Effects of experimental clearcut logging on thermal stratification, dissolved oxygen, and lake trout (*Salvelinus namaycush*) habitat volume in three small boreal forest lakes

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Abstract: Clearcut logging around three 30- to 40-ha dimictic northwestern Ontario lakes was associated with increases of 5% or less in midlake wind speed and no measurable changes in spring and fall circulation efficiency or duration of stratification. Water clarity, indexed as the depth at which photosynthetically active radiation was 1% of surface intensity, declined by 25% after 3 years. Late-summer thermoclines were about 1 m shallower in two lakes after logging, but it was not possible to exclude weather as a factor. None of the lakes showed significant declines in lake trout (*Salvelinus namaycush*) habitat volume. A forested shoreline buffer strip around one of the lakes prevented increases in midlake wind speed but did not prevent declines in water clarity and thermocline depth.

Résumé : La coupe à blanc autour de trois lacs dimictiques de 30 à 40 ha du nord-ouest de l'Ontario a été associée à des augmentations de 5% ou moins de la vitesse du vent au milieu du lac, mais à aucun changement mesurable dans l'efficacité de la circulation printanière et automnale, ni dans la durée de la stratification. La limpidité de l'eau, calculée selon un indice de 1% de la profondeur du rayonnement photosynthétiquement actif, a baissé de 25% après 3 ans. Les thermoclines de la fin de l'été se trouvaient environ 1 m plus haut dans les deux lacs après la coupe, mais on ne pouvait pas exclure le rôle du facteur climatique. Dans aucun lac on n'a noté de baisse significative du volume d'habitat du touladi (*Salvelinus namaycush*). Une bande tampon boisée sur la berge autour d'un des lacs a empêché les augmentations de la vitesse du vent au milieu du lac, mais n'a pas empêché les baisses de la limpidité de l'eau ni le changement de la hauteur de la thermocline.

[Traduit par la Rédaction]

Introduction

For many years, Ontario lake managers have been concerned about the threat posed to lake trout (*Salvelinus namaycush*) populations by nutrient enrichment, sedimentation, and accelerated hypolimnetic oxygen depletion (HOD) following clearcut logging in the boreal forest (Ontario Ministry of Natural Resources 1988). This concern originated in part from studies in the 1960s and 1970s demonstrating the relationships between phosphorus and HOD in temperate lakes and evidence that logging and road construction had caused sedimentation and nutrient enrichment in streams in various parts of North America (e.g., Brown and Krygier 1971; Schindler et al. 1971; Cornett and Rigler 1979). Since then, various studies have reported water quality changes after logging around boreal lakes and streams (Holopainen et al. 1991; Rask et al. 1993, 1998; Carignan et al. 2000). In northwestern Ontario, new clearcuts, roads, and landings were associated with aeolian sediment transport into lakes (Steedman and France 2000) and minor littoral warming (Steedman et al. 1998). There is evidence that deforestation may increase wind exposure on very small lakes (Rask et al. 1993; Scully et al. 2000), but mixing in larger lakes is probably determined by topography, lake morphometry, and water colour (Fee et al. 1996).

Our understanding of lake trout habitat requirements is also evolving. New evidence suggests that lake trout are not fundamentally restricted to the hypolimnion of stratified lakes and that their realized thermal niche can be considerably broader than traditionally inferred from laboratory studies. This may be most apparent in small lakes with simple fish communities, as lake trout are now believed to forage extensively in epilimnetic water if other warmwater predators such as esocids or centrarchids are absent (Snucins and Gunn 1995; Sellers et al. 1998; Vander Zanden et al. 1999). Although our estimates of limiting temperatures for lake trout are being revised upwards, there is little doubt that lake trout are intolerant of dissolved oxygen concentrations below about 5–6 mg·L^{-1} (Sellers et al. 1998). Quantitative indices of lake trout habitat volume have traditionally been based on temperature and dissolved oxygen limits believed to impose significant restrictions on the behaviour or viability of lake trout individuals or populations. Two indices have been widely used by lake managers: (*i*) "optimal" habitat, the volume of water 10°C or colder, with dissolved oxygen of 6 mg·L^{-1} or higher (10/6 habitat), and (*ii*) "usable" habitat, the volume of water 15.5°C or colder, with dissolved

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oxygen of 4 mg·L^{-1} or higher (15.5/4 habitat) (MacLean et al. 1990; Evans et al. 1991; Ryan and Marshall 1994). Schindler et al. (1996) used 10°C and 4 mg dissolved oxygen·L^{-1} to define summer refugia for lake trout in the Experimental Lakes Area.

Five environmental factors have been strongly associated with the volume of lake trout habitat (i.e., cool, highly oxygenated water) in stratified temperate lakes (*i*) lake morphometry, relating primarily to effective fetch and total lake volume, (*ii*) thermocline depth, which determines the partitioning of lake volume into epilimnetic and hypolimnetic waters, (*iii*) HOD caused by decomposition of organic sediments, (*iv*) duration of stratification, the time that the hypolimnion is physically isolated from atmospheric oxygen, and (*v*) circulation efficiency, the amount of dissolved oxygen that is restored to deoxygenated hypolimnetic waters during fall and spring mixing (Molot et al. 1992; Ryan and Marshall 1994; Fee et al. 1996). Catchment deforestation could be expected to modify some or all of these factors through intermediate mechanisms involving decreased water clarity caused by increased loadings of dissolved organic carbon (DOC), nutrients, and sediment and by increased wind-induced mixing.

This study reports the effects of experimental catchment deforestation on wind exposure, water clarity, thermal stratification, and dissolved oxygen in three small northwestern Ontario lake trout lakes. The utility of various lake trout habitat indices was also examined. Water quality impacts are reported in a companion paper (Steedman 2000).

Methods

Study site

The three small headwater lakes monitored in this study (none are formally named) were located in the Ontario Ministry of Natural Resources' Coldwater Lakes Experimental Watersheds in boreal – Great Lakes transition forest on the Canadian shield, approximately 200 km northwest of Thunder Bay, Ontario, Canada (Table 1). The lake catchments had shallow soils with abundant boulders and bedrock outcrops, moderate slopes (15–50%), and relief generally not more than about 60 m. Catchment forests consisted of 75- to 100-year-old jack pine (*Pinus banksiana*), trembling aspen (*Populus tremuloides*), and black spruce (*Picea mariana*), with some paper birch (*Betula papyrifera*), eastern white cedar (*Thuja occidentalis*), red pine (*Pinus resinosa*), and eastern white pine (*Pinus strobus*). The lakes were relatively homogeneous with regard to solute and nutrient content, and extremely dilute, with specific conductance of about 14–22 μS·cm^{-1}, 2–3 mg DOC·L^{-1}, total phosphorus of less than 5 μg·L^{-1}, and chlorophyll *a* of less than 2 μg·L^{-1} (Steedman 2000). The lakes contain native populations of lake trout, white sucker (*Catostomus commersoni*), and five to seven species of smaller fish (primarily Cyprinidae, Gasterosteidae, and Etheostomidae).

Climate

Precipitation, wind speed, and wind direction were monitored at automated meteorological stations on a knoll in an upland clearcut, logged in 1987, 0.5 km northeast of L26 (beginning June 1993) and on rafts in the middle of L26, L39, and L42 200–500 m from shore (beginning May 1995). Before installation of the meteorological stations, rainfall was measured with manual rain gauges in the same locations. R.M. Young 05103 wind monitors were used at all stations, mounted 3 m above the ground at the upland site and 1.5 m above the water surface on the rafts. Data loggers sampled sensor readings at 5-s intervals and recorded hourly means and extremes. Because the climate monitoring network was not fully operational for the first few years of the 1991–1999 study period, surrogate thermal and precipitation indices were derived from available data to allow comparison of weather conditions throughout the study. These included average May–September L42 surface water temperature obtained from 1991–1999 water temperature profiles to provide an integrative index of air temperature during the open-water period and 1992–1999 June–September rainfall to provide an index of summer precipitation. The 1966–1988 climate normals for Atikokan, 70 km southeast of the study lakes, were obtained from Environment Canada.

Experimental clearcut logging

In 1996, after 5 years of predisturbance monitoring, the catchments of L26, L39, and L42 were partially deforested by commercial loggers using a tracked feller buncher and chainsaws. Trees were dragged by skidders to the nearest road and delimbed there (Table 2). L26 had a moderate amount of catchment deforestation (33%), with no disturbance of shoreline forest. L39 and L42 had extensive catchment (60–70%) and shoreline deforestation (40–60%). Some additional logging occurred on the shoreline of L42 and on intermittent stream catchments draining to L26 and L39 in 1998. Catchment and clearcut boundaries extended 100–600 m from the lakeshores. About 5 km of logging roads were constructed in the combined catchments of L39 and L42, and about 2 km of roads were constructed in the catchment of L26.

Limnology

Limnological data, including temperature–oxygen profiles, light profiles, and Secchi depth, were collected at weekly to bimonthly intervals (depending on season and location) at the deepest point of each lake. Most measurements reported here were from the open-water season, roughly mid-April through late October. The dates used for interannual comparisons were standardized for each type of measurement but varied between measurement types depending on the longest period of record that was represented in all study years.

Water temperature and dissolved oxygen profiles were measured with a YSI model 57 or model 58 meter at 1-m or smaller depth intervals. Complete spring or fall circulation was arbitrarily recognized when bottom waters achieved dissolved oxygen concentrations of at least 4 mg·L^{-1} after winter or summer anoxia. Dissolved oxygen levels near equilibrium with air (i.e., at least 8 mg·L^{-1} at 5–10°C) were recognized as evidence of complete and prolonged circulation. Since dissolved oxygen readings near lake bottom may be in error if the probe penetrates flocculent surficial sediments, these observations were based on the highest dissolved oxygen reading within 3 m of the estimated maximum lake depth. These data probably underestimate circulation frequency and extent, particularly for fall dates. Field crews were rarely on the lakes on the date of maximum fall mixing, which often occurred after boat operations were shut down for the winter. Late-fall mixing was inferred if measurements through the ice a month or two later indicated high dissolved oxygen levels near the lake bottom. Thermocline depth was calculated as the bottom of the measurement interval (usually 0.25–0.50 m in the metalimnion) with the highest temperature gradient. The lakes were considered to be thermally stratified when the maximum Brunt–Väisälä frequency N was 20·h^{-1} or more in a temperature–depth profile (MacIntyre and Melack 1995). This corresponded well in these lakes to a surface temperature of 10°C or more, which was used as a surrogate indicator of stratification, interpolated between measurement dates. Where surface water temperatures fluctuated above and below 10°C in the spring and fall, the earliest spring date and latest fall date were used to define strat-

Table 1. Description of the study lakes and summary of experimental catchment disturbance.

	Lake		
	L26	L39	L42
Location	49°07′15″N, 92°08′45″W	49°05′30″N, 92°10′00″W	49°05′00″N, 92°09′30″W
Nominal experimental treatment	Moderate catchment deforestation, shoreline buffer	Extensive catchment and shoreline deforestation	Extensive catchment and shoreline deforestation
Terrestrial catchment area deforested (1996, 1996 + 1998), %	33, 45	62, 77	71, 74
Shoreline length deforested (1996, 1996 + 1998), %	0, 0	62, 62	42, 61
No. of tributary lakes	0	1 (L42)	0
Maximum depth (m)	37	23	18
Surface area (ha)	29	39	26
Lake volume (10^5 m^3)	40.8	46.4	21.5
Average depth	14.1	11.9	8.3
Maximum fetch (m) and orientation	900, N–S	1100, NE–SW	1100, N–S
Total catchment area (ha)	106	194	70
Ratio of terrestrial catchment area to lake surface area	2.6	1.9	1.6
Water renewal time (years)[a]	13.1	8.2	10.5
DOC ($mg \cdot L^{-1}$)[b]	2.1	2.2	2.5
Chlorophyll *a* ($\mu g \cdot L^{-1}$)[b]	1.4	1.7	1.8
Estimated size (average weight, g) of lake trout population[c]	428 (690)	265 (877)	273 (848)

[a]Based on 1995–1997 average local runoff of 0.293 $m \cdot year^{-1}$ (Beaty 1998).
[b]1991–1995 May–September average.
[c]Number of unmarked fish encountered during 1991–1998 fall netting of spawning aggregations on L26 and L42 and in spring netting during 1995–1998 on L39.

Table 2. Description and timing of experimental logging impacts around the study lakes in 1996 and 1998.

		Forestry treatment	
Lake	End of 1991 to 1996 preimpact period	1996: first logging impact[a]	1998: second logging impact
L26: moderate catchment deforestation, with shoreline buffer strip	Aug. 31, 1996	Sept. 1 to Oct, 31, 1996: moderate deforestation of catchment (33% of area) with no disturbance of shoreline forest within 30–90 m of lake; 9-ha stream subcatchment at south end of lake left unharvested	June 1–20, 1998: 9-ha stream subcatchment deforested (this deforested an additional 12% of lake catchment, for a total of 45%); no disturbance of shoreline forest
L39: extensive catchment and shoreline deforestation	July 7, 1996	July 8 to Aug. 15, 1996: extensive deforestation of lake catchment (62%) and shoreline (62%); 19-ha stream subcatchment at south end of lake left unharvested	June 1–15, 1998: 19-ha stream subcatchment deforested (this deforested an additional 16% of lake catchment, for a total of 77%); no additional disturbance of shoreline forest
L42: extensive catchment and shoreline deforestation	June 20, 1996	June 21 to July 31, 1996: extensive deforestation of catchment (71%) and shoreline (42%); temporary 30–40 × 500 m (about 1.8 ha) shoreline buffer strip left along southwest shore of L42	June 1–4, 1998: temporary shoreline buffer strip deforested (this deforested an additional 3% of lake catchment,[b] for a total of 74%, and an additional 19% of shoreline, for a total of 61%)

Note: Catchment areas refer to the terrestrial (i.e., nonlake surface) portions. Shoreline deforestation refers to the length of shoreline where merchantable trees were cut to the water's edge.
[a]The 1996 clearcuts were lightly scarified in July and August 1997, and slash piles were burned in October 1997. The 1998 clearcuts were not scarified.
[b]Deforestation of the temporary buffer strip on L42 also constituted a 1% additional disturbance of the 144-ha L39 catchment. This has been included in the 1998 L39 disturbance figure of 16%.

ification. Detailed digital bathymetric maps (1-m depth resolution) were used to calculate lake strata volumes for lake trout habitat volume indices. In addition to 10/6 and 15.5/4 lake trout habitat (see Introduction), two indices with higher temperature limits were calculated for comparative purposes, based on the observations of Sellers et al. 1998: "20/6 habitat," the volume of water 20°C or colder, with dissolved oxygen of 6 $mg·L^{-1}$ or higher, and "/6 habitat," the volume of water with dissolved oxygen of 6 $mg·L^{-1}$ or higher, with no upper temperature limit. Lake bathymetry was corrected for lake stage on all open-water sampling dates, based on daily interpolation of weekly stage measurements. June lake stages for all lakes in all years were within 0.3 m of the bathymetric datum.

Water transparency was measured at 1-m depth intervals with an upward-facing LICOR cosine-corrected photosynthetically active radiation (PAR) sensor and a LI-1000 data logger. Simultaneous light measurements of ambient light were recorded from a similar sensor located on the boat deck. An epilimnetic light extinction coefficient was calculated as the average incremental light extinction from 1.5 to 6.5 m depth, measured at 1-m intervals. This extinction coefficient did not include light reflection from the water surface. Surface light readings were used to correct sequential submerged readings for minor changes in surface light intensity. An incremental extinction coefficient was not calculated if surface light intensity changed by more than 10% between measurements. The 1% PAR depth was estimated as the depth of the first submerged reading showing ≤1% of surface radiation.

Statistical analyses

This study did not replicate lake treatments and therefore did not estimate between-lake variance associated with forestry impacts, i.e., the responses of each lake to its forestry treatment were analyzed and interpreted independently of the other lakes. However, interannual variability estimates for each lake, before and after logging impacts, were provided by the multiyear study design. Although L39 and L42 might appear to represent replicates of the "extensive deforestation" treatment, L42 was tributary to L39, and the lakes were not hydrologically independent. Data from 1996 were generally excluded from pre- and post-logging comparisons because summer averages from that year often spanned the period of logging.

Continuous wind data were available for the May 12 to October 31 period from 1995 through 1999 (1995–1998 for L26). Wind data had a skewed distribution and were transformed as $\log(n + 1)$ prior to calculation of means and 95% sample confidence intervals. Conventional analysis of variance (ANOVA) of these autocorrelated hourly time series would have been inappropriate, as significance of interannual differences in wind velocity would be greatly exaggerated. Instead, time series intervention analysis was used to estimate shoreline logging impacts on midlake wind velocity. Midlake wind velocity time series spanning upland and shoreline logging activity around L39 and L42 in 1996 and 1998 were modeled with "seasonal" Box–Jenkins or autoregressive integrated moving average (ARIMA) (1,1,1) (0,1,1) 24 models (Box and Jenkins 1976; SPSS, Inc. 1993). These ARIMA models differenced the data (subtracted each observation from the previous one) at hourly and daily lags to stabilize the time series. The models then estimated coefficients for hourly autoregression-1 and moving-average-1 processes and daily moving-average-1 processes to remove remaining autocorrelation in the data. Then, the remaining variance was partitioned according to a binary "dummy" variable set to 0 before the midpoint of each year's logging impact and to 1 after. All ARIMA analyses reported here produced residuals with no significant autocorrelation, a key diagnostic criterion for statistically valid partitioning of variance in autocorrelated time series data. Wind velocity data were not transformed prior to the ARIMA analysis because untransformed data produced better residuals, with impact coefficients in original units of metres per second.

Repeated-measures analysis of variance (RM-ANOVA) was used to estimate treatment effects for lake phenomena that were measured several times each year, including water clarity indices and thermocline depth. These data did not show consistently asymmetrical distributions and were not transformed. Each year's set of weekly to monthly measures was treated as an independent observation for the purposes of this study. RM-ANOVA does not allow missing observations, and in two cases where data were not measured in the required week, data from the previous week were used. Regular ANOVA was used to estimate treatment effects for minimum annual lake trout habitat volume, based on one observation per year. Bonferroni adjustment was used for all post hoc comparisons.

Results

The 1991–1995 prelogging period included a relatively wide range of temperature and precipitation, due in part to the June 15, 1991, eruption of Mt. Pinatubo in the Philippines, which brought unusually cool, wet weather in 1992. In general, the prelogging period had above-normal precipitation. Postlogging precipitation was below normal in 1997 and 1998 but increased in 1999 (Table 3). High winds (i.e., >10 $m·s^{-1}$) occurred in all open-water months, from all directions. The strongest winds tended to blow from the southeast and the southwest and were most frequent in the spring and fall. Wind direction, strength, and seasonal patterns were generally similar between years.

Midlake hourly average and hourly maximum wind velocity increased slightly after logging on the two lakes with extensive shoreline deforestation. However, average wind velocity and 95% confidence intervals varied by less than 15% before and after logging, with highest average velocities in 1997, rather than after maximum deforestation in 1998 (Table 4). Intervention analyses of the L39 and L42 wind velocity time series produced impact coefficients of –0.1 to –0.4 $m·s^{-1}$, with 95% confidence intervals spanning zero, a strong indication that no significant in-year increases in midlake wind were associated with the logging events. Average wind velocity at the upland meteorological station declined about 0.1 $m·s^{-1}·year^{-1}$ from 1995 to 1999, probably due to forest regeneration on surrounding clearcuts. The ratios of midlake to upland average wind velocity (corrected for the upland decline as estimated by linear regression) had a pattern consistent with increased postlogging wind velocity on L39 and L42 (Fig. 1). However, these were small increases of about 5% relative to the upland site.

Lake circulation occurred in late April to mid-May and again in late October to early November. The probability and efficiency of complete circulation were inversely associated with lake depth (Table 5). L26 (37 m deep) circulated in the spring every year after logging compared with only two of six years before logging. It rarely circulated in the fall and did not do so at all after logging. Circulation in L26 never fully restored deep dissolved oxygen levels. Circulation frequency in L39 and L42 was similar before and after logging. L39 (23 m deep) achieved complete spring and fall circulation in all years except fall 1993, with prolonged circulation about half of the time. L42 (18 m deep) had complete, pro-

Table 3. 1991–1999 climate indices.

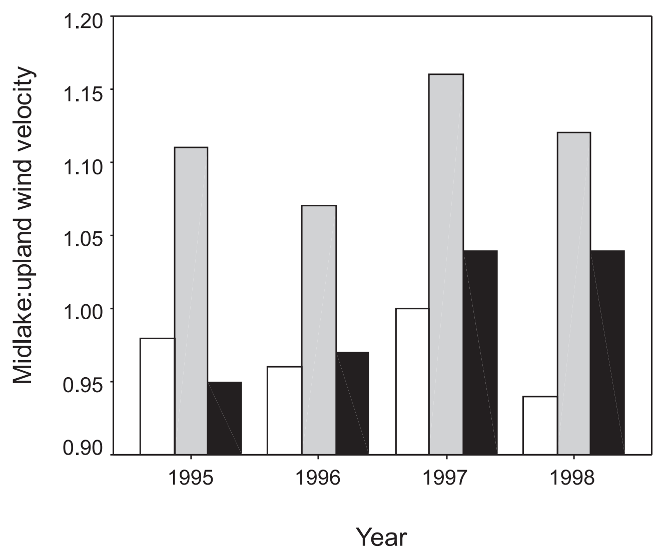
Climate index	Normal[a]	1991	1992	1993	1994	1995	1996	1997	1998	1999
L42 summer average temperature (°C)		20.8	16.2	16.7	18.3	18.7	19.6	17.5	19.3	17.9
March snowpack (mm equivalents)	182[b]				43		130	130	5	113
L42 approximate ice-off date		May 5	May 5	May 1	May 2	May 8	May 16	May 2	Apr. 15	Apr. 28
L42 approximate ice-on date		Nov. 5	Nov. 8	Nov. 15	Nov. 15	Nov. 2	Nov. 10	Nov. 12	Nov. 23	Dec. 1
Average stratification (days)		152	150	154	159	156	147	150	177	159
June–Sept. rainfall (mm)	393		483	414	489	434	440	277	257	409
Annual rainfall (mm)[c]	581				570	623	610	409	504	592

[a]1966–1988 Environment Canada Canadian climate normals at Atikokan, 70 km southeast of the study lakes (http://www.cmc.ec.gc.ca/climate/normals/ONTA006.HTM).
[b]Moisture content of total snowfall.
[c]1994–1999 data are April–October precipitation.

Table 4. 1995–1999 May 12 to October 31 midlake and upland mean and mean hourly maximum wind velocities ($m \cdot s^{-1}$, 95% sample confidence intervals in parentheses).

Lake	Year	Hourly mean	Hourly maximum mean
L26	1995	1.99 (0.38, 5.50)	4.26 (0.98, 12.97)
	1996	1.85 (0.34, 5.07)	4.09 (0.94, 12.40)
	1997	2.06 (0.38, 5.82)	4.46 (1.03, 13.69)
	1998	1.90 (0.32, 5.36)	4.21 (0.94, 12.95)
L39	1995	2.26 (0.48, 6.18)	4.61 (1.23, 13.09)
	1996	2.07 (0.36, 5.95)	4.38 (1.10, 12.80)
	1997	2.39 (0.51, 6.61)	4.85 (1.31, 13.80)
	1998	2.26 (0.50, 6.07)	4.68 (1.29, 13.12)
	1999	2.30 (0.47, 6.44)	4.73 (1.25, 13.59)
L42	1995	1.94 (0.34, 5.45)	4.24 (1.04, 12.44)
	1996	1.88 (0.34, 5.19)	4.17 (1.04, 12.08)
	1997	2.14 (0.47, 5.71)	4.60 (1.25, 12.90)
	1998	2.10 (0.43, 5.75)	4.47 (1.17, 12.76)
	1999	2.08 (0.29, 6.38)	4.43 (1.00, 13.74)
Upland	1995	2.04 (0.53, 5.07)	4.58 (1.71, 10.49)
	1996	1.84 (0.45, 4.55)	4.21 (1.27, 10.93)
	1997	1.87 (0.45, 4.70)	4.36 (1.29, 11.54)
	1998	1.73 (0.44, 4.17)	4.16 (1.26, 10.78)
	1999	1.61 (0.39, 3.89)	3.99 (1.17, 10.47)

Fig. 1. Ratios of midlake to upland average wind velocity for the study lakes L26 (open bars), L39 (gray bars), and L42 (black bars), 1995–1998. Data are May 12 to October 31 annual averages. The 1996–1999 upland wind data were corrected for declines of 0.097 $m \cdot s^{-1} \cdot year^{-1}$ associated with forest regeneration around the upland climate station.



longed circulation in all years except spring 1995, 1995, and 1999.

Stratification duration averaged 156 days (all lakes and all years), spanning April 25 to October 24, and varied from 150 days in 1992 and 1997 to 177 days in 1998 (Table 3). The lakes were stratified an average of 8 days longer in the years after logging, due mainly to relatively warm April temperatures and early ice-off dates in those years. There was no indication that stratification on L39 and L42 was shortened relative to L26 after logging, as could be expected if wind energy increased after shoreline deforestation.

Summer average water clarity as indexed by 1% PAR depth had declined 3–5 m, or about 25%, by the third year after logging in all of the study lakes (Table 6). Measurements of Secchi depth and epilimnetic PAR extinction proved too variable to provide strong evidence of change, but this could also indicate that transparency declines occurred mainly in subepilimnetic waters. Although water clarity measurements varied considerably through the May–August monitoring period, early-summer and late-summer water clarity was generally similar.

Thermocline depths were most variable in early summer and tended to deepen over the summer from about 4 m in early May to about 9 m in late September. Three years after logging, June thermocline depths averaged 1–3 m deeper in all lakes, while September thermocline depths were about 1 m shallower in L26 and L39 but not in L42 (Table 7). However, the three study lakes had shown similar, synchronous changes in late-summer thermocline depths prior to deforestation (Fig. 2), particularly in 1996, which was

Table 5. Occurrence of complete and complete, prolonged spring and fall circulation in the study lakes, 1991–1999, defined as restoration of at least 4.0 (x) or 8.0 (X) mg dissolved oxygen·L^{-1} within 3 m of maximum lake depth.

Lake	Season	1991	1992	1993	1994	1995	1996	1997	1998	1999
L26	Spring	x		x				x	x	x
	Fall		x			x				
L39	Spring	x	X	X	X	x	x	X	X	x
	Fall	x	X		X	X	X	x	X	x
L42	Spring	X	X	X	X	x	x	X	X	x
	Fall	X	X	X	X	X	X	X	X	X

excluded from the analyses in Table 7 because thermoclines that year were potentially influenced by both precut and postcut conditions. If 1996 is included as a "prelogging" year in Table 7, estimates of the 1999 declines in September thermocline depth shrink by about 0.2 m, and the 95% confidence intervals increase by about 40%.

All estimates of postlogging change in minimum lake trout habitat volume had 95% confidence intervals that spanned zero by a substantial margin (Table 8). L42 (which had about 20–25% of the lake trout habitat volume of L26 and L39, depending on the index) was the only lake that showed a consistent pattern of postlogging declines, with 15.5/4 habitat down about 40% in 1998 and 1999.

The indices showed different responses to seasonal changes in thermal stratification and HOD. The 10/6 and 15.5/4 habitat indices showed similar seasonal patterns, with low year-to-year variability and a strong downward trend during the thermally stratified period. The upper temperature limits of these two indices generally excluded epilimnetic water from June to September, and they therefore had upper boundaries somewhere in the thermocline from about 4 to 12 m depth. The 10/6 habitat was always in shorter supply than 15.5/4 habitat during the summer and always disappeared completely in L42 by September or October. The 15.5/4 habitat was never completely eliminated from L42, although it declined to around 10% of lake volume each September in L42 and to 20–40% in L39 and L26. High levels of 10/6 habitat were always restored during the month of October in all lakes. The 20/6 and /6 habitats were always relatively abundant in the study lakes. The 20/6 habitat volume was strongly affected when epilimnion temperature fluctuated above and below 20°C. This produced very high annual and interannual variability in the index during the thermally stratified period. L42 had the lowest amount of 20/6 habitat, with average annual minima of about 20–40% of lake volume. L39 and L26 had average annual minima of about 40–50%. The /6 habitat volume had low variability and was always greater than about 80% of lake volume in all three lakes. This index showed a slight downward trend during thermally stratified period, with annual minima in October.

Discussion

Although midlake wind speeds on the study lakes did not increase much after clearcut logging, average wind velocities close to lee shores probably increased to a greater extent. Shoreline forest should have an effect on wind velocity analogous to that of agricultural shelterbelts. In the lee of forested shelterbelts, wind velocity may be reduced by about 40–80% within five times tree height and by 20–60% between five and 10 times tree height, depending on forest type. On the windward side, velocity is reduced by 20–40% within one to two times tree height (Geiger 1965, cited in Bartholow 1989). A lee shore one day may be a windward shore the next, and on all but the smallest lakes, maximum nearshore wind exposure will be determined mainly by upwind fetch rather than local shoreline features. Where lakes are surrounded by steep terrain, trees are likely to be responsible for only a portion of the total shelter provided by shoreline features (Ying et al. 1994). In flat terrain, removing a shoreline windbreak of dense forest 20 m high would increase effective lake fetch by about 120 m, according to Geiger's (1965) data. Adding 120 m fetch to a lake with 1 km fetch could be expected to increase thermocline depth by about 0.2 m, according to the empirical relationship developed by Shuter et al. (1983). The study lakes showed an opposite effect, suggesting that the influence on thermocline depth of increased wind energy after logging was secondary to other factors such as reduced water clarity. Only small lakes in relatively flat terrain, with fetch less than a few hundred metres, should experience significantly increased wind exposure after shoreline deforestation. This has been shown for a 0.4-ha lake in Finland (Rask et al. 1993) and for a 4-ha lake basin in Michigan (Scully et al. 2000).

Our midlake wind measurements were consistent with data provided in Solinske (1982) but not with France (1997), who interpreted Solinske's (1982) data to suggest that easterly wind velocities on 56-ha Experimental Lakes Area L239 tripled after deforestation along the east shore of that lake. Our analysis of Solinske's (1982) entire July 7 to August 25, 1981, data set showed that easterly and westerly L239 midlake to upland wind speed ratios had broadly overlapping distributions, with means of about 0.89 and 0.81, respectively. Velocity of easterly midlake wind on L239 averaged about 2.2 m·s^{-1} (SD = 0.8 m·s^{-1}), or about 0.6 m·s^{-1} lower than westerly winds during that monitoring period.

The development of relatively deep June thermoclines in all lakes and unusually efficient spring circulation in L26, in 1998 and 1999, were probably not associated with deforestation. Decreased water clarity measured in those years would be more likely to produce shallower thermoclines (Fee et al. 1996). There was no indication that L26, which retained all shoreline forest, experienced increased wind exposure as a result of upland logging. We speculate that deep spring thermoclines in L26 were associated with the warm and windy April and May weather that caused early ice melting in 1998 and 1999. Prolonged spring mixing was not in-

Table 6. Water clarity indices in the study lakes before and after experimental logging, estimated by RM-ANOVA with years as cases and May–August monthly observations as repeated factors within years.

					Mean difference after logging			
Clarity index	Lake	*n* prelogging: years × months	*n* postlogging: years × months	Mean prelogging	1997–1999 pooled	1997	1998	1999
Secchi depth (m)	L26	5 × 4	3 × 4	8.1 (7.6, 8.7)	–0.3 (–1.0, 0.4)	–0.2 (–1.6, 1.2)	–0.0 (–1.4, 1.3)	–0.6 (–2.0, 0.7)
	L39	5 × 4	3 × 4	7.1 (6.2, 8.0)	–0.2 (–1.7, 1.1)	–0.5 (–2.7, 1.7)	0.7 (–1.5, 2.9)	–1.0 (–3.2, 1.2)
	L42	5 × 4	3 × 4	6.7 (5.9, 7.5)	–0.3 (–1.4, 0.7)	–0.8 (–2.8, 1.2)	0.1 (–1.9, 2.2)	–0.4 (–2.4, 1.7)
PAR extinction (m^{-1})	L26	2 × 4	3 × 4	0.22 (0.13, 0.30)	0.03 (–0.00, 0.07)	0.03 (–0.12, 0.17)	0.02 (–0.12, 0.17)	0.05 (–0.10, 0.19)
	L39	2 × 4	3 × 4	0.26 (0.13, 0.40)	0.02 (–0.11, 0.15)	0.01 (–0.22, 0.24)	–0.02 (–0.25, 0.20)	0.08 (–0.15, 0.30)
	L42	2 × 4	3 × 4	0.27 (0.25, 0.28)	–0.01 (–0.03, 0.02)	–0.02 (–0.04, 0.01)	–0.01 (–0.04, 0.02)	0.00 (–0.02, 0.03)
1% PAR depth (m)	L26	2 × 4	3 × 4	17.0 (16.7, 17.2)	–2.6 (–6.2, 0.9)	–1.5 (–1.9, –1.2)*	–2.0 (–2.4, –1.7)*	–4.3 (–4.7, –4.0)*
	L39	2 × 4	3 × 4	14.8 (10.4, 19.1)	–2.2 (–8.5, 4.1)	–1.1 (–8.7, 6.5)	–0.3 (–7.9, 7.2)	–5.2 (–12.8, 2.3)
	L42	2 × 4	3 × 4	13.4 (12.8, 14.1)	–0.9 (–4.7, 2.9)	0.1 (–1.0, 1.2)	–0.0 (–1.1, 1.1)	–2.7 (–3.8, –1.6)*

Note: Monthly observations were means of two to five weekly measurements. The prelogging period of record was 1991–1995 for Secchi depth and 1994–1995 for epilimnetic PAR extinction and 1% PAR depth. The 95% confidence limits are given in parentheses. Asterisks identify differences where the 95% confidence interval does not span zero.

Table 7. Thermocline depths (m) in the study lakes before and after experimental logging, estimated by RM-ANOVA with years as cases and weekly observations in June or September as repeated factors within years.

							Mean difference after logging		
Month	Lake	*n* prelogging: years × observations per year	*n* post-logging: years × observations per year	Calendar weeks used in analysis	Mean prelogging	1997–1999 pooled	1997	1998	1999
June	L26	6 × 4	3 × 4	23–26	4.6 (3.9, 5.3)	1.2 (–0.5, 2.9)	–0.4 (–2.3, 1.4)	2.1 (0.2, 3.9)*	1.3 (–0.5, 3.1)
	L39	6 × 4	3 × 4	23–26	4.7 (3.8, 5.6)	1.7 (–0.4, 3.7)	–0.3 (–2.7, 2.0)	2.1 (–0.2, 4.4)	3.2 (0.9, 5.6)*
	L42	6 × 4	3 × 4	23–26	4.5 (3.5, 5.4)	0.8 (–0.8, 2.4)	–0.3 (–2.9, 2.2)	1.5 (–1.1, 4.1)	1.4 (–1.2, 3.9)
Sept.	L26	5 × 3	3 × 3	36, 37, 39	9.3 (8.8, 9.8)	–0.2 (–1.4, 1.0)	0.6 (–0.6, 1.8)	0.2 (–1.0, 1.4)	–1.4 (–2.6, –0.2)*
	L39	5 × 3	3 × 3	36, 37, 39	9.3 (8.9, 9.7)	–0.2 (–1.1, 0.8)	–0.1 (–1.1, 1.0)	0.5 (–0.5, 1.6)	–1.0 (–2.0, 0.1)
	L42	5 × 3	3 × 3	36, 37, 39	9.5 (9.0, 10.0)	–0.1 (–0.5, 0.8)	0.4 (–0.8, 1.6)	0.2 (–1.0, 1.4)	–0.2 (–1.4, 1.0)

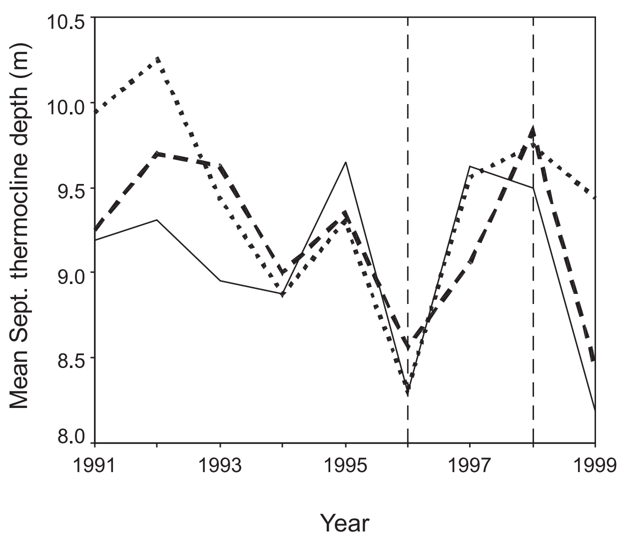
Note: Prelogging (1991–1995) annual cases were contrasted with postlogging (1997–1999) cases. The 1996 data were included as prelogging for June comparisons and excluded from September comparisons. The 95% confidence limits are given in parentheses. Asterisks identify differences where the 95% confidence interval does not span zero.

Table 8. Annual minimum values of various lake trout habitat indices (10^5 m^3) in the study lakes before (1991–1995) and after (1997–1999) logging, estimated by regular ANOVA.

			Mean difference after logging			
Lake	Index	Mean prelogging habitat volume	1997–1999 pooled	1997	1998	1999
L26	10/6	8.6 (7.0, 10.2)	0.7 (–2.6, 4.0)	–0.8 (–7.6, 6.0)	–0.9 (–7.7, 5.9)	3.8 (–3.0, 10.6)
	15.5/4	15.5 (13.8, 17.3)	0.5 (–1.7, 2.7)	0.7 (–6.9, 8.2)	–0.2 (–7.7, 7.3)	1.2 (–6.3, 8.7)
	20/6	20.6 (13.1, 28.0)	–2.6 (–11.5, 6.3)	–4.0 (–35.9, 28.0)	–2.1 (–34.1, 29.8)	–1.7 (–33.7, 30.2)
	/6	30.7 (28.6, 32.8)	0.9 (–2.1, 3.8)	1.6 (–7.3, 10.4)	–1.0 (–9.9, 7.9)	2.1 (–6.8, 10.9)
L39	10/6	3.2 (0.0, 6.0)	–0.3 (–4.1, 3.5)	0.9 (–11.0, 12.8)	–2.5 (–14.4, 9.5)	0.7 (–11.3, 12.6)
	15.5/4	12.6 (10.0, 15.2)	–0.9 (–4.7, 2.9)	–2.2 (–13.2, 8.8)	–2.1 (–13.1, 8.9)	1.7 (–9.3, 12.7)
	20/6	21.3 (9.4, 33.1)	–4.5 (–18.5, 9.6)	–3.5 (–54.2, 47.3)	–6.5 (–57.2, 44.3)	–3.5 (–54.3, 47.2)
	/6	36.9 (34.3, 39.5)	–4.0 (–9.1, 11.6)	–1.3 (–12.4, 9.7)	–2.0 (–13.1, 9.0)	–8.6 (–19.6, 2.5)
L42	10/6	0.0	0.0	0.0	0.0	0.0
	15.5/4	2.7 (2.0, 3.3)	–0.6 (–1.5, 2.1)	–0.3 (–3.1, 2.5)	–0.9 (–3.7, 1.9)	–0.8 (–3.6, 2.1)
	20/6	8.1 (1.5, 14.7)	–3.4 (–11.3, 4.5)	–2.0 (–30.2, 26.1)	–5.0 (–33.2, 23.1)	–3.2 (–31.4, 24.9)
	/6	17.3 (14.4, 20.3)	–1.6 (–6.1, 3.0)	–3.1 (–15.9, 9.7)	1.7 (–11.1, 14.5)	–3.3 (–16.1, 9.5)

Note: The 95% confidence limits are given in parentheses.

Fig. 2. Average September thermocline depths in the study lakes L26 (solid line), L39 (dashed line), and L42 (dotted line), 1991–1999. The vertical dashed lines show years when experimental logging occurred.



volved, since the delay between estimated ice-out and onset of stratification was about 9 days in 1998 and 4 days in 1999, somewhat shorter than the range of 8–15 days observed from 1994 to 1996.

Decreased September thermocline depths in all lakes in 1999 were consistent with postlogging DOC increases of up to 15%, chlorophyll increases of 15–30% (Steedman 2000), and concurrent measurements of decreased water clarity, particularly 1% PAR depth. Unlike thermocline depths, the declines in 1% PAR depth were unprecedented in the 1991–1996 prelogging record and were strongly suggestive of a logging impact. Deep phytoplankton communities that might cause subepilimnetic turbidity showed no significant changes from 1997 to 1999 in any of the study lakes (D. Graham, Department of Civil and Environmental Engineering, University of Kansas, Lawrence, KS 66045, U.S.A., unpublished data). However, decreased water clarity may have been associated with wet weather in 1999, which would have increased catchment exports of DOC and nutrients. Paleolimnological analyses of sediments from the study lakes have indicated that historical sediment accumulation and phytoplankton community composition were more strongly associated with long-term precipitation patterns than with episodic wildfire events in the lake catchments (Blais et al. 1998; Paterson et al. 1998).

Our observations of summer thermocline progression contrast somewhat with those of Schindler (1971) and Ryan and Marshall (1994), who suggested that thermocline depths in northwestern Ontario lakes were generally stable after onset of stratification. Thermocline progression of a metre per month complicates comparative studies where lake profiles are measured over an extended period (e.g., France 1997) and could account for some variation attributed to other factors such as wind exposure and water clarity.

The shoreline buffer strip around L26 probably influenced the response of that lake to clearcut logging by preventing the minor increases in midlake wind velocity that were observed on L39 and L42. However, the buffer strip did not prevent declines in water clarity and fall thermocline depth. These have the potential to increase lake trout habitat volume in the short term but did not appear to do so in this study, perhaps because the impacts were small. Such impacts would be compensatory to declines in habitat volume ultimately expected from increased lake productivity and HOD. Since none of the lakes showed changes in lake trout habitat volume as of 1999, the effectiveness of the shoreline buffer may be a moot point in that regard.

There are no reliable calibrations relating lake trout viability, productivity, age structure, or mortality to lake trout habitat volume indices. As a species, lake trout appear to tolerate a wide range of habitat conditions. Natural occurrence of lake trout in boreal lakes can be predicted reasonably well simply by the presence of thermal stratification, and viable lake trout populations can be found in lakes with habitat volumes far smaller than those proposed by Evans et al. (1991)

as minimum safe values, particularly in lakes with simple fish communities (Marshall and Ryan 1987; Ryan and Marshall 1994; Sellers et al. 1998). However, indices of cool oxygenated water volume are useful for monitoring HOD in oligotrophic shield lakes subjected to catchment disturbance or nutrient inputs. Such indices will be most informative regarding threats to lake trout in lakes with complex epilimnetic fish communities, where lake trout are more likely to be restricted to the hypolimnion. Of the four indices examined, the 15.5/4 index appears to be the most useful general index of cool oxygenated water volume. During the period of thermal stratification, the 15.5/4 index showed a strong seasonal trend and low variability and was greater than zero (i.e., "stayed on the scale") in L42, the shallowest lake trout lake.

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