A regional paleolimnological assessment of the impact of clear-cutting on lakes from the west coast of Vancouver Island, British Columbia

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Abstract: The impact of forest harvesting on lakes within the temperate rainforest on the west coast of Vancouver Island was examined in a paleolimnological study of four lakes that had 35–92% of their watersheds progressively clear-cut over a period of 15–30 years (impact lakes) and four lakes that had experienced little or no known anthropogenic disturbance in their watersheds (reference lakes). Changes in diatom species composition and percent organic matter in the 210Pb-dated sediment cores were compared over the last 100 years in each of the impact lakes before and after the onset of forest harvesting, which began in 1950, and before and after 1950 in the four reference lakes. Only one impact lake showed significant changes in percent organic matter. Significant changes (p < 0.05) in species composition following forest harvesting were detected in all four impact lakes and in one of the four reference lakes. However, the changes in diatom composition following clear-cutting in the impact lakes were small, with changes in the relative abundance of the most common species being maximally 20%, but more typically 3–10%.

Résumé : L’impact de la coupe de la forêt avoisinante sur les lacs dans la forêt pluvieuse tempérée de la côte occidentale de l’île de Vancouver a pu être déterminé par une étude paléolimnologique de quatre lacs dont 35–92% du bassin versant a été progressivement soumis à la coupe à blanc sur une période de 15–30 ans (lacs affectés) et de quatre autres lacs dont les bassins versants ont subi, à notre connaissance, peu ou pas de perturbations anthropiques (lacs témoins). Nous avons examiné les changements de la composition spécifique des diatomées et du contenu en pour cent de matière organique de carottes de sédiments datés au 210Pb au cours des 100 dernières années, soit dans chacun des lacs affectés avant et après la coupe de la forêt qui a débuté en 1950 et dans les quatre lacs témoins avant et après 1950. Seul un des lacs affectés avait subi un changement significatif du pourcentage de matière organique. Des modifications significatives (p < 0.05) de la composition spécifique des diatomées après la coupe de la forêt ont été décelées dans les quatre lacs affectés et un lac témoin ; cependant, ces changements étaient faibles, avec une variation des densités relatives des espèces les plus communes d’au plus 20%, et plus communément de 3–10%.

Introduction

In British Columbia (B.C.), over 90% of the forests are managed under either Timber Supply Areas or Tree Farm Licenses (TFLs), of which the latter are given to individual companies (Pedersen 1995). The TFLs, which account for 8% of the province, contribute approximately 20% of the annual harvest volume (Pedersen 1995) and are concentrated along the coastal regions of Vancouver Island, home to a large portion of the world’s remaining temperate rainforests. In the 1980s and 1990s, provincial forest managers started to view the forest as more than a supply of wood and began to manage with a concern for wilderness, biodiversity (Kimmins 1995), and impacts on aquatic systems (Hartman and Scrivener 1990). In 1995, the Forest Practices Code of British Columbia was established, to promote the conservation of biological diversity through management of the forests, based on ecological units, watersheds, or groups of watersheds (Fenger 1996).

Management of the forested region within a watershed needs to take into account the meteorological, geological, and biological processes that link the terrestrial and aquatic ecosystems. For instance, clear-cutting has been shown in numerous studies to enhance flow, turbidity, concentration of dissolved ions, and temperature in streams (see review in Keenan and Kimmins 1993). The impact of forest harvesting on lakes has been grossly understudied in comparison with research on streams. Furthermore, most studies on the impact of forestry activities on lakes have typically been concerned with short-term dynamics (e.g., Rask et al. 1998). Even in a recent special issue on watershed perturbations (Carignan and Steedman 2000), the majority of studies are only 3 years in duration and few contain any preharvest limnological data.

Paleolimnological techniques can provide a means of assessing impacts associated with forestry activities over a longer time frame, because the sediment record provides an archive of past changes of many biological, chemical, and physical indicators of environmental change. For example, in
northwestern Ontario, lake sediments were used to assess the impacts of logging, fire, and climate on impact and reference lakes over the past 30 years (Paterson et al. 1998). The purpose of this study was to examine the impact of forest-harvesting activities on lakes within the temperate rainforest on the west coast of Vancouver Island. Changes in the sediment organic matter and diatom assemblages preserved in the lake sediments over the last 100 years were compared within four lakes before and after the onset of forest harvesting (impact lakes) and in four lakes in the same region that had experienced little or no known watershed disturbance (reference lakes). Diatom analysis was the main focus of this study, because they are one of the most widely used and sensitive indicators of past changes within lakes (e.g., Stoermer and Smol 1999). This study was designed to detect changes associated with continuous logging practices that occurred over many decades, and provides an assessment of natural variability in the reference lakes and in the impact lakes prior to clear-cutting. Furthermore, changes in the impact lakes (post logging), in comparison with the reference lakes, can be used to determine if the changes in the impact lakes are predominantly attributable to logging activities or potentially related to more regional changes such as climate. A similar study was carried out in the central interior of British Columbia and is presented elsewhere (Laird and Cumming 2001, this issue).

Materials and methods

Study area, sites, and study design

The lakes examined in this study are located on the west coast of Vancouver Island near the towns of Ucluelet and Bamfield on Barkley Sound (Fig. 1). This region lies within the Coastal Western Hemlock biogeoclimatic zone and is dominated by western hemlock (Tsuga heterophylla (Raf.) Sarg.) and western red cedar (Thuja plicata Donn). The area receives very high levels of precipitation (250–500 cm-year⁻¹), falling primarily as rain from October to March, with increasing annual precipitation as one moves inland (Valentine et al. 1978). The dominant soils are acidic ferro-humic podzols, which are low in base cations (Valentine et al. 1978) and nutrients (Clayoquot Sound Scientific Panel 1995), and the landscape is characterized by gentle to extreme slopes.

The lakes in this study were selected from government databases (e.g., Ministry of Forests files, Ministry of Environment fisheries inventory files, maps, and air photos), based on the following criteria: (i) the presence of both reference and logged lakes within an area of similar precipitation, geology, and vegetation; (ii) the absence of large-scale natural disturbances, such as fire, in the last 100 years; and (iii) an attempt to select impact and reference lakes within a similar range of physical (e.g., lake area, watershed area, depth, simple bathymetry) and chemical (e.g., pH, conductivity, nutrients) characteristics (Table 1). In total, 12 lakes were cored. However, based on the results of the ²¹⁰Pb profiles, detailed logging histories of the basins, and concentration and preservation of diatom assemblages in the cores, only eight lakes were analysed in detail. Owing to the difficulty of finding lakes that had experienced no forestry activities in the past 100 years, our reference lakes are defined as having no logging prior to 1980, since this date incorporates most of the prelogging interval in these watersheds. Samples after the onset of logging in the reference lakes were deleted from any analyses. Clear-cutting began in most of the impact lakes in ca. 1950. One of the originally selected impact lakes that experienced logging at this time had low diatom abundances and was not included in the study. In addition, Black Lake, initially selected as a reference lake, had an earlier onset of logging, with associated road building, than originally realized and was therefore analyzed as an impact lake. Thus, three lakes with cutting primarily between 1950 and 1985 (Maggie, Pachena, and Sugsaw lakes) and one lake with cutting between 1970 and 1990 (Black Lake) were included in the analyses.

Field sampling

The lakes were sampled from a Beaver float plane, owing to either no road access or difficult access by logging roads. Two cores were retrieved from the deep basin of each lake, using a modified K-B gravity corer (Glew (1989) equipped with a 60 cm long core tube (internal diameter ~7.6 cm). Approximately 4 kg of extra weight was attached to the approximately 9 kg corer, to provide deeper penetration of the core tubes into the sediments. The cores ranged in length from 35 to 45 cm and were secured upright in padded buckets during transport by plane.

The cores were sectioned vertically into polyethylene whirlpans within 24–36 h of core retrieval. Based on estimations of sedimentation rates, the assumption was made that background ²¹⁰Pb activity would occur by 20 cm in most systems. Thus, each core was sectioned every 0.25 cm for the top 20 cm, and every 0.5 cm for the rest of the core, with the exception of the core from Maggie Lake, which was sectioned every 0.25 cm for the top 30 cm and then every 0.5 cm. Maggie Lake was sectioned in this manner, because initial core evidence suggested that sedimentation was possibly more rapid in this system. Core samples from each lake were placed in large plastic bags, stored in a cooler on ice, and shipped to Queen’s University, Kingston, Ontario, within 48 h of sectioning, where they were stored at 4°C.

Water samples were collected approximately 0.5 m below the water surface of each lake, stored on ice, and shipped to the Pacific Environmental Science Center (Canadian Federal Government Laboratory) in North Vancouver, B.C., within 24 h of collection. Chemical analyses followed standard procedures outlined by the American Public Health Association (1989). Water samples were analyzed for total phosphorus, total nitrogen, nitrate, nitrite, all major anions and cations, dissolved metals (Al, B, Ba, Cd, Co, Cr, Cu, Fe, Mn, Mo, Ni, Pb, Bd, Z, Sr), dissolved inorganic carbon, dissolved organic carbon, alkalinity, and silica. Total phosphorus and nitrogen were estimated from unfiltered samples; all other analyses were based on water samples filtered in the field through a 0.45-μm cellulose acetate filter into acid-washed bottles. Estimates of chlorophyll a were obtained using standard methods based on a 1000-mL sample that was filtered through a 0.45-μm cellulose acetate filter.

²¹⁰Pb dating, percent organic matter, and diatom preparation

From 18 to 22 subsamples of wet sediment from one core from each lake were weighed in a crucible, oven-dried (24 h at 105°C), and reweighed, to determine the percent water and dry weight of the sediment. Samples were ground to a fine dust in the crucible using a pestle and redried overnight at 105°C. Dried sediment was shipped to MYCORE Ltd., Deep River, Ontario, for ²¹⁰Pb analysis. After the ²¹⁰Pb results were received for the first cores, the second core from four of the study lakes was subsampled for ²¹⁰Pb as above, to determine reproducibility of the ²¹⁰Pb profiles for cores with exponential decay profiles (Black and Pachena lakes), as well as for those with less characteristic ²¹⁰Pb profiles (Maggie and Tofuquats lakes).

²¹⁰Pb activity for each sample was estimated by alpha spectrometry, using a ²⁰⁹Pb tracer of known activity. Unsupported ²¹⁰Pb was calculated by subtracting supported ²¹⁰Pb (the baseline activity determined from bottom samples of the core using the guidelines in Binford (1990)) from the total activity at each level. The sedi-
ment chronology and sedimentation rates were calculated from the estimates of $^{210}$Pb activities and estimates of cumulative dry mass (Binford 1990), using the constant rate of supply (CRS) model (Appleby and Oldfield 1978). Chronology for all cores is based on linear interpolation between dated intervals.

The percent organic matter in each core since ca. 1900 was determined in consecutive 0.25-cm intervals (ca. 2- to 3-year resolution) using standard loss-on-ignition methods (Dean 1974). Briefly, a known quantity of dried sediment was heated to 550°C for 2 h in a NEY® model 2-21350 muffle furnace with digital temperature control. The difference between the original weight of the dried sediment and the weight of sediment remaining after ignition was used to estimate the percent organic matter in each sediment sample.

Slides for diatom analysis were prepared using standard techniques (e.g., Wilson et al. 1996). For each sample, at least 400 diatom valves were enumerated along transects on a Leica DMRB microscope under oil immersion at 1000× magnification, using an objective with a numerical aperture of 1.3. Diatom species identification follows references outlined in Wilson et al. (1996).

Statistical analyses

Analysis of similarities (ANOSIM), a nonparametric multivariate statistical test (Clark and Warwick 1994), was used to test the null hypothesis that there was no difference in diatom species composition before and after the onset of forest harvesting in the impact lakes or before and after 1950 in the reference lakes. The approximate date of onset of logging for three of the four impact lakes was 1950. Because the reference lakes were logged after 1980, intervals after the onset of logging in the reference lakes were excluded from the ANOSIM analyses. For example, in Blue Lake, logging started in 1992, thus the intervals included in the "after" group consisted of intervals between 1951 and 1991. Prior to running ANOSIM for Maggie, Toquart, and Sugaw lakes, three to five samples were deleted, to provide approximately equal temporal resolution between intervals to ensure that results were not biased owing to different resolutions before and after 1950.

ANOSIM uses a rank similarity or dissimilarity matrix to calculate within group and across-group differences (Clark and Warwick 1994). A Bray–Curtis similarity coefficient with no species transformations was used as the similarity coefficient in our analyses. Rare taxa (present at less than 2% relative abundance in two or fewer samples) were grouped into diatom genera or other morphological categories (e.g., rare Navicula, rare centrics, rare araphes).

ANOSIM tests were calculated using the statistical package PRIMER 4.0 (Plymouth Marine Laboratory 1996). The outcome of an ANOSIM comparison is termed an $R$ statistic, the value of which may vary between –1 and 1. An $R$ statistic near 0 indicates that the null hypothesis of no differences between the two groups of samples is likely true (i.e., the within- and between-group differences are similar), whereas an $R$ of 1 indicates that samples are always more similar to each other when compared within each group.
Table 1. Summary of the physical and chemical characteristics of the lakes studied.

<table>
<thead>
<tr>
<th></th>
<th>Reference lakes</th>
<th>Impact lakes</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Angora</td>
<td>Blue</td>
</tr>
<tr>
<td>Lake area (km²)</td>
<td>0.31</td>
<td>0.47</td>
</tr>
<tr>
<td>Watershed area (km²)</td>
<td>2.1</td>
<td>5.3</td>
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<tr>
<td>Watershed cut (km²)</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>Watershed cut (%)</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Watershed/lake area</td>
<td>6.8</td>
<td>11.3</td>
</tr>
<tr>
<td>Maximum depth (m)</td>
<td>46.5</td>
<td>11</td>
</tr>
<tr>
<td>Coring depth (m)</td>
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<td>11</td>
</tr>
<tr>
<td>Volume (10⁶ m³)</td>
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<td>na</td>
</tr>
<tr>
<td>Mean watershed slope</td>
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<td>0.05</td>
</tr>
<tr>
<td>pH</td>
<td>8.05</td>
<td>6.96</td>
</tr>
<tr>
<td>Specific conductivity (μS·cm⁻¹)</td>
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<td>24.3</td>
</tr>
<tr>
<td>Alkalinity (mg·L⁻¹)</td>
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<td>4</td>
</tr>
<tr>
<td>Total phosphorus (µg·L⁻¹)</td>
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<td>5</td>
</tr>
<tr>
<td>Total nitrogen (µg·L⁻¹)</td>
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<td>170</td>
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<tr>
<td>Chlorophyll a (µg·L⁻¹)</td>
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<tr>
<td>Silica (mg·L⁻¹)</td>
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<tr>
<td>Calcium (mg·L⁻¹)</td>
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<td>Sodium (mg·L⁻¹)</td>
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<tr>
<td>Chloride (mg·L⁻¹)</td>
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<tr>
<td>Dissolved inorganic carbon (mg·L⁻¹)</td>
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<td>0.7</td>
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<tr>
<td>Dissolved organic carbon (mg·L⁻¹)</td>
<td>3.6</td>
<td>7.6</td>
</tr>
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</table>

Note: Chemical variables are based on one sample taken in August 1996. For reference lakes, watershed cut is before 1980 and, for logged lakes, is total cut. Mean watershed slope is calculated by \( H/L \), where \( H \) is the difference between the highest and lowest point in the basin and \( L \) is the horizontal distance along the longest dimension of the basin parallel to the principal drainage line (Schumm 1956).

Results

\(^{210} \text{Pb} \) profiles

The \(^{210} \text{Pb} \) profiles from the reference and impact lakes all show decreases in activity with depth and all indicate that the cores were of sufficient length to reach supported levels of \(^{210} \text{Pb} \) activity (Fig. 2). Log of total \(^{210} \text{Pb} \) versus cumulative dry weight was plotted to indicate the values from which supported \(^{210} \text{Pb} \) were calculated (Fig. 2). Plots of log unsupported \(^{210} \text{Pb} \) show similar trends to those presented and, for most cores, are indistinguishable post 1900 A.D. from log of total \(^{210} \text{Pb} \), owing to the small and constant amount of supported \(^{210} \text{Pb} \). The \(^{210} \text{Pb} \) profiles do show that all lakes experienced either some changes in their sedimentation rates or changes in sediment mixing over the time frame represented by these cores (i.e., changes in the slope of log \(^{210} \text{Pb} \) versus cumulative dry weight in the region of unsupported \(^{210} \text{Pb} \) activity). During the period of interest in this study (ca. 1900 to present), changes in sedimentation and (or) mixing regimes were minimal in two of the four reference lakes (e.g., Figs. 2a and 2c). In the other two reference lakes (Figs. 2b and 2d), the \(^{210} \text{Pb} \) profiles suggest that there were some changes in the sedimentation rates or sedi-
Fig. 2. Log of total $^{210}$Pb activity versus cumulative dry weight for reference ($a$-$d$) and impact (logged, $e$-$h$) lakes: reference lakes: ($a$) Angora, ($b$) Blue, ($c$) Little Toquart, and ($d$) Toquart; logged lakes: ($e$) Black, ($f$) Maggie, ($g$) Pachena, and ($h$) Sugsaw. Core analyses were completed for profiles shown using solid squares. In four lakes, the second core retrieved from the lake was also dated (open circles). Approximate dates, estimated from the constant rate of supply (CRS) model, are given for points of change in slope and the beginning date of focus of this study (1890–1900). Supported $^{210}$Pb is indicated as samples below the dotted horizontal line.

![Graphs showing log of total $^{210}$Pb activity versus cumulative dry weight for reference and impact lakes.]

For example, in Blue Lake (Fig. 2b), the decrease in the slope of activity versus cumulative dry weight between ~0 and 0.5 g·cm$^{-2}$ could be interpreted as a decrease in sedimentation rate or as a decrease in the average rate of sediment mixing. Similarly, the region of relatively constant activity between 1 and 2.5 g·cm$^{-2}$ in Toquart Lake (Fig. 2d) could be interpreted as a region of increased sediment deposition or of increased mixing.

Three of the four impact lakes show little evidence of changes in sedimentation rate or mixing regime during the time frame of interest of this study (Figs. 2e, 2g, and 2h). This is particularly true for Black Lake (Fig. 2e), whereas Pachena Lake (Fig. 2g) and Sugsaw Lake (Fig. 2h) indicate small changes in slope around 1916 and 1926, respectively, which may indicate either a decrease in sedimentation rate or a decrease in the average rate of mixing. The fourth lake shows major departures from an exponential decay curve (Fig. 2f), suggesting that large changes in sedimentation and (or) mixing occurred in this lake. The CRS model dates the large decline in $^{210}$Pb activity in Maggie Lake at between ca. 1963 and 1967 (Fig. 2f). This portion of the sediment core was associated with large amounts of organic debris, including wood and needles. A smaller event appears to have occurred towards the top of this core (Fig. 2f).

Reproducibility of the $^{210}$Pb profiles from the replicate cores in the deep basin was excellent for all profiles (Figs. 2d–2g). Even though some $^{210}$Pb profiles suggest that either changes in sedimentation rate or mixing occurred, this does not hamper our conclusions, because this study was designed to detect changes associated with continuous logging practices occurring over many decades, and our conclusions are drawn from all lakes, which show similar trends in spe-
cies changes (see below). Overall, the $^{210}$Pb results obtained from the reference and impact lakes suggest that reliable chronologies can be estimated from these cores.

Organic matter changes and resampling results

Only one logged lake (Pachena) showed a significant difference in percent organic matter before and after the onset of clear-cutting (Table 2). Coincident with the onset of logging in the watershed of Pachena Lake, there was a gradual decrease in percent organic matter from approximately 45% to approximately 30% by 1990 (Fig. 3g). After 1990, organic matter began to increase.

Although Maggie Lake did not show a significant difference in organic matter before and after the onset of logging, a distinct but short-lived peak in organic matter was found that corresponded approximately to the onset of logging (Fig. 3f). This distinct peak in organic matter is composed of a 5- to 6-cm layer of woody debris (chunks of wood, twigs, needles) that is estimated (from the two $^{210}$Pb-dated cores) to have been deposited within a short time span (1–4 years) and is associated with large decreases in $^{210}$Pb activity between 1963 and 1967 (Fig. 2f). In 1962, the same time as the onset of logging and road building, Hurricane Freda hit the region, bringing in torrents of rain that may have started debris slides on an already disturbed landscape. Concurrent with the logging and Hurricane Freda, an open pit iron mine upstream from Maggie Lake operated from 1962 to 1968 (Spicer 1999). Consequently, it is impossible to untangle the specific contributions of these various disturbances to the observed changes in sedimentation and organic matter. A later peak in organic matter in the 1990s at Maggie Lake may, in part, be associated with a large January storm in 1992, during which nearly 75 cm of rain fell in the Barkley Sound area (Spicer 1999). The increased flow of water on a disturbed landscape may have resulted in debris slides, as described earlier. Disregarding these peaks in organic matter, there is a general decrease in organic matter from approximately 20% prior to logging to 11–15% from 1970 to 1985 that may be associated with increased erosion of inorganic mineral inputs from the watershed.

In Toquart Lake (reference lake), a large peak in organic matter centered around 1952 was found, with another smaller peak in the 1960s (Fig. 3d). No known anthropogenic disturbances occurred in this watershed during this time interval (John Deniseger, Ministry of the Environment, Nanaimo, B.C., personal communication), thus the cause of the increase in organic matter is unknown. Estimated dates suggest that the organic matter increased over a fairly long time span and, thus, it seems unlikely that the peak is associated with an erosional event. Furthermore, there is no evidence of terrestrial debris in the core and no decline in $^{210}$Pb activity associated with the peak in organic matter, as was observed in Maggie Lake. There was a decline in $^{210}$Pb activity between 5 and 6 cm (estimated date of 1967–1970); however, this decline corresponds to the very end of the peak in organic matter. In addition, the increase in organic matter corresponds to increases in Aulacoseira spp. and araphid diatom taxa between 1950 and 1970, indicating the significance of this event in the history of this lake (Fig. 4d); however, the cause of this change is uncertain.

Diatom species changes and ANOSIM results

The diatom flora was very diverse in all the lakes (Figs. 4 and 5), with the exception of Angora Lake (Fig. 4a), which had only 20 taxa. All other lakes had more than 100 diatom taxa, with up to 170–190 taxa in Blue (Fig. 4b) and Maggie (Fig. 5b) lakes. Dominant taxa, which together typically comprise greater than 50% of the diatom assemblages, include the planktonic centric species Cyclotella stelligera (Cleve and Grunow in Cleve) Van Heurck, Cyclotella pseudostelligera Hustedt, Cyclotella glomerata Bachmann, Aulacoseira distans (Ehrenberg) Kützing, and Aulacoseira distans var. tenella (Nygaard) Florin. Blue and Little Toquart lakes (Figs. 4b and 4c), the two shallowest lakes, are exceptions, with the most abundant taxa being the benthic taxa Brachysira brebissonii Ross and Frustulia rhomboides (Ehrenberg) De Toni; these comprise approximately 20 and 10–15% of the assemblages, respectively. The periphytic taxon Achnanthes minutissima Kützing was abundant in Toquart (Fig. 4d) and Maggie (Fig. 5b) lakes, typically comprising 25–30% of the assemblage in Toquart Lake and 10–20% in Maggie Lake. The genera Navicula Bory and Eunotia Ehrenberg were extremely diverse in the study lakes, with the mean number of taxa across lakes being 26 and 29, respectively, but any specific taxon was rare. Other diverse genera across all lakes include Achnanthes Bory and Cymbella Agardh, whereas the genus Gomphonema Agardh was only diverse and of relative importance in Maggie Lake.

Sediment intervals in each lake were divided a priori into two groups according to the $^{210}$Pb estimated dates: before and after forest harvesting for the impact lakes and between 1950 for the reference lakes. The difference in species assemblages between the two groups within each lake was tested using ANOSIM. The null hypothesis of no difference between diatom assemblages in the two groups could not be accepted in five of the eight lakes (Table 3). All the impact lakes (Black, Maggie, Pachena, and Sugsaw) and one reference lake (Little Toquart) showed a significant difference in species assemblage between the two groups. However, in only three of the lakes, two impact lakes (Black and Pachena) and one reference lake (Little Toquart), were these differences associated with a high R statistic (Table 3). Although the variations in diatom species abundances in the lakes that showed significant differences between the two time periods are often quite small, with changes in the relative abundance of the most common species being maximally 20% (but more typically 3–10% (Figs. 4 and 5)), the variations in diatom community composition are, as expected, more distinct in those lakes that had a high R value. In Pachena Lake (Fig. 5c), there was a distinct increase in the abundance of C. pseudostelligera and C. glomerata, whereas taxa such as A. distans and A. distans var. tenella decreased in abundance; however there appears to be a lag in the response of A. distans with respect to the onset of forest-harvesting activities. In Black Lake (Fig. 5a), the change in the two taxa that contributed the most to the average dissimilarity between the before and after groups (as determined using SIMPER), A. distans var. tenella and C. pseudostelligera, is not strongly associate with the timing of the onset of logging, as in Pachena Lake. However, other taxa with relatively high contributions, such as C. stelligera and A. distans, increase
with the onset of logging, whereas many of the littoral genera (e.g., *Eunotia*, *Achnanthes*, *Cymbella*, and *Brachysira*) decrease. In Maggie and Sugsaw lakes (Fig. 5b and 5d), the significant changes in diatom species composition after logging are the result of minor differences in the mean abundance of many taxa. In Maggie Lake, there were increases in littoral
Fig. 4. Diatom taxa with frequencies greater than 2% in at least two samples in the reference lakes, Angora (a), Blue (b), Little Toquart (c), and Toquart (d). Taxa are ordered according to their contribution to the average dissimilarity (SIMPER score) between the two groups (before and after 1950, excluding samples above the dotted line, which denotes the onset of logging). Consequently, taxa showing the largest changes before and after 1950 are arranged from left to right. Note the different percentage scales for the various taxa.
Fig. 5. Diatom taxa with frequencies greater than 2% in at least two samples in the impact lakes, Black (a), Maggie (b), Pachena (c), and Sugsaw (d). Taxa are ordered according to their contribution to the average dissimilarity (SIMPER score) between the two groups (before and after the onset of logging, denoted by the dotted line). Consequently, taxa showing the largest changes before and after logging are arranged from left to right. Note the different percentage scales for the various taxa.
taxa, such as *Achnanthes pusilla* Grunow in Cleve and Grunow, *Brachysira neoexilis* Lange-Bertalot, *F. rhomboides*, *Navicula* spp., and *Cymbella* spp. (Fig. 5b), associated with the peak in organic matter in the early 1960s (Fig. 3f). This may be a reflection of the transport of more littoral sediment, associated with an erosional event, reaching the center of the lake (see Discussion).

In the one reference lake (Little Toquart) that showed significant changes in diatom species composition, there was an increase in the abundance of *A. distans* var. *tenella* and a decrease in *F. rhomboides* after 1950 (excluding the two samples after the onset of logging in ~1980), with changes in other taxa being quite subtle (Fig. 4c). Although not included in the analyses presented here, after the onset of logging in Little Toquart Lake, there was a distinct increase in *Tabellaria flocculosa* (Roth) Kützing and *A. minutissima*. The other three reference lakes did not show a significant difference in species assemblages before and after 1950. However, in Toquart Lake (Fig. 4d), there was a distinct increase in *Aulacoseira* spp. from approximately 1950 to 1970 that corresponds to the large but unexplained peak in organic matter content (Fig. 3d).

**Ordination results of the physical characteristics of the lakes**

Exploratory PCA analysis of physical characteristics of the lakes did not produce any evidence for differences between the impact and reference lakes, based on their physical characteristics, that could explain the significant diatom changes seen primarily in the impact lakes. The exception was Angora Lake (reference lake), which has much higher slopes within its catchment than any other lake (Table 1) and thus separated out based on this characteristic; however, it did not show any significant changes in diatom species composition. PCA analysis of lake-water chemical variables was also explored, but no unique differences between the reference and impact lakes were found. However, since the chemical data were collected only after clear-cutting began in all lakes, and because the time since the last cut is grossly different between lakes, it is impossible to evaluate the importance or cause of any chemical differences between the lakes with respect to forestry activities.

CVA analysis of two groups of lakes (those with a significant response of diatoms and a high R statistic (two impact lakes: Black and Pachena; one reference lake: Little Toquart) and those with a low R statistic) could not explain the different diatom responses based on any of the physical characteristics of the lakes and their associated watersheds. This may, in part, be due to the small number of lakes examined, the limited number of physical variables included in the analysis, and important variables, such as volume of the lake (unknown for two of the eight lakes) and residence time (unknown for all lakes), not being included.

**Discussion**

The before–after design of this study in both impact and reference lakes from the same biogeoclimatic setting is a means of evaluating changes due to forestry activities, while at the same time attempting to assess if changes may be occurring at the local–regional scale in the absence of forest-harvesting activities. A similar study in Ontario by Paterson et al. (1998) illustrated the importance of this type of design. These authors observed changes in chrysophyte assemblages in both impact and reference lakes, and concluded that the observed changes may be the result of a regional drought in the area and thus may not be due solely to forestry activities or wildfire. In our study, the degree of change in the diatom assemblages was, on average, greater in the impact lakes than in the reference lakes, with the exception of one reference lake. This suggests that forest harvesting has had an impact on lakes that is larger than that seen in most reference lakes, but that natural variability can be as large as the changes seen before and after the onset of logging.

The magnitude and frequency of natural disturbances such as fire and windthrow in a forested region are driven principally by climate and physiography. For example, on the drier east coast of Vancouver Island, fires occur more frequently than on the west coast, where climate is wetter and windthrow and debris slides are the dominant natural disturbances (Clayoquot Sound Scientific Panel 1995). Short-term chemical responses to both windthrow and fire have been recorded in the streams of the Experimental Lakes Area in Ontario (Schindler et al. 1980; Bayley et al. 1992). Schindler et al. (1980) found that the effects on stream runoff and nutrient yields from windthrow disturbance fell within the same range as those from fire, with conditions returning to pre-disturbance levels within 3 years. Although Bayley et al. (1992) found elevated levels of nitrogen (N) and phosphorus (P) in the streams for several years following two fires 6 years apart, the total N and P inputs into Lake 239 were only slightly elevated.

Longer-term studies of natural disturbance on lake ecosystems are rare, but paleolimnological techniques have been used to detect natural forest disturbance from fires and windthrow. For example, Rhodes and Davis (1995) investigated the impact of watershed disturbance on a small oligotrophic pond in Maine over the last 3500 years by examining changes in geochemistry and organic matter (decreases indicating brief pulses of soil erosion), as well as changes in pH.
inferred from changes in the species composition of diatoms. Increases in pH following disturbance were thought to be indicative of increased inputs of base cations from wood ash and mineral soils (Rhodes and Davis 1995). Similarly, the analysis of diatoms in the lake sediments of a small oligotrophic lake in Finland over the past 100 years showed a diatom-inferred increase in pH associated with a fire in 1890; however, the influence of climate on the observed changes could not be conclusively dismissed (Korhola et al. 1996). The apparent sensitivity of these lakes to pH changes associated with fire disturbance may, in part, be due to their poor buffering capacities and topographic location (both are headwater lakes), as opposed to differences in the ratio of watershed area to lake surface area (WA/LA) that differ by an order of magnitude (~50:1 for the Maine pond and ~3:1 for the Finland lake).

The WA/LA and catchment slope were proposed as characteristics that determined the magnitude of lake eutrophication in response to the hemlock decline (Hall and Smol 1993). Hall and Smol (1993) suggest that watersheds with steeper slopes may supply a greater proportion of mineral material to a lake basin than those with gentler slopes. The one impact lake (Pachena) that showed a significant decrease in organic matter after the onset of logging also had the steepest terrain among the impact lakes. The steepness of the terrain, along with a high density of roads (Spicer 1999), likely contributed to the higher proportion of mineral erosion in this basin. However, WA/LA and catchment slope do not appear to explain the significant changes seen in diatom assemblages, even in those lakes with the greatest changes (i.e., impact lakes Black and Pachena and reference lake Little Toquart). Little Toquart Lake is one of the shallowest of the study lakes, which may make it more sensitive to changes. However, Blue Lake (reference lake) is even shallower and did not show any significant change in diatom species composition. The organic-matter profile of Little Toquart Lake does not indicate any natural disturbance from windthrow or debris slides, and fire has not occurred in the basin over this time period. In the absence of such changes, the variability in diatom species may be related to in-lake processes (e.g., population dynamics), although the exact cause remains elusive.

Carignan et al. (2000) proposed an intuitively simple model to explain the degree of impact on water quality from forest harvesting on lakes that is based on the area of watershed cut and lake area. According to this model, the greater the ratio of area harvested divided by either the lake’s volume or its surface area, the greater the impact. Applying this model to our study lakes, the greatest degree of impact should have been seen in Maggie and Sugsaw lakes, whereas, in fact, these are the two lakes with the lowest R statistics and therefore the least impacted. The Carignan et al. (2000) model may not be applicable to our study region, for several reasons: (i) different physiographic and edaphic conditions; (ii) different time frames, since their model is based on 1 year of cut compared with several decades of cutting in this study; or (iii) uncertainties in their model related to deriving preharvest conditions on the water chemistry of the logged lakes from a set of reference lakes.

Although all four impact lakes showed a statistically significant change in diatom assemblage before and after the onset of logging, these changes were relatively subtle even in those lakes with the greatest changes (i.e., a high R statistic). The difference in diatom assemblages before and after forest harvesting typically resulted from small shifts in the percentages of taxa rather than from the appearance or disappearance of taxa. The small stratigraphic changes in diatom taxa reported here are in contrast with the large changes in species assemblages observed in studies of lake acidification (e.g., Cumming et al. 1994), eutrophication (e.g., reviewed in Hall and Smol 1999), and climate change (e.g., Laird et al. 1996). The lack of changes of greater magnitude could be due to several reasons, including the short-term changes, typically in the order of a few years, seen in water chemistry from monitoring studies following forest harvesting, (e.g., Likens et al. 1978; Feller and Kimmins 1984; Hornbeck et al. 1986), and a limited export of nutrients, particularly P.

Podzols, the dominant soil type on the west coast of Vancouver Island, binds P tightly, owing to the low soil pH, and thus release to streams is minimal (Clayoquot Sound Scientific Panel 1995). In general, P is less susceptible to leaching than N (Keenan and Kimmins 1993). Even after whole-tree harvest in a small watershed at the Hubbard Brook Experimental Forest in New Hampshire, negligible P was exported to streams while other nutrients increased dramatically (Yanai 1998). It is well known that P is the primary limiting nutrient in freshwater lakes, and increased loading of P from forest harvesting could result in eutrophication (Likens et al. 1978), with eutrophication resulting in changes to the diatom assemblage (e.g., see review by Hall and Smol 1999). However, clear-cut logging and burning in Carnation Creek, located on the west coast of Vancouver Island near our Bamfield study sites, had only a small impact on periphyton (composed primarily of diatoms) production (Hartman and Scrivener 1990). Short-term responses were seen at some sites during the second year of post logging that were potentially associated with increased P. In areas with richer soils, land clearance associated with logging and later settlement can result in eutrophication and distinct diatom changes, as seen in the study by Fritz et al. (1993).

Although P is typically the limiting nutrient in freshwater lakes, other chemical factors can greatly affect diatoms and other algae in the lakes. Diatom-inferred pH increases have been linked to fire (e.g., Rhodes and Davis 1995) and land clearance associated with agriculture (Renberg et al. 1993), as well as to logging, in some studies (e.g., Davis et al. 1994). All the lakes in these studies were acidic lakes (pH < 6.0), typically with shallow soils in the watershed and, thus, probably with low buffering capacities. The lakes in our study region are generally circumneutral (pH = 6.5–7.8) and, thus, have greater buffering capacities than those in the studies above, which may, in part, account for the minor diatom changes seen in this region. At this time, we cannot infer whether the diatom changes seen in the logged lakes are due to changes in chemical variables such as pH, P, or conductivity. Although an extensive calibration set of chemical variables and diatom taxa exists for over 200 lakes in the interior of British Columbia (Wilson et al. 1996), many of the diatom taxa seen in the Vancouver Island lakes are either not present or are of low occurrence in this dataset, owing to the present paucity of low conductivity oligotrophic lakes in the

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calibration set. Regardless, the relatively minor changes in diatom taxa suggest that diatom-inferred changes in water chemistry associated with logging would be minimal.

Another reason for the small-magnitude changes seen in the diatom assemblages may be related to the rate of revegetation. The rate of recovery of chemical and streamflow changes is strongly linked to the rate of revegetation in the watershed (Marks and Bormann 1972; Likens et al. 1978). Given the high precipitation regime in our study region, and assuming minimal soil disturbance, it is likely that the rate of revegetation would be quite rapid, which would help to account for the lack of response in many lakes. In addition, in any particular year, the percentage of the watershed cut was usually less than 10% and not more than 20%, which may have reduced the impact.

In summary, the minor changes, albeit statistically significant, seen in the diatom assemblages after substantial logging in the watersheds may be due to several reasons, including: (i) our average sampling resolution of 5–10 years may have been too long to detect short-term changes in water chemistry; (ii) the soils in the watersheds of our lakes generally contained low levels of P; (iii) our study lakes were generally well buffered and thus not susceptible to large changes in pH associated with land disturbance; and (iv) rates of revegetation were high and only a low percentage of watershed was cut in any particular year. The significant changes in diatom species composition seen in one reference lake suggest that natural variability can be as great as the variability seen in the impact lakes, and implies that some of the changes seen in the impact lakes may, in part, be due to natural variation. However, the other three reference lakes did not show any significant changes, suggesting that a regional signal is not complicating the impacts from forest harvesting, but rather that there is a local signal affecting the one reference lake.

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