

Impact of logging and natural fires on fish communities of Laurentian Shield lakes

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Abstract: The goal of this study was to determine if natural fires and logging have a significant impact on abundance, growth, and size structure of fish populations in 38 lakes of the Laurentian Shield (Québec, Canada). The watersheds of nine of these lakes underwent logging and nine underwent natural fires, while the 20 remaining lakes were used as references. No significant differences were found among the three lake groups in the catch per unit of effort of the most abundant species: white sucker (*Catostomus commersoni*), northern pike (*Esox lucius*), yellow perch (*Perca flavescens*), lake whitefish (*Coregonus clupeaformis*), fallfish (*Semotilus corporalis*), brook trout (*Salvelinus fontinalis*), walleye (*Stizostedion vitreum*), and burbot (*Lota lota*). No significant difference was found among control, burned, and logged lakes in the back-calculated length of yellow perch, for which age determinations were made. However, we found that the proportions of small yellow perch and white sucker were significantly lower in populations of impacted lakes (burned and logged lakes pooled). The influence of logging and fires remained significant when a series of biotic and abiotic variables on watershed and lake characteristics were accounted for in multiple regression analyses. The lower proportion of small fish in impacted lakes could be due to an increase in postemergence mortality or to a shift of individuals to the pelagic zone.

Résumé : Le but de cette étude était de déterminer si les coupes forestières et les feux de forêt ont un impact significatif sur l'abondance, la croissance et la structure en taille des populations de poissons de 38 lacs du bouclier laurentien (Québec, Canada). Les bassins versants de neuf lacs ont fait l'objet de coupes forestières, ceux de neuf autres ont subi des feux de forêt, alors que les 20 restants ont été utilisés comme référence. Aucune différence significative n'a été observée entre les trois groupes de lacs au niveau des captures par unité d'effort des espèces les plus abondantes : le meunier noir (*Catostomus commersoni*), le grand brochet (*Esox lucius*), la perchaude (*Perca flavescens*), le grand corégone (*Coregonus clupeaformis*), la ouitouche (*Semotilus corporalis*), l'omble de fontaine (*Salvelinus fontinalis*), le doré jaune (*Stizostedion vitreum*) et la lotte (*Lota lota*). Aucune différence significative n'a été observée dans les longueurs rétrocalculées des perchaudes, pour lesquelles la détermination d'âge a été effectuée, entre les lacs témoins, brûlés et coupés. Cependant, une diminution significative de la proportion des individus de petite taille a été observée chez les populations de perchaudes et de meuniers noirs des lacs perturbés (les lacs brûlés et coupés formant un seul groupe). L'influence des coupes et des feux est demeurée significative après que les variables biotiques et abiotiques caractérisant les bassins versants et les lacs ont été considérées dans des analyses de régression multiple. La plus faible proportion des individus de petite taille dans les lacs ayant subi des perturbations pourrait être due à une augmentation de la mortalité post-émergence ou à un déplacement des individus dans la zone pélagique.

Introduction

The effects of logging are well documented in lotic ecosystems (reviewed in Roberge 1996). Some of the most significant effects are the increase in stream flow (Van Der Vinne and Andres 1988b; Heede 1991), nutrient concentrations (Nicolson et al. 1982; Plamondon et al. 1982), and turbidity and sedimentation (Krause 1982; Everest et al. 1987). Road construction and road use associated with logging are often more significant causes of increased sediment yield

than logging itself (Van Der Vinne and Andres 1988a; Miller et al. 1997). Logging may also induce increases in water temperature (Beschta et al. 1987; Garman and Moring 1991), decreases in dissolved oxygen (Ringler and Hall 1975; Murphy and Milner 1997), and increases in primary productivity (Murphy and Hall 1981; Gregory et al. 1987) resulting from canopy removal. Forest fires have also been reported to increase sediment transport (Beatty 1994; Cerdà et al. 1995) and nutrient losses (Spencer and Hauer 1991; Bayley et al. 1992). In lakes, forest removal, either through natural fires or logging, also tends to increase organic carbon, chlorophyll *a*, and limnoplankton biomass (Carignan et al. 2000; Patoine et al. 2000; Planas et al. 2000).

All these changes may impact the top-down and bottom-up trophic interactions, including fish populations, which can reflect short-term changes in limnetic eutrophication due to their ability for rapid growth compensation and their short life history cycles. As fish yield is strongly correlated with lake productivity, forest clearance may increase fish produc-

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tivity (Roberge 1996). Increased sedimentation following deforestation is also likely to have an impact on spawning habitats and therefore on fish recruitment (Everest et al. 1987). Most studies on the impact of logging on fish populations deal with salmonids inhabiting running waters. With the exception of Bérubé and Lévesque (1998), we do not know of any study that investigated the impact of these perturbations on fish population structure of lake ecosystems. Two studies of this issue deal with the potential effect of deforestation on salmonid lake habitats (Gunn and Sein 2000; Steedman and Kuskneriuk 2000).

The objective of our study was to determine if abundance, growth, and size structure of fish populations are correlated with changes in any lake characteristics following wildfires or logging.

Methods

Study area

The data were collected in 38 stratified headwater lakes on the Laurentian Shield, located within a 50 000-km² area around Réservoir Gouin, Québec (for a detailed description of the study area, see Carignan et al. 2000). Our study compared fish communities in lakes with three types of watershed treatments: 20 control lakes with undisturbed watershed, nine "logged" lakes, whose watersheds had undergone forest clearance (8.5–73.2%), and nine "burned" lakes, whose watersheds had been severely burned by fire (50.1–100.0%). Lakes were selected on the basis of comparable size, depth, watershed morphometrics, and time of impact (see Carignan et al. 2000). Most fires and clearcuts occurred in 1995 (with the exception of Lakes C24 and C2, which had partial cuts in spring and summer 1994, respectively). The fish communities of these lakes are generally unexploited or lightly exploited due to the relatively limited access.

Fish sampling

Twenty-one lakes were sampled in 1996 (1 year after impact) and 17 in 1997 (2 years after impact). Each lake was sampled once between June and August. Fish were captured with experimental monofilament gill nets, 102.3 m long × 2.7 m deep, with stretched mesh panels of 20, 24, 33, 36, 50, 60, 76, 90, and 100 mm (filament diameter of 0.17, 0.20, 0.20, 0.20, 0.20, 0.32, 0.32, 0.32, and 0.32 mm, respectively). Gill nets were set perpendicular to the shore, with small and large meshes alternating from the shore among gill nets. The nets were set at regular intervals around the lake, the first net location being randomly located on aerial photographs. The fishing effort was six nets per night for lakes <50 ha, eight nets per night for lakes of 50–100 ha, 10 nets per night for lakes of 100–150 ha, and 12 nets per night for lakes >150 ha. The nets fished for periods of 16–24 h, covering the periods between 18:00 and 09:00. For all fish captured, total length (±1 mm) and weight (±0.1 g) were noted and, when possible, sex was determined by gonad examination. Appropriate bone structures were also removed for further age determination of white sucker, northern pike, yellow perch, brook trout, lake whitefish, and walleye.

Study species

The fish species composition of the study lakes, unknown before the study, was quite diverse (Table 1). The most widespread species were white sucker (31 lakes), northern pike (27 lakes), yellow perch (25 lakes), and lake whitefish (11 lakes). We selected white sucker, yellow perch, and lake whitefish to evaluate the impact of logging or fires on fish population structure because they were the most abundant and frequent species. Northern pike was present in

many lakes, but its abundance (catch per unit of effort (CPUE), number of fish per gill net per night) was too low to build reliable length distributions (mean CPUE: 2.1 ± 1.3 , $n = 27$ lakes).

Relative abundance

Lakes where few individuals were captured (i.e., <15 per species) were not included in the analyses. A one-way analysis of variance (ANOVA) followed by Tukey's multiple range comparison test was performed to determine if there were any significant differences in CPUE among lake groups for species for which sufficient data were available to perform this analysis (e.g., white sucker, northern pike, yellow perch, and lake whitefish). Assuming that logging and fires may have similar effects on abundance (if there is an effect), we pooled the data from logged and burned lakes into a single group, hereafter referred as impacted lakes, to increase sample size of the other species (e.g., fallfish, brook trout, walleye, and burbot). In this case, we used a *t* test to compare control versus impacted lakes. When the data were not normally distributed and (or) the variances heterogenous, the data were $\log(x + 1)$ transformed. The homogeneity of variances was tested with the F_{\max} test (Sokal and Rohlf 1981), while normality was tested with the Kolmogorov–Smirnov–Lilliefors test. The data were not always normally distributed and (or) the variances homogeneous after transformation. We assumed that departures from these assumptions had no marked effect on the *t* test or ANOVA significance levels (Sokal and Rolf 1981).

We also compared the percent CPUE of "small fish" of the three study species and that of age-1+ yellow perch using the same procedure because they appeared to be affected by logging and fires (see Fig. 1 and below for age determination of yellow perch). Small fish corresponded to the first mode of the size frequency distribution, which was estimated visually. In the size frequency distribution, the frequency of each length-class represented the mean of the lake group. For yellow perch, the first mode was clear and corresponded to fish <75 mm in length (see Results section). For white sucker, the first mode was not as clear as for yellow perch but was set to include fish <160 mm. Finally, the first size-class of lake whitefish comprised fish <120 mm.

Age and growth of yellow perch

Opercular bones were used to determine age and growth of yellow perch. Pairs of operculars were placed in a solution of soapy water overnight and then cleaned and dried. A dissecting microscope connected to an image analyser was used for the examination of operculars. The bone structure was submerged in immersion oil and observed at 6.5–10× magnification on a black background with a continuous light ring. An image of the bone was then captured with Sigma Scan Pro[®] (version 4.0) and viewed on a monitor. The true annuli were characterized by a gradual change from the broad opaque summer growth zone to a narrow transparent winter zone, which ends relatively abruptly with a sharp line of discontinuity to the next summer zone (LeCren 1947). This line was taken as the end of the year's growth. Two independent readings were made by the same reader. When the results differed, two additional readings were made; when there was still disagreement, the sample was rejected. The opercular centrum was considered to be the point in the thickest region of the bone where the holes appeared to converge (LeCren 1947). A straight line was then traced perpendicularly between the centrum and the edge of opercula, corresponding to the opercular length. The opercular length as well as successive annual growth increments were measured to the nearest 0.01 mm with the image analyser.

We back-calculated body lengths at previous ages following two proportional back-calculation methods: the scale-proportional hypothesis (SPH) and the body-proportional hypothesis (BPH). A detailed description of these methods can be found in Francis (1990).

Table 1. Occurrence of fish species in the three groups of sampled lakes (control, logged, and burned).

Species	Control	Logged	Burned	Total
White sucker (<i>Catostomus commersoni</i>)	18	6	7	31
Northern pike (<i>Esox lucius</i>)	13	7	7	27
Yellow perch (<i>Perca flavescens</i>)	12	7	6	25
Lake whitefish (<i>Coregonus clupeaformis</i>)	4	3	4	11
Fallfish (<i>Semotilus corporalis</i>)	4	4	2	10
Brook trout (<i>Salvelinus fontinalis</i>)	4	2	1	7
Walleye (<i>Stizostedion vitreum</i>)	4	2	3	9
Burbot (<i>Lota lota</i>)	3	3	1	7
Lake trout (<i>Salvelinus namaycush</i>)	0	1	1	2
Rainbow smelt (<i>Osmerus mordax</i>)	2	0	0	2
Trout-perch (<i>Percopsis omiscomaycus</i>)	1	0	0	1
Brook stickleback (<i>Culaea inconstans</i>)	1	0	0	1
Ninespine stickleback (<i>Pungitius pungitius</i>)	1	0	0	1
Finescale dace (<i>Phoxinus neogaeus</i>)	4	1	0	5
Lake chub (<i>Couesius plumbeus</i>)	3	2	0	5
Golden shiner (<i>Notemigonus crysoleucas</i>)	1	3	1	5
Pearl dace (<i>Margariscus margarita</i>)	4	0	3	7
Blacknose shiner (<i>Notropis heterolepis</i>)	0	1	0	1
Spottail shiner (<i>Notropis hudsonius</i>)	2	0	0	2
Logperch (<i>Percina caprodes</i>)	0	0	1	1
Northern redbelly dace (<i>Phoxinus eos</i>)	2	0	0	2
Common shiner (<i>Luxilus cornutus</i>)	1	0	0	1
Cyprinid sp.	14	1	8	23

Note: Values indicate the number of lakes that each taxa inhabited per category.

In 80% of lakes, the R^2 of the body length–scale length relationships was above 0.95. For this reason, neither back-calculation method was preferred (BPH versus SPH) because no substantial differences were found in back-calculated fish lengths (Pierce et al. 1996). In this paper, we present the results computed with the SPH method. We used back-calculated length-at-age to compare growth among the three lake groups (control, burned, and logged). To reduce the interannual variability, comparisons were made only among fish of the same cohort and of the same lag preceding or following the impact (e.g., age-1 fish 1 year after logging). Since all fish sampling was not done in the same year, this approach allowed us to use 2 years of data in the same analyses (see Results section). This procedure was carried out on the first three cohorts (ages 1–3) for a total of 5 back-calculated years: the 2 years both preceding and following the impacts as well as for the year of the impacts. A one-way ANOVA followed by Tukey's multiple range comparison test was performed for each year in each cohort to determine if there were any significant differences in fish length among lake groups. Finally, we used a t test to compare control and impacted lakes (pooled data from logged and burned lakes).

Multiple regressions

Stepwise multiple linear regressions (forward selection) were used to build models with each of the following dependent variables: percent 1-year-old yellow perch in the population and percent of white sucker <160 mm in the population. No model was built to predict the relative abundance of small lake whitefish in the population because of the low sample size (nine lakes). Watershed and lake morphometry, water quality, phytoplankton, and zooplankton were used as independent variables in statistical analyses (Table 2). Methodological details concerning estimation of these variables can be found in Carignan et al. (2000), Patoine et al. (2000), and Planas et al. (2000). Water quality, phytoplankton, and zooplankton were sampled three times during the summers of 1996 and 1997. For these variables, we used the mean of the three sum-

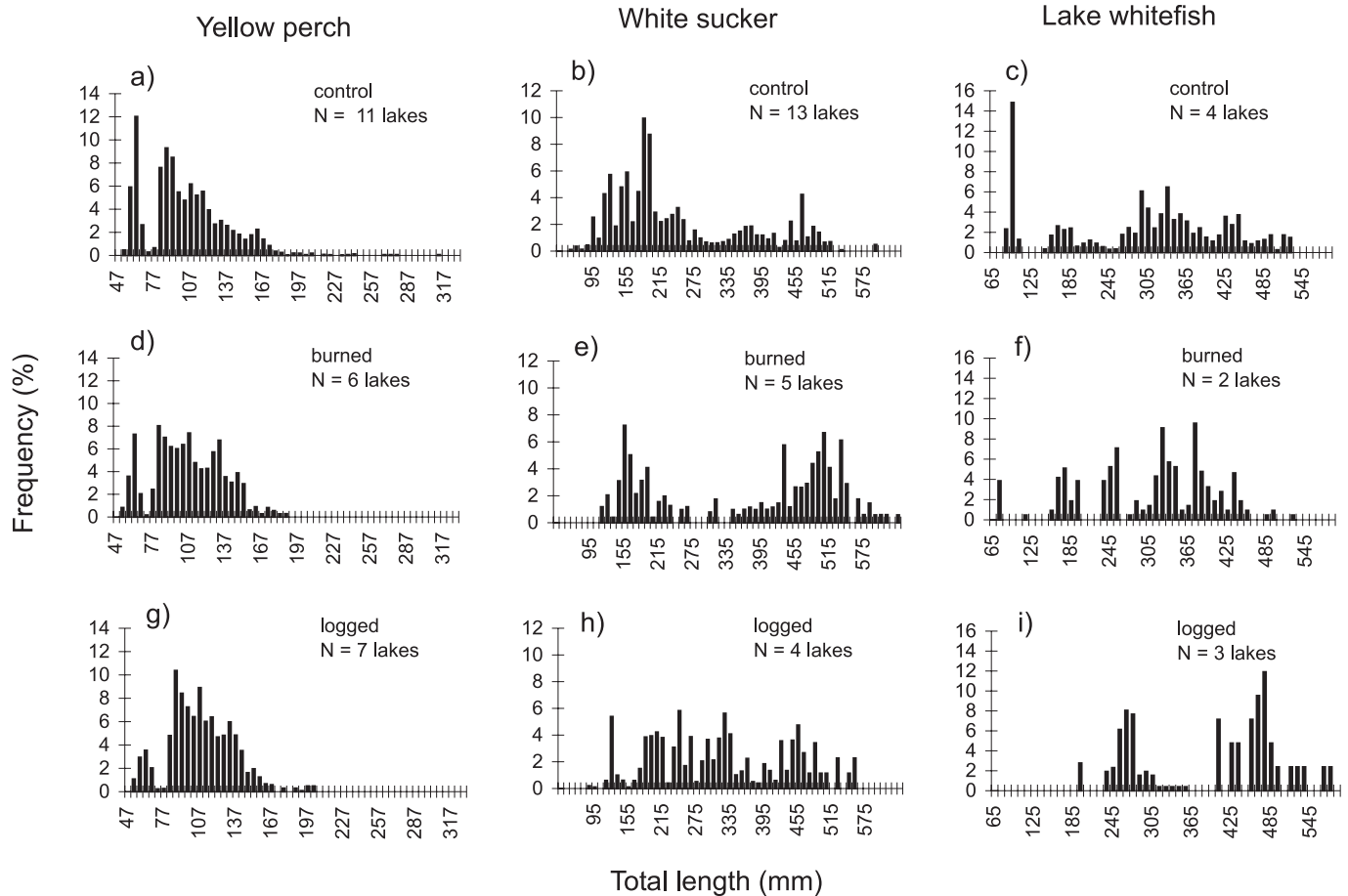
mer samples in statistical analyses (for the summer when fish were sampled). The variable "logging or fires" was entered as a dummy variable where the value 0 was attributed to control lakes and 1 to those that underwent logging or fires. As fish were not sampled simultaneously in all of the lakes during the summer, we also created a variable "sampling date," which was the day of the year. All variables that were not normally distributed were transformed (log or arcsin transformed for proportion). The number of independent variables used in the procedure was always lower than the number of lakes. In the stepwise procedure, the independent variable that enters on the first step is the most highly correlated with the dependent variable. The best model, as indicated by the highest R^2 values and the lowest values of the mean square error associated with the estimate, was retained. Collinearity between the independent variables was evaluated by examination of the pairwise correlation coefficients. Independent variables that were highly correlated (e.g., $r = 0.7$ – 1.0) were not considered simultaneously in regression analyses. The tolerance level given in the Systat regression procedure was also taken into account to prevent collinearity. When collinearity occurred, each collinear variable was used to build a separate model that was then compared with all of the other models. Residual scatterplots, normal probability plots, and partial residual plots were used to determine if the assumptions of the multiple linear regression were satisfied (i.e., normality, linearity, and homoscedasticity of residuals).

Results

Relative abundance

No significant differences were found in CPUE among the three lake groups for white sucker, northern pike, yellow perch, lake whitefish, fallfish, brook trout, walleye, and burbot (Table 3). Similarly, no significant differences were found in CPUE of the most frequent species between control

Fig. 1. Length frequency distribution of yellow perch, white sucker, and lake whitefish populations sampled in 1996 or 1997 in control, burned, and logged lakes. For each length-class, frequency represents the mean of the lake group.



and impacted lakes (i.e., when burned and logged lakes were pooled) (Table 3). The mean percent CPUE of small white sucker (<160 mm) in the population was significantly lower in impacted lakes than in control lakes ($t = -2.15$, $p < 0.05$) (Fig. 1). The mean percent CPUE of small yellow perch (<75 mm) and small lake whitefish (<120 mm) also tended to be lower in impacted than in control lakes, but these differences were not significant (Fig. 1). In lakes sampled in 1996, the mean percent CPUE of 1+ yellow perch in the population was significantly lower in logged lakes than in control lakes ($F = 9.913$, $p < 0.05$), with burned lakes showing an intermediate value (Fig. 2a). Such a difference in mean percent CPUE among lake groups was not observed in 1997 (Fig. 2b). The same pattern was observed for 1+ yellow perch in pooled samples of 1996 and 1997 ($F = 4.289$, $p < 0.05$) (Fig. 2c). A significant increase in the mean percent CPUE of 3+ yellow perch was also observed in burned lakes compared with control lakes (pooled years: $F = 6.024$, $p < 0.01$). As the abundance is expressed as percent CPUE, the significant increase in abundance of 3+ yellow perch is probably due to the significant decrease of 1+ individuals.

Growth of yellow perch

No significant differences were found in the back-calculated length of yellow perch among control,

burned, and logged lakes (Table 4). No significant differences were observed when logged and burned lakes were pooled into impacted lakes (Table 4).

Determinants of the proportion of 1-year-old yellow perch and small white sucker (<160 mm) in the population

The best predictors of the percent CPUE of 1-year-old yellow perch in the population were the biomass of northern pike (–), mean summer lake temperature (+), the dummy variable logging or fires (–), and depth of the epilimnion (+), explaining 29.4, 24.6, 19.5, and 13.1% of the variation, respectively (Table 5). The length of brooks in the watershed (+) explained 0.5% of the variation (Table 5). The abundance of white sucker and the nanophytoplankton density appeared in the model as suppressor variables (i.e., a regressor that is virtually uncorrelated with the dependent variable but is useful in the model because it accounts for some residual variance in the other independent variables that is irrelevant for the prediction of the dependent variable; Tabachnick and Fidell 1983). Given the presence of these suppressor variables (Table 5), this model explained a total of 81.3% of the variation of percent CPUE of 1-year-old yellow perch in the population. The most powerful predictors of the percent CPUE of small white sucker were the density of

Table 2. Independent variables available for multiple regression analyses.

Categorie	Variable	Units	
Geographical	Latitude	Decimal	
	Longitude	Decimal	
	Altitude	m	
Lake morphology	Lake area	km ²	
	Fetch	km	
	Lake perimeter	km	
	Shoreline development		
	Mean lake slope	%	
	Mean littoral slope	%	
	Lake volume	m ³	
	Epilimnion volume	m ³	
	Littoral volume	m ³	
	Littoral area	m ²	
	Maximum depth	m	
	Mean depth	m	
	Epilimnion depth	m	
	Watershed morphology	Watershed area	km ²
Drainage area		km ²	
Watershed perimeter		km	
Drainage density		km·km ⁻²	
Length of brooks on watershed		km	
Runoff		m ³ ·year ⁻¹	
Water residence time		Years	
Mean watershed slope		%	
Marsh on watershed		km ²	
Secchi depth		m	
Physical and chemical	Mean lake temperature	°C	
	Mean lake oxygen	mg·L ⁻¹	
	Mean epilimnion temperature	°C	
	Mean epilimnion oxygen	mg·L ⁻¹	
	Thermocline depth	m	
	Dissolved organic carbon	mg·L ⁻¹	
	Total phosphorus	µg·L ⁻¹	
	Total nitrogen	µg·L ⁻¹	
	Nitrate	µg·L ⁻¹	
	pH		
	Alkalinity	µequiv·L ⁻¹	
	Phytoplankton	Chlorophyll <i>a</i>	µg·L ⁻¹
		Picophytoplankton density	µg·L ⁻¹
Nanophytoplankton density		µg·L ⁻¹	
Microphytoplankton density		µg·L ⁻¹	
Zooplankton	Total volume of particles	mm ³ ·m ⁻³	
	Volume of particles <1000 µm	mm ³ ·m ⁻³	
	Volume of particles >1000 µm	mm ³ ·m ⁻³	
	Ash-free dry weight of particles 50–100, 100–200, 200–500, and >500 µm	mg·m ⁻³	
Fish	Abundance or biomass of northern pike	No·net ⁻¹ or g·net ⁻¹	
	Abundance or biomass of walleye	No·net ⁻¹ or g·net ⁻¹	
	Abundance or biomass of white sucker	No·net ⁻¹ or g·net ⁻¹	

microphytoplankton (+), the dummy variable logging or fire (–), latitude (+), and sampling date (+), which together accounted for 73.6% of the variation (Table 5).

Discussion

Our study indicates that logging and fires affect the abun-

dance of small yellow perch and white sucker. This is confirmed by the continued presence of the dummy variable logging or fires in the multiple regression models after all others biotic and abiotic variables were accounted for. Furthermore, the variable logging or fires explained a substantial proportion of the variation of 1+ yellow perch (19.5%) and small white sucker (24.1%). When logging and fires

Table 3. CPUE of fish species in the three lake groups (control, logged, and burned).

Species	Control	Logged	Burned	p_1	p_2
White sucker ^{a,b}	10.2±12.0 (18)	8.4±9.9 (6)	3.6±2.9 (7)	0.569	0.35
Northern pike ^a	2.0±1.5 (13)	1.9±1.2 (7)	2.6±0.8 (7)	0.3525	0.7096
Yellow perch ^{a,b}	23.4±35.9 (12)	11.7±9.2 (7)	9.6±4.6 (6)	0.7464	0.4611
Lake whitefish ^{a,b}	10.3±5.6 (4)	6.4±7.0 (3)	5.5±8.4 (4)	0.3552	0.1239
Fallfish ^a	5.5±6.1 (4)	1.4±0.6 (4)	1.9±2.6 (2)	nt	0.2921
Brook trout ^a	5.5±6.9 (4)	5.8±7.7 (2)	25.8 (1)	nt	0.4599
Walleye ^a	9.4±4.6 (4)	7.1±0.1 (2)	6.9±3.8 (3)	nt	0.4018
Burbot	0.2±0.1 (3)*	0.3±0.1 (3)	0.8 (1)*	nt	0.3331

Note: Data represent mean ± SD; number of lakes is given in parentheses. p_1 , probability of ANOVA (among control, logged, and burned); p_2 , probability of t test (between control and impacted). *Difference determined by Tukey's test; nt, not tested due to low sample size.

^aANOVA performed on log-transformed data.

^b t test performed on log-transformed data.

were considered separately as independent variables, they did not appear to predict the CPUE of small yellow perch and white sucker in any model. This is probably due to the splitting of samples size (four logged lakes and five burned lakes for white sucker and seven logged lakes and six burned lakes for yellow perch), leading to an increased probability of making a Type II error. In pooling logged and burned lakes, we assumed that both perturbations have similar impacts on small fish.

The impacts of logging are well documented in fish and usually include a decrease in spawning habitat quality. In streams, the increase in temperature and sedimentation after timber harvest can affect egg to fry survival by reducing the oxygen in spawning grounds and by forming physical barriers to emergence (Ringler and Hall 1975; Everest et al. 1987). Forest fires also influence the amount of sediment found in brooks and streams (Beaty 1994; Cerdà et al. 1995). In lakes, the removal of fringing vegetation may also affect spawning habitat quality. For example, fine suspended sediment may be carried around the lake shoreline by water movements and be deposited in spawning grounds (Miller et al. 1997). In some Québec lakes, Bérubé and Lévesque (1998) reported a reduction in sportfishing yield (abundance and biomass) of brook trout after clear-cutting. Possible damage to spawning and nursery habitats was suggested to partially explain these reductions in fishing success.

Most studies on the impact of fires and logging on fish populations have been done on salmonids because they require clean gravel or rubble substrate to spawn, so their reproduction is generally affected by siltation (Berkman and Rabeni 1987). Yellow perch spawn in shallow water of lakes or in tributaries where single, convoluted egg strands are either attached to submerged plants and fallen trees or are deposited on sand and gravel (Scott and Crossman 1973). For this reason, yellow perch are assumed to be less sensitive than trout to the potentially adverse affects of timber harvesting (France 1997); this probably explains why no study has evaluated the impact of logging and fires on this species. In the present study, the reduction in abundance of 1+ yellow perch cannot be attributed to a reduction in egg to fry survival caused by logging or fires. The reduction in abundance was observed in 1996, 1 year after forest harvesting or fires. The 1-year-old yellow perch captured in 1996 were born in spring 1995, before fires and most logging activities

occurred. Consequently, any reduction in the abundance of 1+ yellow perch in the littoral zone would have occurred in the seasons following the emergence of larvae.

White sucker require clean gravel to spawn (Scott and Crossman 1973) and thus could have a response to spawning habitat degradation by siltation similar to that of salmonids (Berkman and Rabeni 1987). As we did not do age determinations for white sucker, we cannot evaluate the effect of fires and logging on the first cohort. However, age-length relationships found in the literature suggest that individuals captured in our gill nets were older than 1 year, including fish under 160 mm. The gillnet selectivity against small fish seems to be responsible for the absence of 1-year-old white sucker in all sampled lakes.

Little information is available on postemergence survival after clearcuts and fires. Depending on the concentration and duration of exposure, suspended sediments can induce physiological stress, reduce growth, and cause direct mortality in fish (Newcombe and MacDonald 1991). In streams where watersheds have been almost completely burned, Bozek and Young (1994) found dead fish following storm flow caused by rain. Fish appeared to have been asphyxiated by sediment that completely obstructed their gills. In streams, the current velocity is strong enough to keep sediment in suspension and thus expose fish to a substantial stress. In contrast, lake basins act as sinks (Miller et al. 1997) where sediment is rapidly deposited. Furthermore, adult fish appear to tolerate suspended sediment concentrations substantially greater than those commonly found in nature (Muncy et al. 1979). For these reasons, it is unlikely that an excess of suspended sediments would have caused direct mortality of fish in burned and logged lakes, especially in logged lakes, due to presence of buffer strips.

Assuming that fine sediment can reach the shores of burned and logged lakes, macroinvertebrate habitats and survival could have been negatively affected, reducing the abundance of food for yellow perch and white sucker. Some reviews have reported a reduction in macroinvertebrate density or diversity following timber harvesting (Gregory et al. 1987; Hartman and Scrivener 1990). Thus, a lower food availability or quality could have influenced the survival of younger fish.

Some studies have reported an increase in salmonid biomass or growth despite a decrease in reproductive habitat

Fig. 2. Age frequency distribution of yellow perch in control, burned, and logged lakes sampled in (a) 1996, (b) 1997, (c) 1996 and 1997 (pooled samples). For each age-class, frequency represents the mean of the lake group.

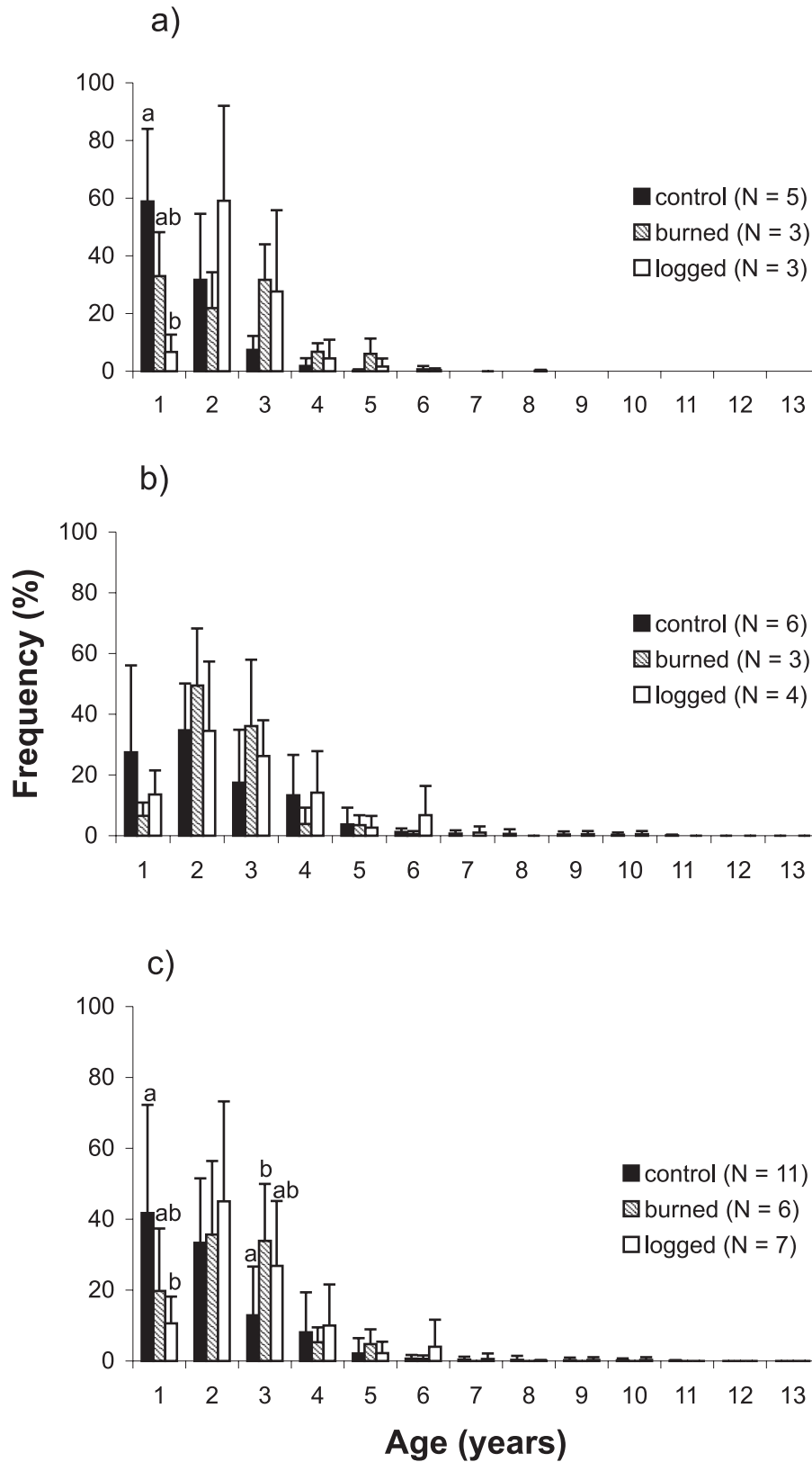


Table 4. Back-calculated length of yellow perch 2 years before, the year of, and 2 years after fire and logging impacts.

Age group	Lake group	2 years before perturbations			1 year before perturbations			Year of perturbations			1 year after perturbations			2 years after perturbations		
		Mean	SD	N	Mean	SD	N	Mean	SD	N	Mean	SD	N	Mean	SD	N
I	Control	48.02	7.41	7	49.18	6.05	9	48.58	8.73	11	52.45	5.23	11	51.16	5.87	6
	Burned	44.46	4.04	5	47.46	10.44	5	49.14	3.45	6	48.02	4.13	6	46.41	4.84	3
	Logged	47.17	4.49	4	47.61	6.34	6	49.89	3.75	7	51.00	3.88	6	50.24	2.95	4
	ANOVA	ns	ns		ns	ns		ns	ns		ns	ns		ns	ns	
	<i>t</i> test	ns	ns		ns	ns		ns	ns		ns	ns		ns	ns	
II	Control	80.37	8.53	5	76.98	11.70	7	79.49	5.74	10	82.48	5.05	11	81.34	5.15	6
	Burned	68.92	7.67	3	70.65	5.49	5	75.34	6.43	5	80.67	7.93	6	79.17	2.84	3
	Logged	81.67	8.61	3	73.32	3.92	4	75.55	7.34	6	81.48	6.69	7	83.30	6.22	4
	ANOVA	ns	ns		ns	ns		ns	ns		ns	ns		ns	ns	
	<i>t</i> test	ns	ns		ns	ns		ns	ns		ns	ns		ns	ns	
III	Control	104.06	24.02	2	104.16	14.57	5	101.92	11.45	7	111.93	13.14	10	109.97	9.84	6
	Burned	102.87	—	1	92.95	13.27	3	94.84	7.27	5	102.37	8.70	5	116.02	14.09	3
	Logged	96.29	—	1	106.47	11.31	3	93.57	3.64	4	102.21	5.87	6	112.75	10.40	4
	ANOVA	ns	ns		ns	ns		ns	ns		ns	ns		ns	ns	
	<i>t</i> test	ns	ns		ns	ns		ns	ns		$p < 0.05$	ns		ns	ns	

Note: Statistical analyses were done on separate treatment levels (ANOVA) and in pooling fires and logged lakes into a single group (*t* test); ns, not significant.

Table 5. Best models predicting the percentage of 1-year-old yellow perch and small white sucker (<160 mm) in the population.

Model	$p > t$	SE	R^2 (%)	Adjusted R^2 (%)	SE _{xy} (%)
% of 1-year-old yellow perch in population =			81.3	73.1	16.6
-2.11	0.0067	0.6766			
-0.18 northern pike biomass ^a	0.0012	0.0505	29.4		
+1.67 mean summer lake temperature ^a	0.0006	0.3899	24.6		
-0.25 logging or fires ^b	0.0142	0.1046	19.5		
+0.15 depth of epilimnion	0.0102	0.1371	13.1		
+0.23 length of brooks in watershed ^a	0.0457	0.0898	0.5		
+0.36 nanophytoplankton density ^{a,c}	0.0184	0.1144	(-0.02)		
-0.34 white sucker abundance ^{a,c}	0.0094	0.0448	(-5.8)		
% of white sucker of length <160 mm in population =			80.2	74	14.5
-20.21	0.0004	4.5602			
+0.66 microphytoplankton density ^a	0.0000	0.1113	30.7		
-0.31 logging or fires ^b	0.0002	0.0645	24.1		
+0.40 latitude	0.0006	0.0928	10.0		
+0.01 sampling date	0.0017	0.0017	8.8		
-0.21 littoral area ^a	0.0728	0.1113	6.7		

quality (Murphy and Hall 1981; Grant et al. 1986). These authors suggested that the loss of reproductive habitat was compensated for by an increase in primary productivity following disturbance. In our study, the increased phosphorus supply in burned lakes caused a significant (80%) upsurge in biological productivity, as evidenced by higher planktonic and attached chlorophyll levels, algal biomass, and zooplankton (Carignan et al. 2000; Patoine et al. 2000; Planas et al. 2000). In contrast, the increased total phosphorus levels in logged lakes did not result in higher productivity, either because higher concentrations of dissolved organic carbon reduced light availability or because of differences in biological availability of phosphorus released by burned and harvested watersheds (Carignan et al. 2000). As no differ-

ence was found in its growth, yellow perch did not respond to the increased productivity in burned lakes.

The gill nets used in this study were chosen to ensure the capture of all species and of a large size range of fish. However, these nets are not efficient in capturing small fish like 1-year-old yellow perch and small white sucker. It is possible that an increase in the catchability of small fish occurred as the summer progressed and the fish grew larger. This could explain the selection of the variable sampling date (+) in the model explaining the percentage of small white sucker in the population (Table 5). The absence of such a relationship in the yellow perch model could be due to specific differences in catchability. The possible increase in catchability of small fish as the summer progressed could explain the

lower abundance of small yellow perch and white sucker in impacted lakes only if these lakes had been systematically sampled at the beginning of the summer and control lakes at the end, which was not the case. Our sampling effort was evenly distributed among treatments through 1996 and 1997. Furthermore, the impact of logging and fires remained significant when sampling date was accounted for in multiple regression analyses. Nevertheless, a detailed estimation of small size-class structure of fish in these lakes would need a more targeted sampling strategy (e.g., smaller mesh size, seine, trawl, or traps). It is also possible that small yellow perch and white sucker move to the pelagic zone following littoral zone disturbances of logged and burned lakes (e.g., following a reduction in macroinvertebrates discussed above); this zone was not sampled with our protocol.

In the model predicting the proportion of small white sucker, the density of microphytoplankton explained a large proportion of the variation, suggesting that this fish is influenced by lake productivity. The biomass of northern pike was the most important determinant of the abundance of small yellow perch. Northern pike is a common predator of yellow perch (Scott and Crossman 1973); a preliminary examination of northern pike stomach contents from this study revealed that small yellow perch were frequently eaten. Mean summer lake temperature and epilimnion depth were also good predictors of the proportion of 1+ yellow perch in the population, probably because they are both related to yellow perch habitat. It is known that growth and survival of perch larvae are positively correlated with temperature (e.g., Guma'a 1978; Wang and Eckmann 1994). The epilimnion depth reflects the availability of habitat selected by young yellow perch; the use of upper waters by juveniles is probably due to temperature requirements for growth (Becker 1983), foraging (Bergman 1987), and (or) avoidance of large piscivores (Werner 1986).

In conclusion, this study suggests that logging and fires have an impact on small yellow perch, white sucker, and perhaps lake whitefish, either through an increase in post-emergence mortality or to a shift of individuals to the pelagic zone. If this is the case, these mortalities may have a cascading effect on the most valuable exploited species like northern pike and walleye, which use yellow perch, white sucker, and lake whitefish as forage fish. We do not know of any study that has investigated the impact of these perturbations in lake ecosystems with such a large sample size and on nonsalmonid species. Our sampling design allowed us to observe significant differences in population structure among lake treatments but not to determine the mechanisms responsible for these patterns. Further studies will be needed to investigate the function of fish populations following forest removal by clear-cutting or burning.

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