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Climatic changes in northwestern Ontario have had a greater effect on erosion and sediment accumulation than logging and fire: Evidence from ^{210}Pb chronology in lake sediments

JULES M. BLAIS[1,*], ROBERT L. FRANCE[2], LYNDA E. KIMPE[3] & R. JACK CORNETT[4]
[1]*Department of Biology, McGill University, 1205 Ave. Docteur Penfield, Montreal, Quebec, Canada, H3A 1B1;* [2]*Graduate School of Design, Harvard University, 48 Quincy Rd., Cambridge, MA, U.S.A., 02138;* [3]*Public Health Sciences, University of Alberta, Edmonton, Alberta, Canada, T6G 2G3;* [4]*Environmental Research, Chalk River Nuclear Labs, Chalk River, Ontario, Canada, K0J 1J0 (*Corresponding author: Department of Biological Sciences, University of Alberta, Edmonton, Alberta, Canada, T6G 2E9)*

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Abstract. Sedimentation patterns in nine lake basins were examined where catchments were either clearcut, burned in recent history, or where there has been no recorded disturbance and the catchments consist of mature forests. Pronounced declines in sedimentation rates were observed in eight of eleven cores after 1980, in reference, clearcut, and burn lakes. The degree of change was positively correlated with the drainage ratio (catchment area: lake area), but was unrelated to land use history. The decline in sedimentation rates after 1980 coincide with a 60% decrease in catchment runoff and precipitation measured over the same time interval at the nearby Experimental Lakes Area. These results indicate that climatic changes over the past 20 years have had a greater effect on catchment erosion than either clearcutting or fire.

Introduction

Recently, interest has developed in determining how catchments respond to varying land-use practices. Most limnological studies of catchment response to disturbance have been based on calculations of stream discharge of suspended and dissolved solids. Such detailed analyses have timescale limitations in that they typically do not extend beyond the duration of a few years. Lake sediment cores, on the other hand, allow us to extend that timescale to study changes in catchment response over the course of decades. Moreover, annual sediment yields from any river can fluctuate by as much as five-fold from one year to the next (Douglas 1969), thus emphasizing the necessity for long-term records.

Lakes are sinks for eroded particles from catchments. Lake sediment cores enable reconstruction of the erosional history of the surrounding landscape. Sediments can be dated using ^{210}Pb radiochronology, which provides accurate measurements of both chronologies in cores, as well as measurements of sedimentation rates throughout the stratigraphy (Oldfield & Appleby 1984). These chronologies of lake sediments provide a good measure of allochthonous inputs (Dearing 1991a). Sediment records have previously shown that sediment yields from denuded catchments exceeded their predisturbance levels by over an order of magnitude (Flower et al. 1989; Dearing 1991b).

One factor that influences a catchment's response to recent forest cutting is climate. Boreal lakes have undergone large hydrological changes over the past 25 years at the Experimental Lakes Area in northwestern Ontario (Schindler et al. 1996), which is situated fairly close (150 km) to our study sites. Between 1970 and 1990, the mean annual temperature rose by 1.6 °C, precipitation declined from 800 mm yr^{-1} to 550 mm yr^{-1}, catchment runoff declined from 400 mm yr^{-1} to $<$150 mm yr^{-1}, and many streams dried up during the ice-free season (Schindler et al. 1996). This resulted in a myriad of effects including reduced stream exports of particulate and dissolved constituents (Schindler et al. 1996). These studies show that changes in stream chemistry after fires were small compared to the long-term change in chemical yields caused by climate warming and drought over the past 20 years (Schindler et al. 1980; Bayley et al. 1992).

The purpose of this study was to use paleolimnological techniques to assess catchment response to forest harvesting and forest fires that have taken place over the last 20 years. We examined ^{210}Pb chronologies and trace metal stratigraphies in nine basins exposed to a variety of catchment disturbances.

Materials and methods

Study area

This study was undertaken in northwestern Ontario in a region transitional between the boreal and Great Lakes/St. Lawrence forest types. Dominant tree species in this area include jack pine (*Pinus banksiana*), black spruce (*Picea mariana*), white birch (*Betula papyrifera*) and trembling aspen (*Populus tremuloides*). The landscape is composed of gently rolling knolls of Precambrian bedrock interspersed with boggy plains and glacial deposits. Coarse-textured podsols and organic soils are thin and widespread in the region (Zoltai 1961; Brunskill & Schindler 1971; Zoladeski & Maycock 1990).

Nine lakes of different watershed disturbance were selected for coring. Two reference lakes, Lake 42 (L42) and L26, were situated within watersheds free of recent forest disturbance (France & Peters 1995; France 1997a), although the charcoal remains of old growth trees suggest that a major wildfire swept through the area over a century ago. These lakes are part of the Coldwater Lakes Study Project (Rusak 1991) and are located 150 km southeast of the Experimental Lakes Area (ELA). Lake 442 is within ELA and has experienced moderate watershed disturbance involving clearcutting from 1975 to 1979 with the exception of a narrow buffer strip. Lakes M-6 and M-4 are located 120 km east of ELA and had their watersheds completely logged in 1983 (France 1997b). This is a region of extensive industrial clearcutting (France et al. 1996). Lakes M-6 and M-4 have distinct double basins with logging roads located in one sub-catchment of each. Both sub-basins of these two lakes were cored for this study. Percy and Little Joe lakes are located along the Trans-Canada Highway just north of ELA and have been exposed to pipeline, rail line and highway related disturbances during the first half of the Twentieth Century. Recently, the drainage basins around both lakes were completely denuded during a massive forest fire in 1980 (Schindler et al. 1986; France 1997b).

Sediment collection and analyses

Short sediment cores were collected from the ice covered lakes in March 1995 using a Glew corer (Glew 1991) and extruded on site at fine intervals using a Glew close interval extruder (Glew 1988). Duplicate cores were collected at all coring sites. Sediments were sectioned at 0.5 cm intervals for the top 5 cms, at 1 cm intervals for the next 5 cms, 2 cm intervals for the next 10 cms, and 5 cm intervals thereafter. Sediments were then sealed in plastic Whirlpak bags and brought back to the laboratory, where they were weighed and stored in a coldroom. Subsamples were placed in preweighed vials and oven dried at 60–80 °C for porosity measurements. This dried sediment was subsequently divided for analyses of ^{210}Pb, trace metals, and loss on ignition.

^{210}Pb analysis

^{210}Pb activity was determined by measuring the activity of the ^{210}Po granddaughter with alpha spectroscopy. Dried sediments were first spiked with a ^{209}Po yield tracer, and then digested in 1:1 v/v HCl:HNO_3 and 5–7 drops of H_2O_2. The mixture was then placed in a teflon bomb and microwaved. The digestate was evaporated to dryness on a hot plate and then brought into solution with 20 ml of HCl (1.5 N). This solution was then filtered, and to the filtrate, 10 ml of an ascorbic acid solution was added to remove interference between pollonium and dissolved iron. The $^{209,210}Po$ solution was then plated on silver disks and counted using alpha spectroscopy. Duplicates of both blanks and reference materials were analyzed with each core. Excess ^{210}Pb was assessed by subtracting the ^{210}Pb activity at depth from the total ^{210}Pb. Reported values of ^{210}Pb activity were decay corrected to the time of core extraction from the lake. Three months elapsed between sediment extraction and ^{210}Pb counting, and it is assumed that secular equilibrium between ^{210}Pb and ^{210}Po was reached.

Trace metals

Trace metal analysis (Pb) was performed by digesting dried sediments in dilute aqua regia ($3H_2O$:3HCl:HNO_3), then reading absorbance spectra on flame atomic absorption (Perkin Elmer 3100). Extraction efficiencies were assessed from a reference standard (NBS #1572, Buffalo River Sediment Standard).

^{210}*Pb model calculations*

Four different models were considered to interpret ^{210}Pb geochronologies in sediment cores (Oldfield & Appleby 1984). The CF:CS or constant flux:constant sedimentation model assumes that both ^{210}Pb flux to sediments and sedimentation rates were constant over time. This is the most restrictive of all ^{210}Pb models, and is the most likely to be in error, especially where changes in sedimentation rates are expected. This model states that the excess ^{210}Pb concentration in sediments (C) varies exponentially with the cumulative dry mass (m) in the sediment core according to the following equation (Robbins 1978):

$$C = C_o\, e^{-km/r} \tag{1}$$

where C_o is surface ^{210}Pb concentration, in Bq g^{-1}, r is a constant sedimentation rate in g cm^{-2} yr^{-1}, and k is the ^{210}Pb decay constant (0.03114 yr^{-1}). This model is applicable only when there is a monotonic decline in ln ^{210}Pb activity with cumulative dry mass (i.e. where there are no inflections in the ln ^{210}Pb – cumulative dry mass curve).

The next model is used to address the issue of sediment mixing at the surface. The rapid steady-state mixing model (RSSM) assumes instantaneous mixing throughout the surface zone of fixed thickness (S, in g cm^{-2}). Thus, C_x in a core with surface mixing is defined as:

$$C_x = C_s\, e^{-k(m-S)/r} \tag{2}$$

This is simply a modification of the CF:CS model which includes a mixing parameter.

The CIC model, or constant initial concentration model, assumes that changes in sedimentation rate are possible, and will result in proportional changes in ^{210}Pb flux to the sediments. This model is mainly applicable when the primary delivery pathway of ^{210}Pb to sediments is by the erosive influx of ^{210}Pb from the lake's catchment, thereby satisfying the assumption that increased sedimentation results in a proportional increase of ^{210}Pb flux. This model likewise assumes a monotonic decline in ln ^{210}Pb activity with cumulative dry mass. The time of deposition of any sediment interval x (T_x) in the core is then defined as:

$$T_x = k^{-1} \ln(C_o/C_x) \tag{3}$$

The CRS model, or constant rate of supply model, assumes that ^{210}Pb is delivered to sediment surfaces at a constant rate. Thus an increase in sedimentation rate would result in ^{210}Pb dilution in the core profile. Therefore, non-monotonic declines would be expected in the ln ^{210}Pb activity – cumulative dry mass curves. A change in slope towards higher values would indicate an increase in sedimentation rate and vise versa. This model is therefore based on ^{210}Pb inventories rather than concentrations. T_x is thus defined as:

$$T_x = k^{-1} \ln(A_o/A_x) \tag{4}$$

where A_o is the excess ^{210}Pb inventory of the entire core, in Bq cm^{-2}, and A_x is the excess ^{210}Pb inventory below sediment interval x. According to this model, we expect ^{210}Pb inventories within a basin, or among basins of a given geographical area, to be independent of sedimentation rate. This is important because the CRS model assumes that ^{210}Pb flux is independent of sediment flux. This model is primarily applicable where the main delivery pathway of ^{210}Pb to lakes is by direct deposition on the lake surface, for which there is abundant support in lakes on the Canadian Shield in Ontario for both ^{210}Pb and stable Pb (Dillon & Evans 1982; Schut et al. 1986; Blais & Kalff 1993). Instantaneous sediment accumulation rates, r (g cm^{-2} yr^{-1}), can be calculated according to the following equation:

$$r = k\, A_x/C \tag{5}$$

Sediment focusing factors were estimated by comparing the ^{210}Pb inventories in sediment cores with measured atmospheric deposition rates of ^{210}Pb assessed in the region of ELA from soil cores (Omelchenko, A. et al.

unpublished report). Sediment inventories (I) in Bq cm^{-2} yr^{-1} of ^{210}Pb were calculated as follows:

$$I = k\ \Sigma\ m_x C_x \quad (6)$$

where m_x and C_x are the dry mass (g cm^{-2}) and concentration of ^{210}Pb at sediment interval x, respectively. In addition, sediment inventories of anthropogenic Pb were measured in each core and compared with published atmospheric fluxes. Anthropogenic Pb (Pb_A), in μg cm^{-2} yr^{-1}, was assessed in the cores as follows:

$$Pb_A = \Sigma(Pb_x - Pb_B)m_x \quad (7)$$

where Pb_x is Pb concentration at sediment interval x, and Pb_B is Pb concentration of background determined at depth.

Sedimentation models

Models were applied to relate hydrology with expected sediment yields from the catchments. The following equation by Fleming (1969) relates annual sediment load with mean annual discharge for catchments of mixed broadleaf and coniferous vegetation:

$$\mathrm{SEDLOAD} = 116.9[\mathrm{DISCHARGE}]^{1.0207} \quad (8)$$

where sedload is expressed in tons per year and discharge is in cubic feet per second. Mean annual river discharge was approximated as the product of the runoff values measured at ELA between 1970 and 1990 in Schindler et al. (1996), and the drainage basin area. The sediment retention coefficient (SRC) is defined as follows (from O'Connor 1988):

$$\mathrm{SRC} = (\mathrm{WRES\ SV}/h_{mean})/(1 + (\mathrm{WRES\ SV}/h_{mean}) \quad (9)$$

where WRES is water residence time (days), SV is mean settling velocity (m/d) and h_{mean} is the lake's mean depth (m).

Using historical lead flux to confirm sedimentation models

Historical lead flux from anthropogenic emissions have been compiled in Edgington and Robbins (1978) and Rosman et al. (1993). These data were applied to calculate expected Pb profiles in our cores and then compared with the measured Pb profiles in order to confirm our sedimentation model results. Expected Pb concentration (Pb_{pred}) for year i is calculated as follows:

$$(Pb_{pred})_i = \Phi_i\ F/r_i \quad (10)$$

where Φ_i is Pb flux in that year, F is a focus factor, and r_i is the sedimentation rate for that year. This simple model assumes that Pb flux to the sediments is equal to the Pb flux to the surface of the lake, catchment contributions of Pb are small, and outflow losses are likewise small. The latter two assumptions are supported by mass balance studies (Schut et al. 1986). The first assumption is supported by the fact that Pb settling velocities in the water column have been estimated to be approximately 1 m day^{-1}, thus the residence times in the water column are in the order of weeks, which is short compared to the sediment histories.

Results and discussion

^{210}*Pb model validation*

The use of different ^{210}Pb dating models has been discussed at length in the literature (e.g. Oldfield & Appleby 1984; Robbins 1978). The CRS model applies when there is evidence that increases in sedimentation rates dilute ^{210}Pb concentrations in sediments and vise versa. Evidence for this phenomenon arises when: (1) inflections in the ln ^{210}Pb – cumulative dry mass activity curves show departures from linearity; (2) ^{210}Pb inventories are not correlated with sedimentation rates; and (3) the sediment ^{210}Pb inventories for a particular region resemble the measured atmospheric flux rate for ^{210}Pb. All of these criteria are met in our data. Firstly, inflections in the ln ^{210}Pb activity – cumulative dry mass curves are clearly evident (Figures 1 to 3). Moreover, these inflections take place at a mean CRS inferred date of 1981 $\pm$1.9 yr SD, n = 8 cores, which shows a highly consistent pattern among lakes in this region regardless of recent catchment disturbance, thereby suggesting a strong external influence. Secondly, the ^{210}Pb inventories among cores are not correlated with sedimentation rate (p = 0.38) which must occur for CIC model dates to be valid. Thirdly, the mean ^{210}Pb inventory calculated in these cores (178 Bq m^{-2} yr^{-1}) is quite close to the calculated atmospheric flux of ^{210}Pb measured at the nearby Experimental Lakes Area (150 Bq m^{-2} yr^{-1}, Omelchenko et al. unpublished report) indicating that catchment export of ^{210}Pb must be small relative to atmospheric ^{210}Pb flux, which is an implicit assumption of the CRS model. These results, coupled with abundant evidence that CRS dates have much better agreement with historically known event horizons than CIC dates (Binford et al. 1993; Blais et al. 1995) provide overwhelming support to the use of CRS model calculations.

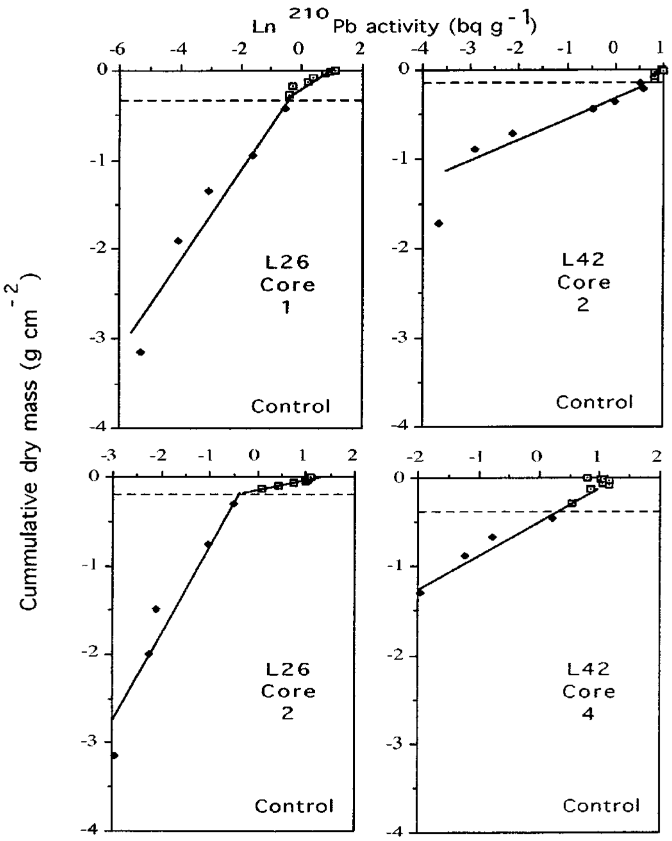

Figure 1. Pb-210 proflies for the cores taken from control lakes. Square symbols are post-1980 and diamond symbols are pre-1980. The dashed line marks 1980 in the chronology.

^{210}*Pb profiles*

The ^{210}Pb decay curves are shown for the reference lakes (Figure 1), for the lakes whose catchments were clearcut (Figure 2), and for the lakes where wildfires swept through the catchments (Figure 3). The ^{210}Pb profiles in these figures reveal that eight of the eleven cores exhibit pronounced inflections in their ln ^{210}Pb – cumulative dry mass curves occurring at the inferred date of 1981 $\pm$1.9 yrs SD. The decline in slope near the surface indicates a decline in sedimentation rate in the most recent portions of the profiles. This pattern is clearly evident in one reference lake (L26) as well as in all the clearcut basins (M6N, M6S, M4E, M4W, L442), and one of the two burn lakes (Percy). These observations are surprising, and appear to contradict previous studies that have documented increased sedimentation rates in response to deforestation (Flower et al. 1989; Dearing 1991b), as one might expect.

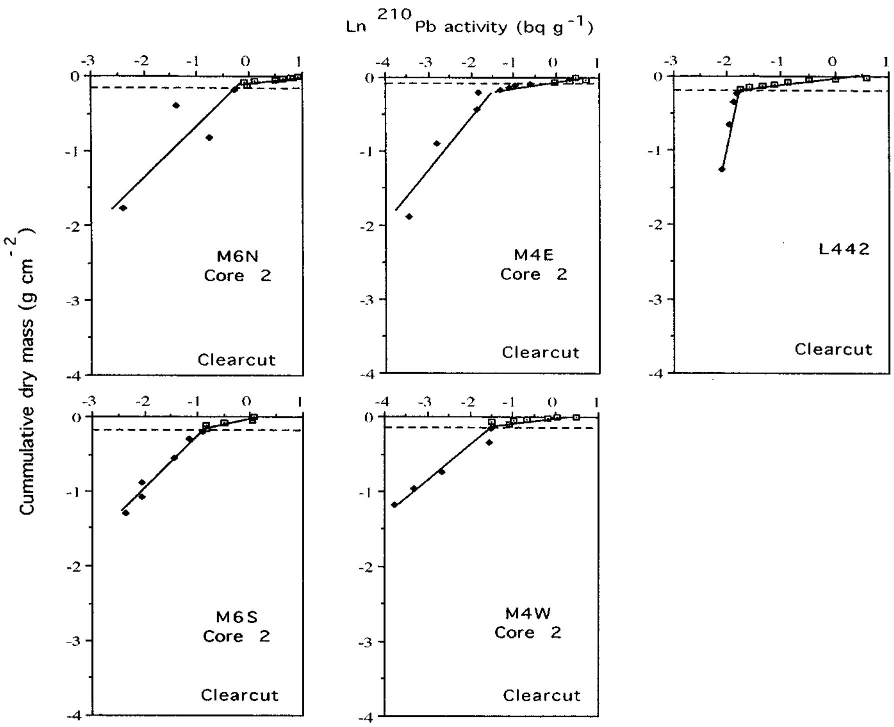

Figure 2. Same as Figure 1, for lakes where catchments were clearcut in the early 1980's.

Sediment core parameters for the ^{210}Pb analysis are in Table 2. Because there is an observed inflection in ^{210}Pb activity with depth consistently at the inferred date of 1980, we calculated sediment accumulation rates (hereafter SAR) for the pre- and post-1980 time periods. This was also appropriate because most catchment disturbances such as clearcutting (in basins M-4E, M-4W, M-6N, M-6S, L442) and fire (Percy and Little Joe) took place in the early to mid 1980s (Table 1). Pre-1980 SARs were significantly greater than post-1980 SARs (paired *t*-test, $t_8 = -2.48$, $p = 0.038$).

Sediment accumulation rates are plotted against the catchment area/lake area ratio (hereafter called the drainage ratio) in Figure 4 for both the pre- and post-1980 time periods. There is a significant positive correlation between the pre-1980 sediment accumulation rates and the drainage ratio ($p < 0.05$) indicating a measureable effect of catchment runoff on sedimentation rates. This effect is not apparent in the post-1980 sedimentation rates, which indicates a reduction in catchment runoff and erosion for these lakes. Also, the reduced sedimentation rates occuring after 1980 are clearly visible in Figure 4.

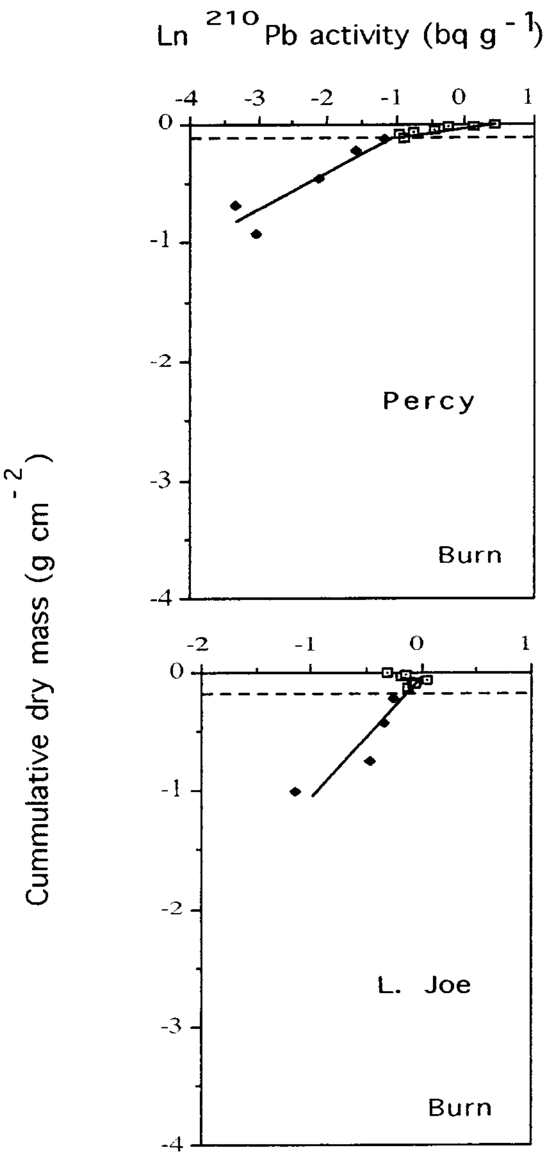

Figure 3. Same as Figure 1, for lakes where catchments were extensively burned in the late 1970's and early 1980's.

Table 1. Characteristics of study lakes.

Lake	Lake area (ha)	Catchment area (ha)	Zmax (m)	Coring depth (m)	% basin disturbance	Lat.	Long.
Reference							
L42	26	44	17	17	0	49°07′	92°08′
L26	30	88	37	36	5 (1987)	49°07′	92°10′
Minor disturbance							
L422	16	145	18	15	90 (1975–79)	49°46′	94°49′
Clearcut							
M6S	19	21	13	13	99 (1983)	49°15′	92°11′
M4W	22	42	12	9	90 (1983)	49°15′	92°12′
Clearcut/road							
M6N	14	76	13	13	99 (1983)	49°15′	92°11′
M4E	10	46	10	10	90 (1983)	49°15′	92°12′
Fire							
Percy (BN1)	39	170	30	19	100 (1980)	49°46′	94°06′
L Joe (BN2)	10	87	–	12	100 (1980)	49°51′	94°51′

Reproducibility of the results and focus corrections

The CRS model provides a means of reconstructing sedimentation patterns over the course of decades by calculating the instantaneous sedimentation rates using eq. 5. Duplicate cores were analyzed from the two reference lakes (L42, L26) to assess within-basin variability in the time trend analysis. SARs calculated from duplicate cores in L42 are shown in Figure 5. Core 4 had consistently higher calculated SARs than core 2 before the focus correction procedure was applied. After correcting for focusing, we observed a convergence between these two replicate ^{210}Pb profiles demonstrating a striking reproducibility in ^{210}Pb among cores (Figure 5). In L26, the agreement between replicate cores is also very high (Figure 6). Moreover, both cores show clear evidence for decreased sediment accumulation rates in the past 25 years.

Long term sedimentation patterns

One way to explain the apparent declines in sedimentation rates is to postulate that there has been a decline in catchment erosion in recent years due to the documented changes in precipitation, evaporation, and runoff. Detailed limnological analyses are available over the past 25 years from the nearby

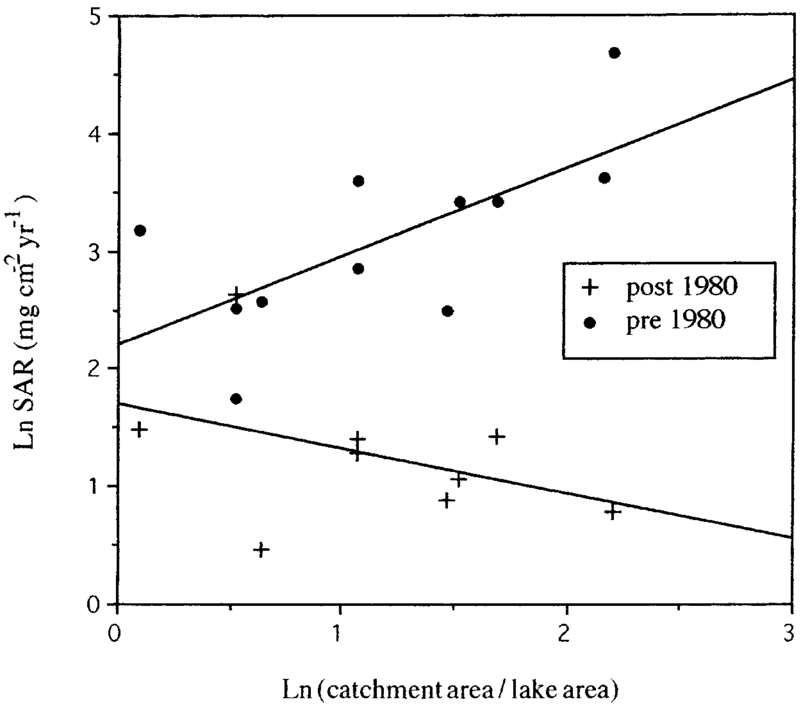

Figure 4. Sediment accumulation rates plotted against the catchment area/lake for the two time intervals, 1900–1980 and 1080–1995. The post-1980 SAR values for L42 core 4 and L. Joe are not shown due to surface mixing which obscured the recent sediments.

Experimental Lakes Area watersheds (Schindler et al. 1996), which allows for a cross-comparison over this recent time interval. We hypothesize that these extensive regional hydrological changes were manifested in the sediment profiles.

Schindler et al. (1996) reported a linear change in annual runoff from approx. 400 mm yr^{-1} in 1970 to approx. 150 mm yr^{-1} in 1990. According to the model relating suspended sediment load with discharge (eq. 8), we would expect a proportional decrease in sediment load over that same time interval (a decline of about 63%). The observed sedimentation rates in the present study have been reduced by about 80% on average (Table 2) in one reference and in all the lakes where the catchments were deforested. The reference lake 42 does not exhibit a decline in sedimentation rate in either of the two cores dated radiometrically. This may be because lake 42 has a relatively small catchment area (Table 1) and it has no channelized inflow (R.J. Steedman, Ontario Ministry of Natural Resources, pers. comm.). The largest changes in sedimentation rates occurred in lakes with the largest drainage ratios (Figure 4). Since the relationship between sediment load and flushing is non-linear

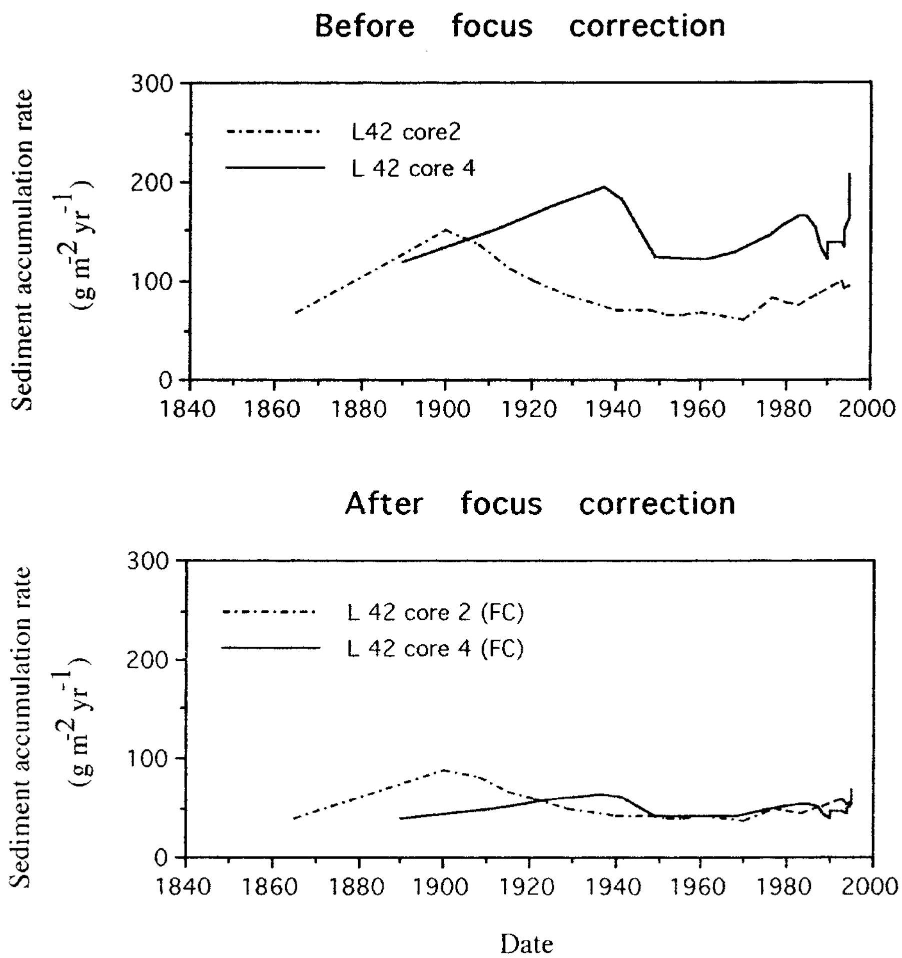

Figure 5. Instantaneous sediment accumulation rates plotted against year for duplicate cores extracted from lake 42. Above is shown the sedimentation rates before a focus correction is applied, and below shows the rates after focus correction. Note the convergence of inferred SARs after the focus correction is applied.

(eq. 8), changes in climate and hydrology will be magnified in large catchments, and thus the greatest changes in hydrology and water chemistry will be observed in lakes with large drainage ratios.

Confirmation of calculated sedimentation rates

The calculated sedimentation rates were examined in relation to the stable Pb profiles to see whether the known historical Pb flux concurs with our measured fluxes. Figure 7 shows cores from reference lakes 42, 26, and ELA

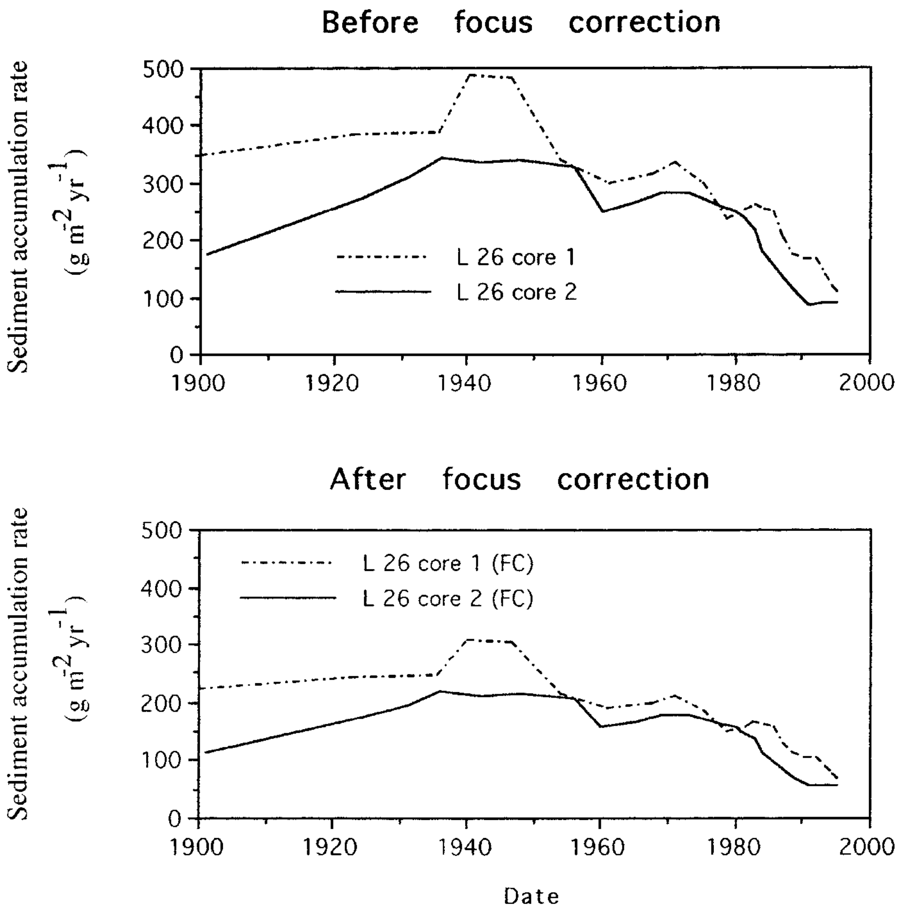

Figure 6. As in Figure 5 for lake 26.

L442. Each lake has a different sedimentation pattern. Lake 42 shows little change in sedimentation rate, whereas L26 shows a sharp decline, and L442 shows a very pronounced decline in SAR in recent years. In lake 42, the observed Pb distribution shows a peak around 1970, and a gradual decline towards the surface which reflects the known atmospheric Pb flux. The predicted curve, though more pronounced at the peak, reflects this pattern. In L26, both the observed and expected Pb curves peak at a later time than the expected date of 1973 as a result of the drop in sedimentation rate near the surface. In lake 442, both the observed and predicted Pb curves increase until the surface, again as a result of the sharp drop in sedimentation rate in the surface horizons of the core.

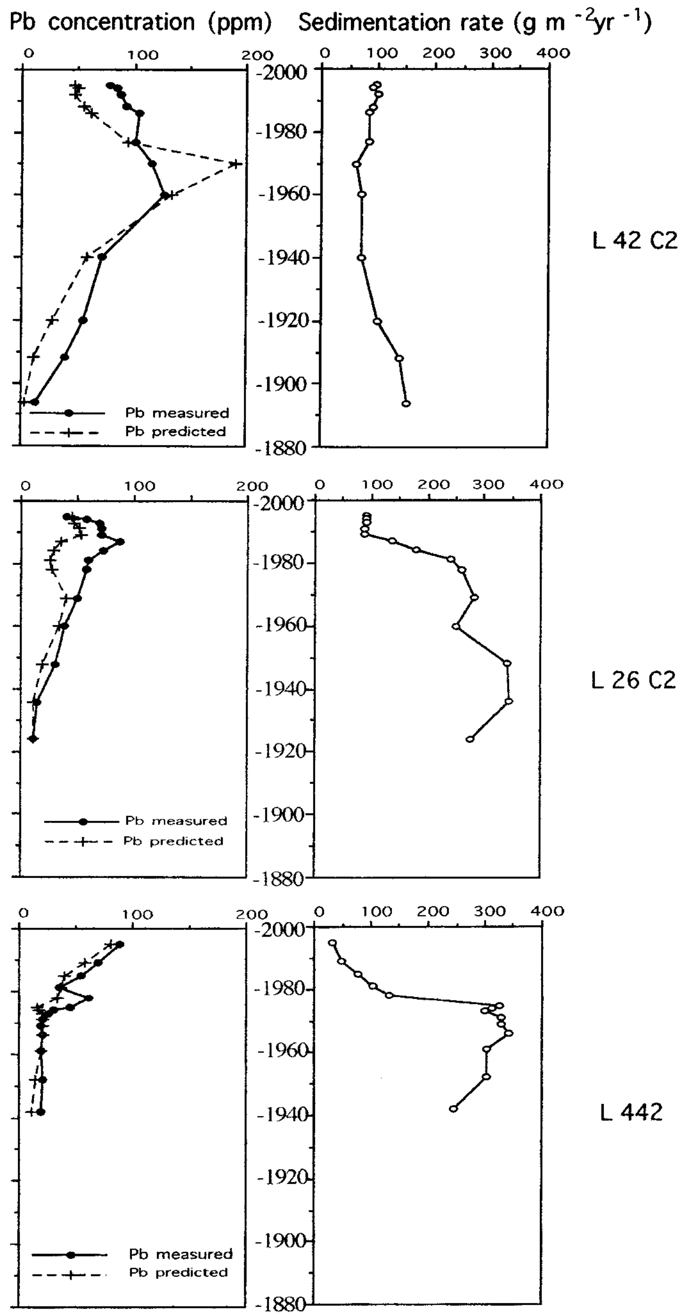

Figure 7. Measured and predicted Pb concentrations according to historical fluxes of anthropogic Pb.

Table 2. Details of the ^{210}Pb analysis. Co is the concentration of ^{210}Pb at the surface of the core, I is the inventory of ^{210}Pb in the core, and SAR is the sediment accumulation rate.

Lake/Core	Co (Bq g^{-1})	I (Bq m^{-2} yr^{-1})	SAR [pre 1980] (mg cm^{-2} yr^{-1})	SAR [post 1980] (mg cm^{-2} yr^{-1})	Focus factor
Reference lakes					
L26 Core 1	3.10	237.4	17.3	4.07	1.58
L26 Core 2	2.97	238.4	36.6	3.65	1.59
L42 Core 2	2.66	255.0	5.75	13.77	1.70
L42 Core 4	2.28	457.6	12.21	Mixed	3.05
Clearcut lakes					
M6N Core 2	2.52	190.8	30.56	4.13	1.27
M6S Core 2	1.11	114.7	23.75	4.42	0.76
M4E Core 2	1.59	61.2	30.26	2.89	0.41
M4W Core 2	1.63	70.5	13.13	1.59	0.47
L442	1.79	83.3	107.3	2.21	0.55
Burn lakes					
Percy (BN1)	1.56	51.1	12.07	2.43	0.34
L. Joe (BN2)	1.15	199.2	36.76	Mixed	1.33

The purpose of this exercise is two-fold. Firstly, it demonstrates that interpretations of concentration profiles in sediment cores without considering flux measurements is problematic. Figure 7 depicts three very different concentration profiles which each depict similar flux patterns. Secondly, it serves as a means of confirming sediment flux measurements calculated using ^{210}Pb chronology. There are few crosschecks available to us to corroborate ^{210}Pb derived dates and sedimentation rates. A simple method is performed here to assess ^{210}Pb derived sedimentation rates using the stable Pb profile as a reference. The results of the analysis in Figure 7 demonstrate that drastic declines in sedimentation rates, which we suggest is the result of the recent drying trend, can in some instances act to enhance the concentrations of contaminants such as Pb in the surface sediments in spite of recent declines in Pb emission over the last two decades. Although this was shown in the present study to be the case for Pb, it may also apply to other banned compounds such as DDT, PCBs, and a host of pesticides no longer in use such as toxaphene and chlordane. One implication of this work is that banned substances may be continuing to increase in concentration in surface sediments despite the fact that they are no longer in use.

Conclusions

Declines in sedimentation rates after 1980 are shown to coincide with decreased precipitation and higher temperatures over the past twenty-five years in northwestern Ontario lakes (Schindler et al. 1996). Although suspended sediment loads in rivers as well as the loading of nutrients and other dissolved solids have been observed to increase after logging and fire within these areas (e.g. Beaty 1994), this study of sediment profiles spanning several decades shows that the longer-term changes in catchment hydrology have profoundly affected the catchment yields of eroded materials and exceeded land use effects in these areas.

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