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A Palaeoecological Evaluation of the Acidification Status of Lakes in the North-West and West of Ireland

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1 INTRODUCTION

Acid deposition has caused strong acidification of lakes on susceptible geology in the Galloway region of south-west Scotland, Cumbria and north Wales (eg. Battarbee *et al.* 1988, Battarbee *et al.* 1989). Elsewhere in acid sensitive areas of the UK, lake acidification is widespread but is usually less severe. The most sensitive lakes have lake calcium concentrations of, or below c. 40 μeq l^{-1} and the critical loading of non-marine sulphate deposition for these sites is <5 g S m^{-2} yr^{-1} (Battarbee 1989). We have also shown (Flower *et al.* 1992) that lakes in the north-west of Scotland, where deposited sulphate varies between 0.4-0.8 g S m^{-2} yr^{-1}, are not significantly affected by acid deposition. However, little is known about the situation in the far north-west and west of Ireland where sensitive lakes lie in areas where acid deposition is considered to be low. This paper concerns lakes in this region (Figure 1.1).

Modelling studies indicate that H^+ and SO_x are transported to Ireland mainly from the east, principally from the UK and central Europe (Bailey *et al.* 1986). A small component comes from the west, mainly by return of polluted air that originated in the east, but also possibly from North America. There is also some acid deposition derived locally by power utilities on the Shannon River, principally the coal-fired station at Moneypoint and the oil-fired plant at Tarbet (C. Jordan pers. comm.). Peat-fired power stations in the region are small and are unlikely to contribute a significant sulphur load.

Precipitation chemistry in Ireland shows considerable spatial variation (Jordan and Enlander 1990). In the south-east of the country rainfall is strongly acidic (pH 4.6-4.8) but the average annual weighted mean pH of precipitation for the south-west (Valencia) is around 5.0. This latter value is one of the highest mean pH values recorded for any European country (Bailey *et al.* 1986). Precipitation data for Lough Navar, an EMEP monitoring site in Northern Ireland, but only some 25 km from the west coast, also show an annual weighted mean pH value of around 5.0. As in the south of Ireland, rainfall is more acid in the east of Northern Ireland with an annual weighted mean pH of 4.7 (1986-1988) and values below 4.0 are frequently recorded in the Silent Valley (Mourne Mountains).

Sea salts have a major effect on precipitation chemistry over Ireland; this effect is greatest in the western coastal regions. The effect of location on precipitation quality is demonstrated by comparing data (Table 1.1) from an eastern station (Silent Valley) with those from a north-western site (Lough Navar).

There is also considerable regional variation in non-marine sulphate deposition across Ireland with highest values occurring in the south-west around the Shannon estuary and in the north-east. Lowest deposition occurs in the north-west along the coastal fringe of Donegal.

The high deposition area around the Shannon probably reflects the local point sources of sulphur emissions noted above. According to Jordan and Enlander (1990) sulphur deposition in western Ireland can reach >1 g S m^{-2} yr^{-1} due to local emissions from the Moneypoint and Tarbet power stations. This rate may be too high as data from Gortglass, less than 20 km to the east, indicates a deposition rate of <0.4 g S m^{-2} yr^{-1} (Bowman 1991). The high deposition area in the north-east probably results from the same air masses that cause high deposition values in the south-west of Scotland and north-west of England together with local point sources on the north-east coast. Rainfall and deposited sulphate for point locations across Ireland are given in Table 1.2 and the regional picture of sulphate deposition as interpolated from 13 monitoring sites in 1987 is shown in Figure 1.2. Apart from the two high deposition areas shown on this map other points to note are that the lowest deposition is not in the far south-west and that there is a very sharp decline in deposition from east to west across County Fermanagh.

To assess some of the consequences of acid deposition in western Ireland studies of acid sensitive lakes and streams throughout Ireland (Bowman 1986, 1991) and initial surveys of diatoms in western Ireland (Flower unpubl.) were carried out. Results indicated that lakes in sensitive catchments (usually on granite) were largely unaffected by the apparent low levels of acid deposition impacting the region. Preliminary diatom analysis of a sediment core from Lough Barra, an acid lake in Donegal (Figure 1.1), indicated no recent acidification. In contrast, water chemistry data showed that comparable lakes on the eastern margin of Ireland had non-marine sulphate concentrations at least twice those of western sites and diatom analysis of a sediment core from Blue Lough in the Mourne Mountains of north-east Ireland, showed strong recent acidification (Patrick *et al.* 1989).

Many acid sensitive lakes in western Ireland are less than 200 km from strongly acidified lakes in Galloway, south-west Scotland. This proximity suggests that either sensitive Irish lakes are affected by acid deposition and both the initial study sites and modelled deposition figures are unrepresentative, or that the effects of acid deposition undergo a rapid decline in a westerly direction from south-west Scotland. These possibilities were tested by detailed palaeolimnological analysis of four sites in granitic regions of western Ireland: three in County Donegal and one in County Galway (Figure 1.1).

Table 1.1 *Precipitation quality at two stations in the north of Ireland.* **[Silent valley is in the east and Loch Navar is in the west; data are as weighed annual mean values in µeq l^{-1}; sulphate is non-marine S. Data are supplied by Dr J. Irwin of the Warren Springs Laboratory]**

	H^+		SO_4^{2-}		Ca^{2+}		Cl^-	
	Loch Navar	**Silent Valley**	**Loch Navar**	**Silent Valley**	**Loch Navar**	**Silent Valley**	**Loch Navar**	**Silent Valley**
1988	9.6	14.3	14.0	45.0	21.0	30.0	408.0	136.0
1987	9.5	26.9	16.0	51.0	11.0	21.0	125.0	118.0
1986	11.0	21.0	19.0	41.0	17.0	17.0	293.0	186.0

Table 1.2 *Annual rainfall and deposited non-marine sulphate for selected sites in Ireland.* **[Values refer to mg sulphur deposited m^{-2} yr^{-1}; data supplied by Dr C. Jordan, Freshwater Biological Investigation Unit, Antrim, Northern Ireland.] (r/f = rainfall).**

	Valencia		Rosslare		Belmullet		Loch Navar	
	r/f (mm)	S	r/f (mm)	S	r/f (mm)	S	r/f (mm)	S
1988	1581	0.34	867	0.41	1263	0.51	1992	0.33
1987	1351	0.41	835	0.53	1084	0.53	1144	0.29
1986	1155	0.44	902	0.55	1442	0.45	1439	0.43

Figure 1.1 Study site location map

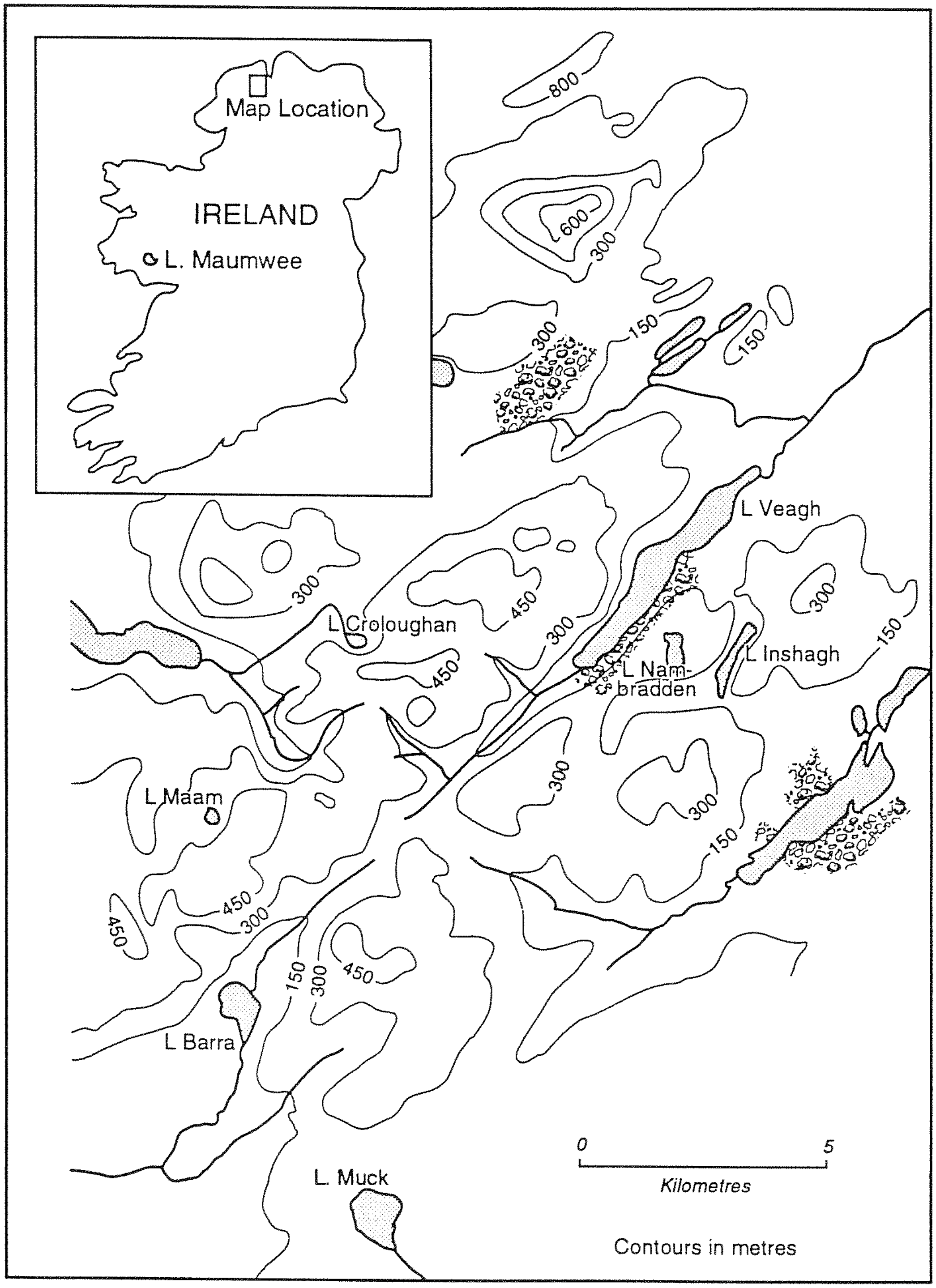
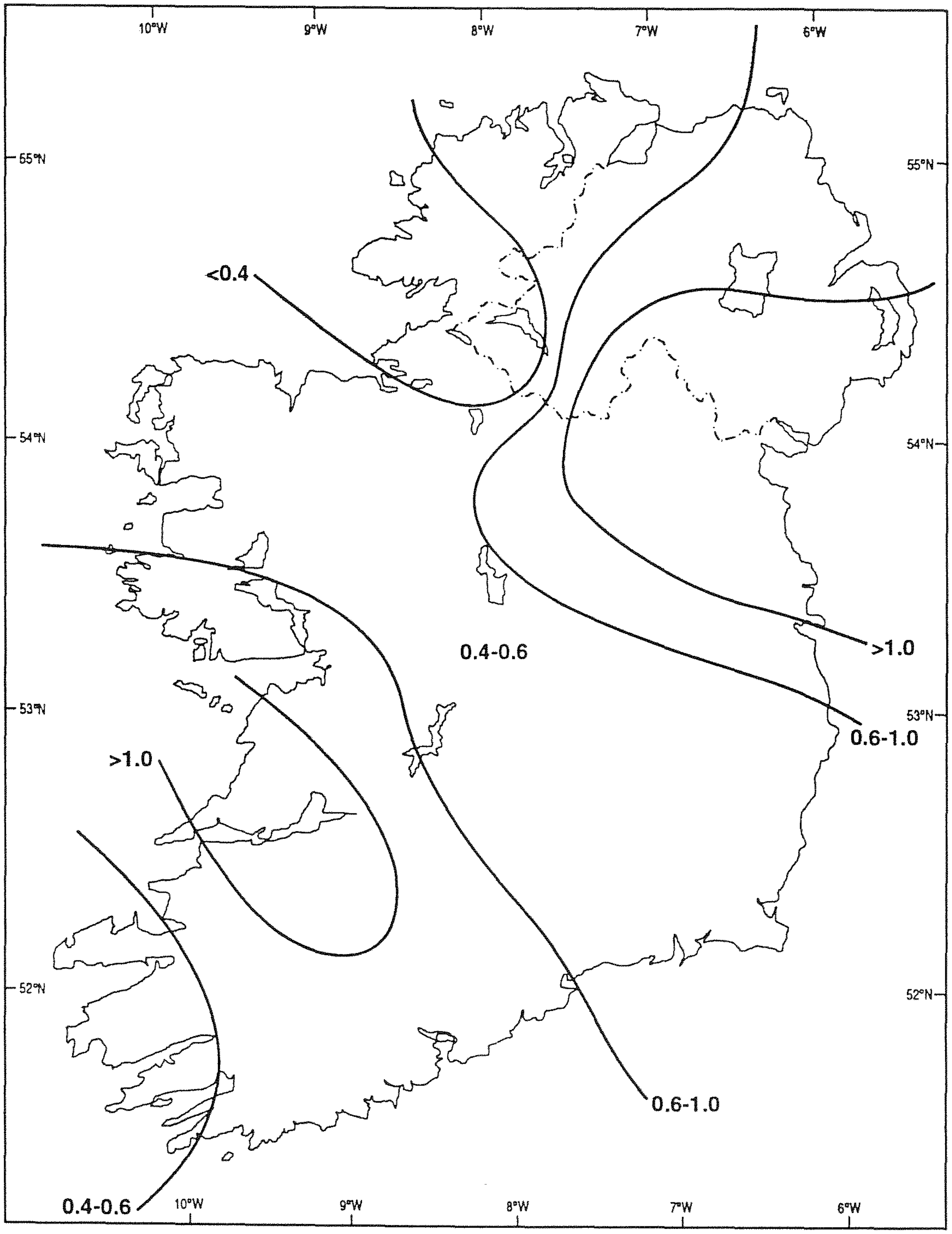


Figure 1.2 Sulphur deposition map of Ireland (g S m^{-2} yr^{-1}) (source: Jordan 1990)



2 LOUGH VEAGH, DONEGAL

Site description

Lough Veagh lies in the Glenveagh National Park in County Donegal, north-west Ireland (Figure 1.1). It lies at an altitude of only 40 m but hills within the catchment rise to over 600 m. The area experiences a cool oceanic climate with mean (1973-1976) annual rainfall of 1500 mm (Telford 1977). Frosts are uncommon.

The entire Lough Veagh catchment lies on Devonian granites and because of their great geological age these rocks are often heavily faulted and contain numerous dolerite and basalt dykes. Catchment soils are dominated by blanket peats, often of considerable thickness and in many areas partially eroded into peat hags (Telford 1977). The blanket peat lies directly on granite rock or on minerogenic, often podzolised, sub-soils. Vegetation over much of the catchment is a typical acid moorland community where *Molinia caerulea* and *Calluna vulgaris* are common. However, there are substantial stands of native oak woodland within the catchment, particularly in the Black Burn valley and on hill slopes around the southern end of the lough. Land-use within the catchment was particularly intense during the early part of the nineteenth century (Telford 1977), being characterised by heavy grazing of sheep. Furthermore, during the 1840s a large number of drainage ditches were constructed, remains of these can still be seen on hillslopes on the west side of the lough. Land-use intensity declined after 1861 when over forty families were ejected from the Glen Veagh estate. However, late in the nineteenth century both red deer and *Rhododendron* were introduced into the estate which have served to degrade further the remaining natural woodland.

Lough Veagh occupies a geological fault line but is glacially over-deepened and is blocked by morainic material at the northern end from where the outflow (Owencarron River) emerges. The lough possesses several sub-basins as shown by the bathymetric map (Figure 2.1); a maximum depth of about 48 m occurs in the southern basin.

Several reports (eg. Macan and Lund 1954, Allott 1985) show that the phytoplankton of Lough Veagh is characterised by *Dinobryon*, *Ankistrodesmus falcatus, Quadrigula* and *Scenedesmus* species and by the diatoms *Cyclotella praetermissa* (*C. radiosa*), *Tabellaria flocculosa* and *Rhizosolenia longiseta*. The phytoplankton was more extensively surveyed between 1987-1989 (Bowman 1991) and maximum phytoplankton biomass (143 mm^3 m^{-3}) was shown to coincide with development of a *Ceratium hirundinella* population. Planktonic and benthic invertebrates are described in Reynolds (1983), Twomey (1984) and Coyle and Murray (1989). The lough supports a good stock of trout but sea trout and salmon runs have been curtailed by installations on the outflow and lower loughs.

The biota of Lough Veagh is indicative of an infertile soft water lake and this view is supported by the water chemistry (Table 2.1 and Bowman 1991). The water chemistry data for May and September 1988 show some interesting differences that can be mainly attributed to relatively greater additions of wind-borne sea salts in the period immediately preceding the May sampling. Hence the early period is characterised by relatively high sodium and chloride concentrations but lower pH and alkalinity. There are only small differences between the concentrations of cations derived mainly from catchment sources (ie. calcium and potassium) on the two sampling dates. Of all the metal ions only aluminium is present in the May 1988 sample at a concentration lower than in the September sample, despite the lower pH. This is probably caused by dilution of the mainly terrestrially-derived aluminium ions by high precipitation that preceeded the May sampling.

Table 2.1 *Water chemistry data for Lough Veagh as sampled in May and September 1988*

		May 1988	September 1988
pH		5.9	6.69
Conductivity	µS cm^{-1}	83.0	*
Alkalinity (Alk$_e$)	µeq l^{-1}	30.0	39.5
Calcium	µeq l^{-1}	80.0	77.0
Magnesium	µeq l^{-1}	144.0	117.0
Sodium	µeq l^{-1}	542.0	385.0
Potassium	µeq l^{-1}	12.0	11.0
Sulphate	µeq l^{-1}	87.0	77.0
Chloride	µeq l^{-1}	690.0	488.0
Aluminium (total acid soluble)	µg l^{-1}	16.0	35.2
Aluminium (labile)	µg l^{-1}	4.0	*
Colour (absorbance)	250 nm	*	0.24

* = not determined

Methods

Sediment cores were collected from Lough Veagh in June 1988 using a mini-Mackereth corer (Mackereth 1969). Several coring positions were investigated: in about 7 m of water towards the north end of the lough; in c 15 m depth in the mid-lough sub-basin and on a fairly flat area of lake bed about 250 m off the eastern shore (Figure 2.1). The first (core VEAG1) contained a considerable amount of sand and gravel throughout the sediment column; the lower portion of the second (core VEAG2) consisted of late-glacial grey clay; but the last (core VEAG3) was composed of fairly homogeneous black organic sediment and used as the master core for dating and diatom analysis. Sediment core analysis was undertaken using techniques described in Stevenson *et al* (1987), and for carbonaceous particles, Rose (1990).

Lithostratigraphy

Core VEAG3 was composed of fine (particles generally <0.1 mm in diameter) black detrital sediment with small amounts of sand and clay and some plant fragments. In the 11-16 cm depth section sediment was distinctly gritty in texture. Profiles of sediment wet density, percentage dry weight and loss on ignition (LOI) are shown in Figure 2.2. There is considerable down-core variation in percentage dry weight and organic matter (as indicated by the LOI curve). The greatest changes begin around 16 cm depth as percentage dry weight increases sharply from average values of around 15% to a peak of almost 40% at 14 cm depth. Organic content shows a corresponding decline from values around 20-25% and reaches a minimum value of c. 10% at 14 cm. However, above this depth organic matter rapidly increases to c. 40% by 11 cm depth, a value which is maintained to the core top.

These down-core changes in percentage dry weight and organic matter reflect important changes in the quantity and composition of particulate material entering the lake from sources within the catchment. The major changes around 15 cm depth may relate to significant land-use changes within the catchment that have resulted in a pulse of minerogenic material entering the lake. This pulse is followed by an increased and sustained influx of organic material, almost certainly derived from accelerated erosion of catchment peat.

Dating

Sediment samples from core VEAG3 were analysed for ^{210}Pb, ^{226}Ra, ^{137}Cs, ^{134}Cs and ^{241}Am by gamma spectrometry (Appleby *et al.* 1986). The ^{210}Pb and ^{226}Ra results are given in Table 2.2 The ^{137}Cs, ^{134}Cs and ^{241}Am results are given in Table 2.3. The ^{134}Cs derives from Chernobyl fallout and has been corrected for decay since 5th May 1986.

Above the layer of dense sediment (11-16 cm depth) unsupported ^{210}Pb activity in the core shows some irregularities but the overall trend is exponential and there is accordingly little systematic variation between dates calculated by the CRS and CIC ^{210}Pb dating models (Appleby and Oldfield 1978). Dates determined by the two models are compared in Figure 2.3. Just above the dense sediment layer, CRS dates are a little older than CIC model dates, but this may simply reflect errors in estimates of the unsupported ^{210}Pb below this level. ^{210}Pb assays within and below the dense sediment layer revealed no unsupported ^{210}Pb, and dates could not be calculated. Dates above this level are given in Table 2.4 and have been calculated using the mean constant sediment accumulation rate of 0.0143 g cm^{-2} yr^{-1} given by the two models.

There is some confirmation of the ^{210}Pb dates from the ^{241}Am record. A small ^{241}Am peak was detected at 3.75 cm (Table 2.3), suggesting a date of c. 1963 for this level. The high surficial ^{134}Cs activity (Table 2.3) shows that the ^{137}Cs record near the core top is dominated by Chernobyl fallout. The mobility of ^{137}Cs within the sediment column is shown by the presence of relatively high ^{137}Cs activities down to depths which clearly predate the 1954 onset of ^{137}Cs fallout from nuclear weapons testing.

The top of the the band of dense sediment between 11 and 16 cm depth (Figure 2.2) is dated to c. 1912. Assuming that this sediment band was the result of soil inwash it is not unreasonable to infer that it relates to a catchment event that began in the middle or late-nineteenth century.

Table 2.2 Lough Veagh: ^{210}Pb and ^{226}Ra data

Depth	Dry mass	^{210}Pb Conc.				^{226}Ra Conc.	
		Total		Unsupp.			
cm	g cm^{-2}	pCi g^{-1}	±	pCi g^{-1}	±	pCi g^{-1}	±
0.25	0.020	38.87	1.51	38.77	1.53	0.10	0.26
2.75	0.232	36.71	1.72	36.11	1.74	0.60	0.28
3.25	0.277	21.01	1.72	19.98	1.74	1.03	0.26
3.75	0.322	16.63	1.15	16.38	1.16	0.25	0.18
5.75	0.500	16.35	1.06	14.99	1.08	1.36	0.23
7.75	0.687	8.63	0.58	7.60	0.60	1.03	0.15
9.75	0.928	6.97	0.45	5.84	0.46	1.13	0.08
10.75	1.054	5.30	0.46	4.20	0.47	1.10	0.10
14.75	2.304	0.60	0.18	-0.69	0.18	1.29	0.04
16.50	2.747	2.41	0.26	0.25	0.27	2.16	0.08
20.50	3.326	1.35	0.17	-0.19	0.18	1.54	0.06

Table 2.3 Lough Veagh: ^{137}Cs, ^{134}Cs and ^{241}Am data

Depth	^{137}Cs Conc.		^{134}Cs Conc.		^{214}Am Conc.	
cm	pCi g^{-1}	±	pCi g^{-1}	±	pCi g^{-1}	±
0.25	45.25	0.59	20.72	0.97	0.00	0.00
2.75	23.10	0.52	0.00	0.00	0.00	0.00
3.25	17.79	0.48	0.00	0.00	0.00	0.00
3.75	11.66	0.33	0.00	0.00	0.08	0.05
5.75	4.25	0.25	0.00	0.00	0.00	0.00
7.75	2.29	0.13	0.00	0.00	0.00	0.00
9.75	1.29	0.08	0.00	0.00	0.00	0.00
10.75	0.92	0.09	0.00	0.00	0.00	0.00
14.75	0.12	0.03	0.00	0.00	0.00	0.00
16.50	0.21	0.04	0.00	0.00	0.00	0.00
20.50	0.01	0.03	0.00	0.00	0.00	0.00

Table 2.4 Lough Veagh: ^{210}Pb chronology

Depth	**Cum. dry mass**	**Chronology**			**Sedimentation rate**		
		Date	**Age**				
cm	**g cm^{-2}**	**AD**	**yr**	**±**	**g cm^{-2}yr^{-1}**	**cm yr^{-1}**	**± (%)**
0.00	0.000	1988	0		0.014	0.145	9.2
0.50	0.041	1985	3	1			
1.00	0.083	1981	7	2			
1.50	0.126	1978	10	2			
2.00	0.168	1974	14	2			
2.50	0.211	1971	17	2			
3.00	0.254	1967	21	2			
3.50	0.299	1964	24	2			
4.00	0.344	1961	27	3			
4.50	0.388	1958	30	3			
5.00	0.432	1955	33	3			
5.50	0.477	1952	36	3			
6.00	0.522	1949	39	4			
6.50	0.569	1946	42	4			
7.00	0.616	1943	45	4			
7.50	0.663	1940	48	4			
8.00	0.717	1937	51	5			
8.50	0.777	1934	54	5			
9.00	0.837	1929	59	5			
9.50	0.898	1925	63	6			
10.00	0.959	1921	67	6			
10.50	1.022	1916	72	7			
11.00	1.090	1912	76	7			

Diatom Analysis and pH reconstruction

Figure 2.4 presents a summary diatom diagram for core VEAG3. Below 16 cm depth the diatom flora is fairly stable, the main variation being in the complementary changes in the relative abundances of two closely related planktonic diatoms, *Cyclotella kuetzingiana* and its variety v. *minor*. *C. kuetzingiana* v. *minor* is not clearly defined in the literature but is recognised here, using the light microscope, by its small size (diameter <10 µm) and an abundance of processes, or indentations/silica nodules and processes in the central area. The prevalence of the nominate variety above 40 cm depth is demonstrated in the diagram and could be of ecological significance. However, this change in the abundance of *Cyclotella* taxa probably reflects only a small change in water quality and, as far as is known, both diatoms have the same pH preference. In the basal two levels, below 60 cm depth, *Achnanthes minutissima* shows a small increase in frequency.

The main diatom changes in this core begin at 16 cm depth as *C. kuetzingiana* sharply declines by almost 50% as several periphytic taxa, most notably *Brachysira vitrea*, increase in abundance. The diatoms, *Aulacoseira distans* v. *tenella* and *A. lirata* v. *alpigena* also increase in abundance in the upper section of the core. The acidophilous diatom *Frustulia rhomboides* v. *saxonica* shows a small abundance peak at 16 cm depth. Other taxa such as *Fragilaria virescens* v. *exigua* show an initial decrease in abundance above 15 cm depth but increase again above c. 6 cm depth. Other taxa show variable responses over the upper 16 cm section of the core.

Compared with diatoms in plankton samples collected in the 1950s and 1970s (Macan and Lund 1954, Flanagan and Toner 1975) the sedimentary diatoms show a relatively low abundance of *Tabellaria flocculosa* and an absence of *Rhizosolenia longiseta*. These differences can probably be accounted for by selectivity and representational problems of plankton sampling and by poor preservation of the former taxon within the sediment.

The changes in diatom composition of the core seem mainly related to the soil inwash event, the record of which begins at about 16 cm depth (see above). The diatom plankton is clearly diminished by this event as would be expected by a decrease in water transparency caused by an increase in turbidity in the water column. Furthermore, the increased abundances of *Aulacoseira* taxa after the initial inwash of minerogenic material could indicate a sustained increase in water colour by dissolved humic material staining the lough water (as might result from increased erosion of blanket peat into the lough).

The lake water pH over the period represented by the core was reconstructed using multiple regression (MR) (Flower 1986) and weighted averaging (WA) (Birks *et al.* 1990) methods (Figure 2.5).

Between 20 cm depth and the core base, both WA and MR methods show that pH deviated little from around 6.3 (Figure 2.5). At 16 cm depth (late-nineteenth century) the MR-inferred pH declines by 0.4 of a pH unit to 5.9 but the WA method indicates only a slight (0.1 pH unit) decline at this depth. Above 16 cm depth inferred pH values increase and by 5 cm depth (1949) reach the pre-disturbance values. A further decline in inferred pH occurs above 5 cm, with the WA method indicating the most change with a 0.2 pH unit decline. The surface sediment inferred pH values of 6.0 (WA) and 6.2 (MR) are within the range of measured lake water values (pH 5.9 - 6.7).

Both the MR and WA methods have a standard error of about 0.3 pH units (Flower 1986, Birks *et al.* 1990) and the inferred pH changes for Lough Veagh are not considered significant. There is no consistent post-1860 acidification trend apparent in this lake. The MR-inferred decline at c 16 cm depth reflects a reduction in the proportion of planktonic *Cyclotella kuetzingiana* at this point, a change probably linked with the inorganic inwash recorded at this depth rather than with a lake water pH change *per se*. We have shown previously (Battarbee and Flower 1985) that inwashed catchment diatoms can cause errors in inferred pH values for lake water. Thus, it is likely that the inwash event had negligible effect on water acidity and that small increases in acidophilous diatoms such as *Frustulia rhomboides* v. *saxonica*, associated with the inwash, were derived from strongly acid environments within the lake catchment. Furthermore, the acidophilous diatoms show declining or inconsistent frequency trends in the most recent sediment (post-1900). The WA method which is less sensitive to changes in acidophilous and indifferent diatom group proportions, indicates very little pH change at around 16 cm depth. It is probable however that the inwash caused a general increase in lake water colour as indicated by the presence of several *Aulacoseira* taxa and the *Cyclotella* decline. A more sophisticated multivariate analysis of the floristic data could produce a significant response to water colour change following the nineteenth century inwash event.

Carbonaceous particle analysis

The carbonaceous particle profile for the core VEAG3 is shown in Figure 2.6. Below 10 cm depth or before the early-1900s contamination was very low with less than 200 particles g^{-1} dry weight of sediment. Above 10 cm depth the carbonaceous particle concentration rises, slowly until 5 cm depth (1952) but then rapidly to a peak of almost 12,000 particles g^{-1} dry sediment at 1 cm depth or the mid-1980s. In the upper 1 cm of sediment the concentration declines sharply to pre-1960 levels.

Pollen analysis

Preliminary examination of the pollen in core VEAG3 showed that preservation was poor and that pollen concentration was low, particularly between 10 and 20 cm depth. Hence, a full analysis of the core was not undertaken.

Geochemistry

There are changes in sediment geochemistry in the gritty sediment layer between 12 and 16 cm (Figure 2.7). Both the potassium and sodium concentrations increase considerably, although the magnesium and calcium concentrations hardly change. Apart from these features, the major cation concentrations vary little over the length of the core.

The main feature of the trace metal profiles is the strong increase in zinc and lead concentration above 12 cm (Figure 2.8). The trace metal concentrations respond little, if at all, to the change in sediment constitution in the 12-16 cm layer, except that zinc and nickel may drop a little. The increase in zinc and lead above 12 cm dated to around 1900, is maintained to the sediment surface. The origin of these trace metals in the lake must be deposition from the atmosphere, as there are no significant wastewater sources at this remote location.

Discussion

Despite stable inferred pH values for Lough Veagh during the nineteenth century, the diatom record indicates that the lough experienced slight but progressive water quality change during this period. This is indicated by the gradual replacement of *C. kuetzingiana* by *C. kuetzingiana* v. *minor*. These species are closely related and have the same optimum pH for the purposes of the WA method of pH reconstruction. The significance of this change is unclear. After the mid-nineteenth century, effects of accelerated sediment inwash, resulting from catchment disturbance, tentatively dated to the late-nineteenth century (16 cm depth) are clearly recorded in the core. The effects of this inwash persist to the present day. As in the case of Loch Teanga in the Outer Hebrides (Flower *et al.* 1992), it is argued that the inwash has probably had little real effect on lake water acidity although the diatom plankton component of the assemblage is reduced following this event. The small increase in acidophilous diatoms after the inwash peak could well represent influx of allochthonous material from the catchment.

Since the mid-nineteenth century the MR-inferred pH for Lough Veagh shows a significant fall coinciding with the inwash event, but there is no significant effect on the WA-inferred pH. This difference arises from the method of calculation and the latter is mostly likely to be correct since it is based on the individual species pH optima. It is the decline of *C. kuetzingiana* which drives the MR pH down, but WA-inferred values show little change. This is because both *C. kuetzingiana* and *B. vitrea* have similar optima in the modern calibration data set (Birks *et al.* 1990). There is no significant change in the inferred pH of Lough Veagh over the post-1900 period, hence acid deposition at this site has been insufficient to cause water acidification.

The cause of accelerated inwash in the late-nineteenth century is unknown but it is thought that land-use changes within the catchment were responsible. Possible causes include increased grazing pressure and consequent peat erosion in the catchment particularly during the eighteenth century (cf. Telford 1977). Alternatively, the establishment of peatland drainage channels on slopes leading down to the lough in the nineteenth century, or the construction of Glen Veagh Castle, could have promoted soil erosion.

Although there is no evidence for recent sustained acidification in Lough Veagh there are several factors which hinder the detection of acidification by atmospheric pollution at this site. Catchment disturbance, probably beginning in the late-nineteenth century has caused significant changes in the diatom assemblage which could mask any small effects resulting from acid deposition. The calcium concentration in this lake is higher than in any of the acidified sites in Galloway and hence the site should be better buffered against the effects of acid deposition. The calcium concentration in Lough Veagh is similar to those in Lochs Chon and Tinker in the Trossachs, sites where the diatom record indicates moderate acidification (Kreiser *et al.* 1992). However, at these sites alkalinity is considerably lower than in Lough Veagh which re-enforces the diatom evidence that the Scottish sites are acidified by acid deposition.

These conclusions are supported by a calculation of the acidification status of the lake using the Henriksen (1982) equation:

where: acidification = 0.93 (Ca + Mg)-14-Alkalinity
for Lough Veagh, = $0.93\,(54 + 12)^{1}$-14-35 = $\underline{12}$
[1] Mean concentrations of Ca and Mg (Table 2.1) less seasalt components.

This acidification value calculated from the relationship between base cations and the alkalinity status in Lough Veagh indicates only slight acidification. The index value is small compared with values calculated for sites known to be strongly acidified in Scotland (cf. Harriman and Wells 1985). The Henriksen acidification index can be modified by factors other than by acid deposition. Sea salts can affect ionic equilibria and if only the low seasalt (low chloride) sample is used to compute the index the result is higher at 16. Dissolved organic carbon (DOC) in lake water can contribute 4-6 μeq of acidity mg^{-1} (Harriman pers. comm.). In sites comparable to Lough Veagh DOC is closely related to absorbance (at 250 nm) and for the Lough Veagh low seasalt sample DOC is about 4.2 mg l^{-1} and contributes at least 17 μeq l^{-1} of acidity. Hence acidification of this lake as inferred by the Henriksen equation can be entirely accounted for by DOC acidity. In Lough Veagh the main source of DOC is clearly catchment peatland drainage.

Application of Battarbee's (1989) diatom acidification model to Lough Veagh also suggests that the lake is not acidified and this remains the case if lake water calcium concentration is corrected for seasalt effects.

Although recent acidification of Lough Veagh can not be demonstrated by diatom or water chemistry analyses, the lake is less oligotrophic than many smaller lakes in the surrounding Donegal uplands. This is presumably because of enhanced supply of base cations and other ions from the basic intrusions which traverse the Lough Veagh catchment. Hence, before discounting any ecological effects of acid deposition in north-west Ireland, sites with lower calcium concentrations and zero alkalinity waters must be examined.

Lough Veagh - conclusions

1) Diatom analysis of the Lough Veagh core has demonstrated that acid deposition has not exceeded the critical loading factor for this site.

2) Application of the Henriksen Equation to Lough Veagh water chemistry indicates slight acidification, but seasalt effects and DOC acidity make the acidification index questionable.

3) The diatom acidification model (Battarbee 1989) predicts the lake to be unacidified.

4) Land-use changes around Lough Veagh have resulted in major inwash of both minerogenic and organic soils. Resulting turbidity effects possibly caused suppression of the planktonic diatom flora.

5) Carbonaceous particle and trace metal records clearly indicate that despite an absence of recent acidification Lough Veagh is moderately contaminated by atmospheric pollution.

Figure 2.1 Lough Veagh: bathymetry

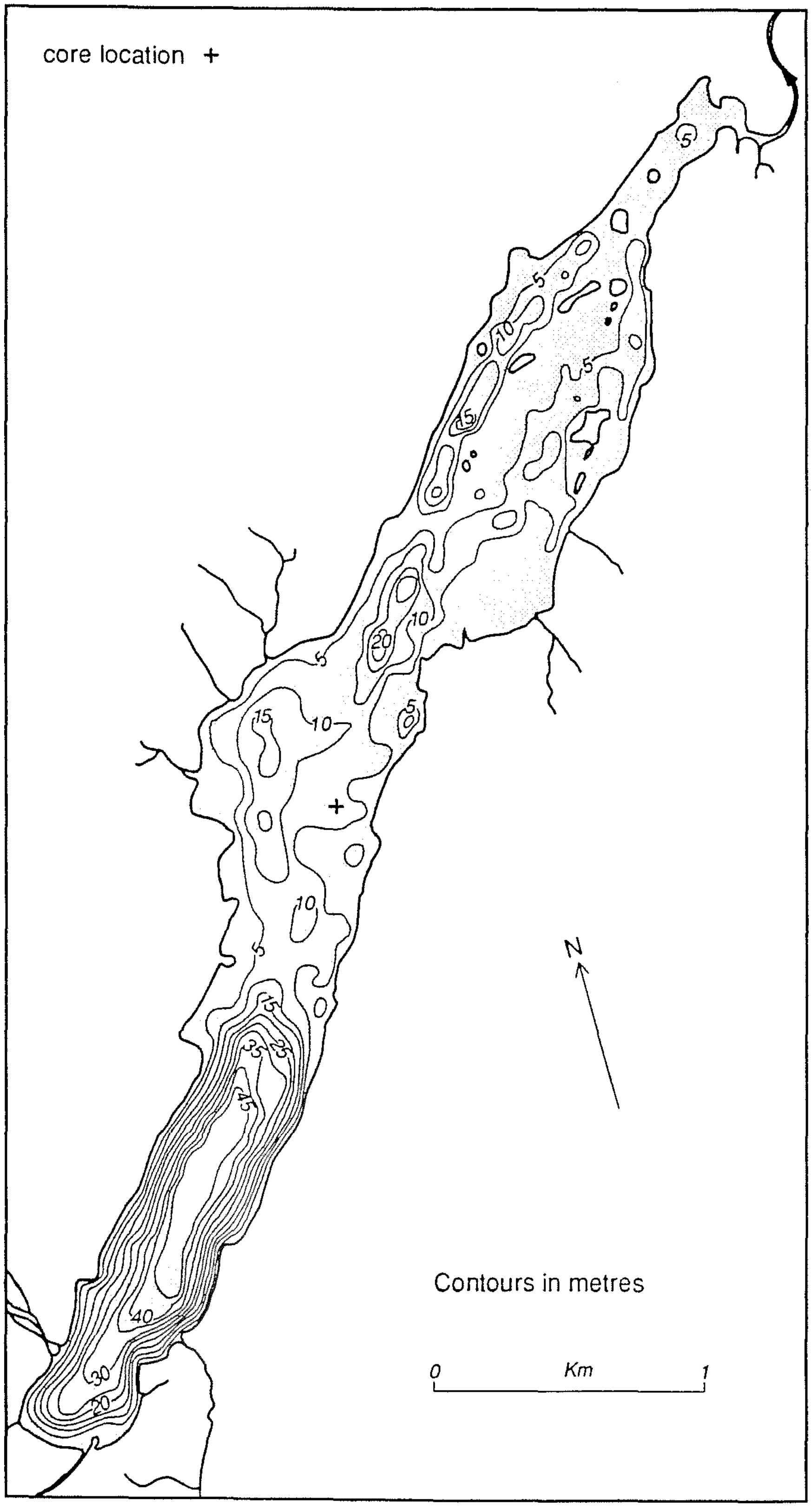


Figure 2.2 **Lough Veagh: Lithostratigraphy**

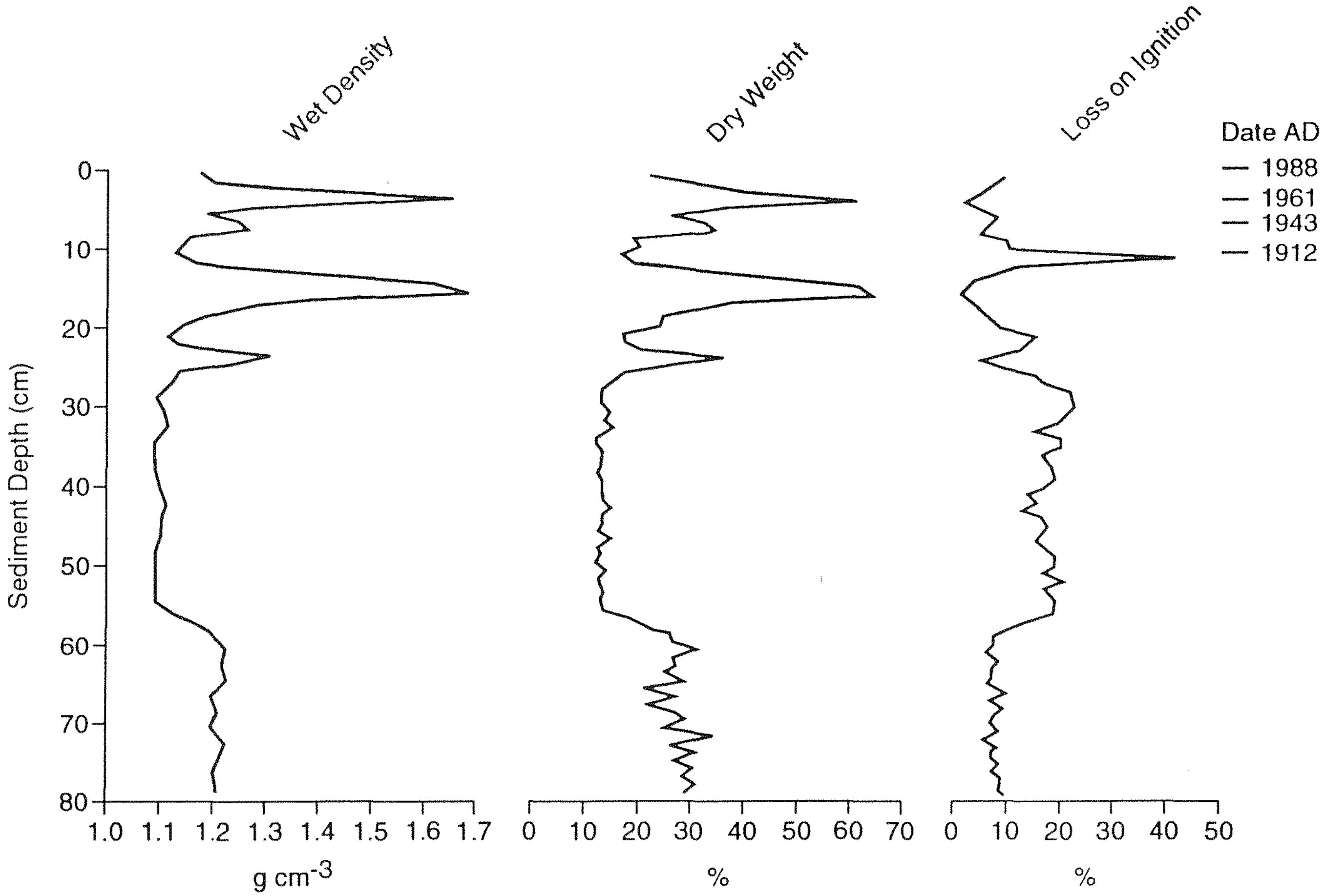


Figure 2.3 Lough Veagh: ^{210}Pb chronology

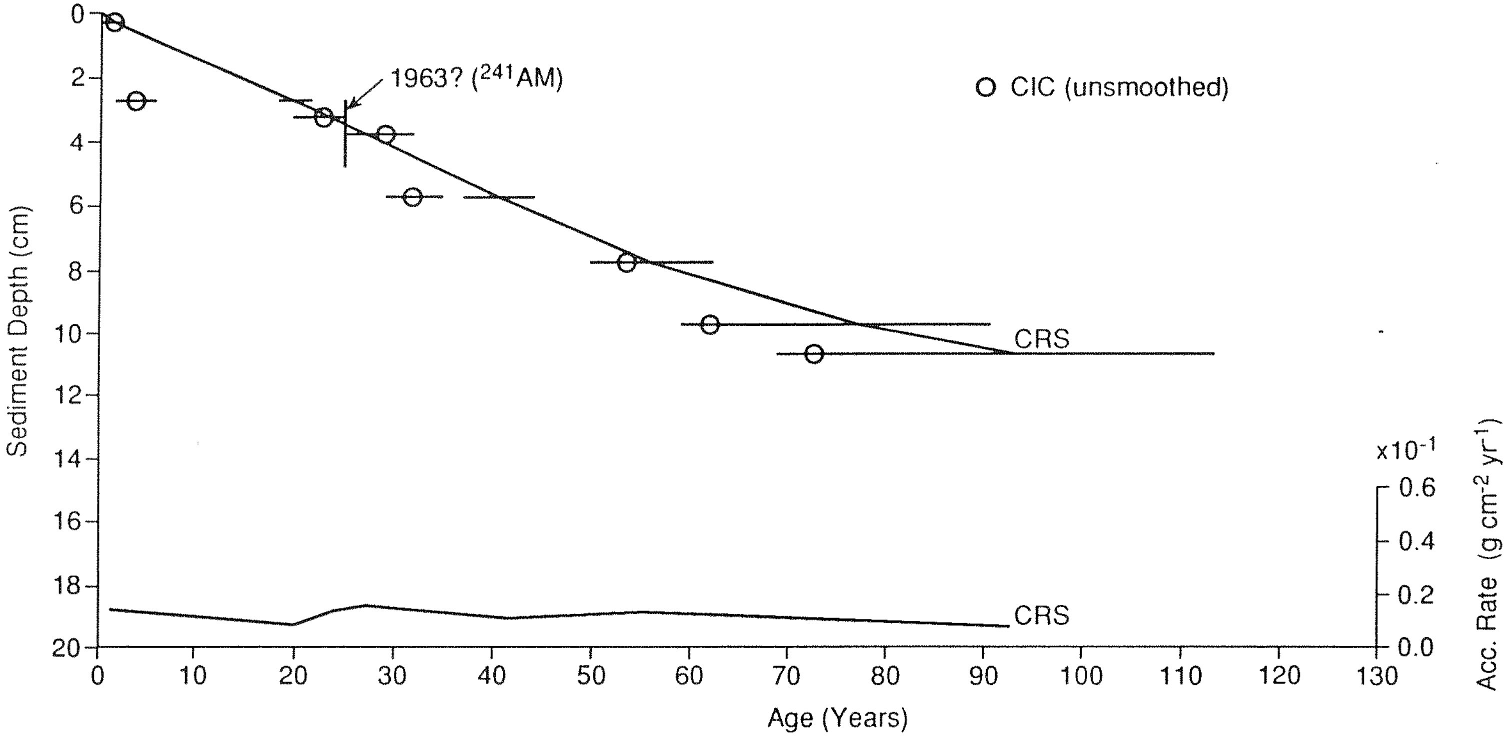


Figure 2.4 Lough Veagh: summary diatom diagram

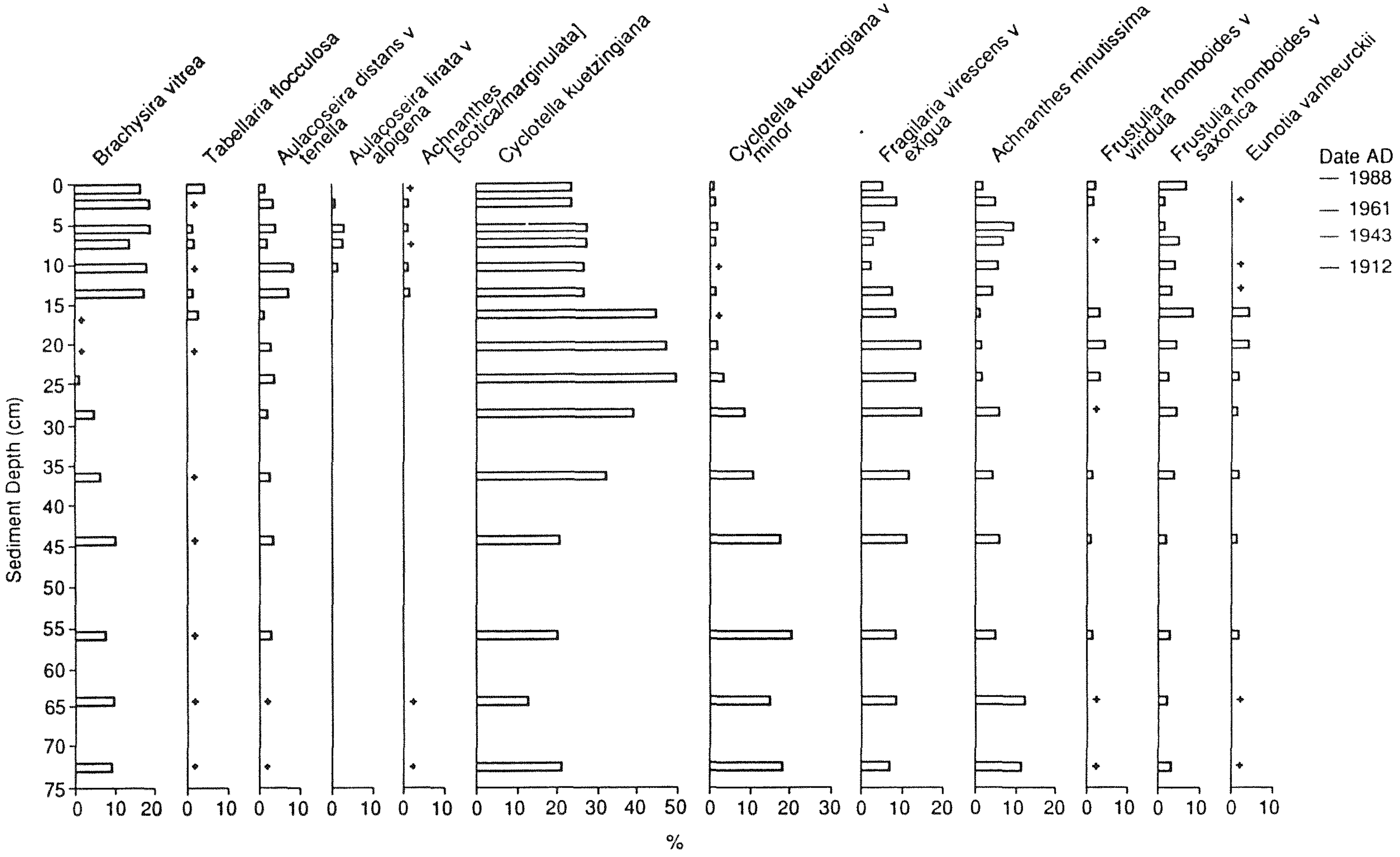


Figure 2.5 Lough Veagh: pH reconstructions

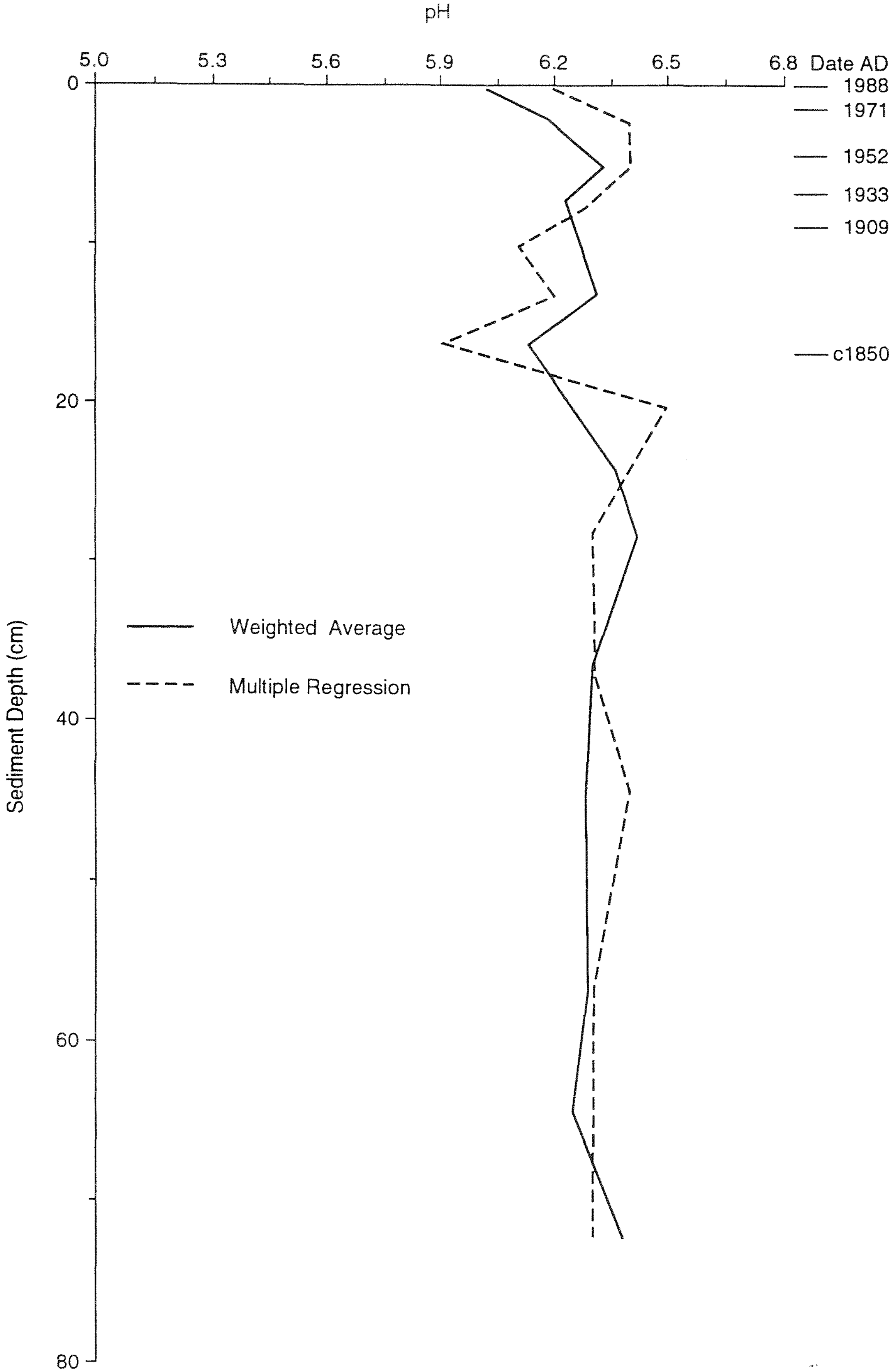


Figure 2.6 Lough Veagh: variation of carbonaceous particle concentrations with depth

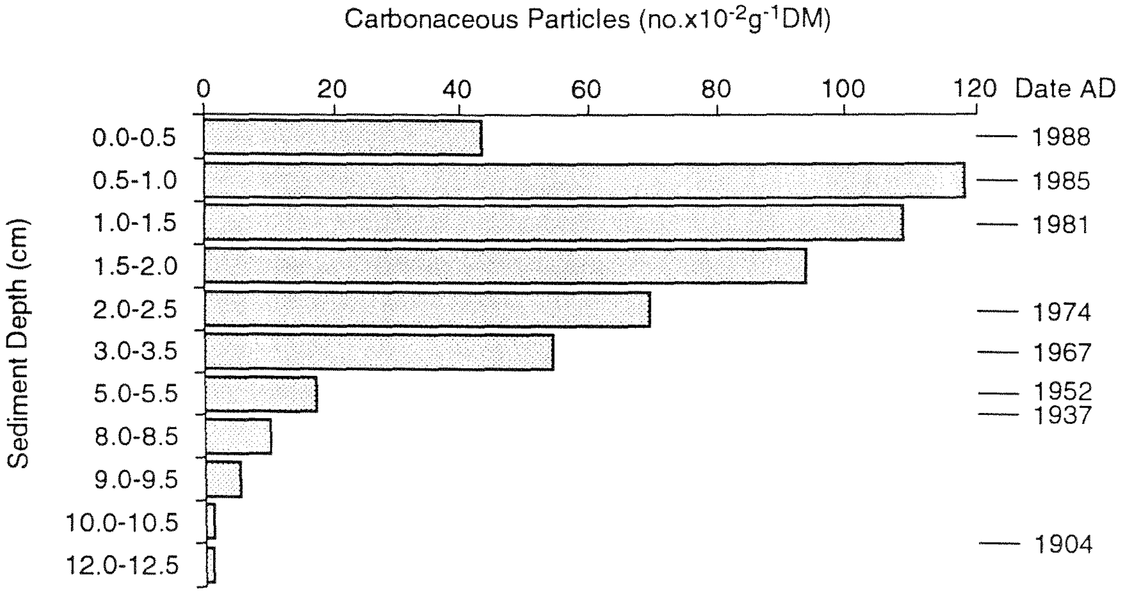
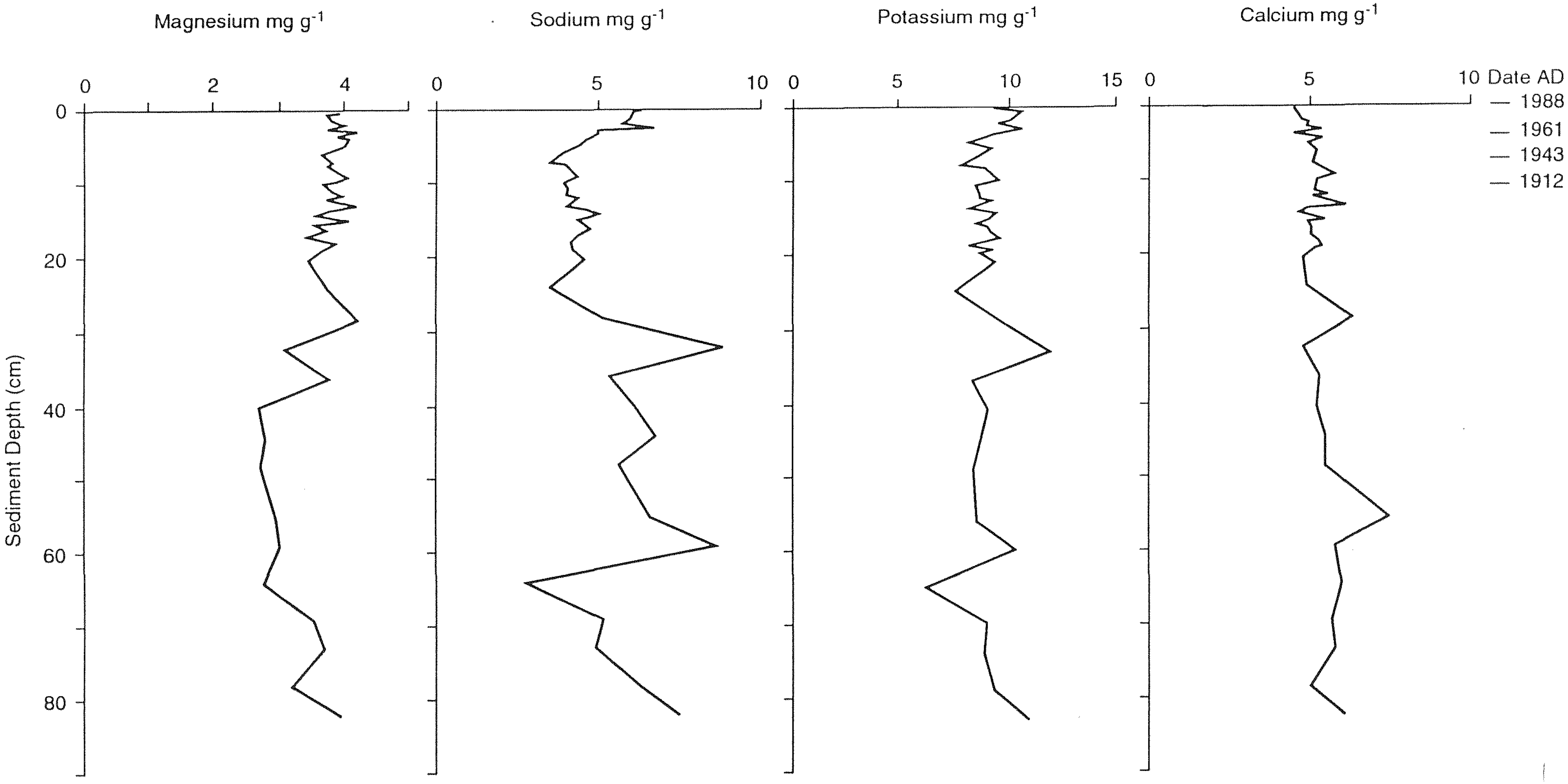


Figure 2.7 Lough Veagh: variation of major ion concentrations with depth



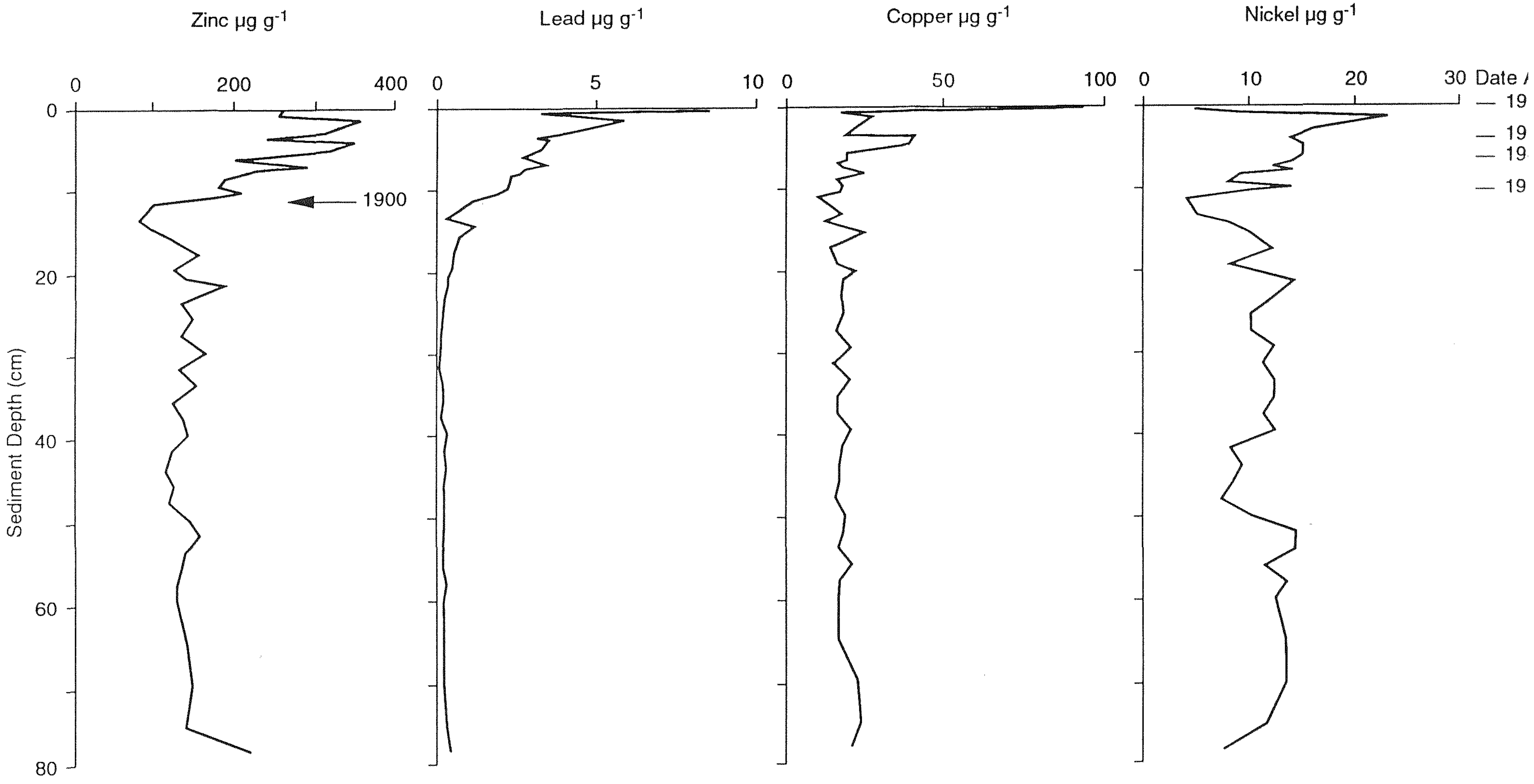


Figure 2.8 Lough Veagh: variation of trace metal concentrations with depth

3 LOUGH MAAM, DONEGAL

Site description

Lough Maam is a small oligotrophic lake located in the granitic uplands of the Glenveagh National Park in Donegal and lies some 10 km south west of Lough Veagh (Figure 1.1). The lake is about 250 m in diameter and has a maximum depth of 8.5 m near its centre; the site was not surveyed in detail but a approximate bathymetry is available (Figure 3.1). It occupies a fairly sheltered position in a corrie on the north side of Slieve Snaght at 436 m altitude. Its catchment is small and consists of a steep granite escarpment rising from the southern margin of the lake; less steep slopes form the east and west margins and the outflow lies at the northern margin. As far as is known there are no basic intrusions within the Lough Maam catchment. Catchment terrain varies between smooth exposures of granite, slipped granite blocks and morainic material. Blanket peat occurs on less steep areas and there is only a small amount of peat erosion in the catchment today. Catchment vegetation is dominated by *Calluna vulgaris*. The lake was well known locally for its trout but apparently these have declined in recent years, presumably as a result of either over-exploitation or poor water quality.

Water chemistry for Lough Maam (Table 3.1) is typical for a site located in an acid moorland catchment influenced by inputs of wind-borne sea salts. The lake water is acid and possessed zero alkalinity on both sampling occasions. However, there were major differences between concentrations of common ions in the May and September samples. Sodium and chloride concentrations in the former sample are more than twice those in the latter period which points to a significant input of sea salts prior to sampling in May. Similarly over half the calcium concentration in the May sample is attributable to marine-derived calcium. The lack of a marked pH depression in the May sample probably reflects the relatively small amount of peaty soils in the catchment for exchange processes with sea salts to operate extensively.

Table 3.1 *Water chemistry data for Lough Maam as sampled in May and September 1988*

		May 1988	September 1988
pH		5.0	5.14
Conductivity	μS cm^{-1}	88.3	*
Alkalinity (Alk_e)	μeq l^{-1}	0.0	0.0
Calcium	μeq l^{-1}	63.8	25.0
Magnesium	μeq l^{-1}	122.0	45.0
Sodium	μeq l^{-1}	507.0	206.0
Potassium	μeq l^{-1}	13.1	8.0
Sulphate	μeq l^{-1}	81.1	52.0
Chloride	μeq l^{-1}	721.9	257.0
Aluminium (total acid soluble)	μg l^{-1}	50.0	44.0
Aluminium (labile)	μg l^{-1}	*	30.1
Colour (absorbance)	250 nm	0.09	0.20

* = not determined

Methods

In May 1987 three sediment cores were collected from the central area of the lake using a mini-Mackereth corer (Mackereth 1969). The cores were analysed using the techniques described in Stevenson *et al* (1987) and an additional analysis, for carbonaceous particles derived from fossil fuel combustion (Rose 1990), was also carried out.

Lithostratigraphy

The appearance of sediment in all three cores was similar, being composed of organic fine black detrital sediment (colour 10YR 2/2) throughout. Profiles of sediment percentage dry weight, LOI and wet density were determined and are shown for each core in Figures 3.2-3.4. The LOI profile for core 3 (MAAM3) shows most variation with a pronounced peak in the upper 25 cm of sediment. The post-LOI peak sediment was absent in core 2 and core 1 was rather short. Hence, core Maam 3 was selected as the master core for further analysis.

In core 3 (Figure 3.4) the LOI values show a strong increase above 22 cm depth from 55% to over 70% between 16 and 10 cm depth. These high values as well as smaller LOI peaks at around 70 and 50 cm depth probably result from inwash pulses of eroded peat. Small peaks in sediment percentage dry weight are caused by minerogenic inwash pulses and occur particularly in the 8-16 cm depth section. Past disturbances within the catchment are probably responsible for these inwash pulses (cf. Stevenson *et al.* 1990). Declining LOI values could reflect catchment stabilization through regeneration of vegetation, possibly following a reduction or cessation of catchment burning and/or sheep grazing. Unlike Lough Veagh there is no evidence of any major inwash pulse of mineral soil at this site.

Dating

The unsupported ^{210}Pb activity versus depth data (Table 3.2) yields a non-linear profile, with a distinct reduction in the gradient above 7 cm depth. Consequently, there is a significant divergence between dates calculated using the CRS and CIC ^{210}Pb dating models (Appleby and Oldfield 1978). The chronology given by these two models are shown in Figure 3.5. Both suggest an acceleration in sediment accumulation rates, but whereas the CRS model indicates a steady acceleration over the past 50 years, the CIC model indicates a more abrupt increase during the 1950s. The core has an unusually low ^{210}Pb inventory of 5.2 pCi cm^{-2}, representing a mean ^{210}Pb flux of 0.16 pCi cm^{-2} yr^{-1}, less than a third of the value for other sites in the region. In view of this it is not possible to choose between the two sets of dates on the basis of ^{210}Pb data alone.

The high surficial ^{134}Cs activity (Table 3.3) shows that the high ^{137}Cs activities near the core top are due to Chernobyl fallout. Using the ^{134}Cs activity to partition the total ^{137}Cs activity into Chernobyl and weapons test components, the latter does not have a well defined sub-surface peak, and it is of little chronological value. There appear to be traces of ^{241}Am (Table 3.3) at a depth of c. 8 cm. A date of 1963 for this level is reasonably consistent with the CIC model dates. The data are however sparse and it was considered safer to construct a mean chronology based on both ^{210}Pb dating models. Dates calculated in this way are given in Table 3.4.

The core chronology indicates that above 7 cm sediment depth (1952) the sediment accumulation rate, though still relatively low, increased almost two-fold. The irregular nature of the LOI profile (Figure 3.4) over this section of the core indicates that both peat and minerogenic inwash has accelerated over this period. However, the highest rate of sediment accumulation is likely to have occurred over the elevated LOI core section between 10 and 25 cm depth which, unfortunately, is not dateable by ^{210}Pb measurement but must precede the twentieth century.

Table 3.2 Lough Maam: ^{210}Pb and ^{226}Ra data

Depth	Dry mass	^{210}Pb Conc.				^{226}Ra Conc.	
		Total		Unsupp.			
cm	g cm^{-2}	pCi g^{-1}	±	pCi g^{-1}	±	pCi g^{-1}	±
0.25	0.019	11.44	0.95	11.44	1.02	0.00	0.38
2.25	0.149	10.34	0.98	9.21	1.07	1.13	0.44
3.75	0.244	7.20	0.94	6.00	0.99	1.20	0.32
4.75	0.317	8.68	0.63	7.85	0.67	0.83	0.23
5.75	0.389	6.47	0.60	4.98	0.67	1.49	0.30
6.75	0.465	7.42	0.76	6.12	0.79	1.30	0.23
7.75	0.549	6.38	0.65	4.92	0.71	1.46	0.28
8.75	0.628	4.62	0.71	3.44	0.81	1.18	0.39
10.75	0.782	3.26	0.33	2.08	0.35	1.18	0.12
13.75	1.032	0.96	0.27	0.26	0.29	0.70	0.11
16.75	1.274	1.01	0.34	0.29	0.36	0.72	0.12
19.25	1.471	0.61	0.51	-0.11	0.54	0.72	0.19

Table 3.3 Lough Maam: ^{137}Cs, ^{134}Cs and ^{241}Am data

Depth	^{137}Cs Conc.		^{134}Cs Conc.		^{214}Am Conc.	
cm	pCi g^{-1}	±	pCi g^{-1}	±	pCi g^{-1}	±
0.25	13.56	0.36	4.81	0.69	0.09	0.04
2.25	4.01	0.36	0.00	0.00	0.00	0.00
3.75	1.71	0.19	0.00	0.00	0.06	0.04
4.75	1.16	0.19	0.00	0.00	0.00	0.00
5.75	0.96	0.21	0.00	0.00	0.00	0.00
6.75	1.30	0.16	0.00	0.00	0.00	0.00
7.75	0.71	0.21	0.00	0.00	0.11	0.04
8.75	0.61	0.24	0.00	0.00	0.10	0.05
10.75	0.39	0.11	0.00	0.00	0.00	0.00
13.75	0.24	0.06	0.00	0.00	0.00	0.00
16.75	0.11	0.07	0.00	0.00	0.00	0.00
19.25	0.00	0.00	0.00	0.00	0.00	0.00

Table 3.4 Lough Maam: ^{210}Pb chronology

Depth	Cum. dry mass	Chronology			Sedimentation rate		
		Date	Age				
cm	g cm^{-2}	AD	yr	±	g cm^{-2}yr^{-1}	cm yr^{-1}	± (%)
0.00	0.000	1988	0				
0.50	0.035	1985	3	1			
1.00	0.068	1983	5	2			
1.50	0.100	1981	7	2			
2.00	0.133	1979	9	2			
2.50	0.165	1976	12	2			
3.00	0.197	1974	14	2			
3.50	0.228	1972	16	2	0.014	0.203	10.4
4.00	0.262	1969	19	3			
4.50	0.298	1967	21	3			
5.00	0.335	1964	24	3			
5.50	0.371	1961	27	3			
6.00	0.408	1959	29	4			
6.50	0.446	1956	32	4			
7.00	0.486	1952	36	5			
7.50	0.528	1945	43	6			
8.00	0.569	1940	48	7			
8.50	0.601	1935	53	8			
9.00	0.648	1930	58	10	0.008	0.094	20.6
9.50	0.686	1925	63	13			
10.00	0.724	1919	69	14			
10.50	0.763	1914	74	15			
11.00	0.804	1909	79	16			
11.50	0.845	1903	85	18			

Diatom analysis and pH reconstruction

A summary diatom diagram is presented in Figure 3.6. Between the core base and 23 cm depth the diatom taxa show very little change in their frequency distributions. *Brachysira vitrea* and *Fragilaria virescens* v. *exigua* are the most common taxa over this section of the core. At, and above 21 cm depth marked changes in frequency occur. Initially the frequency of *F. virescens* v. *exigua* almost doubles and *Aulacoseira lirata* v *alpigena* disappears from the record. The frequency of *F. virescens* v. *exigua* begins to decline at around 13 cm depth and again at c. 11 cm. The upper depth falls within the ^{210}Pb sediment chronology which dates the change to 1909. At this time other changes occur: *Cymbella lunata* and *Achnanthes minutissima* decline whilst *Cymbella perpusilla* begins to increase. By 7 cm depth or 1952, *F. virescens* v. *exigua* has declined markedly and corresponds with a decline in the frequency of *Brachysira vitrea*. Also at this time *Achnanthes marginulata* and *Navicula leptostriata* show small increases in abundance. *Frustulia rhomboides* v. *saxonica* increases from the mid-1960s (c. 5 cm depth) and in the most recent (upper 3 cm) sediment which post-dates 1974, frequencies of *Tabellaria quadriseptata*, *Brachysira brebissonii* and *Eunotia flexuosa* increase.

The lake appears to have never supported a significant diatom phytoplankton community within the past few hundred years. However, a few planktonic *Cyclotella* and possibly planktonic *Aulacoseira* taxa occur in the basal section of the core.

The down core variation in diatom concentration (Figure 3.7) is rather irregular but with no major changes and diatom preservation is good throughout the core. There is however some correspondence between changes in sediment LOI and diatom abundance. The earliest diatom concentration decline is at 55 cm depth which coincides with a small peak in organic matter (Figure 3.4). Above 45 cm depth there is another, but larger, decline in diatom abundance which also corresponds with small peaks in organic matter as well as small changes in sediment percentage dry weight (Figure 3.4). At around 26 cm depth, organic matter increases markedly and the peak value coincides with the lowest diatom concentration value recorded in the core. Above this point diatom concentration increases but levels-off at c. 11 cm depth (early-twentieth century) and then fluctuates to the core top.

The declines in diatom concentration are most probably caused by inwash of peat and occasionally minerogenic material from the catchment. These erosional pulses have occurred at various periods during the recent history of Lough Maam and began well before the period spanned by ^{210}Pb dating.

pH was reconstructed from the sedimentary diatom assemblages using both the MR (Flower 1986) and the statistically more robust, WA (Birks *et al.* 1990) methods (Figure 3.8). The inferred pH profiles produced by the two methods show similar trends but WA pH values are consistently lower than those calculated by MR, by between 0.6-0.8 pH units. The current measured pH of Lough Maam is between 5.0-5.14 and is very close to the WA-inferred pH of 5.2 for the most recent sediment. The difference is well within the root mean squared error of the WA method (Birks *et al.* 1990). This suggests that the WA reconstruction technique is the most appropriate for this site. The overestimation of pH produced by the MR method is probably a result of the relatively high percentages of pH indifferent taxa in the Lough Maam sediments.

Both pH profiles indicate that Lough Maam is a recently acidified lake and that the acidification began some 90-100 years ago. The WA data show a pH decline of about 0.4 pH units (pH 5.6 to 5.2) from the late-nineteenth century. There is no evidence of any recovery and reversal of pH decline in the most recent sediment.

Carbonaceous particle analysis

The down-core variation in carbonaceous particle concentrations (Figure 3.9) shows that these particles increase gradually in abundance throughout the core from near detection point concentration at 9.5 cm depth (1925). Concentrations rise gradually until 5 cm depth (1964) and then sharply increase up to a maximum concentration of more than 8,000 particles g^{-1} dry sediment at 1 cm depth (1983). A small fall-off occurs in the upper 1 cm of sediment representing the post-1983 period.

Pollen analysis

The major feature of the summary pollen frequency diagram (Figure 3.10) is the low level of arboreal pollen throughout the core. Similarly, there is little change in frequencies of *Calluna vulgaris* and Gramineae throughout the core (Figure 3.11) with *Calluna* contributing around 60% to the total pollen sum. Of the aquatic pollens (Figure 3.12), *Isoetes lacustris* spores are low throughout but show a small decline above about 40 cm depth.

The vegetation record indicates that land-use change has been minimal at this site. If catchment management was intensified (eg. for sheep grazing) the *Calluna* : Gramineae ratio would be expected to increase towards the core top. The boulder-strewn and precipitous nature of the catchment makes it highly unsuitable for exploitation by sheep although it is likely that burning of the catchment vegetation has occurred occasionally in the past. The pollen diagram gives little evidence of the inwash events suggested by the LOI profile, only the *Isoetes* decline at c. 40 cm depth could provide evidence for turbidity/siltation effects of the first inwash event. There is no pollen record of the more major and recent inwash beginning at c. 26 cm depth. *Isoetes* abundance is considered to be susceptible to erosional inwash, but in this lake it can only grow in a very limited marginal location, and its spore frequencies are unlikely to provide a particularly good record of catchment erosion.

Geochemistry

The magnesium and sodium concentrations drop between 10 and 20 cm, with magnesium increasing strongly above 10 cm, while sodium returns to values found deeper in the core (Figure 3.13). The calcium concentration also rises above 10 cm and all the major cation concentrations drop towards the sediment surface. These changes appear to be associated with variations in peat erosion.

All the trace metal concentrations, especially zinc and lead, increase above 14 cm (Figure 3.14). Because of the changes in sediment constitution above 20 cm, as shown by the cation results, the depth at which trace metal contamination commences in Lough Maam must be selected carefully. If it is assumed that the changes in sediment constitution affect both the major cation and trace metals, then the inter-relationships between these species can be used to establish the point at which trace metal contamination begins. Figure 3.15 shows that zinc and lead contamination starts at around 14 cm depth, which corresponds to the middle of the nineteenth century. The only source of trace metal contamination in this remote lake is deposition from the atmosphere.

The trace metal results show that Lough Maam has been contaminated by material deposited from the atmosphere since last century. It is assumed that the trace metal profile in a sediment core is a surrogate for acid stress and as there is no drop in zinc concentration towards the sediment surface, as there is in strongly acidified lakes, the results indicate that the lake pH has not fallen below 5.0.

Discussion

The diatom record and inferred pH history of Lough Maam show that the lake has been recently acidified and that the acidification process began around the turn of this century. The overall pH decline since c. 1900 is between 0.4 and 0.5 pH units, depending on the pH reconstruction method used. This decline is statistically significant (cf. Birks *et al.* 1990) and indicates a moderate lake acidification which contrasts to unacidified Lough Veagh nearby. The degree of acidification is however considerably less than that found in sensitive lakes in heavily polluted areas such as Galloway, south-west Scotland. Lough Maam began to acidify rather later than these heavily impacted sites which suggests that either acid atmospheric pollution was later arriving in north-west Ireland or that acidic deposition rates were initially insufficient to exceed the natural buffering capacity of the Lough Maam system. The latter explanation is favoured since low calcium concentration (Table 3.1) indicates that this lake is as sensitive as most of the Scottish sites. On the other hand trace metal contamination typically preceeds acidification and this is the case for Lough Maam where contamination began in about the mid-nineteenth century.

As assessed by the carbonaceous particle concentration and trace metal records, Lough Maam is somewhat less contaminated than Lough Veagh, but almost an order of magnitude less than strongly acidified lakes in south-west Scotland. Compared with Loch Teanga in the Outer Hebrides where acidic deposition is low (Flower *et al.* 1992) Lough Maam is about three times more contaminated. However, the timing of the major increase in particle contamination in the 1970s is similar between sites. The results substantiate earlier observations that recent levels of particle contamination accord with the degree of atmospheric pollution as assessed by the rate of non-marine sulphate deposition.

Another approach to assessing the acidification status of Lough Maam is to use the water chemistry data and apply the equation of Henriksen (1982). Using the mean of the two samples from Table 3.1 this gives a low acidification value of 9. However, one sample is strongly affected by a seasalt intrusion. If only the water chemistry sample with low sea salts is used in the calculation then the acidification index becomes zero, indicating no acidification. If dissolved organic carbon (DOC) acidity is taken into account the index becomes negative indicating surplus acid neutralizing capacity. Compared with the diatom evidence of an acidification of around 0.4 pH units the Henriksen equation clearly underestimates acidification and this underscores the problem of applying the equation to sites where water chemistry data are limited and sea salts dominate water quality.

The potential acidification status of Lough Maam can also be estimated by applying the diatom acidification model of lake water calcium concentration to total deposited non-marine sulphur (Battarbee 1989). This model predicts that if the mean lake water calcium concentration and a sulphur deposition rate of 0.5 g m^{-2} yr^{-1} are used, then Lough Maam should not be acidified. Underestimation of the rate of deposited sulphur in the Lough Maam catchment will cause the model to under-predict acidification, but if the lowest seasalt corrected calcium concentration, rather than mean calcium concentration in lake water is used, the model then clearly suggests that the lake is acidified and so accords with the diatom-based pH reconstruction.

The lithostratigraphic changes in the Lough Maam core provide evidence of changes in land-use which began well before the period encompassed by the ^{210}Pb chronology. However, it is clear that the main peat inwash period pre-dates, and is unrelated to, onset of lake acidification at the turn of this century.

Lough Maam: Conclusions

1) Diatom analysis of the Lough Maam core has demonstrated that acid deposition exceeded critical loading for the site some 80 years ago and the lake is now moderately acidified.

2) Application of the Henriksen Equation to Lough Maam water chemistry indicates slight or no lake acidification.

3) The diatom acidification model (Battarbee 1989) predicts the lake to be unacidified, but when lake water calcium values are corrected for the effect of sea salts or the lowest calcium values are used, the model correctly predicts its acidification.

4) The timing of land-use changes around Lough Maam is unrelated to the chronology of acidification.

5) Carbonaceous particle and trace metal records indicate that Lough Maam is moderately impacted by atmospheric pollution.

Figure 3.1 Lough Maam: bathymetry

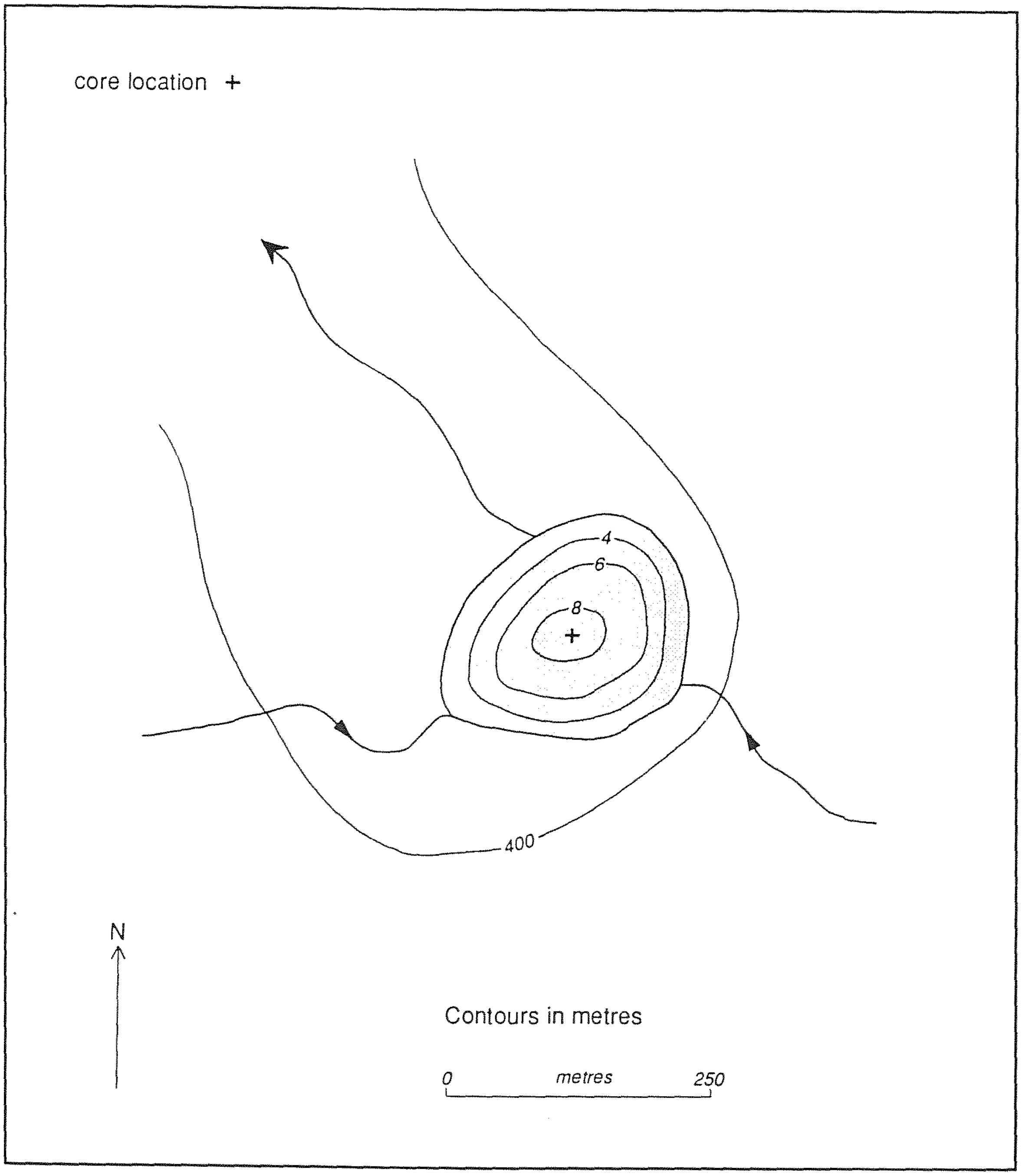


Figure 3.2 Lough Maam: lithostratigraphy core MAAM1

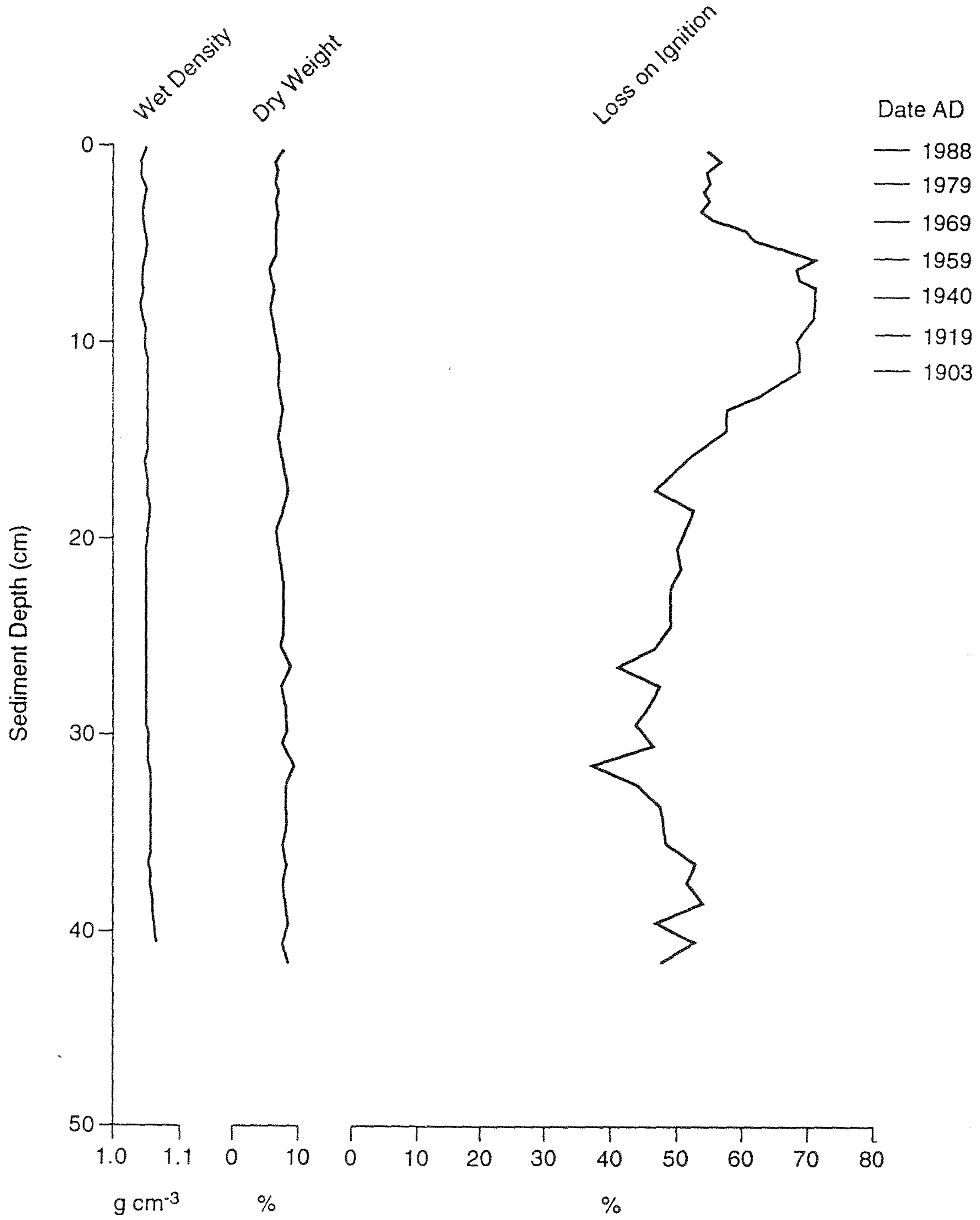


Figure 3.3 Lough Maam: lithostratigraphy core MAAM2

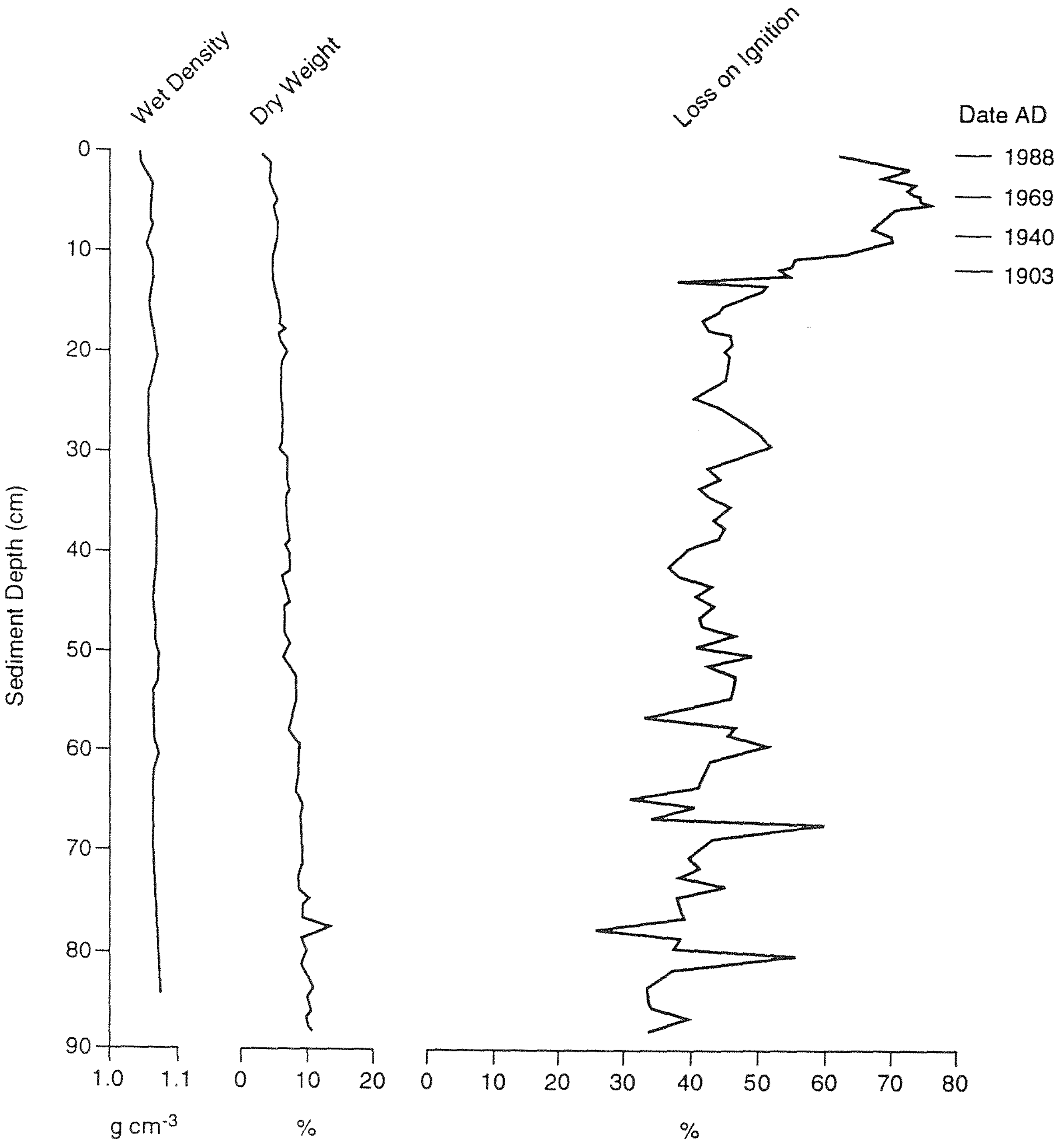


Figure 3.4 Lough Maam: lithostratigraphy core MAAM3

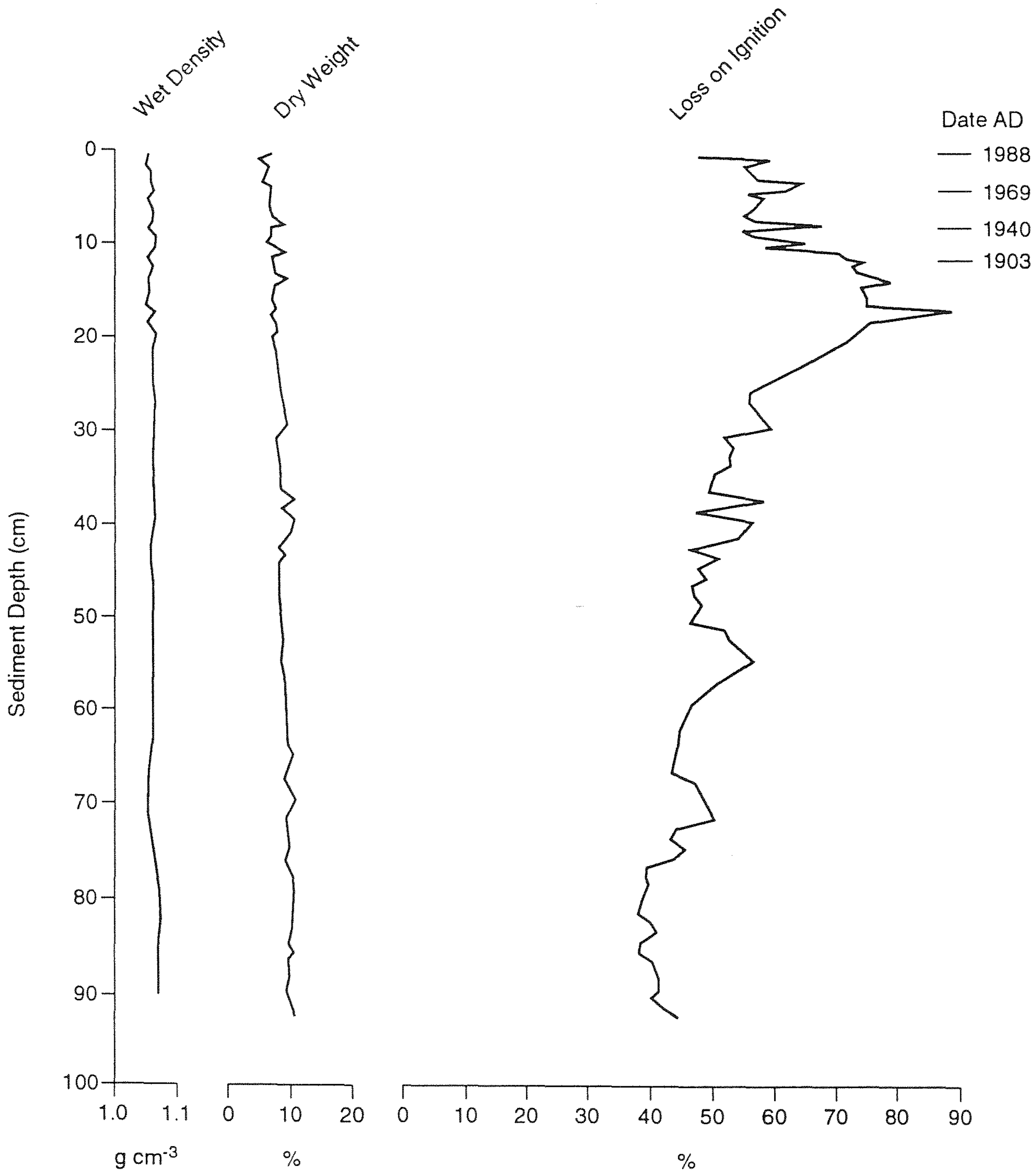


Figure 3.5 **Lough Maam: ^{210}Pb chronology**

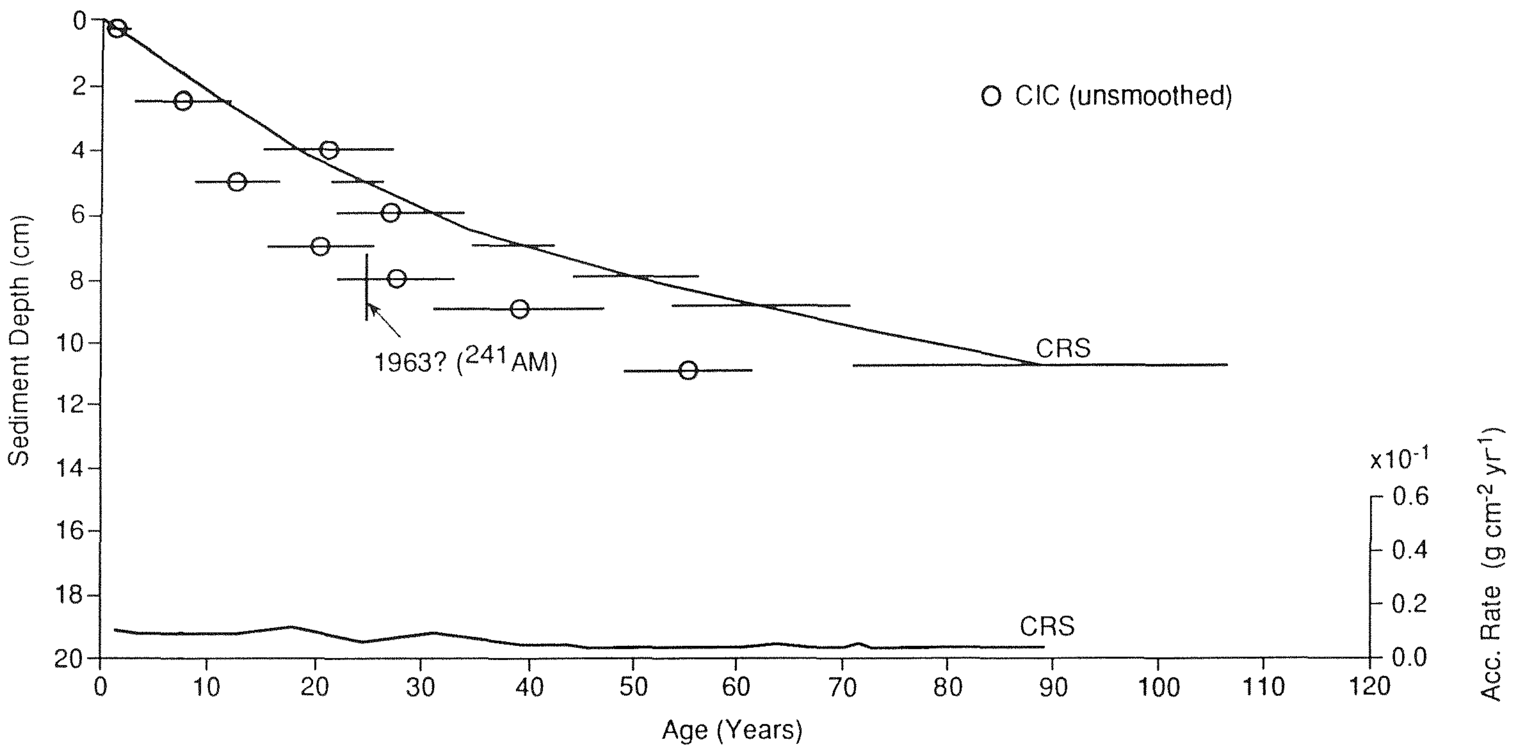


Figure 3.6 Lough Maam: summary diatom diagram

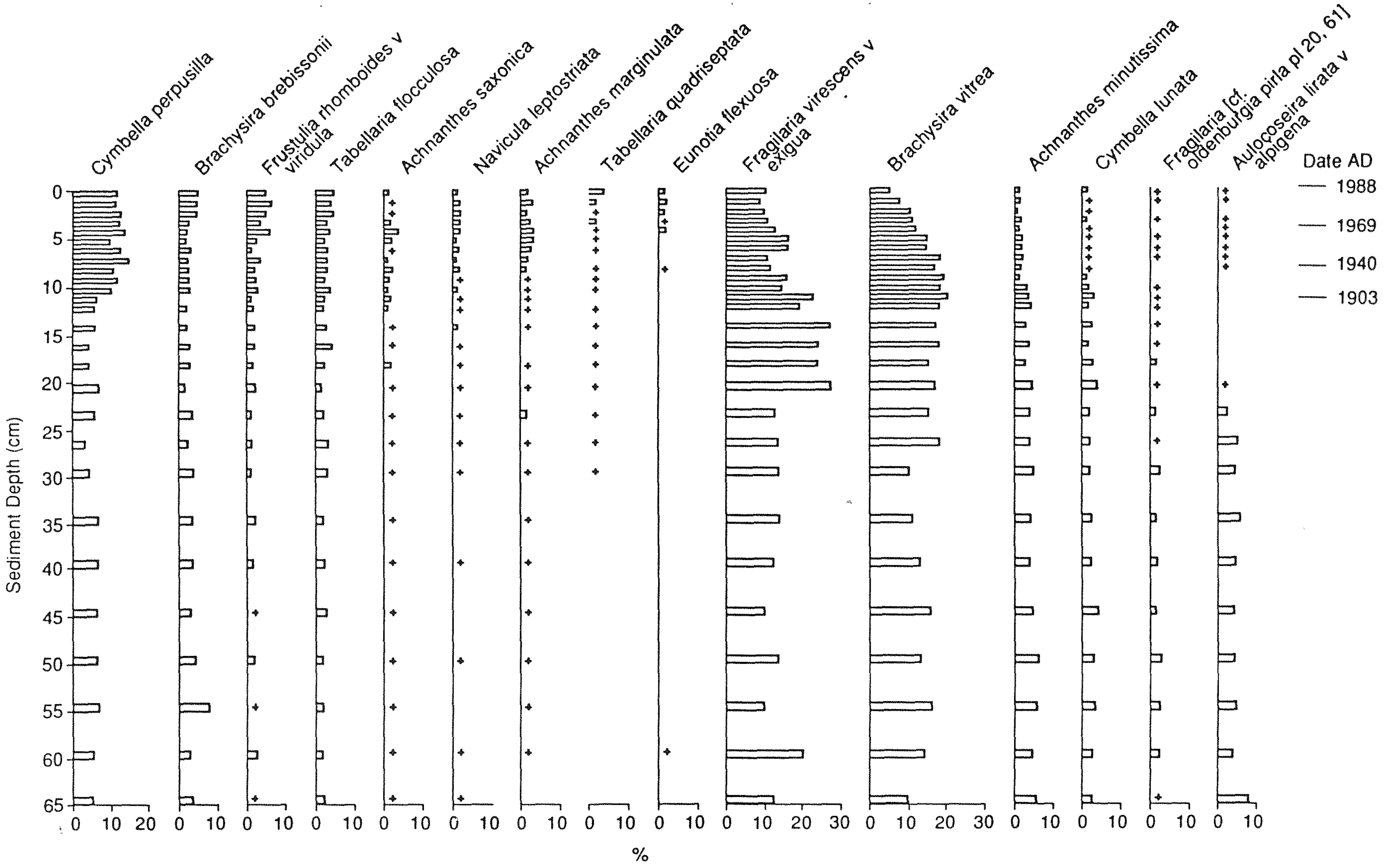


Figure 3.7 Lough Maam: variation of diatom concentrations with depth

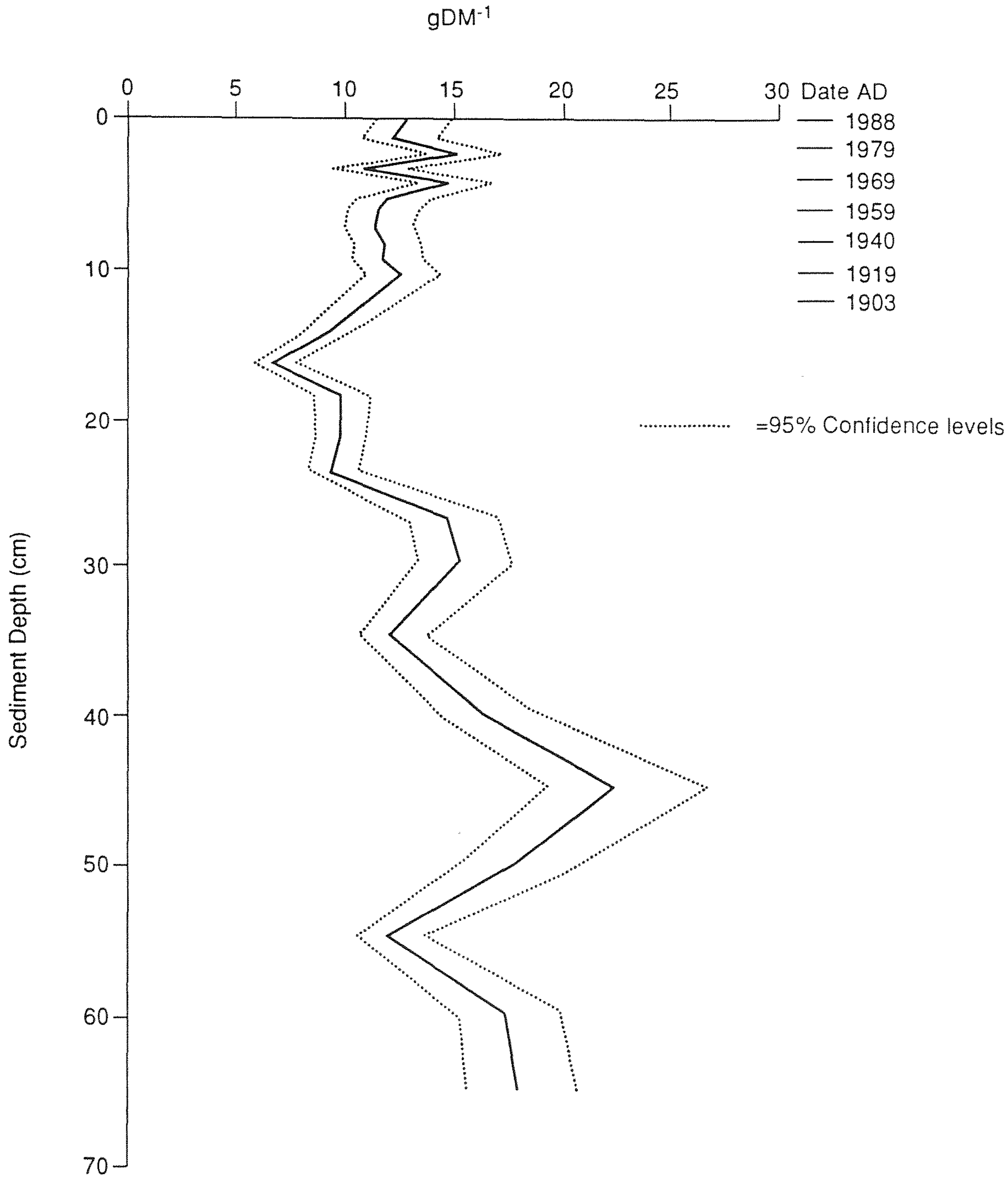


Figure 3.8 Lough Maam: pH reconstructions

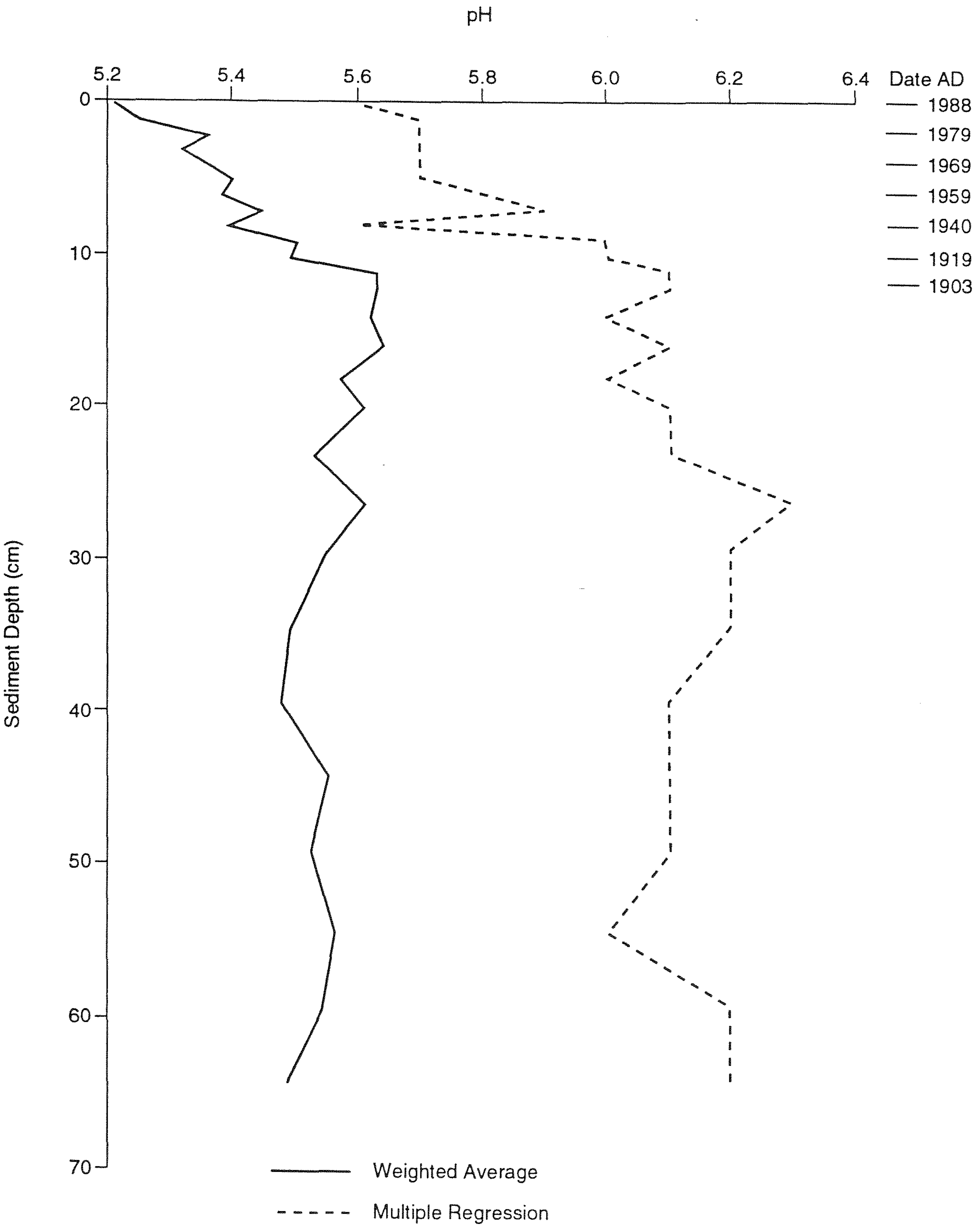


Figure 3.9 Lough Maam: variation of carbonaceous particle concentrations with depth

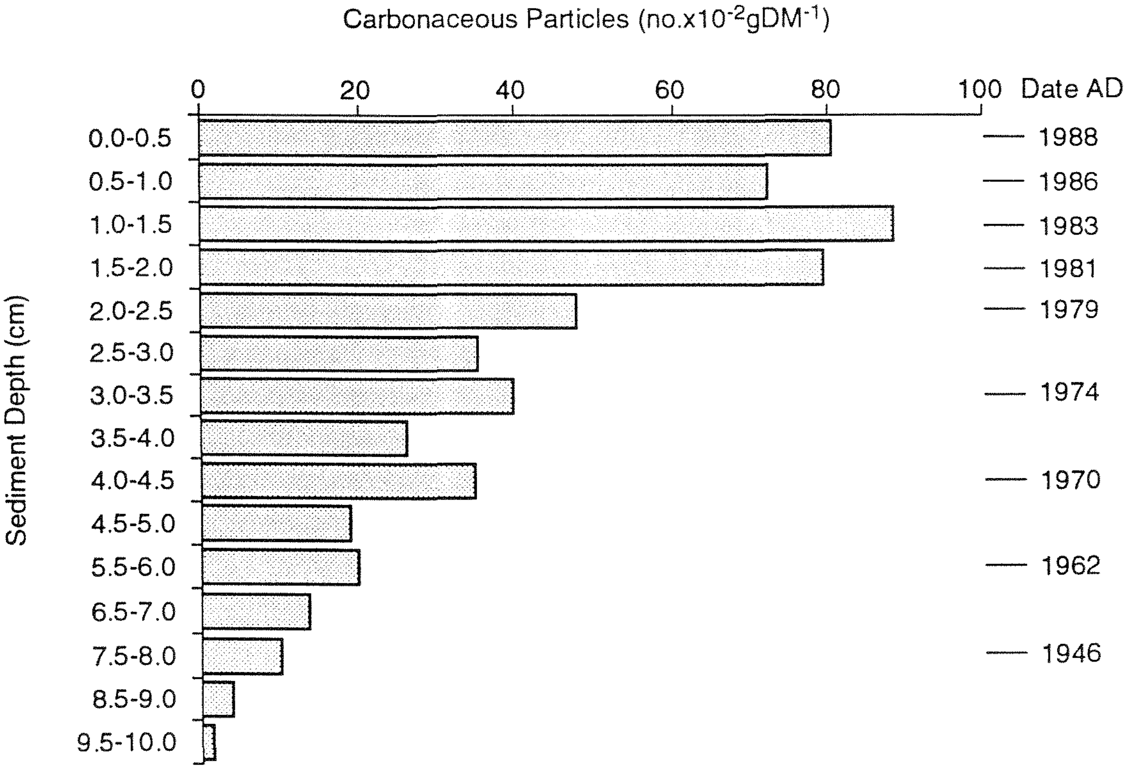


Figure 3.10 Lough Maam: summary pollen diagram

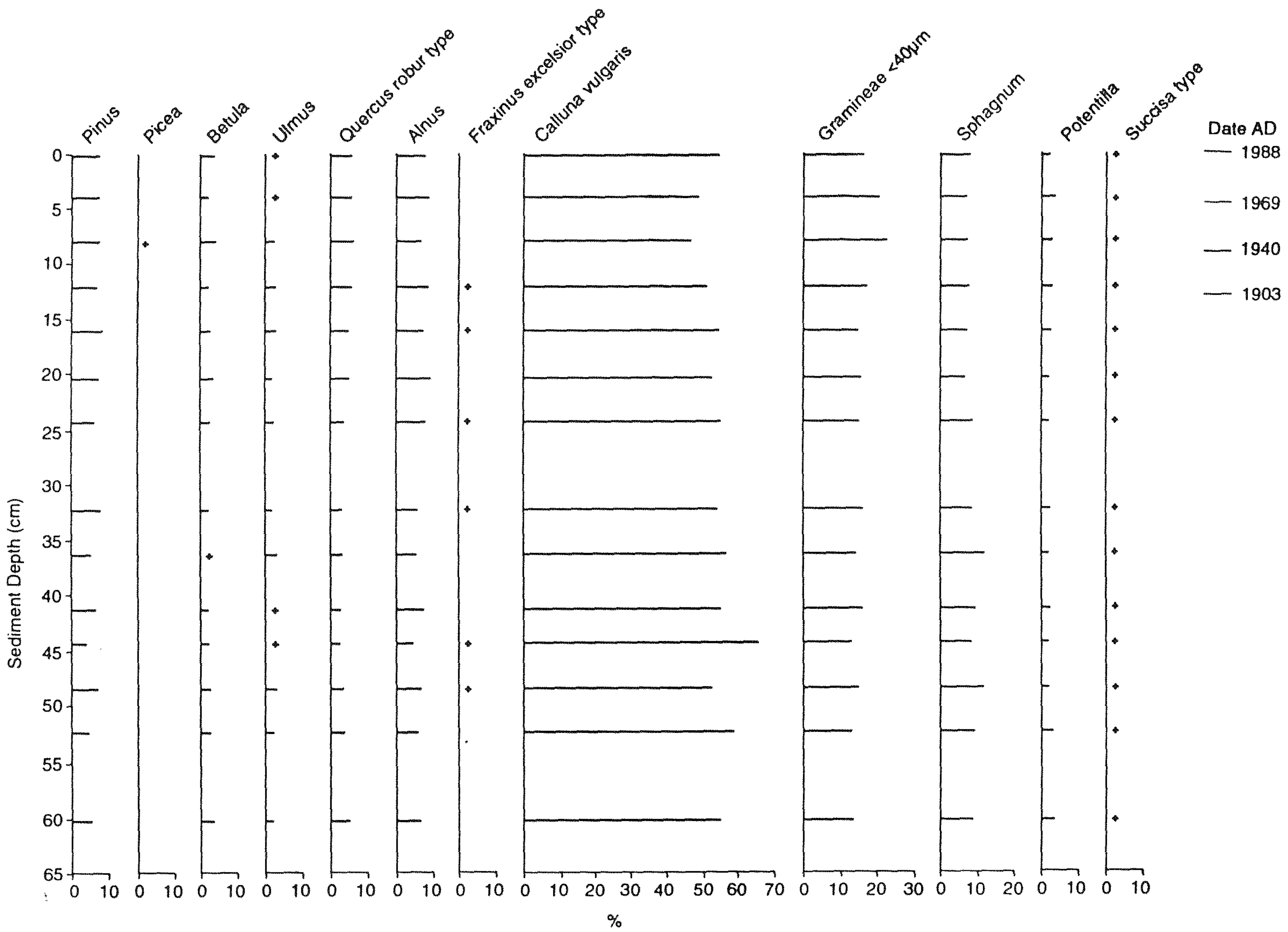


Figure 3.11 Lough Maam: *Calluna* : Gramineae ratio

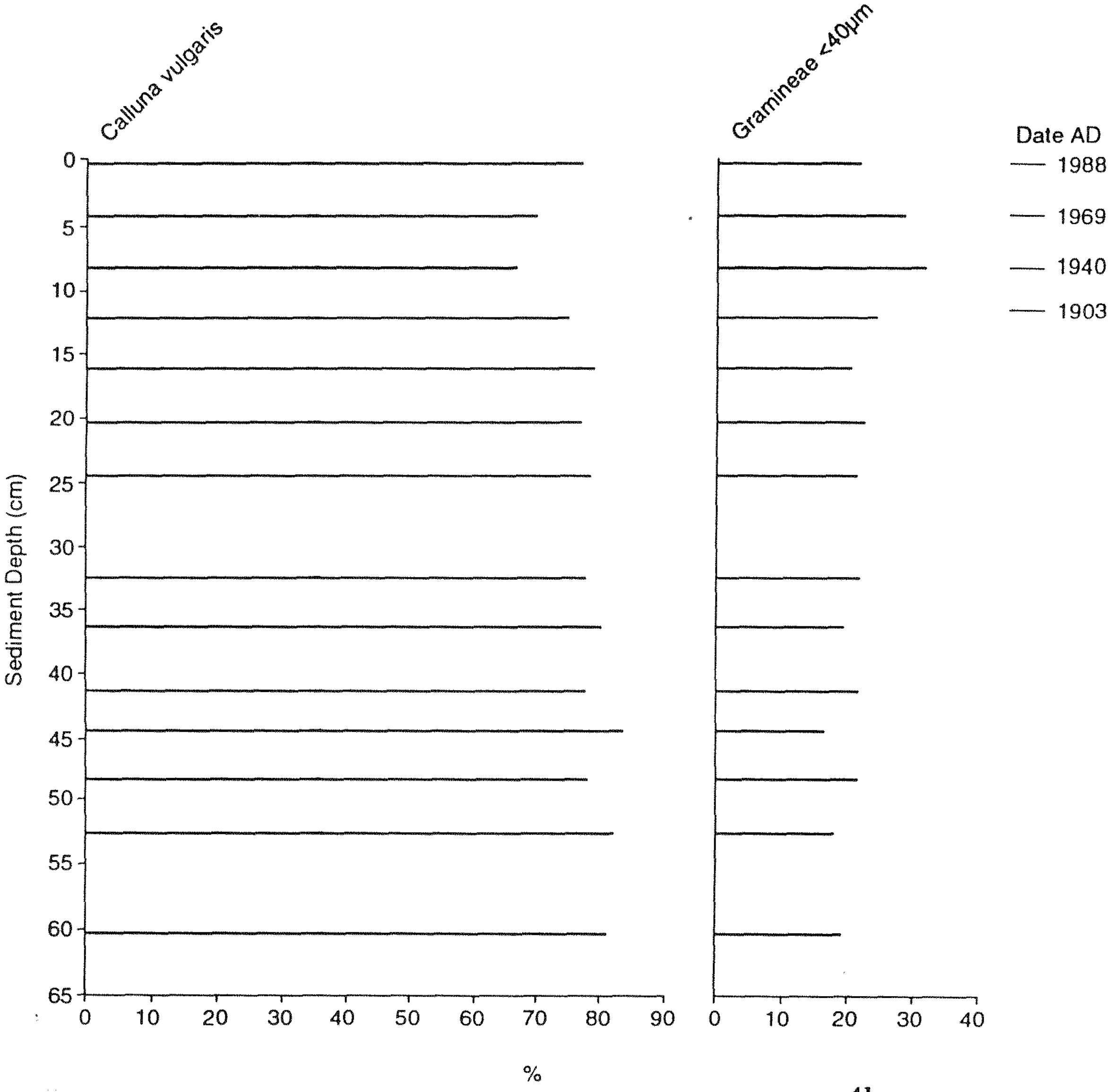


Figure 3.12 Lough Maam: aquatic pollen, summary diagram

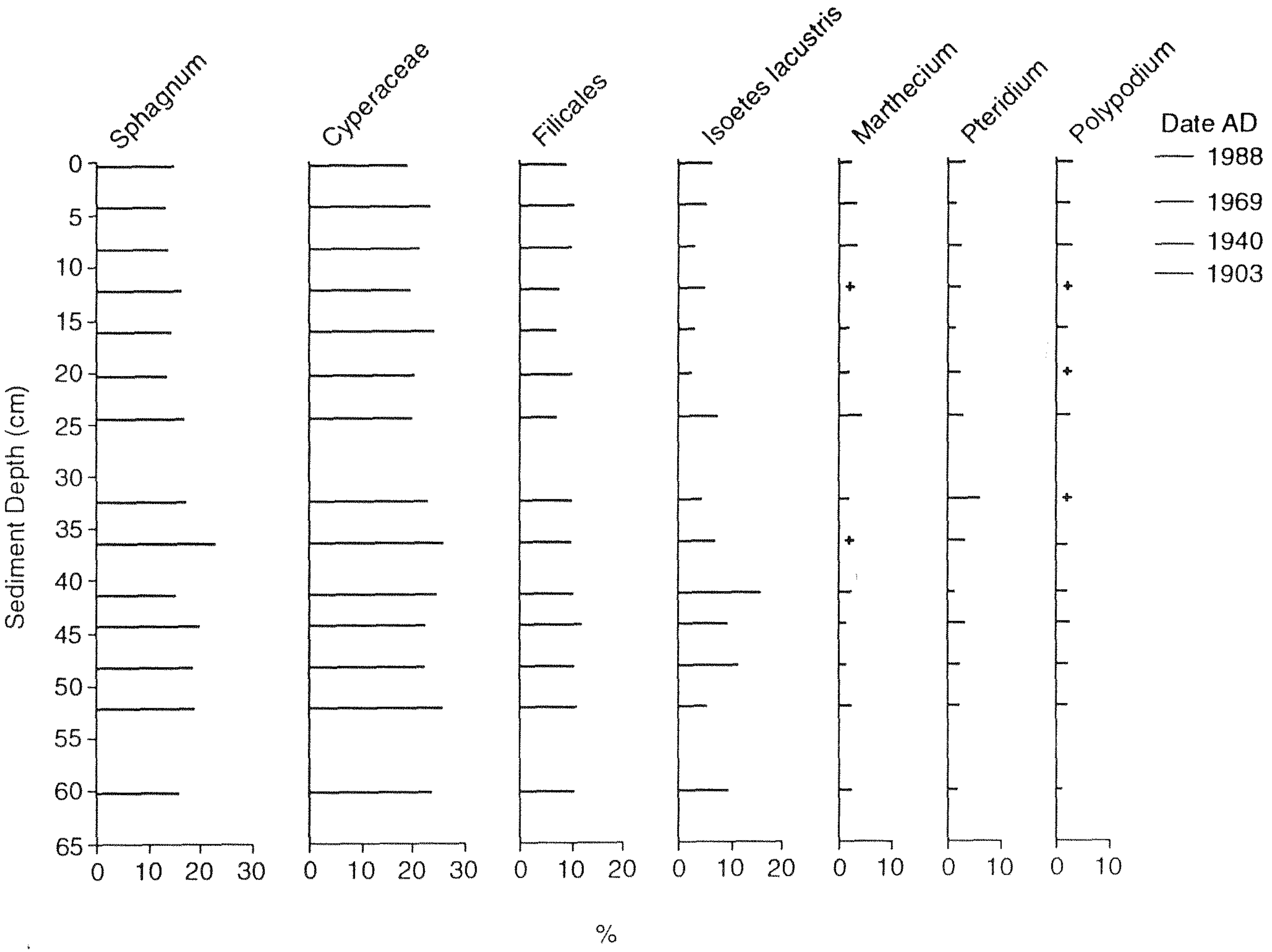


Figure 3.13 Lough Maam: variation of major ion concentrations with depth

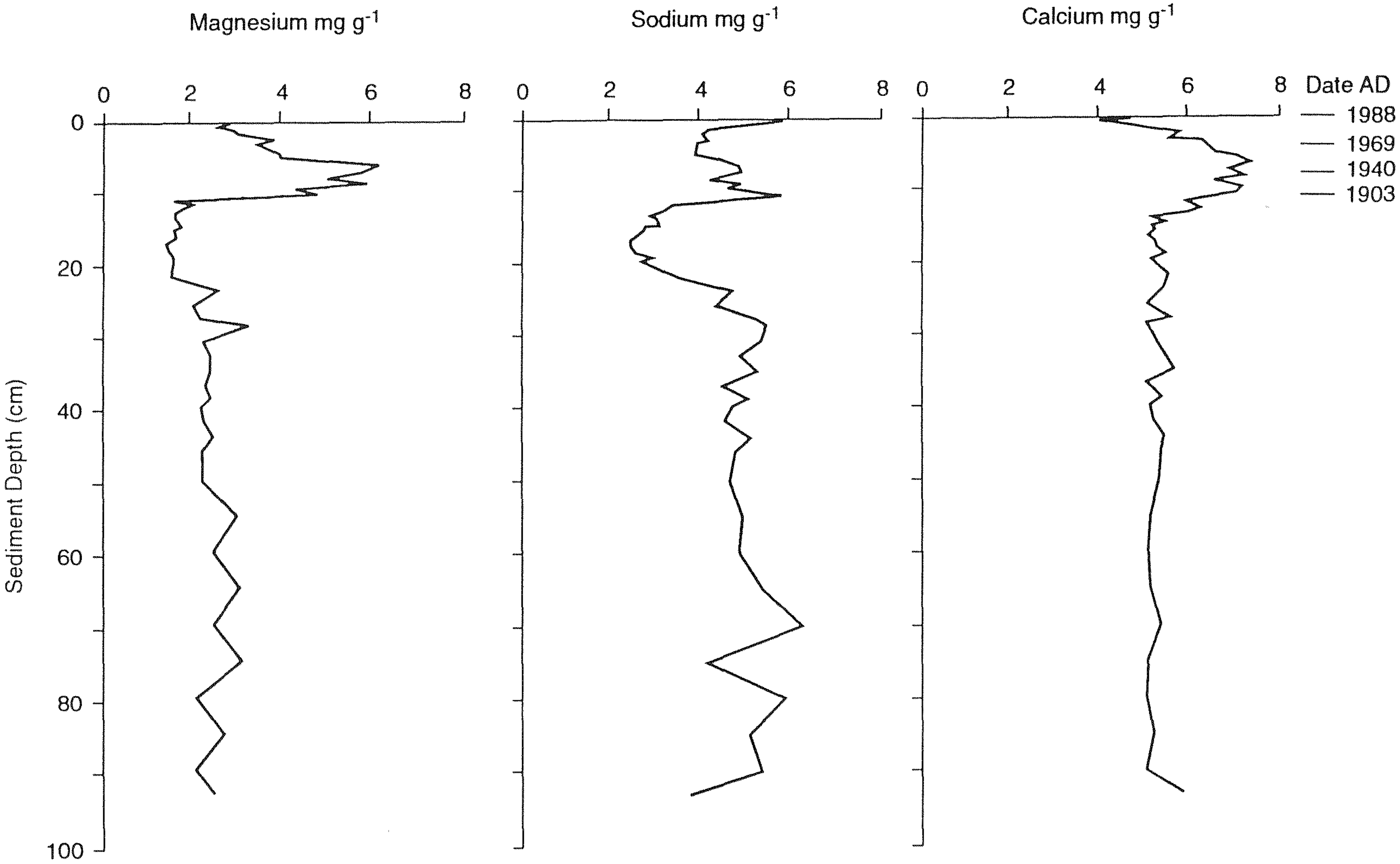


Figure 3.14 Lough Maam: variation of trace metal concentrations with depth

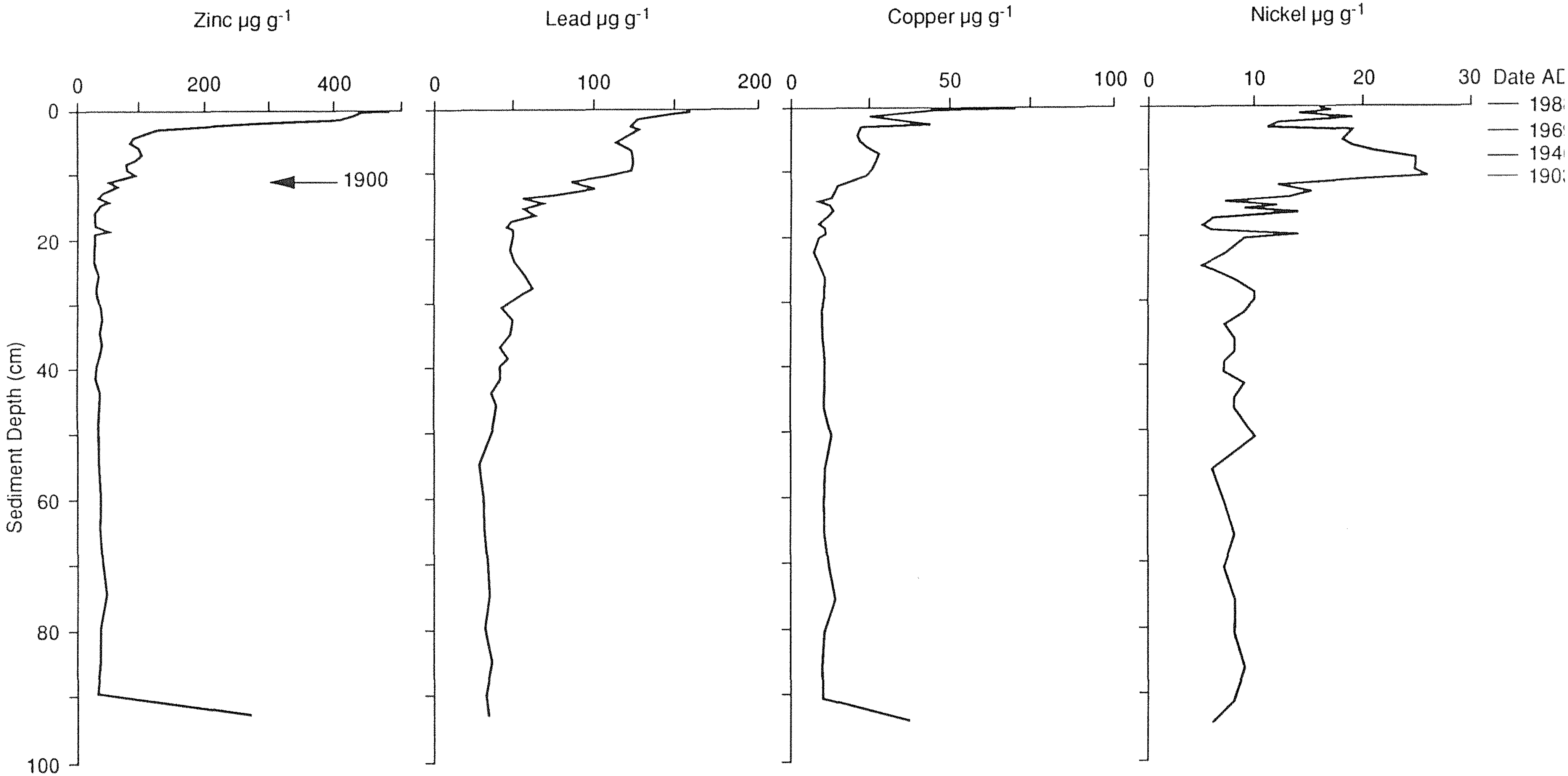
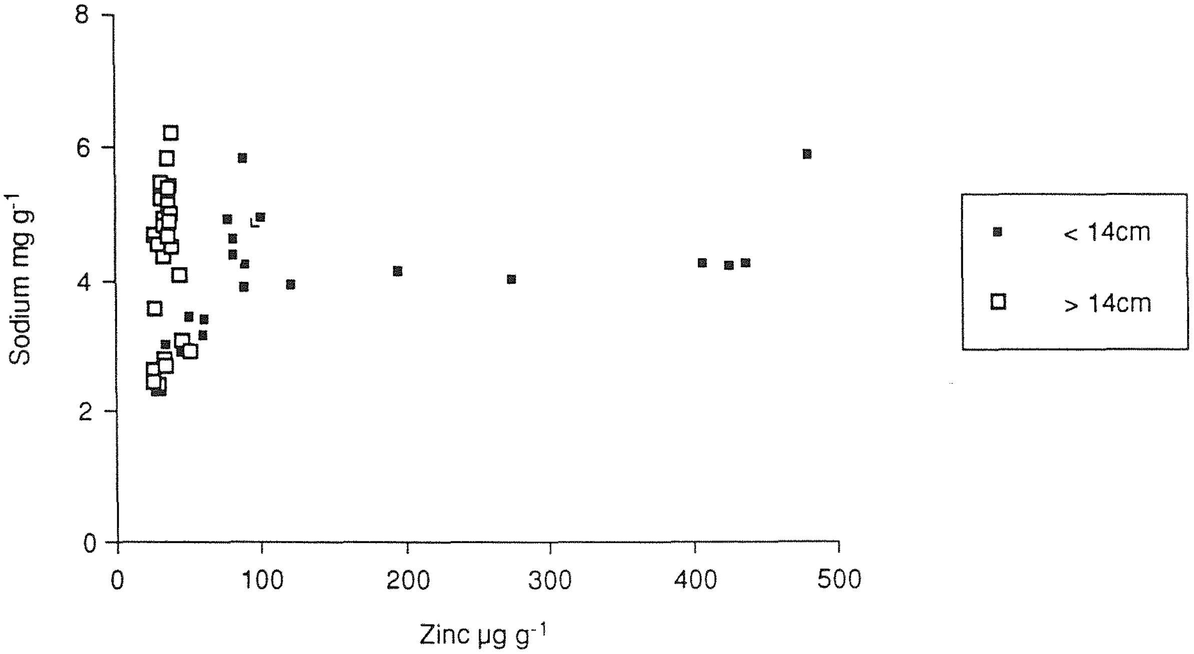
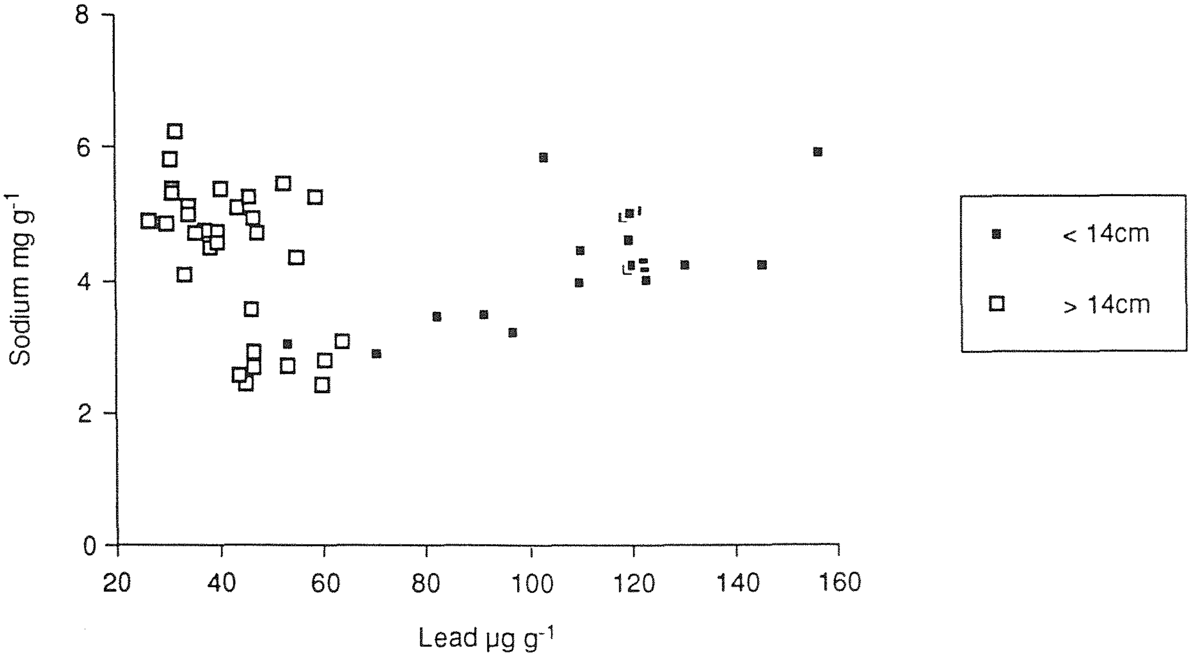


Figure 3.15 Lough Maam: relationships between zinc and sodium and lead and sodium concentrations in the sediment





4 LOUGH MUCK, Donegal.

Site description

Lough Muck is an upland lake located in the Donegal uplands less than 10 km south of Lough Maam. It lies at an altitude of 210 m (Figure 1.1). Lough Muck is considerably deeper (maximum depth c. 19 m) than Lough Maam and has a considerably larger catchment. The lake was not surveyed in detail but an approximate bathymetry is available (Figure 4.1). The catchment is predominantly granitic and consists of considerable areas of fairly flat blanket peat which in some places is undergoing, or has undergone, active erosion. Catchment vegetation is dominated by *Molinia* grassland with *Calluna vulgaris* occurring on drier ground. Afforestation is generally extensive on lower slopes below the catchment and by 1987 had proceeded to the point where the outflow stream leaves the lake.

Water chemistry for Lough Muck (Table 4.1) is typical for a site located in an acid moorland catchment influenced by inputs of wind-borne sea salts. The lake water is acid and possessed zero alkalinity on both sampling occasions. However, there are major differences between concentrations of common ions in the May and September samples. Sodium and chloride concentrations in the former sample are about 50% greater than those in the latter period which points to a significant input of sea salts prior to sampling in May as at Lough Maam. Similarly, most of the difference by which the May calcium concentration exceeded that measured in September is attributable to marine-derived calcium.

Lough Muck seems to be less affected by sea salts than Lough Maam since the absolute sodium and chloride concentrations were considerably lower in the former on each sampling occasion. Lough Muck has a lower pH (in the May sample) and this could reflect a greater contribution to total acidity by organic acids since water colour is greater than in Lough Maam. Direct comparison of dissolved matter concentrations between sites is however made difficult by differences in water residence time. Furthermore, extensive blanket peat deposits in the Lough Muck catchment should allow considerable ion exchange to occur between incoming precipitation and catchment soils. Total acid soluble aluminium is about four-fold higher in Lough Maam and, assuming similar deposited acidity, this must reflect lower availability of aluminium in the Lough Muck catchment. Possibly, in this catchment, less runoff comes into direct contact with minerogenic aluminium rich soils.

Table 4.1 *Water chemistry data for Lough Muck as sampled in May and September 1988.*

		May 1988	September 1988
pH		4.93	5.07
Conductivity	μS cm^{-1}	90.8	*
Alkalinity (Alk_e)	μeq l^{-1}	0.0	0.0
Calcium	μeq l^{-1}	55.4	48.0
Magnesium	μeq l^{-1}	136.7	120.0
Sodium	μeq l^{-1}	604.7	421.0
Potassium	μeq l^{-1}	12.0	10.0
Sulphate	μeq l^{-1}	91.5	87.0
Chloride	μeq l^{-1}	744.5	547.0
Aluminium (total acid soluble)	μg l^{-1}	9.0	11.5
Aluminium (labile)	μg l^{-1}	*	*
Colour (absorbance)	250 nm	0.15	0.22

* = not determined

Methods

Three sediment cores where taken from Lough Muck in May 1988 using a mini-Mackereth corer (Mackereth 1969). Sediment at the deepest point in the lake was late-glacial grey clay, unsuitable for sampling. Soft organic sediment was located in a shallower area of the lake towards the west shore and two cores, both about 90 cm long, were taken from 9 m depth (Figure 4.1). Methods of core analysis follow Stevenson *et al.* (1987) and for carbonaceous particles, Rose (1990).

Lithostratigraphy

The appearance of sediment in all three cores was similar, being composed of fine black detrital sediment (colour 10YR 2/2) in the upper 10-15 cm and slightly lighter colour (10YR 2/1) below. The sediment was very organic but appeared to contain a little more silty matter towards the base. Core 1 had a slightly disturbed mud-water interface and was not analysed further.

For cores 2 and 3 profiles of sediment percentage dry weight, LOI and wet density (Figures 4.2, 4.3) show that most variation occurs in the LOI profile. This must reflect past variations in the rate of peat erosion within the catchment. Despite cores being collected from within a few metres of one another LOI profiles are rather different. In core 2 the organic matter increases gradually from about 70 cm depth to a peak between 35 and 40 cm depth where values reach almost 60%. There is also one isolated spike of organic matter at 12 cm otherwise LOI values show a general decline from 35 cm depth to the core top. Although taken from within 10 m of core 2, core 3 has a somewhat different LOI profile. Organic matter does not increase until 45 cm depth but rapidly reaches a peak of over 60% at 40 cm depth. Above this depth values decline irregularly until 10 cm where organic matter content again sharply increases to values around 60% before declining at 3 cm depth. A single high value occurs in the upper 0.5 cm of sediment.

Core 2 (MUCK2) exhibits less irregularity in LOI values and was selected for further analysis. The changes in LOI values are interpreted as evidence of different phases of enhanced peat erosion in the catchment. The earliest and most important record of increased peat erosion begins at c. 70 cm where sediment LOI starts to increase. Above 35 cm the peak LOI values generally decline, possibly reflecting regeneration of catchment vegetation following a reduction or cessation of catchment burning and/or sheep grazing. Unlike Lough Veagh there is no evidence of any major inwash pulse of mineral soil at this site. However, microscopic examination of the sediment does show that levels with high LOI values contain larger and very angular quartz fragments and that these fragments are more frequent towards the core top.

Dating

Sediment samples from core MUCK2 were analysed for ^{210}Pb, ^{226}Ra, ^{137}Cs, ^{134}Cs and ^{241}Am by gamma spectrometry (Appleby *et al.* 1986). The ^{210}Pb and ^{226}Ra results are given in Table 4.2. The ^{137}Cs, ^{134}CS and ^{241}Am results are given in Table 4.3. The ^{134}Cs derives from Chernobyl fallout and has been corrected for decay since May 1986.

The chronology calculated using the CRS and CIC ^{210}Pb dating models (Appleby and Oldfield 1978) is shown in Figure 4.4 and suggest a modest acceleration in sediment accumulation rate over the past 40 years. There is a small relative decline in unsupported ^{210}Pb between 11.75 and 8.75 cm depth which indicates a small dilution, probably by soil inwash, at this level. The ^{210}Pb inventory of core MUCK2 is 38.9 pCi cm^{-2}, representing a mean atmospheric flux of 1.21 pCi cm^{-2} yr^{-1}. This is about twice the expected atmospheric flux and suggests significant sediment focusing at the core site. In view of this, it is not possible to choose between the two dating methods on the ^{210}Pb data alone.

The high surficial ^{134}Cs activity (Table 4.3) shows that the ^{137}Cs record near the core top is dominated by Chernobyl fallout. Using the ^{134}Cs activity to partition the total ^{137}Cs activity into a Chernobyl component and a weapons component, the latter has a well defined peak at a depth of 5.25 cm which has been only partly obscured by the Chernobyl ^{137}Cs. Since the ^{241}Am activity (Table 4.3) has a clearly defined peak at a depth of 4.25 cm, these two levels can be dated with confidence to the mid-1960s. Figure 4.4 shows that this date is in good agreement with the CRS model ^{210}Pb dates. Since both dating models suggest similar accumulation rates beneath this level, the CRS model would appear to provide a reliable basis for dating the core. Dates calculated using this model are given in Table 4.4

Table 4.2 Lough Muck: ^{210}Pb and ^{226}Ra data

Depth	Dry mass	^{210}Pb Conc.				^{226}Ra Conc.	
		Total		Unsupp.			
cm	g cm^{-2}	pCi g^{-1}	±	pCi g^{-1}	±	pCi g^{-1}	±
0.50	0.0354	54.71	2.82	54.71	2.86	0.00	0.46
2.75	0.2032	48.20	1.39	47.27	1.48	0.93	0.51
4.25	0.3177	50.62	1.86	50.17	1.91	0.45	0.42
5.25	0.4002	46.17	1.79	44.07	1.94	2.10	0.75
6.25	0.4865	35.74	1.52	34.66	1.56	1.08	0.36
8.75	0.7134	17.42	1.13	15.47	1.19	1.95	0.36
11.75	1.0116	15.46	0.85	13.12	0.89	2.34	0.27
14.75	1.3211	9.92	0.49	7.22	0.52	2.70	0.18
20.50	1.9516	3.07	0.29	1.06	0.33	2.01	0.15
26.50	2.6163	3.64	0.27	0.34	0.31	3.30	0.15
30.50	3.0603	2.98	0.27	0.04	0.31	2.94	0.15

Table 4.3 Lough Muck: ^{137}Cs, ^{134}Cs and ^{241}Am data

Depth	^{137}Cs Conc.		^{134}Cs Conc.		^{241}Am Conc.	
cm	pCi g^{-1}	±	pCi g^{-1}	±	pCi g^{-1}	±
0.50	103.17	1.20	59.20	2.31	0.16	0.11
2.75	63.58	0.67	31.50	2.03	0.09	0.05
4.25	30.01	0.71	3.97	0.83	0.67	0.13
5.25	42.10	0.71	1.93	0.59	0.35	0.10
6.25	31.96	0.51	0.00	0.00	0.28	0.07
8.75	8.56	0.27	0.00	0.00	0.00	0.00
11.75	4.32	0.17	0.00	0.00	0.00	0.00
14.75	2.42	0.09	0.00	0.00	0.00	0.00
20.50	0.74	0.06	0.00	0.00	0.00	0.00
26.50	0.76	0.05	0.00	0.00	0.00	0.00
30.50	0.18	0.04	0.00	0.00	0.00	0.00

Table 4.4 Lough Muck: ^{210}Pb chronology

Depth	Cum. dry mass	Chronology			Sedimentation rate		
		Date	Age				
cm	g cm^{-2}	AD	yr	±	g cm^{-2}yr^{-1}	cm yr^{-1}	± (%)
0.00	0.0000	1988	0				
0.50	0.0354	1986	2	2	0.0210	0.284	5.3
1.00	0.0727	1984	4	2	0.0205	0.276	5.0
1.50	0.1100	1983	5	2	0.0200	0.268	4.8
2.00	0.1473	1981	7	2	0.0195	0.260	4.5
2.50	0.1846	1979	9	2	0.0189	0.252	4.3
3.00	0.2223	1977	11	2	0.0179	0.237	4.3
3.50	0.2605	1974	14	2	0.0164	0.214	4.5
4.00	0.2986	1972	16	2	0.0149	0.191	4.8
4.50	0.3383	1969	19	2	0.0139	0.174	5.1
5.00	0.3796	1966	22	2	0.0136	0.164	5.5
5.50	0.4218	1963	25	2	0.0135	0.158	5.6
6.00	0.4649	1960	28	2	0.0138	0.156	5.6
6.50	0.5092	1957	31	2	0.0146	0.161	5.8
7.00	0.5546	1954	34	2	0.0158	0.173	6.3
7.50	0.5999	1952	36	2	0.0171	0.184	6.8
8.00	0.6453	1949	39	2	0.0184	0.196	7.3
8.50	0.6907	1946	42	2	0.0197	0.207	7.8
9.00	0.7383	1943	45	2	0.0198	0.206	8.1
9.50	0.7880	1940	48	2	0.0187	0.194	8.2
10.00	0.8377	1937	51	2	0.0176	0.181	8.3
10.50	0.8874	1934	54	2	0.0165	0.168	8.4
11.00	0.9371	1931	57	3	0.0154	0.155	8.5
11.50	0.9868	1929	59	3	0.0144	0.143	8.6
12.00	1.0374	1925	63	3	0.0137	0.134	8.9
12.50	1.0890	1921	67	3	0.0133	0.130	9.4
13.00	1.1406	1917	71	3	0.0131	0.125	9.9
13.50	1.1921	1913	75	4			
14.00	1.2437	1909	79	4			
14.50	1.2953	1905	83	4			
15.00	1.3485	1901	87	5			
17.00	1.5678	1885	103	8	0.0131	0.120	18.7
19.00	1.7871	1868	120	11			
21.00	2.0070	1851	137	14			
23.00	2.2286	1835	153	16			

Diatom analysis and pH reconstruction

A summary diatom diagram for core MUCK2 is shown in Figure 4.5. Below about 40 cm depth in the core diatoms are scarce and poorly preserved. However, above this level preservation is generally reasonable and from this point to about 14 cm depth (1905) the sedimentary diatom assemblage is dominated by *Aulacoseira distans* v. *tenella* and *Cyclotella kuetzingiana. Fragilaria virescens* v. *exigua* and *Brachysira vitrea* are also common. Several taxa show consistent frequency changes in this section with *C. kuetzingiana* declining from c. 23 cm depth (1835), *B. vitrea* from c. 21 cm depth (1851) and *Cymbella lunata* from 18 cm depth (1876). Other taxa show corresponding increases, *A. distans* v. *tenella* at 23 cm, *F. virescens* v. *exigua* and *Frustulia rhomboides* v. *viridula* at 21 cm depth. Above this depth these taxa begin to decline in abundance with *B. vitrea* declining first at around 20 cm depth (1860), then *A. distans* v. *alpigena* at 13 cm depth (1917) followed by both *C. kuetzingiana* and *F. virescens* v. *exigua* at 5 cm depth (1966).

Above 14 cm depth marked changes occur in the diatom taxa frequencies. At 12 cm depth (1925) *A. distans* v. *tenella* is largely replaced by *C. kuetzingiana*. At around 8 cm depth (1946) *Asterionella ralfsii, Tabellaria quadriseptata* and *Cymbella perpusilla* frequencies increase sharply to the core top. At this depth other species such as the *Frustulia* taxa and *F. virescens* v. *exigua* start to decline in abundance. *C. kuetzingiana* shows an unusual distribution: despite the declining frequency abundance trend from 23 cm depth, frequencies more than double at the 8 and 12 cm depth levels before declining again to the core top. These two levels bracket a sharp inwash peak at 11 cm depth (see Figure 4.2) and the 10.5-11.0 cm depth level contained too few and poorly preserved diatoms for reliable counting.

Where preservation was reasonable diatom concentrations were fairly uniform at about 5 x 10^{7} cells g^{-1} dry sediment (Figure 4.6). This value is rather low compared with other similar sites in the region (eg. Lough Maam) and an order of magnitude lower than diatom concentrations in Cairngorm lake sediments (cf. Jones *et al.* 1992). The low concentrations found in Lough Muck probably reflect a combination of low diatom productivity and a relatively high rate of catchment erosion causing dilution of the diatoms in the sediments. Furthermore, poor diatom preservation at certain levels in the core must account for the scarcity of diatom valves around 11 cm and 40 cm depth.

The diatom abundance minimum at 11 cm depth coincides with a sharp peak in sediment LOI (see Figure 4.2) and with very poor diatom preservation. Below 40 cm in the core diatom preservation again deteriorates, but is reasonable between 35 and 40 cm depth, the section where LOI values are highest. In other acid lakes where past inwash events have occurred diatoms are often scarce because of dilution effects but preservation is largely unaffected (eg. Battarbee and Flower 1985). In the Lough Muck core, dilution effects are difficult to detect around the earlier inwash peak probably because of the combination of sediment inwash and diatom dissolution effects. Even so, it is clear that inwash *per se* is not solely responsible for poor diatom preservation in this core and preliminary examination of core MUCK3 revealed a similar record with diatoms virtually absent from below the inwash peak at 45 cm depth. Other cores from acid lakes with poor diatom preservation are thought to be influenced by ground water movements (eg. Flower *et al.* 1988), but this is unlikely in a relatively large rock basin site such as Lough Muck. A sub-surface spring could be a possible local cause of water movements within sediments but analysis of a suite of cores taken from across the Lough Muck basin is required to test this suggestion.

Using the chronology provided by ^{210}Pb analysis the pH history of Lough Muck can be inferred from the early-nineteenth century to the present (1987). Past pH values were reconstructed from the diatom assemblages using both the MR (Flower 1986) and WA (Birks *et al.* 1990) methods (Figure 4.7). The inferred pH profiles produced by both methods show similar trends with most of the

WA pH values being less than values calculated using MR, markedly so around 10 cm depth. The current measured pH of Lough Muck is 4.9-5.0 which is close to the WA and MR inferred pH of 4.7 for the most recent sediment, the difference is within the root mean squared error of the WA method (Birks *et al.* 1990). Both profiles show clearly that Lough Muck is recently acidified with pH declining from around 5.5 in the nineteenth century to 4.7 by the mid-1980s.

The onset date of acidification is difficult to assign precisely and differs according to the method employed. WA indicates a clear and consistent declining pH trend above 18 cm depth (1875) but the initial small decline at 20 cm (1860) could indicate the first effects of acidification. MR indicates pH decline above 20 cm depth (1860). Although changes in individual diatom taxa (see above) indicate some slight acidification effects earlier in the nineteenth century (as *C. kuetzingiana* and *B. vitrea* decline) the changes only become manifest as a clear pH shift in the late-nineteenth century. There is no evidence of any recovery or increase in inferred pH from the most recent sediment.

A clear anomaly in the historical pH curves calculated according to the WA and MR methods occurs between 8 and 12 cm depth (1949-1925) where MR pH values are between 0.4 and 0.7 units greater than those calculated by WA. These two levels have high frequencies of *C. kuetzingiana* which must increase the calculated pH values since this species is characteristic of less acid conditions. This has a particularly marked effect on MR values where *C. kuetzingiana* is classified as being indifferent to pH yet most of the associated assemblage is acidophilous. Though less marked, pH values calculated using the WA method must also be higher for this section of the core since *C. kuetzingiana* has a higher pH optimum (6.29) than *Aulacoseira distans* v. *tenella* (5.78), the taxon which seems to be most affected by poor preservation. Therefore it is probable that the high frequency of *C. kuetzingiana* in these levels is an artifact caused by dissolution of less well silicified taxa, or from reworking of older sediments, with the result that pH values for the period 1925-1949 are probably unreliable. Although a source of *C. kuetzingiana* valves from reworked very old sediment can not be excluded, reworking of sediment from elsewhere in the period spanned by the core is untenable since assembalge frequencies are incompatible and differential reworking is highly unlikely.

The high *C. kuetzingiana* frequency in the levels which bracket 11 cm depth can be shown to be misleading by converting the data to concentration values (Table 4.5). This removes the peak of *C. kuetzingiana* at 12 cm depth since its concentration at this level is not significantly different from concentrations at 8 or 16 cm depth. The diatom concentration data indicate an irregularly declining trend from 26 cm depth (1830) and this is most marked between 8 and 2.5 cm depth (1949-1979). At 16 cm depth the sedimentary concentration of *C. kuetzingiana* is particularly low, probably for reasons other than water acidity or poor preservation. Diatom accumulation rate data offer the best guide to estimating past supply rates of diatoms to the sediments but problems of valve dissolution and peat inwash around 11 cm depth as well as the lack of ^{210}Pb measurements at this particular level make such calculations unreliable for this core.

Table 4.5 Diatom percentage frequencies and concentration data for Lough Muck core 2.

Depth (cm)	Total diatom conc. (cells 10^{-6} g^{-1})	% *Cyclotella Kuetzingiana*	*Cyclotella* conc. (cells x 10^{-5} g^{-1})	95% confidence limits
0.5 - 1.0	6.25	4	2.5	+0.85 - -0.70
2.5 - 3.0	6.55	3	1.9	+1.00 - -0.80
4.5 - 5.0	5.46	7	3.8	+0.80 - -0.70
8.0 - 8.5	3.57	14	4.9	+0.50 - -0.40
12.0 - 12.5	2.08	22	4.5	+0.30 - -0.20
16.0 - 16.5	4.57	4	1.8	+0.50 - -0.40
20.0 - 21.0	5.37	9	4.8	+0.60 - -0.50
26.5 - 26.0	4.06	14	5.6	+0.60 - -0.40
36.0 - 37.0	3.95	15	5.9	+0.50 - -0.45

Carbonaceous particle analysis

The carbonaceous particle profile (Figure 4.8) for Lough Muck shows very low levels of contamination below 21 cm depth or before the mid-nineteenth century. Above this level particle concentration increases to the late 1940s. The trend of particle increase is then checked as concentrations fluctuate between 4000-6000 g^{-1} dry sediment until the early 1980s when a major concentration peak of c. 9000 particles g^{-1} dry sediment occurs at 1.5-2.0 cm. Particle concentration then reverts to pre-peak levels in the top 1.5 cm of the core.

The maximum concentration of carbonaceous particles in Lough Muck sediment is about the same as that recorded in nearby Lough Maam although somewhat less that that recorded in Lough Veagh. Differences in surface sediment particle concentration probably arise from variations in sediment accumulation rate and particle focusing effects.

Pollen analysis

The summary pollen diagram (Figure 4.9) indicates a fairly uniform pollen assemblage composition throughout the sediment core. The most marked change occurs in the frequency of *Isoetes* spores which shows a marked decline between 68 and 45 cm depth. The *Isoetes* decline corresponds closely with increasing organic content of the sediment and it is very likely that this plant has been suppressed by increased water turbidity from inwashed eroded peat.

It is notable that although the landscape around Lough Muck is essentially treeless, tree pollen constitutes c. 25% of the down-core pollen sum. *Pinus* pollen usually exceeds 5% (Figure 4.9) of this sum and since natural *Pinus* woodland is long extinct in this part of Ireland its only likely source, before recent planting, during the last 200 years is from reworked and inwashed peat. The frequency of *Pinus* pollen does not however increase with increasing organic content of the sediment and this indicates that only the quantity, rather than the source, of peat eroded into the lakes has changed markedly through time. Similarly, neither *Calluna* nor Gramineae pollen frequencies change substantially over the inwash peak around 35 cm depth. Nevertheless, by about 25 cm depth or the early-nineteenth century the *Calluna* : Gramineae ratio (Figure 4.10) begins to change and by the surface sediment (1988) Gramineae pollen is more frequent than that of *Calluna*. Interpretation of this ratio is made difficult by the influx of old pollen (see above), but a declining ratio towards the core top is compatible with the hypothesis that the vegetation has been modified by management practices such as heather burning.

Geochemistry

Although the dry weight and LOI decrease above 10 cm and 35 cm respectively (Figure 4.2), the major cation concentrations are fairly constant, particularly above 30 cm (Figure 4.11). This makes interpretation of the trace metal profiles straightforward). The zinc and lead concentrations increase above 20 cm depth (1860) (Figure 4.12). There is a marked surface effect, where all the trace metal concentrations are particularly high in the top 0.5 cm. The elevated trace metal concentrations in the more recent sediment is clearly shown by plotting zinc and lead against a tracer of minerogenic erosion such as sodium (Figure 4.13). An important secondary feature of the zinc profile is the drop in concentration above 8 cm (1950). As there are no major changes in the chemical constitution, as indicated by the constant major cation concentrations, this drop in zinc concentration is likely to result from a reduction in zinc sedimentary efficiency as the pH drops below 5.5-5.0.

The trace metal results show that Lough Muck has been contaminated by material deposited from the atmosphere since the middle of the nineteenth century and that the associated acid stress on the lake has been sufficient to cause lake water pH to fall below 5.0.

Discussion.

The diatom record and inferred pH history of Lough Muck show that the lake has been recently acidified and that the acidification process began in the middle to late-nineteenth century. Despite problems of calculating pH values in the 1910-1950 period due to differential diatom preservation, the overall pH decline since the mid-nineteenth century can be established as 0.8 pH units. The degree of acidification is greater than in nearby Lough Maam but less than in heavily polluted areas such as the Galloway hills of south-west Scotland. Lough Muck began to acidify rather earlier than Lough Maam but at a similar time to sites in south-west Scotland. These observations suggest that acid deposition initially affected both north-west Ireland and south-west Scotland at about the same time in the last century but the effect was less in Ireland, probably because of lower deposition rates of acid pollutants. Differences in catchment ANC probably account for differences in the degree and onset dates of acidification in the same geographical area.

The mean calcium concentration in Lough Muck today is around 50 $\mu eq\ l^{-1}$ which is higher than in Lough Maam (Table 3.1). The chemistry data in Table 4.1 refer to only two samples, one of which is strongly affected by a seasalt intrusion. Even if the lowest available calcium concentration value for Lough Muck (48 $\mu eq\ l^{-1}$) is utilised the value indicates that the lake is less, rather than more, sensitive to acidification than is the less-acidified Lough Maam. Furthermore, the diatom acidification model of calcium concentration to deposited sulphur (Battarbee 1989) predicts that Lough Muck should not be acidified at all.

Two factors probably account for the conflicting conclusions regarding the acidification status of Lough Muck. Firstly, it is likely that even the lowest calcium measurement is relatively high and more measurements are required, and secondly, the interpolated non-marine sulphate deposition of around 0.5 $g\ S\ m^{-2}\ yr^{-1}$ for the region is too low because enhancing factors such as altitude are ignored.

The carbonaceous particle and trace metal record indicates that atmospheric deposition has affected Lough Muck to a similar extent to Lough Maam. Concentrations are somewhat lower but the overall sediment accumulation rate is higher in Lough Muck. The results substantiate observations at Lough Maam, that levels of contamination in Donegal are lower than in strongly acidified Scottish lakes, but are considerably more than in non-acidified Loch Teanga in the Outer Hebrides where deposition of atmospheric acidity is lower.

The acidification value for Lough Muck as determined by the equation of Henriksen (1982) is small and is derived from water chemistry data where one sample is strongly affected by a seasalt intrusion. If the water chemistry sample with low sea salts is used in the calculation then the acidification index is 24, indicating moderate acidification. Hence, despite problems of applying the Henriksen Index to lakes influenced by sea salts the equation seems to give correct results when this influence is minimal. However, DOC contributes about 18 μeq l^{-1} of acidity and if this is used to correct the acidification index for the low seasalt sample the value is reduced to about 6.

According to the diatom acidification model (Battarbee 1989), Lough Muck should be unacidified. Only if the seasalt corrected calcium concentration is substituted for total calcium concentration in the model does Lough Muck move from unaffected to transitional in response to a sulphur deposition of 0.5 g m^{-2} yr^{-1}. Acidification of this site is therefore not clearly predicted by either this model or the Henriksen equation. This could be caused by underestimation of lake water calcium concentration or annual sulphur deposition. The recent acidification is incompatible with the timing of inferred peat inwash although peat drainage acidity is a long term factor since Lough Muck has always been at least 0.5 pH units more acid than nearby Lough Maam, where blanket peat is scarce.

Calcium concentration could well be underestimated since only two measurements are available, although calcium *per se* would cause the Henriksen Equation to predict a lesser degree of acidification. Information concerning deposited sulphur for the Lough Muck catchment was obtained by interpolation of data from monitoring stations in the north-west of Ireland (Figure 1.2). These stations are located at low altitude and hence without corrections for orographic rainfall and occult precipitation interpolated annual sulphur deposition probably underestimates values for high altitude sites (eg. Fowler *et al.* 1989).

However, despite uncertainties of sulphur deposition and lake calcium concentration, the magnitude of the diatom changes in the Lough Muck core provide unequivocal evidence that the site has significantly acidified since the middle to late-nineteenth century.

As at other upland acidified lakes, the chronology of land-use change in the Lough Muck catchment, as inferred from the lithostratigraphic and pollen record, does not coincide with the onset of acidification. The sediment core shows several peaks in lithostratigraphic characteristics that seem to be related to catchment erosion events. There is however little evidence of vegetation change around the lake over the time period represented by the core. It is probable that changes in organic content of the sediment result from accelerated peat erosion due to sheep grazing and/or heather burning.

Declining organic matter over the dated upper section (26-0 cm) of the core coincides with a period of accelerating sediment accumulation. This is contrary to the expectation that accumulation rate should decline after the organic inwash peak at 35-40 cm depth and could indicate that erosion source(s) have changed through time with disproportionately more minerogenic material entering the lake over the last c. 150 years. This observation raises the possibility that variation in minerogenic erosion has determined the sediment LOI characteristics and that the LOI peaks result from an absence of inwashed minerogenic material rather than an increase in eroded peat. Some evidence against this is provided by the diatom record where abundance is very low at 10.5-11.0 cm depth, immediately following a LOI spike. However, the acceleration in sediment accumulation at this point is small and can not account for the observed scarcity of diatoms. Diatom evidence of valve dilution by soil inwash is equivocal elsewhere in the core and between 35 and 40 cm depth, the highest LOI section, diatom concentration is not markedly diminished although valves are not well preserved. Furthermore, diatoms are entirely absent below about 45 cm depth and continue to be so even below 70 cm depth where LOI values are low and stable at around 30%. The two other sediment cores recovered from Lough Muck were checked and in both diatoms were not present in the deeper levels of sediment. It is clear that at this site diatom absence is not primarily caused by dilution effects from inwashed soil but to poor preservation. Diatom preservation is usually excellent in acid lake sediments and in this case it can only be postulated that poor preservation is caused by slow water movement through the deposited sediment. A sub-surface spring near the coring site is one explanation but clearly further analysis including multi-coring of the lake basin is required to substantiate this possibility.

Stronger evidence that the LOI peak at 35-40 cm depth results from increased peat erosion rather than a reduced minerogenic influx is provided by the *Isoetes* profile. This profile is entirely consistent with the accelerated peat inwash hypothesis as abundance of this plant declines from c. 70 cm depth to very low levels by 45 cm depth. The peat inwash peak is probably not earlier than the medieval period and the rate of sediment accumulation could have been rapid over the middle section of the core. Although peat inwash peaked in this period the increase in sediment accumulation rate in the upper dated section of the core must indicate a shift to minerogenic erosion sources within the catchment. Overall it is likely that the recent history of Lough Muck was characterised initially by a strong increase in catchment peat erosion followed by gradual erosion of minerogenic sub-soils, possibly exposed by the preceding process.

Lough Muck: Conclusions

1) Diatom analysis of the Lough Muck core has demonstrated that acid deposition exceeded critical loading for the site some 100 years ago and the lake is now moderately to strongly acidified.

2) Application of the Henriksen Equation to Lough Muck water chemistry indicates moderate acidification of the lake.

3) The diatom acidification model (Battarbee 1989) predicts the lake to be unacidified unless lake water calcium is corrected for seasalt effects.

4) The timing of land-use changes around Lough Muck can not account for acidification.

5) Carbonaceous particle and trace metal records indicated that Lough Muck is moderately impacted by atmospheric pollution.

Figure 4.1 Lough Muck: bathymetry

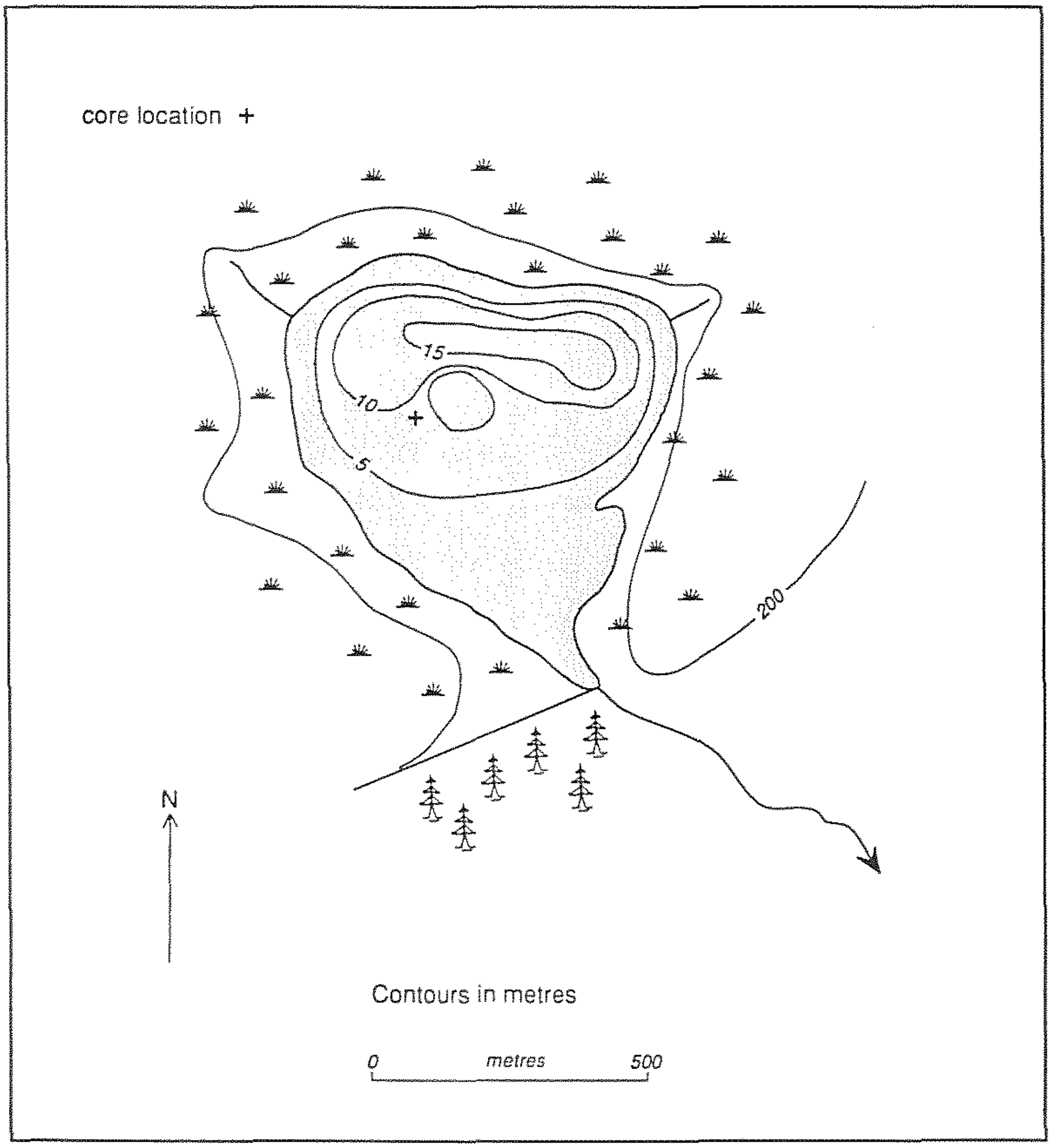


Figure 4.2 Lough Muck: lithostratigraphy core MUCK2

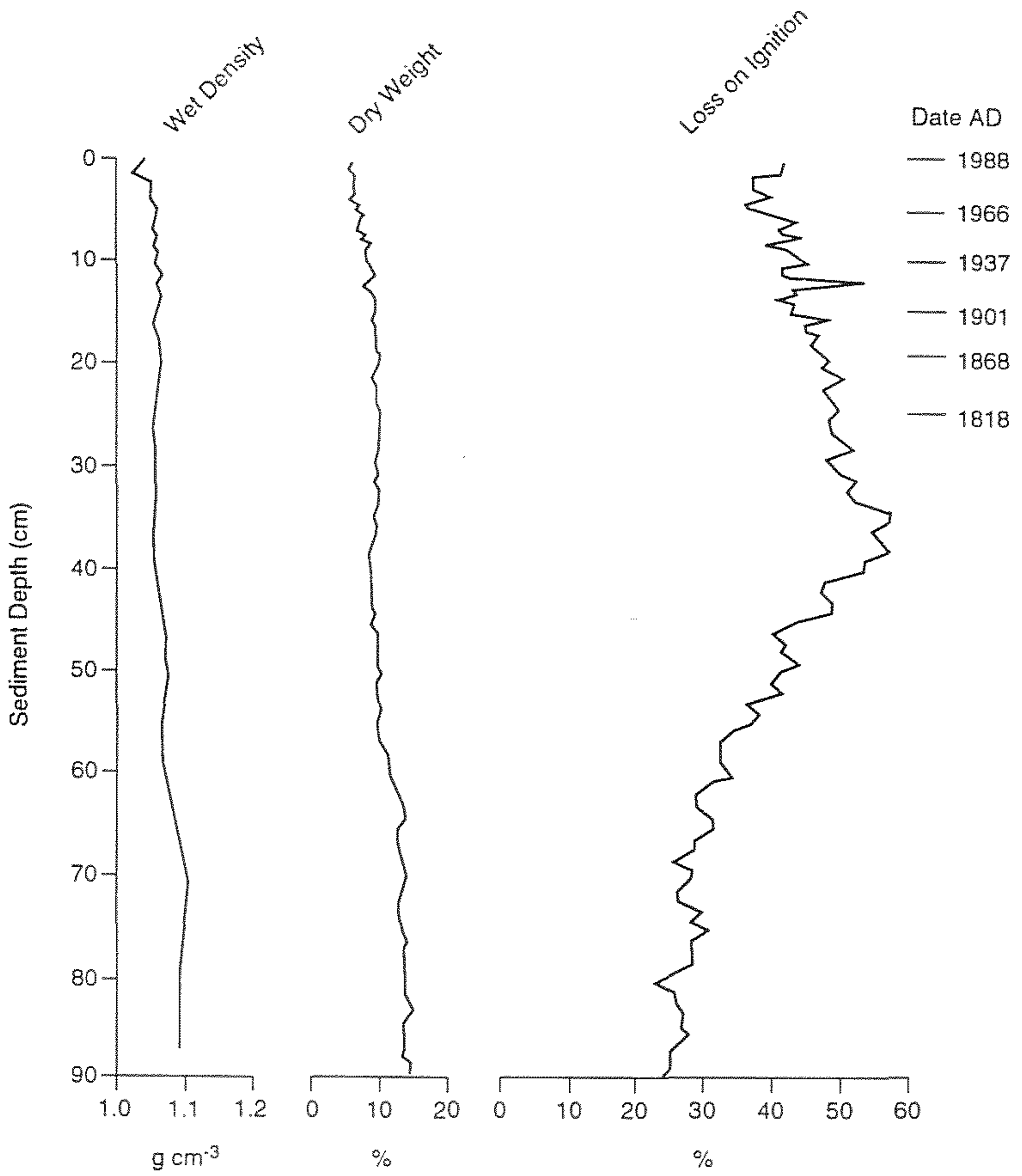


Figure 4.3 Lough Muck: lithostratigraphy core MUCK3

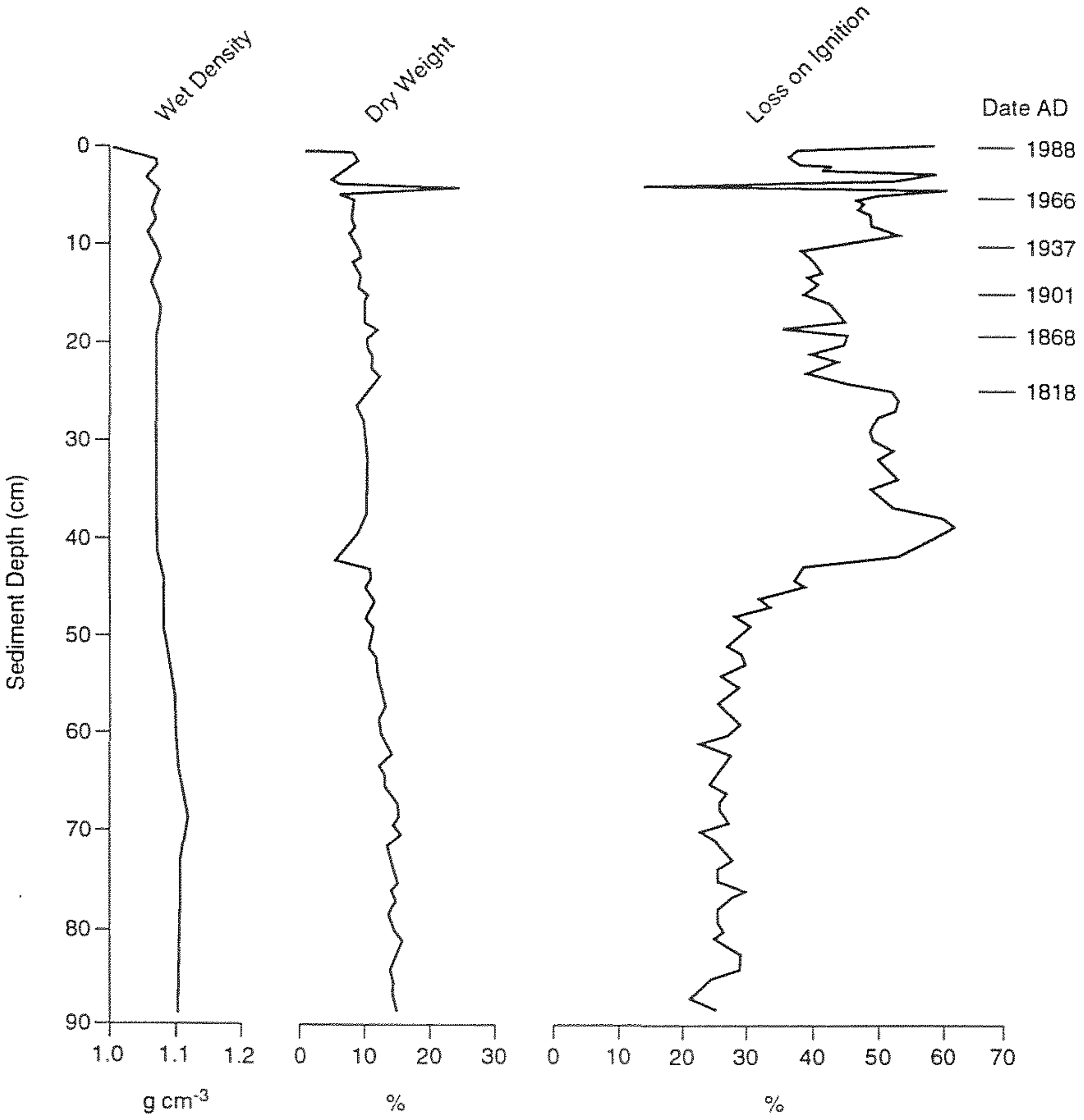


Figure 4.4 Lough Muck: ^{210}Pb chronology

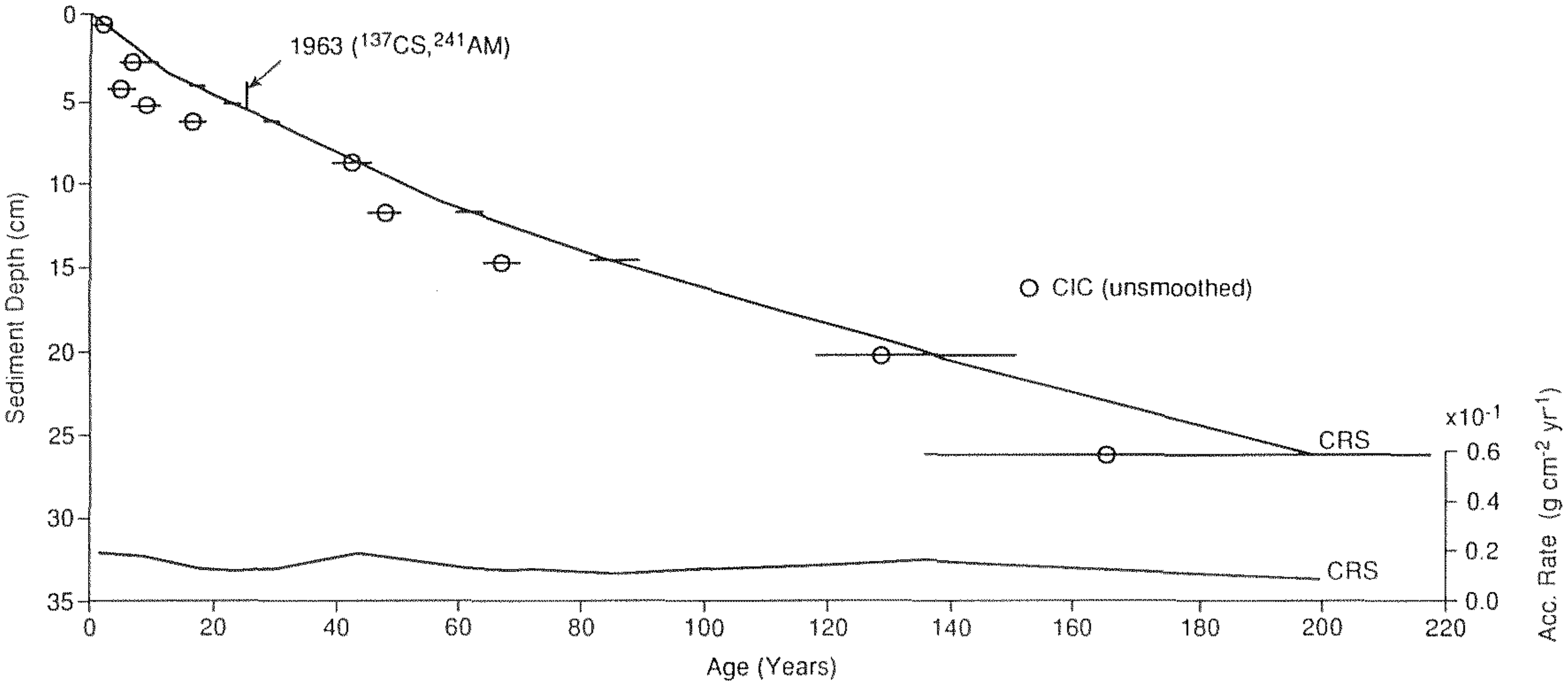


Figure 4.5 **Lough Muck: summary diatom diagram**

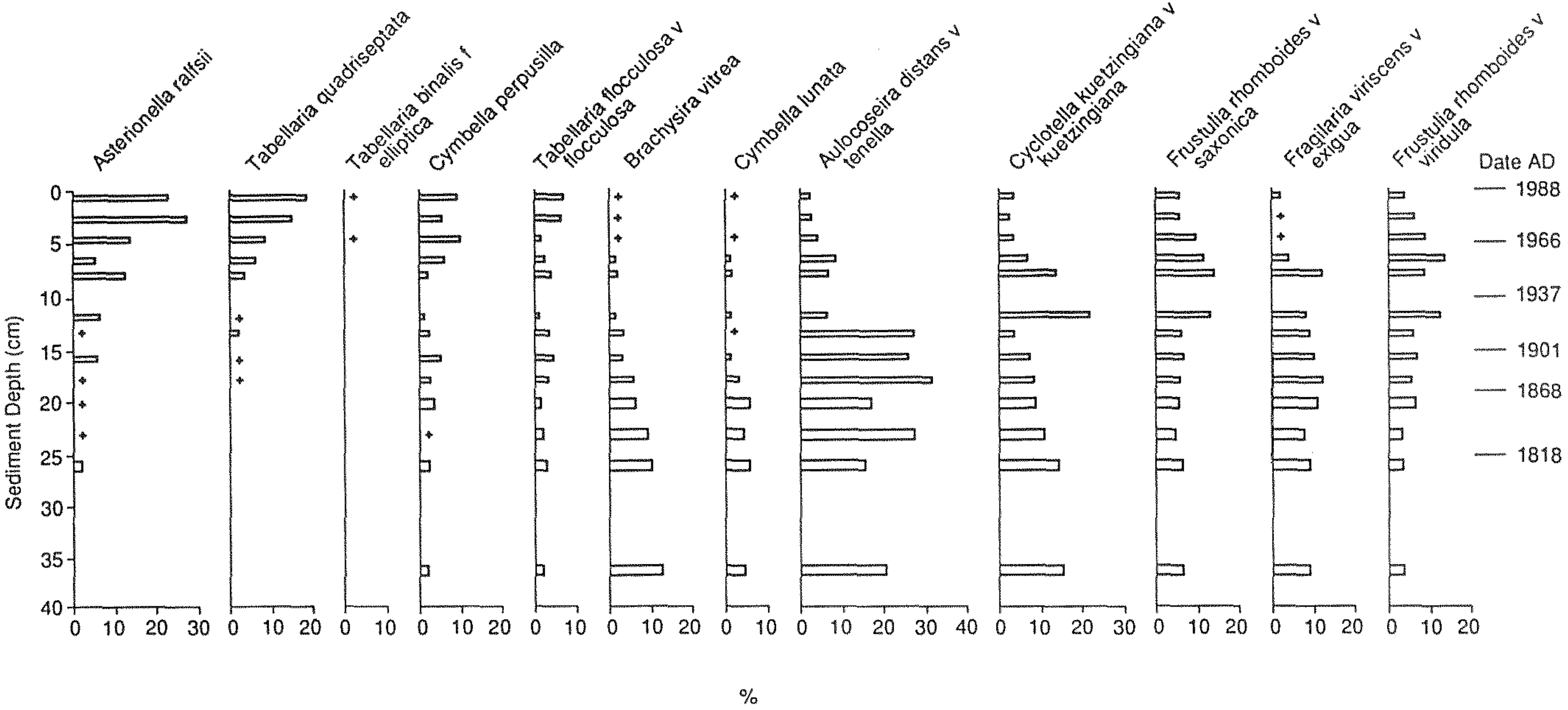


Figure 4.6 Lough Muck: variation of diatom concentrations with depth

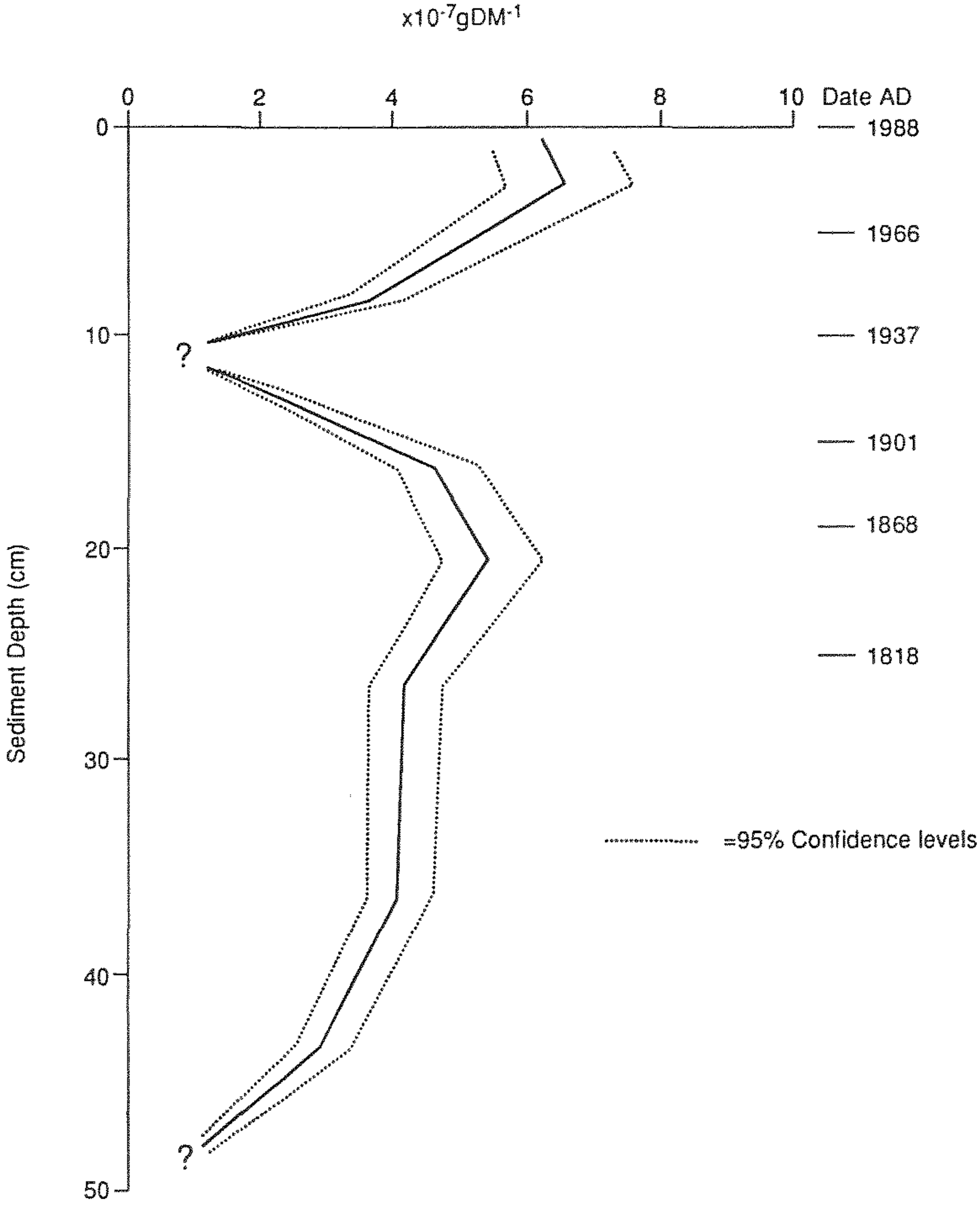


Figure 4.7 Lough Muck: pH reconstructions

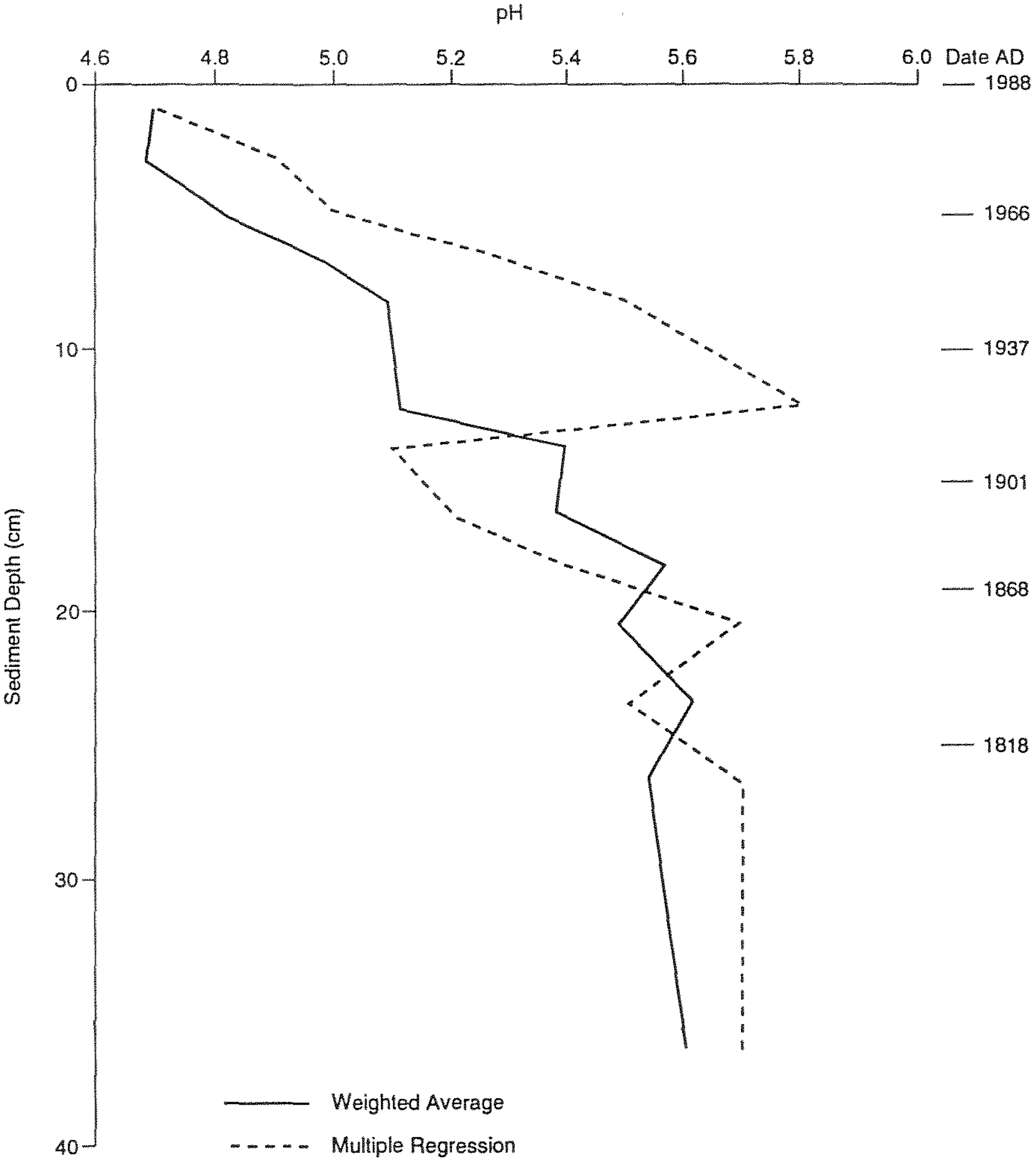


Figure 4.8 Lough Muck: variation of carbonaceous particle concentrations with depth

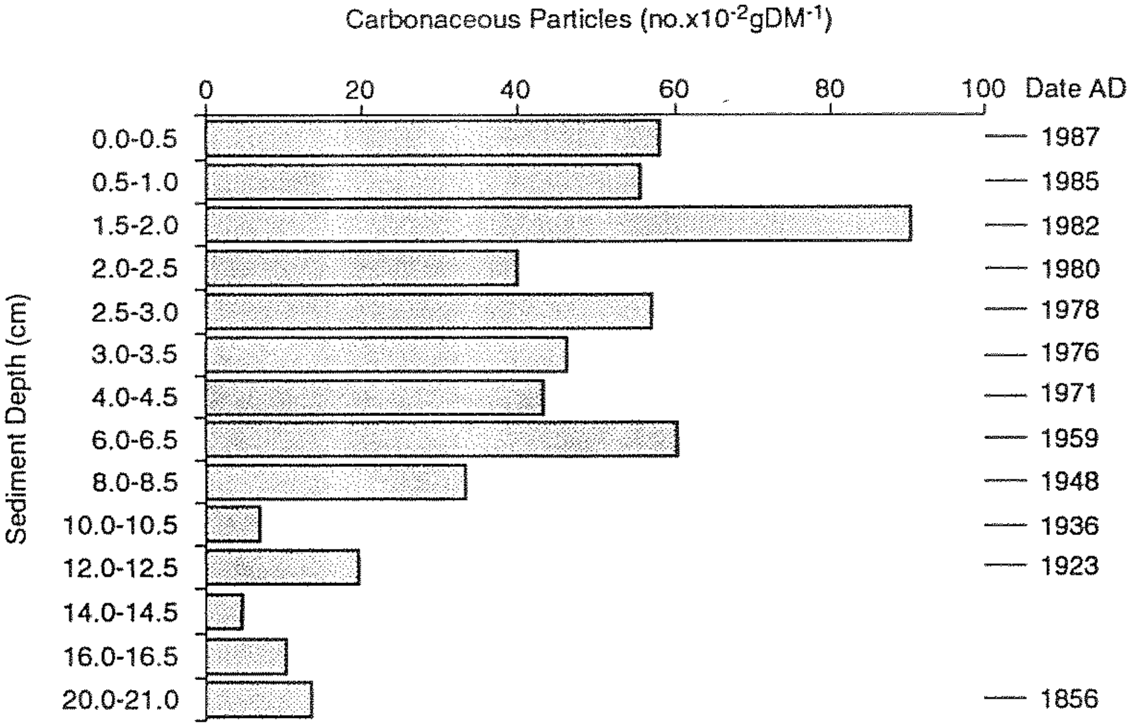


Figure 4.9 Lough Muck: summary pollen diagram

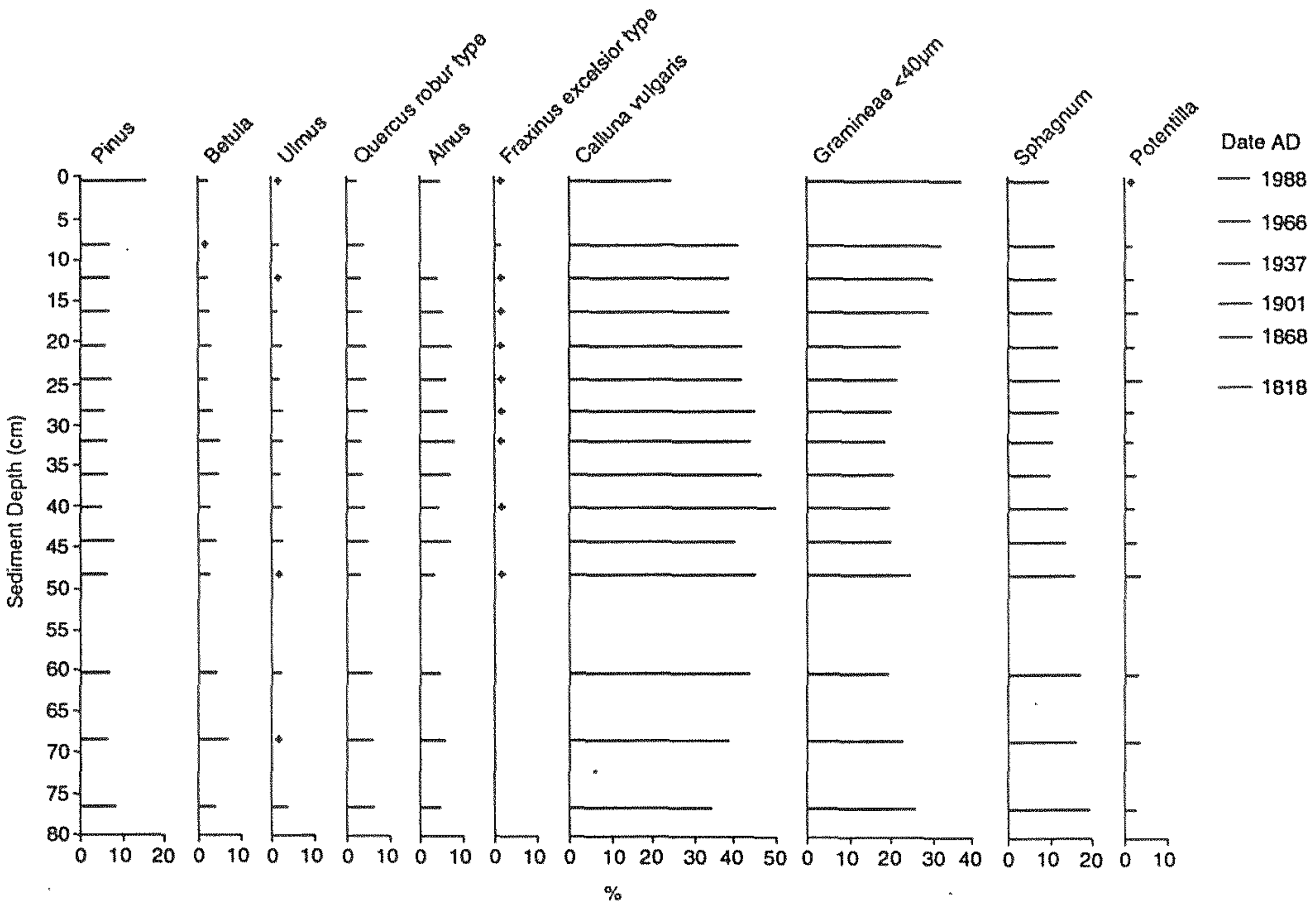


Figure 4.10 Lough Muck: *Calluna* : Gramineae ratio

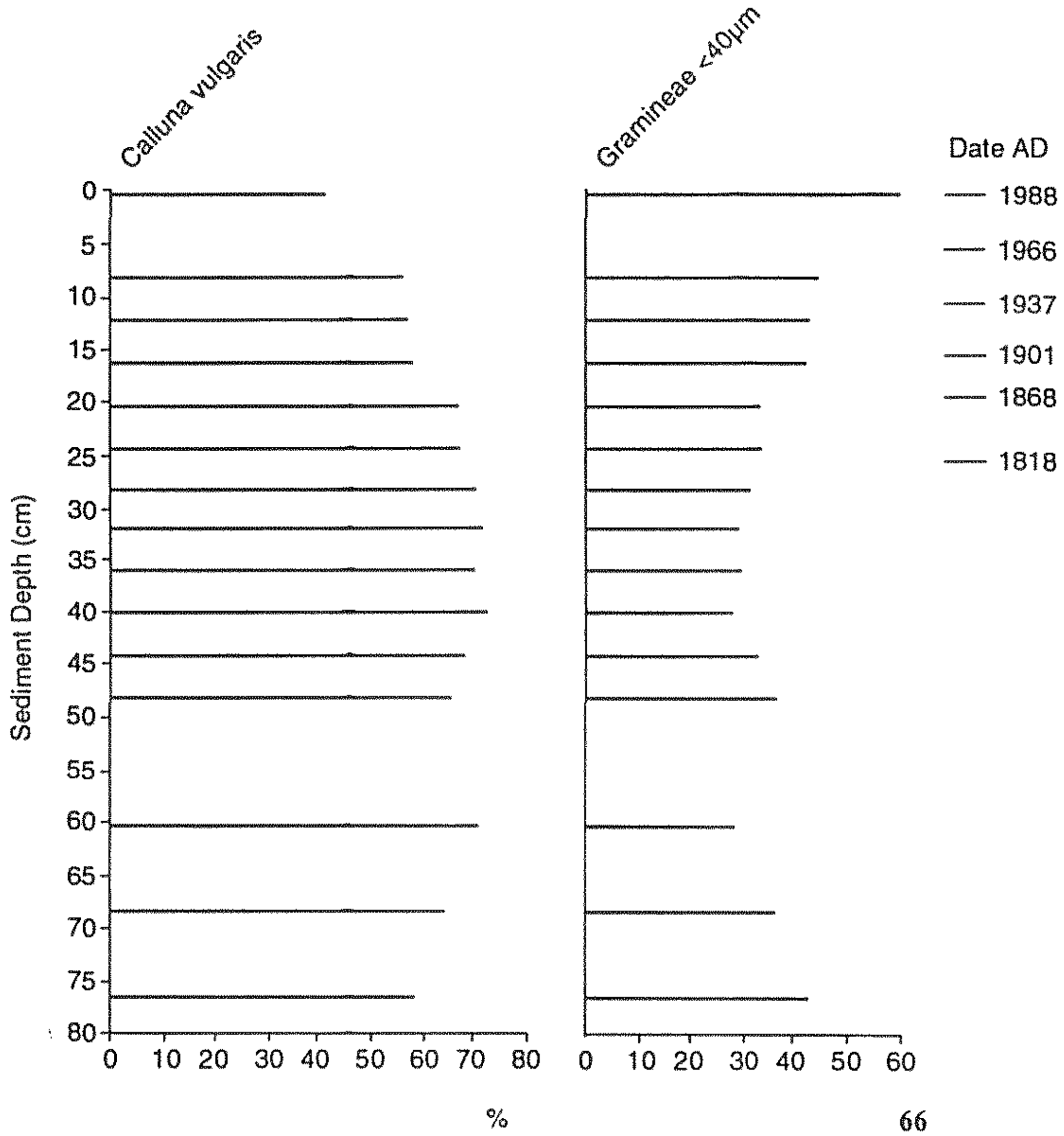


Figure 4.11 Lough Muck: variation of major ion concentrations with depth

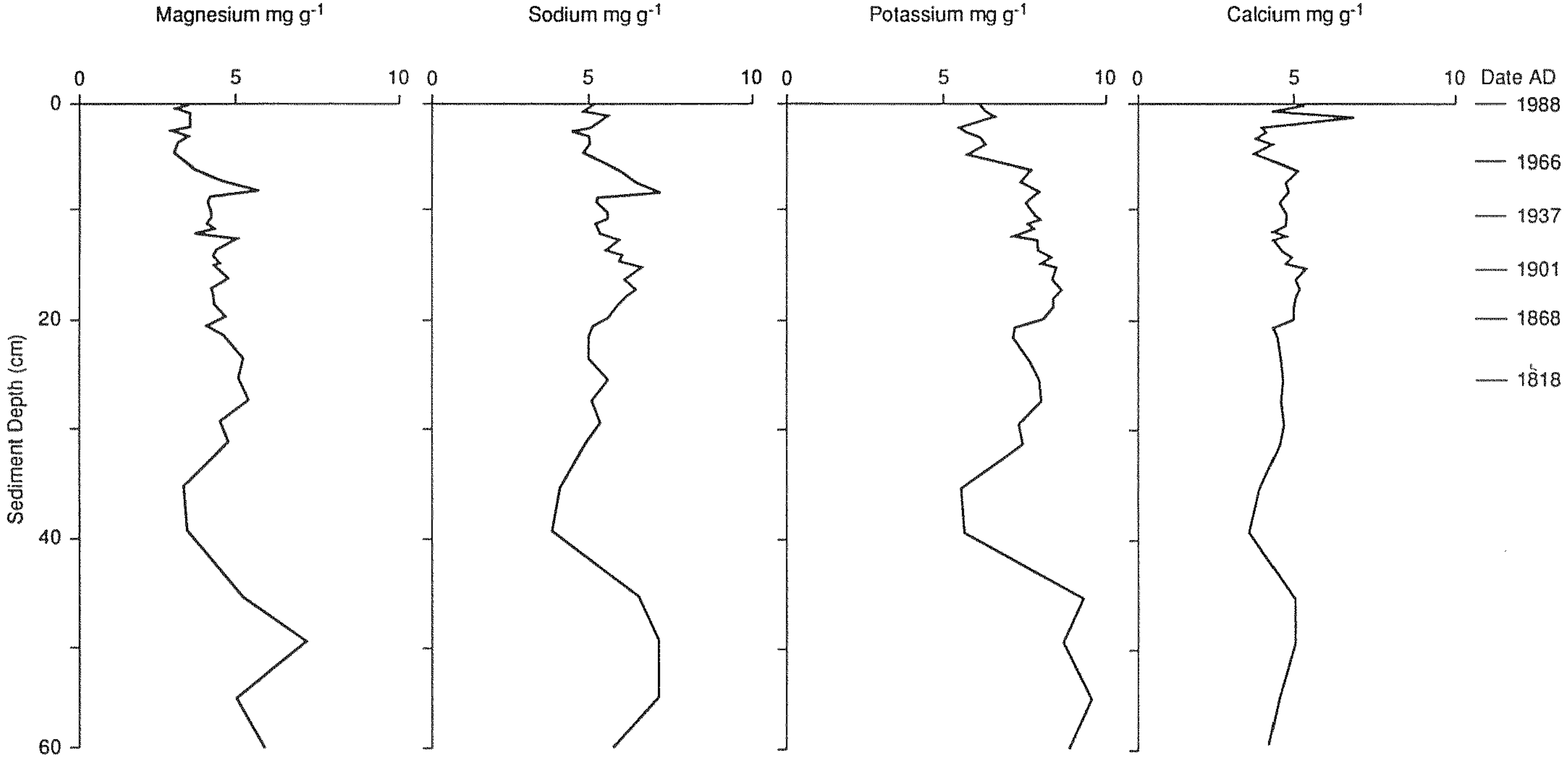
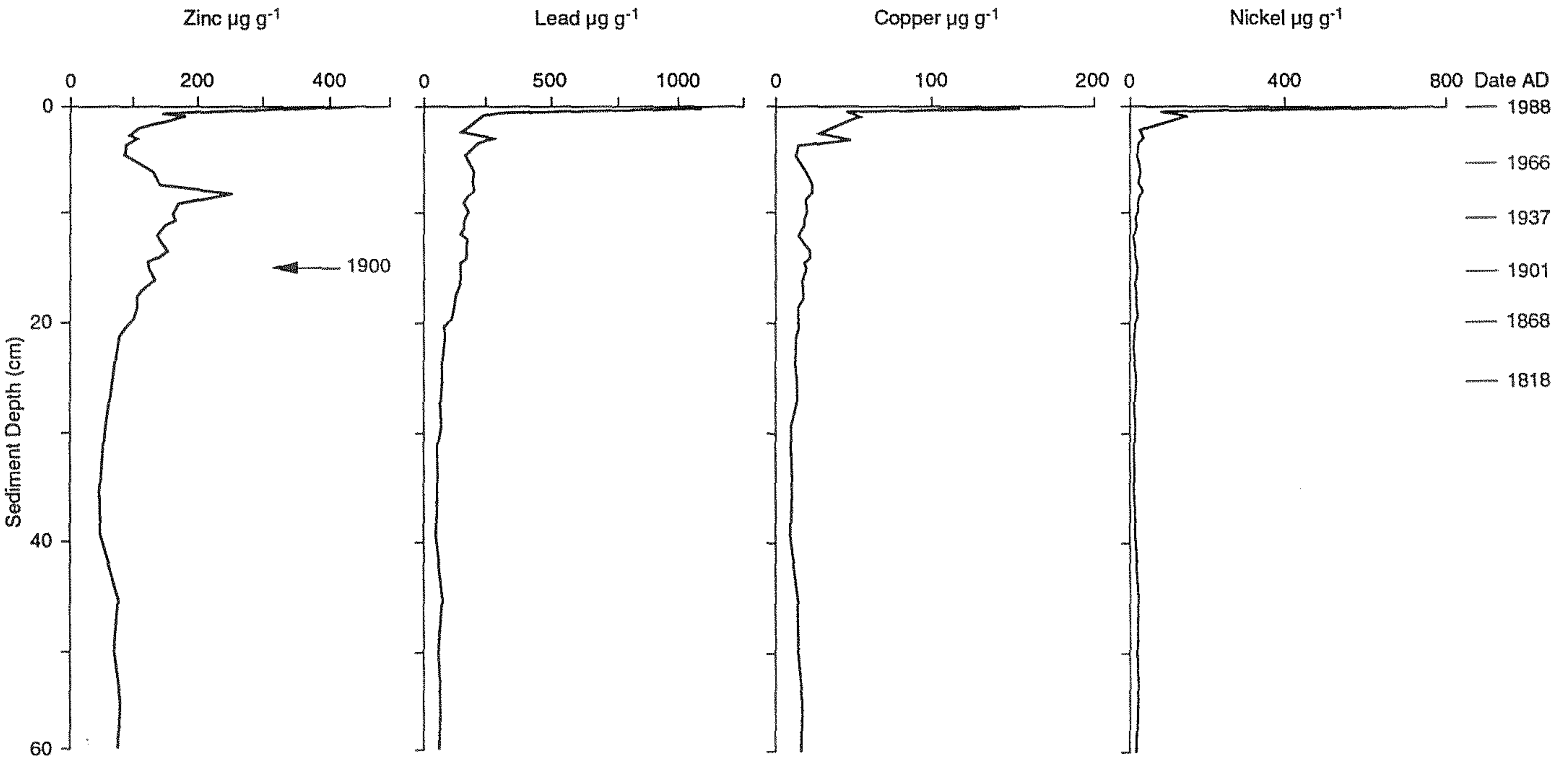


Figure 4.12 **Lough Muck: variation of trace metal concentrations with depth**



5 LOUGH MAUMWEE, COUNTY GALWAY.

Lough Maumwee lies on granite rocks near Maam Cross, County Galway (Figure 1.1). This location provides for comparison with lakes selected in Donegal since the extent and effects of acid deposition are expected to be different (Bailey *et al.* 1986). According to the non-marine sulphur deposition map for Ireland (Figure 1.2), the Lough Maumwee area receives a sulphur loading of 0.6-0.8 g m^{-2} yr^{-1} or <0.4 g m^{-2} yr^{-1}, according to different authorities (Jordan and Enlander 1990, Bowman 1991, respectively). This loading seems to reflect the effect of sulphur emissions from electricity generation by power utilities at Moneypoint and Tarbet on the Shannon Estuary. In 1984 and 1985 the weighted mean pH rainfall in the area was 4.6 and 4.4, respectively (Bowman 1986).

Site description

Lough Maumwee is a relatively shallow lowland (altitude 48 m) lake measuring about 1.0 by 0.6 km and has a maximum depth of almost 8 m near the north-eastern shore (see Bowman 1987). Catchment geology consists of quartzites and granite. Blanket peat is abundant and the moorland vegetation is dominated by *Molinia* grassland. Peat cutting is extensive in the western part of the catchment and some sheep grazing is practised. The catchment has probably been burnt periodically.

The lake water is moderately coloured by dissolved humic material but supports populations of trout, salmon, minnow and eels (Bowman 1987). The aquatic vegetation of Lough Maumwee is fairly sparse but is unusual in supporting a stand of *Cladium mariscus*, a plant characteristic of richer environments, and *Eriocaulon septangulare* (the pipewort) which occurs in the littoral zone.

Water chemistry of Lough Maumwee was measured on two occasions in 1988 (Table 5.1) and more data for 1984-1989 are available in Bowman (1986, 1991). The data show that the lake is mildly acidic (pH c. 6) and that calcium and alkalinity values are relatively high. The water chemistry is broadly similar to that of Lough Veagh. As with the other Irish sites in Donegal a major difference is apparent between the water chemistry on the two 1988 sampling dates. Again, the high concentrations of sodium and chloride ions in the May sample indicates a seasalt incursion into the lake prior to sampling.

The phytoplankton and benthic invertebrates of Lough Maumwee were extensively surveyed in 1987-1989 (Bowman 1991). Flagellate algae dominated the phytoplankton with *Peridinium inconspicuum* being particularly common. A maximum biomass of 80 mm^3 m^3 was reached in July of each year. Plecoptera and Chironomids dominated the benthic fauna of the lake.

Table 5.1 Water chemistry data for Lough Maumwee as sampled in May and September 1988

		May 1988	September 1988
pH		6.01	5.94
Conductivity	μS cm^{-1}	95.0	*
Alkalinity (Alk_e)	μeq l^{-1}	31.0	50.0
Calcium	μeq l^{-1}	102.0	56.0
Magnesium	μeq l^{-1}	132.0	64.0
Sodium	μeq l^{-1}	648.0	226.0
Potassium	μeq l^{-1}	13.0	7.0
Sulphate	μeq l^{-1}	92.0	40.0
Chloride	μeq l^{-1}	800.0	293.0
Aluminium (total acid soluble)	μg l^{-1}	6.0	7.0
Aluminium (labile)	μg l^{-1}	*	1.0
Colour (absorbance)	250 nm	0.12	0.20

* = not determined

Methods

An 83 cm long core (core MAUM1) from Lough Maumwee was analysed according to the methods of Stevenson *et al.* (1987).

Lithostratigraphy

The core consisted of fine black detrital sediment (colour 10YR 2/2) in the upper 6 cm; below 32 cm depth the sediment texture changed slightly with sand grains becoming more frequent. Gross stratigraphic changes in sediment percentage dry weight, LOI and wet density are shown in Figure 5.1. Changes are relatively minor with LOI values varying around c. 25%. However, small peaks in the percentage dry weight curve indicate relatively minor pulses of minerogenic inwash material at 80, 61, and c. 38 cm depth. These peaks correspond with low LOI values. Above the 38 cm minerogenic peak, sediment organic matter increases irregularly from about 10% to between about 25% and 30% and rising to a single peak of 35% at 25 cm depth. There is a slight decrease in LOI over the upper 5 cm of sediment.

These down core changes in gross sediment stratigraphy reflect a sustained period of catchment disturbance, probably by peat cutting and also by burning to improve grazing. It is likely that the minerogenic pulses arise from erosion following the cutting of peat to the level of minerogenic subsoils.

Dating

Sediment samples from Lough Maumwee core MAUM1 were analysed for ^{210}Pb, ^{226}Ra, ^{137}Cs, ^{134}Cs and ^{241}Am (Appleby *et al.* 1986). The ^{210}Pb and ^{226}Ra results are given in Table 5.2. The ^{137}Cs, ^{134}Cs and ^{241}Am results are given in Table 5.3. The ^{134}Cs derives from Chernobyl fallout and has been corrected for decay since May 1986.

Figure 5.2 plots dates calculated using the CRS and CIC ^{210}Pb dating models (Appleby and Oldfield 1978). Above 8 cm (dated 1960) there is good agreement between the two sets of dates, indicating a constant sedimentation rate during the past 30 years of c. 0.027 g cm^{-2} yr^{-1}. Below this level there is a small divergence between dates produced by the two models, indicating a period of moderate acceleration in sediment accumulation. The CRS model suggests a more abrupt change sometime after 1954. Prior to 1930, both models indicate a constant sedimentation rate of c. 0.015 g cm^{-2} yr^{-1}, except for an earlier episode of accelerated accumulation dated by CRS to c. 1890. Because of the apparent dilution of ^{210}Pb this episode could not be dated by the CIC model. The sediments associated with this event have elevated ^{226}Ra and ^{40}K activities, and it may relate to an inwash of more minerogenic sediment.

The unsupported ^{210}Pb inventory of the core is 16.7 pCi g^{-1}, corresponding to a mean ^{210}Pb flux of 0.52 pCi cm^{-2} yr^{-1}. This is comparable to the expected atmospheric flux. In view of this, and the direct evidence of ^{210}Pb dilution near the core base, the CRS model dates are thought to be more reliable than the CIC dates and are given in Table 5.4.

The artificial fallout isotope ^{137}Cs was of no chronological value in this core. The high surficial ^{134}Cs activity (Table 5.3) reflects the effect of Chernobyl fallout. Using the ^{134}Cs activity to identify that component of the total ^{137}Cs activity derived from Chernobyl fallout, shows that the weapons test component also has its maximum value in the surficial sediment and thus does not provide a reliable record of atmospheric deposition. ^{241}Am was not detectable in the Lough Maumwee core.

Table 5.2 Lough Maumwee: ^{210}Pb and ^{226}Ra data

Depth	Dry mass	^{210}Pb Conc.				^{226}Ra Conc.	
		Total		Unsupp.			
cm	g cm^{-2}	pCi g^{-1}	±	pCi g^{-1}	±	pCi g^{-1}	±
0.50	0.0532	19.35	1.05	18.90	1.06	0.45	0.15
2.75	0.2757	15.26	1.12	14.05	1.16	1.21	0.30
5.75	0.5595	12.46	1.12	9.63	1.18	2.83	0.37
7.75	0.7452	10.85	0.81	8.65	0.82	2.20	0.15
10.75	1.0472	9.23	0.66	6.80	0.68	2.43	0.15
14.25	1.4104	7.01	0.65	5.19	0.66	1.82	0.13
16.75	1.6801	4.78	0.42	2.82	0.45	1.96	0.15
20.50	2.0906	3.82	0.35	0.56	0.37	3.26	0.12
25.50	2.6093	3.55	0.30	1.12	0.31	2.43	0.09
31.50	3.3463	2.55	0.22	-0.23	0.23	2.78	0.07

Table 5.3 Lough Maumwee: ^{137}Cs and ^{134}Cs data

Depth	^{137}Cs Conc.		^{134}Cs Conc.	
cm	pCi g^{-1}	±	pCi g^{-1}	±
0.50	35.41	0.43	11.67	0.64
2.75	15.66	0.36	2.30	0.54
5.75	5.47	0.31	0.00	0.00
7.75	3.23	0.16	0.00	0.00
10.75	1.53	0.13	0.00	0.00
14.25	0.96	0.11	0.00	0.00
16.75	0.63	0.13	0.00	0.00
20.50	0.44	0.07	0.00	0.00
25.50	0.08	0.05	0.00	0.00
31.50	0.05	0.03	0.00	0.00

Table 5.4 Lough Maumwee: ^{210}Pb chronology

Depth	Cum. dry mass	Chronology			Sedimentation rate		
		Date	Age				
cm	g cm^{-2}	AD	yr	±	g cm^{-2}yr^{-1}	cm yr^{-1}	± (%)
0.00	0.0000	1988	0				
1.00	0.1026	1984	4	2	0.0260	0.262	7.1
2.00	0.2015	1980	8	2	0.0264	0.270	8.0
3.00	0.2993	1977	11	2	0.0268	0.278	8.9
4.00	0.3939	1973	15	2	0.0273	0.286	10.3
5.00	0.4885	1970	18	2	0.0277	0.293	11.6
6.00	0.5827	1966	22	2	0.0277	0.294	12.3
7.00	0.6756	1963	25	2	0.0263	0.273	11.2
8.00	0.7704	1959	29	2	0.0249	0.254	10.5
9.00	0.8710	1955	33	2	0.0236	0.238	10.8
10.00	0.9717	1950	38	3	0.0223	0.221	11.2
11.00	1.0731	1946	42	3	0.0209	0.204	11.9
12.00	1.1769	1940	48	3	0.0191	0.185	13.2
13.00	1.2807	1934	54	4	0.0173	0.166	14.6
14.00	1.3845	1929	59	4	0.0155	0.147	16.0
15.00	1.4913	1922	66	5	0.0153	0.144	18.3
16.00	1.5992	1915	73	6	0.0157	0.145	20.8
17.00	1.7075	1908	80	7	0.0177	0.163	25.3
18.00	1.8169	1903	85	8	0.0247	0.230	35.9
19.00	1.9264	1897	91	9	0.0317	0.297	46.4
20.00	2.0359	1891	97	10	0.0387	0.364	56.9
21.00	2.1425	1886	102	11	0.0389	0.366	60.1
22.00	2.2462	1881	107	13	0.0323	0.302	56.1
23.00	2.3499	1875	113	14	0.0256	0.238	52.1
24.00	2.4537	1870	118	15	0.0190	0.175	48.1
25.00	2.5574	1864	124	17	0.0123	0.111	44.1

Diatom analysis

Diatom taxa frequency profiles show remarkably little variation throughout the core (Figure 5.3). At all levels the diatom assemblage is dominated by *Brachysira vitrea* and *Fragilaria virescens* v. *exigua*. Frequencies vary by less than ± 5% between levels, a degree of variation which is little more than would be expected by making replicate counts on sub-samples from the same level (cf. Renberg 1990). The only changes apparent in the core are slightly increased abundance of *Navicula cocconeiformis* and the appearance of *Eunotia exigua* at and above 22 cm depth (c. 1880).

The diatom concentration in this core shows considerable variation (Figure 5.4). Low concentration values between 35 and 40 cm depth are probably the result of dilution associated with minerogenic inwash (38 cm peak). Otherwise, there is no clear correspondence between diatom concentration and lithostratigraphic changes. In this core changes in rate of sediment accumulation probably have a controlling effect on valve concentrations and the diatom taxa frequency diagram offers no evidence of changing diatom productivity in the lake. Over the dated section of the core (upper 25 cm) the c. threefold increase in sediment accumulation rate between 25 and 20 cm depth (1864-1891) does indeed coincide with declining diatom concentration. The same link between sediment accumulation rate and diatom concentration occurs between 14 cm depth and the core top (1929-1988), however, this concentration decline is disproportionately greater than in the earlier period.

As predicted by the stable species composition of diatom assemblages in this core there is very little change in inferred pH history for the lake (Figure 5.5) calculated by either the weighted averaging (WA) (Birks *et al.* 1990) or multiple regression methods (MR) (Flower 1986). Inferred pH changes by both methods are 0.1 pH units or less and are well within the statistical error of the method (Birks *et al.* 1990). The surface sediment pH is 6.0 for the MR method and coincides almost exactly with the measured pH for the lake in 1988. WA pH values follow approximately the same trend as MR values but, at each level, are consistently about 0.3 pH units lower.

Carbonaceous particle analysis

The carbonaceous particle profile (Figure 5.6) for Lough Maumwee shows very low levels of contamination below 18 cm depth or before the turn of this century. Above this level particle concentration gradually increases until 4 cm depth (late 1960s) and then sharply increases to a maximum of almost 5,000 particles g^{-1} dry sediment at 1.0-1.5 cm depth or the early 1980s. There is a very slight decline in particle concentration in the top 1 cm of sediment.

The maximum concentration of carbonaceous particles in Lough Maumwee sediment is less than that recorded in the Donegal sites, but about three times higher than in Loch Teanga in the Outer Hebrides (cf. Flower *et al.* 1992). Compared with contaminated sites in south-west Scotland where particle concentration can exceed 100,000 g^{-1} dry sediment (Flower *et al.* 1990), Lough Maumwee is not heavily impacted by this form of atmospheric pollution.

Pollen analysis

The summary pollen diagram (Figure 5.7) shows no major changes in frequency. However, it is notable that although the landscape around Lough Maumwee is essentially treeless, tree pollen constitutes c. 30% of the down-core pollen sum and that above about 60 cm depth, *Pinus* pollen usually exceeds 10%. Since natural *Pinus* woodland is long extinct in this part of Ireland the only likely source of its pollen is from reworking of fossil material by peat extraction. Ruderal pollen types are also relatively high throughout the profile, again suggesting widespread disturbance of catchment soils.

The *Calluna* : Gramineae ratio (Figure 5.8) initially increases up to c. 45 cm but declines a little above this depth to the core top. Interpretation of this ratio is made difficult if influx of old pollen is significant, as seems to be the case at this site. However, a declining ratio towards the core top is compatible with the hypothesis that the vegetation has been modified by management practises.

With regard to the pollen record of aquatic plants (Figure 5.9) it is interesting to note that *Isoetes* spores are in relatively high abundance throughout most of the core section indicating that erosional influxes of soil particulates has not been large enough to produce a sustained impact on the distribution of this plant. Interestingly, the two levels of low frequency of *Isoetes* spores at 55 and 31 cm closely follow soil inwash pulses indicated by lithostratigraphic data.

Geochemistry

There is little change in calcium and potassium concentrations throughout the core (Figure 5.10). However, the magnesium concentration drops and sodium increases above 35 cm. Because of these changes in chemical constitution, it must be established whether increases in trace metals above 30 cm depth (Figure 5.11) are due to contamination or changes in sediment constitution. Figure 5.12 shows that the true point where the zinc and lead contamination starts is 20 cm (1892), not 30 cm. The sedimentary concentrations of zinc and lead rise a little above 30 cm but increase more markedly above 20 cm. These changes must be due to contamination from the atmosphere since there are no independent catchment sources for these trace metals.

These results show that Lough Maumwee has been receiving trace metals deposited from a contaminated atmosphere since the late-nineteenth century. Any associated reduction in lake water pH as a result of acid stress has been insignificant.

Discussion

The diatom frequency changes observed during the recent history of Lough Maumwee are small and show that the lake is not acidified by atmospheric pollution. Consequently, the inferred lake water pH values show no decline.

The sedimentary trace metal and carbonaceous particle records do, however, show that the lake is moderately impacted by atmospheric pollution, but clearly acid deposition in the area has not exceeded the critical load of the Lough Maumwee system. The water chemistry indicates that the site is not highly susceptible to the effects of acid deposition. It possesses significant alkalinity, and calcium concentrations are similar to Lough Veagh. Despite its quartzite/granite catchment, aspects of the aquatic vegetation, such as the occurrence of *Cladium mariscus*, further indicate that there is some additional and local supply of base cations from the catchment. It is possible that basic intrusions occur in the catchment, as found in the Lough Veagh drainage basin.

Application of the equation of Henriksen (1982) reveals a negative acidification value. If the water chemistry sample with low sea salts (Table 5.1) is used in the calculation then the acidification index becomes more strongly negative; and if DOC acidity is taken into account the index becomes still more negative indicating a greater surplus acid neutralising capacity. In whatever form the Henriksen Equation is used Lough Maumwee is shown to be unacidified. Furthermore, if the uncorrected calcium concentration (56 μeq l^{-1} - Table 5.1) and sulphur deposition of 0.7 g m^{-2} yr^{-1} are compared with the predictive diatom acidification model (Battarbee 1989), Lough Maumwee clearly falls into the unacidified category. All three techniques - diatom analysis, the Henriksen Index and the sensitivity model all indicate that Lough Maumwee is unacidified.

Lough Maumwee: conclusions

1) Diatom analysis of the Lough Maumwee sediment core has demonstrated that acid deposition has not exceeded the critical loading necessary to cause lake acidification.

2) Application of the Henriksen Equation to Lough Maumwee water chemistry indicates no lake acidification.

3) The diatom acidification model (Battarbee 1989) predicts the lake to be unacidified.

4) Land-use changes around Lough Maumwee as inferred from sediment core lithostratigraphy and pollen analysis have not caused any lake acidification.

5) Carbonaceous particle and trace metal records indicate that Lough Maumwee is moderately impacted by atmospheric pollution.

Figure 5.1 Lough Maumwee: lithostratigraphy

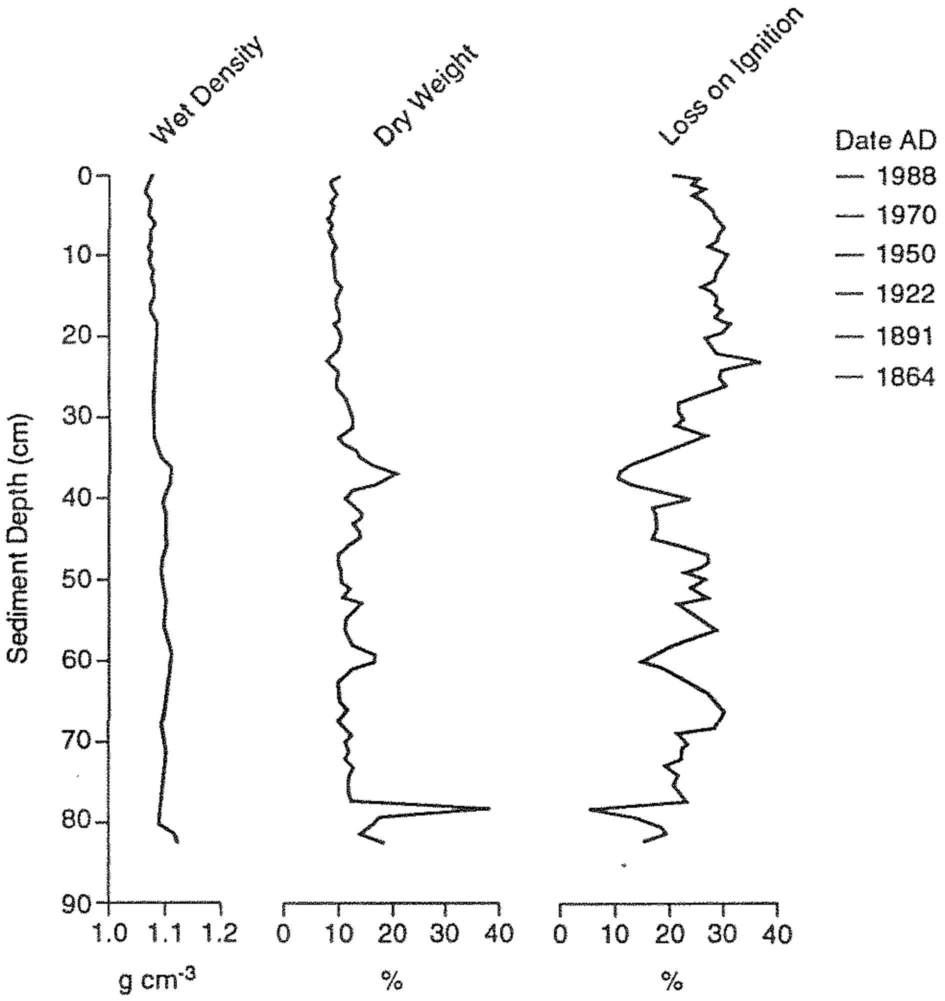


Figure 5.2 Lough Maumwee: ^{210}Pb chronology

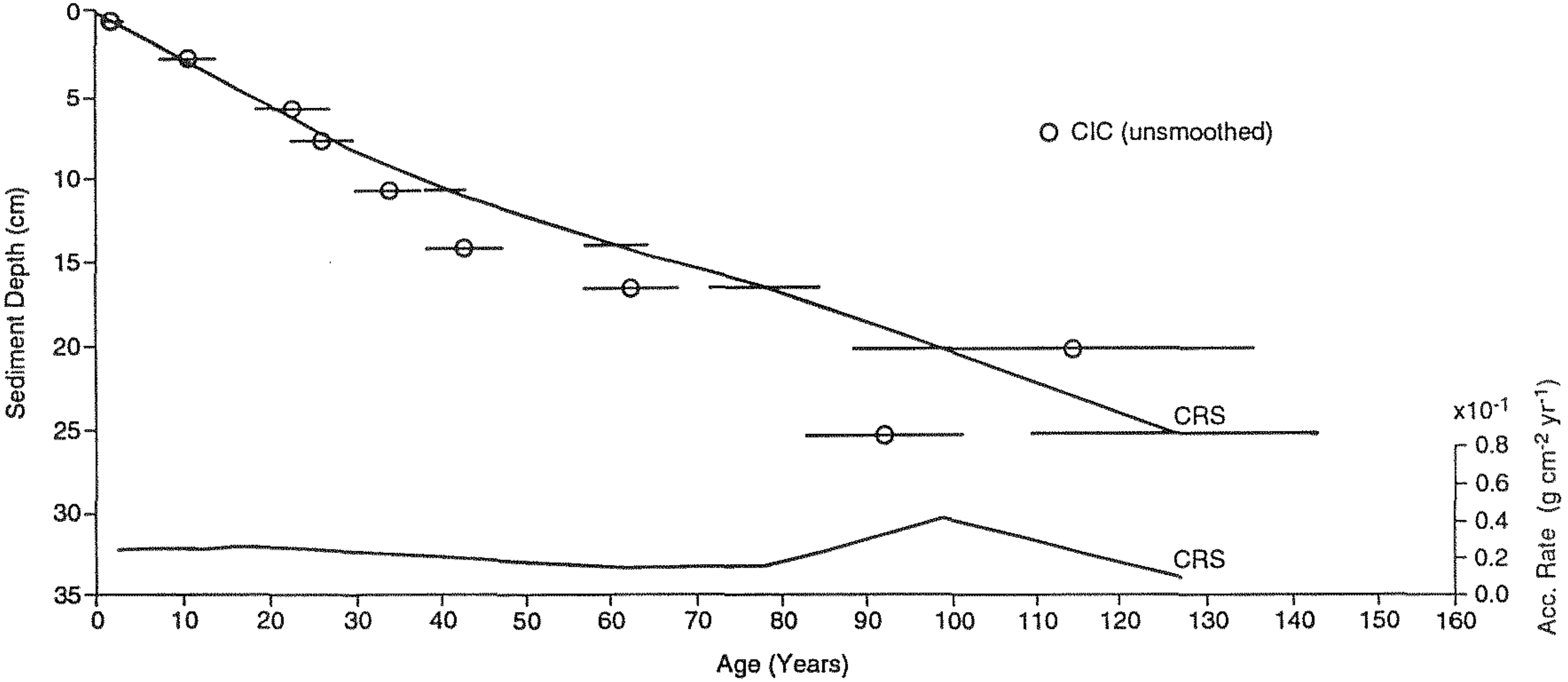


Figure 5.3 Lough Maumwee: summary diatom diagram

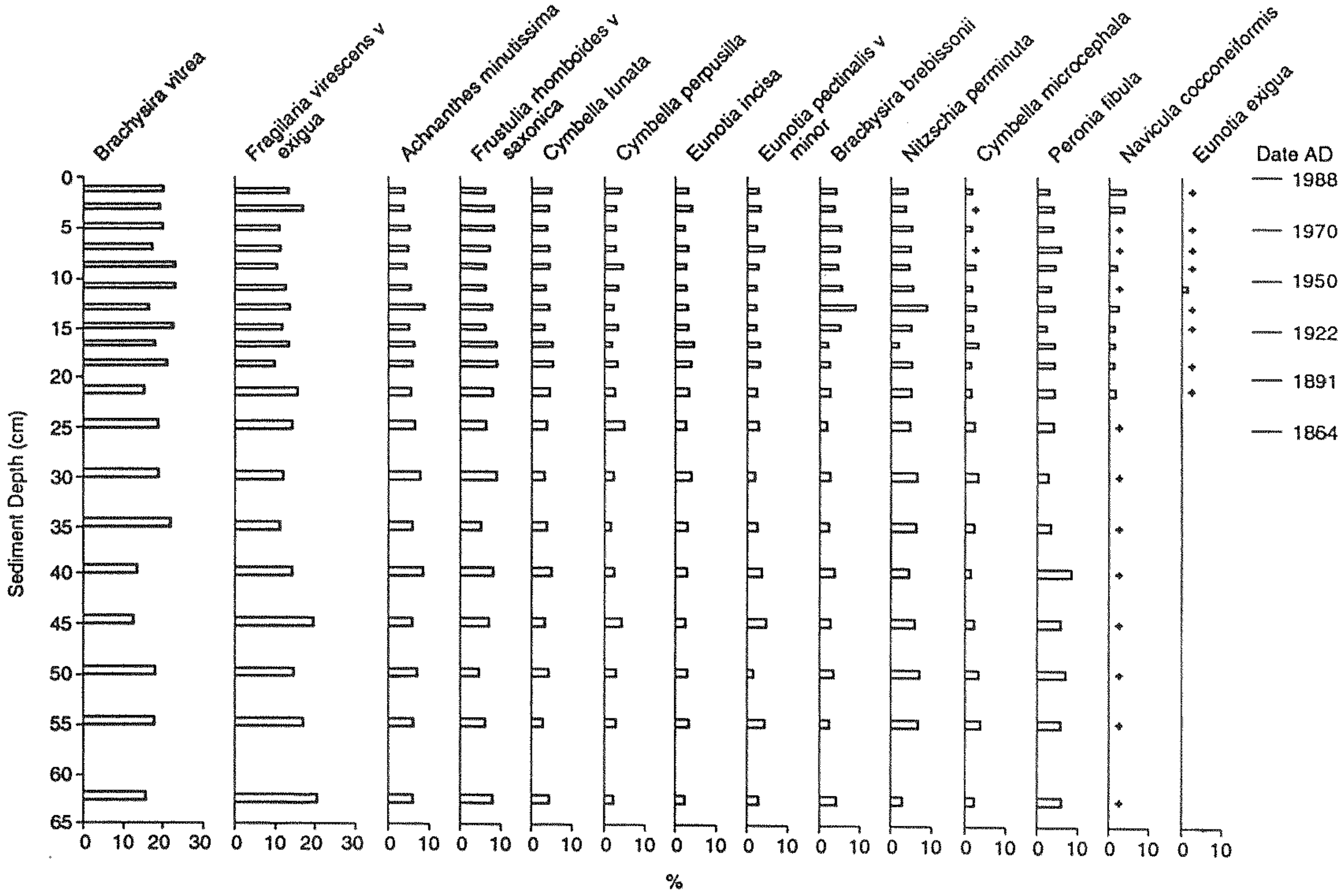


Figure 5.4 Lough Maumwee: variation of diatom concentrations with depth

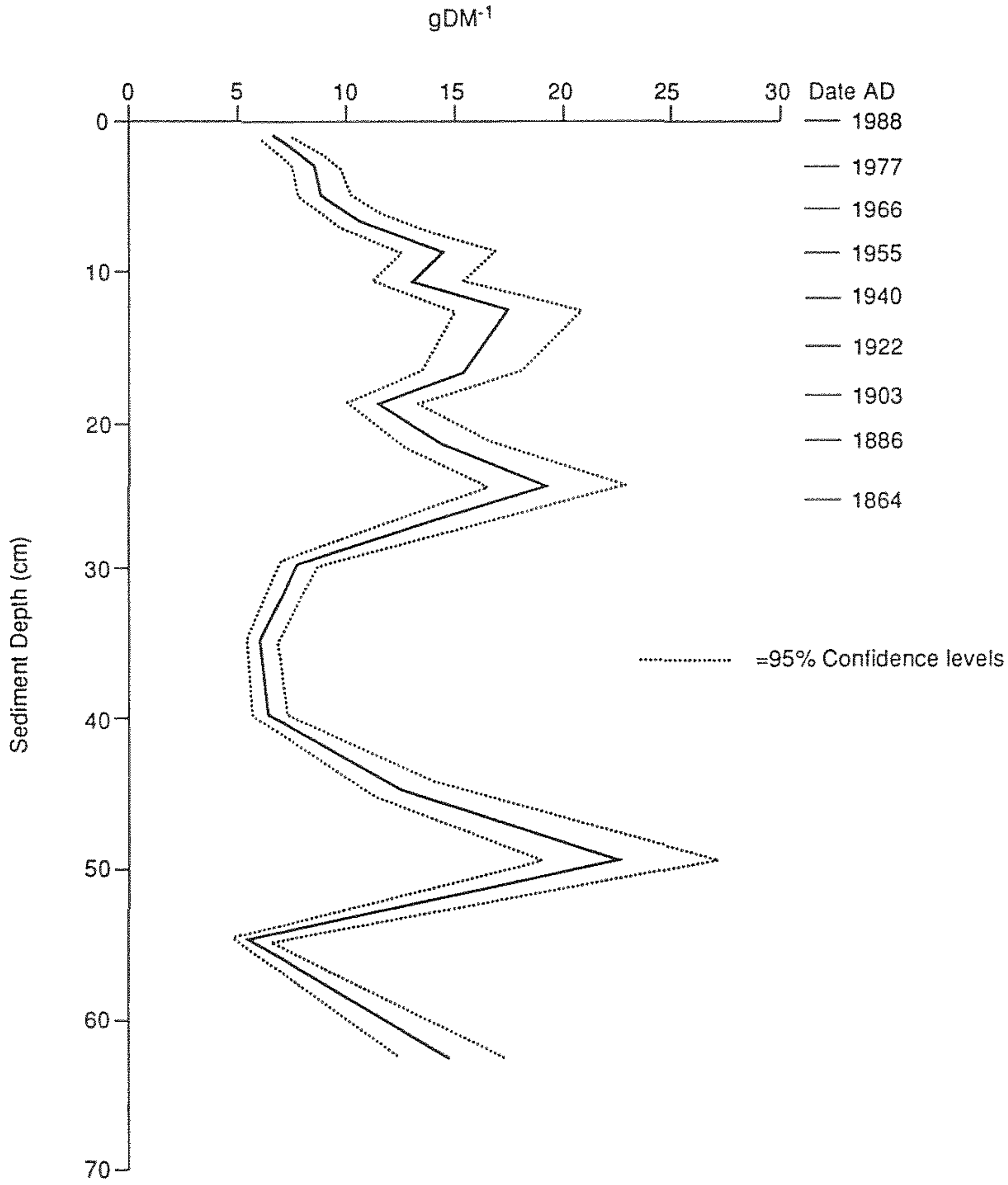


Figure 5.5 Lough Maumwee: pH reconstructions

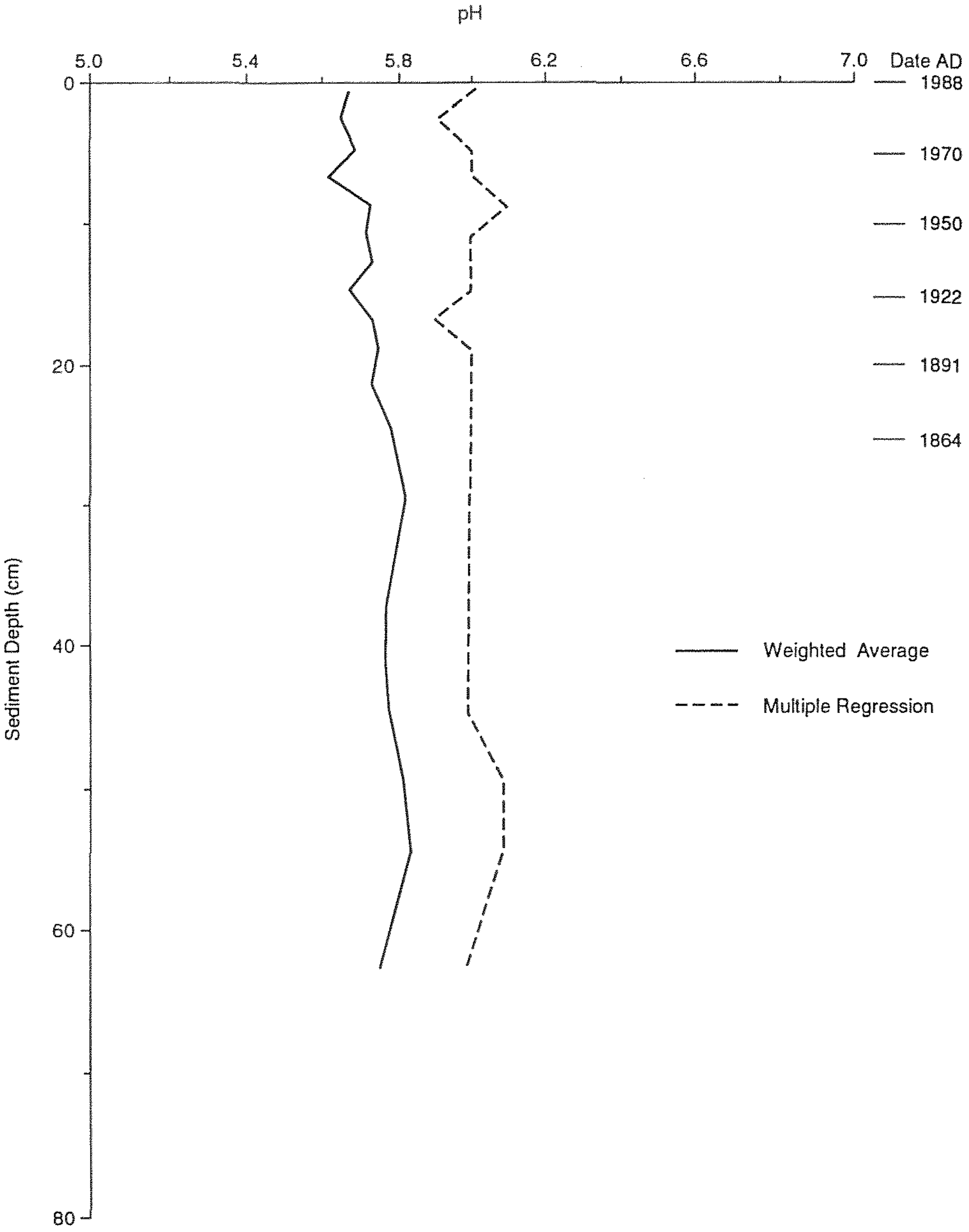


Figure 5.6 Lough Maumwee: variation of carbonaceous particle concentrations with depth

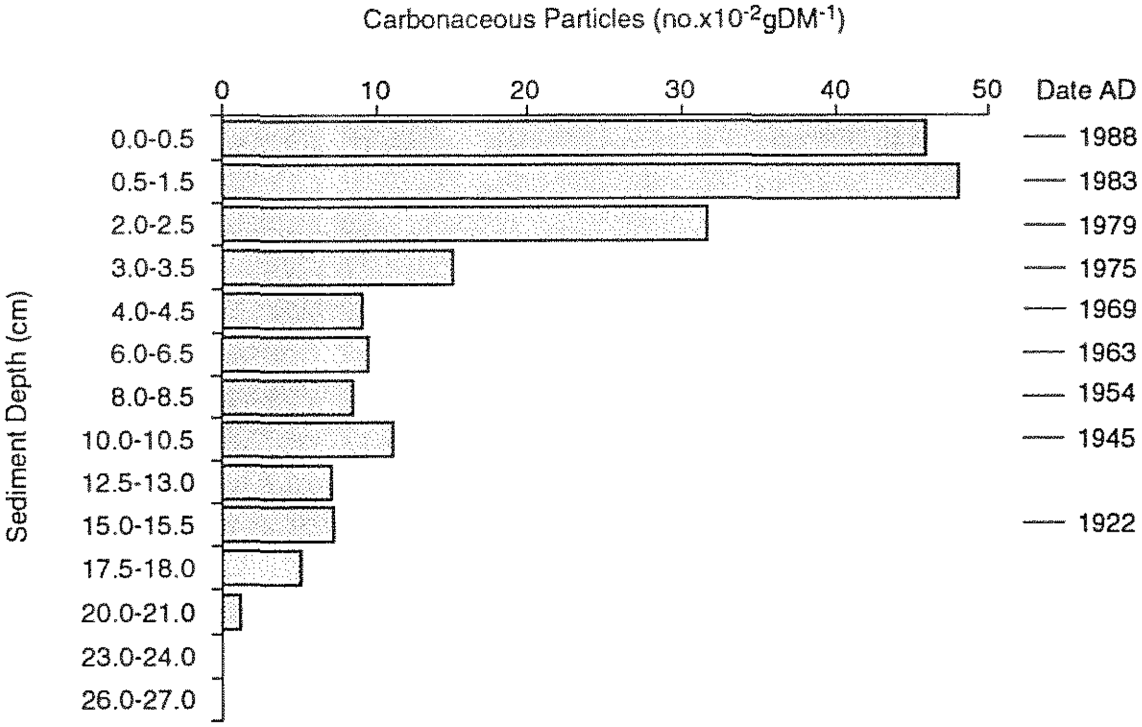


Figure 5.7 Lough Maumwee: summary pollen diagram

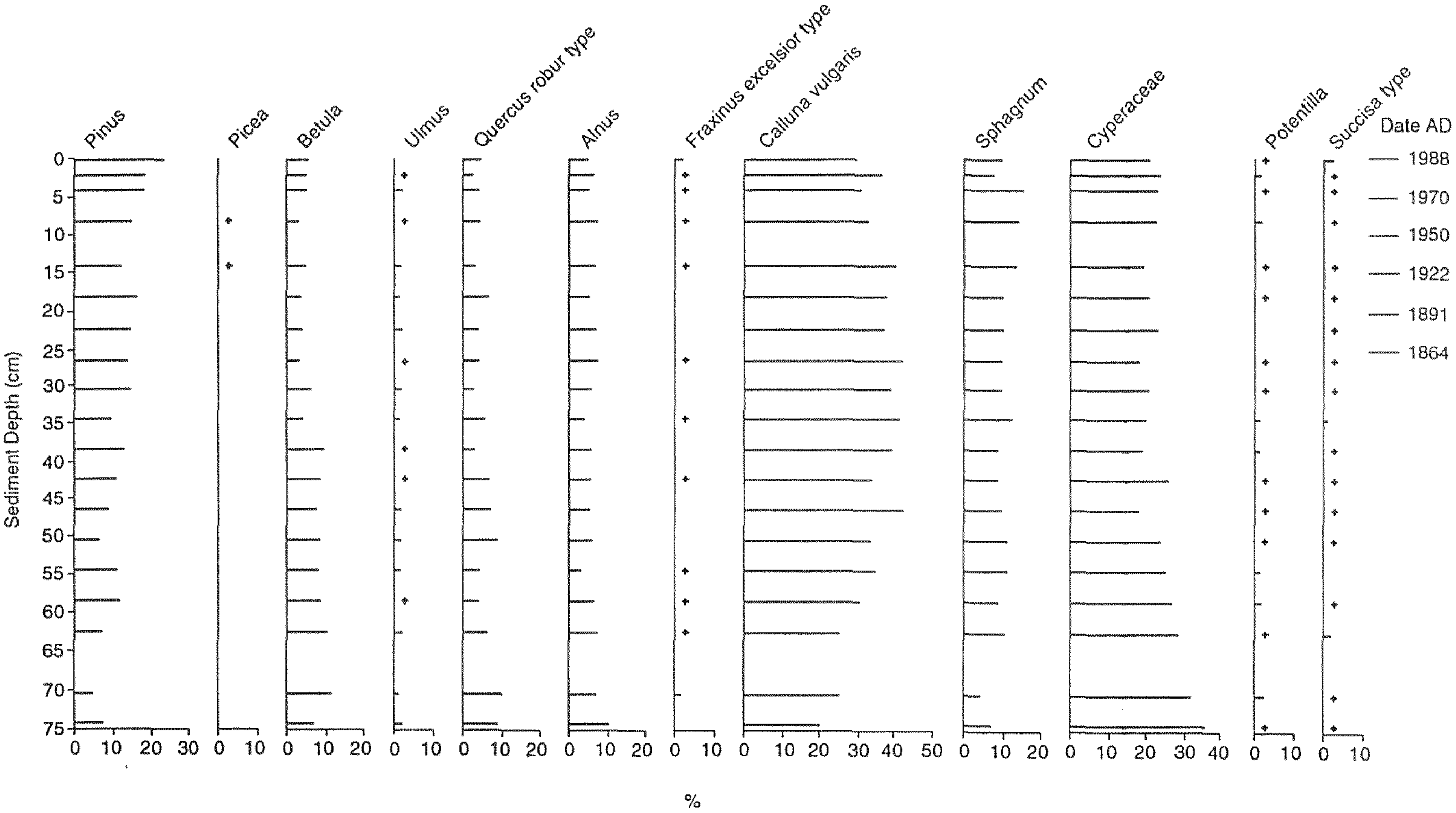


Figure 5.8 Lough Maumwee: *Calluna* : Gramineae ratio

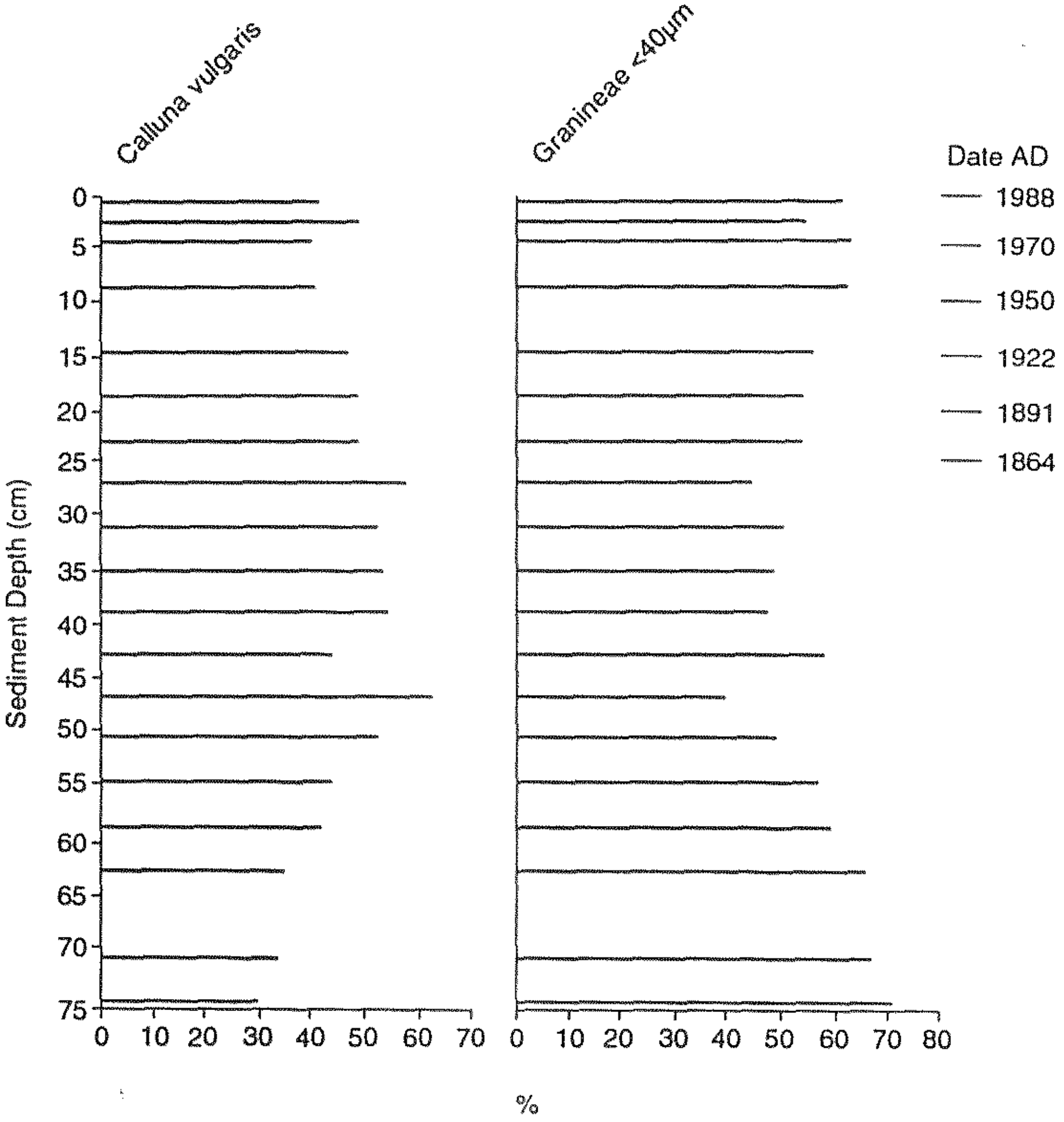


Figure 5.9 Lough Maumwee: aquatic pollen, summary diagram

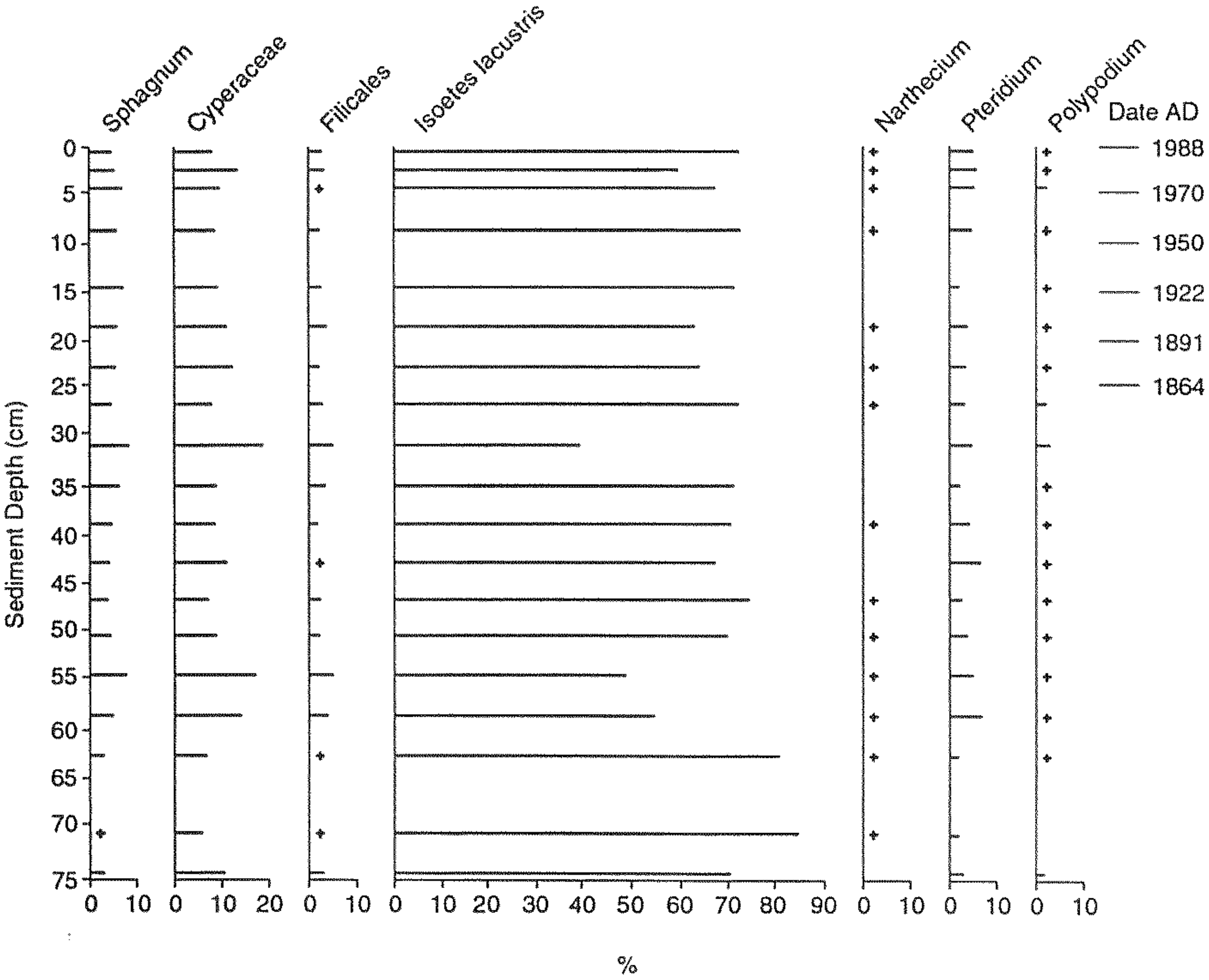


Figure 5.10 Lough Maumwee: variation of major ion concentrations with depth

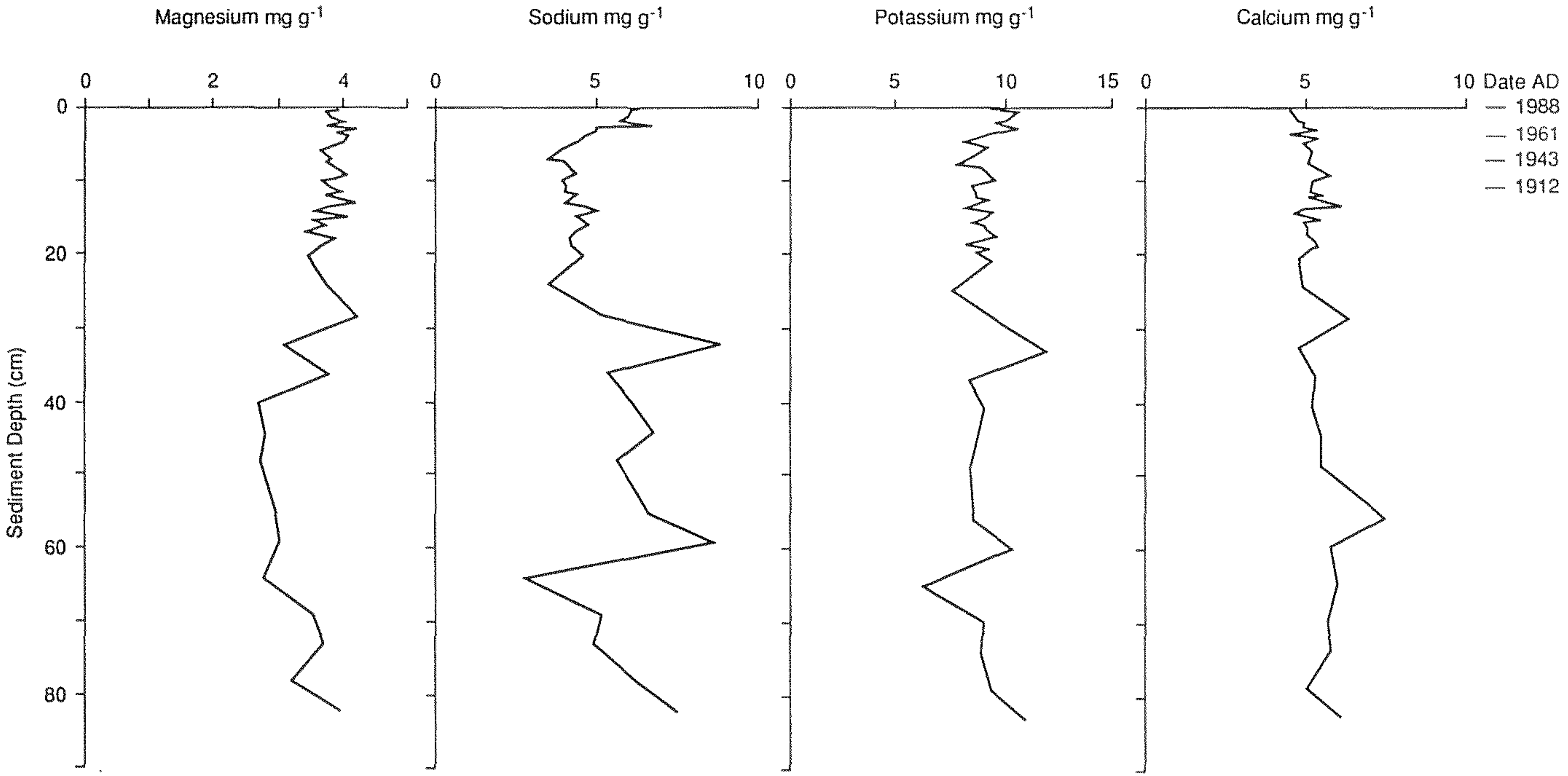


Figure 5.11 Lough Maumwee: variation of trace metal concentrations with depth

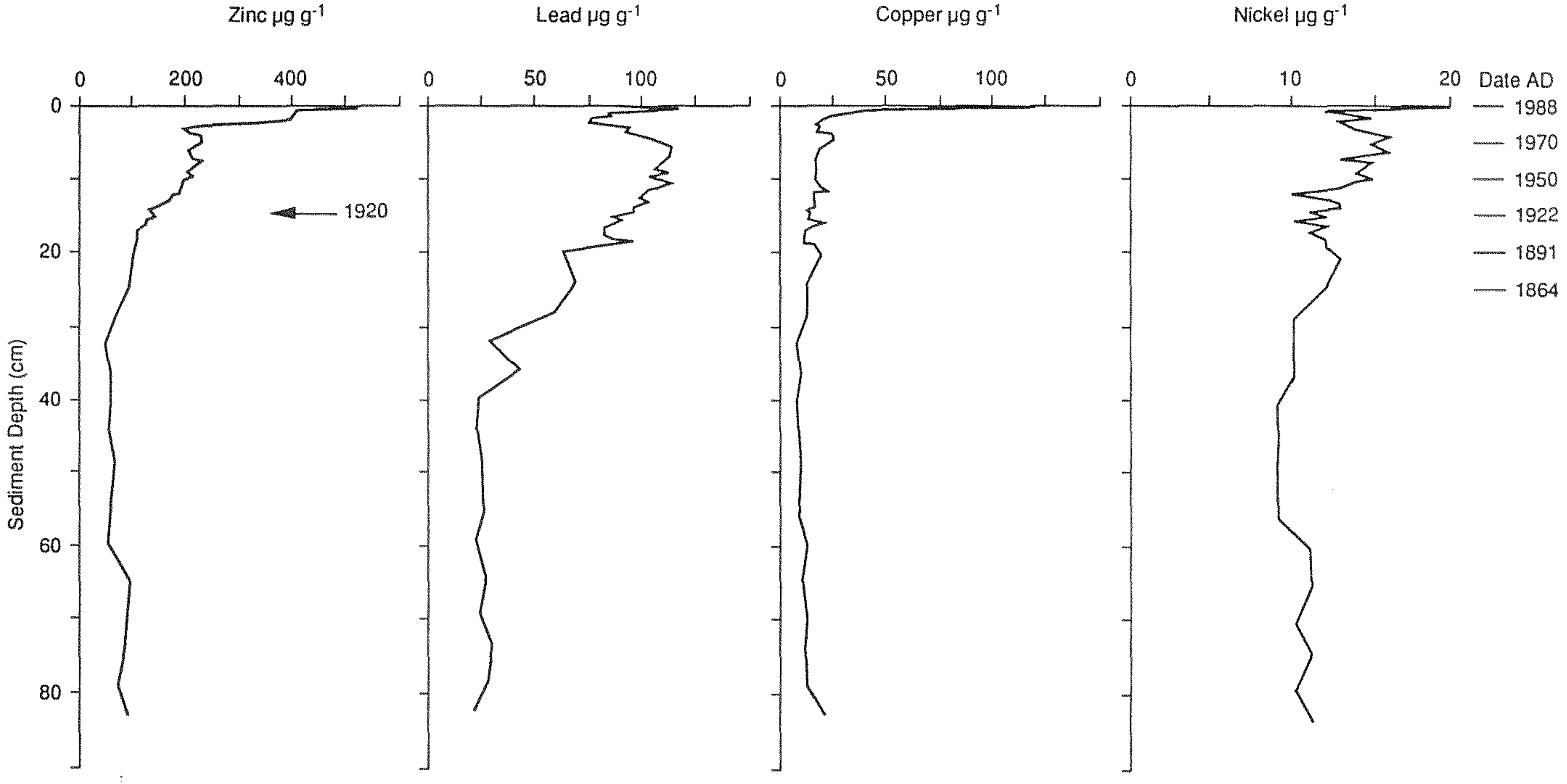
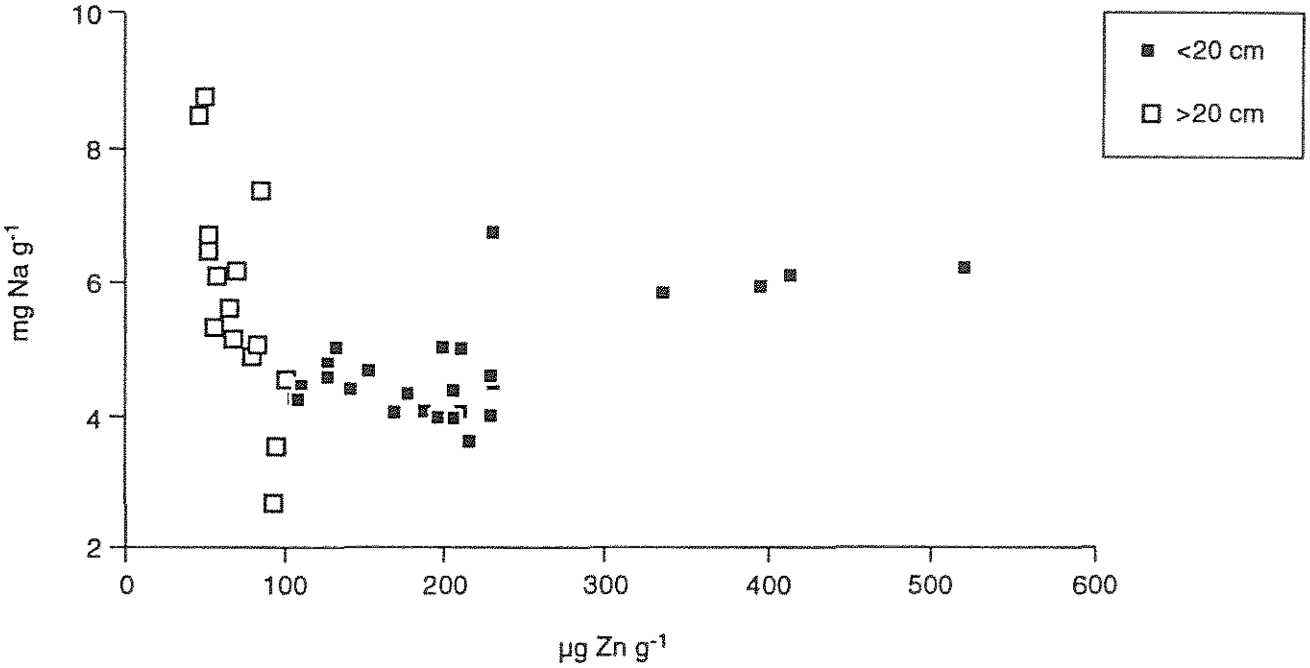


Figure 5.12 Lough Maumwee: relationship between zinc and sodium concentrations in the sediment



6 CONCLUSIONS OF THE IRISH LAKE SURVEY

The acidification status of the four lakes in western Ireland varies according to local characteristics and not to variations in acid deposition. All four lakes lie in areas mapped as highly sensitive to acid deposition as classified by Chadwick and Kuylenstierna (1990) on a consideration of geology, land-use and precipitation. This map gives no indication of relative local differences in lake sensitivity between sites in the most sensitive category.

Diatom analysis clearly shows that whilst Lough Veagh and Lough Maumwee remain unacidified, Lough Maam is moderate acidified and nearby Lough Muck is significantly acidified. The current acidification status of these four lakes is summarized in Table 6.1 which shows that all four are contaminated by atmospheric pollutants but only two have acidified. The two unacidified sites have higher lake water calcium concentrations, indicating less sensitivity to acid deposition. The trace metal contamination flux is highest for Lough Maumwee. This is because the total sediment influx is considerably higher at this site and in fact this lake is less affected than the Donegal sites.

The difference in responses to acid deposition shown by the two acidified lakes, Lough Muck and Lough Maam, is not readily explained. The lakes are in close proximity to one another and have similar water chemistries. In fact non-marine calcium concentrations are higher in the most acidified site, Lough Muck. The between-site differences must reflect either local differences in acid deposition or additional sources of increased acidity at Lough Muck from surrounding peatlands. There are two principal catchment differences between the sites, firstly, Lough Maam is located in a relatively sheltered north-west facing small corrie, whereas Lough Muck is located towards the centre of an exposed granite plateau; secondly, the catchment of the former is rocky and precipitous whereas that of the latter is fairly flat and covered with blanket peat. It is possible that at Lough Maam aspect could play a role in locally reducing the quantity of acid deposition, but trace metal contamination is similar at the two sites. The fact that Lough Muck has always been consiberably more acid than Lough Maam suggests that acid peat drainage has significantly influenced water acidity at the former site. Reconstructed pH for Lough Muck in the nineteenth century is about 5.6 and indicates that alkalinity was already close to zero when atmospheric pollution began to affect the site. In such circumstances internal buffering of any increased precipitation acidity would have been minimal and lake acidification would have proceeded rapidly.

In comparison with the worst effected areas in the British Isles, western and north-western Ireland appear to be little affected by surface water acidification. Significant acidification (>0.3 pH units) has so far only been clearly demonstrated for two lakes in the Donegal uplands. Considerable variation in catchment acid neutralising capacity as well as possible local variations in acid deposition serve to protect many lakes in the region from acidification. In the case of the Donegal sites, the Archean granite rocks are heavily faulted and contain many basic intrusions providing considerable catchment acid neutralising capacity. Such features are uncommon in the strongly acidified area of Galloway, south-west Scotland, where the granites are Devonian in age. The most sensitive Galloway lakes are considerably more acidified and are more heavily contaminated by carbonaceous particles and trace metals than the most affected Donegal lake (Lough Muck). For example, since the nineteenth century the acidity of Loch Enoch in Galloway has increased by c. 40 μeq H^+ l^{-1} compared with 16 μeq H^+ l^{-1} in Lough Muck. Unacidified lakes in regions with low levels of atmospheric pollution, as indicated by low sulphur deposition (<0.4 g m^{-2} yr^{-1}), are considerably less contaminated by atmospherically-derived trace metals and carbonaceous particles. At Loch Teanga in the Outer Hebrides sulphur deposition is <0.4 g m^{-2} yr^{-1} and the post-1980 influx of carbonaceous particles is five-fold less than in Lough Muck. The Irish sites demonstrate that regional patterns of sulphur deposition in Ireland and the United Kingdom do give some guide to the occurrence of acidified lakes but acidification of any particular site is strongly determined by local characteristics.

Table 6.1 *A summary of the impact of atmospheric pollution on four Irish lakes with differing sensitivities to acidification.* [Degree of acidification is assessed using values produced by the weighted averaging model (Birks et al. 1990). Calcium is the lowest seasalt corrected concentration in µeq l^{-1}. Carbonaceous particle concentration refers to that in the surface sediment (0.0-0.5 cm depth). Non-marine sulphate deposition is for regional data in 1987 (Jordan 1990), a site-specific estimate of S deposition from Bowman (1991) is given in parenthesis.]

	Lough Veagh	Lough Maam	Lough Muck	Lough Maumwee
Non-marine S deposition (mg m^{-2} yr^{-1})	0.5 (0.1-0.2)	0.5	0.5	0.7 (0.2-0.3)
Non-marine Ca concentration (µeq l^{-1})	57	14	26	44
Degree of acidification				
pH	0	0.4	0.9	0
H^+	0	3	16	0
Carbonaceous particle conc. (gDM^{-1})	4000	8000	5800	4600
Cumulative down-core Zn contamination since 1900 (mg m^{-2})	1160	1022	1185	2772*

* This apparently high value can be partly accounted for by the relatively high rate of sediment accumulation in this lough.

7 SUMMARY

1) Diatom analysis of cores from four western Irish lakes has demonstrated that acid deposition has exceeded the critical load necessary to cause acidification at two of these sites.

2) Trace metal and fossil fuel-derived carbonaceous particle concentration records indicate that all four lakes are contaminated by atmospheric pollutants.

3) Application of the Henriksen equation to predict lake acidification gives inconsistent results. Marked seasalt and brown water effects at the acidified sites interfere with ionic equilibria and influence water acidity so that assumptions made in the equation are invalidated.

4) The diatom acidification model (Battarbee 1989) predicts these lakes to be unacidified or, if lake water calcium concentrations are corrected for seasalt effects, in a transitional state.

5) Past land-use changes in each of the four lake catchments, as inferred from core lithostratigraphy and pollen analysis, has not significantly affected lake acidity.

6) Regionally computed sulphur deposition rates are not related to the observed degree of lake acidification. Differences in lake sensitivity to acidification must reflect local (eg. lake water calcium concentrations) rather than regional characteristics.

7) Because the Donegal granites are geologically old, faulting and basic intrusions are common, consequently many lakes in this highly sensitive area may remain unacidified.

8) Compared with strongly acidified areas in the United Kingdom, even the most acidified Irish lake (Lough Muck) has suffered an acidity increase only 25% of that recorded in Loch Enoch (Galloway, south-west Scotland). In atmospherically clean regions of the United Kingdom, such as north-west Scotland where sensitive lakes remain unacidified, both regional sulphur deposition and trace metal and carbonaceous particle contamination are less than at the Donegal sites.

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