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The suitability of a south Pennine (UK) reservoir as an archive of recent environmental change

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Abstract

The suitability of a south Pennine reservoir as an archive of recent industrial pollution (Pb deposition) and vegetation change was assessed by comparing the sediment record of Pb and pollen with a local blanket peat profile, and the modelled regional SO_2 deposition since 1840. The pollen-based record of vegetation change from the reservoir sediments was obscured by high inputs of eroded peat from the surrounding catchment. Total fluxes of Pb from the catchment into the reservoir varied between 0.05 and 2.67 kg km^{-2} year^{-1} during a 7 year period of increased peat erosion (1976–1984). The presence of concentration peaks in the Pb profile of the blanket peat may have been caused by changes in sulphide or redox chemistry within the peat profile. Large variations in influxes of Pb to the reservoir occurred during periods of increased peat erosion, suggesting the record of aerial pollution deposition has been obscured by terrestrial inputs. Extensive areas of blanket peat in the south Pennines have been subject to denudation, suggesting reservoirs in the region and other areas of high erosion and sediment flux are unsuitable for producing accurate records of the aerial deposition of pollen rain and Pb pollution. The ecological implications of highly variable fluxes of heavy metal contaminants from extensively eroded blanket bogs to ecosystems downstream are poorly understood.

Introduction

Until the middle of the 18th century, settlements surrounding the Pennine uplands of northern England, such as Manchester and Salford, did not differ markedly in size from medieval times (Douglas et al. 2002). However, the initiation of international trade dominated by cotton resulted in a huge expansion in the urban population, increased pollution emissions and demand for resources from the adjacent countryside. Large areas of the Pennine uplands are covered by blanket bog, and anthropogenic influences in the form of air pollution deposition, overgrazing by sheep, and trampling by visitors has been shown to have caused much of the ecological degradation and peat erosion on these moors since the late 18th century (Tallis 1998; Yeloff 2002).

Environmental change in the south Pennines after the beginning of the industrial revolution has been reconstructed from analyses of blanket peat stratigraphy (Tallis 1964, 1985, 1987; Yeloff 2002). However, due to the slow accumulation of blanket peat, these studies lack the sampling resolution to precisely identify the timing of relatively abrupt changes in the moorland environment within the last 200 years, and so understand their causes more clearly. Geochemical records of pollution deposition in peat also suffer problems related to post-depositional mobility and diagenesis (e.g., Jones and Hao 1993).

An area with few natural lakes, the south Pennines are dotted with numerous small artificial water supply reservoirs, the first of which was constructed in 1797 (Shotbolt et al. 2000). Reservoirs have relatively high sedimentation rates due to their riverine hydrodynamic nature and large catchment: reservoir area ratio (Van Metre and Callender 1997), giving a potential for high temporal sampling resolution. The characteristically rapid sedimentation rate has also been suggested to minimise diagenesis within the sediment column, especially in sediments with large inputs of organic matter (Callender 2000). Anderson et al. (1988) suggested that records of atmospheric air pollution in the reservoir sediments of the southern Pennines would be limited by the availability of undisturbed sediment, and regular drawdown of the water level during the summer months may be a significant factor in the reworking and disturbance of sediments. However, Shotbolt et al. (2000, 2001) demonstrated that with careful site selection, it is possible to obtain sediment cores which have been subject to minimal stratigraphic disturbance. This study aims to assess the use of a reservoir for reconstructing recent changes in vegetation composition and industrial air pollution deposition in the south Pennines by comparing the sediment record (palynology and geochemistry) of a south Pennine reservoir (March Haigh, Figure 1) with both an undisturbed profile taken from the blanket peat within its catchment, and the modelled regional record of industrial SO_2 air pollution deposition.

Site and methods

March Haigh Reservoir (1°59′ W, 53°36′ N) lies west of Huddersfield, in West Yorkshire (United Kingdom) (Figure 1). The reservoir was built as a canal-feeder in 1838, and has consequently not been subject to frequent drawdown of water level during the summer months or to the consequent reworking of sediments (Stott 1984). It therefore should contain a relatively undisturbed sediment record.

The catchment of March Haigh (Figure 1) drains into the River Colne, which runs adjacent to the Huddersfield Narrow Canal. The catchment area, excluding the reservoir area of 5.2 ha, is 263.6 ha; a catchment: reservoir area ratio of 51:1. Carboniferous Millstone Grits dominate the

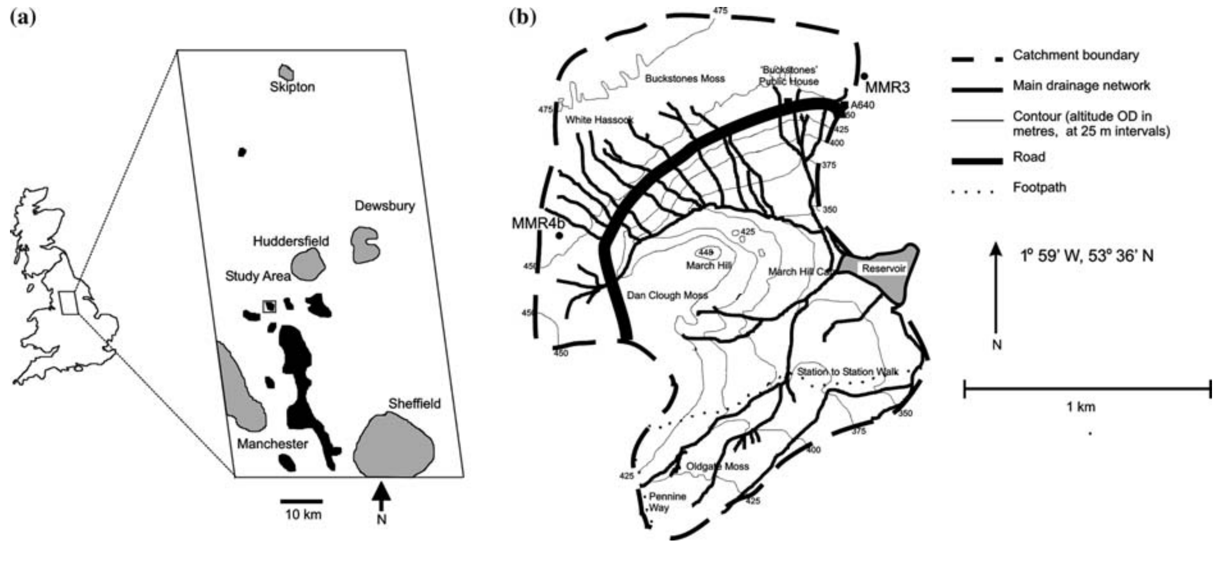

Figure 1. The study area: (a) the southern Pennies. Dark shading indicates deep peat deposits based on (Lindsay et al. 1998), urban areas are represented by grey shading; (b) the March Haigh Catchment. Latitude/longitude coordiantes refer to North corner of the March Haigh dam wall.

bedrock that underlies the catchment. The majority of the upper slopes of the catchment is covered in blanket peat, whilst the low relief area around the reservoir has podzolic soils. The upper slopes are dominated by a mixture of acid grassland vegetation types including: *Nardus*, *Molinia* and *Eriophorum* spp. grass were interspersed with patches of *Empetrum nigrum*. The steep slopes between the reservoir and Buckstones Moss are covered in a mixture of *Pteridium aquilinum* and *Cladonia* lichens. *P. aquilinum* also occurrs on the gentler slopes of March Hill Carr. *Rubus chamaemorus* and *Sphagnum recurvum* are encountered in wetter areas such as the sides of gullies.

Areas of denuded peat in the March Haigh catchment are concentrated around Buckstones Moss, which forms a small proportion of the catchment area. Studies of erosion and sediment flux within the catchment (Anderson et al. 1989; Yeloff 2002; Yeloff et al. 2005), have shown that peat erosion initiated on Buckstones Moss ca. 1959 was due to a combination of severe moorland fires and overgrazing by sheep. This resulted in increased sedimentation of eroded peat becoming evident in the reservoir by 1963. The period 1976–1984 is characterised by a high rate of peat erosion, with 2849 m^3 of peat being deposited in the reservoir during this time.

Field work at March Haigh Reservoir was aided by British Waterways conducting engineering work on the dam during the August 1999. This meant that the reservoir was almost entirely drained and that the sediments could be surveyed and cored on foot. Sediment cores were taken using a 4 cm diameter piston corer, or, where the surface of the sediment was too hard and dry for this, a 6 cm diameter Dutch Auger was used. Core MAR13 was selected for geochemical and palynological analyses as it reached the base of the sediments, was the longest (370 cm) and contained much of the variation in lithology shown in the other cores. MAR13 was sampled at ca. 3 cm intervals using a slicing plate. Activity of the radionuclides ^{210}Pb and ^{137}Cs was assayed in selected sub-samples of MAR13 at the University of Durham, using an EG and G Ortec well detector. Peak activity of ^{137}Cs occurs at ca. 287 cm depth, this peak is probably due to the 1963 fallout maximum [Full details of ^{137}Cs measurements are given in Yeloff et al. (2005)]. The ^{210}Pb age calculations using the constant rate of supply (CRS) (Appleby and Oldfield 1978) dating calculation are shown in Table 1, and the age–depth model of MAR13 is shown in Figure 2. Full details of field and laboratory work including sedimentological analyses (bulk density, loss-on-ignition) are given in Yeloff et al. (2005).

Pollen analysis was carried out by simmering in KOH and processed according to the sieving and swirling technique of Hunt (1985). Slides were traversed at 400× magnification, at least 200 pollen and spores were counted on each slide, and identified according to Moore et al. (1991). Pollen degradation (Delcourt and Delcourt 1980) and palynofacies (Combaz 1964) may be used to indicate depositional environment, and the presence of

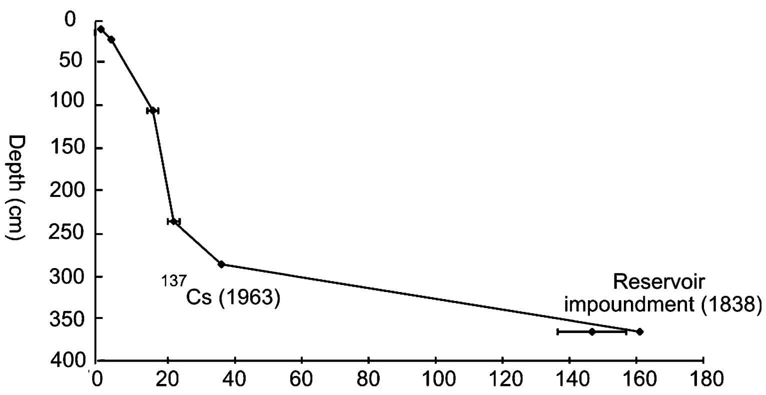

Figure 2. Age–depth plot of sediment core MAR13. Horizontal bars indicate error in ^{2}10Pb age estimates, this is the uncertainty in estimating the area of the net peak in gamma emission for ^{2}10Pb (at an energy of 45 keV). The ^{1}37Cs based date of 1963 for 287 cm depth is also indicated.

Table 1. ^{210}Pb age calculations for sediment core MAR13. Determination of mean dry bulk density is detailed in Yeloff et al. (2005).

Depth (cm)	Unsupported ^{210}Pb activity (Bq g^{-1})	Mean dry bulk density (g cm^{-3})	^{210}Pb Inventory below depth (Bq g^{-1})	Age (Year before 1999)	Error ($\pm$ year)
12.0	0.031	0.49	1.977	0	0
24.0	0.027	0.26	1.794	3	0
107.2	0.009	0.18	1.220	15	2
237.1	0.029	0.27	1.004	22	2
366.2	0.019	0.3	0.020	147	10

Total inventory 1.977

reworked organic material. During pollen counting, the state of degradation of each pollen grain or spore in the reservoir sediment samples was assessed at 400× magnification, according to the categories of Delcourt and Delcourt (1980) and Hart (1986). In a separate exercise, at least 100 palynofacies (deposited microscopic organic particles) were counted on pollen slides according to a classification derived from Combaz (1964); Batten (1982) and Hunt and Coles (1987).

The main components of industrial air pollution NO_x, NH_x, SO_x and O_3 are volatile compounds, and when deposited in peat or lake sediments they may be chemically reduced, the efficiency of reduction will depend on environmental conditions. Ozone will remain a gas, and no residue will remain in sediments. In studies of past air pollution deposition on bogs, alternative indicators have been used which reflect air pollution deposition both spatially and temporally. These have been used as indicators, as they are considered environmentally robust (i.e., immobile in the sediment/peat profile, and not susceptible to diagenesis) after deposition.

Pb and mineral magnetic measurements have been used as indications of air pollution deposition in this study. Pb is one of the trace elements enriched in coal, and lead has been used as a petrol additive from the 1920s to the 1990s in Britain (Gilbertson et al. 1997). A sizeable proportion of fly ash spherules resulting from the high temperature combustion of fossil fuels in facilities such as power stations are composed of the mineral magnetite. This can be identified in peat profiles using mineral magnetic parameters described in Thompson et al. (1980), and Thompson and Oldfield (1986). Studies such as Richardson (1986) have shown that clear pollution trends can be derived from the mineral magnetic record of peat. Additionally, the underlying lithology of Millstone Grit is of a magnetically impoverished nature (Hutchinson 1995), suggesting that any variations in mass-specific magnetic susceptibility (χ) in the sediments of March Haigh may be reflecting the content of air pollution-derived trace metal contaminants.

To estimate Pb content, approximately 0.5 g of sediment was taken from selected sub-samples at approximately 10 cm intervals of MAR13. They were then ground using a pestle and mortar, weighed to the nearest 0.001 g and digested in concentrated HNO_3 using a microwave oven. The resulting digestion was filtered and then diluted with ultra-pure water. The concentration of Pb in each digestion was determined by atomic-absorption flame photometry, with use of a graphite furnace to improve measurement sensitivity. All glassware was acid-washed prior to use, to ensure minimal contamination. Replicates of subsamples enabled measurement error to be estimated as 7%.

Air-dried samples were gently disaggregated using a pestle and mortar, sieved to select the < 2 mm diameter particle size fraction, and packed into 10 cm^3 plastic pots. Mass-specific magnetic susceptibility was measured at low frequency (χ_{lf}) using a Bartington MS2 meter calibrated with a $MnCO_3$ standard (Gale and Hoare 1991).

Peat samples were taken from the catchment of March Haigh during October, 2000. Peat samples were taken by inserting a plastic monolith box of 50 cm depth into an exposed peat face, or by digging a small pit and taking a monolith from the pit face. Monolith samples were described, and placed in a domestic chest freezer at −40 °C until frozen solid. An electric saw was used to sub-sample one monolith from each site (monolith A), at ca. 5 mm intervals. The length of each sub-sample was corrected for expansion upon freezing. The Pb content of air dry sub-samples at 1 cm intervals of the vertical peat profile was measured by the method described above. Correlation (based on peat humification) with a ^{210}Pb (CRS) dated profile from site MMR 4b (Figure 1b) enabled age and accumulation rates to be calculated. The rise in air pollution deposition corresponding with the end of the first phase of the industrial revolution ca. 1840 (Thompson et al. 1980; Thompson and Oldfield 1986), has been identified using volume magnetic susceptibility measurements (Figure 3a). Volume magnetic susceptibility κ was measured by pressing a cleaned field probe against the face of the air dried sample and recording magnetic susceptibility using the attached Bartington MS2 meter. The value of κ was calculated by comparison with the average of two air readings as described in Dearing (1994).The age–depth model of the core from site MMR3 (Figure 1b) is shown in Figure 3b. Due to a hiatus in the peat profile of MMR4b, the accuracy of dates between 1840 and 1940 should be viewed with some caution. Pollen counting and measurement of Pb content of air dry sub-samples at 1 cm intervals of the vertical peat profile of MMR3 was conducted by the methods described above. Full details of field and laboratory work on the profile MMR3 including magnetic susceptibility measurements and pollen analyses are given in Yeloff (2002).

Results

March Haigh Reservoir sediments

Due to the relatively low sedimentation rate prior to the early 1960's, the early sediment deposition to March Haigh Reservoir (Figure 4) is represented

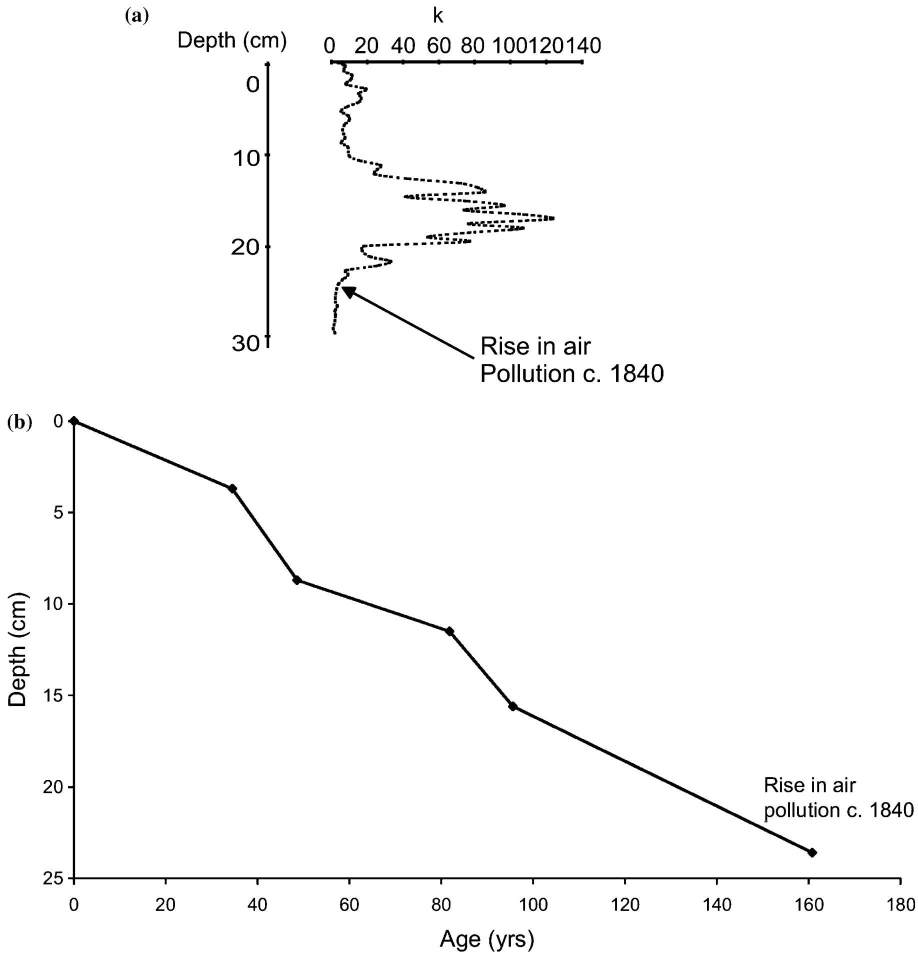

Figure 3. (a) volume magnetic susceptibility (κ) measurements of peat profile MMR, showing position of the rise in industrial air pollution deposition at 23.4 cm; (b) Age–depth plot of peat profile MMR3.

by a coarser sampling time resolution. Noticeable variation in the pollen record occurs during the period 1976–1984, as represented by increased values of the moorland pollen type Ericaceae and reductions in Poaceae. Proportions of corroded and degraded pollen and spores (expressed as a percentage of the pollen sum) also rise during this time. The palynofacies categories fungal debris, pollen and spores (expressed as a percentage of the palynofacies sum) also increase. This has been shown to be a period of increased peat erosion in the catchment (Yeloff et al. 2005), and correlates with the high sedimentation rate.

Pb influx to the reservoir sediments (Figure 4) varied between 1 and 222 μg cm^{-3} year^{-1}. Pb concentrations varied between 2 and 264 μg g^{-1}. This is slightly higher than maximum loadings of 200 μg g^{-1} in the sediments of two other southern Pennine reservoirs, Howden and Watersheddles (Anderson et al. 1988; Shotbolt et al. 2000). A small peak is noticeable during the period 1881–1895. Pb influx is highly variable after 1975, with two short-duration maxima noticeable in the late 1970's – early 1980's. This appears to be confirmed by the records of Pb and χ in the March Haigh sediments (Figure 4). The exception to this is the peak in χ around 1978, which then decreases during the period of highest Pb influx. The size fraction measured may be the reason for the inconsistency. χ is a measure of the magnetic properties of the <2 mm fraction of the sediments. After 1978, there was an influx of sediment almost entirely composed of eroded peat (Yeloff et al. 2005), and little fine particulate inorganic material. Reworked peat, mostly composed of plant debris >2 mm diameter would produce a very low magnetic susceptibility signal. In contrast, Pb measurements are taken from the total sample including the larger plant debris fraction. Pb in air pollution deposition binds easily to plant material (e.g., Yang et al. 2002). High Pb values in the sediments would result from the inwash of plant debris eroded from the air pollution-contaminated surface layers of the blanket bog.

Catchment peat

Figure 5 summarises the pollen analysis of MMR3, which has already been described in detail

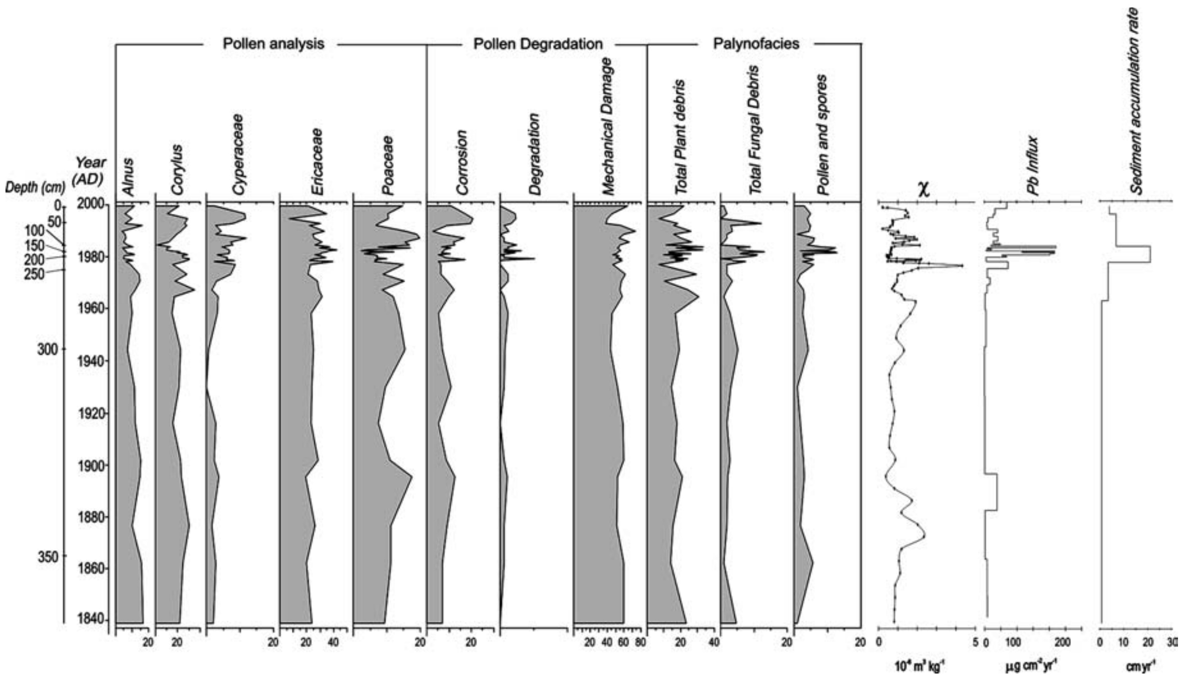

Figure 4. March Haigh Reservoir sediments (MAR13): selected pollen types (expressed as % of total land pollen); pollen degradation (expressed as % of total land pollen), palynofacies (expressed as % of total palynofacies); mass-specific magnetic susceptibility measurements (χ); Pb influx; and sediment accumulation rate.

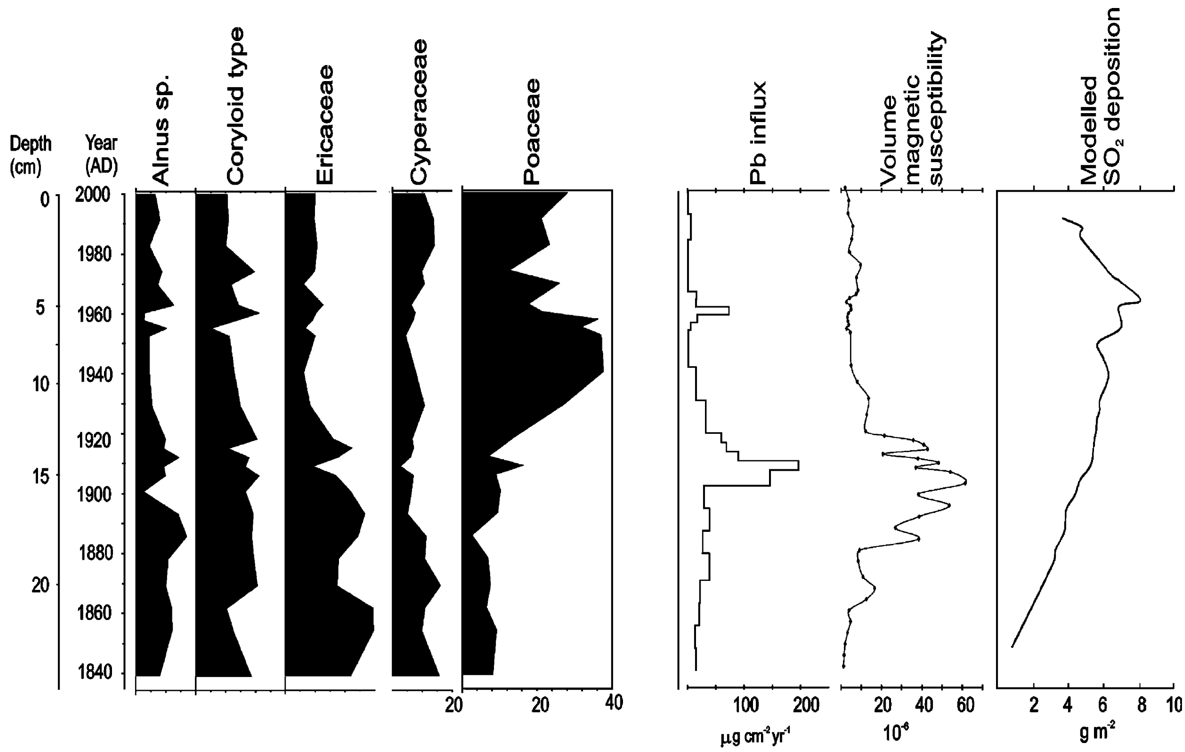

Figure 5. Blanket peat profile from site MMR3: selected pollen types (expressed as % of total land pollen); volume magnetic susceptibility measurements; Pb influx; modelled regional SO_2 deposition (Skeffington et al. 1997).

by Yeloff et al. (2002). With the exception of the air pollution 'takeoff' ca. 1840, the precision of dates prior to 1940 should be viewed with caution, due to the uncertainty in the ^{210}Pb chronology of core MMR4b. Prior to ca. 1918, the moorland vegetation is dominated by Ericaceae, which is then replaced by grass. A decrease in abundance of local moorland vegetation types ca. 1959 suggests the onset of vegetation removal and the initiation of peat erosion.

Pb influx to the blanket peat (Figure 5) varied between 0 and 197 $\mu g\ cm^{-3}\ year^{-1}$. Pb concentrations varied between 6 and 542 $\mu g\ g^{-1}$. This is comparable with three other peat sites in the south Pennines (Buxton, Featherbed Moss, Ringinglow Bog) sampled by Livett et al. (1979), which had maximum concentrations of 481, 534 and 688 $\mu g\ g^{-1}$, respectively. From 1840 onwards, Pb influx gradually increases. A sharp rise occurs in the 1900's, which peaks around 1910. A second peak occurs around 1960. Magnetic susceptibility may also be used to indicate variations in air pollution deposition on blanket peats (Thompson et al. 1980). There is a similarity between the pollution records of magnetic susceptibility and Pb, with the exception of the ca. 1960 peak in Pb influx, which has not been recorded in the magnetic susceptibility profile.

Skeffington et al. (1997) combined the model of Mylona (1993), and English coal consumption figures to estimate emissions of SO_2 over the UK, and deposition on the south Pennine region (Figure 5). Modelled pollution emission and deposition shows steadily rising levels from the mid-19th century, a peak in the late 1960's, and then a decline till 1991.

Discussion

Changes in vegetation composition

In comparison to the record from the catchment peat, the pollen–vegetation record of the March Haigh Reservoir sediments has clearly been disturbed, and variations are related to changes in the flux of sediments into the reservoir. This is especially noticeable during the period 1976–1984, when increased erosion resulted in a transfer of a large amount of eroded organic material from the surrounding catchment to the reservoir. The high proportion of pollen and spores transported from the eroding blanket bog and re-deposited in the reservoir sediments (Yeloff and Hunt 2005) prevents the reservoir record from being a reliable archive of vegetation change in the surrounding area. The contamination of the lake sediment record in recent sediments by re-deposited pollen has also been noticed in natural lakes in the British Isles (Pennington et al. 1972; Dearing et al. 1981; Bradshaw and McGee 1988; Huang and O'Connell 2000).

Table 2. The total influx of Pb to the March Haigh sediments during 1976–1984: Sediment characteristics and Pb measurements from Yeloff et al. (2005); bulk density estimate based on measurements of 45 sub-samples; Pb concentrations based on measurements of 14 sub-samples. The trap efficiency of March Haigh Reservoir was estimated at 99%, based on the equation of Brown (1944).

Volume of sediment layer (m^3)	2849
Average bulk density (g cm^{-3})	0.18 ± 0.15
Mean Pb concentration ($\mu g\ g^{-1}$)	29.7
Minimum Pb concentration ($\mu g\ g^{-1}$)	1.8
Maximum Pb concentration ($\mu g\ g^{-1}$)	96.0
Area of catchment (km^2)	2.636
Total influx of Pb into March Haigh Reservoir 1976–1984 (kg km^{-2} $year^{-1}$):	
Mean	0.82
Minimum	0.05
Maximum	2.67

The record of air pollution deposition

When compared with modelled estimates of regional SO_2 deposition, the record of air pollution deposition (Pb influx and magnetic susceptibility) on to the March Haigh blanket peat differs, suggesting that the peat record may not be recording air pollution deposition effectively. The Pb influx record is composed of two distinguishable peaks, and these may have been caused by changes in sulphide and redox chemistry controlling the movement of trace metals within the peat profile (Clymo et al. 1990; Yang et al. 2001). However, both the record of regional pollution deposition and the blanket peat profile do not exhibit the high variation evident in the reservoir sediments. The concentration of contaminants in the March Haigh sediments corresponds clearly with changes in erosion rates within the catchment.

The March Haigh sediments, despite being 'internally undisturbed', i.e., not subject to summer drawdown and other processes, may be recording the flux of eroded material through the aquatic system to the point of obscuring any evidence of air deposition. The blanket peats of the south Pennines cover a large area. The area of moorland in the Peak District National Park, covering a major part of the south Pennines has been estimated at 52,252 ha (Phillips et al. 1981), with approximately 38% of the peat blanket being severely eroded (Anderson and Tallis 1981). The results of this study suggest that many of the reservoirs in the region, and in other areas of high erosion and sediment flux may be unsuitable as archives for producing accurate records of the aerial deposition of pollen rain and pollution.

Sediment fluxes to lakes and reservoirs within the eroded blanket peat catchments of the southern Pennines are governed by: (1) variations in runoff from storm events, dominated by overland and near-surface flow processes (Gardiner 1983; Labadz et al. 1991); (2) the availability of sediment from the eroding bog surfaces. The removal of the contaminated surface layers of peat, and redeposition downstream by the interaction of these two processes, has produced an erratic and highly varied record of fluxes of contaminants within the March Haigh catchment. The record of Pb deposition in the reservoir sediments during the period 1976–1984 (Figure 4) suggests that large releases of trace metals from the blanket bog surface can occur in a short space of time. Estimates of the total flux of Pb from the catchment into the reservoir sediments vary between 0.05 and 2.67 kg km^{-2} $year^{-1}$ for this period (Table 2). The accuracy of these estimates should be viewed with caution, as they are based on Pb measurements from one sediment core, and do not take into account spatial variations in Pb concentrations within the reservoir due to sediment focusing and other processes. The majority of blanket peat erosion in the March Haigh catchment occurring since 1948 has been on Buckstones Moss in the north of the catchment (Anderson et al. 1989), which occupies a minor proportion of the total catchment area. This suggests that Pb fluxes will be subject to significant spatial variation within the catchment.

Increased fluxes of trace metal pollution during storm events have also been observed in a Canadian peatland (Branfireun et al. 1996), and this study suggests that blanket peat areas subject to erosion represent a highly variable source of contamination for aquatic environments downstream. As much as 350,000 ha of blanket bogs in the British Isles may be in an eroded state (Tallis 1998). However, the ecological implications of highly variable fluxes of heavy metal contaminants from extensively eroded blanket bogs to ecosystems downstream are poorly understood.

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