. 2 and 3). By 1998, ε_{PAR} values in burnt lakes were approaching those of cut lakes. The best relationship between DOC or ε_{PAR} and watershed properties for burnt lakes included DR or VOL/WA (a surrogate for the residence time) as independent variables and explained 55–70% of the variance. By 1998, due to the increasing ε_{PAR} in burnt lakes, biological production may have been light limited in some lakes, since the midsummer epilimnion depth exceeded the compensation depth for net photosynthesis (1% incident PAR).

Several mechanisms may explain the contrasting DOC and ε_{PAR} responses of cut and burnt lakes to forest clearance. The increase in annual runoff following the loss of forest cover in Québec's boreal forest has been estimated at about 200 mm (Plamondon 1993), or 40% of the long-term runoff at our study site. In eastern Canada, this high runoff

Fig. 2 (*concluded*).

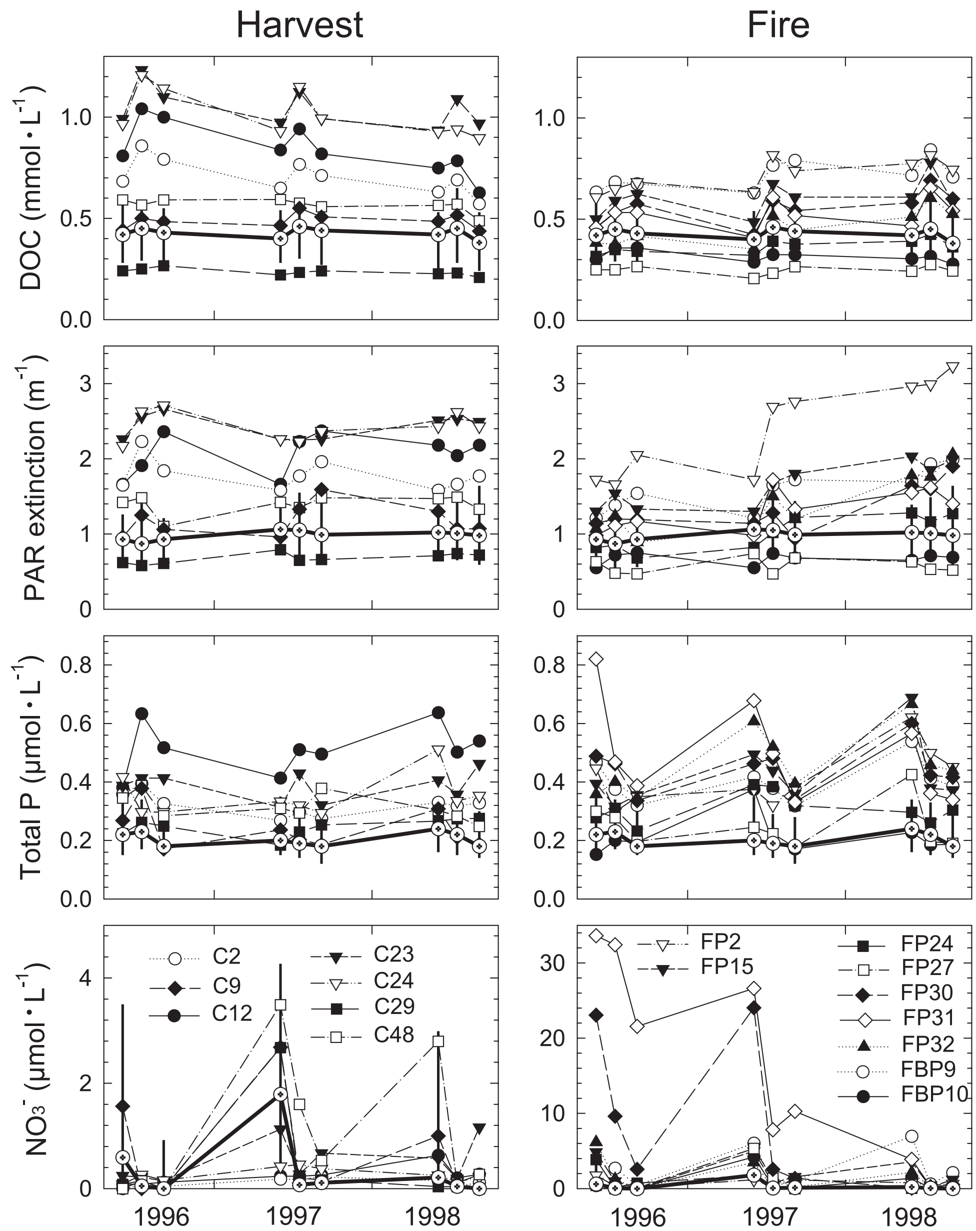


is expected to persist for decades after forest clearance by harvesting or fire (Plamondon 1993). Increased runoff should increase the transfer of terrestrial DOC and colour to lakes by increasing the development of superficial flow paths through organic-rich soil layers (Mulholland et al. 1990; D'Arcy and Carignan 1997). If the transfer of terrestrial DOC and colour to lakes had been strictly caused by a rise in the water table and an increase in superficial flow path, DOC and ε_{PAR} should have been highest in burnt lakes, since their drainage areas had twice as much forest cleared as those of cut lakes. The facts do not support such a simple hydrological control of DOC and colour export to lakes affected by harvest or fire. DOC was higher in cut lakes, where runoff must have been lower than in the nearly completely deforested burnt catchments, and decreased rapidly in some of them. Moreover, DOC and ε_{PAR} were low in 1996 in burnt lakes and increased afterwards. Compared with fire, harvesting leaves large quantities of easily leached or decomposed organic material, which may have produced part of the excess DOC initially observed in cut lakes only. This fresh DOC may be less coloured than humic DOC derived from upper soil horizons, since the significant decrease in DOC seen in cut lakes during the first 3 years is not paralleled by a corresponding decrease in ε_{PAR}. We can only speculate on the reasons explaining low initial concentrations and increasing trends of DOC and colour in burnt lakes. Low export of DOC and colour from drainage areas may reflect the nearly complete loss of important organic C sources (seasonal litterfall from trees, shrubs, and herbaceous vegetation) and the partial combustion of the organic forest floor. Some DOC and colour may have been sorbed by the large quantities of charcoal left on the ground after fire (Wardle et al. 1998). The large N plus P pulse associated with fire may also have stimulated the degradation of terrestrial DOC and colour, both on land and in the water.

Phosphorus and nitrogen

TP (yearly averages) was significantly higher in cut and burnt lakes than in reference lakes (Fig. 2). A significant (RMA) seasonal pattern was observed in reference and burnt lakes but not in cut lakes (Fig. 3). Concentrations were usually at a maximum in springtime and declined by ~20% during later months in reference lakes and by ~50% in several burnt lakes. Compared with reference lakes, springtime and annual average TP concentrations in burnt lakes showed a significant (RMA) increasing trend from 1996 to 1998. In reference lakes, 52% of the variability in springtime log TP is explained by log DR and 74% by a combination of log DR and –log ΣLA (Table 2, model 12). Note, however, that an equally successful model involves log DR and –log CA (r^2 = 0.74). The negative influence of lake size may be interpreted as resulting from the important contribution of fringing vegetation to P loading in small lakes (Cole et al. 1990). These different models stress the point that such empirical

Fig. 3. Evolution of dissolved constituents and ε_{PAR} during the first 3 years following forest clearance by harvesting or fire. The thick solid lines indicate median concentrations plus and minus the 5th and 95th percentiles for reference lakes.

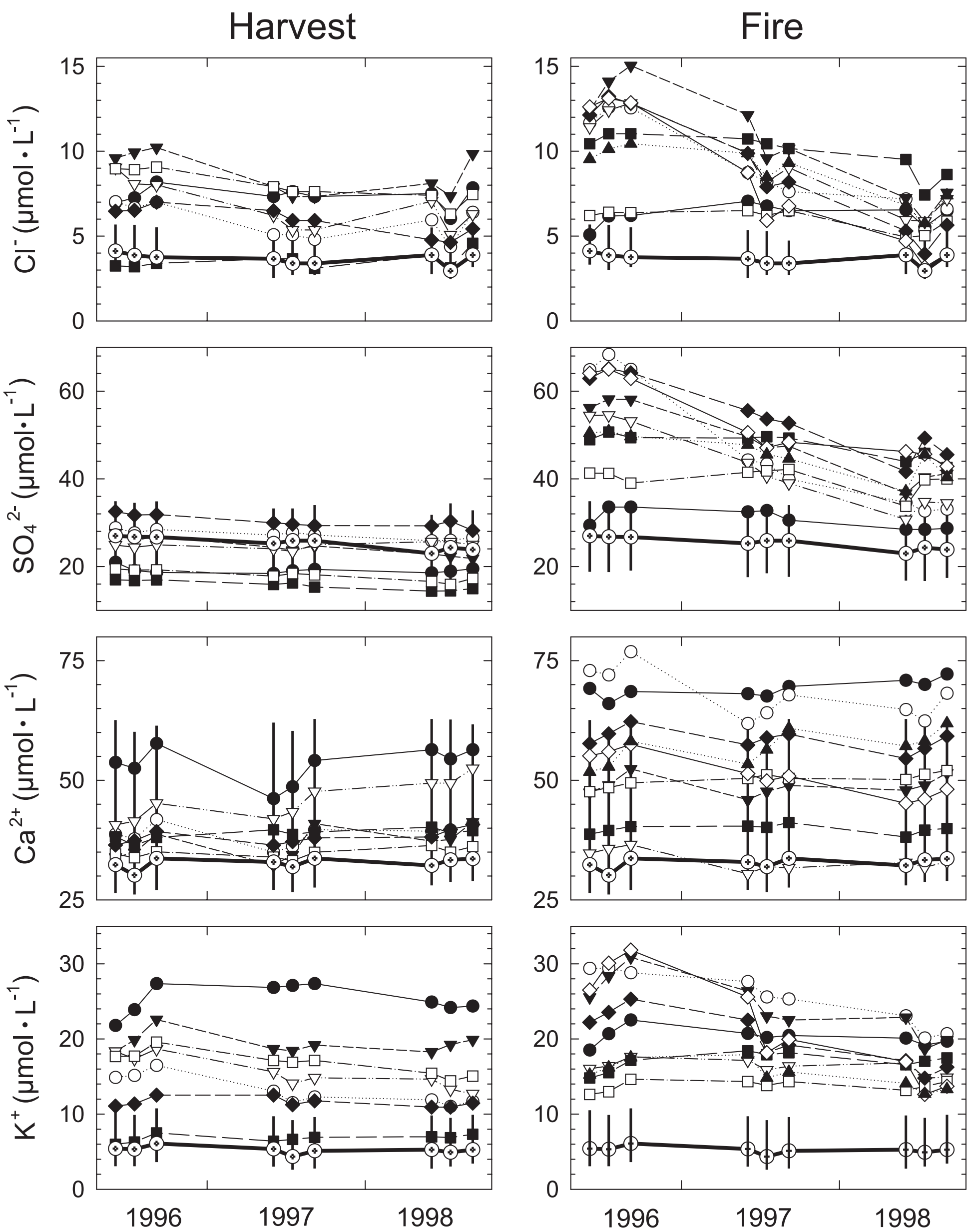


relationships must be interpreted with caution due to the high degree of autocorrelation between morphometric lake and watershed variables and to the large number of independent variables.

TP concentrations in several of the cut and burnt lakes were higher than the range observed in reference lakes (Fig. 3). As found for DOC and ε_{PAR}, these high TP concentrations cannot be attributed solely to the slightly higher DRs of cut and burnt lakes compared with reference lakes. As found above for DOC in cut lakes, TP was directly proportional to the DR and CUT/ΣLA (Fig. 6; Table 2, model 13, r^2 = 0.57) and to CUT/VOL (model 14, r^2 = 0.70), as found above for DOC and ε_{PAR} in cut lakes. Similar relationships are found for burnt lakes (models 15 and 16). Lake FP15 was a clear outlier in model 16, however, and was omitted from this regression. Exceptionally, a large headwater lake drains into Lake FP15 and contributes to intercept some of the P load arriving from the burnt catchment. Regression slopes and intercepts calculated separately for cut and burnt lakes are not significantly different and both groups of lakes are shown together in model 17 and Fig. 6.

Forest clearance by fire and harvesting appears to cause similar TP increases in surface waters when expressed as a function of DR and of the CUT/ΣLA (Fig. 6). The chemical form and bioavailability of P supplied from harvested and burnt drainage areas may not be identical, however. The rapid seasonal decrease in TP observed in burnt lakes is indicative of a high bioavailability compared with cut lakes, where no consistent seasonal decrease in TP occurs. Moreover, burnt lakes produced more planktonic chlorophyll *a*

Fig. 3 (*concluded*).



per unit TP than cut lakes did (Planas et al. 2000). Springtime and mean annual TP concentrations in burnt and cut lakes may take several more years to reach background values, since they did not decrease between 1996 and 1998.

Harvesting had no significant effect on NO_3^- concentrations (RMA). In both reference and cut lakes, NO_3^- was generally low in the spring (<0.04–3 μmol·L^{-1}) and, in a majority of cases, decreased below the detection limit by midsummer (Fig. 3). Springtime NO_3^- concentrations in reference lakes were weakly but significantly related to ZM (r^2 = 0.30). Other variables expected to be related to NO_3^- (latitude, longitude, DR, residence time) played no significant role. NO_3^- concentrations were weakly but nonsignificantly related to indicators of acid deposition (SO_4^{2-} (positive), alkalinity (negative)). Fire had a different and more varied effect on NO_3^- concentrations (Fig. 3; note that the scales are different for cut and burnt lakes). Very high springtime concentrations (>20 μmol·L^{-1}) were measured in 1996 in two lakes (FP30 and FP31) and intermediate concentrations were observed in other lakes (FP15, FP24, and FP32), whereas NO_3^- did not exceed reference levels in some other lakes (FP2, FBP9, FBP10, and FP27). A combination of log(FIR/ΣLA) and log(catchment slope) explained 90% of the variability in springtime NO_3^- in burnt lakes (Table 2, model 18). This NO_3^- pulse was short-lived, however; by 1998, NO_3^- concentrations in all burnt lakes were near those observed in reference and cut lakes.

NH_4^+ concentrations were systematically low (<0.07–1.5 μmol·L^{-1}) and were unrelated to pH (data not shown). Although concentrations were low, median values were significantly different in burnt lakes (0.6 μmol·L^{-1}), cut lakes (0.4 μmol·L^{-1}), and reference lakes (0.3 μmol·L^{-1}). TN was

Fig. 4. DOC versus DR for reference (○), cut (+, % harvested), and burnt lakes (●). The solid line shows the relationship between DOC and the DR for the reference lakes only (1996 values). The broken lines show the 95% confidence interval of predictions.

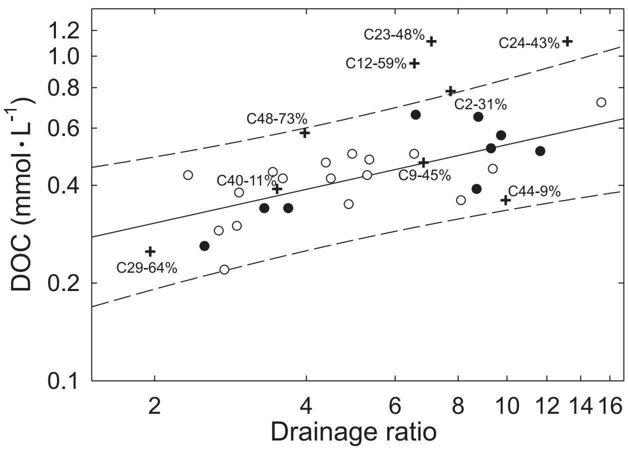


Fig. 5. Observed versus predicted DOC and light attenuation (ε_{PAR}) in reference (○) and cut lakes (+, % harvested) using volumetric impact models 3 and 6 (Table 2).

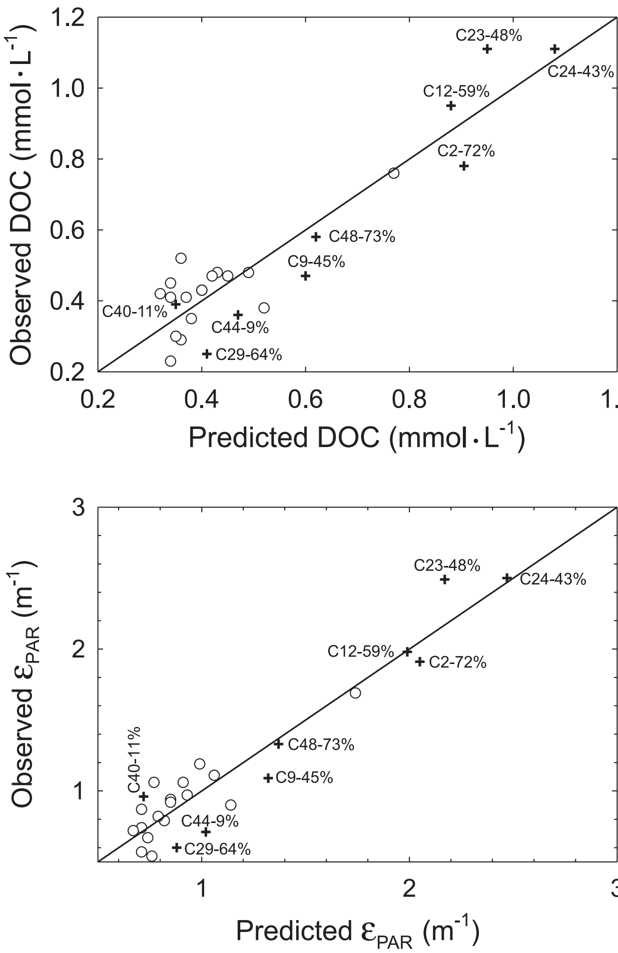
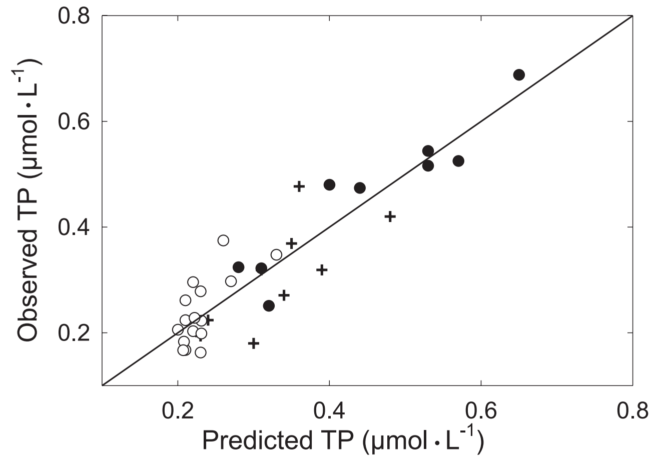


Fig. 6. Observed versus predicted springtime TP concentrations using the areal impact model 17 (Table 2) in reference (○), cut (+), and burnt lakes (●).



highest in burnt lakes and lowest in reference lakes (Fig. 2). Regression models for total organic nitrogen (TON = TN – (NO_3^- + NH_4^+)) for reference lakes were similar to those found for DOC, ε_{PAR}, and TP. Average TON concentrations were slightly but significantly higher in cut and burnt lakes (19.9 and 19.7 μmol·L^{-1}) than in reference lakes (15.9 μmol·L^{-1}). In cut lakes, TON was closely related to CUT:VOL, as found for DOC, ε_{PAR}, and TP (Table 2, model 19).

Major ions

The concentrations of several major ions (Ca^{2+}, Mg^{2+}, K^+, Cl^-) and alkalinity in reference lakes were lower in 1997 than in 1996 and 1998, likely because of a higher runoff in 1997. Annual variability in reference lakes was small for $Ca^{2+}$and Mg^{2+} (<1%), which are usually well buffered by the soil matrix, but pronounced for K^+, Cl^-, and alkalinity (9–19%). SO_4^{2-} decreased significantly (RMA) in reference lakes during the 3 years (–7%), while Na^+ increased significantly (+6%). Local variations in bedrock geology and till composition probably cause most of the spatial variability in base cations, since no significant relationships were found between cations and SO_4^{2-} concentrations in reference lakes or between cations and watershed properties, including the type and thickness of superficial deposits. A geological influence is apparent for Ca^{2+}, Mg^{2+}, and alkalinity. Six lakes (N55, N89, N122, P106, P107, FBP9) located on a particular granodioritic gneiss formation had alkalinities and Ca^{2+} and Mg^{2+} (but not Na^+ and K^+) concentrations that were up to twice as high as those of other lakes in their group.

K^+ and Cl^- concentrations were much higher in cut and burnt lakes than in reference lakes (Figs. 2 and 3) and were directly proportional to the amount of forest cleared by fire or harvest, divided by the lake's surface area or volume (Table 2, models 20–23). The different slopes of the harvest and fire models suggest that logging exports more K^+ and Cl^- than fire to surface and groundwaters. The relationships are not significantly different (analysis of covariance), however, and both perturbations may have comparable effects (Lamontagne et al. 2000). K^+ and Cl^- decreased significantly between 1996 and 1998 in cut and burnt lakes and should reach reference levels in a few more years. Cut lakes had

Fig. 7. Evolution of DOC, TP, K^+, and ε_{PAR} in Lakes P25 (○), P106 (▲), and P110 (●) logged between 1996 and 1998 (see text for timing). The thick solid lines with dotted circles show median values plus and minus the 5th and 95th percentiles for the reference lakes.

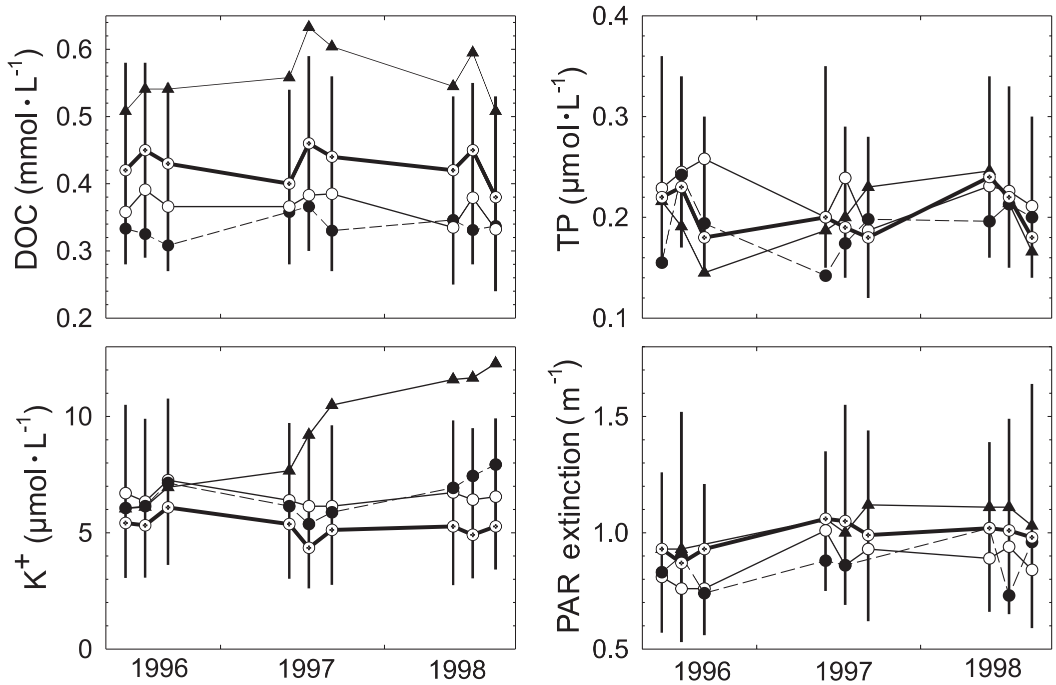


Table 3. Predicted changes in ε_{PAR}, DOC, TP, and K^+ following harvesting in lakes P25, P106, and P110.

Lake	DR	Clearcut (km^2)	Lake area (km^2)	Lake volume (m^3)	AIR ($m^2 \cdot m^{-2}$)	VIR ($m^2 \cdot m^{-3}$)	$\Delta\varepsilon_{PAR}$ (m^{-1})	ΔDOC ($mmol \cdot L^{-1}$)	ΔTP ($\mu mol \cdot L^{-1}$)	ΔK^+ ($\mu mol \cdot L^{-1}$)
P25	4.50	0.28	0.33	1.55×10^6	0.85	0.18	0.11±0.02	0.05±0.01	0.022±0.012	2.7±1.7
P106	2.97	0.32	0.41	1.89×10^6	0.78	0.16	0.10±0.02	0.04±0.01	0.020±0.011	2.5±1.5
P110	5.93	1.06	0.81	3.74×10^6	1.30	0.28	0.17±0.04	0.07±0.02	0.034±0.018	4.3±2.6

Note: The areal impact ratio (AIR) is the impacted surface area divided by the lake surface area). The volumetric impact ratio (VIR) is the impacted surface area divided by the lake volume. Expected changes in ε*subromanPAR*, DOC, TP, and K^+ are calculated from models 6, 3, 13, and 20 (Table 2).

slightly but significantly more Na^+ than reference and burnt lakes. The cause for this difference is uncertain. Important Na losses by volatilization during fire may explain lower Na^+ concentrations in burnt lakes. On the other hand, the fact that a relatively mobile ion such as Na^+ did not decrease in cut lakes between 1996 and 1998 suggests that slightly higher Na^+ concentrations in cut lakes may be due to differences in geology.

Ca^{2+} and Mg^{2+} concentrations in burnt lakes were significantly higher (two- to four-fold) than in reference and cut lakes (Figs. 2 and 3). The box plots in Fig. 2 suggest that harvesting had no significant effect on Ca^{2+} and Mg^{2+}. This conclusion should be considered with caution, as other facts point to the contrary. Although Ca^{2+} concentrations in cut lakes were within the range of concentrations observed in reference lakes, they were systematically above the median reference concentration (Fig. 3). Furthermore, lakes located on granodioritic gneiss contributed most of the variability in Ca^{2+} concentrations. When these lakes and the negligibly impacted Lakes C40 and C44 are removed from the ANOVA, the three groups of lakes become significantly different, with mean Ca^{2+} concentrations of 30.8, 39.6, and 50.7 $\mu mol \cdot L^{-1}$ in reference, cut, and burnt lakes, respectively (1996 values). Finally, Ca^{2+} (but not Mg^{2+}) is significantly related to CUT/VOL and CUT/ΣLA in cut lakes (Table 2, model 24). These observations indicate that Ca^{2+} is lost at an elevated rate, not only from burnt catchments but also from harvested ones.

Contrary to what was observed for K^+ and Cl^-, Ca^{2+} and Mg^{2+} concentrations did not change significantly with time in cut lakes compared with reference lakes. However, some individual lakes (C24, C29, C40) showed significant increases. This behaviour is expected, since divalent cations are less mobile than K^+ and Cl^- in soils and may take longer to reach surface waters. Wood contains relatively large amounts of Ca compared with other nutrients, and important quantities of Ca can be held for a few years in decaying branches and boles. Trends were more erratic in burnt lakes. As a group, and compared with reference lakes, Ca^{2+} decreased slightly but significantly with time. There were, however, significant increases in Lakes FBP10, FP27, and FP31 and significant decreases in Lakes FBP9, FP2, and FP30.

In reference lakes, SO_4^{2-} was positively related to longitude (Table 2, model 25, $r^2 = 0.52$) and negatively related to alkalinity ($r^2 = 0.33$ and 0.58 when high-alkalinity outliers N55, N89, and N122 are removed), reflecting a well-known H_2SO_4 deposition gradient in this region due to the presence of important point sources of SO_2 (Sudbury, Rouyn-Noranda) located 200–400 km to the west. A similar relationship was found for alkalinity (model 26) when high-alkalinity reference Lakes N55, N89, and N122 were removed from the data set. Note that the slope of the alkalinity model (26) is of opposite sign and twice as high as that of the SO_4^{2-} model (18). The correct acid–base stoichiometry of both relationships indicates that they are not statistical artefacts. Much higher SO_4^{2-} concentrations were observed in burnt lakes than in reference and cut lakes. High SO_4^{2-} levels in lakes draining burnt catchments were likely caused by the partial combustion of the LFH layer and B horizon, which contain 30 times more organic S than the above-ground biomass in conifer forests (Houle and Carignan 1992). In burnt lakes, SO_4^{2-} was weakly but significantly related to the impact ratio ($r^2 = 0.48$). Spatial variability in fire intensity, in humus thickness, and in the degree of humus combustion probably explains the poorer fit between SO_4^{2-} and the impact ratio.

Response of lakes harvested during the study

Four of the study lakes were logged between 1996 and 1998. Lake P106 had 24% of its catchment logged in the winter of 1996–1997, Lake P25 had 18% of its catchment logged in the summer of 1997, and Lake P110 had 22% of its catchment logged in the winter of 1997–1998. Lake P106 showed clear K^+ and Cl^- signals (Fig. 7, Cl^- not shown) arriving during the summer of 1997. In Lake P110, K^+ and Cl^- began to reach the lake during the summer of 1998, by which time they had not yet arrived in Lake P25. Lake P25 had narrow cut blocks mostly fringing the limit of its watershed. In this lake, K^+ and Cl^- liberated from branches and leaves may not have had the time to reach Lake P25 by September of 1998.

Clear-cutting had minimal effects on DOC, ε_{PAR}, and TP in these lakes (Fig. 7). Compared with median reference values, slight increases were apparent only in Lake P106 and perhaps in Lake P110. In Lake P106, after 1996, DOC and ε_{PAR} increased by about 0.05 $mmol \cdot L^{-1}$ and $0.2 \cdot m^{-1}$, respectively, relative to median reference values. TP increased by about 0.03 $\mu mol \cdot L^{-1}$ and K^+ by 5 $\mu mol \cdot L^{-1}$. In Lake P110, DOC and ε_{PAR} appear to have increased at most by about 0.03 $mmol \cdot L^{-1}$ and $0.1 \cdot m^{-1}$, respectively, after 1996. Except for K^+ and Cl^-, these changes were not significant. Such limited effects are consistent with the predictions (Table 3) of the impact models proposed above (see Figs. 5 and 6 and Table 2 (models 3, 6, 13 and 20)). According to these models, in Lakes P106 and P110, DOC should not have increased by more than 0.04–0.07 $mmol \cdot L^{-1}$, ε_{PAR} should not have increased by more than $0.10–0.17 \cdot m^{-1}$, and TP should not have increased by more than 0.02–0.03 $\mu mol \cdot L^{-1}$. The increase in K^+ observed in Lake P106 is also near the range predicted by model 20.

Differential effects of fire and harvest: management implications

Our observations indicate that Boreal Shield lakes respond predictably, but in some cases differently, to disturbance by fire and harvesting. Three years after impact, some responses to these disturbances appear to be similar (TP, TON, K^+, Cl^-, Ca^{2+}), while others are clearly different (DOC, ε_{PAR}, NO_3^-, SO_4^{2-}). Some effects, such as the K^+, Cl^-, NO_3^-, and SO_4^{2-} pulses that follow disturbance by fire or harvesting, are apparently short-lived, while others (DOC, ε_{PAR}, TP), which may be driven at least in part by hydrological changes, appear to evolve on a longer time scale. Some of these responses, such as the increase in DOC, ε_{PAR}, and TP in cut lakes, should be given particular attention because they appear to be long-lived and because they may adversely influence other key limnological properties. Reduced water clarity reduces mixing depth (Fee et al. 1996) and the depth of the euphotic zone, where photosynthesis can occur. In our study lakes, high DOC loads from cut lakes are associated with high methyl mercury in zooplankton and fish (Garcia and Carignan 1999, 2000). The simple impact ratio models shown above can be used, along with water quality guidelines, to define allowable cuts at the watershed scale.

Acknowledgements

We thank the Canadian Network of Centres of Excellence on Sustainable Forest Management, NSERC, and SSHRC for financial support. The late R.H Peters assisted with the study design. The forest companies Abitibi-Consolidated, Cartons Saint-Laurent, and Donohue and Kruger provided land-use data. Pierre Gagnon and Christiane Hudon (Centre Saint-Laurent) kindly supplied the repeated-measurements analyses. Pierre Magnan, Jaap Kalff, Benrnadette Pinel-Alloul, Dolors Planas, and numerous students assisted at various stages of the project.

References

Bayley, S.E., Schindler, D.W., Beaty, K.G., Parker, B.R., and Stainton, M.P. 1992. Effects of multiple fires on nutrient yields from streams draining boreal forest and fen watersheds. Can. J. Fish. Aquat. Sci. **49**: 577–583.

Cole, J.J., Caraco, N.F., and Likens, G.E. 1990. Short-range atmospheric transport: a significant source of phosphorus to an oligotrophic lakes. Limnol. Oceanogr. **35**: 1230–1237.

D’Arcy, P., and Carignan, R. 1997. Influence of watershed topography on water quality in southeastern Québec Shield lakes. Can. J. Fish. Aquat. Sci. **54**: 2215–2227.

D’Elia, C.F., Steudler, P.A., and Corwin, N. 1977. Determination of total nitrogen in aqueous samples using persulfate digestion. Limnol. Oceanogr. **22**: 760–764.

Engstrom, D.R. 1987. Influence of vegetation and hydrology on the humus budgets of Labrador lakes. Can. J. Fish. Aquat. Sci. **44**: 1306–1314.

Fee, E.J., Hecky, R.E., Kasian, S.E.M., and Cruikshank, D.R. 1996. Effect of lake size, water clarity, and climatic variability on mixing depths in Canadian Shield lakes. Limnol. Oceanogr. **41**: 912–920.

Franklin, J.F. 1993. Preserving biodiversity: species, ecosystems, or landscapes? Ecol. Appl. **3**: 202–205.

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

Garcia, E., and Carignan, R. 2000. Mercury concentrations in northern pike (*Esox lucius*) from boreal lakes with logged, burned, or undisturbed catchments. Can. J. Fish. Aquat. Sci. **57**(Suppl. 2): 129–135.

Houle, D., and Carignan, R. 1992. Sulfur speciation and distribution in soils and aboveground biomass of a boreal coniferous forest. Biogeochemistry **16**: 63–82.

Hunter, M.L., Jr. 1991. Coping with ignorance: the coarse-filter approach strategy for maintaining biodiversity. *In* Balancing on the brink of extinction. *Edited by* K.A. Kohm. Island Press, Washington, D.C.

Hunter, M.L., Jr. 1993. Natural fire regimes as spatial models for managing boreal forests. Biol. Conserv. **65**: 115–120.

Lamontagne, S., Carignan, R., D'Arcy, P., Prairie, Y.T., and Paré, D. 2000. Element export in runoff from eastern Canadian Boreal Shield drainage basins following forest harvesting and wildfires. Can. J. Fish. Aquat. Sci. **57**(Suppl. 2): 118–128.

Likens, G.E., Bormann, F.H., Johnson, N.M., Fisher, D.W., and Pierce, R.S. 1970. Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook watershed ecosystem. Ecol. Monogr. **40**: 23-47.

Mulholland, P.J., Wilson, G.V., and Jardine, P.M. 1990. Hydrogeochemical response of a forested watershed to storms: effects of preferential flow along shallow and deep pathways. Water Resour. Res. **26**: 321–3036.

Natural Resources Canada. 1996. The state of Canada's forests. Cat. Fol-6/1996E. Natural Resources Canada, Canadian Forest Service, Ottawa, Ont.

Nicolson, J.A., Foster, N.W., and Morrison, I.K. 1982. Forest harvesting effects on water quality and nutrient status in the boreal forest. Can. Hydrol. Symp. **82**: 71–89.

Patoine, A, Pinel-Alloul, B., Prepas, E.E., and Carignan, R. 2000. Do logging and forest fires influence zooplankton biomass in Canadian Boreal Shield lakes? Can. J. Fish. Aquat. Sci. **57**(Suppl. 2): 155–164.

Plamondon, A.P. 1993. Influence des coupes forestières sur le régime d'écoulement de l'eau et sa qualité. Rapport pour le Ministère des forêts du Québec. Université Laval, Laval, Qué.

Planas, D., Desrosiers, M., Groulx, S.-R., Paquet, S., and Carignan, R. 2000. Benthic and pelagic algal responses in eastern Canadian Boreal Shield lakes following harvesting and wildfires. Can. J. Fish. Aquat. Sci. **57**(Suppl. 2): 136–145.

Rask, M.R., Arvola, L., and Salonen, K. 1993. Effects of catchment deforestation and burning on the limnology of a small forest lake in southern Finland. Verh. Int. Ver. Limnol. **25**: 525–528.

Rask, M.R., Nyberg, K., Markkanen, S.L., and Ojala, A. 1998. Forestry in catchments: effects on water quality, plankton, zoobenthos and fish in small lakes. Boreal Environ. Res. **3**: 75–86.

Rasmussen, J.B., Godbout, L., and Schallenberg, M. 1989. The humic content of lake water and its relationship to watershed and lake morphometry. Limnol. Oceanogr. **34**: 1336–1343.

Schindler, D.W. 1971. A hypothesis to explain differences and similarities among lakes in the Experimental Lakes Area, northwestern Ontario. J. Fish. Res. Board Can. **28**: 295–301.

Stainton, M.P., Capel, M.J., and Armstrong, F.A.J. 1977. The chemical analysis of fresh water. 2nd ed. Can. Fish. Mar. Serv. Misc. Spec. Publ. No. 25.

St-Onge, I., and Magnan, P. 2000. Impact of logging and natural fires on fish communities of Laurentian Shield lakes. Can. J. Fish. Aquat. Sci. **57**(Suppl. 2): 165–174.

Wardle, D.A., Zackrisson, O., and Nilsson, M.C. 1998. The charcoal effect in boreal forests: mechanisms and ecological consequences. Oecologia, **115**: 419–426.

Wright, R.F. 1976. The impact of forest fire on the nutrient influxes to small lakes in northeastern Minnesota. Ecology, **57**: 649–663.