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Deep chlorophyll maxima in small boreal forest lakes after experimental catchment and shoreline logging

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In situ profiles of chlorophyll fluorescence combined with depth-specific water chemistry and biotic analysis were used to assess deep chlorophyll maxima (DCM) in four boreal shield lakes that were subjected to different levels of watershed disturbance by clear-cut logging. Phytoplankton communities within the DCM varied among lakes, but were generally comprised of chrysophytes, diatoms, and cryptophytes. One lake also had deep cyanobacterial populations. Relative *in situ* fluorescence (absolute fluorescence at each peak minus background fluorescence) at the fluorescence peaks was significantly correlated with the level of chlorophyll *a* at the peak ($r^2 = 0.81$, $p < 0.025$) for all lakes. In general, DCM peak heights were consistent in each lake over the four-year post-logging sampling period, whereas DCM depths declined slightly in the most heavily logged lakes associated with increases in dissolved organic carbon. Therefore, on the basis of DCM data, only mild effects on lake water quality were apparent after the logging activity.

Introduction

Deep chlorophyll maxima (DCM) are found in many thermally stratified lakes and reservoirs, and are generally associated with aggregations of algae and other photosynthetic organisms (Fee 1976, Moll and Stoermer 1982, deNoyelles and Likens 1985, Lindholm 1992, Reynolds 1992). DCM are typically located in the metalimnion or the upper hypolimnion, with their size, vertical position, and composition varying with thermocline depth, light penetration, lake fertility, localized focusing of nutrients, and the oxygen status of the hypolimnion (Moll and Stoermer 1982,

Lindholm 1992). Because they usually reside in or around the metalimnion where nutrient and other resource gradients are steepest, DCM can respond sensitively to even minor changes in nutrient status or water clarity and be bio-indicators of changes in lake water quality (e.g., Battoe 1985, Kettle *et al*. 1987, Christensen *et al*. 1995).

Disturbance of forested catchments can alter lake water quality and phytoplankton abundance (Rask *et al*. 1993, Rask *et al*. 1998, Carignan *et al*. 2000), but associated responses in DCM, as indexed by *in situ* fluorometry, have not previously been reported. *In situ* fluorometry is a useful tool for water quality monitoring because it can be performed quickly in the field and provides a rapid index of phytoplankton distribution in a water column. It also provides a continuous profile of the product of physical and chemical conditions and, therefore, it can detect local changes in water chemistry that might be missed through water monitoring at specific depths. The method has its limitations (*see* Salonen *et al*. 1999); however, the method is fast and simple and, with calibration, can provide "real-time" data on DCM communities and other conditions in a water column.

This paper presents results from a four-year post-disturbance sampling program assessing DCM using *in situ* fluorometry in four lakes with different levels of catchment disturbance due to logging. Steedman (2000) had showed that only small changes in water quality were noted in the study lakes that had major watershed disturbance; however, no supporting biological data or depth-specific profiling were reported. This paper, therefore, presents such supporting data, updating and confirming the previous observations on lake water quality during and post watershed deforestation.

Materials and methods

Study sites

The lakes monitored in this study are located in the Ontario Ministry of Natural Resources' Coldwater Lakes Experimental Watersheds located in the boreal-Great Lakes transition forest on the Canadian Shield (Steedman 2000). Lakes L20, L26, and L39 are comparatively deep (Table 1) and are aerobic to depths greater than 20 m (below the euphotic zone), whereas lake L42 is shallower than the other lakes (~17 m) and is the only lake where the euphotic zone crosses into an anoxic hypolimnion. All four lakes are oligotrophic and have 1% PAR (photosynthetically active radiation) depths between about 7 to 20 m.

In 1996, the catchments of L26, L39 and L42 were partially clearcut using a tracked feller buncher and chainsaws (Table 1). Trees were dragged by skidders to the nearest road and de-limbed. L26 had moderate catchment disturbance (33%), with no disturbance of shoreline forest, whereas L39 and L42 had extensive catchment (60% to 70%) and shoreline disturbance (40% to 60%). In 1998, additional logging occurred on the shoreline of L42 (19%

Table 1. Study lakes basin and catchment morphometry.

Lake	L20	L26	L39	L42
Max. depth (m)	32	37	23	18
Surface area (ha)	57	29	39	26
Lake volume (10^6 m^3)	7.4	4.1	4.6	2.2
Total catchment (ha)	524	106	194	70
Catchment/Surface area	8.2	2.6	1.9	1.6
% Net terrestrial disturbance (1996/1998)	7	45	77	74
Experimental role	Reference	Moderate catchment disturbancebut no shoreline disturbance	Extensive catchment and shoreline disturbance	Extensive catchment and shoreline disturbance

of shoreline length; 3% of catchment), and on stream catchments draining to L26 (12% of catchment) and L39 (15% of catchment). Catchment and clearcut boundaries extended between 100 and 600 m from the lake shorelines. About 5 km of logging roads were constructed in the catchments of L39 and L42, and about 2 km of roads were constructed around L26. See Steedman and Kushneriuk (2000) for further detail on these lakes.

L20 was monitored as a reference lake in this study. L26 is in the catchment of L20, therefore deforestation around L26 affected about 7% of the L20 catchment area. Although L20 was used a reference lake here, it was not ideal because its catchment-to-lake surface ratio was higher than the other lakes (which may impact its response to drought conditions) and it was slightly more colored than the other lakes. Regardless, it was exposed to the same general weather conditions as the other lakes and provided a fair contrast to the lakes with deforestation in the study.

Water chemistry and fluorescence profiles

Detailed *in situ* fluorometry, in conjunction with water chemistry analyses, was performed during seven extended site visits over the four-year study period. The visits were in the Augusts of 1997 through 2000 for yearly comparisons, and in late May, early July, and mid-September of 1998 for seasonal comparisons. Additional water quality samples were collected and analyzed (typically bi-weekly) associated with the regular monitoring program. Some pre-2000 monitoring data from the lakes were reported previously by Steedman (2000). Late summer was chosen as the time for *in situ* fluorometric profiling because total phosphorus (TP) and chlorophyll *a* levels in regional lakes tend to be most stable at that time of year, and because water column nutrient, oxygen, and thermal gradients are usually strongest (Fee *et al*. 1978, deNoyelles and Likens 1985, France *et al*. 1995).

Each major fluorometry sampling event included the measurement of vertical *in situ* fluorescence profiles, dissolved oxygen (DO), pH, temperature, and light intensity; and the collection of discrete water samples for chemical determinations and for phytoplankton identification. Fluorescence profiles were always measured first using a Turner Model 10-005R field fluorometer equipped with a depth-calibrated hose and continuous-flow pump apparatus (Kettle *et al*. 1987). Fluorescence profiles were developed by lowering the hose through the water column at a constant rate, while continuously pumping water to the surface and through the fluorometer. The profiles were digitized from field chart recorder printouts using internally developed computer software upon returning to the laboratory (to allow computer-based image comparisons).

DO, pH and temperature were then measured using a Horiba Water Checker (Horiba Instruments), and light intensity was determined using a LI-COR spherical quantum sensor (Model LI-185A). For each case, the instrument probe was passed down the water column with readings being recorded either at 1-m intervals or at specific depths where fluorescence maxima had been noted during the fluorescence scan. Discrete samples for water chemistry and phytoplankton identification were collected last, using the pump-and-hose apparatus associated with the fluorometer.

Discrete samples were collected for the following analyses: total nitrogen (TN), total phosphorus (TP), dissolved organic carbon (DOC), chlorophyll *a*, and phytoplankton identification. Typically between five and eight samples were collected per profile per sampling event, representing one to three samples from the epilimnion, two to five samples from the vicinity of the observed fluorescence maxima (the DCM peak), and one or two samples from the deep hypolimnion. Water samples were either collected in 500-ml screw-capped amber bottles (TN, TP, DOC, and chlorophyll *a*), or 100-ml clear screw-capped bottles (phytoplankton). The samples were temporarily stored on ice in the dark prior to returning to the laboratory for sample processing.

All samples for phytoplankton characterization were preserved at the time of sampling using Lugol's iodine solution. Samples were then pre-settled and phytoplankton was identified to the genus-level using an Olympus inverted microscope (400× magnification) and

associated counting cells. Details of sample processing methods and analytical procedures have been described elsewhere (Graham *et al.* 1999a, Graham *et al.* 1999b, Steedman 2000).

Results and discussion

DCM phytoplankton composition in the study lakes

Phytoplankton communities observed in the DCM of the four lakes were not atypical of late summer phytoplankton commonly seen in oligotrophic, north temperate lakes (Findlay *et al.* 2001). Although absolute phytoplankton numbers were not determined in detail, clear trends in genera and algae numbers associated with the observed DCM peaks were apparent upon examination of the preserved samples. With the exception of L42, the lakes were dominated (typically > 90% of observed organisms per peak) by chrysophyte, diatom and cryptophyte genera (Table 2). The DCM in L42, unlike the other lakes, was dominated by *Limnothrix rosea*, a cyanobacterial species that is well adapted to low light conditions in anoxic hypolimnion (Meffert 1988). In general (except L42 in 1997), the phytoplankton composition of the DCM in each lake was consistent from year-to-year over the four-year study period.

Correlation between field relative fluorescence (RF) and chlorophyll *a*

To provide data comparable with other studies on DCM, field fluorescence was calibrated to analyzed levels of chlorophyll *a*. Relative *in situ* fluorescence (RF; the absolute fluorescence at each peak minus background fluorescence) significantly correlated with chlorophyll *a* in the four lakes ($r^2 = 0.81$, $p < 0.025$) with one RF unit being equal to 6.3 μg l^{-1} of extracted chlorophyll *a*. RF observations in the lakes ranged from 0.1 to about 5.5 units, which is low compared with most lakes that develop late-summer DCM (Moffett 1991).

Table 2. Dominant phytoplankton in the DCM of the study lakes. DCM depths are noted in parentheses. Multiple depths are provided to describe "local" peaks within the DCM of each lake.

Lake	1997	1998	1999	2000
20	(7.0 m) *Uroglena*, *Cryptomonas*	(7.0 m) *Cyclotella*, *Cryptomonas*, *Aphanocapsa*	(8.6 m) *Uroglena*, *Cryptomonas*, *Synura*	(5.0 m) *Uroglena*
26	(15.5 m) *Uroglena*	(15.0 m) *Cyclotella*, *Cryptomonas*	(13.0 m) *Uroglena*, *Synura*	(12.0 m) *Cyclotella*, *Cryptomonas*
39	(15.5 m) *Ankistrodesmus*, *Ankistrodesmus*, *Asterionella*, *Cryptomonas*, *Mallomonas*	(15.0 m) *Cryptomonas*, *Asterionella*, *Mallomonas*,	(11.0 m) *Cryptomonas*, *Chrysosphaerella*	(13.0 m) *Ankistrodesmus*, *Uroglena*
42	(16.0 m) *Cryptomonas*, *Asterionella*, *Ankistrodesmus*	(12.0 m) *Cryptomonas* (17.0 m) *Limnothrix*, *Cryptomonas*	(13.0 m) *Cryptomonas*, *Mallomonas* (15.4 m) *Limnothrix*, *Cryptomonas* (16.8 m) *Limnothrix*	(16.0 m) *Limnothrix*, *Cryptomonas*

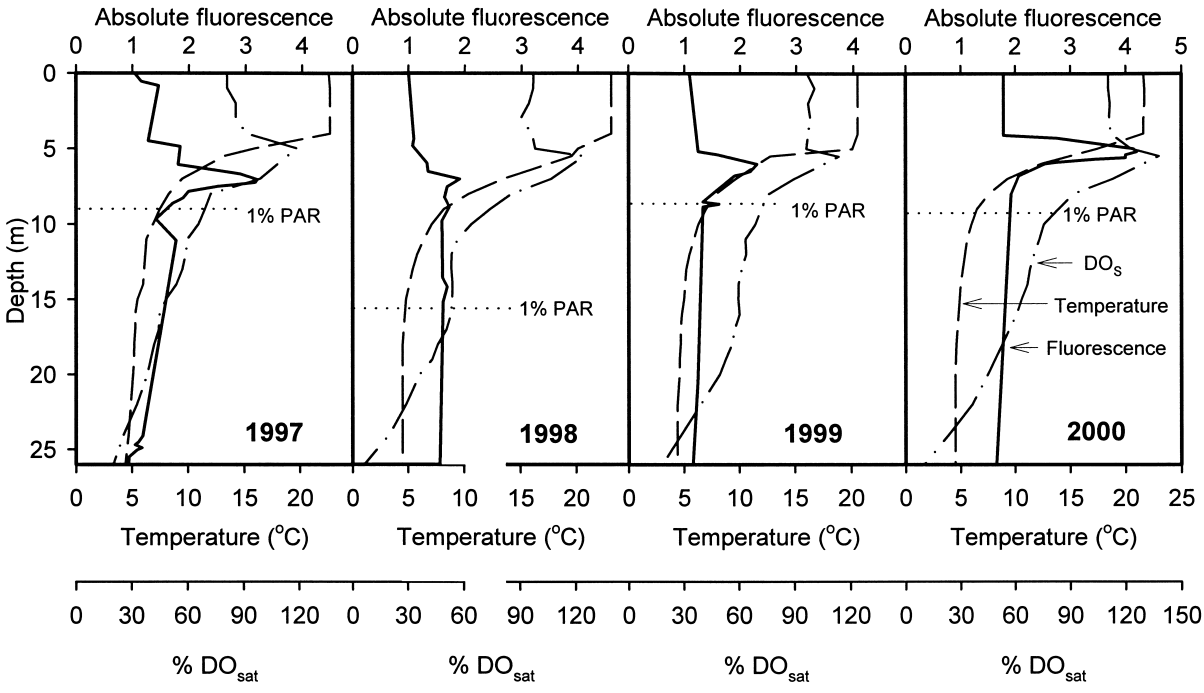

Fig. 1. Vertical fluorescence and temperature profiles for L20 from early August 1997, 1998, 1999 and 2000. The depth to 1% PAR is noted by the dashed lines.

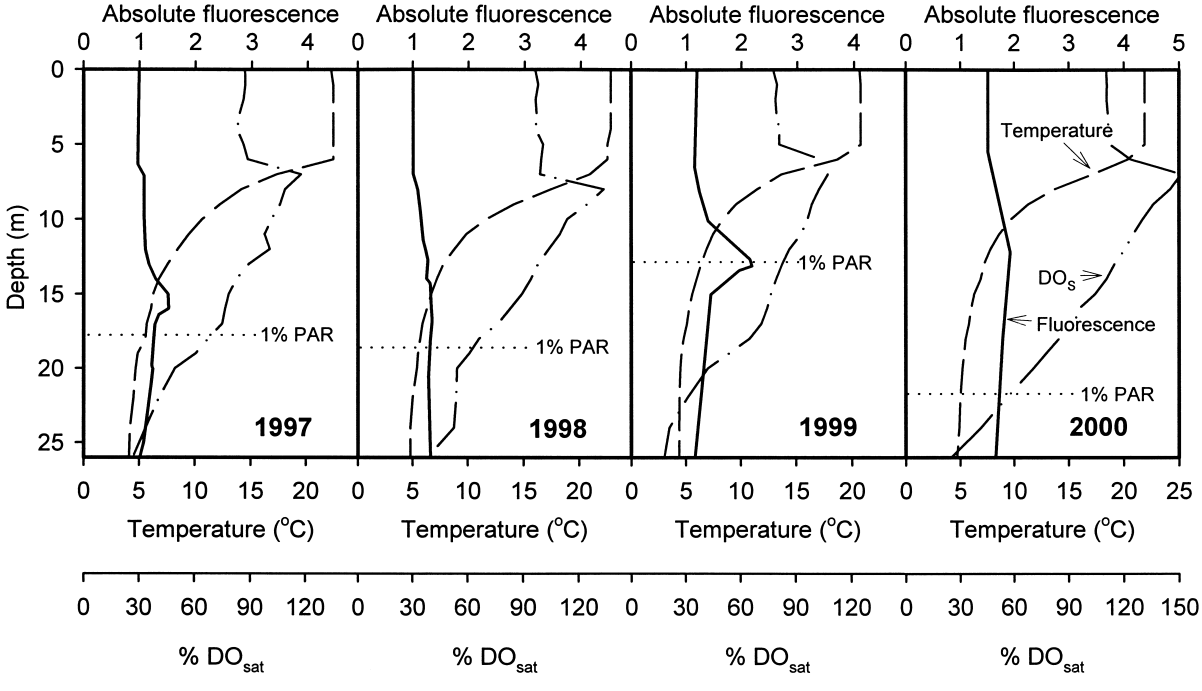

Fig. 2. Vertical fluorescence and temperature profiles for L26 from early August 1997, 1998, 1999 and 2000. The depth to 1% PAR is noted by the dashed lines.

Post-logging trends in August DCM from 1997–2000

Late-summer DCM peak heights and shapes varied slightly among years, but did not show any major long-term trends in any of the lakes between 1997 and 2000 (Figs. 1–4). Similarly, chlorophyll *a* levels were generally unchanged over the monitoring period (*see* Table 3). Late-summer DCM peaks were always small in the

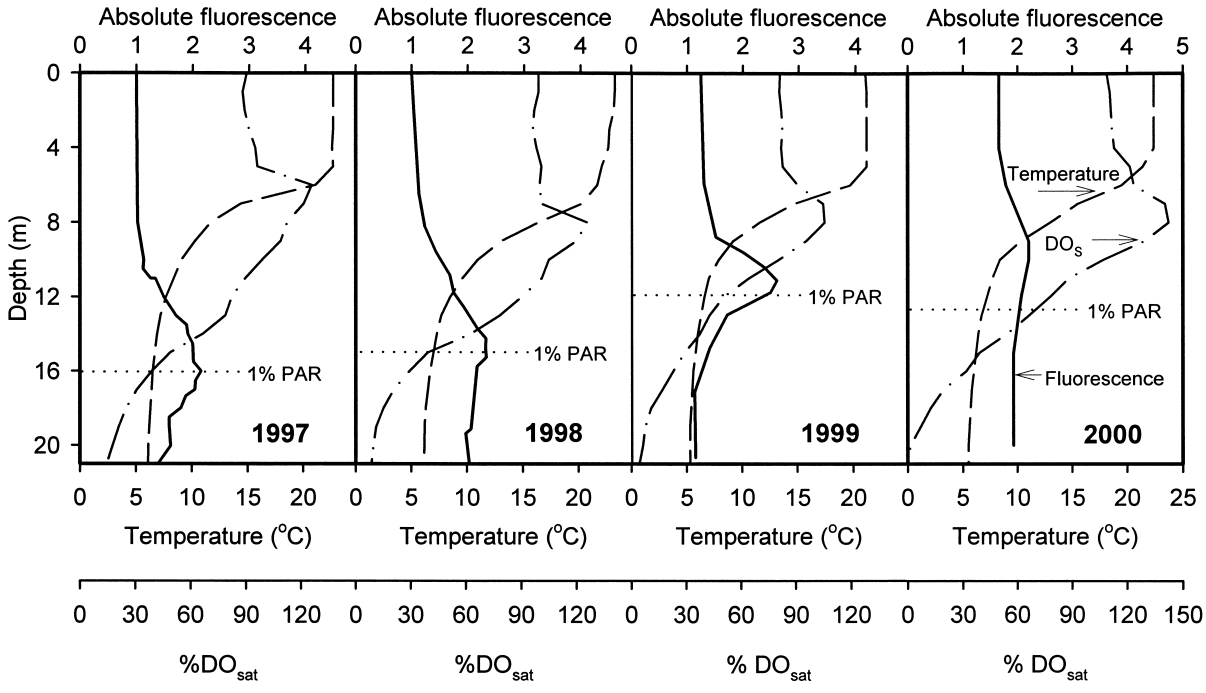

Fig. 3. Vertical fluorescence and temperature profiles for L39 from early August 1997, 1998, 1999 and 2000. The depth to 1% PAR is noted by the dashed lines.

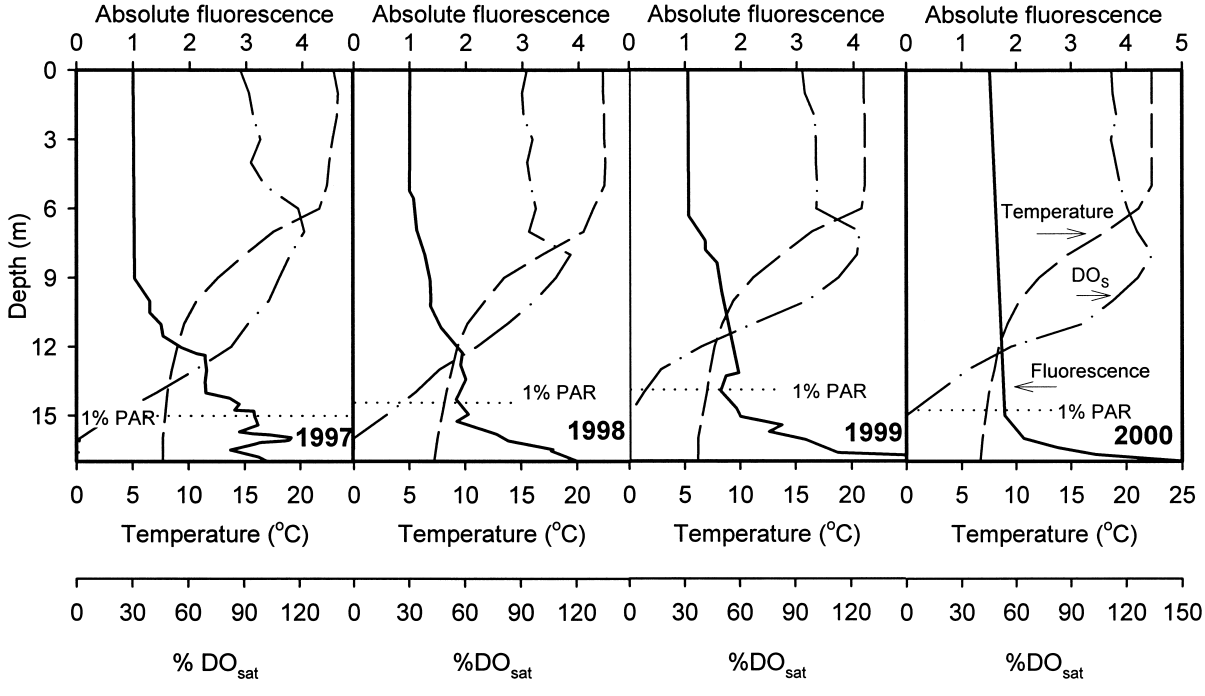

Fig. 4. Vertical fluorescence and temperature profiles for L42 from early August 1997, 1998, 1999 and 2000. The depth to 1% PAR is noted by the dashed lines.

deeper lakes with a mean RF value of 1.2, and larger in L42 with a mean RF value of 3.8. The L42 DCM was always at or near the top of the hypolimnion, and did not change position in the water column through the 1997–2000 monitoring period. In contrast, DCM depths generally

trended shallower in the deeper lakes from 1997–2000 (Figs. 1–3), roughly corresponding with increases in DOC, especially in the logged lakes (*see also* Table 3). Figure 5 shows that the 1% PAR and DCM depths were significantly correlated with water column DOC levels in L20, L26, and L39.

Seasonal changes in DCM in 1998

Table 4 summarizes DCM peak sizes from all four study-years and includes seasonal data from 1998 collected between May and September (seasonal patterns were similar in other years; unpublished results). Although DCM peaks were always small in the deeper lakes (L20, L26 and L39), they were consistently greatest in May and became smaller as summer proceeded. With exception of L20, DCM in the deeper lakes showed little seasonal variation in depth and were usually located near the 1% PAR depth of each lake (Table 4 and 1998 data in Figs. 1–4). The vertical location of the DCM in L20 was most variable and was usually located at about one-half the depth of the DCM in the other deep lakes.

In contrast, the DCM in L42 was physically larger than in the deeper lakes and was consistently present at the bottom of the lake either just above or in the anoxic hypolimnion. DCM peak sizes in L42 were always largest in the late summer (August).

Relationships amongst logging, water chemistry, and DCM communities

Previous studies have shown that DCM communities can change dramatically due to small changes in nutrient and/or chemical conditions in thermally stratified lakes (Battoe 1985, Shortreed and Stockner 1990, Moffett 1991, Christensen *et al*. 1995). In this study, we had hypothesized that changes in DCM size, depth, and composition might result due to changes in water

Table 3. Summary of 1991–2000 May-to-August water-column average DOC and chlorophyll *a* concentrations, and 1% PAR depths (1994–2000 data) in the study lakes. Approximate 95% confidence interval (2 S.D.) is shown in parentheses.

Year	L20	L26	L39	L42
DOC (mg-C l^{-1})[a]				
1991–1996	4.0 (0.1)	2.1 (0.1)	2.2 (0.1)	2.4 (0.3)
1997	3.1 (0.3)	1.8 (0.1)	1.9 (0.1)	2.0 (0.1)
1998	3.9 (0.2)	2.4 (0.3)	2.5 (0.3)	2.6 (0.2)
1999	3.9 (0.4)	2.4 (0.3)	2.6 (0.3)	3.0 (0.5)
2000	4.2 (0.4)	2.3 (0.2)	2.7 (0.2)	3.0 (0.4)
Chlorophyll *a* (μg l^{-1})				
1991–1996	1.5 (0.2)	1.3 (0.1)	2.0 (0.3)	2.9 (0.4)
1997	1.3 (0.3)	1.3 (0.3)	1.9 (0.4)	2.5 (0.7)
1998	1.5 (0.3)	1.2 (0.4)	1.8 (0.7)	2.7 (1.5)
1999	1.9 (0.4)	1.6 (0.4)	1.8 (0.5)	2.6 (1.1)
2000	N/A[b]	1.2 (0.4)	2.2 (0.6)	2.8 (1.2)
1% PAR depth (m)				
1994–1996	7.8 (1.4)	17.0 (2.0)	14.8 (2.0)	13.4 (2.2)
1997	8.0 (1.2)	15.4 (2.0)	13.6 (2.4)	13.5 (1.3)
1998	9.2 (1.2)	14.9 (1.8)	14.4 (1.2)	13.4 (1.3)
1999	6.5 (1.6)	12.7 (2.2)	9.8 (1.8)	11.0 (1.3)
2000	9.5[c]	17.4 (1.6)	13.6 (1.1)	14.1 (1.4)

Notes: [a] Mean water column DOC and chlorophyll *a* levels, typically based on bi-weekly samples collected at 0.5, 5.5, 10.5, and 15.5 m depth between mid-May and mid-August in each year.
[b] N/A = not available.
[c] One June measurement.

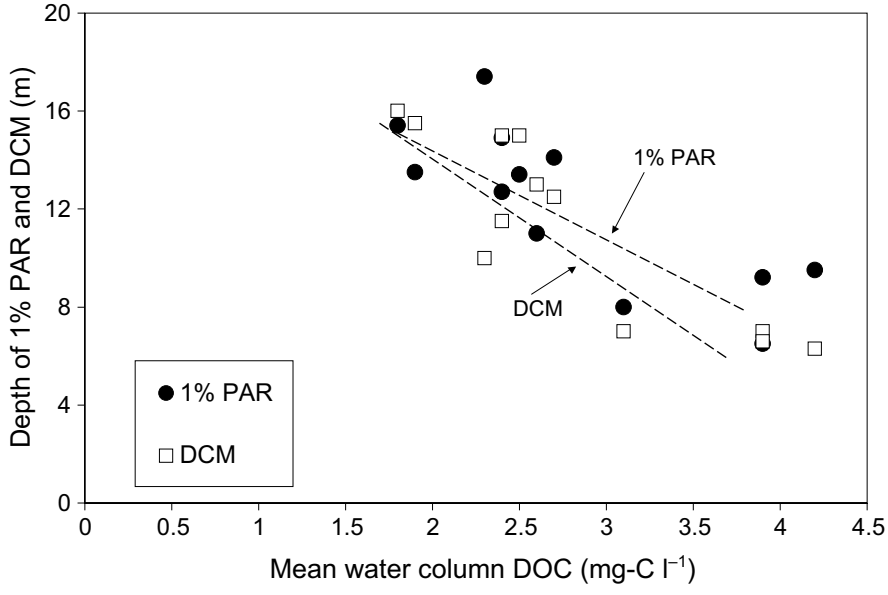

Fig. 5. Relationships between water column DOC level and the depths of 1% PAR (r^2 = 0.64, p < 0.025) and the DCM (r^2 = 0.77, p < 0.025) in L20, L26, and L39 between 1997 and 2000. Trend lines are noted.

quality resulting from major watershed logging activity. However, DCM profile and water chemistry results reported here, and previous physical and chemical data reported by Steedman (2000) indicate that water conditions did not change substantially in any of the study-lakes. Small changes in DCM vertical position were observed, however the changes appeared to be more associated with varying regional weather patterns rather than experimental forestry operations. For example, 1997 was a very dry year in the study area (Steedman and Kushneriuk 2000), which resulted in reduced ambient lake DOC levels in that year. As a consequence, in the year we expected to see the greatest impact of deforestation on water quality (and in the DCM), we primarily saw the effects of the 1997 drought; i.e., dry conditions overwhelmed any affect on the DCM that may have been evident after 1996 logging activity.

The subsequent years of increasing DOC and lower DCM depths probably resulted from both recovery from the dry weather period and a small increase in DOC due to catchment disturbance (Steedman 2000), although this cannot be proved absolutely. Between 1997 and 1999, total N (averaged over the whole water column based four discrete samples per event) increased by about 17%, DOC increased by about 8%, and 1% PAR depths decreased by about 25% in L26, L39 and L42, whereas total P and chlorophyll *a* remained unchanged (Steedman 2000, Steedman and Kushneriuk 2000). These results are corroborated by our DCM observations. The upward repositioning of DCM generally observed in our monitoring program was small and erratic, but consistent with reduced water clarity associated with increased ambient DOC levels. Our DCM data further indicate that no undetected changes, which might have been missed in the previous

Table 4. DCM-peak intensity (RF) and mean peak depth (m), denoted in parentheses, in the four lakes from 1997 to 2000.

Lake	Aug. 1997	28 May 1998	3 July 1998	10 Aug. 1998	5 Sep. 1998	Aug. 1999	Aug. 2000
L20	2.1 (7.0)	1.5 (9.0)	1.0 (13.0)	1.2 (7.0)	0.4 (8.0)	1.3 (6.6)	2.4 (6.3)
L26	1.3 (16.0)	1.7 (19.0)	1.5 (17.5)	0.9 (15.0)	0.5 (16.0)	1.3 (11.5)	0.6 (10.0)
L39	0.6 (15.5)	1.8 (15.0)	1.2 (14.5)	1.4 (15.0)	0.6 (14.0)	1.0 (13.0)	0.4 (12.5)
L42	3.0 (16.3)	2.8 (16.0)	2.2 (15.5)	5.0 (17.0)	2.2 (16.8)	3.9 (16.5)	3.4 (16.5)

depth-specific monitoring program, in vertical water quality were apparent in each lake (Steedman 2000).

Application of *in situ* fluorometry in water quality studies

In general, our *in situ* fluorometry results were consistent with other water quality data in this study. RF signals were strongly correlated with chlorophyll *a*, and provided a compatible picture of water quality changes to that indicated by water chemistry monitoring. *In situ* fluorometry was very useful in latter years of the study because real-time predictions of water quality conditions could be made "in the boat" using DCM data without corroborating chemical analyses. We suggest that this simple technique be considered for water quality monitoring in similar studies in the future.

Conclusions

Previously reported measurements of lake stratification and water quality on the four study-lakes indicated that the lakes were not substantially impacted by experimental logging activities (Steedman and Kushneriuk 2000, Steedman 2000). Although L20 was not an ideal reference lake for this study, all lake monitoring, including water chemistry, microscopic observations of phytoplankton communities, and DCM measurements, indicated that no major differences in water quality existed among the four lakes, even lakes with heavily logged watersheds. Further, our results indicate that DCM measurement is a useful addition to water quality monitoring because it can provide rapid, real-time data, and provides corroborating whole-water column water quality data that are complementary other monitoring methods.

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