A Reconnaissance of Contaminants in the Fraser River Drainage Basin Using Dated Sediment Cores from Six Lakes.

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Prepared for:

Environment Canada
Environmental Conservation Branch
Aquatic and Atmospheric Sciences Division
700-1200 West 73rd Avenue
Vancouver, BC V6P 6H9

Prepared by:

R.W. Macdonald and D.W. Paton

Institute of Ocean Sciences, P.O. Box 6000
Sidney, B.C. V8L 4B2

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Aquatic and Atmospheric Sciences Division
Environmental Conservation Branch
Environment Canada
700-1200 West 73rd Avenue
Vancouver, B.C.
V6P 6H9

Abstract

Sediment cores have been collected from six lakes in the Fraser River drainage basin (Moose, Stuart, Kamloops, Nicola, Chilko and Harrison Lakes). Each core has been dated using ^{210}Pb and, where possible, by counting varves or measuring ^{137}Cs. The dated sediments have been analyzed for a suite of priority contaminants including metals, polynuclear aromatic hydrocarbons and organochlorine contaminants. A preliminary survey shows Pb to be a ubiquitous contaminant, usually found in minor amounts consistent with its use as a gasoline fuel additive. Other metals are found as minor contaminants in lake sediments, the clearest record being Hg contamination in Stuart Lake during the 1940s originating from a mine on Pinchi Lake. PAHs are found to contaminate Kamloops Lake and Harrison Lake, with the source of contamination probably being local. Organochlorine compounds are generally found in low concentrations consistent with long range transport. Exceptions are DDT in Nicola Lake, which was contaminated during local applications in the 1970's and 80's, and PCDD/Fs in Kamloops Lake which derive primarily from the pulpmill when it used chlorine as bleaching agent.

Résumé

Des carottes de sédiments ont été prélevées dans six lacs du bassin hydrographique du Fraser (lacs Moose, Stuart, Kamloops, Nicola, Chilko et Harrison). On a effectué la datation au 210Pb et, lorsque la chose était possible, on a compté les varves ou mesuré le 137Cs. Les sédiments ainsi datés ont été analysés pour voir s’ils contenaient une suite de contaminants d’intérêt prioritaire dont les métaux, les hydrocarbures aromatiques polycycliques et les organochlorés. Une étude préliminaire révèle que le plomb est un contaminant très répandu, qui se trouve habituellement en faibles quantités correspondant à son utilisation comme additif à l’essence. On a trouvé d’autres métaux comme contaminants mineurs dans les sédiments lacustres, l’exemple le plus probant étant la contamination par le mercure dans le lac Stuart durant les années 40, à partir d’une mine sur les bords du lac Pinchi. Les HAP contaminent les lacs Kamloops et Harrison; leur source est probablement locale. On trouve habituellement de faibles concentrations de composés organochlorés, ce qui correspond au transport sur longue distance. Il y a deux exceptions : le lac Nicola, qui était contaminé par le DDT à la suite d’une ou de plusieurs applications locales, et le lac Kamloops dont les PCDD/F proviennent principalement de l’usine de pâte, lorsque celle-ci utilisait le chlore comme agent de blanchiment.

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1.0 Introduction

It is well known that lake sediments have the potential to provide both regional and local contaminant histories (for a few examples of an extensive literature see [Vilanova, 1995 #1116; Muir, 1996 #1352; Maatela, 1990 #43; Wakeham, 1980 #381; Müller, 1977 #468; Eisenreich, 1989 #524; Christensen, 1993 #893; Sanders, 1992 #894; Oliver, 1989 #896; Renberg, 1994 #941]. Therefore, as part of the Fraser River Action Plan sediment cores were collected from six lakes in the Fraser Drainage basin (Figure 1). The cores were dated using 210Pb as the primary dating tool. In the case of some cores, the 210Pb dating was evaluated using 137Cs and by counting laminae, or varves, which were observed in two of the lakes. In addition to dating, chemical analyses were carried out on the sectioned cores for metals, hydrocarbons, and organochlorines. Due to constraints both in the budget and in the size of individual core section samples, a selected subset of the core sections was analysed for each contaminant. The primary objective of this study was to provide a preliminary assessment of the history, scale and potential sources of contamination in each lake, and in the Fraser River drainage basin in general, by examining contaminant trends within cores and comparing these between lakes. Surprisingly, there appear to be few published data on contaminants in BC lake sediments with which to compare the results reported here.

Lakes can be impacted by contaminants from at least two sources: 1) Long-range atmospheric transport from outside the drainage basin with deposition either directly into the lake or indirectly into the lake via the drainage basin; 2) Contaminant entry directly or indirectly into the lake from local industrial or municipal activities within the drainage basin. Often, the source of the contaminant can be surmised either from the patterns of contamination (for example congener distribution of PCBs) or from the known history of contaminating sources within the drainage (for example Hg mining or the establishment of a pulp mill). In the following report, we provide the preliminary findings of this study, focusing on the clearest examples of contaminant trends evident in the sediment cores.

2.0 Geographical and Hydrographical Description of the Lakes

The six Fraser Basin Lakes chosen for the study were Moose, Stuart, Chilko, Kamloops, Nicola and Harrison (Figure 1). As shown in Table 1 the characteristics of the lakes likely to affect contaminant and sediment capture vary widely. For example, at one extreme Kamloops is a flow-through lake little more than a widening and deepening of the Thomson River. It therefore has an exceptionally large watershed and short bulk residence time for the water. In contrast, Chilko Lake has a much smaller watershed, a larger lake volume, and a bulk residence time of about 17 years. Of the six lakes, only Kamloops Lake and Moose Lake sediments exhibited varves suggesting a strong seasonal pattern undisturbed by sediment mixing in these two lakes (cf. [Pharo, 1979 #407; St. John, 1996 #1338; Desloges, 1995 #891].

3.0 Methods

3.1 Sediment Sampling

Sediment cores were collected from a small boat using a KB gravity corer with an acrylic tube (10 cm i.d.), and coring locations were determined using a hand-held GPS instrument. Coring sites were generally situated in bathymetric depressions toward the center of the lakes and well away from any obvious disturbing influences like streams flowing in at the side. In the case of Kamloops and Moose Lakes, the sites were chosen based on estimated sedimentation rates from the literature [Pharo, 1979 #407; Desloges, 1995 #891]. A summary of sample locations and dates is given in Table 2.

To minimize sediment disturbance during coring, the sediments were penetrated slowly (<0.3 m/s) with a clear-vented tube without a core cutter or catcher. When the weight came off the corer as it penetrated the sediments, a flat valve dropped down to maintain a seal at the top of tube. As the recovered core reached the water surface an end cap was placed over the bottom of the tube before bringing it above the water. The core was maintained in an upright position until subsectioned on site within several hours. Subsectioning was effected by placing the corer onto a piston and extruding the core upwards incrementally. The sediment was subsampled into plastic bags for ^{210}Pb and metals analyses and cleaned glass jars for organochlorine or PAH determination. Stainless steel subsectioning tools were cleaned between samples (tap water, distilled water, acetone), and sediment from the outer 1 cm of the core was discarded. The glass jars were stringently precleaned (detergent rinse, 4-h soak in 2% RBS, rinse, distilled water, rinse, and baked overnight at 350 °C overnight). To allow for a range in sedimentation rates without producing an unmanageable number of samples, subsamples were collected as follows: at 1 cm intervals for the first 10 cm, 2 cm intervals from 10 to 20 cm and 5 cm intervals from deeper in the core.

3.2 Sediment Analyses

^{210}Pb determination: All ^{210}Pb determinations were performed by Flett Research Ltd. ^{210}Pb was determined by measuring ^{210}Po [Eakins, 1978 #919] using a 300-mm silicon surface barrier detector with a Canberra 8180 MCA. Counting errors, maintained at less than 10%, were verified by replicate analyses on sediment samples.

^{137}Cs determination: All ^{137}Cs determinations were performed by Flett Reseach Ltd. ^{137}Cs was measured using gamma spectroscopy with either a Ge(Li) detector or a NaI(Tl) detector.

For determinations using the Ge(Li) detector, dried sediment samples were sealed in 50 x 9 mm polystyrene plastic petri dishes. Each dish was placed on an Ortec Ge(Li) coaxial detector having a crystal diameter of 58 mm , an active volume of 114 cm^3, a relative (to NaI) efficiency of 23.3 % and completely surrounded by 4 inch lead shielding. A gamma spectrum was acquired by counting each sample for a minimum of 24 hours. The 662 Kev line of ^{137}Cs (85 % with ^{137}Ba m) was resolved using Nuclear Data Accu-Spec gamma spectroscopy software. ^{137}Cs was quantified by comparing the sample count rate with that of a NIST Gamma-Ray Emission-Rate Standard: Spiked Clay counted in the same geometry as the samples.

For determinations using the NaI(Tl) detector, dried sediment samples (60 °C overnight) were placed in 56 mm diameter aluminum foil planchettes, compressed to about 15 PSI and the height measured. The planchette was placed upon a 3” X 3" NaI(Tl) gamma detector contained within 2” of lead shielding. A gamma spectrum was acquired for a minimum of 24 hr and the counting efficiency calculated against a graded height series of ^{137}Cs spiked river sediments (NIST) of the same diameter as the samples.

Signal acquisition was via a Canberra 8180 MCA using a 1000 channel window.

Metals determination: Metals analyses were performed by ERI (Elemental Research Inc.). A total sediment digestion was performed using a combination of hydrofluoric, nitric and perchloric acids. The sample was dried at 105 °C for 2 hours to obtain a dry weight. About 0.1g of dried sample was added to a teflon digestion vessel, 2 mL of HF (Seastar) and 0.2 mL of HCl (Seastar) were added, and the bomb was sealed and heated overnight on a hot plate. The sample was then fumed off, 1 mL of HNO_3 (Seastar) added, and the bomb was recapped and heated. After two hours, the sample was fumed off again and 2 mL of HCl (Seastar) was added. After a final 1 hour of heating the sample was made up to 100 mL with 18 MΩ deionized water. The digestate was analysed by Inductively Coupled Plasma Mass Spectrometry (ICPMS) for the elements of interest. Calibration was performed using a single point traceable to multielement SPEX standard solution.

An instrument blank was analysed at the beginning of each analytical run to monitor instrument cleanliness. A preparation blank was analysed with each analytical run to monitor reagent purity.

Calibration was performed at the beginning of each analytical run. Certified Reference Materials were analysed at a rate of 1:10 (CRM:sample) or two per batch whichever was greater. Duplicate samples were analysed for 10% of the samples, or two per batch whichever was greater.

Arsenic was determined from a Nitric Acid extraction since Arsenic produces volatile fluorides and can not be quantitatively recovered from a total digest.

Hg determination: Hg analyses reported here were performed by Flett Research Ltd. For sediment sample analyses, approximately 1.0 mL of wet sediment was added to a 20.0 mL test tube and three mL of acid (7:3 HNO_3:H_2SO_4) was added to the samples. One sample was run in triplicate with one of the triplicate samples being spiked with 300 ng of Hg to evaluate spike recovery. The magnitude of the spike was chosen to be at least 10 times the expected amount of Hg in the sample. The other two samples were used to evaluate precision. Marbles were placed on top of the test tubes and the tubes are allowed to sit for one hour. The samples were then put into an aluminum heating block with a lead sheet wrapped around them as a draft shield, and refluxed overnight (but not allowed to boil). The next day, after cooling, 150 mL of BrCl was added to each sample, and they were transferred into acid-cleaned 5 mL teflon vials for storage.

The Hg was determined using a bubbler/gold trap/fluorescence detector analytical system. Drierite pre-traps were installed after the bubblers to remove water vapor and acid fumes, which otherwise may collect in the gold traps and be released into the fluorescence cell to condense and give a false peak due to scattering of the excitation radiation. Before analysis, the drierite pre-traps were purged of Hg by filling the bubblers with about 70 mL of 1% HCl and 1 mL of SnCl and bubbling overnight with Hg-free N_2 at about 100 $mL.min^{-1}$

For sample analysis, One mL each of SnCl and $NH_20H.HCL$ was added to the bubblers and they were bubbled for 20 minutes to purge any Hg in the reagents. The SnCl is used to reduce the mercury to Hg^0 so it can be trapped on gold traps, and $NH_20H.HCL$ is used to destroy the BrCl because free halogens will damage the gold traps and result in low or irreproducible results thereafter. Two hundred mL of sample was then added to the bubblers and Hg-free gold traps installed at the end of each bubbler. The samples were bubbled with Hg-free N_2 at approximately 200 mL.min-1 for 20 minutes, at which time the traps were removed and tightly sealed with teflon caps. To analyze the mercury contained on the gold trap, a nichrome wire coil was placed around the trap, and the trap was inserted into the analytical train between the incoming Hg-free helium and the fluorescence detector. Helium flow through the column was at a rate of about 30 mL.min-1. As soon as the trap was installed, the integrator, which controls the heat and the fan, was turned on. Gas was allowed to pass through the traps for 2 minutes to remove condensed water after which electrical current (10 VAC) was applied to the coil for 4.0 minutes thermally desorbing the Hg as Hg^0. The coil was then turned off, the fan was turned on for 2 minutes and, finally, when the fan was turned off the integrator reported the peak area. The trap was removed from the heating coil, teflon caps were replaced on the trap, and the next trap to be analyzed was installed. System blanks were run at the beginning and at the end of the day and a 1 ng standard was run before the samples. Analytical blanks were also run to determine how much mercury each reagent contributed to the final result. Deviation from the mean of duplicate samples ranged from 0 - 9 % in the concentration range of 17 to 39 ng Hg/ g (wet weight sediment). At 30 ng

Hg/g (wet weight sediment), the blank was usually between 1 and 2% of the measured concentration. Recoveries ranged between 89% and 104 % over 4 analytical days when measuring nearshore ocean sediments.

Hydrocarbon determination: Sediment samples were analyzed by Axys Analytical Services Ltd for the following classes of compounds; parent and alkylated polycyclic aromatic hydrocarbons (PAH); alkanes; chlorinated pesticides; PCBs (as Aroclors) and PCB congeners; and toxaphene.

The sample handling protocols, sample workup procedures and instrumental analyses have been documented in the AXYS data reports as are the results for QA/QC samples (procedural blanks, analysis duplicates, and internal reference materials).

Sediment samples were thoroughly homogenized using a solvent-rinsed spatula. Any rocks over 0.5 cm in diameter were removed and samples were stored at -20 °C until analyzed. Immediately prior to analysis, homogenized samples were thawed, stirred thoroughly and subsampled for analysis. Samples were analyzed wet and a separate subsample was taken for moisture determination (dried overnight at 105°C and weighed) so that data could be reported on a dry weight basis.

Briefly, the method consisted of first spiking each sample with an aliquot of surrogate standard solution containing nine perdeuterated PAHs and one perdeuterated alkylated PAH. An aliquot of a solution containing four deuterated alkanes was also added for analysis of alkanes. Each sample was ground with sodium sulphate, packed in a glass column and eluted with methanol followed by dichloromethane. The eluate was transferred to a separatory funnel and washed with solvent-extracted, distilled water. Extracts were fractionated on silica gel into two fractions: a non-polar fraction containing alkanes, and a polar fraction containing parent and alkylated PAHs. Prior to instrumental analysis, an aliquot of recovery standard containing three perdeuterated PAHs was added to each fraction. Each fraction was analyzed by high resolution gas chromatography with low resolution mass spectrometric detection (HRGC/LRMS) using a Finnigan INCOS 50 mass spectrometer equipped with a Varian 3400 GC, a CTC autosampler and a DG 10 data system.

Organochlorine determination: Organochlorine determinations reported here were performed by Axys Analytical Laboratories. (Analyses for PCDD/Fs and PCBs in Kamloops Lake sediments were analyzed at the Regional Dioxin Laboratory at IOS and the methods are described separately [Macdonald, 1997 #1382]). Briefly, samples were first spiked with a solution containing several ^{13}C-labeled surrogate standards including hexachlorobenzene, gamma-BHC, p,p'-DDE, p,p'-DDT, Mirex, CB101, CB180 and CB209 (for analysis of PCBs, non-polar and moderately polar pesticides and toxaphene) and deuterium-labeled endosulphan (for analysis of highly polar pesticides).

Each sample was solvent-extracted on a shaker table, and the extracts were washed in a separatory funnel with solvent-extracted distilled water. Extracts were dried over anhydrous sodium sulphate and concentrated. Extracts were separated on Florisil into two fractions, a combined Fl + F2 and an F3. The PCBs, non-polar and moderately polar pesticides were collected in Fl +F2, while the polar pesticides were collected in F3. Each fraction was prepared for instrumental analysis by transferring to an autosampler vial and adding an aliquot of recovery standard (^{13}C-labeled CB153).

The Florisil Fl +F2 fraction was analyzed for PCBs as Aroclor equivalents, PCB congeners and non-polar / moderately polar pesticides by high resolution gas chromatography with high resolution mass spectrometric detection (HRGC/HRMS). Fl + F2 was analyzed for toxaphene by HRGC/LRMS operated in the electron capture negative ion mode (ECNI). The most polar pesticides, collected in the third Florisil fraction (F3), were analyzed by high resolution gas chromatography with electron capture detection (HRGC/ECD).

HRGC/HRMS analysis of Aroclors, PCB congeners and pesticides was carried out on a VG 70SE high resolution mass spectrometer equipped with a Hewlett Packard 5890 GC, a CTC autosampler and a VAX workstation. Toxaphene was analyzed using a Finnigan INCOS 50 mass spectrometer equipped with a Varian 3400 GC. Polar chlorinated pesticides in the F3 were analyzed using a Hewlett Packard 5890 GC with a ^{63}Ni electron capture detector (GC/ECD). Toxaphene was quantified by the summation of 21 peaks in the chromatogram.

Samples were worked up in batches of approximately 9 samples in addition to one known sample (a spiked sediment sample), one analysis duplicate and a procedural blank.. Each batch went from sample workup through instrumentation as a unit, and on to data interpretation and final reports. The sample results were reviewed and evaluated in relation to the QA/QC samples worked up at the same time.

The procedural blanks demonstrated non-detectable or low background levels of the target compounds.

Approximately 10 % of the samples were analyzed in duplicate. Agreement within each set of duplicates was acceptable and generally satisfied the Axys criterion of +/-(20% + Method Detection Limits) as well as the contractual specifications of +/-30%. Results for duplicates are reported along with the analysis results in data report appendices.

The recovery of each surrogate standard is monitored by comparing its response to that of the recovery standard added just prior to instrumental analysis. Surrogate recoveries between 50-120% were specified and in most cases, this criterion was satisfied. But some pesticide surrogate recoveries did not meet the criterion, despite repeat analysis. This is likely due to a sample matrix effect. Data accuracy should not be affected since analyte concentrations are automatically corrected for incomplete surrogate recoveries.

Batches of samples included a spiked sediment sample. The analysis of these "known" samples were generally within the acceptable percent recovery range of 70-130%. All spiked sediment samples were prepared at Axys by spiking a solution of authentic target compounds (either PCB Aroclors and pesticides; or PAHs and alkanes) into a weighed amount of in-house reference sediment (well homogenized and analyzed unspiked in-house).

Detection limits were calculated on a sample-specific basis and have been reported with each sample. The detection limit was calculated as the concentration corresponding to the area reject. The area reject, determined from the ion chromatogram of each compound, is the area of a peak with height three times the maximum height of the noise. Only peaks with responses greater than three times the background noise level have been quantified.

3.3 Sediment Dating and Data Presentation

Sedimentation rates have been determined using the ^{210}Pb profiles. Where possible, these dates have been corroborated by counting laminae clearly evident in some cores and by measuring ^{137}Cs. For Moose, Harrison and Kamloops Lakes, previous studies have evaluated sedimentation rate [Pharo, 1979 #407; St. John, 1996 #1338; Desloges, 1991 #892; Desloges, 1995 #891] and rates determined in these studies have also been compared with our estimates. The sedimentation rates listed in Table 3 have been used to convert depth in sediment to approximate time of sedimentation as given in plots of contaminants for the various lakes.

In the lake-by-lake discussion below, we provide plots only for the salient features that show up in the data set. For many elements or compounds there is no apparent contaminant record, or the contaminant concentrations are below or near the detection limit (see Table 4 a,b,c for a summary of the data set). Where two cores from the same lake tell essentially the same story, we generally plot the data from only one core to illustrate the trend(s). In the figures and text, Total DDT refers to Σ (o,p’-DDE + p,p’-DDE +o,p’-DDD + p,p’-DDD + p,p’-DDT o,p’-DDE); Total HCH refers to Σ (αHCH + βHCH + γHCH); Total PAH refers to Σ 178-278 parent PAH total excluding perylyene (phenanthrene, anthracene, fluoranthene, pyrene, benz*_a_**anthracene, chrysene and triphenylene, benzo*_b/j/k_**fluoranthene, benzo*_e_**pyrene, benzo*_a_**pyrene, indeno*_1,2,3-cd_**pyrene, benzo*_ghi_**perylene and dibenz*_a,h_**anthracene) and; Total alkanes refers to the sum of C_{12} to C_{36} alkanes.

4.0 Results and Discussion

4.1 Moose Lake (Figure 2)

The average sedimentation rates determined for the Moose Lake sediment cores appear to be reliable because the ^{210}Pb data are well behaved (Figures 3,4), agree well with the independent estimate based on counting laminae and with other work done in this lake [Desloges, 1995 #891]. There is some evidence of a change in sedimentation rate at site M-2 where laminae suggest slower rates in the top 7-8 cm of the core; this observation is mirrored by the ^{210}Pb data which show a shallower slope in that region (Figures 3,4). We have not tried to account for sedimentation rate variations in the present report but have simply used the ^{210}Pb average to establish time horizons. In addition to the agreement between varves and ^{210}Pb dates for average sedimentation ratets within cores, the contaminant records discussed below show good agreement between the two cores when viewed in the time dimension.

There is evidence of contamination of Moose Lake sediments by several heavy metals starting in about 1945-1950. Particularly clear is the contamination by Zn and Ni (Figure 5) and, to a lesser degree contamination by Cu and Co. The cores corroborate one another, showing parallel contamination trends in these metals with time. A potential source of contaminant metal may be the highway, and in particular road salt used on the highway. We are presently investigating the history of the highway construction and determining the composition of materials used to maintain it.

Lead (not shown) also exhibits minor contamination, particularly at M-2, in the interval 1950 to 1980. The trend in Pb differs from that of the other four contaminating metals discussed above. Gross Pb contamination has previously been found in Kootenay Lake due to local mining and the disposal of tailings into the lake [Macdonald, 1994 #1381]. However, activities such as mining play no role in Moose Lake, and the predominant source of the contaminant Pb is likely to be the automobile traffic along the north side of the lake. It has long been known that leaded gasoline produced an environmental signal starting in North America about 1927 when lead was first used as an anti-knock agent (in the case of Moose Lake, the date of entry of contaminant Pb will probably coincide with the date of road construction). Peak emissions occurred in the 1970s and, after about 1979, Pb was phased out of such use (for a summary of the use of lead in gasoline in BC see [Macdonald, 1991 #1380]). Therefore, the Pb observed in the sediments has probably entered the lake either directly as automobile exhaust aerosol or indirectly in runoff from the highway and soil adjacent to it.

Hg shows very low concentrations in the sediments (Figure 6). Nevertheless, there does seem to be a contamination signal in the upper layers of the M-2 core, with maximum values being achieved in the late 1970s to early 1980s.

Organochlorine compounds in Moose Lake sediments are found at very low concentrations, or not detected at all (Table 4a). DDT, HCB and HCH show the clearest trend with time (Figures 7-9). Total DDT enters the lake sediments after about 1940. Peak concentrations were achieved in the late 1960s, a few years after DDT was reported to be used in aerial spray operations in BC (1946-1962; [Prebble, 1975 #1378]). The total DDT in the sediments is distributed among DDT, and its metabolic products DDD and DDE, and therefore appears to be old and partially degraded (cf. [Zinck, 1975 #1383; Addison, 1978 #1384]. Clearly, the DDT in the lake could derive from large-scale aerial spraying from regions upwind of the lake, or from local small-scale use. We are currently trying to establish the record of DDT use for this and other regions of BC. HCH (Figure 8) and HCB (Figure 9) are detected in the sediments after about 1960 which is somewhat later than the introduction of HCH as a pesticide, in about 1945. This observed offset can easily be explained by the method detection limit which was about 0.01 ng/g for HCH for the 14-16 cm (ca 1957) sample. The dramatic decrease in HCH concentration during the late 1980s matches the global decrease in atmospheric concentrations of HCH at that time [Jantunen, 1995 #999].

Analyses of fish collected in 1991 led Donald et al. [Donald, 1993 #397] to conclude that there might potentially have been a PCB spill into Moose Lake from either the highway, railway, or electric power transmission line situated along the lake and its principal inlet river. Therefore, we analyzed the sediments with the intention of using the PCB record as a way to determine the source of PCB in the fish. Our analyses show the PCB concentrations to be extremely low in the sediments (Figure 10). Most of the PCB congeners were found to be at or below detection limits and the total concentrations were below those found in a number of northern lakes [Muir, 1996 #1352]. We infer from these data that it is unlikely that PCBs in the higher trophic levels of Moose Lake come from a contaminating event (i.e. a spill). Rather, it is more likely that long-range atmospheric transport is the source of the PCB found in the food chain, and the high levels in fish are supported by the trophic structure in the lake and the lake yield (Table 1) and not by exceptional inputs of PCB (e.g., see [Kidd, 1995 #1193]). Low PCB concentrations observed in

Moose Lake sediments may, in part, be caused by high sedimentation rates relative to other lakes where PCB records have been found (e.g., [Muir, 1996 #1352]). It is possible that future work using larger sample sizes could be used to develop a better PCB congener profile in these lake sediments.

PAH concentrations in Moose Lake sediments appear generally to be very low (Figure 11) and we see no obvious trend with time. In particular, the surface sediments do not suggest any detectable recent contaminant PAH additions. The n-alkanes (not plotted; see Table 4a) also are inordinately low in concentration throughout the Moose Lake cores. There may be a slight enrichment of alkane in sediments dating later than about 1960, but this might be due to diagenetic losses within the sediments rather than augmented recent inputs. Therefore, sediment trends provide no conclusive evidence of contaminant hydrocarbon entering the lake either from anthropogenic combustion sources or from petroleum (e.g., from road runoff).

4.2 Stuart Lake (Figure 12)

Sedimentation rates determined by ^{210}Pb appear to be reliable because, for both S-1 (Figure 13) and S-2 (Figure 14), the ^{210}Pb profiles exhibit an exponential decrease with depth in the core and the cores are sufficiently deep to achieve reasonable estimates of supported ^{210}Pb concentrations. Neither core had any apparent stratification (laminae) and both were generally massive, porous, gelatinous fine mud. In the case of S-2, ^{137}Cs was measured for a sub-set of the sections and its date of entry into the sediments and peak concentration are consistent with the ^{210}Pb assigned time frame (Figure 15, Table 3).

Most of the heavy metals in Stuart Lake sediments show very slight trends or no trends at all with time. There appear to be minor enrichments in Pb (Figure 16) after about 1940, peaking in the 1970s and 1980s and declining after that. As in the case of Moose Lake, this scenario is consistent with the source being emissions from the use of leaded gasoline. Zn may also show slight enrichments after about 1940 (Figure 17). The clearest case of metal contamination in Stuart Lake is seen for Hg in both sediment cores (Figure 18). Contaminant Hg first enters the sediments in the 1940s, and has been decreasing since that time. The signal is slightly stronger at

S-1, but there appears to have been a more protracted pulse at S-2 which has not yet witnessed a decrease to background. The entry of Hg into Stuart Lake coincides with the operation of a Hg mine at Pinchi Lake from 1940 to 1944 (the outflow of Pinchi Lake enters Stuart Lake via Pinchi Creek; Figure12). What is surprising is how far the contamination extends; it was found at the S-1 site, over 25 km away from the Pinchi River inlet and upstream from the Stuart Lake outflow. More recently, the mine reopened and operated from 1968 to 1975 but this later opening does not seem to be accompanied by a Hg pulse, probably due to a comprehensive set of environmental controls in the later mining operation. Integrating the Hg contaminant burdens for both cores (60 to 100 ng/cm^2) and assuming that accumulating regions comprise half of the lake (i.e. 180 km^2) we estimate that the contaminant Hg in Stuart Lake sediments could be produced by the addition of 100 to 200 kg Hg which is a very small portion of the war-time production (1,914,000 kg; Mineral Statistics, Land Management and Policy Branch).

Organochlorine (and PAH) determinations were made for only four sections of the S-1 core (0-1cm; 5-6cm; 9-10cm; 50-55cm) and for the surface section of core S-2. These few data points suggest that Stuart Lake sediments are similar to the more detailed profiles observed in Moose Lake. For example, surface concentration of total DDT in S-1 is low, but detectable, and a relatively higher value of 1.25 ng/g is observed in the 5-6 cm horizon (centering at about 1961). The deeper sections contain no detectable organochlorines, consistent with their age (mid 1930s or earlier).

Total PAH concentration is about four times higher in Stuart Lake than in Moose Lake sediments (Table 4c). However, the profiles do not appear to vary much with depth, so it is likely that the higher values in Stuart Lake are supported by natural inputs and the lake geochemistry.

4.3 Chilko Lake (Figure 19)

Sedimentation rates in both cores were difficult to determine with certainty. In core C-1, the ^{210}Pb profile appears well behaved (i.e, exponential decrease with depth (Figure 20)). However, the estimated sedimentation rate reported in Table 3 is probably reliable within only a factor of two because the calculated rate is sensitive to choice of supported ^{210}Pb. Unfortunately, we were not able to measure supported ^{210}Pb from deep sections of the core because the corer did not penetrate deeply enough to get to background. At C-3, the ^{210}Pb profile (Figure 21) suggests that this site has been disturbed by a recent slump or a turbidity flow as evidenced by a region of low ^{210}Pb in the top 3 cm. At the time of sampling, a recently-scarred slope on a steep area bordering the lake was observed and this may have been responsible for the sedimentation episode in the lake. Beneath the surface layer, from 3 cm to 30 cm, the ^{210}Pb profile follows approximately an exponential decay with depth in the core but the sediments clearly appear to have been disturbed. As in the case of C-1, the calculated sedimentation rate is sensitive to choice of supported ^{210}Pb. At both sites it is clear that the ^{210}Pb flux is very low (Table 3), probably because much of the region is frozen or glaciated rock limiting the ^{222}Rn fluxes.

There is very little evidence of any trends with depth (time) of metals, including Hg, in Chilko Lake sediments, suggesting little or no detectable contaminant inputs. Similarly, for the organochlorines there are a few detectable concentrations in upper portions of the sediments but the values are low (Table 4b). PAHs appear also to be low (Table 4c), similar to the Moose Lake concentrations, and there are no obvious trends with depth in the core.

4.4 Kamloops Lake (Figure 22)

The average sedimentation rates determined by ^{210}Pb appear reasonable based on the exponential decay with depth (Figures 23, 24) and the independent estimate of sedimentation rate available from counting laminae for the Kamloops Lake cores (Table 3). Furthermore, our sedimentation rates agree well with the general trend in sedimentation rate down the axis of lake established by [Pharo, 1979 #407] who counted varves in six sediment cores. However, both the K-1 and K-2 cores exhibit larger laminae in the top 2 to 3 cm which, if produced by either a turbidity flow or a couple of years of relatively high sedimentation rates, may generate offsets between the ^{210}Pb average dates and those according to the laminae. Here, for convenience, we have simply used the ^{210}Pb average dates to construct contaminant histories. For the general trends in sediment contaminants during the past century, this will cause little error. But, for a more careful look at the most recent contaminant history (last 20 years or so), the laminae will probably prove to be more reliable (see [Macdonald, 1997 #1382]).

For most of the metals there is little evidence of contaminants entering the Kamloops Lake. Pb (Figure 25) may show a very slight enrichment after about 1950 with a peak in about 1980 and generally decreasing since that time. As before, the Pb trend in sediments is entirely consistent with the known use of Pb as an antiknock agent in gasoline. Hg also clearly shows enrichment in the mid to late 1980s. We have no idea what the source may have been. If we assume the Hg enrichment at the maximum in core K-1 is typical, then the total sediment budget of 3.4 x 10^8 kg/yr ([Macdonald, 1997 #1382] implies a maximum contaminant Hg flux to the sediments of Kamloops Lake of about 7 kg/yr.

The organic contaminant analyses of K-1 and K-2 sediments provide the strongest examples of contamination in this study. As shown elsewhere ([Macdonald, 1997 #1382]) the lake has been impacted by polychlorinated dioxins and furans (PCDD/Fs) and a few specific PCBs (13, 15 and 37) which clearly have originated for the most part from the Weyerhaueser pulp mill which was established on the Thompson River above Kamloops Lake in 1965. However, some of the PCDD/Fs seem to have come from the use of wood preservatives (e.g., pentachlorophenol), probably on the power poles or railway ties that run the length of either side of the lake. Since the dioxins, furans and PCBs originating from the pulp mill have been discussed in detail elsewhere they will not be further examined here. Kamloops Lake sediments also contain small quantities of PCBs (Figure 27) consistent with atmospheric transport, and low levels of contamination by DDT, HCB and HCH as was the case elsewhere (Table 4b). Detectable amounts of dieldrin and endrin were found in horizons from the late 1970 to mid 1980s for sediments collected at K-1. However, for the pesticides (and hydrocarbons), only four horizons from each of the two cores were analyzed so that, while we are able to identify contamination, we do not have a clear picture of the historical trends.

The PAHs and alkanes show considerable contamination in Kamloops Lake. For total parent PAH, the background concentrations are probably around 35 ng/g. In the upper levels of both cores, total PAH was found to be in the range of 100-250 ng/g (n=6). Likewise, the total n-alkanes are two to five times higher in the upper levels of the cores, and the alkyl PAHs also clearly show an enrichment in recent sediments. These contaminant hydrocarbons could come from a variety of sources including beehive burners and hog fuel burning at the mill, inputs of petroleum hydrocarbons from various sources including Kamloops, and from disturbance of the drainage basin above Kamloops Lake. Further analyses of hydrocarbons in Kamloops Lake together with an examination of PAH profiles using multivariate techniques may resolve these sources as will analyses of Thompson River sediments above Kamloops Lake.

4.5 Nicola Lake (Figure 28)

The ^{210}Pb profile for Nicola Lake shows a fair amount of scatter (Figure 29), but there is an exponential trend with depth and the calculated sedimentation rate appears to be reasonably well constrained (Table 3). We did not have enough sample to run a complete ^{137}Cs profile (Figure 30); nevertheless, the sedimentation rate estimated by ^{137}Cs date of entry agrees well with the ^{210}Pb estimate (Figure 30, Table 3).

With the exception of Pb, Nicola Lake does not show any noteworthy contamination by metals (Hg was not analyzed in this core). The naturally low Pb concentrations in Nicola Lake sediments (about 8 μg/g) show only minor contamination of an extra 1 to 3 μg/g Pb with the strongest contaminant inputs after about 1940 (Figure 31). As with the other lakes, the Pb profile is understandable as a general environmental Pb contamination due to leaded gasoline use and this Pb may have derived from cars using the road adjacent to the lake or from more distant, and heavier-trafficked areas.

Nicola Lake has very clearly been contaminated by organochlorine pesticides, the most striking example being DDT (Figure 32) for which Nicola Lake sediments exhibit the greatest DDT contamination, exceeding all other lakes studied by at least two orders of magnitude (Table 4b). The DDT appears to be partially degraded (i.e., it includes substantial quantities of DDE and DDD) and the date of entry of DDT into this lake appears to have been during the late 1970s to mid 1980s. These dates are significantly later than the known use of DDT for aerial spraying in BC (1946-1962 [Prebble, 1975 #1378]). Based on the total amount of DDT in the sediments at this site, and assuming that about half of the lake’s area is presently accumulating sediments, the sediment inventory of DDT is very approximately estimated at 50 kg. The anomalous amount of DDT and the timing of its entry into the lake lead us to conclude that it derives from local use or disposal rather than long-range atmospheric transport. The sediment profile suggests that the DDT may have come from two local applications, one in the late 1970s and one in the mid 1980s.

Of the other pesticides, HCH shows the clearest record; most of the contaminant enters the sediments after about 1970, peaks in the 1980s and declines in the most recent sample analyzed (Figure 33). This history is consistent with the known emission history and the recent atmospheric trends for HCH [Jantunen, 1995 #999]. PCB is also found in Nicola Lake sediments, but in very low concentrations consistent with long-range atmospheric transport.

PAH exhibits increased sediment concentrations during the 1900s with maximal values achieved during the 1950s (Figure 34). Since the 1950s, the PAH concentration has decreased to values not much different than the pre-industrial levels. We have not examined the PAH data in detail to determine likely sources, but the trend in the Nicola Lake sediment is similar to those observed elsewhere in western North America [Wakeham, 1980 #381; Carpenter, 1989 #552]. It has previously been suggested that observed trends in PAH can be explained by increasing use of coal up until the 1950s at which time coal was substituted with liquid fuels for domestic use (e.g., see [Christensen, 1993 #893]. The total alkanes also show elevated concentrations toward the surface of the Nicola Lake core (Figure 35), but the trend with time clearly differs from that of total PAH. As in the case of Kamloops Lake, the total alkane enrichments in recent Nicola Lake sediments could be supported by local petroleum inputs, runoff from highways and from human activities (e.g., logging, farming) in the Nicola drainage basin. On the other hand, they could reflect diagenetic loss of labile or metabolizable alkanes as the sediments age.

4.6 Harrison Lake (Figure 36)

The ^{210}Pb data show scatter (Figure 37) but the sedimentation rate appears reasonably well constrained according to the confidence intervals on the regression slope (Table 3). We do not have sediment laminae or ^{137}Cs against which to evaluate these sedimentation rates but the contaminant histories, outlined below, seem to be consistent with the derived sediment rate.

Pb shows recent sediment contamination (Figure 38) similar to many of the other lakes. The contaminant Pb is first detecable after the 1940s, peaks in the region corresponding to 1970 to about 1985 and shows recent declines. Again, this is consistent with the leaded gasoline source. Surprisingly, Cr and Ni (Figure 39) also show slight enrichments in the lake sediments after about 1970 but we have no idea what the source might be.

Contamination of Harrison Lake sediments by organochlorine compounds appears to be slight, but detectable, and trends are consistent with long-range atmospheric transport. For example, total DDT (Figure 40) shows a classic envelope, entering the sediments probably in the late 1940s or early 1950s, maximizing in the early 1960s when most of the aerial spraying occurred in various parts of BC [Prebble, 1975 #1378], and then decreases to the present. Likewise, PCB (Figure 41) enters sediments somewhat earlier than DDT and reaches a maximum later, in the mid to late 1970s. Other organochlorines, such as HCH and HCB are also detected in sediments after 1950 but the concentrations are very low.

The hydrocarbon trends in the Harrison Lake sediments (Figures 42 and 43) are reminiscent of those in Nicola Lake (Figures 35, 36). PAH in sediments increases as early as horizons dated at the mid 1880s; it reaches a maximum during the 1950s and thereafter decreases to the present. Concentrations of PAH near the surface of the core are significantly elevated above the background deep in the core (Table 4c). The source of the PAH contributing to these increases is not clear. It could come from the substitution of coal for liquid fuels in the 1950s, but it could also derive in part from local activities associated with logging such as beehive burners, and slash burning. The relatively early entry of higher concentrations of PAH into the Harrison Lake sediments, before the turn of the century, suggests the likelihood of long-standing, local sources. The total alkane profile, which parallels that observed in Nicola Lake sediments, differs from PAH in that it shows detectable sediment elevations only after about 1940, and the concentrations increase linearly with time since then. As noted earlier, this could reflect enhanced inputs or degradation of alkane with time in the accumulating sediments.

5.0 Conclusions

The chemical records measured in dated sediment cores collected in six lakes of the Fraser drainage basin illustrate that these lakes have received contaminants from both long-range transport and from local sources. In general, the lake sediments can be considered relatively pristine and contaminant levels are extremely low except where local sources have impacted a particular lake (see Tables 4, 5, 6 for a summary of the background and contaminant concentrations in the lake sediments). Of the two sources - long range and local - it is the local sources, when they occur, that are of far greater signficance.

Pb exhibits an ubiquitous contamination that follows the history of Pb use as an anti-knock agent in automobiles. How far this Pb may have travelled before entering a particular lake is difficult to assess because almost all of the lakes have highways or roads running along their margins. In addition, it is likely that Pb has entered lakes partly from the atmosphere and partly from indirect washout off roads within the various watersheds. Stuart Lake is exceptional in showing a strong Hg contamination event which, according to our dating, derives from a Hg mine operated at Pinchi Lake in 1940-1944. This mine disposed much of its tailings into Pinchi Lake, and it is likely that contaminant Hg was delivered to Stuart Lake via Pinchi Creek which drains into it. Hg also appears as a detecable contaminant in Kamloops Lake, however the source is as yet unknown. There is little evidence of contamination by other metals in the various lakes with the exception of Zn and Ni in Moose Lake and Cr and Ni in Harrison Lake. Sources for these contaminants are presently unknown and under investigation.

The organochlorine contaminants are found as trace contaminants in virtually all of the lakes (Table 4b). The history of contamination, as derived from the dated sediment cores, is consistent with the known emission histories both in North America and globally. These low concentrations of contaminants, therefore, probably derive from a combination of sources incorporating both local and long-range transport. However, in addition to this low, ubiquitous contamination, there appear to have been at least two significant local sources of organochlorine contamination. As discussed elsewhere [Macdonald, 1997 #1382], a pulpmill has introduced large quantities of PCDD/Fs into Kamloops Lake, particularly during the 1970s and 1980s. Technical changes at the mill have significantly reduced this source during the 1990s. In Nicola Lake a large amount of DDT entered the sediments in the late 1970s to mid 1980s. The timing of entry and amount of

DDT in the sediments set this lake apart from the others, and suggest very strongly that the DDT derives from local, not regional or global, use.

Background concentrations of hydrocarbon vary widely between lakes (Table 4c). As shown by the figures, a case can be made for PAH contamination in Kamloops, Nicola and Harrison Lakes where the PAH contamination is identified by a maximum in the 1950s. In contrast to PAH, alkanes enrichments in sediments can be detected after about 1950 and the alkane concentrations continue to increase linearly with time up to the present. However, it is not yet clear whether these alkane enrichments stem from increased inputs of petrogenic hydrocarbon or whether they are simply the apparent result of diagenetic loss with depth (time) in the core. Although shifting from coal to liquid fuels for domestic heating during the 1950s has been inferred as a likely cause of observed PAH maxima in sediments dating from that decade, in the cases of strongest hydrocarbon contamination (Kamloops Lake, Harrison Lake, Nicola Lake) we also cannot discount inputs from local sources such as wood burning associated with the logging and pulp and paper industries. Efforts to determine these various sources by using compound distribution together with multivariate techniques is in progress.

6.0 References

Addison, R.F., D.E. Willis, The metabolism by rainbow trout (Salmo gairdnerii) of p,p'[14C]DDT and some of its possible degradation products labeled with 14C, *Toxicology and Applied Pharmacology, 43*, 303-315, 1978.

Carpenter, R., and L. Peterson, Chemical cycling in Washington's coastal zone, in *Coastal Oceanography of Washington and Oregon*, edited by M.R. Landry, and B.M. Hickey, pp. 367-509, Elsevier Science Publishers, Amsterdam, 1989.

Christensen, E.R., and X. Zhang, Sources of polycyclic aromatic hydrocarbons to Lake Michigan determined from sedimentary records, *Environmental Science and Technology*, *27* (1), 139-146, 1993.

Desloges, J.R., Gilbert, R., The sedimentary record of Moose Lake: implications for late-glacial and Little Ice Age glacier activity in the Mt. Robson area, British Columbia, *Canadian Journal of Earth Sciences*, *32*, 65-78, 1995.

Desloges, J.R., and R. Gilbert, Sedimentary record of Harrison Lake: implications for deglaciation in southwestern British Columbia, *Canadian Journal of Earth Sciences*, *28*, 800-815, 1991.

Donald, D.B., R. Bailey, R. Crosley, D. Muir, P. Shaw, and J. Syrgiannis, Polychlorinated biphenyls and organochlorine pesticides in the aquatic environment along the continental divide region of Alberta and British Columbia, Inland Waters Directorate, 1993.

Eakins, J.D., and R.T. Morrison, A new procedure for the determination of lead-210 in lake and marine sediments, *International Journal of Applied Radiation and Isotopes*, *29*, 531-536, 1978.

Eisenreich, S.J., P.D. Capel, J.A. Robbins, and R. Bourbonniere, Accumulation and diagenesis of chlorinated hydrocarbons in lacustrine sediments, *Environmental Science and Technology*, *23*, 1116-1126, 1989.

Jantunen, L.M., and T.F. Bidleman, Reversal of the air-water gas exchange direction of hexachlorocyclohexanes in the Bering and Chukchi Seas, *Environmental Science and Technology*, *29*, 1081-1089, 1995.

Kidd, K.A., D.W. Schindler, D.C.G. Muir, W.L. Lockhart, and R.H. Hesslein, High concentrations of toxaphene in fishes from a subarctic lake, *Science*, *269*, 240-242, 1995.

Maatela, P., J. Paasivirta, M.A. Grachev, and E.B. Karabanov, Organic chlorine compounds in lake sediments. V. Bottom of Baikal near a pulp mill., *Chemosphere*, *21* (12), 1381-1384, 1990.

Macdonald, R.W., E.C. Carmack, and C.H. Pharo, Sediment records of man's activities in the Kootenay Lake drainage basin, *Water Pollution Research Journal of Canada*, *29* (1), 103-116, 1994.

Macdonald, R.W., M.G. Ikonomou, and D.W. Paton, Historical inputs of PCDDs, PCDFs and PCBs to a British Columbia interior lake: the effect of environmental controls on pulpmill emissions, *Environmental Science and Technology*, *In Press*, 1998.

Macdonald, R.W., D.M. Macdonald, M.O. O'Brien, and C. Gobeil, Accumulation of heavy metals (Pb, Zn, Cu, Cd), carbon and nitrogen in sediments from Strait of Georgia, B.C., Canada, *Marine Chemistry*, *34*, 109-135, 1991.

Muir, D.C.G., A. Omelchenko, N.P. Grift, D.A. Savoie, W.L. Lockhart, P. Wilkinson, and G.J. Brunskill, Spatial trends and historical deposition of polychlorinated biphenyls in Canadian midlatitude and arctic lake sediments, *Environmental Science and Technology*, *30*, 3609-3617, 1996.

Müller, G., G. Grimmer, and H. Böhnke, Sedimentary record of heavy metals and polycyclic aromatic hydrocarbonin Lake Constance, *Naturwissenschaffen*, *64*, 427-431, 1977.

Oliver, B.G., M.N. Charlton, and R.W. Durham, Distribution, redistribution, and geochronology of polychlorinated biphenyl congeners and other chlorinated hydrocarbons in Lake Ontario sediments, *Environmental Science and Technology*, *23* (2), 200-208, 1989.

Pharo, C.H., and E.C. Carmack, Sedimentation processes in a short residence-time intermontane lake, Kamloops Lake, British Columbia, *Sedimentology*, *26*, 523-541, 1979.

Prebble, M.L., *Aerial Control of Forest Insects in Canada*, 330 pp., Department of the Environment, Ottawa, 1975.

Renberg, I., M.W. Persson, and O. Emteryd, Pre-industrial atmospheric lead contamination detected in Swedish lake sediments, *Nature*, *368*, 323-326, 1994.

Sanders, G., K.C. Jones, J. Hamilton-Taylor, and H. Dörr, Historical inputs of polychlorinated biphenyls and other organochlorines to a dated lacustrine sediment core in rural England, *Environmental Science and Technology*, *26* (9), 1815-1821, 1992.

St. John, B.E., E.C. Carmack, R.J. Daley, C.B.J. Gray, and C.J. Pharo, The limnology of Kamloops Lake, B.C., Inland Waters Directorate, Vancouver, 1976.

Vilanova, R., P. Fernandez, J.O. Grimalt, P. Appleby, N. Rose, and R.W. Battarbee, The use of high mountain lake sedimentary records for the study of European environmental change, in *Organic geochemistry: developments and applications to energy, climate, environment and human history, 17th International Meeting on Geochemistry*, edited by J.O. Grimalt, and C. Dorronsoro, pp. 575-577, AIGOA, Spain, 1995.

Wakeham, S.G., C. Schaffner, and W. Giger, Polycyclic aromatic hydrocarbons in recent lake sediments - I. Compounds having anthropogenic origins, *Geochimica et Cosmochimica Acta*, *44*, 403-413, 1980.

Zinck, M.E., R.F. Addison, The effect of temperature on the rate of conversion of p,p'-DDT to p,p'DDE in brook trout (Salvelinus fontinalis), *Canadian Journal of Biochemistry*, *53*, 636-639, 1975.

TABLES

Table 1 **Hydrographical and limnological characteristics of the lakes.**

	Moose	Stuart	Chilko	Kamloops	Nicola	Harrison
Area (km^2)	13.9	358	200	52.1	24.9	510
Volume (km^3)	0.73	9.3	23	3.70	0.57	80.5
Mean Depth (m)	52	26	137	71	23.5	158
Maximum Depth (m)	87	95	366	143	54.9	270
Bulk residence time (yr)	1.6	2.3	17	0.16	3.0	5.6
Discharge (km^3/yr)	0.46	4.1	1.3	22	0.19	14.3
Watershed area (km^2)	1640	14600	2110	40386	2990	7870
Lake Yield (m/yr)	33	11.5	7.7	422	7.6	28
Watershed Yield (m/yr)	0.28	0.28	0.62	0.54	0.06	1.8
Lake Elevation (m)	1032	680	1172	336	627	10
References	2,3,4,6	2,3,4	2,3,4,5	1,2,3	2,3,4,5	2,3,4,7

1. [Pharo, 1979 #407]
2. Provincial Maps giving Bathymetry, areas, volumes and hypsometry
3. Surface Water data (EN-36-407)
4. Historical Streamflow Summary (EN-36-418/7)
5. Provincial Lake file (Richard Dobrowski, pers. comm.).
6. [Desloges, 1995 #891]
7. [Desloges, 1991 #892].

Table 2 Coring locations, dates and depths.

Core	Latitude	Longitude	Depth (m)	Core Length (cm)	Date Collected
H-1	49 34.519N	121 52.980W	258	73	Nov 4/93
N-1	50 11.131N	120 29.899W	50	55	Nov 5/93
K-1	50 46.67N	120 45.76W	94	73	Jun 8/94
K-2	50 44.808N	120 40.611W	91	69	Jun 8/94
M-1	52 57.132N	118 54.395W	80	44	Jun 10/94
M-2	52 57.727N	118 56.225W	74	59	Jun 10/94
S-1	54 39.669N	124 50.873W	31	75	Jun 12/94
S-2	54 31.01N	124 32.01W	45	85	Jun 12/94
C-1	51 29.344N	124 12.054W	153	20	Jun16/94
C-3	51 26.650N	124 11.836W	162	41	Jun 17/94

Table 3 Sedimentation rates at the core sites.

Lake	Sed Rate	95% CI	Sed Rate	^{210}Pb Flux	Varves	^{137}Cs
	g/cm^2/yr	g/cm^2/yr	cm/yr	dpm/cm^2/yr	cm/yr	cm/yr
Moose-1	0.33	0.27 - 0.44	0.31	2.2	0.35	
Moose-2	0.20	0.18 - 0.24	0.214	1.9	0.26	
Stuart-1	0.040	0.033 - 0.050	0.16	0.59	n/a	
Stuart-2	0.030	0.026 - 0.035	0.13	0.67	n/a	0.14
Chilko-1	0.064	0.050 - 0.089	0.14	0.36	n/a	
Chilko-3	0.11? (slump)		0.22?	0.59	n/a	
Kamloops-1	0.26	0.19 - 0.38	0.51	1.8	0.45	
Kamloops-2	0.32	0.24 - 0.46	0.49	1.7	0.55	
Nicola-1	0.090	0.071 - 0.12	0.38	0.27	n/a	0.35
Harrison-1	0.18	0.15 - 0.22	0.34	1.5	n/a	

Table 4 a. **Summary of metal data for the lake sediments[1].** Units are Tg g^{-1} except Hg which is ng g^{-1}.

Metal (µg/g)	Moose	Stuart	Chilko	Kamloops	Nicola	Harrison
Pb (background)	31.0 ± 2.1	15.5 ± 0.86	13.2 ± 2.8	26.3 ± 2.6	8.4 ± 0.6	17.1±1.2(8)
Pb (maximum)	**38.1**	**22.3**	20	**37.5**	**11.3**	**26.5**
Zn (background)	86.7 ± 6.4	119 ± 9	164 ± 15	147 ± 13	87.0 ± 6.6	182 ± 10
Zn (maximum)	**130**	136	188	173	103	**213**
Cu (background)	35.0 ± 1.9	47.6 ± 4.7	83.8 ± 8.5	62.3 ± 5.1	77.4 ± 6.9	105 ± 6
Cu (maximum)	**50.3**	**65.9**	92.6	69.9	89.8	**124**
Ni (background)	41.8 ± 4.6	91 ±8	46.0 ± 11.7	79.6 ± 8.5	45.8 ± 4.3	42.5 ± 1.2
Ni (maximum)	**134**	**117**	63	89.9	52.9	**59.5**
Co (background)	17.9 ± 1.2	17.0 ± 0.7	27.7 ± 3.5	30.0 ± 2.5	19.3 ± 1.7	28.2 ± 1.4
Co (maximum)	**32.2**	**22.3**	32.3	35	22.5	29.5
Cr (background)	113 ± 4	78.5 ± 8.5	84.0 ± 7.5	151 ± 10	106 ± 4	79.4 ± 4.2
Cr (maximum)	**140**	102	102	169	**132**	**110**
Hg (background)	12.5 ± 1.3	19.4 ± 3.3	29.6 ± 7.5	20 ± 3		
Hg (maximum)	**24**	**158**	42	**52**		

1. Background values were calculated as the average of the bottom 3-4 samples for the cores (± 1 sd). Bold numbers identify where a case can be made for significant contamination based on the metals profile in the core.

Table 4 b. Summary of organochlorine contaminant maximum concentrations for the lake sediments.

Compound ng/g	Moose	Stuart	Chilko	Kamloops	Nicola	Harrison
number of samples	18	5	5	8	11	12
HCB	0.2	0.44	0.4	0.17	1.5	0.06
Total HCH	0.14	0.09	0.13	0.29	6.1	0.06
Heptachlor	nd	nd	nd	nd	nd	nd
Aldrin	nd	nd	nd	nd	0.14	nd
Chlordane	0.01	0.07	0.02	nd	3.1	0.06
Total DDT	1.73	1.25	0.32	1.95	323	8.7
Total Nonachlor	0.04	0.06	0.06	nd	1.4	0.05
Mirex	0.007	0.008	0.01	nd	0.25	nd
Dieldrin	0.06	nd	nd	0.92	nd	nd
Endrin	nd	nd	nd	2.6	nd	nd
Methoxychlor	nd	nd	nd	nd	nd	nd
Toxaphene	nd	nd	0.24	nd	nd	nd
Total PCB	1.6	0.68	0.94	7.7	1.49	1.5

Table 4 c. Summary of hydrocarbon data for the lake sediments[1].

Compound ng/g	Moose	Stuart	Chilko	Kamloops	Nicola	Harrison
number of samples	18	5	5	8	11	12
Background Total PAH	24.5 ± 20	125 (est)	29 ± 8	35 (est)	46 ± 20	7 ± 2
Highest PAH	**45**	209	38	**245**	**109**	**201**
Background Total Alkane	1155 ±320	7500 (est)	1600 ± 890	3800 (est)	11250 (est)	1790 ±270
Highest Total Alkane	2053	8943	3170	19300	20600	4880

1. Background values were calculated as the average of the bottom 3-4 samples for the cores (± 1 sd). Bold numbers identify where a case can be made for significant contamination based on the hydrocarbon profile in the core.

FIGURES

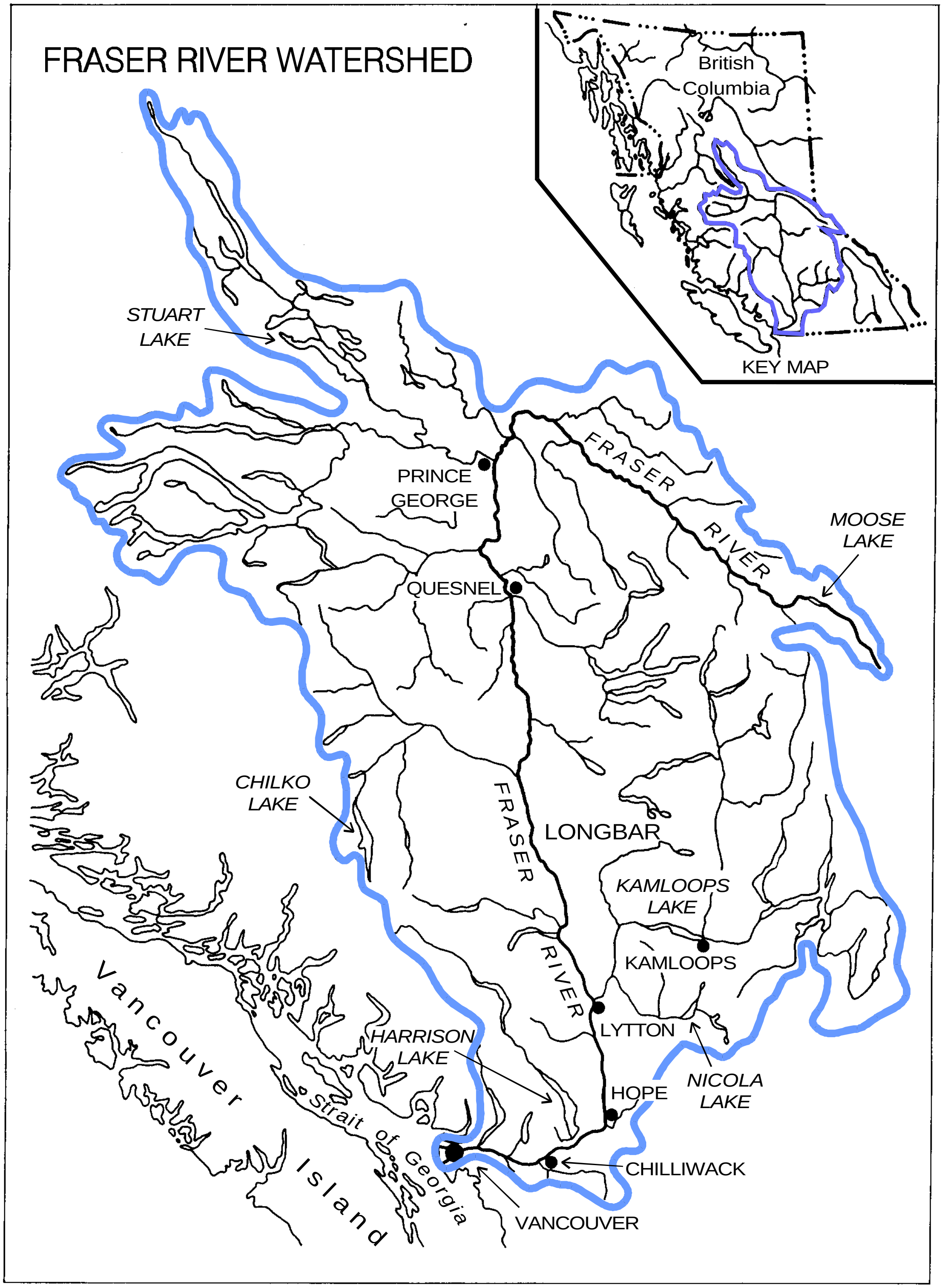

Figure 1. **The location of the six Fraser Basin lakes studied.**

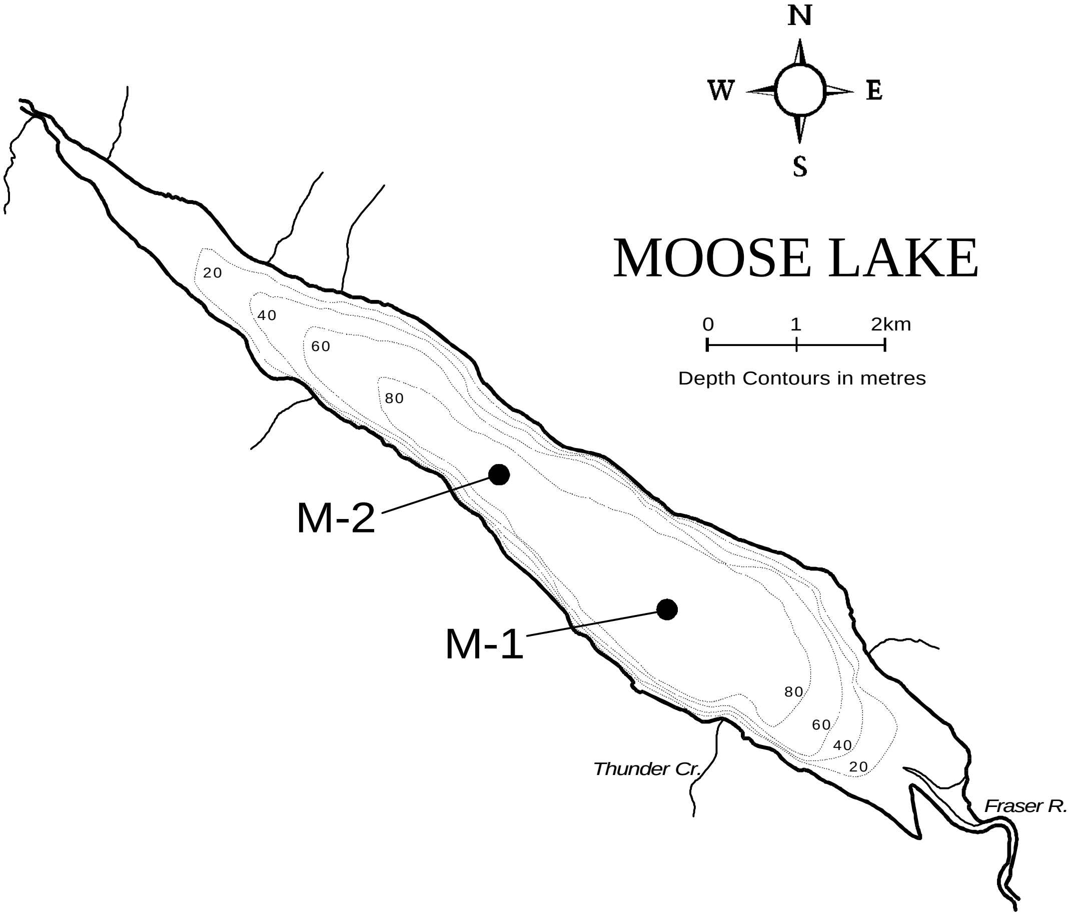

Figure 2. **Moose Lake sampling locations.**

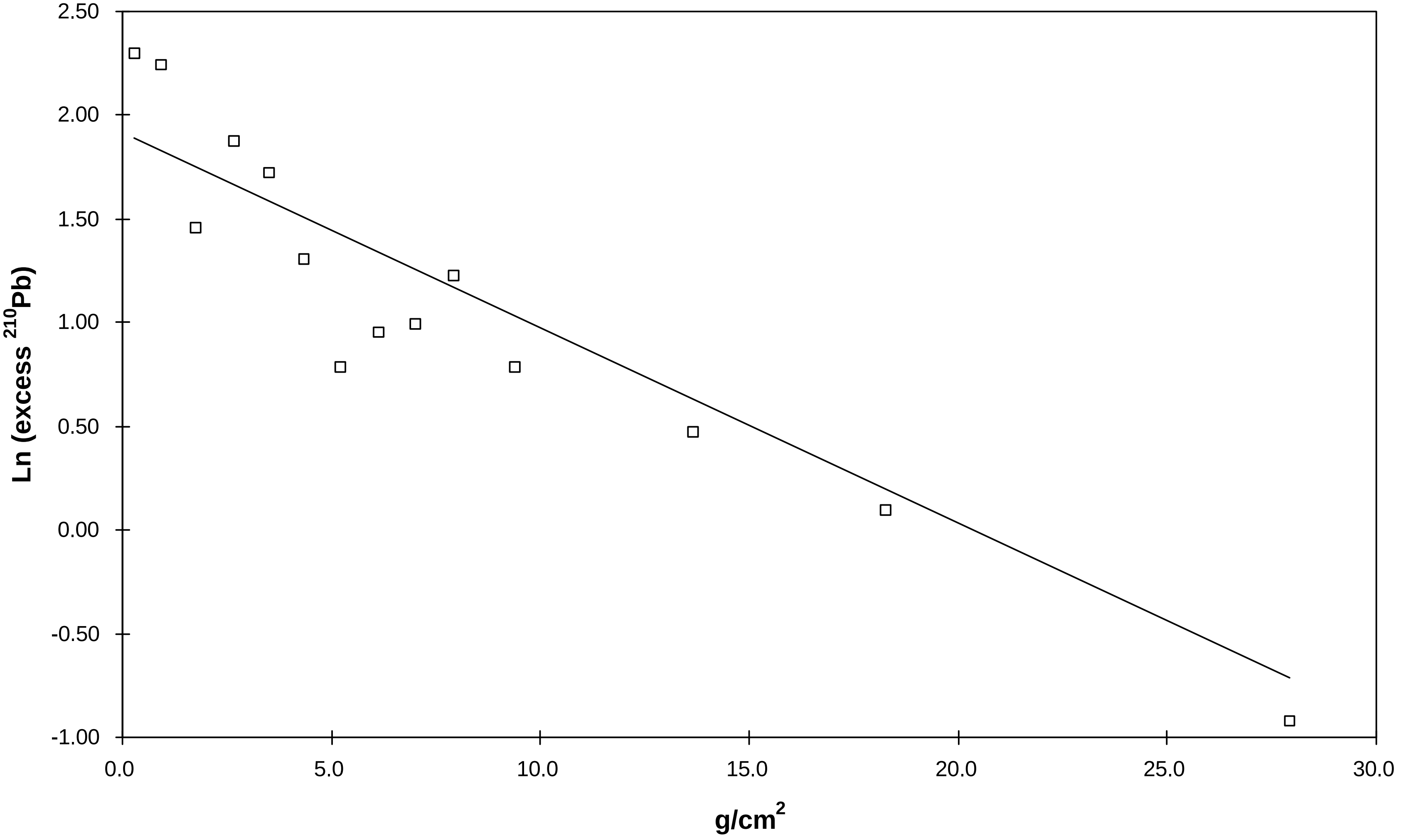

Figure 3. ^{210}**Pb plot for location Moose-1.**

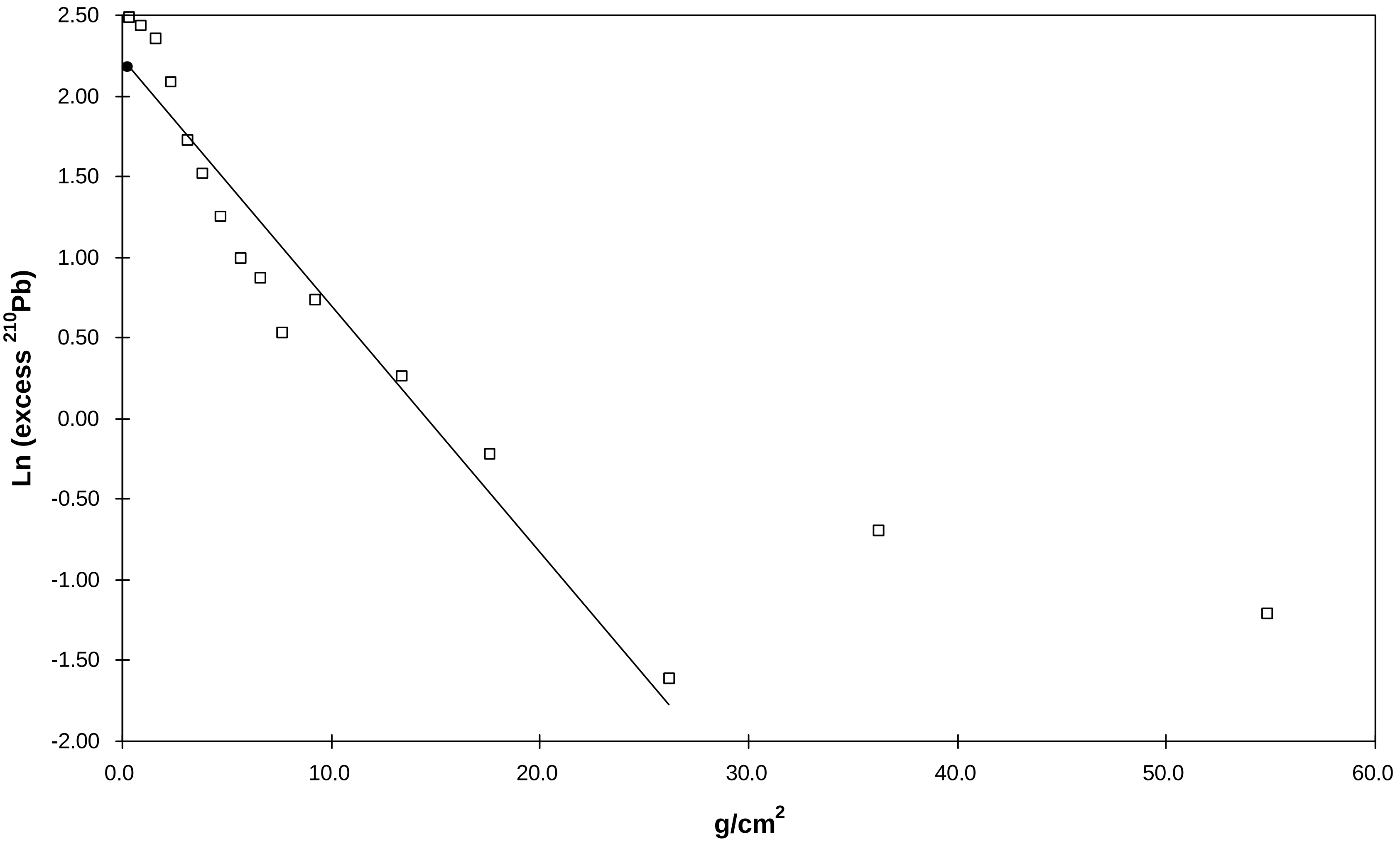

Figure 4. ^{210}**Pb plot for location Moose-2.**

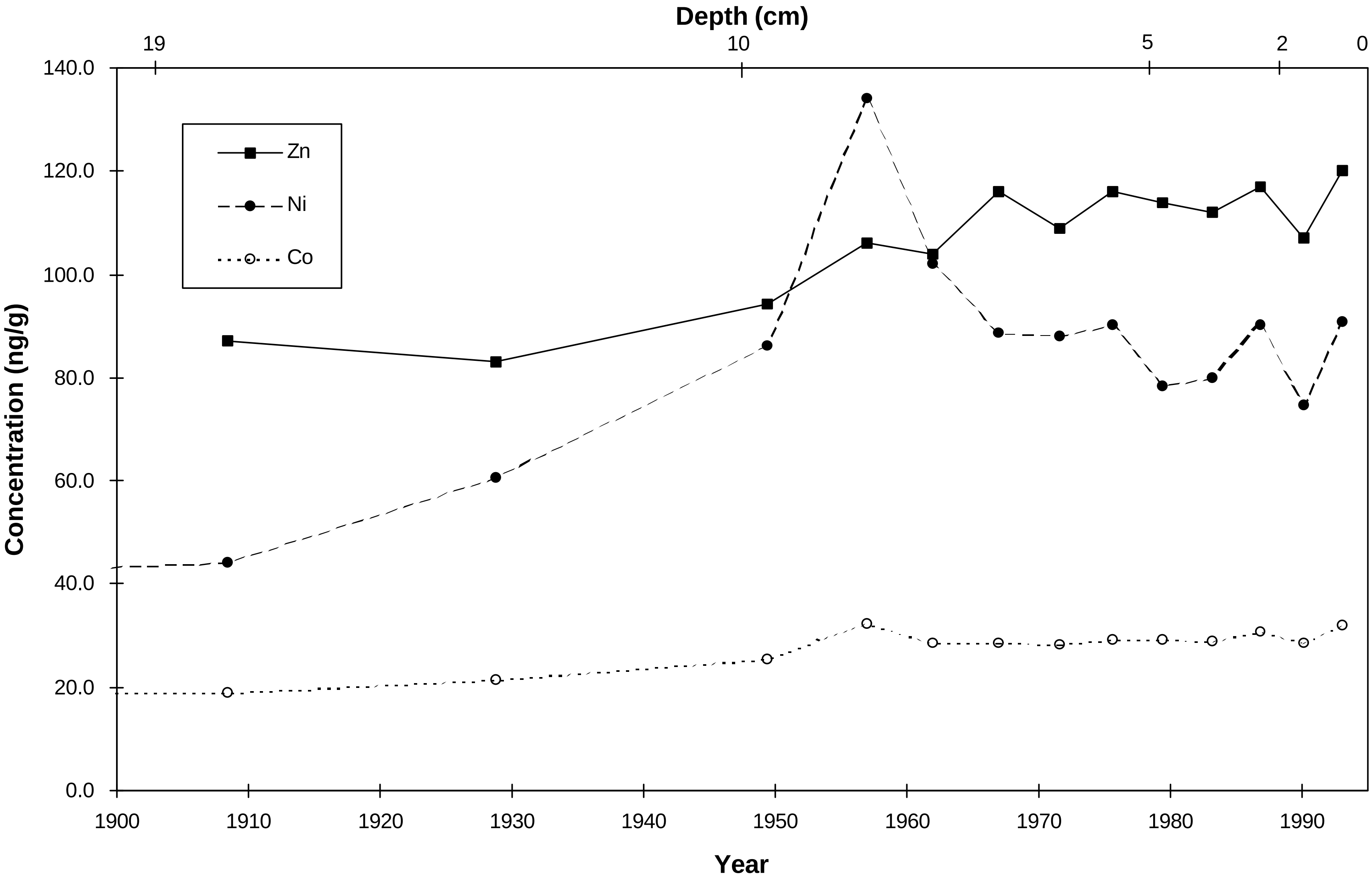

Figure 5. **Zn, Co and Ni trends with time in Moose-2 sediments.**

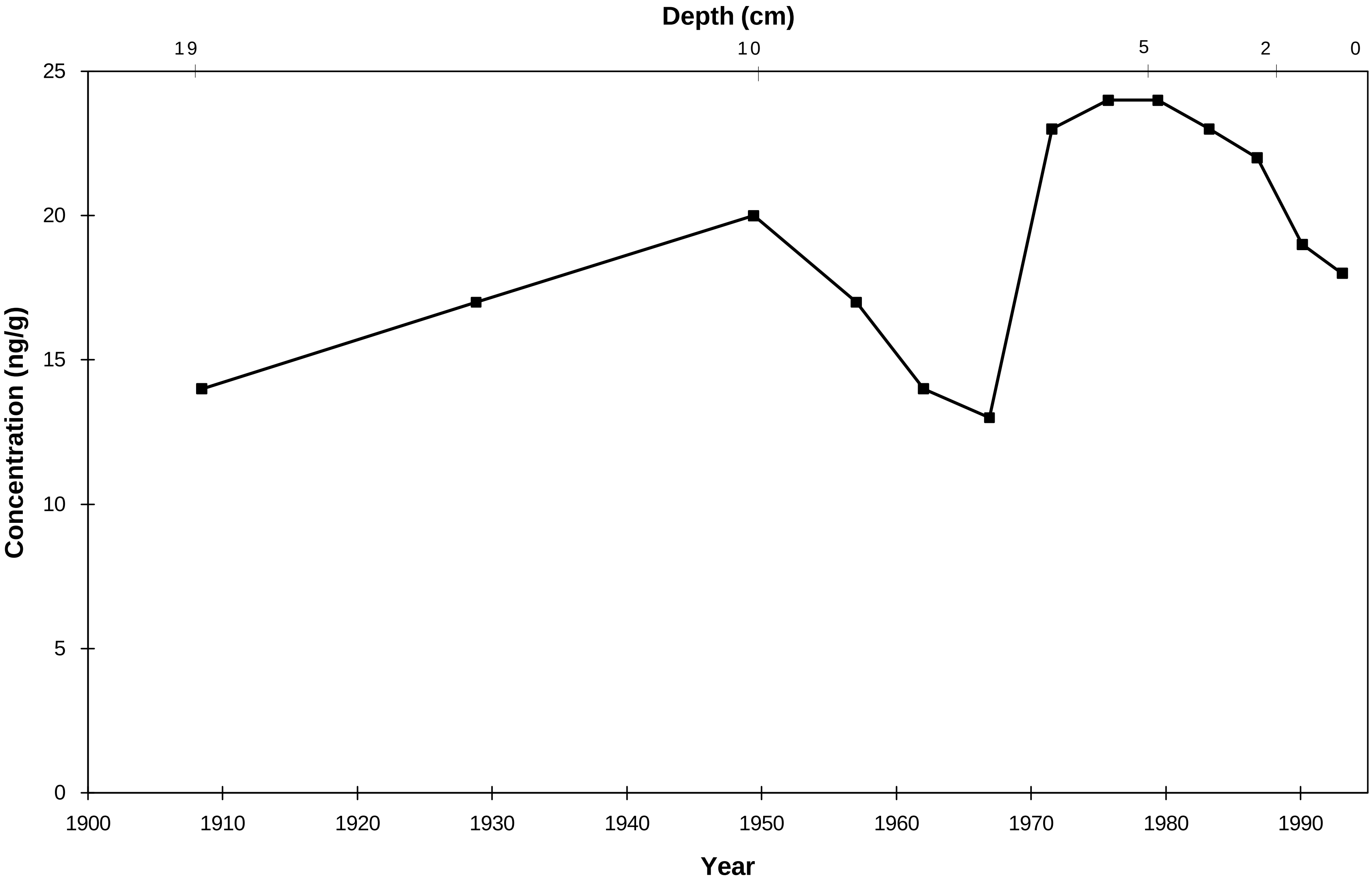

Figure 6. **Hg trend with time in Moose-2 sediments.**

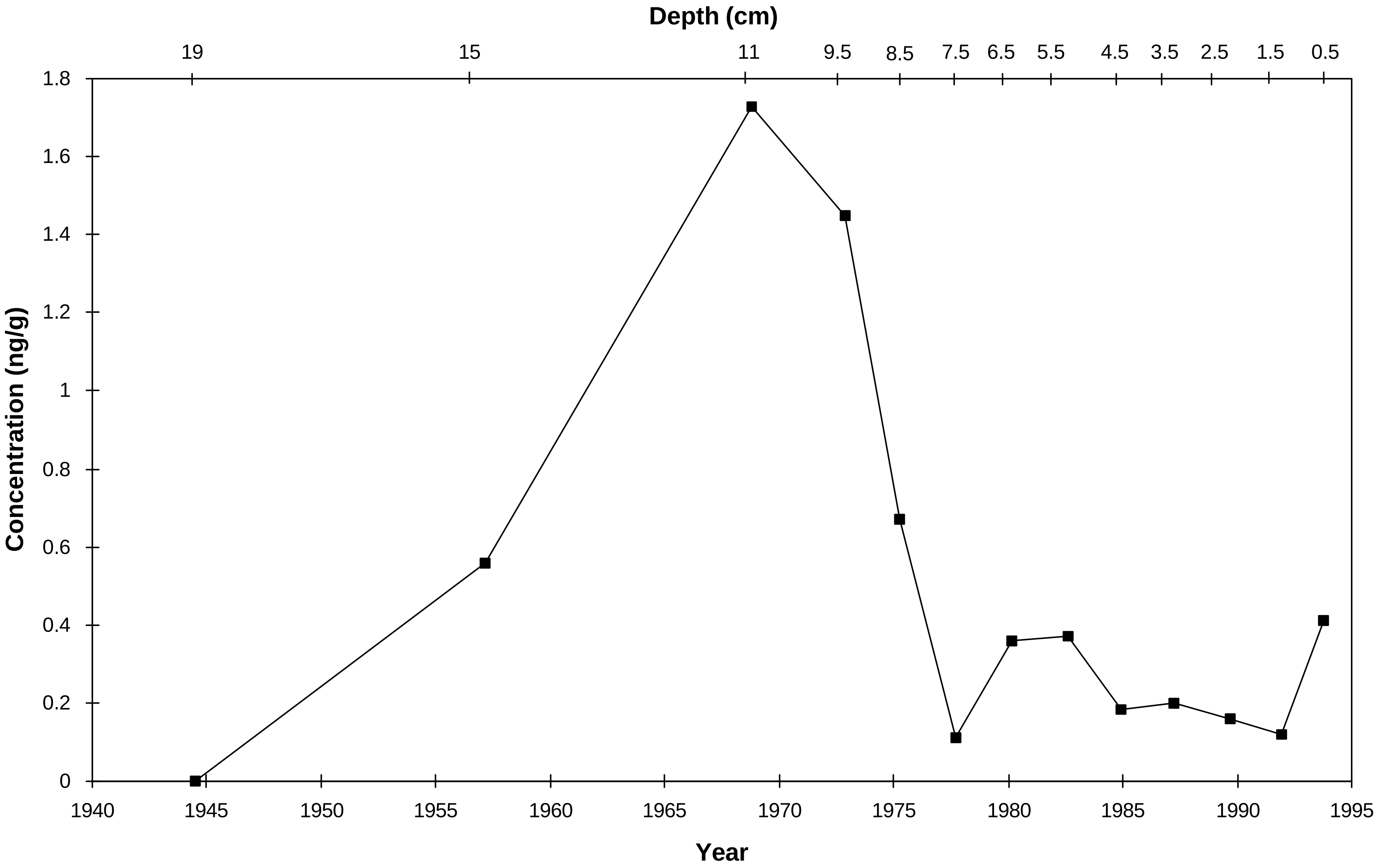

Figure 7. **Total DDT trend with time in Moose-1 sediments.**

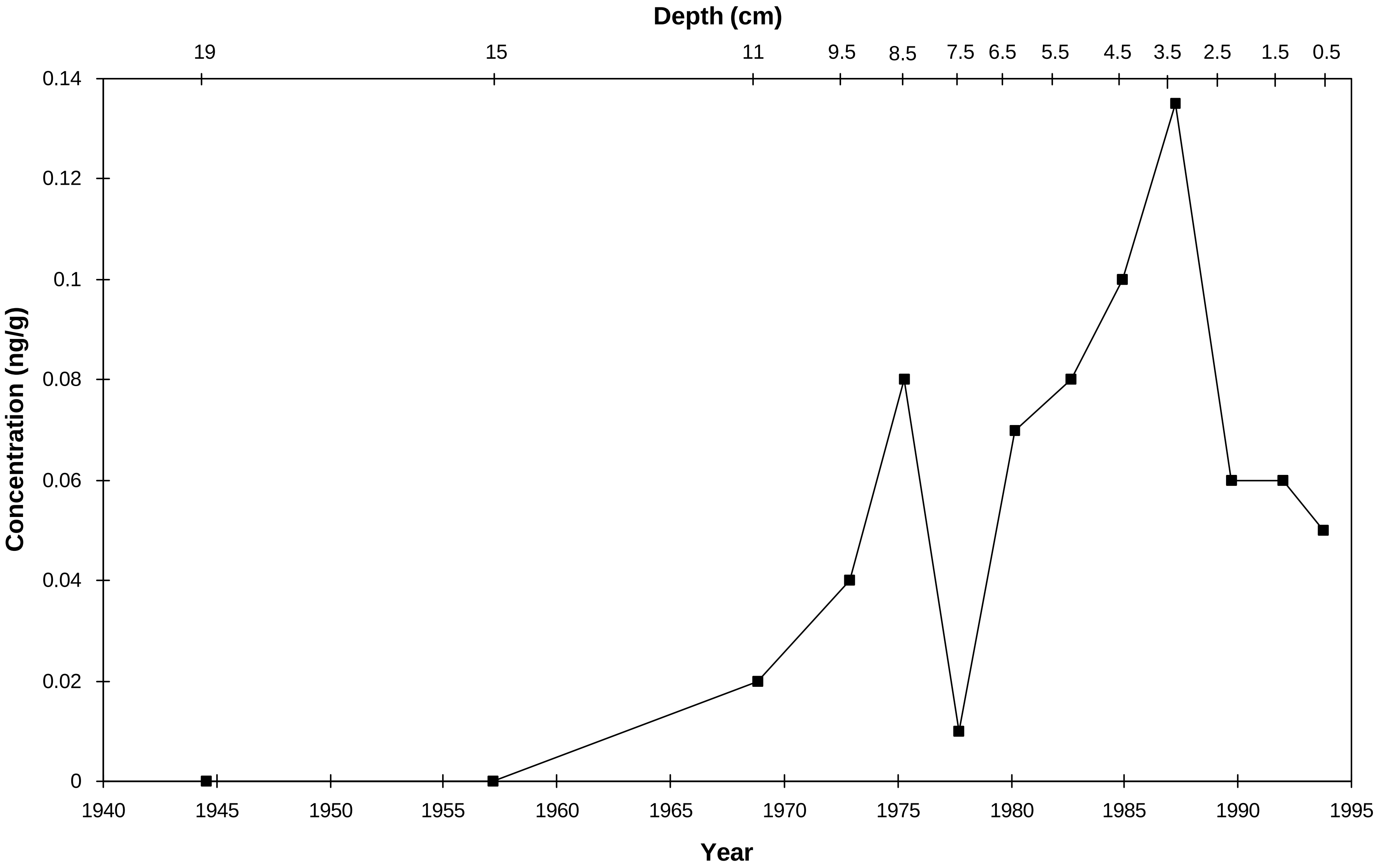

Figure 8. **Total HCH (alpha plus gamma) trend with time in Moose-1 sediments.**

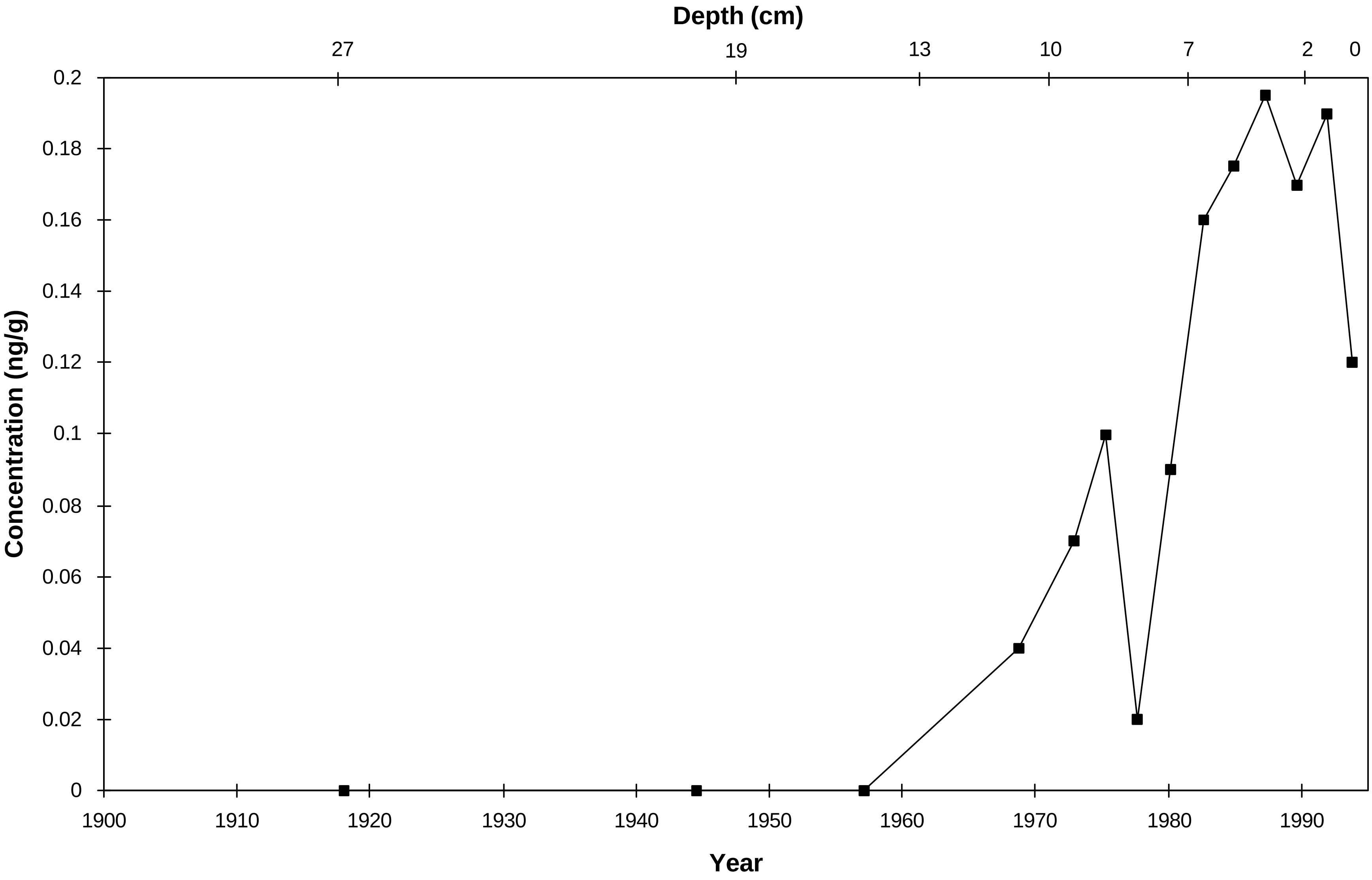

Figure 9. **HCB trend with time in Moose-1 sediments.**

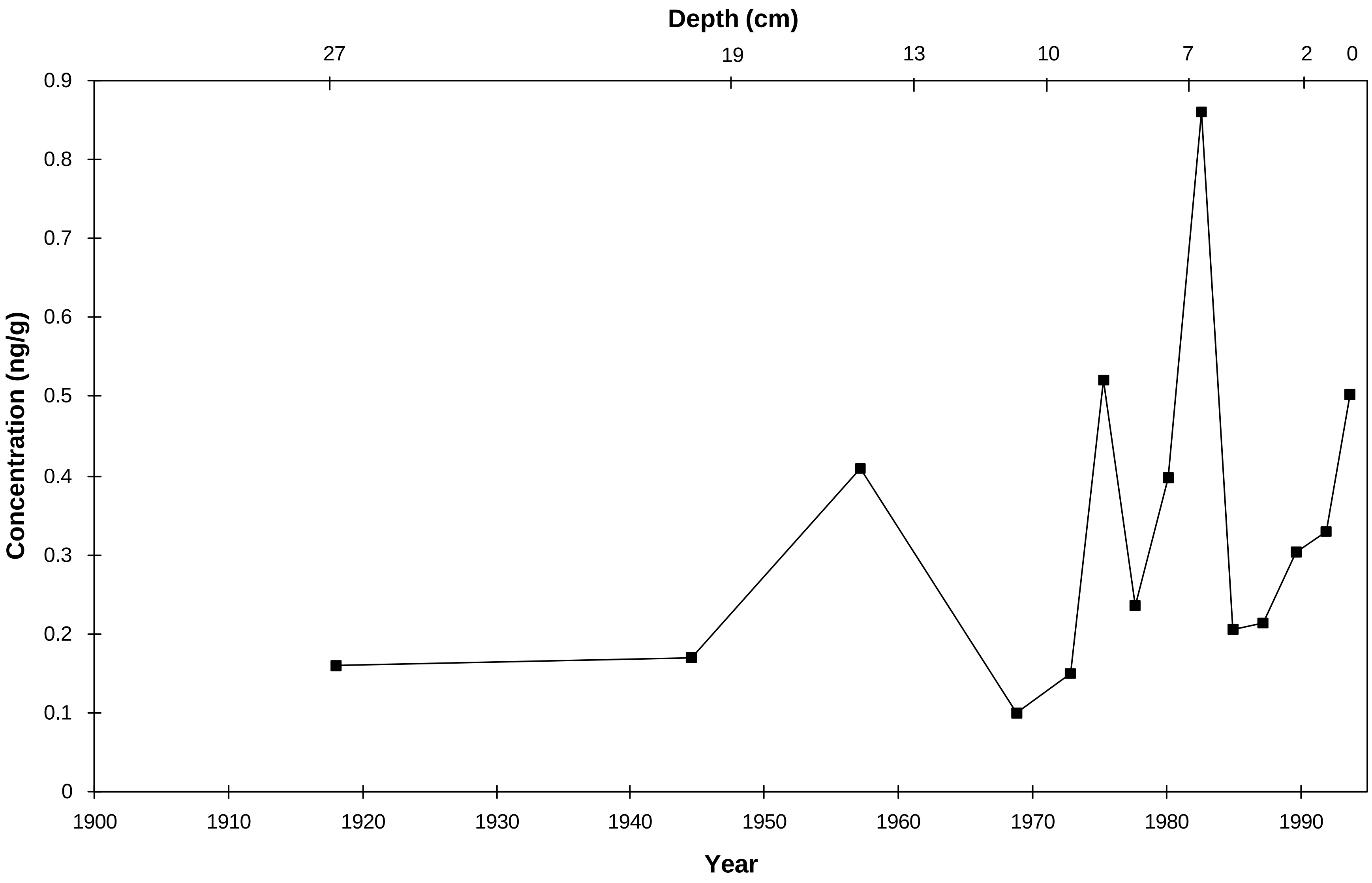

Figure 10. **Total PCB (sum congeners) with time in Moose-1 sediments.**

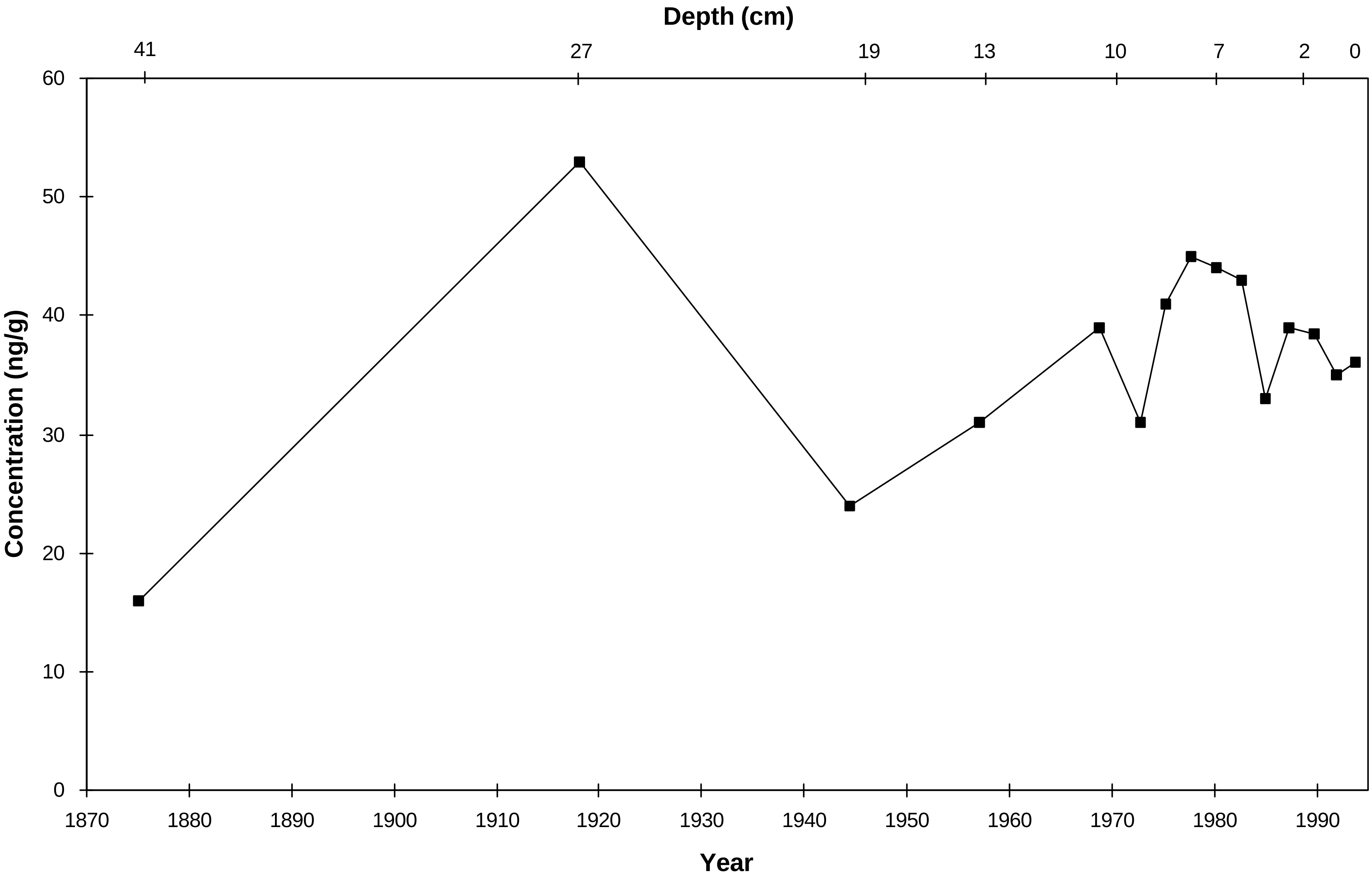

Figure 11. **Total PAH trend with time in Moose-1 sediments.**

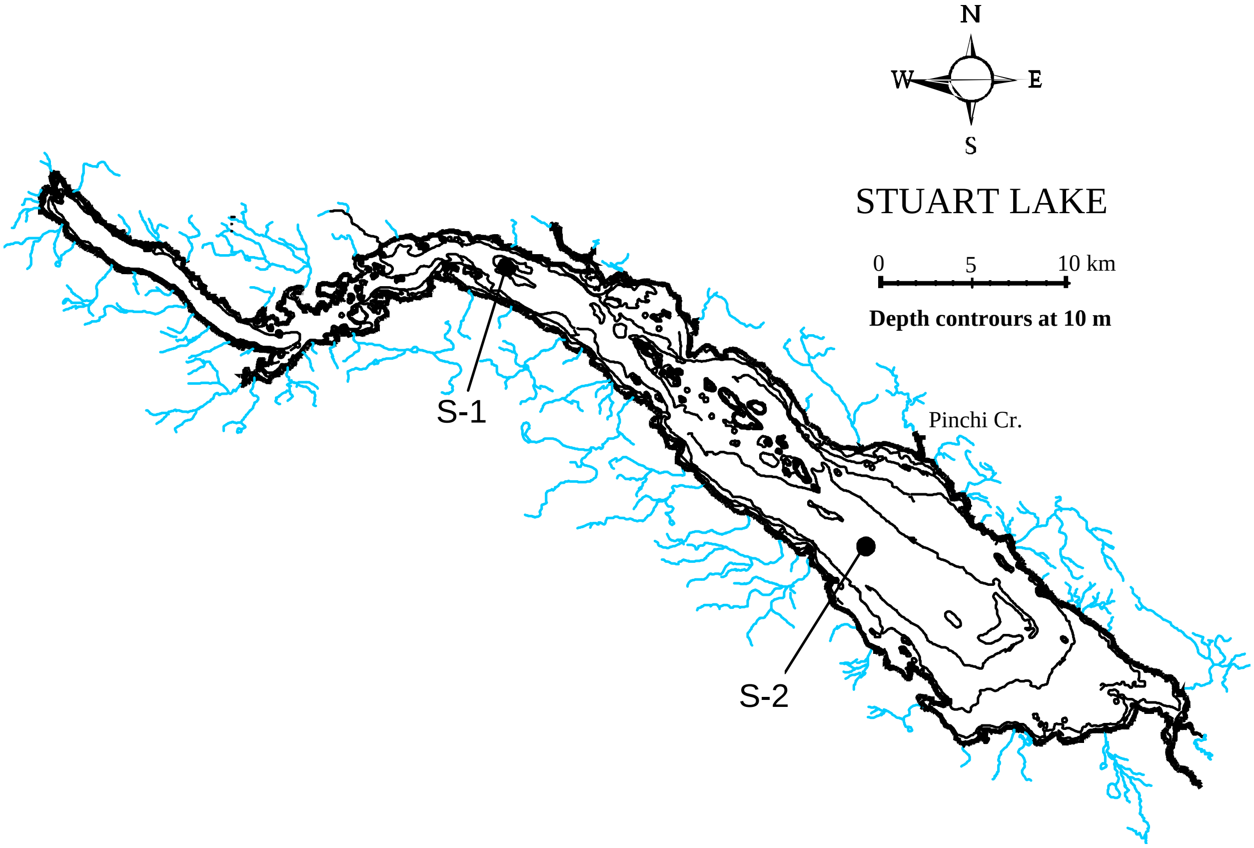

Figure 12. Stuart Lake sampling locations.

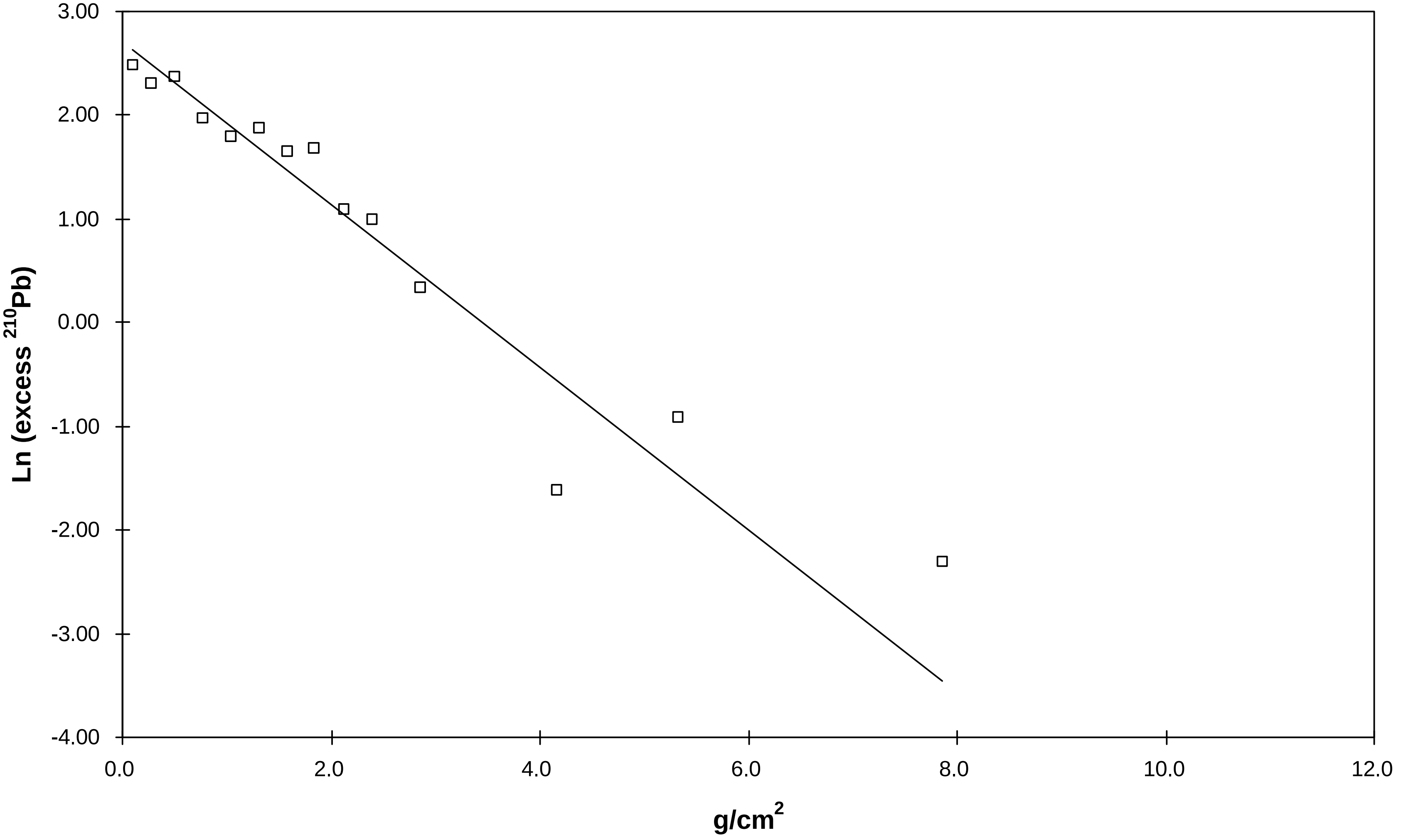

Figure 13. ^{210}Pb plot for location Stuart-1.

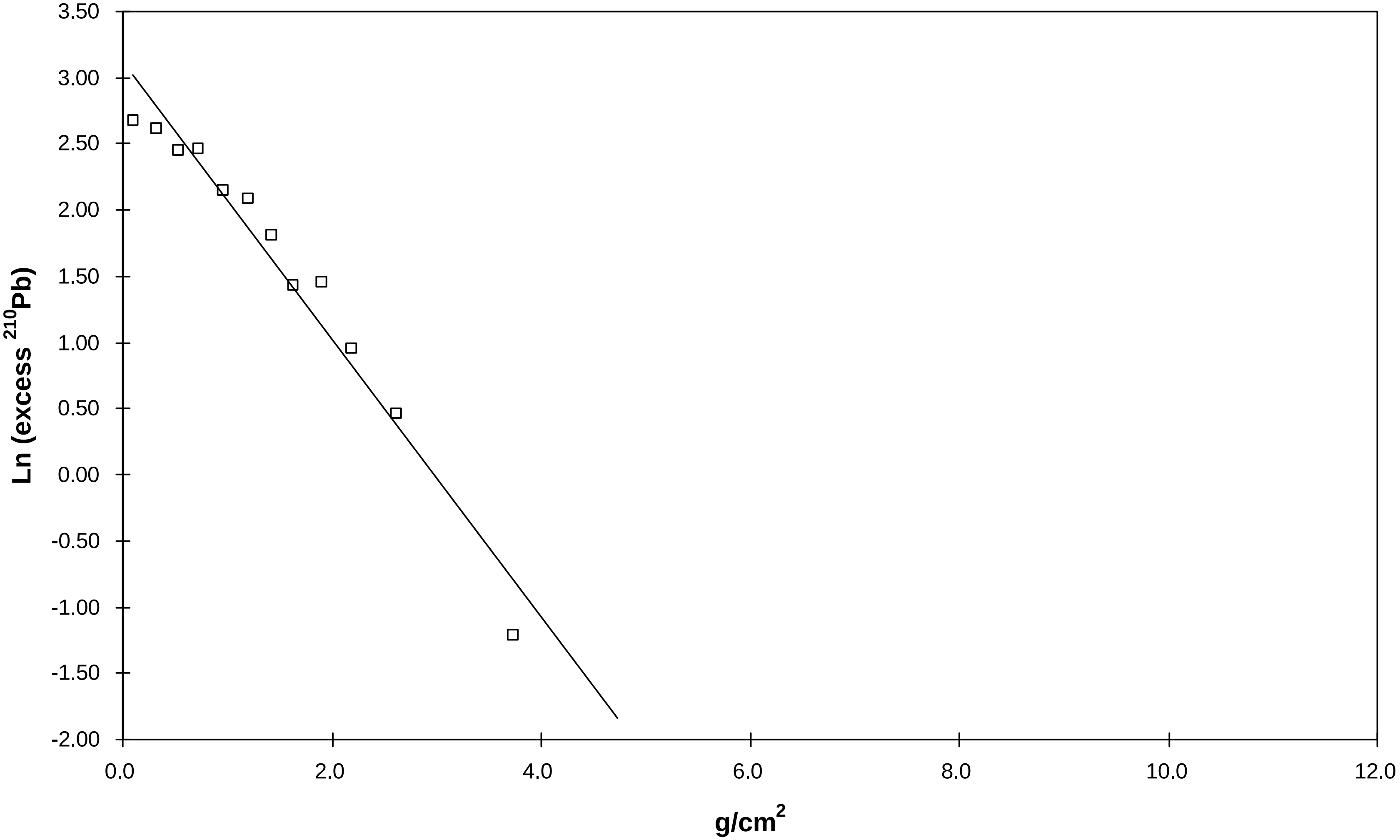

Figure 14. ^{210}**Pb plot for location Stuart-2.**

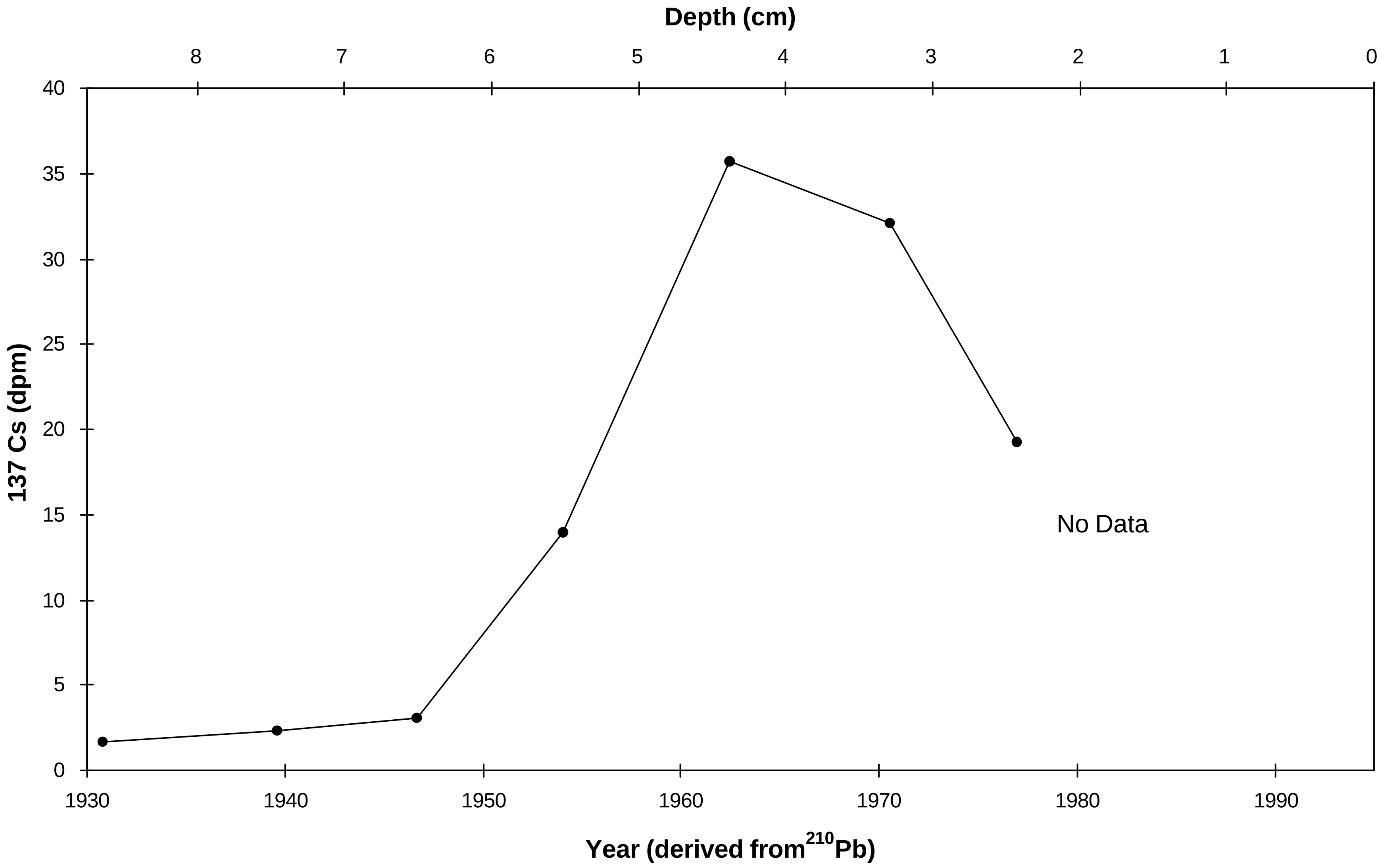

Figure 15. ^{137}Cs concentration vs ^{210}Pb age in the core Stuart-2.

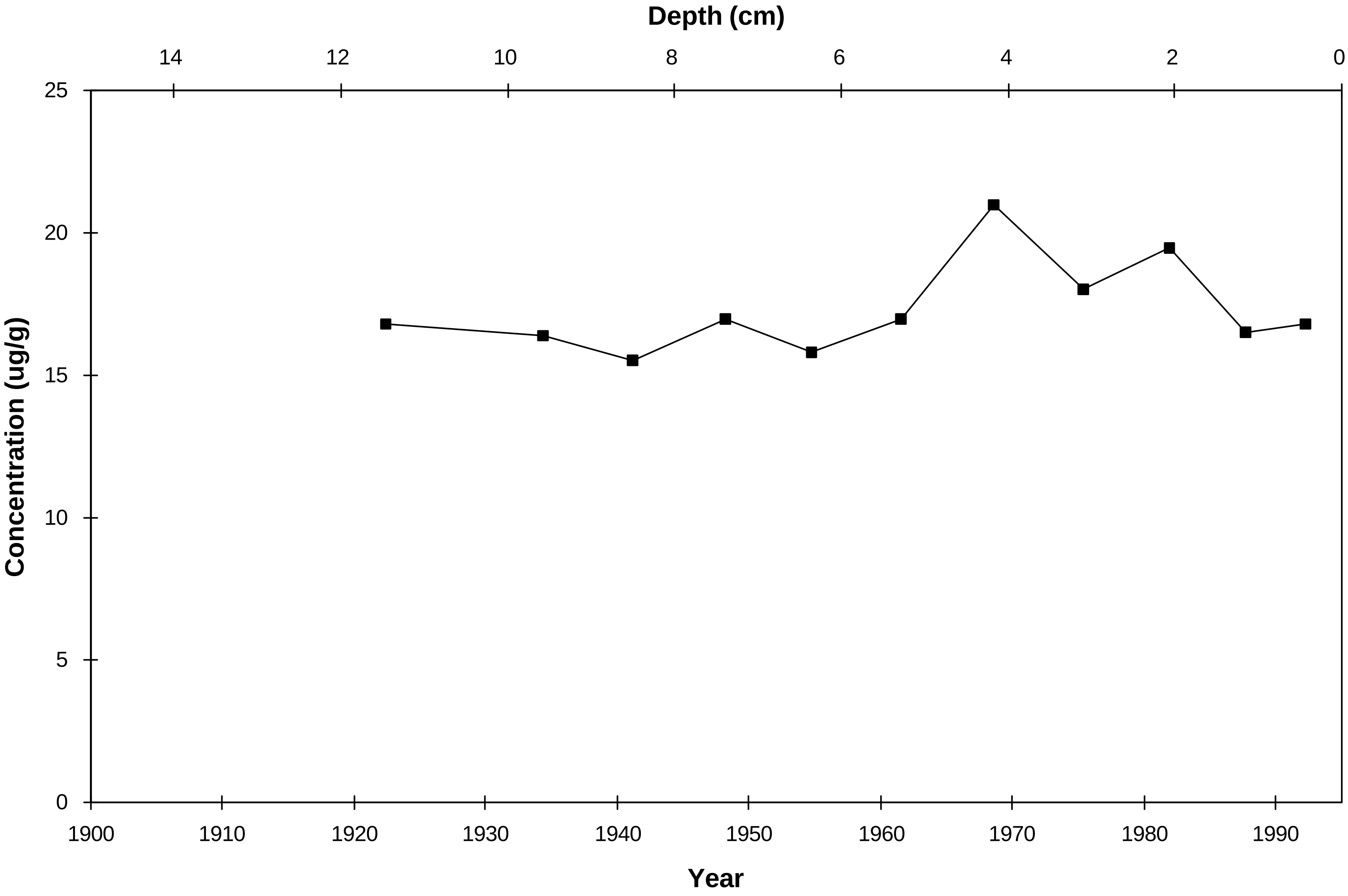

Figure 16. **Pb trends with time in Stuart-1 sediments.**

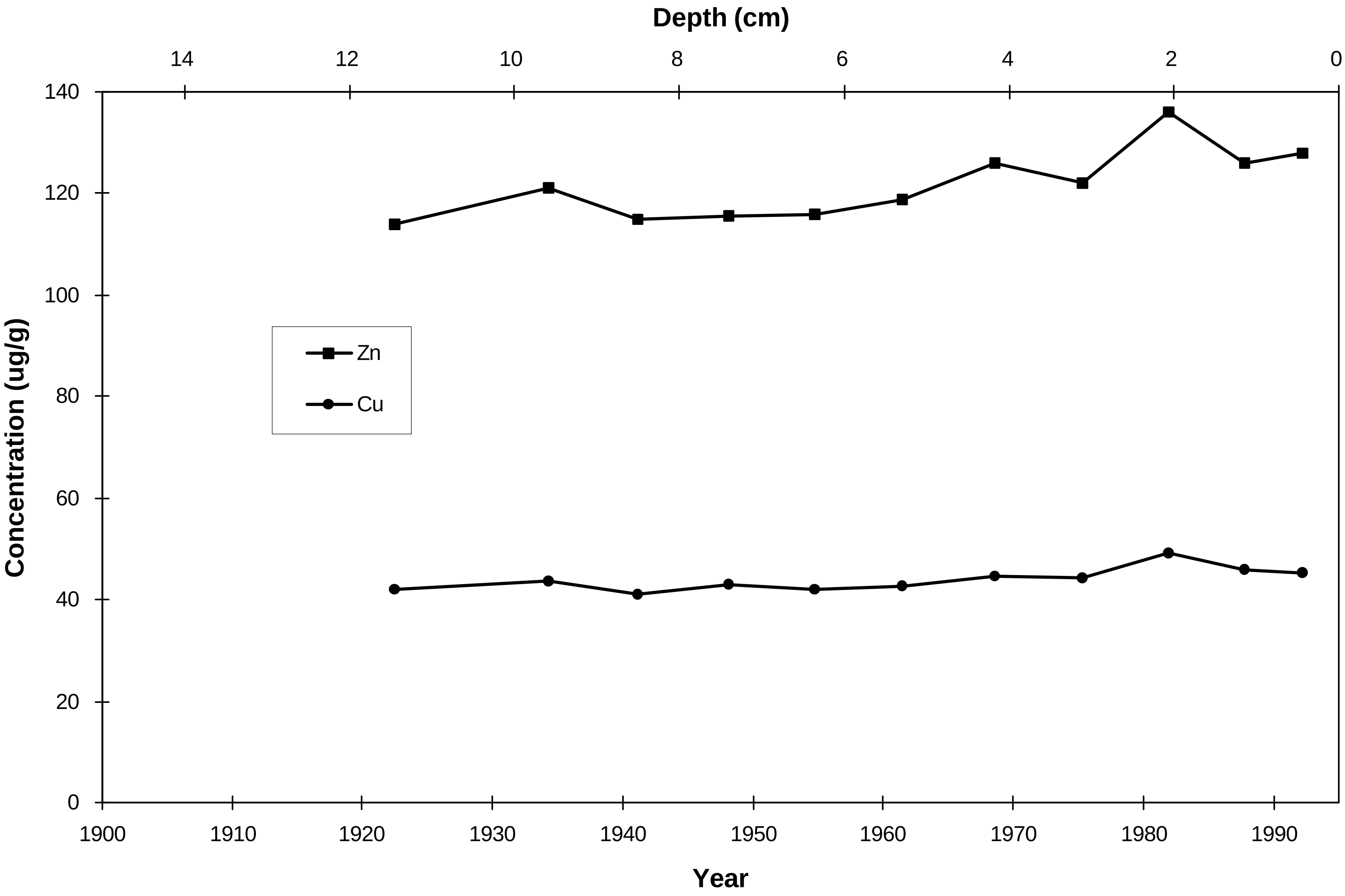

Figure 17. **Zn and Cu trends with time in Stuart-1 sediments.**

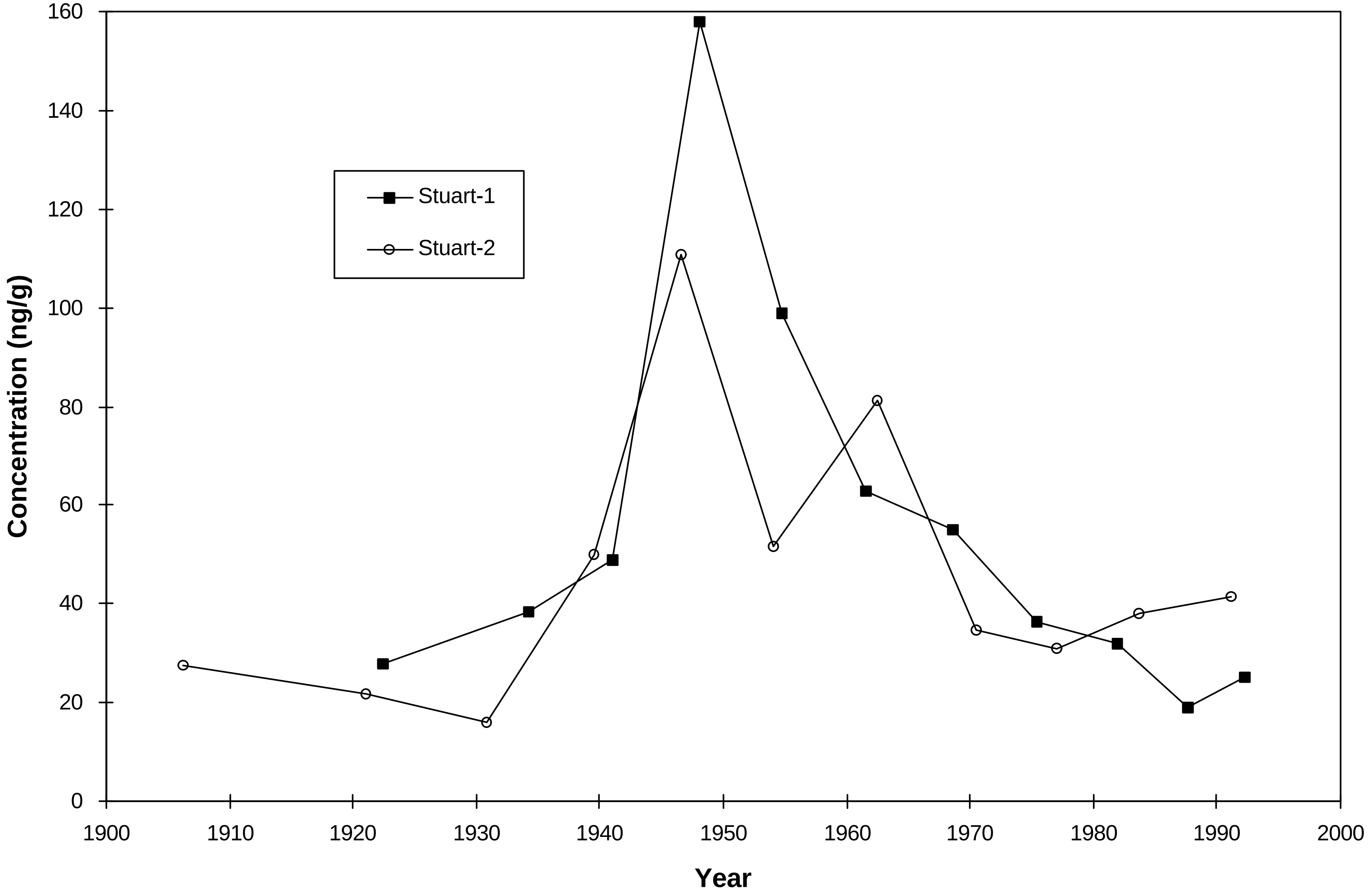

Figure 18. Hg trends with time in Stuart-1 and Stuart-2 sediments.

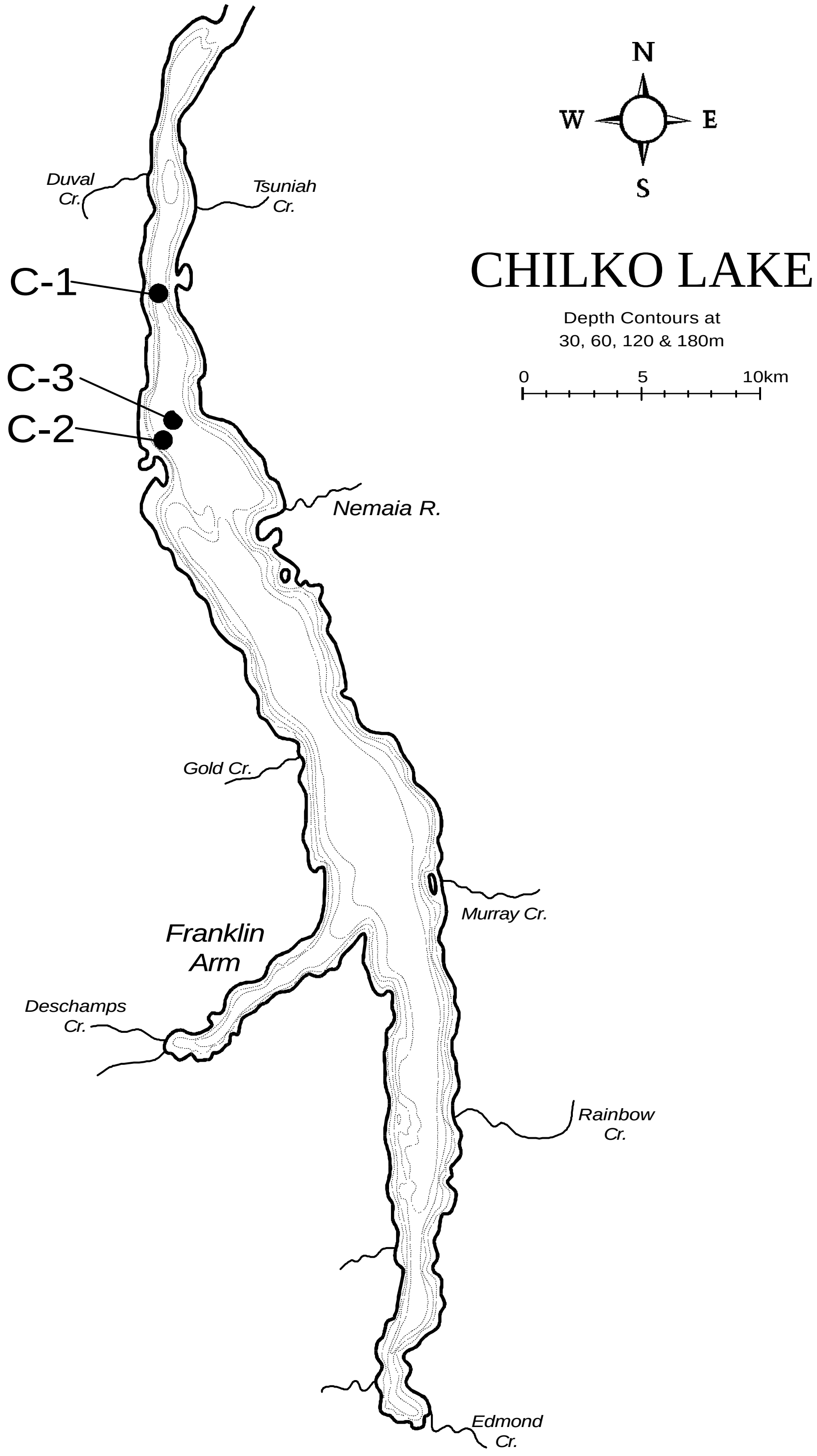

Figure 19. **Chilko Lake sampling locations.**

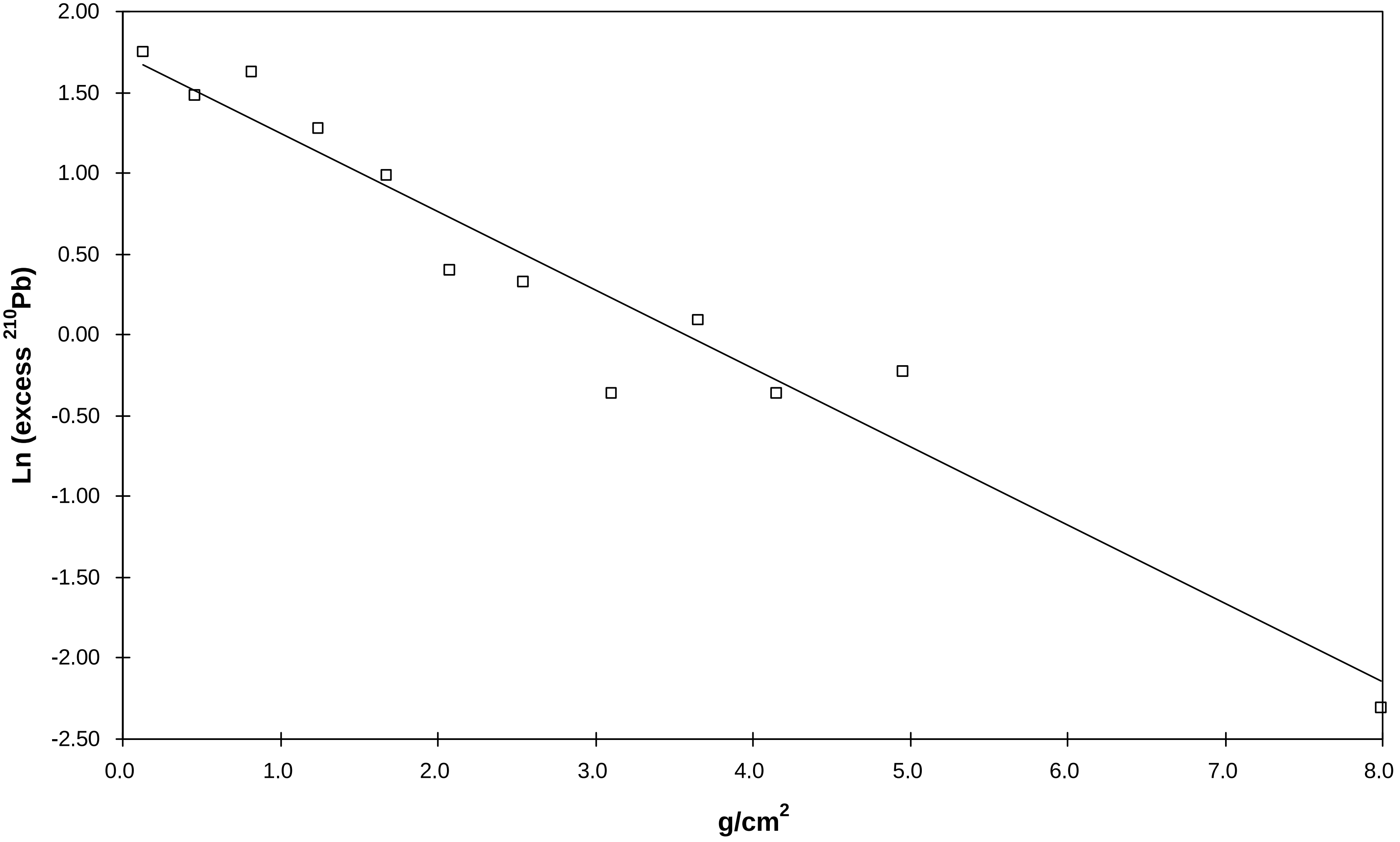

Figure 20. ^{210}**Pb plot for location Chilko-1.**

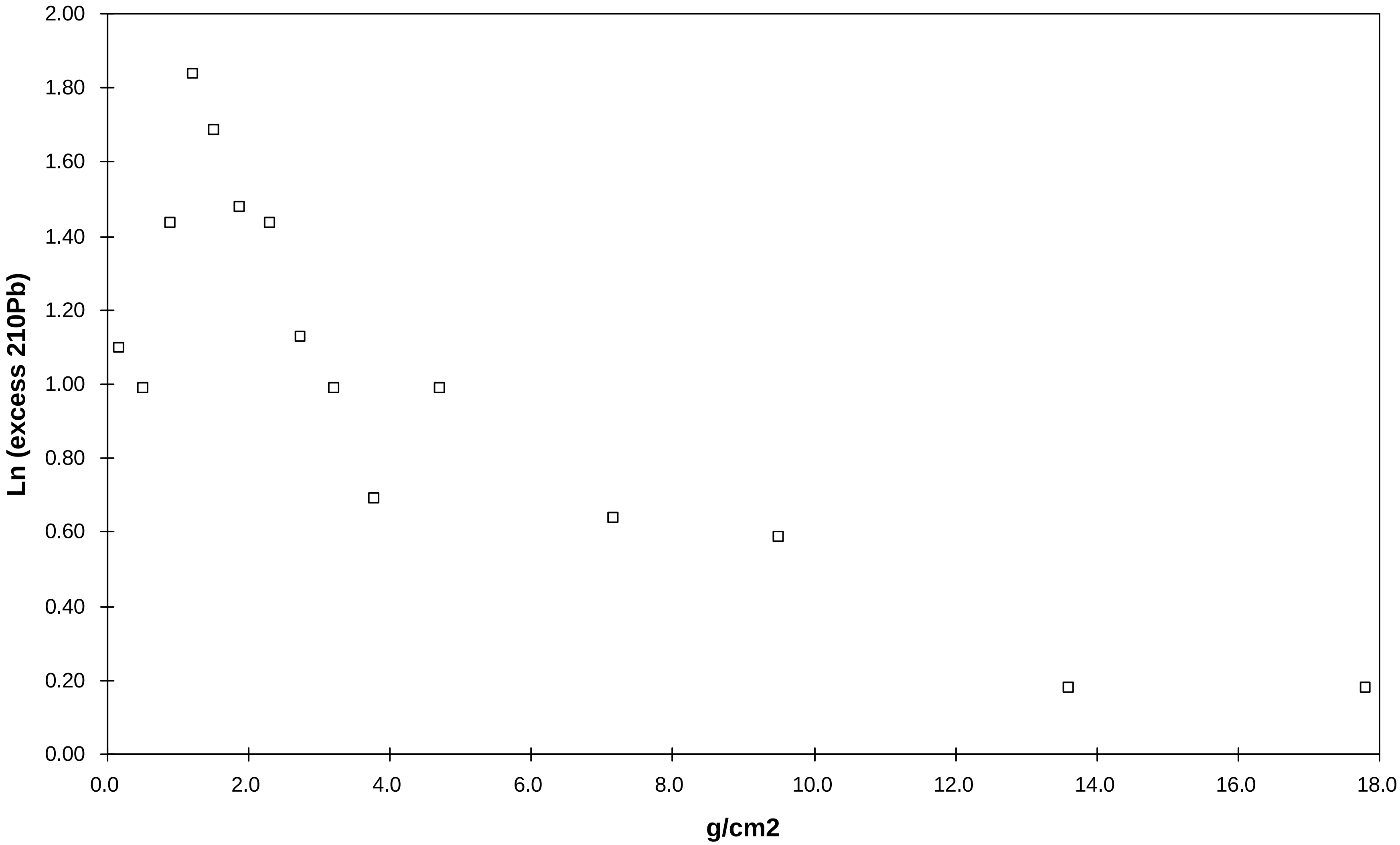

Figure 21. ^{210}**Pb plot for location Chilko-3.**

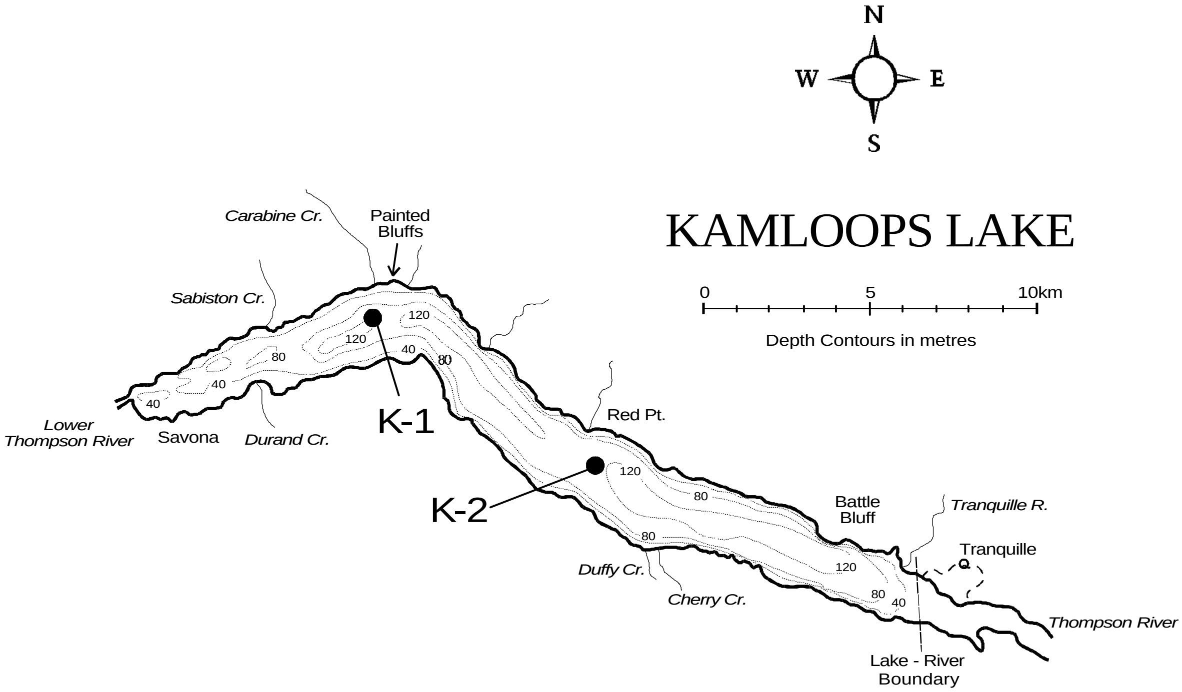

Figure 22. **Kamloops Lake sampling locations.**

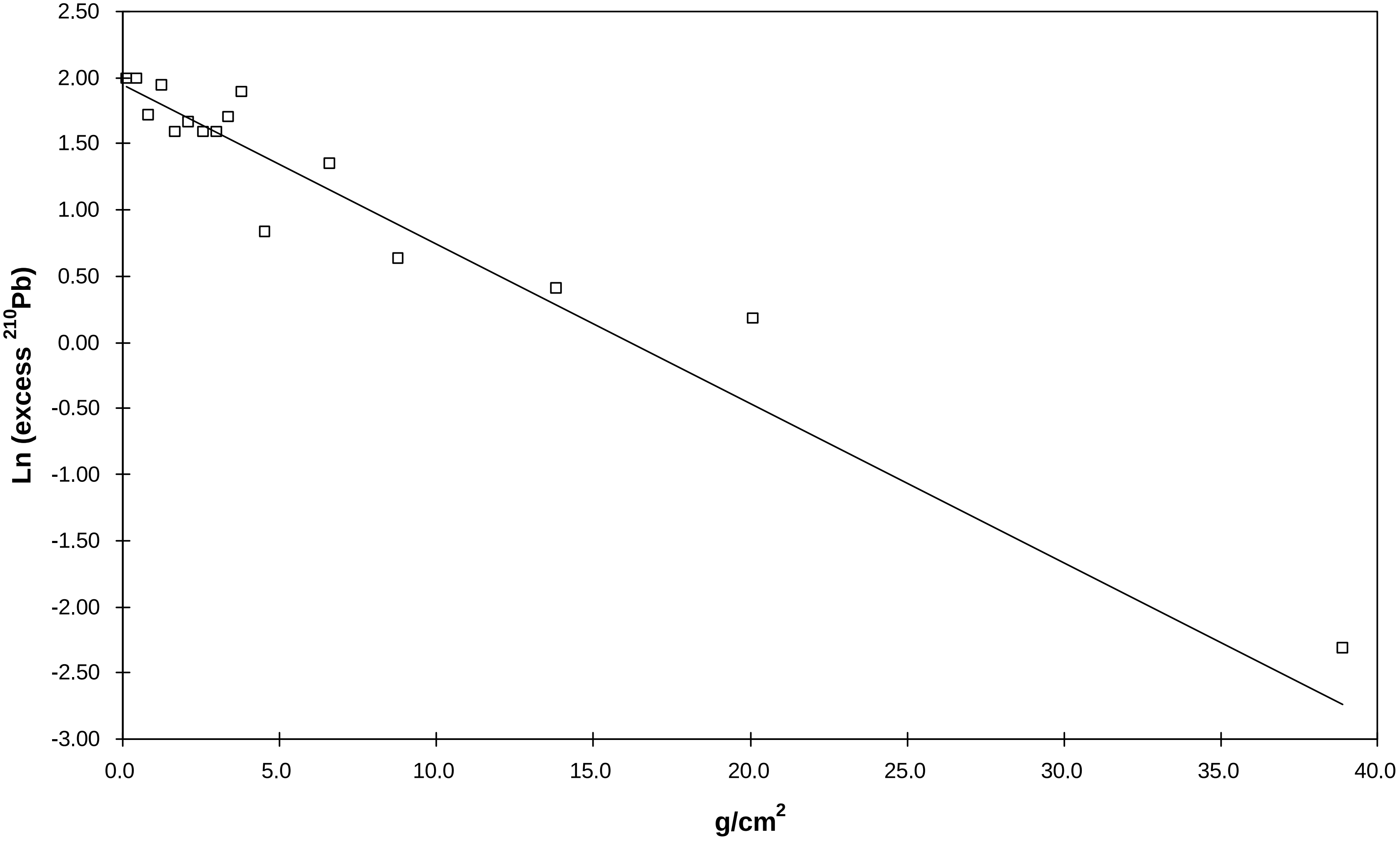

Figure 23. ^{210}**Pb plot for location Kamloops-1.**

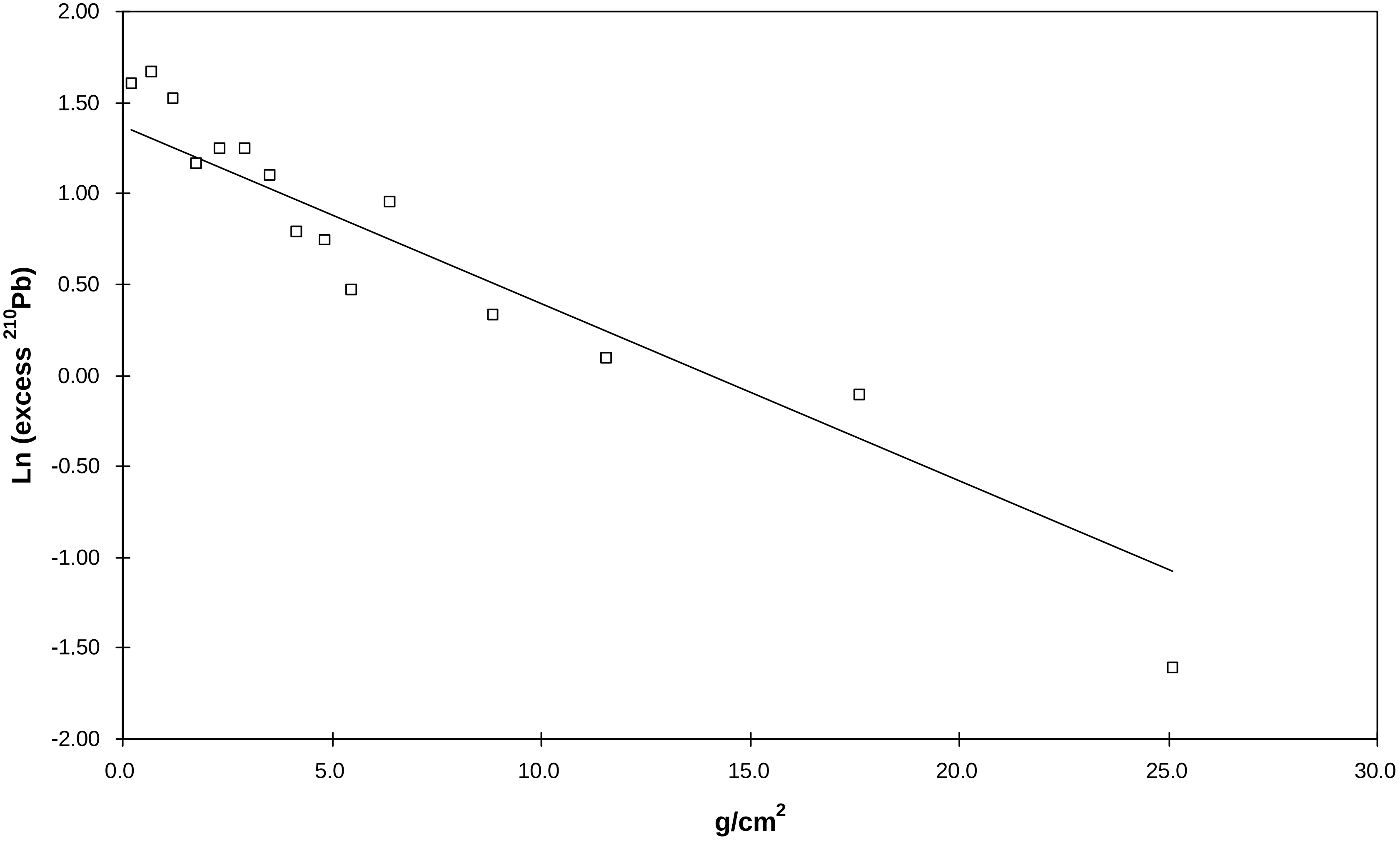

Figure 24. ^{210}**Pb plot for location Kamloops-2.**

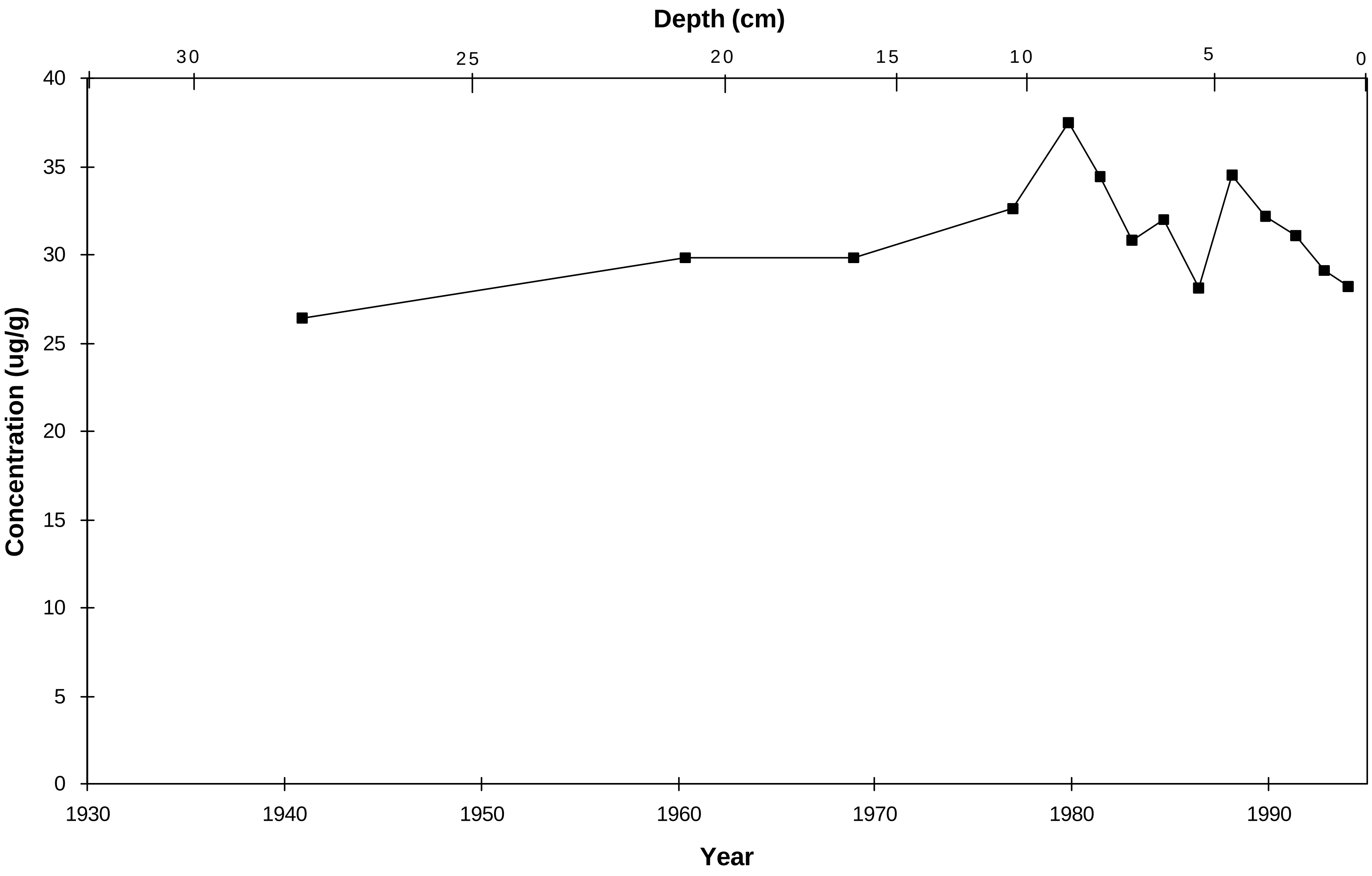

Figure 25. **Pb trend with time in Kamloops-1 sediments.**

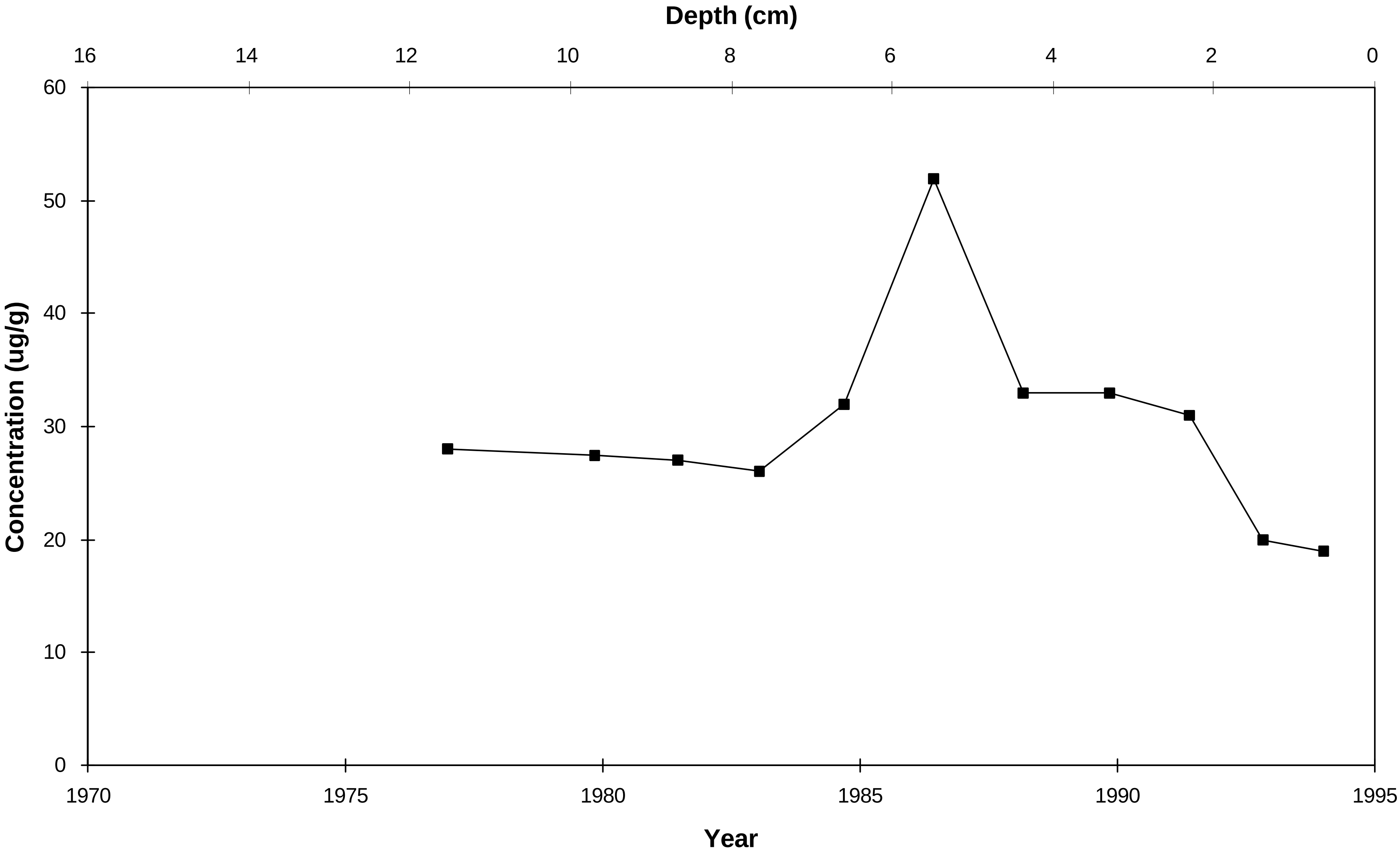

Figure 26. **Hg trend with time in Kamloops-1 sediments.**

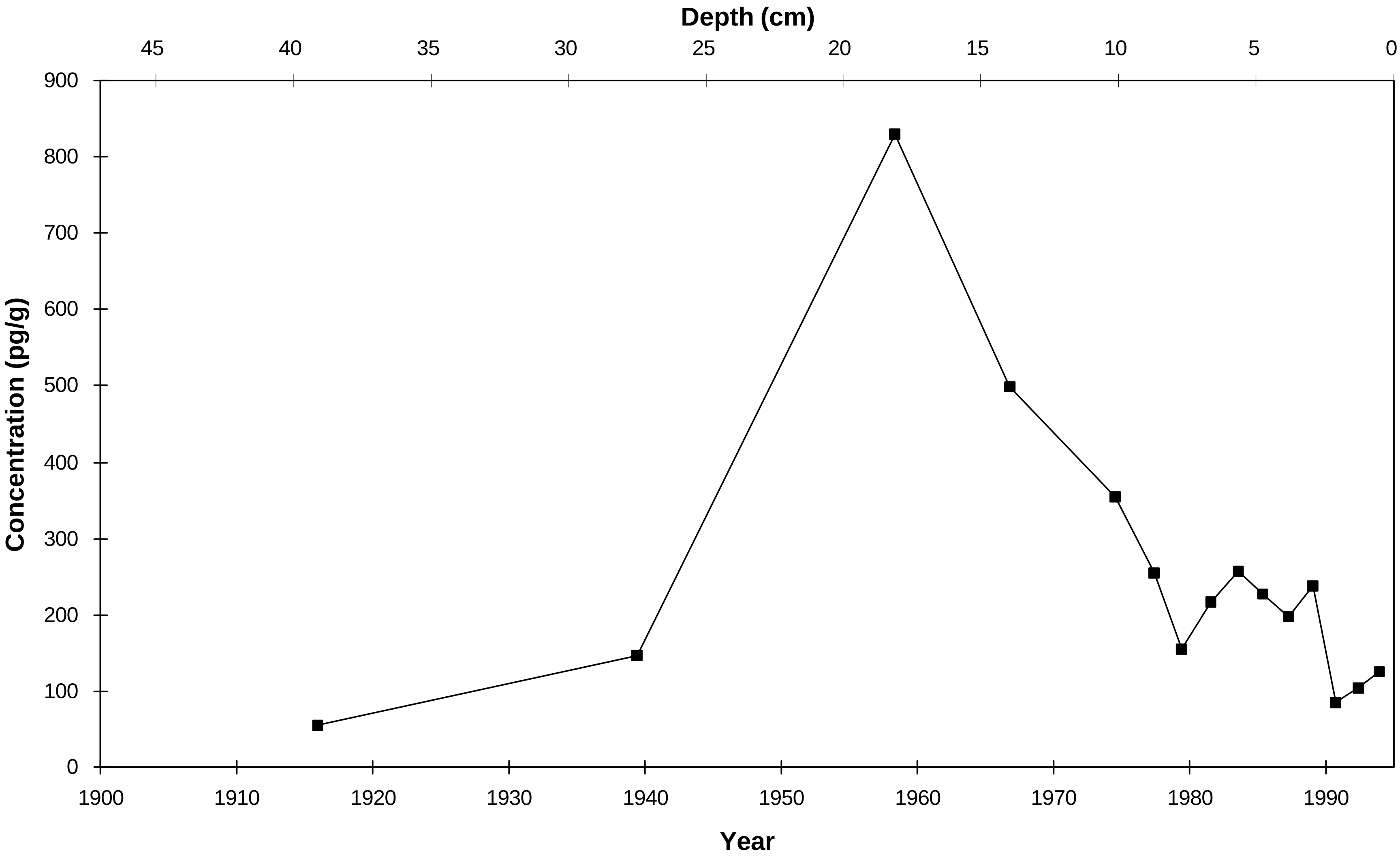

Figure 27. **Total PCB (excluding PCB 13, 15 and 37) trend with time in Kamloops-2 sediments.**

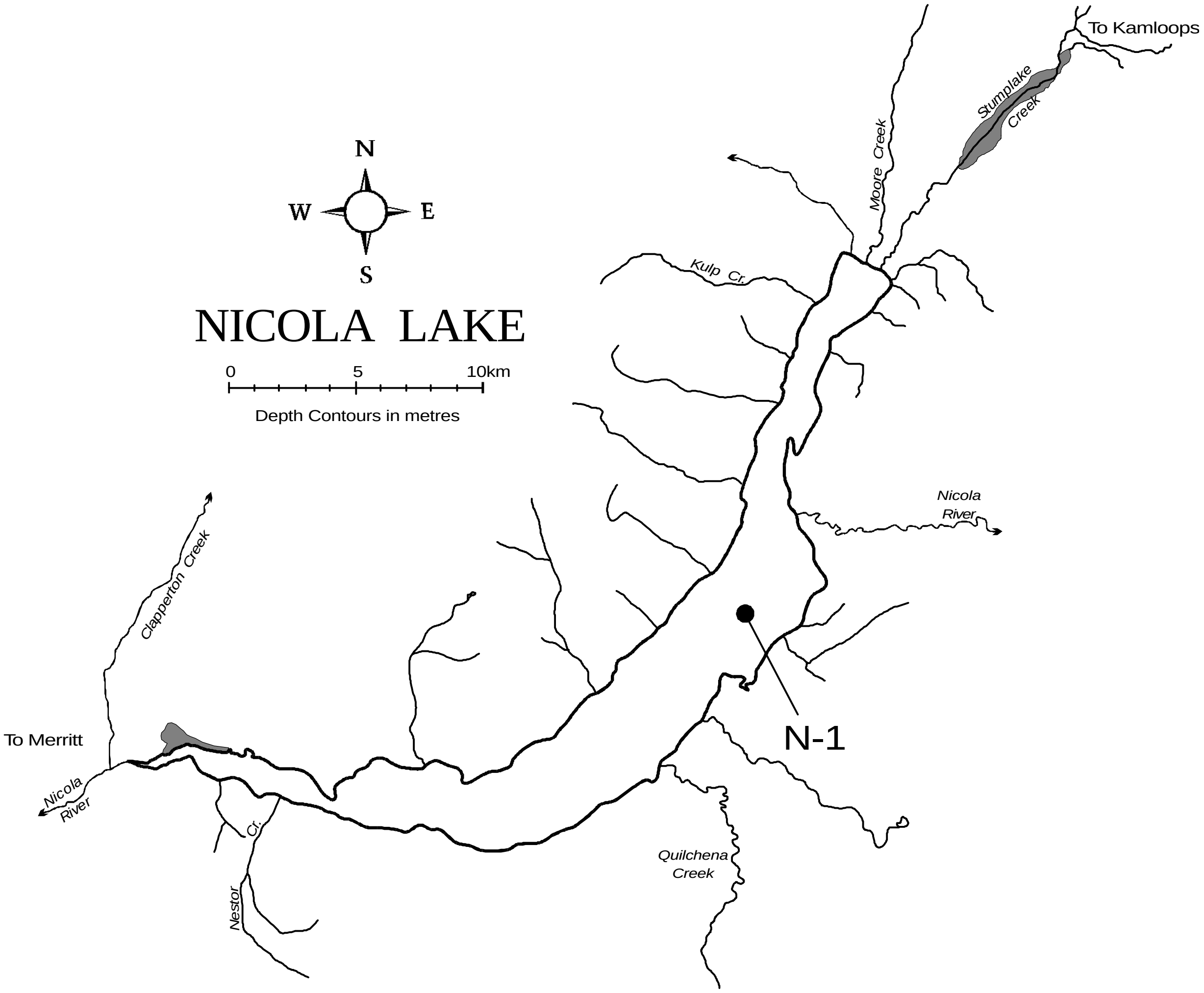

Figure 28. **Nicola Lake sampling location.**

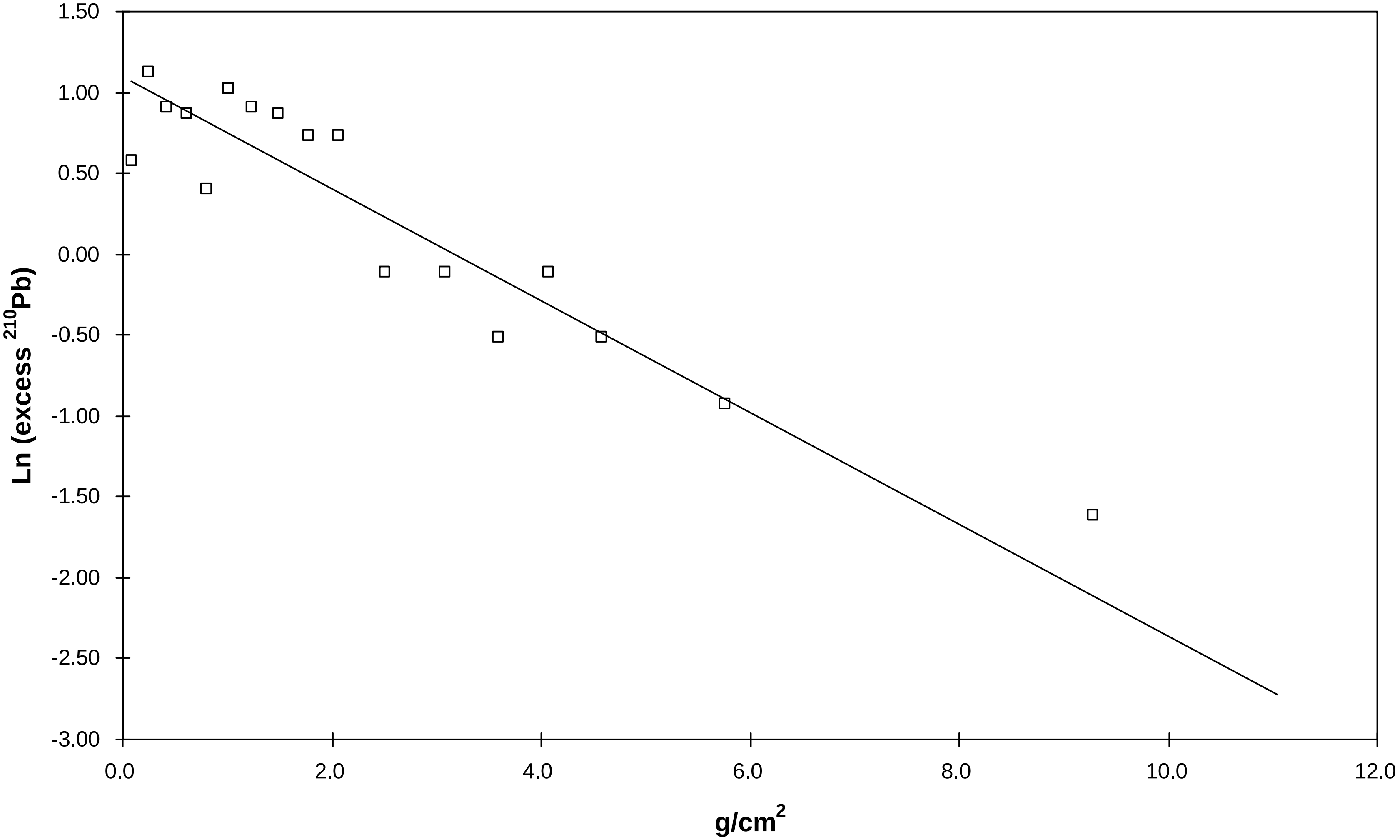

Figure 29. ^{210}**Pb plot for location Nicola-1.**

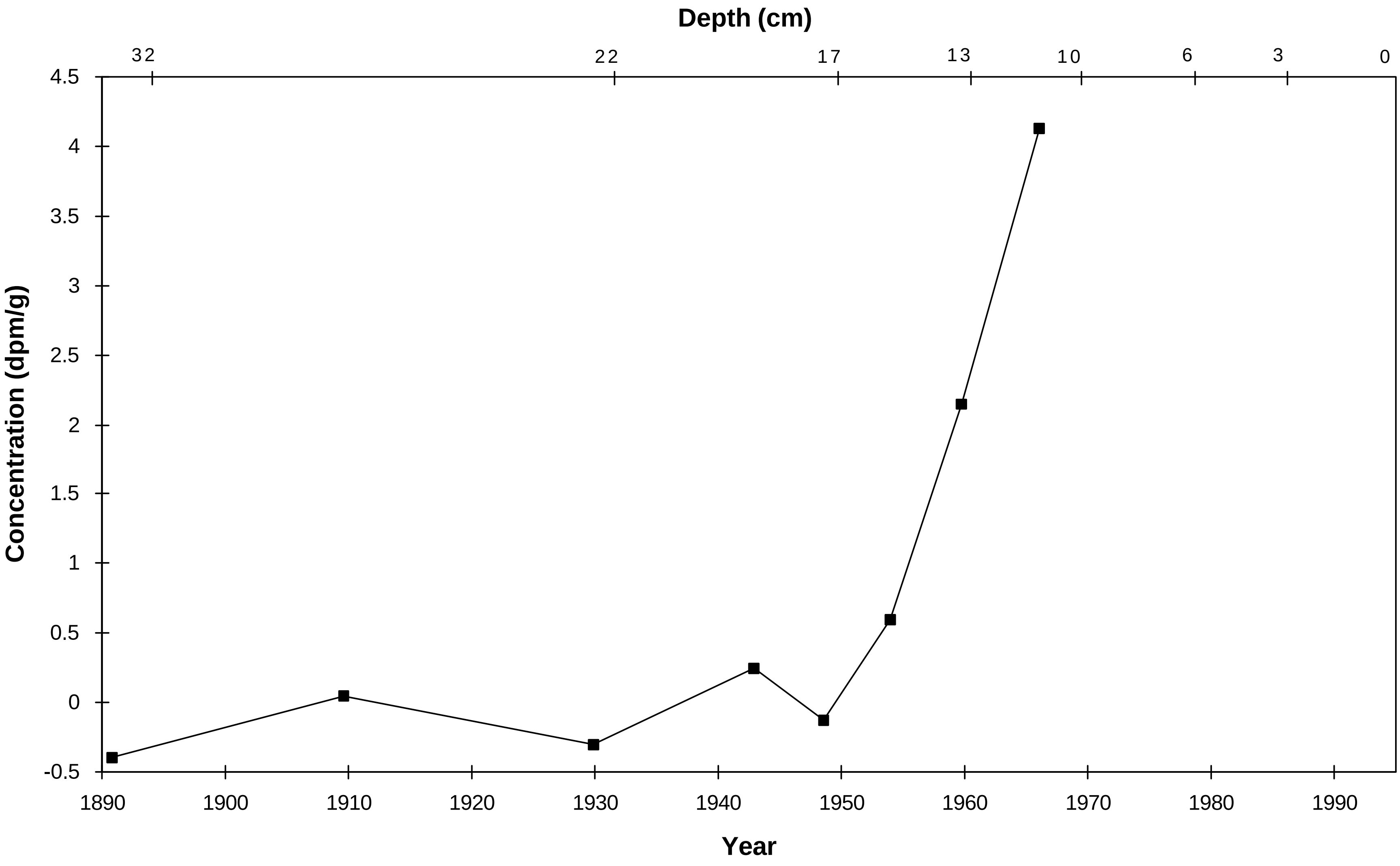

Figure 30. ^{137}Cs concentration as a function of time in Nicola-1 sediments (using the ^{210}Pb dates).

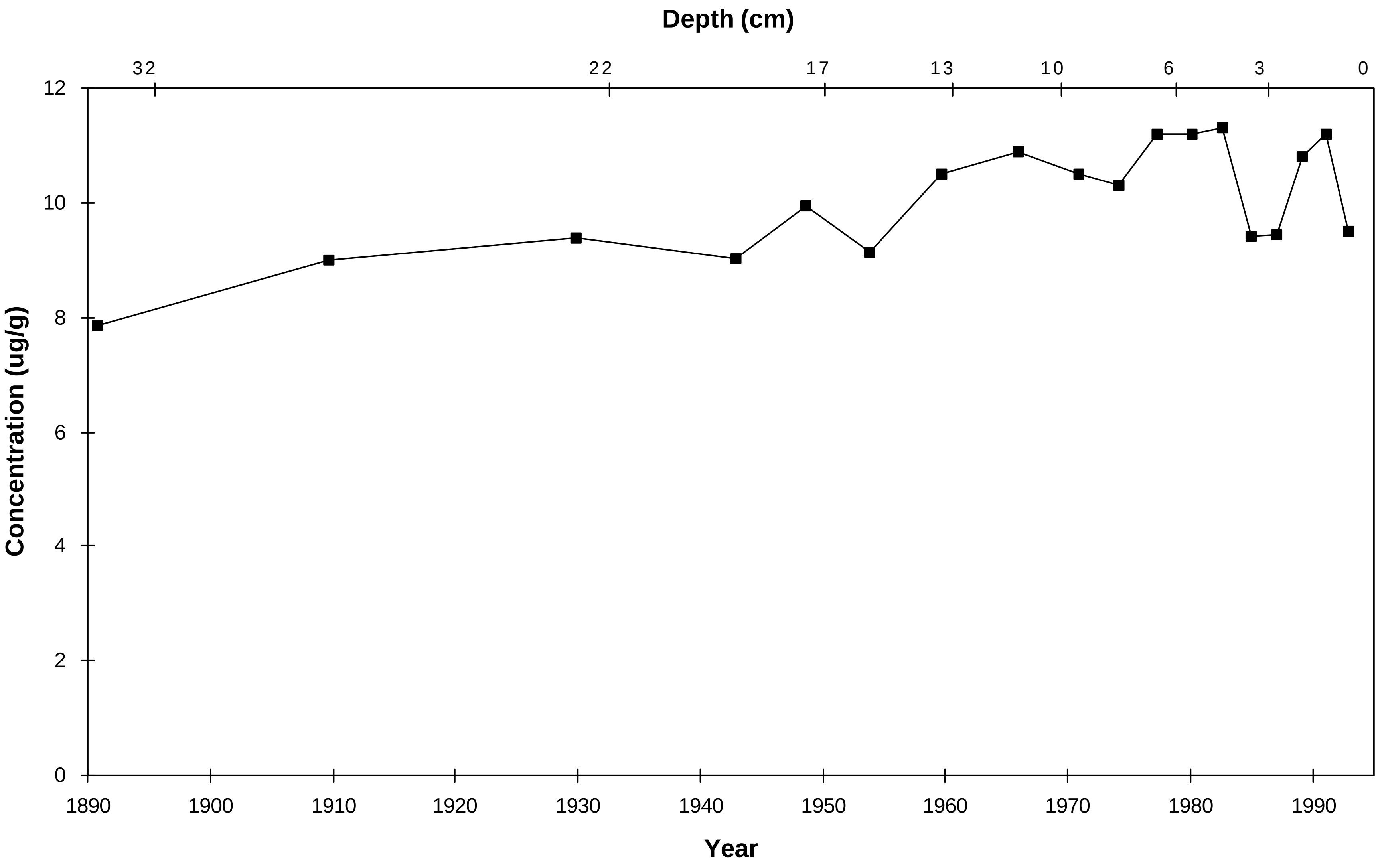

Figure 31. **Pb trend with time in Nicola-1 sediments.**

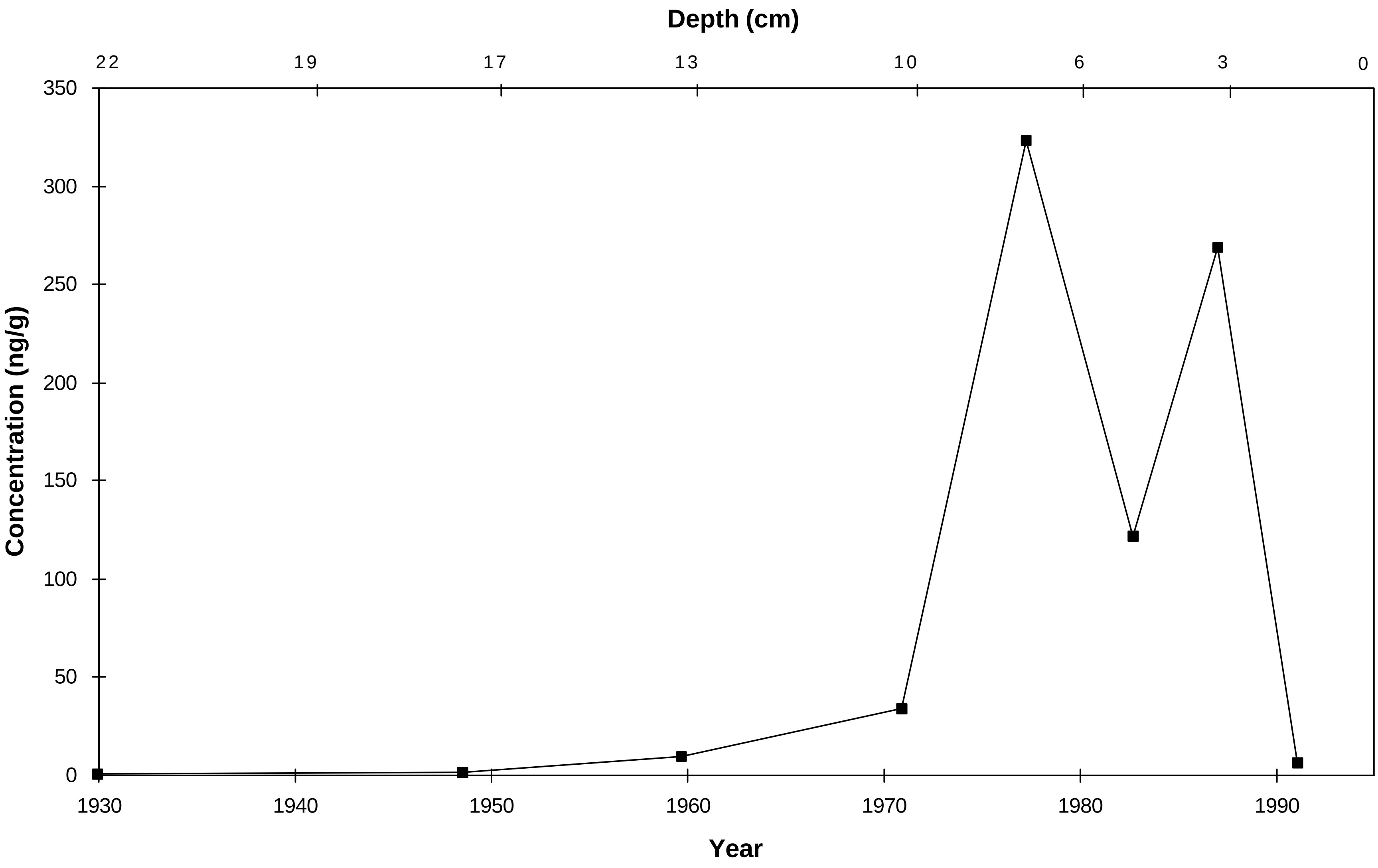

Figure 32. **Total DDT trend with time in Nicola-1 sediments.**

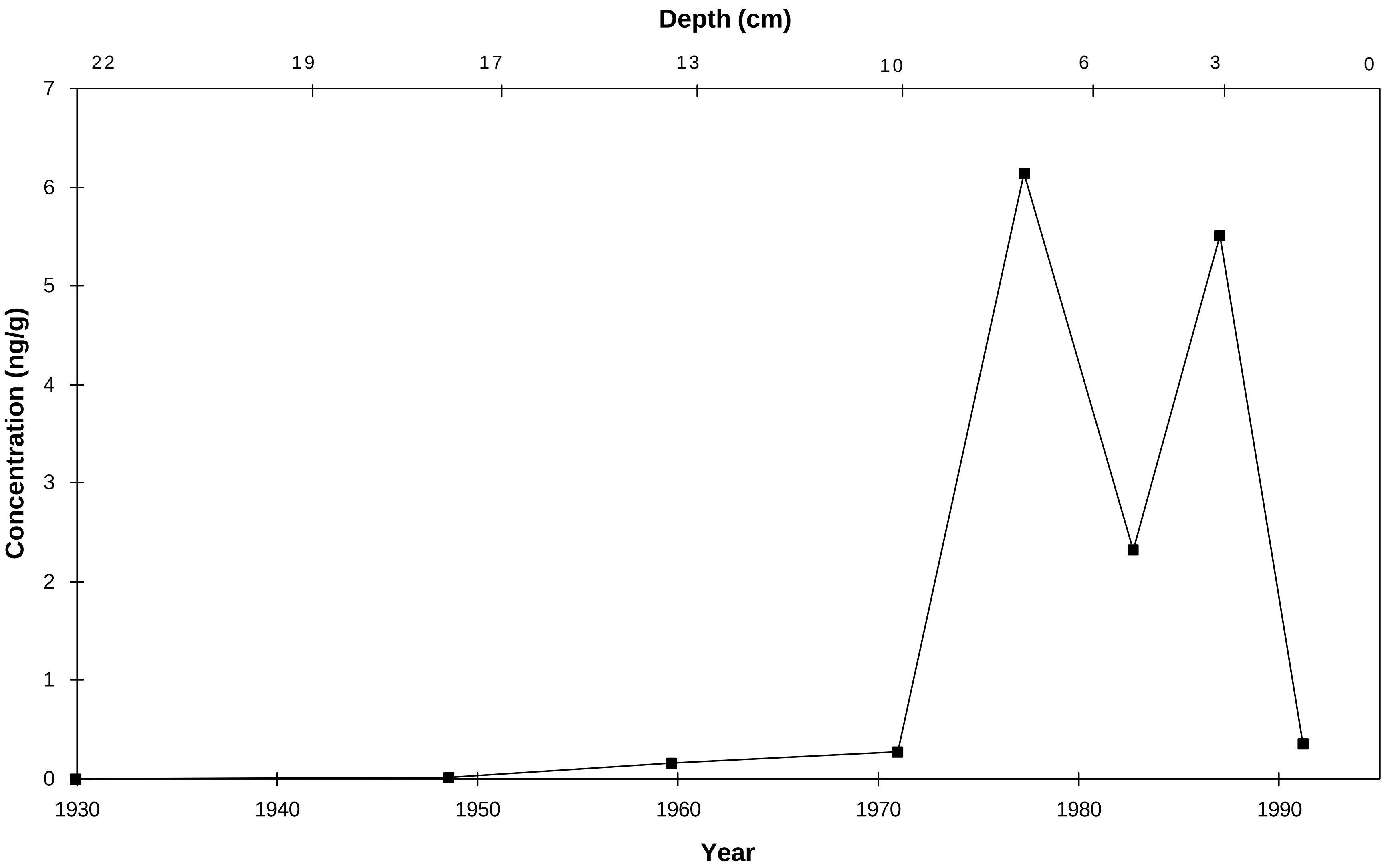

Figure 33. Total HCH trend with time in Nicola-1 sediments.

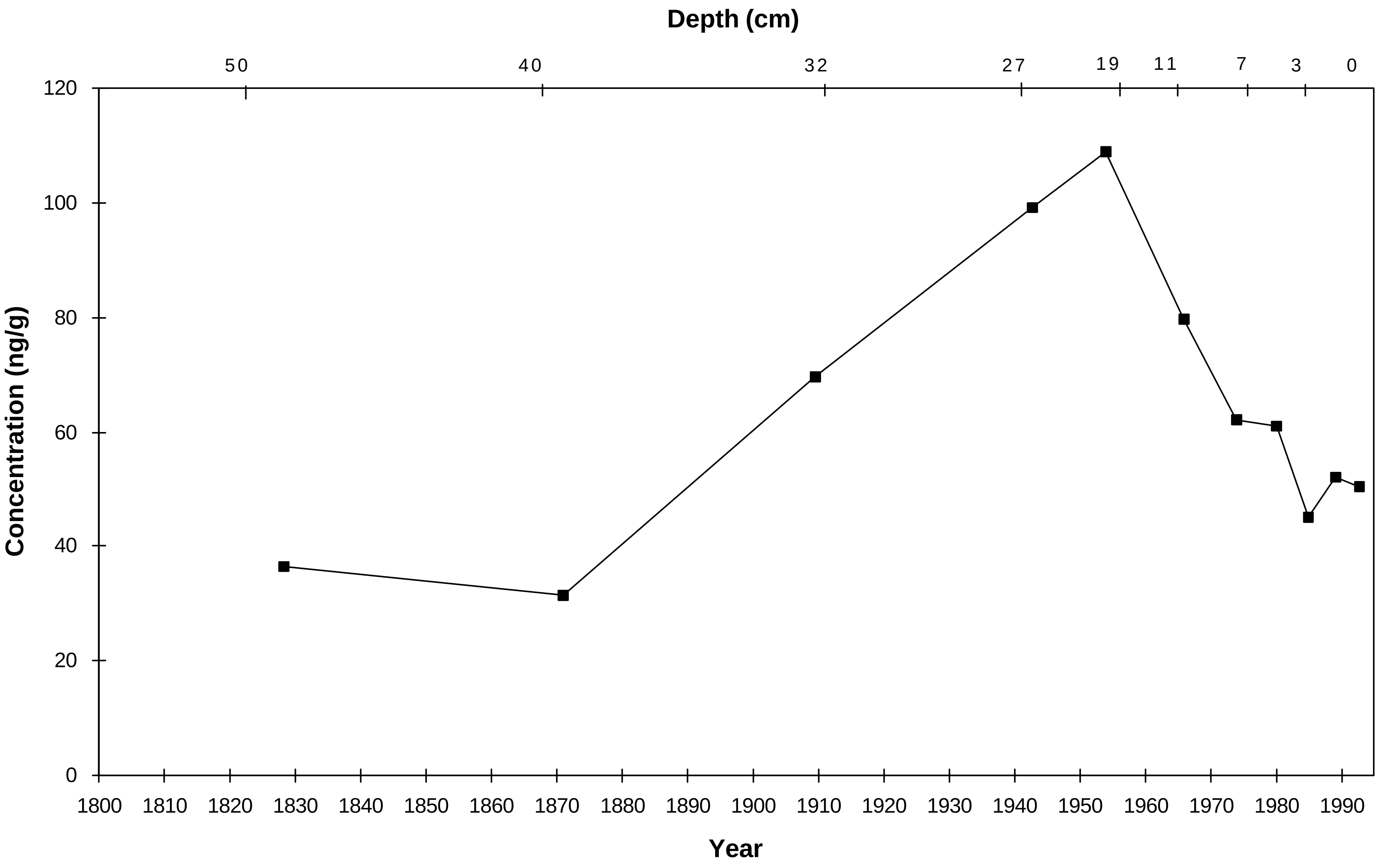

Figure 34. Total PAH trend with time in Nicola-1 sediments.

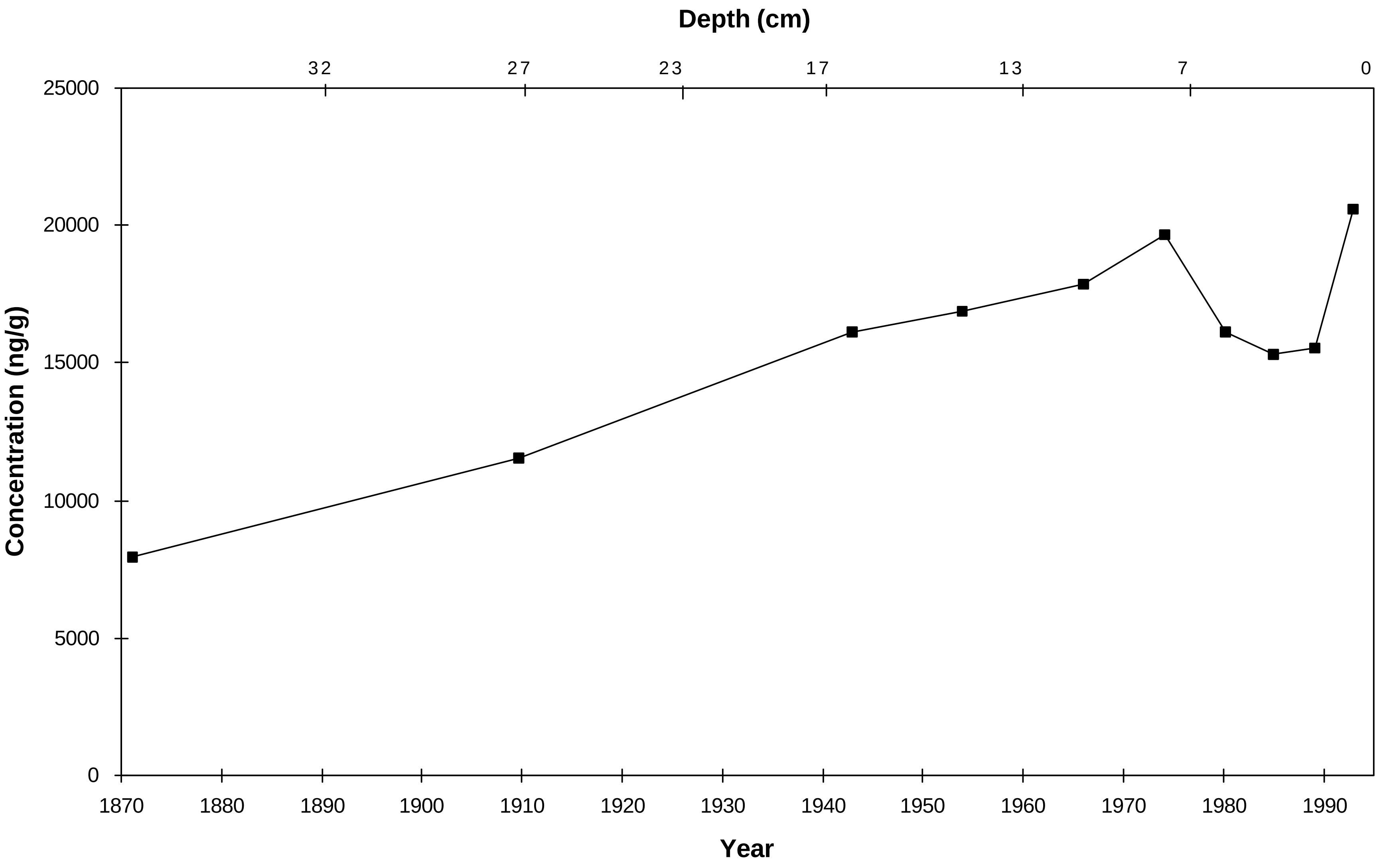

Figure 35. **Total alkane trend with time in Nicola-1 sediments.**

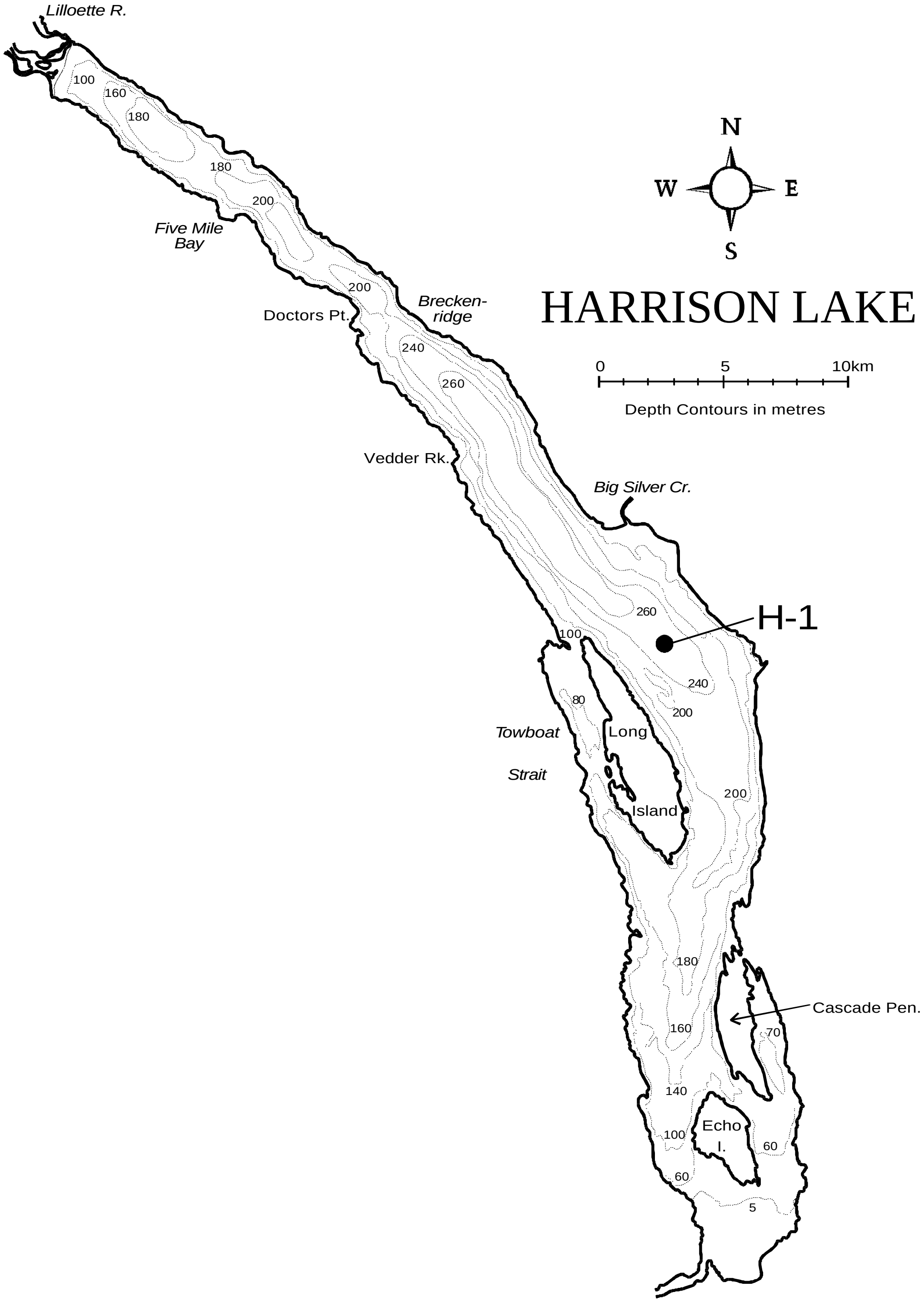

Figure 36. Harrison Lake sampling location.

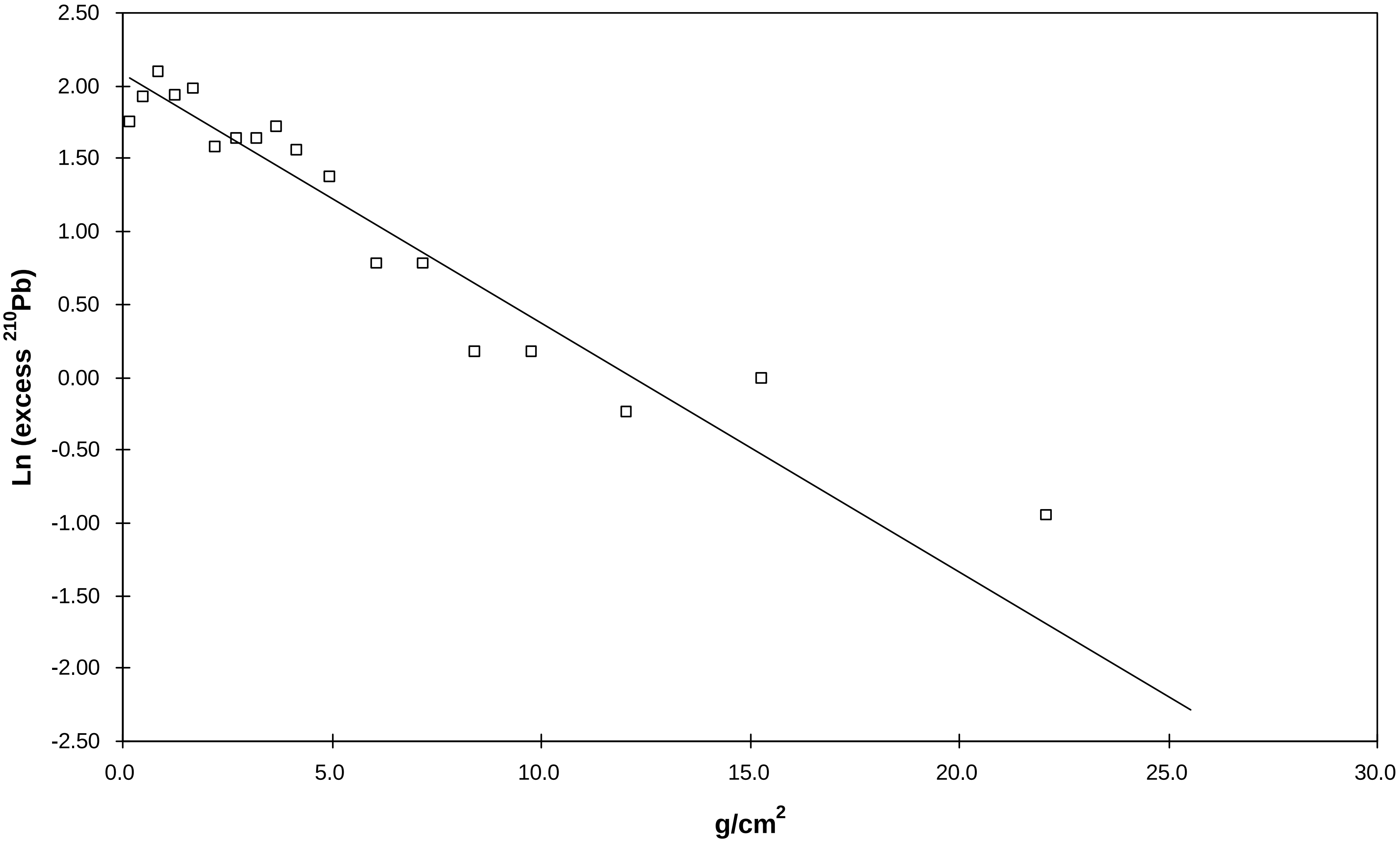

Figure 37. **^{210}Pb plot for location Harrison-1.**

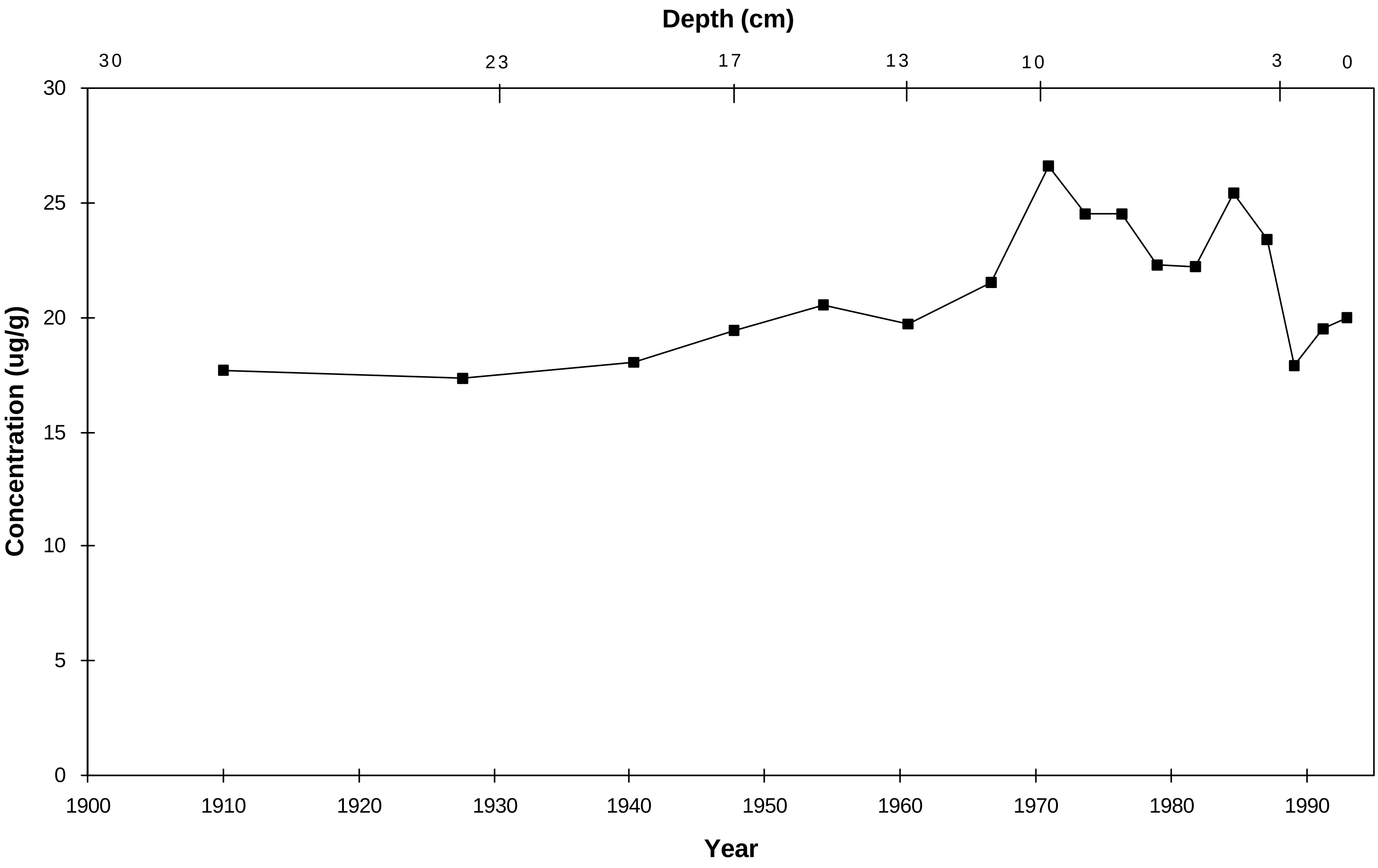

Figure 38. **Pb trend with time in Harrison-1 sediments.**

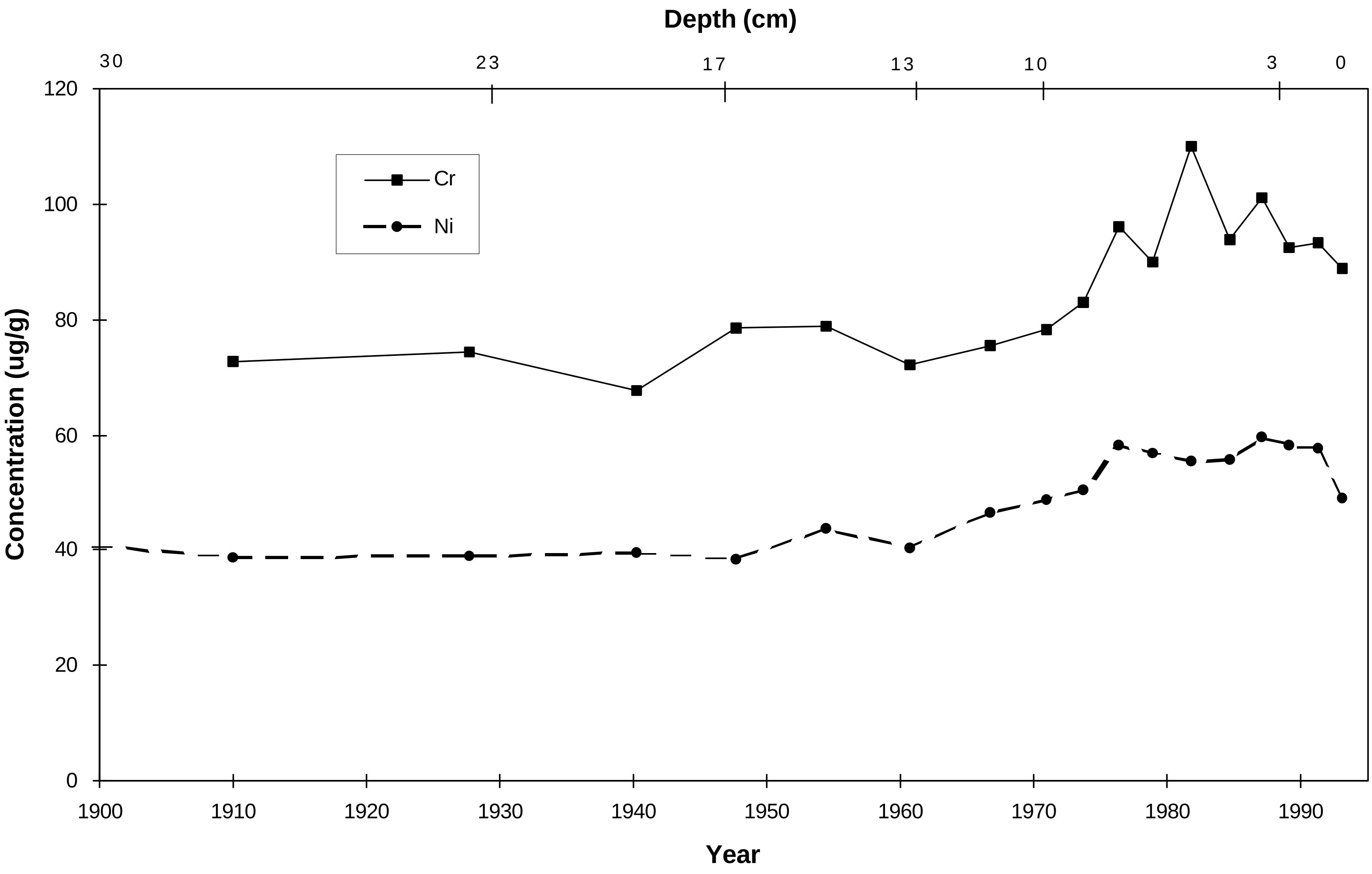

Figure 39. Cr and Ni trends with time in Harrison-1 sediments.

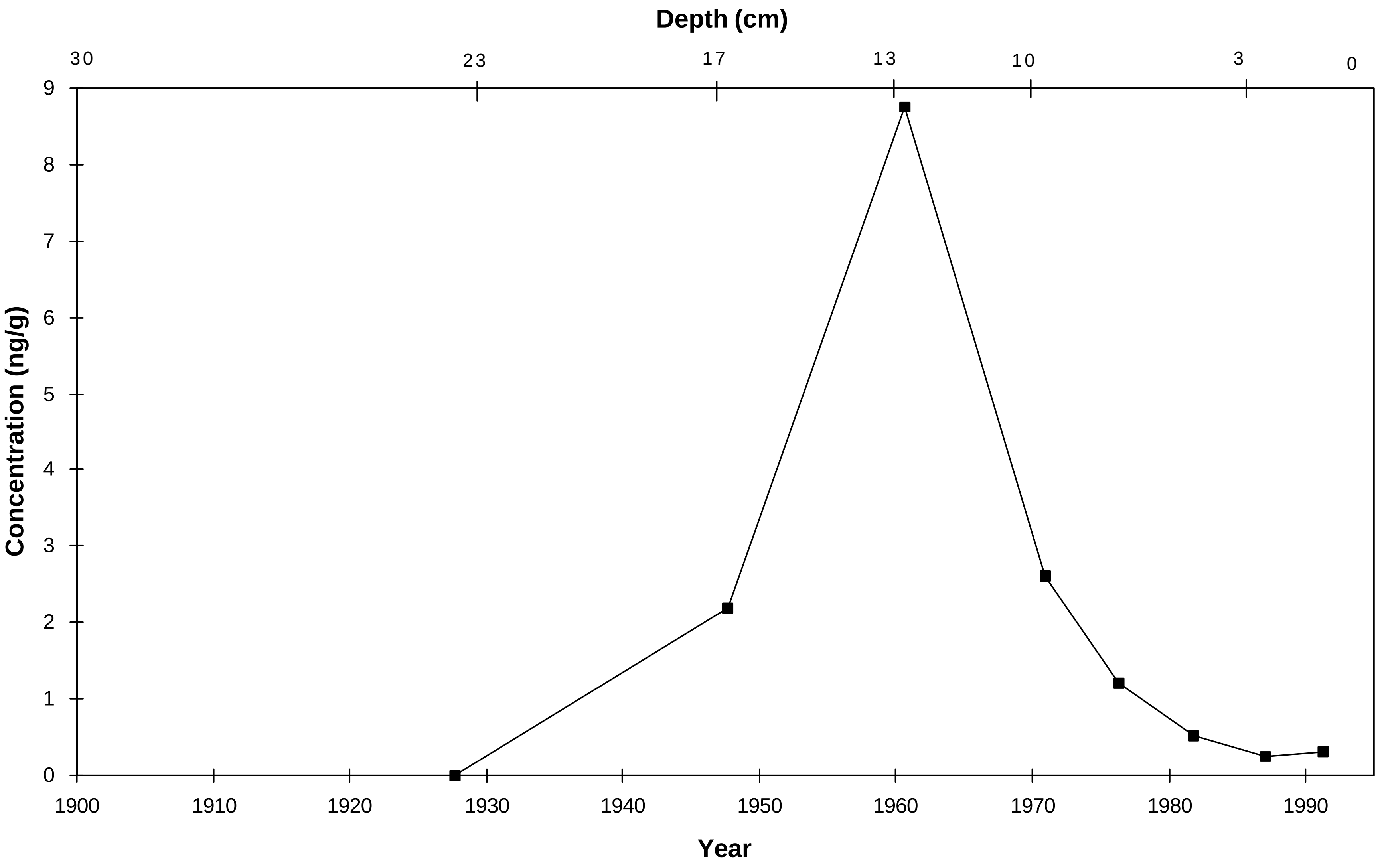

Figure 40. **Total DDT trend with time in Harrison-1 sediments.**

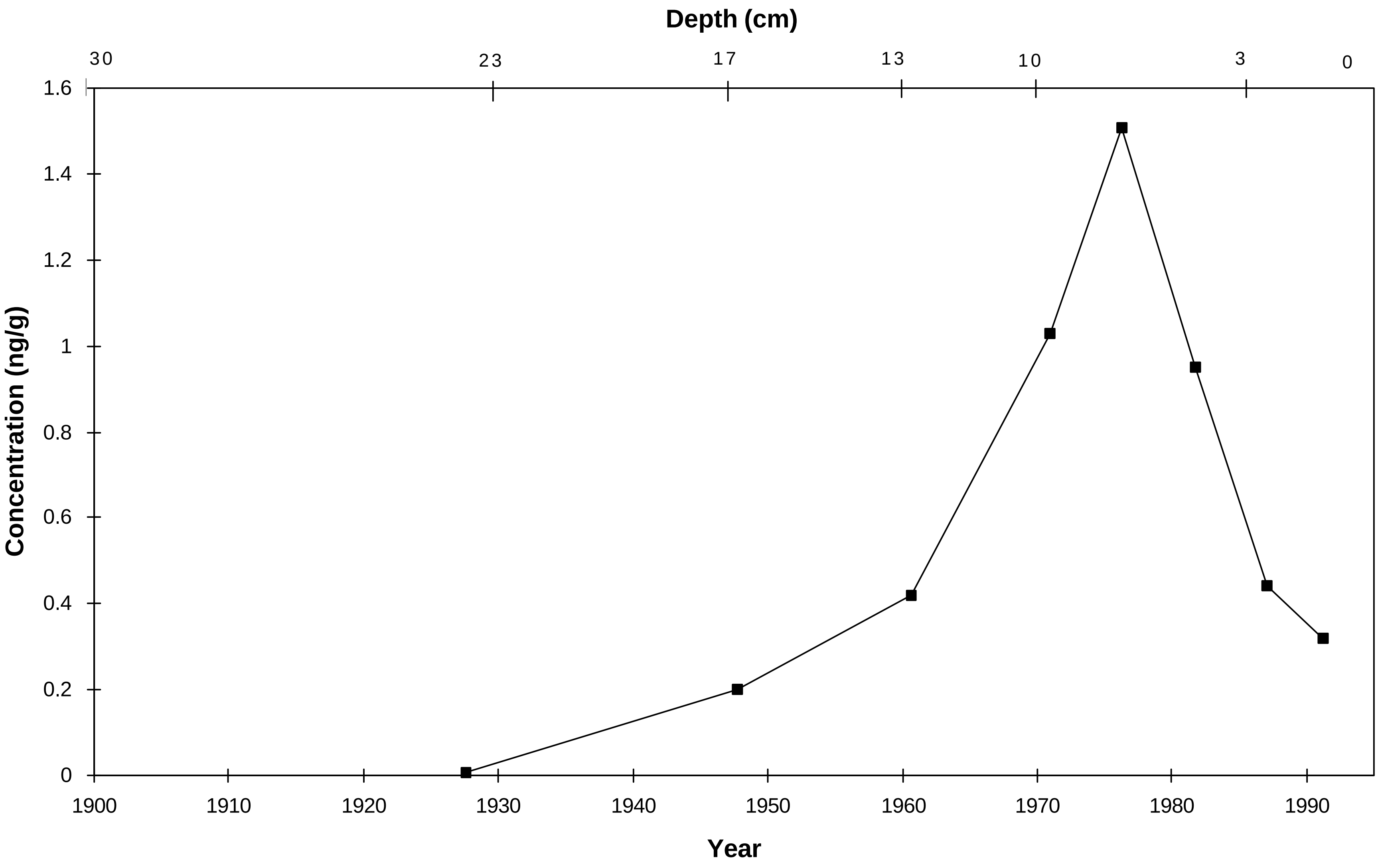

Figure 41. **Total PCB trend with time in Harrison-1 sediments.**

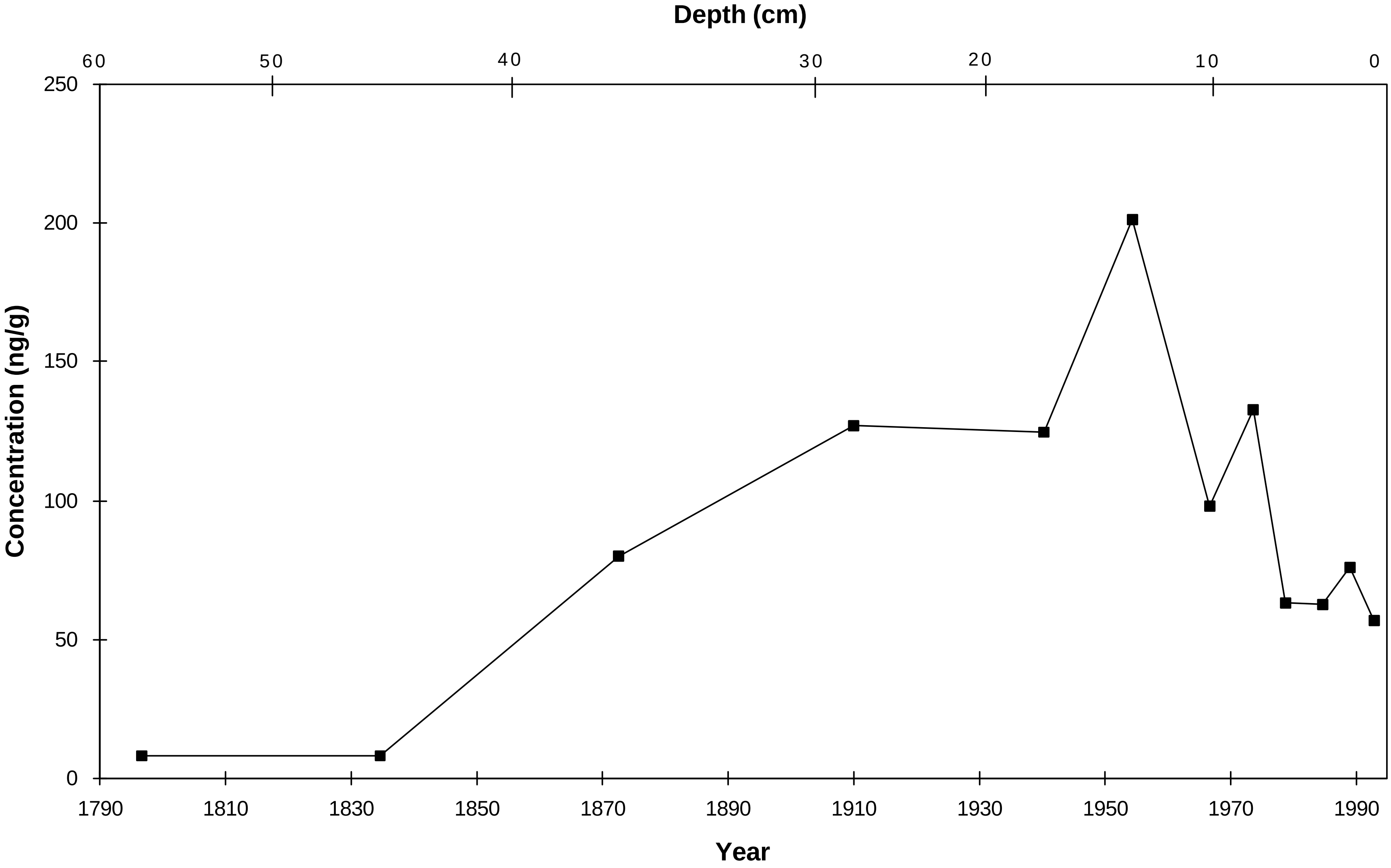

Figure 42. **Total PAH trend with time in Harrison-1 sediments.**

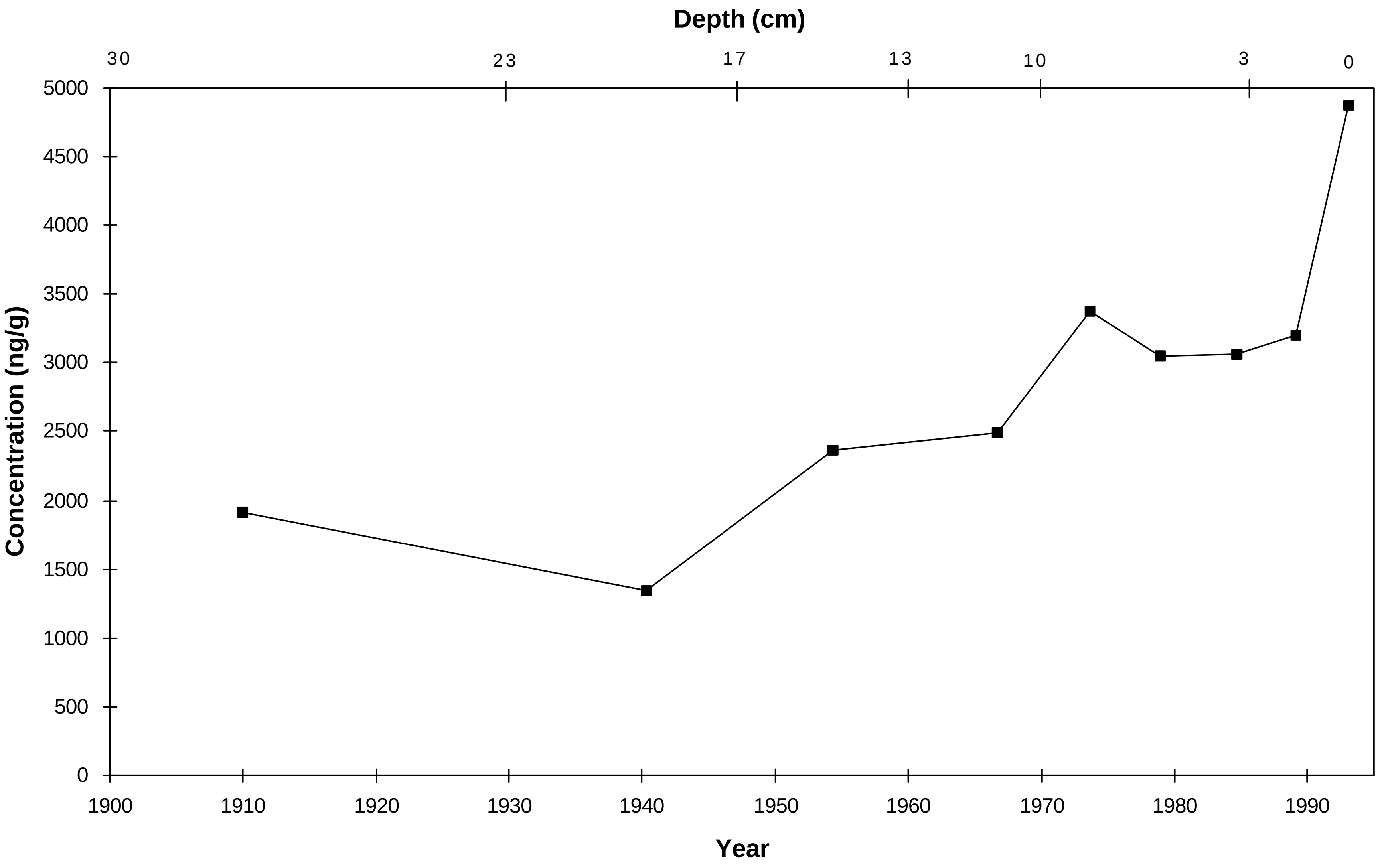

Figure 43. **Total Alkane trend with time in Harrison-1 sediments.**