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Element Export in Runoff from Eastern Canadian Boreal Shield Drainage Basins Following Forest Harvesting and Wildfires

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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EXECUTIVE SUMMARY

Element export from 9 harvested, 9 burnt, and 16 reference boreal Shield lake basins in Haute-Mauricie, Québec, was estimated for the three years following harvesting or fires. Element export from the drainage area of the basins was evaluated using lake sampling surveys, estimated hydrological budgets, and estimated nutrient retention during lake transit. Increases in the export K^+ (3 to 8-fold reference), total N (2 to 3-fold reference), and total P (2-fold reference) were similar between harvested and burnt drainage areas. However, harvested drainage areas exported more dissolved organic carbon (DOC; 2 to 3-fold reference) while burnt drainage areas exported more divalent base cations (Ca^{2+} and Mg^{2+}; 2 to 3-fold reference), NO_3^- (up to 17-fold reference) and SO_4^{2-} (3 to 4-fold reference). Cumulative losses of K^+ in runoff during the three years of the study were of a similar magnitude as volatilization losses during fires and corresponded to ~33% of the losses by biomass removal during harvesting. While N and P exports in runoff following fires or harvesting represented negligible loss of nutrients for the forest, they were important inputs to lakes. The increased export of DOC following forest harvesting but not following forest fires indicates that these disturbances have different impacts on biogeochemical cycles in the boreal Shield forest.

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INTRODUCTION

Forest harvesting and forest fires are important disturbances in the boreal Shield forest in eastern Canada (Natural Resources Canada 1996). Annually, in the Province of Québec, ~1% of the boreal forest is harvested and <0.1% to 0.4% is affected by forest fires (Ministère des Ressources naturelles 1996). Some current forest harvesting management initiatives seek to emulate the effect of natural disturbances during forestry operations. However, the similarity of wildfires and harvesting on element loss from drainage areas[1] and their relative impacts on downstream freshwater ecosystems are not well known. Biogeochemical cycles are affected differently by forest fires and tree harvesting. Although forest fires and harvesting both result in increased runoff from reduced evapotranspiration (Wright 1976; Verry 1986; Bayley et al. 1992), forest fires tend to mineralize the organic soil (Smith 1970) while this layer is better preserved following harvesting. Whether the organic soil will be preserved following a disturbance will have important implications for soil mineralization and plant colonization processes, which are important controls on nutrient losses in runoff (Vitousek 1981).

In Shield conifer forests, losses of base cations (K^+, Ca^{2+}, and Mg^{2+}), nitrate (NO_3^-), and total phosphorous (TP) above background rates may last for 5 to 9 years following wildfires (Wright 1976; Bayley and Schindler 1991; Bayley et al. 1992). A similar effect is usually observed following harvesting (Nicolson et al. 1982; Plamondon et al. 1982; Nicolson 1988a). In some unproductive forests, however, the export of nutrients is occasionally unaffected by harvesting (Vitousek 1981; Nicolson 1988a). Increased export of dissolved organic carbon (DOC) is suspected to be important in harvested Shield basins (Krause 1982) but has not been well characterized. Changes in DOC inputs have been shown to have important implications for the physical, chemical, and biological properties of Shield lakes (Schindler et al. 1996, 1997).

We have studied the response of 16 reference and 18 impacted Shield lake basins in Haute-Mauricie, Québec, from 1996 to 1998. The drainage area of nine basins had been commercially harvested and nine others had sustained forest fires in 1995. We have estimated element export rates from the drainage area of the basins using time series of element concentration from lakes. Our objectives were two-fold; first, we compared element loss rates from harvested and burnt drainage areas relative to reference drainage areas. Second, we evaluated whether the cumulative export of nutrients in runoff in perturbed basins during the study was a significant loss relative to element volatilization during fires or biomass removal during harvesting.

METHODS

The study area (~ 50 000 km^2) is centered on Gouin Reservoir, Québec (48°50'N, 75°00'W), at the transition of the boreal mixed and boreal conifer forest (see Carignan et al. *this*

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[1] Drainage area (*DA*) = terrestrial part; lake area (*LA*); basin area (*BA*) = *DA* + *LA*

volume). The landscape is typical of the Canadian Shield, with a low relief, thin surficial deposits, and numerous lakes and wetlands. Extensive forestry operations are ongoing throughout the study area, where three large wildfires (30 000 ha to 50 000 ha, involving over 1 000 lakes) occurred during the summer of 1995. This provided a unique opportunity to simultaneously compare the effect of both disturbances on similar lakes. Thirthy-eight lake basins were selected on the basis of comparable size, depth, and morphometry (Carignan et al. *this volume*). Four of the original twenty reference basins (drainage areas undisturbed for the last 75 years) were excluded from the analysis because they were subsequently harvested in 1997 or 1998. Nine basins had 50% – 100% (average = 90%) of their drainage area severely burnt (Table 1) and nine other basins had 8.5% – 73% (average = 47%) of their drainage area harvested in 1995. Fourteen of the 34 lakes are second-order (i.e., receive inputs from usptream lakes). However, with the exception of lake FP15, the ratio of upstream lake area to study lake area is small (Table 1). Forest harvesting was usually in the form of 200 – 300 ha cuts with protection of regeneration and soils (CPRS), with 20 meters buffer strips fringing streams, lakes, and wetlands. The principal species harvested were black spruce (*Picea mariana* (Mill.) B.S.P.) and white birch (*Betula papyrifera* Marsh.). Salvage cuts did not occur in burnt lake basins.

Table 1. Morphometry of 34 harvested, burnt and reference basins in Haute-Mauricie (mean and range).

	Harvested	Burnt	Reference
	All basins		
LA (km^2)	0.56 (0.18 – 2.3)	0.40 (0.17 – 0.64)	0.43 (0.15 – 0.80)
DA (km^2)	3.4 (0.59 – 10)	4.9 (0.57 – 20)†	2.0 (0.45 – 4.5)
$DA{:}LA$	7.2 (2.0 – 14)	12 (2.5 – 41)†	5.4 (2.3 – 15)
Mean depth (m)	4.3 (2.1 – 7.5)	5.8 (4.2 – 10)	4.3 (2.1 – 8.5)
%DA cut or burnt	47% (8.5 – 73%)	91% (50 – 100%)	0% (0 – 1%)
n	9	9	16
	2nd-order basins only		
LA (km^2)	0.88 (0.18 – 2.3)	0.42 (0.23 – 0.57)	0.57 (0.19 – 0.80)
$\sum LA_k$ (km^2)	0.13 (0.01 – 0.28)	0.41 (0.04 – 1.5)‡	0.08 (0.005 – 0.023)
$\sum LA_k{:}LA$	0.15 (0.08 – 0.31)	0.85 (0.14 – 3.2)‡	0.13 (0.01 – 0.11)
$\sum DA_k$ (km^2)	1.0 (0.21 – 1.4)	4.3 (0.58 – 16)‡	0.71 (0.082 – 1.0)
DDA (km^2)	4.3 (1.3 – 9.2)	3.4 (1.8 – 4.2)	2.3 (0.98 – 3.7)
Mean depth (m)	4.0 (2.9 – 5.8)	5.4 (4.6 – 7.3)	4.5 (2.8 – 5.4)
%DA cut or burnt	55% (43 – 73%)	97% (92 – 100%)	0%
n	4	5	5

† without Lake FP15: mean DA = 3.1 km^2 and mean $DA{:}LA$ = 7.9

‡ without Lake FP15: $\sum LA_k$ = 0.12 km^2, $\sum LA_k{:}LA$ = 0.27 and $\sum DA_k$ = 1.5 km^2

Lake sampling

Duplicate integrated water samples of the euphotic zone (i.e., depth of 1% light penetration, usually 2 – 5 m) were collected three times per year at or near maximum lake depth. Lakes were sampled in spring within two weeks of ice-out (late May to early June), in mid-summer (July), and early fall (late August to mid-September). Samples were filtered within 12 hours when necessary (Gelman 0.45 µm filter membrane), stored at 4°C, and usually analyzed within 48 hours using standard methods (Carignan et al., *this volume*). Measured properties included temperature, light penetration, dissolved oxygen, dissolved organic carbon (DOC), total phosphorus and nitrogen (TP, TN), nitrate (NO_3^-), ammonium (NH_4^+), major ions (Ca^{2+}, Mg^{2+}, Na^+, K^+, SO_4^{2-}, Cl^-), conductivity, pH, and alkalinity.

Estimation of drainage area element export rates

Drainage area element export rates (*D*; $mol\cdot$ or $eq\cdot ha^{-1}\cdot y^{-1}$) were estimated by retrocalculation (see below) using the time series of element concentration from the lakes. The first step involved in retrocalculation was to estimate runoff from the lake basins. The lake basins are remote and could not be gauged with traditional hydrological techniques. Runoff from each lake basin was estimated using three gauged river basins with natural flow regimes in the study area (the Gatineau, Bell, and Chamouchouane rivers; Table 2). Three time intervals were defined, corresponding to the mid-point in time between lake sampling surveys (*spring* – 1 January to 15 June; *summer* – 15 June to 15 August; *fall* – 15 August to 31 December). For each lake basin, runoff (*r*) for a given time interval was the mean from the reference river basins weighed as a function of the distance between the lake basin and the geographical center of the reference river basins:

$$r_{i,j,k} = \frac{\sum_{x=1}^{3}\left(\frac{1}{d_{i,x}^{2}}\cdot r_{j,k,x}\right)}{\sum_{x=1}^{3}\frac{1}{d_{i,x}^{2}}} \qquad (1),$$

where $r_{i,j,k}$ is runoff from the i^{th} lake basin for the j^{th} year and k^{th} sampling interval, $r_{j,k,x}$ runoff from the x^{th} reference river basin for given period, and $d_{i,x}$ the distance between the i^{th} lake basin and the x^{th} reference river basin.

Impact of harvesting and fires on runoff

Forest harvesting and forest fires usually increase runoff from Precambrian Shield basins (Verry 1986; Plamondon 1993). In general, runoff is more strongly affected when

Table 2. Characteristics of the reference river basins used to estimate runoff from the study lake basins. All reference river basins have natural flow regimes. Coordinates are for the geographical center of the river basins (not the gauging station locations). Daily runoff from 1996 to 1998 obtained from: Direction du milieu hydrique, Ministère de l'Environnement et de la Faune, 2360 chemin Sainte-Foy, Sainte-Foy (Québec), G1V 4H2.

	Bell	Gatineau	Chamouchouane
Long. (N)	76°38'	75°17'	73°27'
Lat. (W)	48°41'	47°44'	49°24'
Environment Canada Gauging Station No.	03AC004	02LG005	02RF001
Basin area (km^2)	22 200	6 840	15 300
1996 runoff (mm)	533	544	609
1997 runoff (mm)	641	692	750
1998 runoff (mm)	463	460	546
Long-term mean runoff	555	582	623

Table 3. Location of four precipitation monitoring stations bounding the Haute-Mauricie region, with the wet-only deposition of selected elements. Monthly deposition for 1990 to 1996 obtained from: Direction du milieu atmosphérique, Ministère de l'Environnement et de la Faune, 675 boul. René-Lévesque est, 5^{ieme} étage, Québec (Québec), G1R 5V7.

	La Morandière	Lebel-sur-Quevillon	Parent - S	Pémonca
Environment Canada Station No.	7094026	7094275	7075799	7065900
Long. (N)	77°37'	76°58'	74°37'	72°42'
Lat. (W)	48°37'	49°03'	47°55'	48°45'
SO_4^{2-} (eq ha^{-1} y^{-1})	325	269	262	174
NO_3^- (eq ha^{-1} y^{-1})	174	134	140	123
Ca^{2+} (eq ha^{-1} y^{-1})	55	50	35	45
H^+ (eq ha^{-1} y^{-1})	310	220	260	250
Cl^- (eq ha^{-1} y^{-1})	14	23	17	23

evapotranspiration is an important component of the hydrological budget. In Haute-Mauricie, evapotranspiration (~475 mm) is an important proportion of precipitation inputs (~1 000 mm) and tree removal by harvesting or fires should yield an important increase in yearly runoff. Plamondon (1993) estimated that the potential for an increase in runoff was 200 mm following harvesting in this region (i.e., a 38% increase of total yearly runoff). We used Plamondon's estimate as the increase in runoff from drainage areas following 100% tree removal. In each disturbed drainage area, the increase in runoff was proportional to the area burnt or harvested (i.e., a 2 mm increase in runoff for each percent of the drainage area burnt or harvested). The increase in runoff was equally distributed between the three yearly sampling intervals.

Wet deposition to lake surfaces

Direct deposition to lake surfaces is an important component of lake nutrient budgets and was estimated in a manner similar to lake basin runoff. The Quebec Ministry of the Environment maintains a network of wet-only deposition chemistry stations throughout the Province (Jacques and Boulet 1988). Four stations bounding the study area were used to estimate direct precipitation inputs to the lakes surfaces (Table 3). Atmospheric deposition (*P*; mol· or eq·ha^{-1}·y^{-1}) to the surface of each lake was determined using the same weighing procedure as the one used for estimating runoff (but with four reference stations instead of three reference river basins). A gradient of SO_4^{2-} deposition occurs from the west to the east of the study area, in part because of the occurrence of smelters approximately 300 km to the west, in the Abitibi region. The provincial network measures the input of H^+, major cations, Cl^-, SO_4^{2-}, NH_4^+, and NO_3^-. Total phosphorus, DOC, and TN inputs were estimated from the literature (Table 4). Precipitation chemistry data were available only for the first year of the study (1996); the average deposition from 1990 – 96 was used for the 1997 – 98 budgets.

Table 4. Precipitation input of TP, TN, and DOC at various locations on the Canadian Precambrian Shield.

	TP (mol ha^{-1} y^{-1})	TN (mol ha^{-1} y^{-1})	DOC (mol ha^{-1} y^{-1})	Reference
ELA, NW ON	8.2	400	2300	1
Harp Lake, ON	24	1143		2
South-Central ON	9.3 - 13	714 – 786		3
NW Ontario	2.9			4
Clear Lake, ON	11			5
Harp Lake, ON	5.7		725	6
Used in this study	6	400	1000	

1 – Linsey et al. 1987; 2 – Nicholls and Cox 1978; 3 – Scheider et al. 1979; 4 – Foster 1974; 5 – Schindler and Nighswander 1970; 6 – Dillon and Molot 1997.

Retrocalculation of D from a lake mass-balance model

The general element mass-balance for the lakes was defined as:

$$\frac{d[x]}{dt} = \left(J_D + J_P - J_o - S \right) / V \qquad (2),$$

where $d[x]/dt$ is the change in lakewater concentration of chemical species x over time ($\mathrm{mol \cdot m^{-3} \cdot y^{-1}}$), J_D the load from the drainage area ($\mathrm{mol \cdot y^{-1}}$), J_P the load from atmospheric deposition to the lake surface, J_o the loss via outflow, S the apparent sedimentation loss ($\mathrm{mol \cdot y^{-1}}$), and V the lake volume ($\mathrm{m^3}$). By rearranging and setting (2) as a finite difference equation:

$$V \cdot \Delta[x] = J_D + J_P - J_o - S \qquad (3),$$

where $\Delta[x]$ *is the fall concentration of x minus the fall concentration of x in the previous year*. Thus, $V \cdot \Delta[x]$ (or ΔM) is the change in the mass of x in the lake for a given year (in $\mathrm{mol \cdot y^{-1}}$ or $\mathrm{eq \cdot y^{-1}}$). Concentration data was not available for 1995. We first assumed that fall concentrations in 1995 and 1996 were similar in reference lakes. This approximation is reasonable because only small changes in element concentration were observed in the reference lakes between 1996 and 1998 (Carignan et al., *this volume*). Two approaches were taken to estimate ΔM in 1996 in harvested and burnt basins. First, empirical models relating basin morphometry, latitude, and longitude to fall element concentrations were evaluated for the reference lakes. For the chemical species with a strong correlation with basin location or morphometry (SO_4^{2-}, TP, and DOC), models were used to estimate pre-impact fall concentrations in 1995 in the harvested and burnt basins (Table 5). For the other species (Ca^{2+}, Mg^{2+}, Na^+, K^+, NO_3^-, Cl^-, and TN), fall 1995 concentrations were approximated as the average fall concentration in 1996 in reference lakes.

Table 5. Empirical models used to estimate fall 1995 SO_4^{2-}, TP and DOC concentrations for lakes in harvested and burnt basins. The models were determined by relating fall 1996 element concentration in 20 reference lakes to basin morphometry, latitude, and longitude using least-square multiple regression. Concentrations in $\mathrm{mg \cdot L^{-1}}$ for SO_4^{2-}-S and DOC and $\mathrm{mg \cdot L^{-1}}$ for TP. Latitude (*lat*) and longitude (*long*) expressed in decimal form (i.e., 48°30'N = 48.50) to the north and west respectively. Z_m – mean depth (m); τ – estimated water residence time ($\mathrm{y^{-1}}$).

Model	r^2	P	SE of estimate
$\log \mathrm{DOC} = 0.513 + 0.268 \cdot \log (DA{:}LA)^{*}$	0.25	0.025	0.0987
$\log SO_4^{2-}\text{-S} = -6.71 + 0.0869 \cdot long^{***} + 0.233 \cdot \log (Z_m)^{**}$	0.72	0.000	0.0524
$\log \mathrm{TP} = 5.30 - 0.300 \cdot \log (\tau)^{***} - 0.0925 \cdot lat^{*}$	0.61	0.000	0.072

* $P < 0.05$

** $P < 0.01$

*** $P < 0.001$

We assumed that outflow concentrations equaled euphotic zone concentrations and calculated the annual loss of dissolved species at the outflow as:

$$J_o = \sum_{t=1}^{3}\{ (r_t \cdot LA + r_t \cdot DA + r_t \cdot DA \cdot f \cdot E) \cdot [x]_t \} \qquad (4),$$

where r_t is the runoff estimated for a given sampling interval (m), f the proportion of the drainage area cut or burnt, E the coefficient of increase in runoff following harvesting or fire (0.0666 m·[sampling interval]$^{-1}$ or 0.2 m·y^{-1}), and $[x]_t$ the euphotic zone element concentration for a given sampling interval.

To estimate sedimentation losses (*S*), chemical species were classed into 'conservative' (Ca^{2+}, Mg^{2+}, Na^+, K^+, Cl^-) and 'non-conservative' (NO_3^-, SO_4^{2-}, DOC, TP, TN) groups based on their expected in-lake behavior. For non-conservative chemical species, it was assumed that some retention occurred during lake transit. A search of the literature was made to obtain representative mass-transfer coefficients (*v*) in Shield lakes for each species (Table 6). Mass transfer coefficients (in m y^{-1}) are usually determined empirically from mass-balance budgets and can be visualized as a net yearly flux from the water column to the sediments proportional to the mass of a given solute contained in *v* meters of the water column. This is an apparent sedimentation rate because losses through other processes (phytoplankton uptake, transformation to other forms, etc) are also included but do not necessarily involve 'sedimentation' to the lake bottom. Using similar approximations as in Carignan and Tessier (1988), a second estimate of *v* for SO_4^{2-} and NO_3^- was obtained by estimating their reduction rate in Shield lake sediments (Table 6). For conservative chemical species *S* always equaled zero. For non-conservative species:

$$S = LA \cdot v \cdot \overline{[x]} \qquad \text{(Reckow and Chapra 1983; 5),}$$

where $\overline{[x]}$ is the average euphotic zone concentration of a given chemical species during a given year.

By setting J_D as $D{\cdot}DA$ and, similarly, J_P as $P{\cdot}LA$ (*P* = atmospheric deposition rate; mol· or eq·ha^{-1}·y^{-1}), and by rearranging eq. 3 to solve for *D*:

$$D = \frac{\Delta M + J_O + S - P \cdot LA}{DA} \qquad (6).$$

In the second-order lake basins, however, eq. 6 will tend to overestimate *D* if direct element deposition to upstream lakes is an important part of budgets, or will underestimate *D* if sedimentation loss in upstream lakes is important. Thus, for second-order lake basins, a modification of eq. 6 was used to include direct atmospheric deposition to, and element retention within, upstream lakes. In a first step, the *DA* of each second-order basin was separated into a direct drainage area (*DDA*) and, for each upstream lake, in sub-basins. In a second step, retention (*R*; [mass in – mass out] / mass in) in upstream lakes had to be evaluated with a model that did not require that element concentrations in the lakes was known. With a steady-state approximation (i.e., by assuming ΔM~0 in upstream lakes), the proportion of the load from precipitation and the drainage area that was retained in upstream lakes could be evaluated from the hydrological properties of the sub-basins using:

$$R = \frac{v}{v + q_s} \qquad \text{(Rechow and Chapra 1983; 7),}$$

where q_s is the areal water discharge ($r \cdot [LA + DA] / LA$) for a given upstream lake sub-basin. Assuming that *D* is constant in the *DDA* and in the *DA* of each sub-basin, for the case with *k* sub-basins eq. 6 modifies to:

$$D = \frac{\Delta M + J_O + S - P \cdot \{ \sum_{k=1}^{n} [LA_k \cdot (1 - R_k)] + LA \}}{\sum_{k=1}^{n} [DA_k \cdot (1 - R_k)] + DDA} \qquad (8),$$

where *LA* is the lake area of the second-order lake, LA_k the lake area in sub-basin *k*, DA_k the drainage area of sub-basin *k*, and R_k the retention coefficient of the lake in sub-basin *k*.

Although eq. 8 appears a drastic modification to eq. 6, with the exception of the Lake FP15 basin, the *D* estimates obtained with either equation were similar. Using eq. 8 and excluding Lake FP15, the average *D* for second-order basins was higher by 1% for SO_4^{2-}, 5% for DOC, 9% for NO_3^-, 13% for TN, and 13% for TP than when using eq. 6. For Lake FP15 only, *D* was higher by 3%, 18%, 112%, 28%, and 62% for the same chemical species respectively when using eq. 8. Although using a steady-state model to estimate *R* may not be accurate for the disturbed basins, with the exception of Lake FP15, the potential for direct precipitation inputs and retention within upstream lakes is small.

Estimation of flux per unit of disturbed drainage area

Drainage areas were impacted to various degrees by harvesting or fires (Table 1). To compare impacts between basins, element export rates standardized on a per unit of disturbed drainage area (D_d) were estimated. D_d was estimated using:

$$D = F_d \cdot D_d + F_u \cdot D_u \qquad (9),$$

where F_d and F_u are the proportion of the drainage area that is disturbed and undisturbed respectively, and D_u is the drainage area element flux rate expected under undisturbed conditions. Rearranging eq. 9 yields,

$$D_d = \frac{D - F_u \cdot D_u}{F_d} \qquad (10).$$

D_u was approximated as the average D in the reference basins during a given year. Because estimates of D_d could have a large error when F_d is low, D_d for lakes C40 and C29 (which only had 8.5% and 10% of their drainage area harvested respectively) was not estimated.

The statistical significance for the difference in D_d between harvested and burnt basins was evaluated using *t*-tests. Because there was a limited overlap in the proportion of the drainage areas impacted by each type of disturbance, the tests were only aimed at determining the likelihood that the observed differences were due to chance (i.e., not to "prove" in a statistical sense that Shield basins respond differently to fire or harvesting). The statistical comparison between D_d from impacted basins and D from reference basins was more complex because the average D from reference basins is already used to estimate D_d. For this later comparison, the statistical significance for the difference in fluxes was evaluated using a permutation test (Legendre and Legendre 1998). For each trial (10 000 in total), reference basins were randomly split in two groups. The mean D in one subset of reference basins was used to estimate D_d in disturbed basins. The second subset of reference basins was used in a permutation test – which yields a result similar to a *t*-test (Legendre and Legendre 1998).

Table 6. Mass transfer coefficients for several chemical species in Shield lakes.

SO_4^{2-} ($m\ y^{-1}$)	NO_3^- ($m\ y^{-1}$)	TN ($m\ y^{-1}$)	TP ($m\ y^{-1}$)	DOC ($m\ y^{-1}$)	Reference
0.4 – 0.54	7.4 – 9.2				1
	2.4 – 12.5	3.0 – 4.5			2
			4.7 – 13.7		3
				2.3 – 4.1	4
				2.7	5
0.64	6.6				6
0.40	7.4	3.0	7.9	2.3	*Used in this study*

1 – Kelly et al. 1987; 2 – Molot and Dillon 1993; 3 – Dillon and Molot and Dillon 1997b; 5 – Curtis and Schindler 1997; 6 – from estimated reduction rates in Shield lake sediment.

Scenarios of element loss from drainage areas

Increased runoff following tree removal and sedimentation losses in lakes were expected to be important factors when determining element export rates from drainage areas (eqs 4 and 5). However, runoff and element retention in lakes could only be approximated with our study design. We therefore *D* and D_d estimated using four scenarios of catchment runoff and element retention in lakes. The four scenarios were: 1) no increase in runoff occurred following fires or harvesting and no retention in lakes; 2) only an increase in runoff occurred; 3) only retention in lakes occurred; 4) both an increase in runoff and retention in lakes occurred. These scenarios were designed to provide a plausible range of element export rates from the drainage areas.

DATA ANALYSIS

Harvesting and fires generally increased element export from harvested and burnt drainage areas relative to reference drainage areas (Fig. 1). In harvested basins, the magnitude of the increase in *D* was positively related to the percent drainage area cut for Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}, Cl^{-}, and TP (Fig. 1). In burnt basins, relationships between the percent of the drainage area disturbed and *D* could not be tested because most drainage areas were nearly completely burnt (Fig. 1). With the exception of TN and one outlier for Cl^{-}, export rates from reference drainage areas had a smaller range than in disturbed drainage areas.

When element export rates were standardized on a per unit drainage area disturbed (D_d), some similarities and contrasts were found between harvested and burnt drainage areas (Fig. 2). In general, D_d values were similar between burnt and harvested drainage areas for K^{+} (3 to 8-fold reference), TN (2 to 3-fold reference), and TP (2-fold reference). However, D_d in burnt drainage areas was higher for divalent base cations (Ca^{2+} and Mg^{2+}; 2 to 3-fold reference), SO_4^{2-} (3 to 4-fold reference), and NO_3^{-} (up to 17-fold reference). D_d in harvested drainage areas was higher for Na^{+} (0.4 to 2.0-fold reference) and DOC (2 to 3-fold reference). D_d for Cl^{-} was initially highest in burnt drainage areas but decreased below rates in harvested drainage areas by 1998.

D_d from harvested and burnt drainage areas were highest for most elements in 1996 and tended to decline in 1997 and 1998 (Fig. 2). Noticeable departures from this trend were small peaks in NO_3^{-} export from harvested areas and in Na^{+} and DOC exports from burnt areas in 1997. The export of divalent cations, Na^{+}, SO_4^{2-}, NO_3^{-}, and DOC from reference drainage areas was also highest in 1997, the year with the highest runoff (Table 2). In 1998, three years after fires or harvesting had occurred, export rates from harvested and burnt drainage areas tended to be still above rates in reference drainage areas.

Cumulative exports with different mass-balance scenarios

The cumulative exports (as D_d for harvested and burnt basins) from 1996 to 1998 were evaluated for the four scenarios of runoff and element retention in lakes (Table 7). In general, the cumulative export of conservative chemical species (i.e., Ca^{2+}, Mg^{2+}, K^+, Na^+, and Cl^-) increased by 17% to 30% when the possibility of increased runoff was taken into account. For non-conservative species, whether increased runoff (scenario 2) or loss through sedimentation (scenario 3) yielded the largest cumulative export depended on the v of a particular species (Table 7). For species with a low v (i.e., SO_4^{2-}), increased runoff was more important than sedimentation losses. For species with a high v (i.e., TN and TP), sedimentation losses were more important than increased runoff (Table 7). Cumulative exports were highest when both increased runoff and element retention in lakes were taken into account (scenario 4).

Comparison of cumulative exports estimated from the four mass-balance scenarios reveals that the main patterns in element export following harvesting or fires were independent of the two most important assumptions used in the model. Even using the most conservative scenario 1, the cumulative exports (as D_d) from burnt areas for Ca^{2+}, Mg^{2+}, K^+, SO_4^{2-}, NO_3^-, Cl^-, TP, and TN were still 1.6- to 8-fold the cumulative export from reference drainage areas (Table 7). Also with scenario 1, the cumulative exports (as D_d) from harvested areas for Na^+, K^+, Cl^-, TP, TN, and DOC were still 1.6- to 4.6-fold the cumulative export from reference drainage areas.

Uncertainty on TP flux estimates

Despite a relatively small error on average fluxes (Fig. 2), TP export estimates are the most uncertain because both precipitation input to lake surfaces and sedimentation losses were important components of the budgets (Table 8). Both the atmospheric deposition of TP per unit lake surface (P_{TP}) and the TP mass-transfer coefficient (v_{TP}) were taken from the literature and may be biased estimates for our lake population. A sensitivity analysis was made to determine the influence of variation in P_{TP} and v_{TP} on TP flux estimates. The range for natural variability in both parameters was estimated as ±50% (Tables 4 and 5). In addition to the sensitivity analysis, D for TP (D_{TP}) was also estimated using Prairie's (1988) model. Prairie's model assumes that net TP sedimentation is a function of both the load and lake content of TP (instead of just the lake content in eq. 6). This approach appears conceptually more sound for this element (Prairie 1988, 1989).

Variability in v_{TP} and P_{TP} substantially affected the magnitude of the estimated TP export rates in harvested, burnt and reference drainage areas (Table 9). The mean D_{TP} for reference drainage areas was the most sensitive to variability in P_{TP} and v_{TP} (range = 1.2 to 8.0 mol ha^{-1} y^{-1} or 3.7 to 25 mg m^{-2} y^{-1}). The estimates of D_{TP} obtained with Prairie's model were generally lower than the estimates of D_{TP} obtained with our model (Table 9). The discrepancy between models was largest in reference basins, where the average D_{TP} ranged from 3.8 to 4.8 mol·ha^{-1}·y^{-1} (12 to

Table 7. Cumulative exports from 1996 to 1998 from harvested and burnt drainage areas (as D_d) and from reference drainage areas in Haute-Mauricie according to four mass-balance models: 1) – no sedimentation losses and no increased runoff in disturbed areas; 2) – with increased runoff; 3) – with sedimentation losses; 4) – with both increased runoff and sedimentation losses.

Model	Ca^{2+} (eq ha^{-1})	Mg^{2+} (eq ha^{-1})	Na^{+} (eq ha^{-1})	K^{+} (eq ha^{-1})	SO_4^{2-} (eq ha^{-1})	NO_3^{-} (eq ha^{-1})	Cl^{-} (eq ha^{-1})	TP (mol ha^{-1})	TN (mol ha^{-1})	DOC (mol ha^{-1})
					Harvested					
1 ($v = 0$; $E = 0$)	1 900	700	940	580	710	<15	200	7.7	360	20 000
2 ($v = 0$; $E \neq 0$)	2 300	910	1 100	680	990	<15	240	9.6	460	25 000
3 ($v \neq 0$; $E = 0$)	1 900	700	940	580	710	<20	200	23	590	28 000
4 ($v \neq 0$; $E \neq 0$)	2 300	910	1 100	680	990	<20	240	25	710	32 000
					Burnt					
1 ($v = 0$; $E = 0$)	2 500	1 500	610	540	2 000	75	210	6.8	330	9 700
2 ($v = 0$; $E \neq 0$)	3 200	1 800	780	660	2 600	91	260	9.0	450	13 000
3 ($v \neq 0$; $E = 0$)	2 500	1 500	610	540	2 200	136	210	21	580	15 000
4 ($v \neq 0$; $E \neq 0$)	3 200	1 800	780	660	2 700	154	260	23	710	18 000
					Reference					
1 ($v = 0$; $E = 0$)	1 600	760	580	110	870	<15	71	1.4	79	7 700
3 ($v \neq 0$; $E = 0$)	1 600	760	580	110	1 010	<15	71	13	350	14 000

15 mg·m^{-2}·y^{-1}) with our model (using v_{TP} = 7.9 and P_{TP} = 6) and from 1.2 to 1.5 mol·ha^{-1}·y^{-1} (3.7 to 4.7 mg·m^{-2}·y^{-1}) with Prairie's model. Relative to reference basins, the increase in D_{TP} in harvested and burnt basins was largest with Prairie's model (Table 9). Although the magnitude of D_{TP} varied substantially using different models, or sets of parameters, the tendency for D_{TP} in harvested and burnt drainage areas to be larger than in reference drainage areas was always maintained.

Table 8. Mean values for the different components of the element mass-balance for harvested, burnt, and reference basins for Ca^{2+}, TP, and DOC in 1996 (using scenario 4). All rates standardized in mol or eq·[ha of drainage area]$^{-1}$·y^{-1}.

	Harvested (n = 9)	Burnt (n = 9)	Reference (n = 16)
		Ca^{2+} (eq·[ha drainage area]$^{-1}$ ·y^{-1})	
$+ J_o / DA$	577	909	498
$- J_P / DA$	8.2	9.6	12
$+ S / DA$	0	0	0
$+ \Delta M / DA$	7.0	368	0*
D	576	1 267	486
		TP (mol·[ha drainage area^{-1}]·y^{-1})	
$+ J_o / DA$	2.4	3.4	1.7
$- J_P / DA$	1.2	1.1	1.5
$+ S / DA$	4.6	4.1	4.2
$+ \Delta M / DA$	0.61	0.70	0*
D	6.4	7.1	4.4
		DOC (mol·[ha drainage area]$^{-1}$·y^{-1})	
$+ J_o / DA$	4 863	4 009	2 963
$- J_P / DA$	199	182	249
$+ S / DA$	2 407	1 494	2 205
$+ \Delta M / DA$	896	-186	0†
D	7 967	5 135	4 919

† By definition in reference basins for 1996

y^{-1}). Mean fluxes in mol P·[ha of drainage area]$^{-1}$·y^{-1}.

	Equation 6			Prairie (1988)
	Minimum flux $P_{TP} = 9$ $v_{TP} = 3.95$	Used in the study $P_{TP} = 6$ $v_{TP} = 7.9$	Maximum flux $P_{TP} = 3$; $v_{TP} = 11.9$	
		1996		
Harvested	3.5	6.4	9.3	4.0
Burnt	4.3	7.1	9.9	5.1
Reference	1.5	4.4	7.3	1.3
		1997		
Harvested	3.0	5.7	8.5	3.4
Burnt	4.7	7.7	11	5.6
Reference	1.2	3.8	6.4	1.2
		1998		
Harvested	2.7	5.7	8.7	2.9
Burnt	4.2	7.3	10	4.8
Reference	1.8	4.8	7.8	1.5

Contrasts in nutrient export mechanisms between harvested and burnt basins

The patterns in N, P, and S flux between harvested and burnt drainage areas may be more complex than our measurements suggest because these elements can be exported in both organic and inorganic forms. For example, assuming a DOC:DON ratio of ~55 in Shield stream water (Allan et al. 1993), DON exports represented ~82% of cumulative TN exports for harvested drainage areas, ~46% for burnt drainage areas, and ~73% for reference drainage areas. Thus, while cumulative TN exports were similar between burnt and harvested drainage areas, N was exported mostly in organic forms following harvesting and in both mineral and organic forms following fires. Although the loading of nutrients increased in both harvested and burnt basins, zooplankton (Patoine et al., *this volume*) and benthic algae (Planas et al., *this volume*) responded differently to the perturbations. We hypothesize that the different response of lake biota to the perturbations was caused, in part, by nutrients being supplied in different forms in harvested and burnt basins.

Different mechanisms were responsible for the transport of cations from the forest to the lakes in harvested and burnt basins. The magnitude of cation export from drainage areas is controlled by the availability of mobile anions (Reuss and Johnson 1986). In reference drainage areas, dissociated organic acids (i.e., a component of DOC) and SO_4^{2-} were the two important anions in runoff (Table 10). In contrast, SO_4^{2-} was the dominant anion in runoff from burnt drainage areas and dissociated organic acids the main anions in runoff from harvested drainage areas (Table 10). Even when their export increased, Cl^- and NO_3^- were only a small component of the anion flux (Table 10). Thus, although cation export increased in both harvested and burnt drainage areas, the mechanisms involved were different. Increased cation export was caused by increased leaching of inorganic anions in burnt basins and increased leaching of organic anions in harvested basins.

Table 10. Cumulative cation and anion flux between 1996 and 1998 in harvested (as D_d), burnt (as D_d), and reference drainage areas in Haute-Mauricie. All fluxes in eq·ha^{-1}.

	Harvested	Burnt	Reference
Ca^{2+}	2 300	3 200	1 600
Mg^{2+}	910	1 800	760
Na^+	1 100	780	580
K^+	680	660	110
Cation flux	4 990	6 440	3 050
SO_4^{2-}	990	2 700	1 010
NO_3^-	5	150	1
Cl^-	240	260	71
Mineral anion flux†	1 235	3 110	1 082
Cations – mineral anions	3 755	3 330	1 968
Organic anions‡	2 700 – 3 200	1 400 – 1 700	1 200 – 1 400

† excluding HCO_3^-

‡ Estimated using the method of Oliver et al. (1983) for a pH of runoff varying between 5.0 (the lower value) and 6.0 (the higher value).

Element export from reference drainage areas

Element export rates estimated from Haute-Mauricie reference drainage areas fall within the range found in other small Precambrian Shield basins (Table 11). In general, the pattern in the export of major cations and anions from Haute-Mauricie drainage areas is intermediate between patterns found at the Experimental Lakes Area (ELA) and the Turkey Lakes Area (TLA), both in Ontario. The ELA is located in a relatively pristine area of the Precambrian Shield (Schindler et al. 1976) while the TLA receives important S inputs of anthropogenic origin (Clair et al. 1995). At the three sites, Ca^{2+} and Mg^{2+} are the main cations in runoff (Table 11). However, the amount of Ca^{2+} exported is variable between sites and appears related to SO_4^{2-} export. Calcium and SO_4^{2-} exports are lowest at the ELA, intermediate in Haute-Mauricie, and highest at the TLA. A gradient in SO_4^{2-} deposition also occurs within the Haute-Mauricie region and SO_4^{2-} export rates in reference basins were positively related to wet SO_4^{2-} deposition rates (export [eq·ha^{-1}·y^{-1}] = -65 + 1.5·wet deposition [eq·ha^{-1}·y^{-1}]; $P = 0.02$; $r^2 = 0.34$). Most reference Haute-Mauricie drainage areas exported more SO_4^{2-} than was supplied by wet deposition, suggesting an important dry deposition component or desorption of past S inputs (Houle and Carignan 1995). Patterns in cation and anion flux are markedly different between Haute-mauricie and the Sogndal catchments in Norway, which are under a maritime influence (Table 11). Overall, the patterns in element flux in reference Haute-Mauricie drainage areas is consistent with a continental Precambrian Shield setting subjected to past or present anthropogenic S inputs.

Table 11. Element export in runoff from Precambrian Shield basins. The Turkey lakes and Dorset areas receive moderate to elevated acid rain inputs. The Sogndal catchment is under maritime influence (i.e., elevated sea-salt inputs). Fluxes in eq ha^{-1} y^{-1} or mol ha^{-1} y^{-1}.

	Haute-Mauricie, PQ (mean and range)	ELA, ON	Turkey Lakes, ON	Dorset, ON (mean and range)	Sogndal, Norway
Ca^{2+}	521 (353 – 823)	232	1 188	–	161
Mg^{2+}	253 (175 – 389)	174	236	–	91
Na^{+}	193 (126 – 270)	116	157	–	425
K^{+}	37 (10 – 86)	20	46	–	35
SO$_4^{2-}$	339 (212 – 516)	76	767	–	209
NO$_3^{-}$	-- (<5 – 8)	2.5	211	–	9
Cl^{-}	24 (17 – 70)	15	61	–	409
TP	4.3 (2.8 – 6.4)	1.3	–	2.6 (0.6 – 8.2)	–
TN	116 (40 – 174)	66	–	–	–
DOC	4 800 (3 130 – 6 890)	2 900	–	3 900 (825 – 7 600)	–
Reference	1	2,3	4	5	6

1 – this study; 2 – Schindler et al. 1976 (Rawson L., northwest inflow); 3 – Bayley et al. 1992 (Rawson L. northwest inflow); 4 – Nicolson 1988b; 5 – Dillon and Molot 1990; 6 – Frogner 1989.

MANAGEMENT APPLICATIONS

Significance of nutrient export in runoff to forest and lake nutrient budgets

The cumulative loss of nutrients in runoff for the three years following the disturbances was compared to losses through volatilization during fires and through biomass removal during harvesting (Table 12). The losses in runoff used here are minimum estimates because elevated exports will likely occur for several more years. Potassium was the element with the highest loss in runoff relative to volatilization or biomass removal (25% to 66% of total losses; Table 12). Calcium and Mg losses in runoff were of intermediate importance, representing 8% to 40% of total losses. Losses of N and P in runoff were negligible relative to losses through volatilization or biomass removal (1% - 5% of total losses). Similar patterns in base cations exports were observed following forest fires (Bayley and Schindler 1991) and harvesting (Nicolson et al. 1982) at the ELA. Potassium export in runoff following fires and harvesting can be a significant loss for boreal Shield forests.

While the increase in N and P export in runoff following harvesting or fires appeared a negligible loss of nutrient for the forest, they represented a significant input to lakes. The load of

TN and TP to lakes in burnt and harvested basins increased by 50% on average following the disturbances, with the loads doubling in some lakes (Lamontagne, unpublished data). While the P load is a strong determinant of total primary production in Shield lakes (Schindler 1974), the N:P ratio can influence the composition of algal populations (Schindler 1977). Although TP export rates per unit of harvested or burnt drainage area were similar, additional work is required to determine in which form P is exported to lakes following fires or harvesting.

Clear-cuts over large areas are often justified on the grounds that natural disturbance, such as forest fires, also remove trees over large surfaces. However, for the export of elements from forests to lakes, forest harvesting and forest fires are not similar disturbances in Haute-Mauricie. The largest difference between these disturbances is an important increase in DOC following harvesting but not following fires. DOC from terrestrial sources is colored and influences the optical and physical properties of small lakes (Fee et al. 1996; Schindler et al. 1996). Lower light penetration and shallower thermoclines may affect algal production and the volume of lake water available to some aquatic organisms. Increased DOC export following harvesting could be a vector for the transport of some contaminants to lakes, such as methyl mercury (Garcia and Carignan 1999). Carignan et al. (1999) have provided a quantitative relationship linking DOC to an impact factor defined as the harvested area in the watershed : lake volume ratio. They have found that DOC will not exceed the range of natural variability, and that excess DOC (and presumably mercury) problems can be avoided if this ratio does not exceed 0.5. This guideline can easily be applied in the field to potentially sensitive lakes by limiting harvest size in a given pass.

Table 12. Nutrient losses following forest fires or harvesting in boreal Shield forests. All data in kg ha^{-1}.

	Fire		Harvesting		
	Volatilization†	Runoff§	*Picea mariana*¶	*Betula papyrifera*¶	Runoff§
Ca	73	32	130 – 190	93 – 210	14
Mg	20	13	6 – 15	14 – 31	1.8
K	14	22	51 – 85	37 – 91	22
N	456	5.0	28 – 110	84 – 160	5.0
P	<40‡	0.4	34 – 47	4 – 14	0.3

† Estimated from soil pits (forest floor and 0-20 cm mineral soil) in areas burnt during the 1995 Belleplage fire relative to nearby reference areas in 1997 (Paré, unpublished data). Volatilization losses were approximated as the difference in nutrient mass between soil pits from burnt and reference areas, minus the cumulative nutrient losses in runoff from burnt areas (this study) for 1996 and 1997. The Belleplage fire occurred within the study area.

‡ From MacLein et al. 1983.

§ Difference in cumulative exports between disturbed and reference drainage areas from 1996 to 1998 in this study.

¶ Bole (lower boundary) and whole-tree (upper boundary) biomass at the time of harvest for pure stands in Québec; (Paré, unpublished data from several sources).

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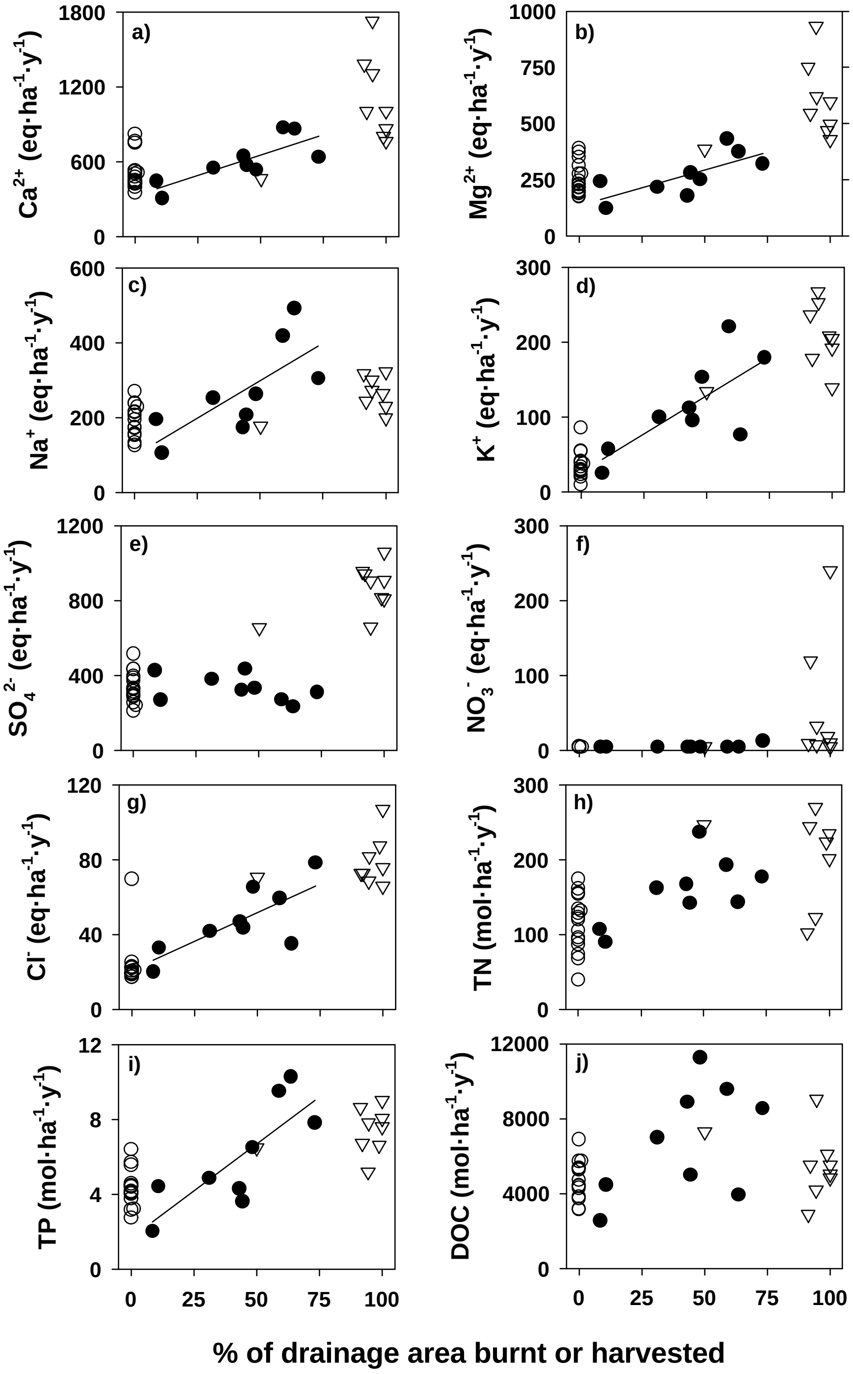

Figure 1. Element export rates from harvested (dark circles), burnt (open triangles), and reference drainage areas (open circles) in Haute-Mauricie (mean rates for the three years following fires or harvesting). Least-square linear regressions between element export rates and the percent of drainage area disturbed were evaluated for the harvested basins (results in brackets in the following). a) Ca^{2+} ($P = 0.01$; $r = 0.80$); b) Mg^{2+} ($P = 0.03$; $r = 0.73$); c) Na^{+} ($P = 0.03$; $r = 0.73$); d) K^{+} ($P = 0.02$; $r = 0.74$); e) SO_4^{2-} (n.s.); f) NO_3^{-} (n.s.); g) Cl^{-} ($P = 0.02$; $r = 0.76$); h) TN (n.s.); i) TP ($P = 0.01$; $r = 0.80$); j) DOC (n.s.).

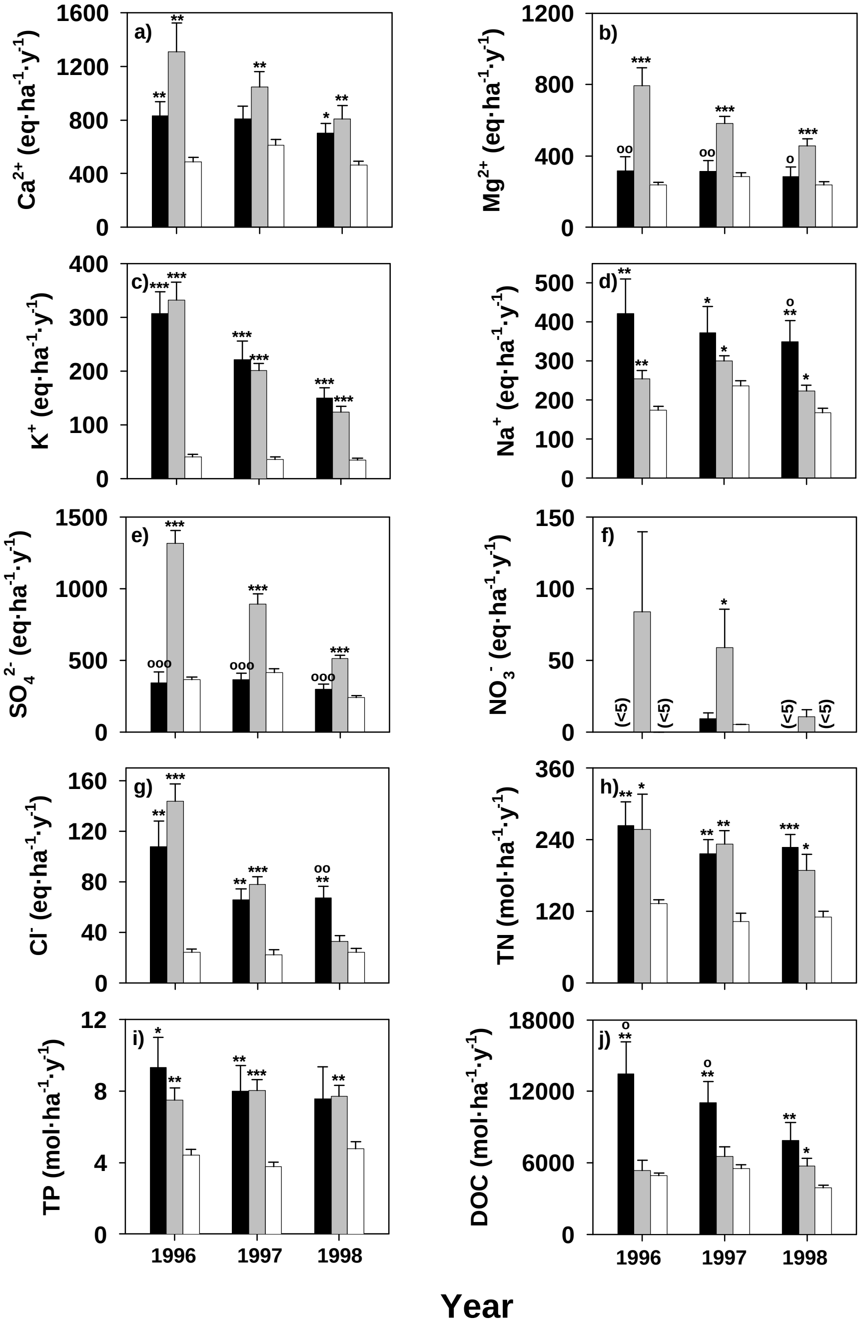

Figure 2. Annual export rates from harvested (solid) and burnt (gray) drainage areas corrected for the percent of the drainage area disturbed. Statistical significance tests between disturbed and reference drainage areas (in white) expressed with (*) and between harvested and burnt drainage areas with ('o'). * or 'o' = $P < 0.05$; ** or 'oo' = $P < 0.01$; *** or 'ooo' = $P < 0.001$.