PROJECT REPORT
1999-35

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ISBN 1-55261-050-0

Comparative Impacts of Fire and Forest Harvesting on Water Quality in Boreal Shield Lakes

SFM Network Project: Water-quality: causes of natural variability and impacts of watershed disturbances in the boreal forest lakes of Québec

by

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October 1999

EXECUTIVE SUMMARY

Dissolved organic carbon (DOC), light penetration, nutrients and major ions were monitored in boreal shield lakes for three years following their simultaneous impact by clearcuts (9 "cut lakes") or fires (9 "burnt lakes"). Sixteen similar undisturbed lakes served as references, while the catchments of 4 other ones were harvested during the study. Dissolved organic carbon (DOC) and the light attenuation coefficient (ε_{PAR}) were up to 3-fold higher in cut lakes than in reference and burnt lakes. Compared to median values for reference lakes, cut and burnt lakes had higher concentrations of total phosphorus (TP, 2 to 3-fold), total organic nitrogen (TON, 2-fold), K^+, Cl^- and Ca^{2+} (up to 6-fold). Nitrate and SO_4^{2-} concentrations were respectively up to 60-fold and 6-fold higher in burnt lakes than in reference and cut lakes. In reference lakes, ε_{PAR}, DOC, TP and TON were related to catchment properties (drainage ratio, drainage density). In impacted lakes, the concentrations of most constituents were directly proportional to the area harvested or burnt divided by the lake's volume or area. These simple impact models, shown in figures 5 and 6, correctly predicted the weak effects observed in lakes harvested during the study; they can be used, along with water quality guidelines, to define allowable cuts and cutting schedules at the watershed scale.

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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 Cartons Saint-Laurent, Abitibi-Consolidated, Donohue and Kruger kindly provided land-use data. P. Magnan, J. Kalff, B. Pinel-Alloul, D. Planas and numerous students assisted at various stages of the project.

INTRODUCTION

Aquatic ecosystems of the Canadian boreal forest are influenced by recurring (50-300 y) 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 from more or less arbitrary harvest patterns to 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 will preserve ecosystem integrity. These important assumptions remain to be tested, however, for both terrestrial and aquatic ecosystems.

Given the diversity in vegetation, land forms 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; Plamondon et al. 1982; Nicolson et al. 1982; Verry 1986; Bayley et al. 1992), while a minority of studies have focused on lakes (Wright 1976; Rask 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 color. Few studies were designed, however, to compare the effects of forest removal by fire and clearcutting, or to provide quantitative impact models linking the type and extent of watershed disturbance to water quality in lakes.

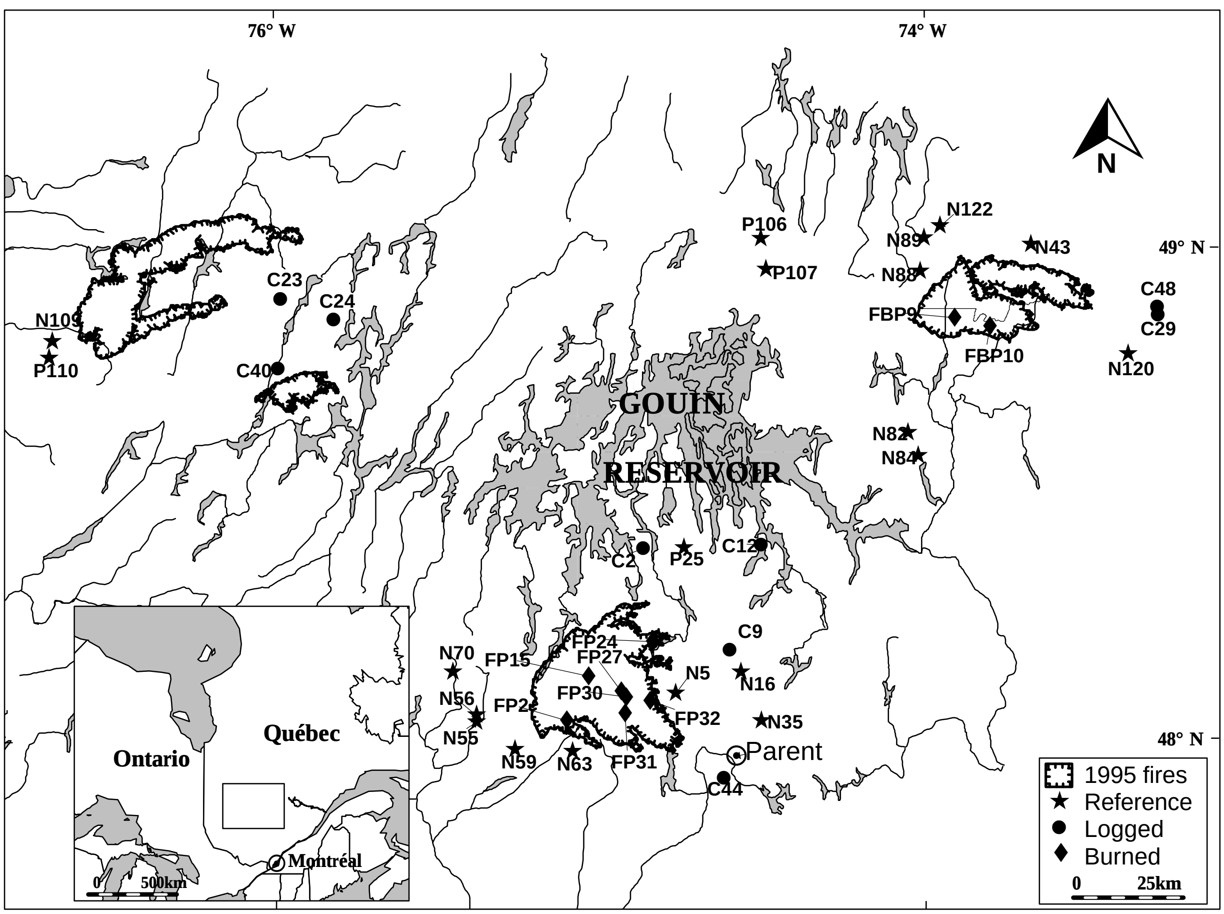
During the summer of 1995, four major wildfires totalling 1 620 km^2 occurred in a portion of Québec's boreal forest undergoing extensive harvesting operations. The large number of lakes (~800) simultaneously impacted by fire or harvest provided a unique opportunity to compare and to model the effects of both disturbances on water quality and on biotic communities in similar lakes (see Patoine et al.; Planas et al., and St-Onge and Magnan, *this issue*). The present report compares some of the chemical and physical changes that have occurred in the lakes during the first three years after impact. A companion paper (Lamontagne et al. *this issue)* compares element export from the catchments of these lakes.

Study region

The lakes are located in a 30 000 km^2 area surrounding Réservoir Gouin 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 north-east quadrant of the region. The low (40-150 m net elevation) rolling hills are covered by glacial deposits that are generally thin (0-2m), but which 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 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. *this issue*). Runoff was exceptionally high in 1997, and low in 1998, compared to its long term average of 586 mm

The fire-impacted lakes are located in two of the four large 1995 wildfires (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.

Figure 1. Location of the study lakes.



METHODS

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), 9 “burnt lakes” which had most of their catchment burnt in 1995 (“FP” and “FBP” lakes), 9 “cut lakes” which had 9% to 72% of their catchment logged in 1995 (“C” lakes), and 4 lakes which had 5 to 21 % of their catchment logged during the study (“P” lakes). The following criteria were used in the selection process:

Forest age: > 50 y.
Wetlands: < 6% of the drainage area.
Drainage ratio: 2 to 15.
Hydrological order: 1
Fetch: > 1 km.
Impact by fire of logging in 1995.
Fire intensity: Medium to severe, no salvage logging.
Maximum depth: >5 m.

All retained lakes met these conditions, with the following exceptions: Lakes C43 and N120 had 40 and 45 years old tree stands in their catchments. Note that stands aged > 70 y on Québec’s vegetation maps can be considerably older. Lake order is not always exactly 1, as seventeen 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.

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, 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). Residence times were not corrected for increased runoff due to loss of forest cover.

Sampling and chemistry

The lakes were sampled three times during the ice-free season (a few days after ice-out in late May or early June, in mid-July and mid-September). The penetration of light available to photosynthesis (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 analysed (within 72 h). Samples for DOC measurements and occasional color (absorbance at 440 nm) measurements were filtered on washed Gelman Supor 0.45 µm membranes and kept at 4 °C. Dissolved organic carbon was measured (Shimadzu TOC-5000) within 72 h by infra-red 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 pre-combusted GF/C filters. Total phosphorus was measured using the molybdenum-blue method (Stainton et al. 1977), after autoclaving 50 ml samples with 0.5 g of potassium persulfate for one hour at 120 °C. Nitrate 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 was measured as NO_3^- after alkaline persulfate digestion of 50 ml samples at 120 °C (D'Elia et al. 1977). Total N was not measured in June and July of 1997. Chloride, 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.

Statistical analyses

The significance of between group (reference, cut and burnt lakes) differences was assessed using one-way ANOVA with a non-parametric Kruskal-Wallis test when distributions were significantly skewed. Differences between years within a given group were tested using two-way ANOVA. Multiple regressions were performed on normalized ($\log_{10}$-transformed) data when necessary, using a stepwise forward variable selection procedure with an F-ratio of 4. Unless otherwise noted, significance tests and regression parameters are significant at the $p < 0.05$ level. Variable definitions and units are summarized in Table 1.

Table 1. Watershed characteristics and morphometry of the study lakes. Attributes with significantly different between-group medians are shown in bold.

Variable	Reference (n=17)				Harvested (1995, n = 9)				Harvested (1996-1997, n = 4)				Burnt (1995, n = 9)			
	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 area (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 ($WA = CA+\Sigma LA$)	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 ($DR = CA/\Sigma LA$)	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
Maximum depth (Zmax, m)	11	12	7	21	12	14	5	30	12	14	10	23	15	17	10	34
Residence time (τ, y)	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, deg.)	5.8	6.37	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 (area=CON)	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 (area=LEA)	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 (y)			40	>80			50	>90			50	70			50	70

[*]The catchment is defined here as the terrestrial portion of the watershed.

DATA ANALYSIS

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 drainage ratio, the water residence time and catchment slope (Schindler 1971; Engstrom 1987; Rasmussen et al. 1989; D'Arcy and Carignan 1997). 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 m vs. 5.8 m) than burnt lakes (F lakes); burnt lakes had less conifers (51.2% of DA) than the three other groups (65.3 to 74%). The significant difference in mean depth between reference and burnt lakes is due to the inclusion of one particularly deep lake (FBP27, 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.

We believe that the study lakes are representative of a large number of lakes in the area. A systematic inventory of the 1,549 lakes occurring in four quadrants totalling 3,350 km^2 located within the study region shows that 30% of the water surface area is composed of lakes having the same range of LA than our study lakes. 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.

Water quality

Mean annual chemical properties and ε_{PAR} for reference, cut and burnt lakes are compared between group and between years in Figure 2. In order to increase the homogeneity of cut lakes group, lakes C44 and C40, which had only 9% and 11 % of their drainage area harvested, are excluded from the 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 of 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 to those of burnt lakes (91%). The effects of harvesting and fire shown in Fig. 2 are therefore not directly comparable.

DOC and light penetration

In reference lakes, DOC (3-year averages) was significantly related ($r^2 = 0.49$) to the drainage ratio (logDR), as usually found (Schindler 1971, Engstrom 1987, Rasmussen et al. 1989; Carignan and D'Arcy 1997). Goodness-of-fit was improved, however, when when drainage density (DD = km of permanent streams per km^2 of catchment) was added as an independent variable (r^2 = 0.61, Table 2, model 1). Similar, but more accurate relationships were found between ε_{PAR} and logDR (r^2 = 0.67) and a combination of log(DR) and logDD (r^2 = 0.78, Table 2, model 2).

Figure 2a. Box plots showing median values with the 25th and 75th, and the 5th and 95th percentiles of annual average concentrations and of the light (PAR) extinction coefficient in reference (open boxes, n = 16), cut (hashed boxes, n = 7) and burnt (shaded boxes, n = 9) lakes. The thick horizontal lines indicate between treatment differences: for a given year, boxes with different levels are significantly different. Numbers on top of the boxes indicate between years differences: boxes with different numbers are significantly different. Two lakes (C40 and C44) where only a small portion of the catchment had been harvested are excluded from these analyses.

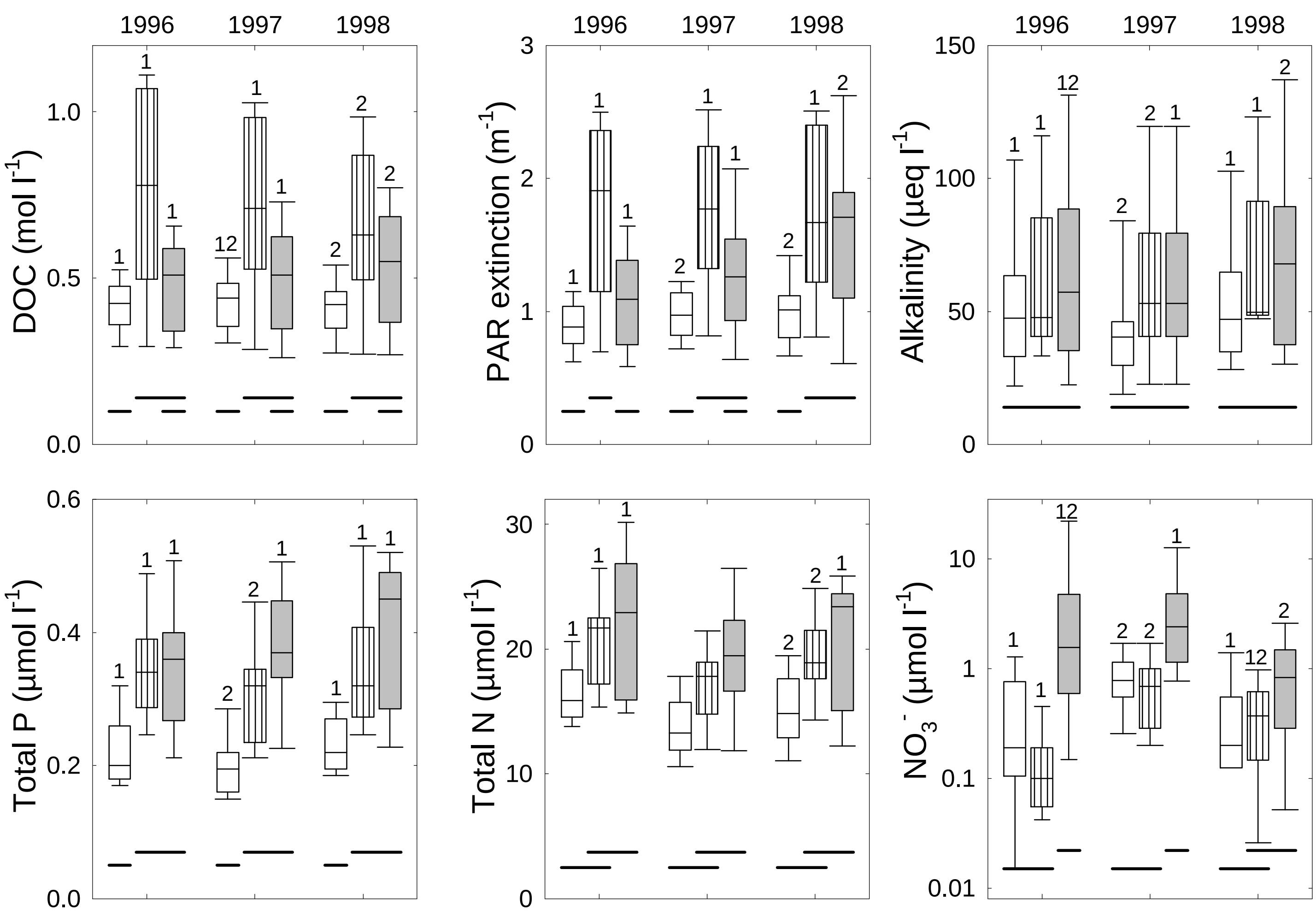


Figure 2b. Legend as in figure 2a.

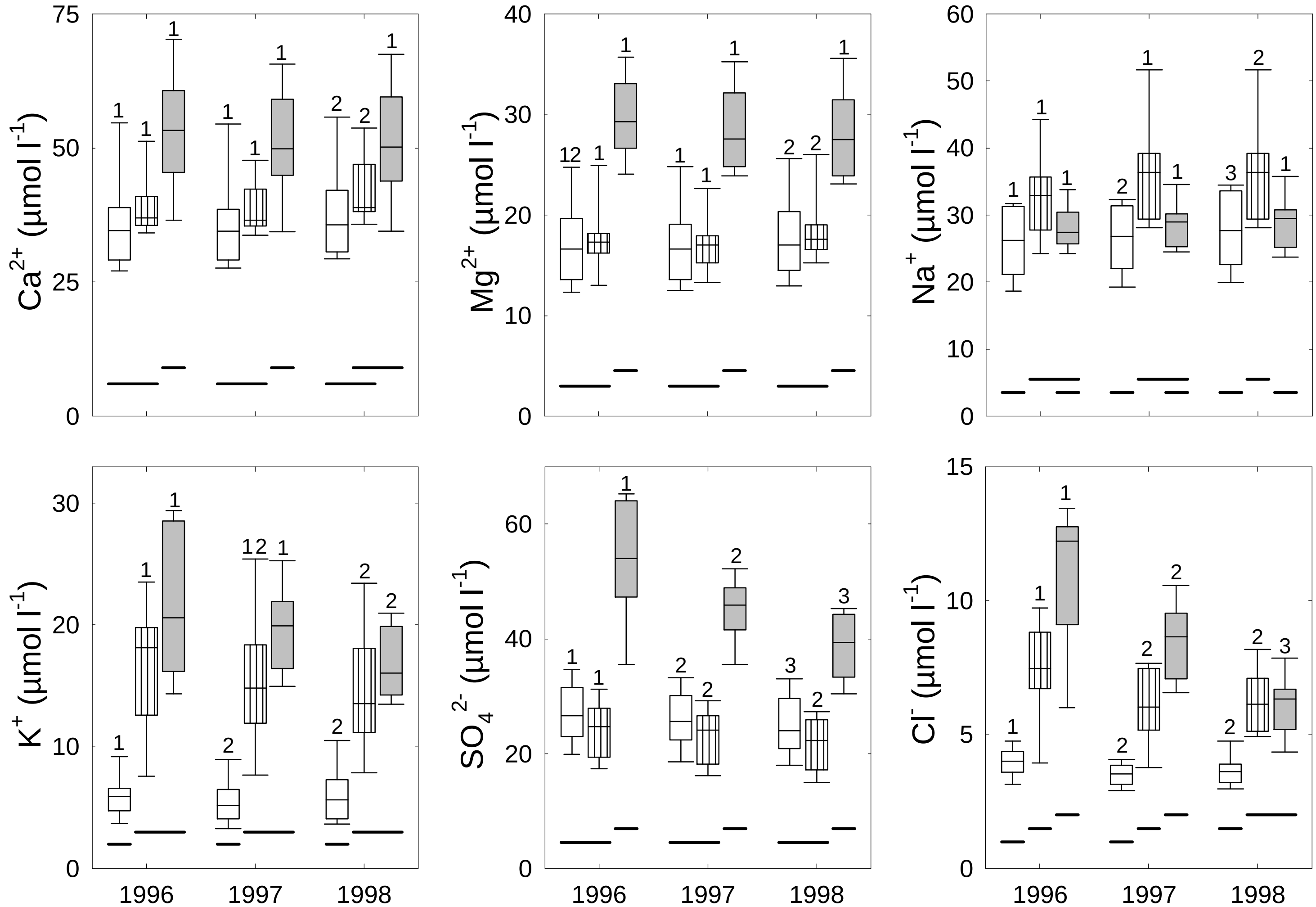


Table 2. Linear regression equations. R = reference lakes; C = cut lakes, F = burnt lakes. sp = springtime values only. SE = standard error of estimates.

No.	Lakes	Year	Model	r^2	SE
1	R	96-98	DOC = 0.48±0.10 log(DR) – 0.13±0.06 log(DD+0.2)	0.61	0.07
2	R	96-98	ε_{PAR} = 1.36±0.21 log(DR) – 0.32±0.12 log(DD+0.2)	0.78	0.14
3	C	96	DOC = 0.48±0.06 (CUT/VOL, m^2/m^3) + 0.26±0.07	0.87	0.12
4	C	97	DOC = 0.45±0.06 (CUT/VOL, m^2/m^3) + 0.25±0.07	0.89	0.10
5	C	98	DOC = 0.34±0.07 (CUT/VOL, m^2/m^3) + 0.25±0.08	0.79	0.13
6	C	96	ε_{PAR} = 1.05±0.18 (CUT/VOL, m^2/m^3) + 0.63±0.19	0.83	0.33
7	C	97	ε_{PAR} = 0.93±0.13 (CUT/VOL, m^2/m^3) + 0.72±0.13	0.89	0.23
8	C	98	ε_{PAR} = 0.94±0.15 (CUT/VOL, m^2/m^3) + 0.82±0.17	0.82	0.30
9	C	96	DOC = 0.16±0.04 (CUT/ΣLA) + 0.23±0.13	0.69	0.19
10	C	97	DOC = 0.15±0.04 (CUT/ΣLA) + 0.21±0.11	0.73	0.17
11	C	98	DOC = 0.10±0.03 (CUT/ΣLA) + 0.27±0.12	0.58	0.18
12	R	96-98sp	log(TP) = 0.54±0.09 log(DR) – 0.07±0.02 log(DD+0.2) – 1.0±0.02	0.72	0.06
13	C+F	96-98sp	TP = 0.041±0.004 (CUT or FIR)/ ΣLA + 0.19±0.02	0.85	0.06
14	C+F	96-98sp	TP = 0.063±0.004 (CUT or FIR)/ VOL + 0.19±0.03	0.80	0.06
15	R+C+F		log VOL = 1.14±0.11 log AL + 0.67±0.05	0.75	0.16
16	R	96-98	log TON = 0.40±0.09 log(DR) – 0.06±0.02 log(DD+.2) + 0.87±0.06	0.61	0.06
17	C	96	TON = 5.0±1.2 CUT/VOL + 14.2±1.4	0.71	2.4
18	R	96-98	SO_4^{2-} = 4.7±1.2 LONG - 329±90	0.52	4.0
19	R	96-98	Alkalinity = - 9.2±2.7 LONG + 728±236	0.44	9.8
20	C	96	K^+ = 3.3±1.0 CUT/ΣLA + 5.5±2.9	0.62	4.5
21	C	96	Cl^- = 1.0±0.3 CUT/ΣLA + 4.0±1.0	0.58	1.6
22	F	96	K^+ = 1.4±0.5 FIR/ΣLA + 12.7±3.7	0.51	4.9
23	F	96	Cl^- = 0.61±0.24 FIR/ΣLA + 6.8±1.7	0.48	2.2
24	C	96	Ca^{2+} = 1.9±0.7 CUT/ΣLA + 31.0±2.2	0.51	3.4

Most cut lakes had strikingly higher DOC concentrations and ε_{PAR} than reference lakes and showed a significant decreasing trend in DOC (but not ε_{PAR}) with time (Fig. 2). Dissolved organic carbon concentrations in burnt lakes were intermediate, but did not differ significantly from those of reference and cut lakes. In contrast to cut lakes, DOC and ε_{PAR} increased significantly with time in burnt lakes. Dissolved organic carbon concentrations showed a significant (two-way ANOVA, $p < 0.05$) seasonal pattern in most of the reference, cut and burnt lakes, with mid-summer values rising 5-15% above those of spring and late summer (Fig. 3). The 400-700 nm light extinction coefficient (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 colored 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.

Figure 3a. Evolution of concentrations of DOC, TP, NO_3^- and ε_{PAR} during the first three years following forest clearance by harvesting or fire. The thick grey lines indicate median concentrations ± the 5th and 95th percentiles for reference lakes.

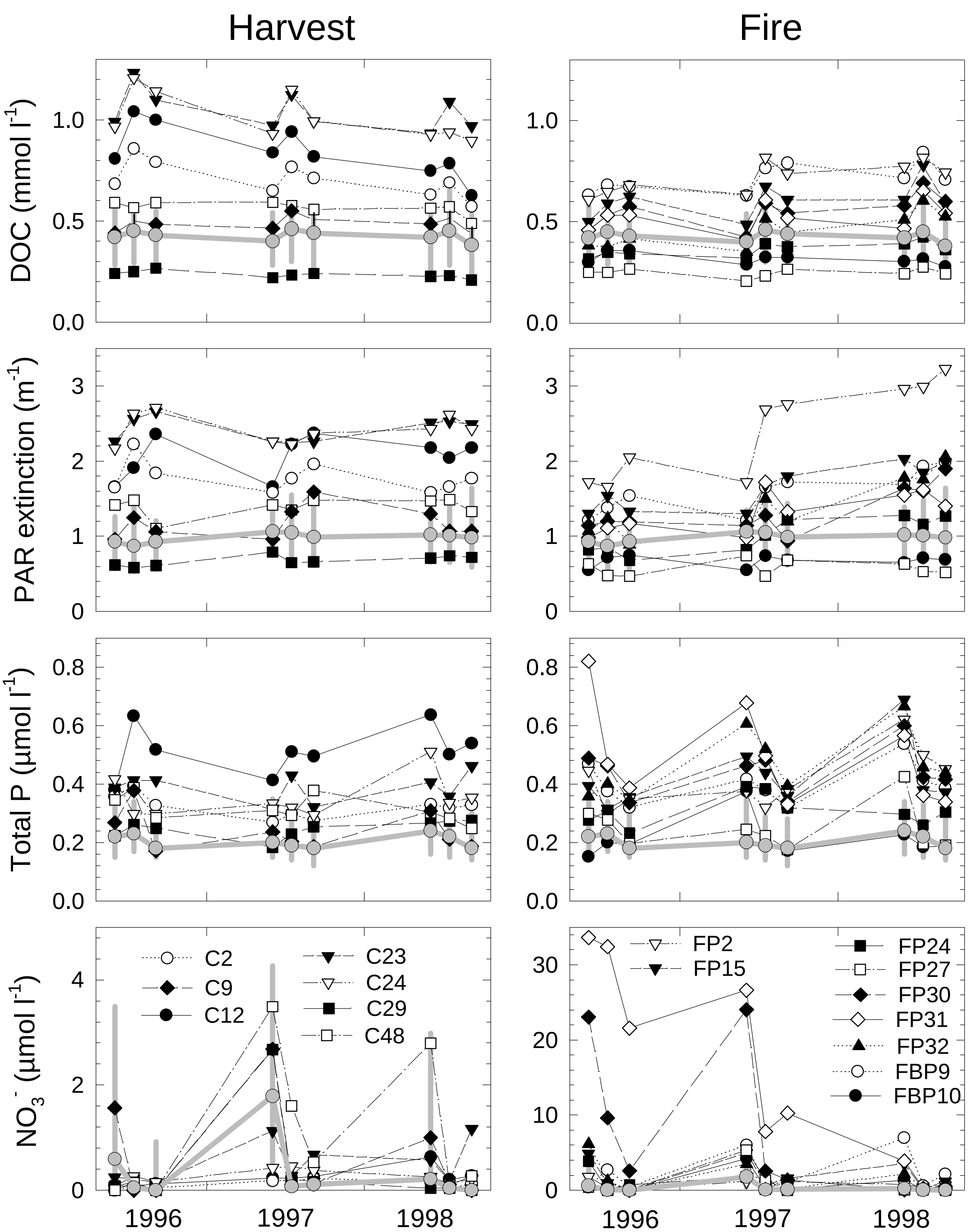
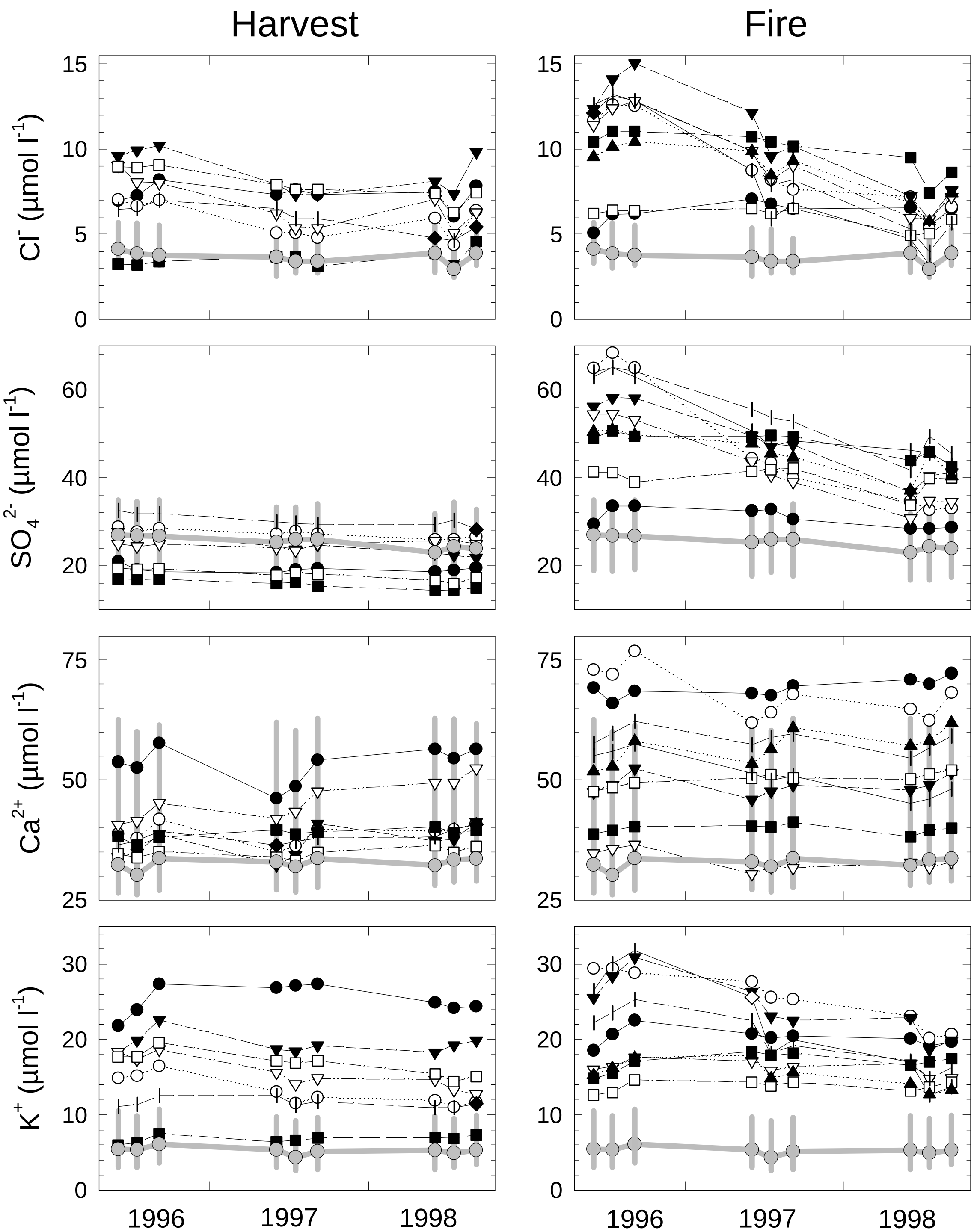
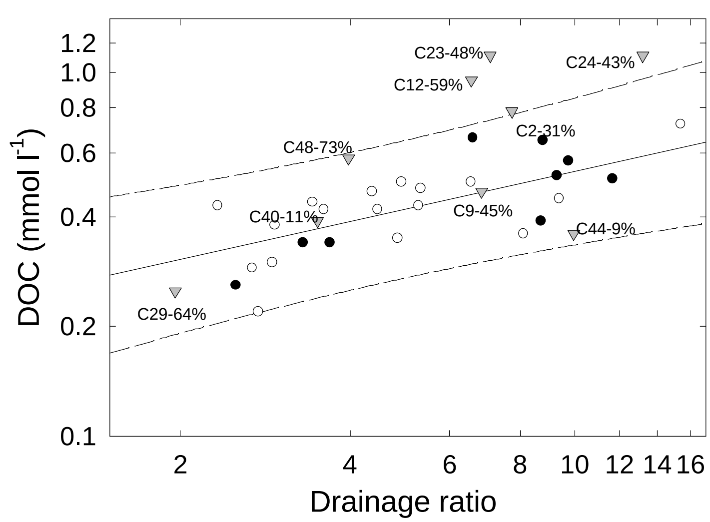


Figure 3b. Legend as in figure 3a for Cl^-, SO_4^{2-}, Ca^{2+} and K^+.



The high DOC concentrations and ε_{PAR} values observed in cut lakes (Fig. 2) cannot be entirely attributed to their higher median DR ratio (Table 1) compared to reference lakes. Inclusion of the cut lakes on a linear regression plot of DOC *vs.* 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. These lakes either have a low DR or had only a small fraction of their drainage area harvested. Figure 4 suggests that COD in cut lakes will significantly exceed the range of natural variability only when their drainage ratio 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, but lies well within the model's prediction limits. Lake C9 is deeper, however, and has a longer residence time (1.1 yr) than lakes C2, C12, C23 and C24 (0.4 to 0.7 yr). 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.

Figure 4. Relationship (solid line) between DOC and the drainage ratio for reference lakes only (1996 values). The dashed lines show the 95% confidence interval for predictions. Filled circles: burnt lakes; triangles: cut lakes with the percentage of the catchment harvested.



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 a volumetric impact ratio, defined as the harvested surface area : lake volume ratio CUT:VOL (m^2/m^3) and explains 87% of the variability in DOC (model 3, Fig. 5). The slope of this relationship 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 be substituted for a more easily obtained areal impact ratio (CUT:ΣLA), but with a loss of precision (models 9-11).

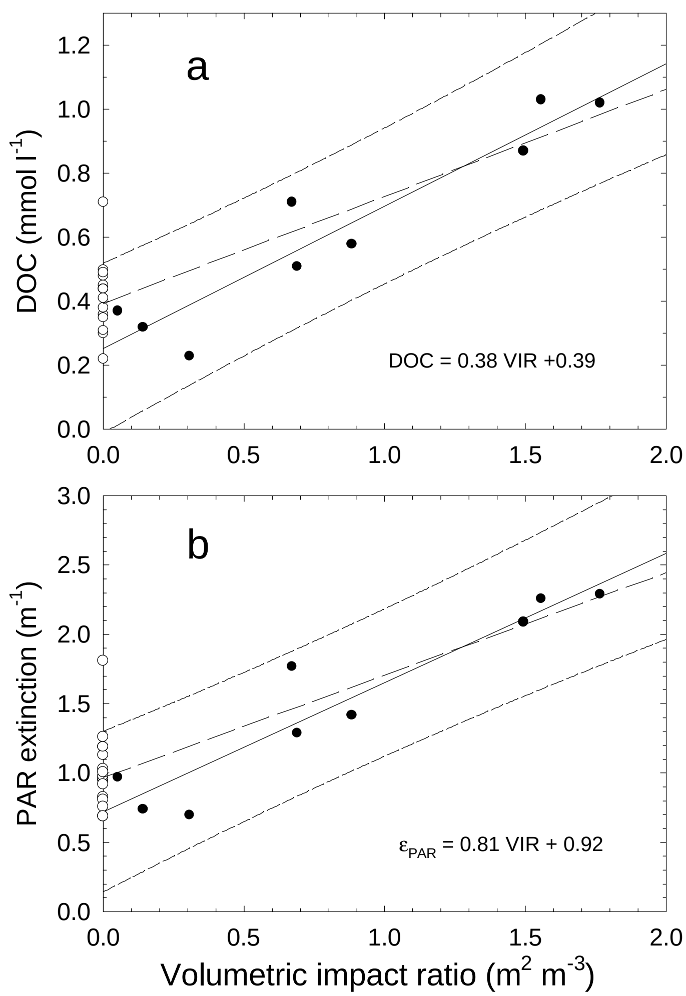
Models 3-11 (Fig. 5) are slightly biased since their intercepts are about 30% lower than the mean DOC and ε_{PAR} of reference lakes. The bias arises because two of the three lakes having the lowest impact ratios also have unusually low drainage ratios (C29, DR = 1.96; C40, DR = 3.50). More realistic regression lines and equations including the reference lakes are also shown on Fig. 5. Variables appearing in the regression models found for reference lakes (saturation index, drainage ratio) did not improve the impact ratio models for cut lakes, possibly because of their high degree of correlation with CUT, VOL and ΣLA, and because population size was too small to resolve watershed effects that were subtle compared to those caused by a massive loss of forest cover.

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 *vs.* time (n = 9 data points) for individual lakes shows that the decrease is significant in lakes C2 (-0.07 mmol y^{-1}), C12 (-0.11 mmol y^{-1}), C24 (-0.09 mmol y^{-1}) and C29 (-0.015 mmol y^{-1}). It is also marginally significant in lake C48 (-0.02 mmol y^{-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} y^{-1}) and C40 (+0.07 m^{-1} y^{-1}). During the same interval, DOC changed significantly, by a much smaller amount (-0.006 mmol y^{-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}. Dissolved organic carbon concentrations in fire-impacted lakes were near those of reference lakes one year after impact (Figs. 2-4), and increased instead of decreased afterwards. DOC increased in all lakes except FBP10, but the change was significant only in lakes FP2 and FP32 when individual lakes were examined by linear regression. Likewise, ε_{PAR} was near reference levels in 1996, but increased significantly afterwards in all burnt lakes except lakes FBP10 and FP 27 (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% to 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 mid-summer 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 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 color to lakes by increasing the development of superficial flowpaths through organic-rich soil layers (Mulholland et al. 1990; D'Arcy and Carignan 1997). If the transfer of terrestrial DOC and color to lakes had been strictly caused by a rise in the water table and an increase in superficial flowpath, 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 color 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 to fire, harvesting leaves large quantities of easily decomposable organic material which may have produced part of the excess DOC initially observed in cut lakes only. This fresh DOC may be less colored than humic DOC derived from upper soil horizons since the significant decrease in DOC seen in cut lakes during the first three 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 color in burnt lakes.

Figure 5. Relationships (solid lines) and 95% confidence interval (short-dashed lines) between a) DOC and b) ε_{PAR}, and the volumetric impact ratio (VIR = area cleared : lake volume) for cut lakes (n = 9, models 8 and 11 in Table 3). The long-dashed dashed lines show the least squares relationships obtained when reference lakes (circles) are included.



Low export of DOC and color from drainage areas may reflect the nearly complete loss of important organic carbon sources (seasonal litterfall from trees, shrubs and herbaceous vegetation) and the partial combustion of the organic forest floor. Some DOC and color may have been sorbed by the large quantities of charcoal left on the ground after fire (Wardle et al. 1998). The large N+P pulse associated with fire may also have stimulated the degradation of terrestrial DOC and color, both on land and in the water.

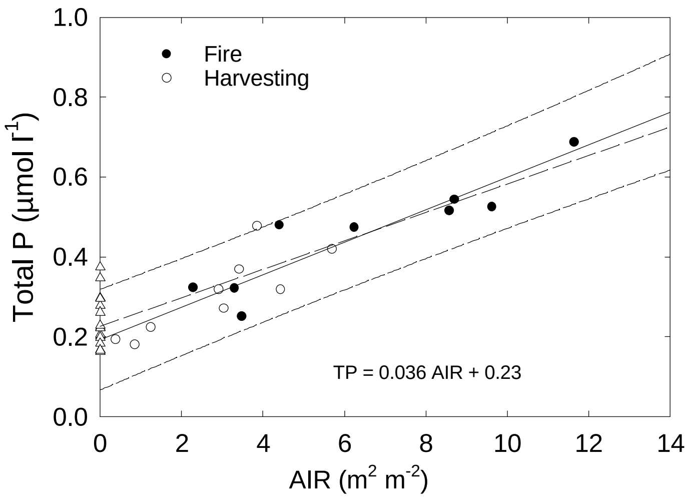
Phosphorus and nitrogen

Total phosphorus (yearly averages) was significantly higher in cut and burnt lakes than in reference lakes (Fig.2). A significant 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 a striking ~50% in burnt lakes. Springtime and annual average TP concentrations in cut and burnt lakes showed no significant decreasing or increasing trend from 1996 to 1998. In reference lakes, 52% of the variability in springtime TP is explained by DR, and 72% by a combination of log(DR) and log(DD) (model 12). Note, however, that an equally successful model involves +logDR and -ΣLA (r^2 = 0.75), or +logDR 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 relationships must be interpreted with caution, due to the high degree of correlation between morphometric lake and watershed variables and to the large number of independent variables.

Total P 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 drainage ratios of cut and burnt lakes compared to reference lakes. For both cut and burnt lakes, TP was directly proportional to the areal impact ratio (Fig. 6, model 13, r^2 = 0.85) and to the volumetric impact ratio (model 14, r^2 = 0.80), as found above for COD and ε_{PAR} in cut lakes. Regression slopes and intercepts calculated separately for cut and burnt lakes are not significantly different. Lake FP15 was a clear outlier in model 14, however, and was omitted from this regression. Exceptionally, a large headwater lake drains into lake FP15 and contributes to dilute most of the P load arriving from the burnt catchment. We attempted to circumvent this problem by estimating the volume of higher order lakes present in burnt catchments using the VOL *vs.* LA relationship calculated for our study lakes (model 15). The r^2 of the new regression (r^2 = 0.80, model 14 including FP15) did not surpass that of the more easily applied model 13.

Forest clearance by fire and harvesting result in similar TP concentrations in surface waters, as TP in cut and burnt lakes fall on the same line in Fig. 6 when expressed as a function of the relative surface area of forest cleared per m^2 or m^3 of lakewater. 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 to cut lakes, where no consistent seasonal decrease in TP occurs. Moreover, burnt lakes produced more planktonic chlorophyll *a* per unit TP than cut lakes did (Planas et al. *this issue*). 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.

Figure 6. Relationship (solid line) and 95% confidence interval (short-dashed lines) between total phosphorus and the aeral impact ratio (AIR = aread cleared by fire or harvest : lake area). The long-dashed dashed lines show the least squares relationship obtained when reference lakes (triangles) are included. Filled circles: fire-impacted lakes; open circles: harvest-impacted lakes.



Harvesting had no detectable effect on NO_3^- concentrations. In both reference and cut lakes, NO_3^- was generally low in the spring (< 0.04 to 3 µmol l^{-1}) and, in a majority of cases, decreased below detection limit by mid-summer (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, drainage ratio, residence time) played no significant role. Nitrate concentrations were weakly, but non significantly related to indicators of acid deposition (SO_4^{2-}, positive, and alkalinity, negative). Fire had a different and more varied effect on NO_3^- concentrations (Fig. 3, note that 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), 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). This NO_3^- pulse was short-lasted. By 1998, NO_3^- concentrations in all burnt lakes were near those observed in reference and cut lakes.

Ammonium concentrations were systematically low (< 0.07 to 1.5 µmol l^{-1}) and were unrelated to pH. Median concentrations 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}). A significant positive correlations with chlorophyll *a*, which was also highest in burnt lakes (Planas et al. *this issue*), suggests that between group differences are a sampling artefact due to N mineralisation by bacteria and zooplankton during the relatively long delay (up to 12 h) between sample collection and filtration.

Total nitrogen was highest in burnt lakes, and lowest in reference lakes (Fig. 2). Regression models for total organic nitrogen [TON = TN – (NO_3^- + NH_4^+)] in reference lakes (model 16) were similar to those found for DOC, ε_{PAR} and TP. Average TON concentrations were significantly higher in cut and burnt lakes (19.9 and 19.7 µmol l^{-1}) than in reference lakes (15.9 µmol l^{-1}). From 1996 to 1998, TON decreased slightly (6%) but significantly (ANCOVA) in cut lakes only. In cut lakes, TON was closely related to the volumetric impact ratio (CUT:VOL), as found for DOC, ε_{PAR} and TP (model 17).

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. The differences were 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%). Sulfate decreased in reference lakes during the three years (-7%) while Na^+ increased (+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, 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.

Potassium and chloride 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 (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, however, and both perturbations may have comparable effects (Lamontagne et al. *this issue*). Potassium and Cl^- decreased significantly between 1996 and 1998 and should reach reference levels in a few more years. Cut lakes had 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.

Calcium and magnesium concentrations in burnt lakes were significantly higher (2- to 4-fold) than in reference and cut lakes (Figs. 2 and 3). The box plots of 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 for 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 µmol l^{-1} in reference, cut and burnt lakes, respectively (1996 values). Finally, Ca^{2+} (but not Mg^{2+}) is significantly related to the areal and volumetric impact ratios in cut lakes (model 24). These observations indicate that Ca^{2+} is lost at an elevated rate not only from burnt catchments, but also from harvested ones.

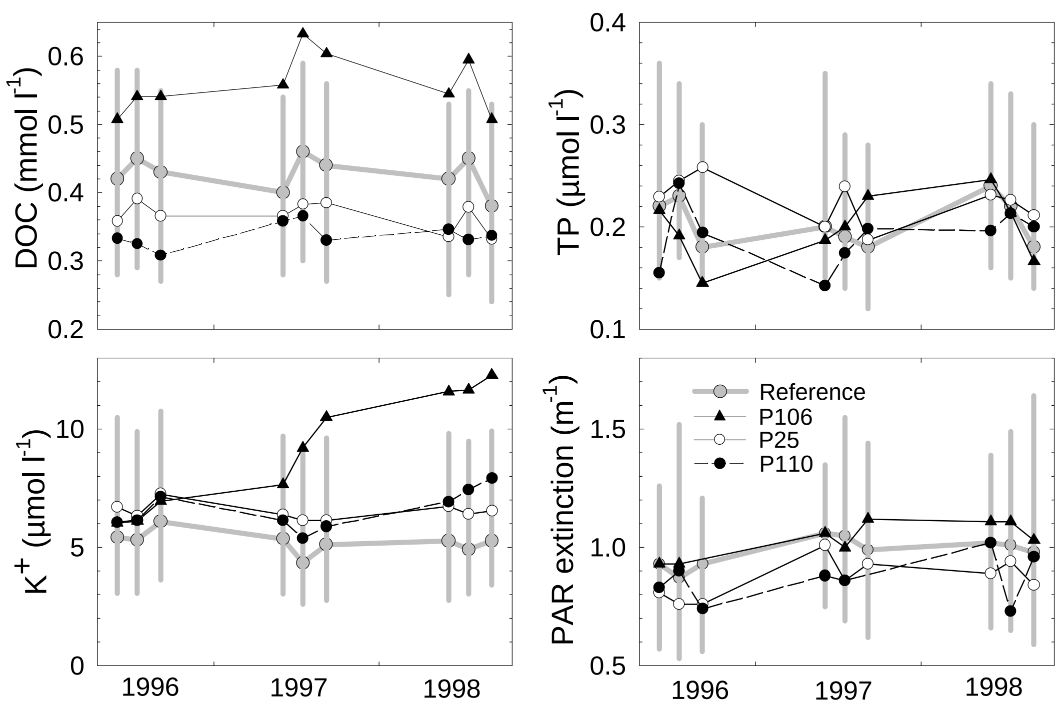
Contrary to what was observed for K^+ and Cl^-, Ca^{2+} and Mg^{2+} concentrations tended to increase slightly with time in all cut lakes. The increase was significant in lakes C24, C29 and C40. This behavior is expected since divalent cations are less mobile than K^+ and Cl^- in soils and may take longer to reach surface waters. Wood contains relatively large amounts of Ca compared to 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, where significant increases (FBP10, FP27, FP31) or decreases (FBP9, FP2, FP30) are observed.

In reference lakes, SO_4^{2-} was positively related to longitude ($r^2 = 0.52$, model 18), 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 19) when high-alkalinity reference lakes N55, N89 and N122 were removed from the data set. Note that the slope of the alkalinity model (19) is 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 explain the poorer fit between SO_4^{2-} and the impact ratio.

Response of lakes harvested during the study

Four of our study lakes were logged between 1996 and 1998. One of these (P107) had only 6% of its catchment logged during the winter of 1997-98 and is not considered here. Lake P106 had 24% of its catchment logged in the winter of 1996-97. 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 1997-98. 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 it 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.

Figure 7. Evolution of DOC, total phosphorus, potassium and ε_{PAR} in three lakes logged between 1996 and 1998 (see text for timing). The thick grey lines show median values ± the 5th and 95th percentiles for reference lakes.



Clearcutting had minimal effects on DOC, ε_{PAR} and TP in these lakes (Fig. 7). Compared to 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 l^{-1} and 0.2 m^{-1}, respectively, relative to median reference values. Total P increased by about 0.03 µmol l^{-1}, and K^+ by 5 µmol l^{-1}. In lake P110, DOC and ε_{PAR} appear to have increased at most by about 0.03 mmol l^{-1} and 0.1 m^{-1} 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 Fig. 5, 6 and models 20-21 of Table 2). According to these models, in lakes P106 and P110, DOC should not have increased by more than 0.06 to 0.11 mmol l^{-1}, ε_{PAR} should not have increased by more than 0.14 to 0.23 m^{-1}, and TP by more than 0.03 to 0.05 µmol l^{-1}. The increase in K^+ observed in lake P106 is also near the range predicted by model 20.

Table 3. Predicted changes in ε_{PAR}, DOC, TP and K^+ following harvesting in lakes P25, P106 and P110. 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 (m^2) divided by the lake volume (m^3). Models used to calculate the expected changes in ε_{PAR}, DOC and TP are given in the legends of Figs. 5 and 6. Model 43 (Table 2) Was used for K^+.

Lake	Drainage ratio	Clearcut (km^2)	Lake area (km^2)	Lake volume (m^3)	AIR (m^2/m^2)	VIR (m^2/m^3)	$\Delta\varepsilon_{PAR}$ (m^{-1})	ΔDOC (mmol l^{-1})	ΔTP (µmol l^{-1})	ΔK^+ (µmol l^{-1})
P25	4.50	0.28	0.33	1.55E06	0.85	0.18	0.15±0.04	0.07±0.02	0.029±0.005	2.7±1.7
P106	2.97	0.32	0.41	1.89E06	0.78	0.16	0.14±0.04	0.06±0.02	0.028±0.005	2.5±1.5
P110	5.93	1.06	0.81	3.74E06	1.30	0.28	0.23±0.07	0.11±0.03	0.047±0.008	4.3±2.6

MANAGEMENT APPLICATIONS

Differential effects of fire and harvest

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-lasted, 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-lasted 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, *this issue*). The simple impact ratio models shown above (Figs. 5 and 6) can be used, along with water quality guidelines, to define allowable cuts at the watershed scale.

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