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Water Quality Investigation of Loweswater, Cumbria

Final Report to the Environment Agency

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Executive Summary

1. This is the final report to the Environment Agency: Water Quality Investigation of Loweswater, Cumbria. The project employs palaeolimnological techniques to evaluate the extent of eutrophication at Loweswater.

2. The existing published and unpublished literature on Loweswater was reviewed. Certain aspects of the water chemistry indicate mesotrophic conditions whilst others suggest slightly eutrophic ones. The phytoplankton survey data indicate that Loweswater is a mesotrophic to mildly eutrophic lake with abundant algal biomass and relatively abundant blue-green algal populations. The current diatom flora is typical of mesotrophic lakes. However, the diverse macrophyte flora indicates that a range of chemical conditions exist in Loweswater, supporting species typical of oligotrophic, mesotrophic and eutrophic waters. Furthermore, the bottom fauna and the leeches and oligochaetes are indicative of a productive lake, whilst the dominant corixid water bugs species are more typical of an unproductive waterbody. It is, therefore, difficult to define the trophic status of Loweswater consistently. The lake is naturally more fertile than many others in the English Lake District, because of its relatively lowland catchment with well developed soils.

3. From the available lake water chemistry data, there is perhaps some evidence of slight enrichment during the last century. Nutrient data are sporadic but nitrate concentrations appear to have risen, probably associated with diffuse inputs from agricultural intensification. Phosphorus concentrations appear to remain generally low with occasional high values of TP in recent years which could be cause for concern, although data are scarce and need augmenting. Available data for the lake inflows indicate that the main inflow of Dub Beck provides the major source of nutrient loading to the lake. The literature review supported the need for further investigation into the water quality and ecology of Loweswater.

4. This report describes the lithostratigrapy, radiometric dating, geochemistry and fossil diatom assemblages in twenty-five selected levels of a sediment core from the deep basin of the lake. Diatom transfer functions are applied to the diatom data to generate quantitative reconstructions of total phosphorus (TP) concentrations for the lake, following taxonomic harmonization between the training sets and core species data. The TP reconstructions are calculated using a Northwest European calibration set of 152 lakes (Bennion *et al*., 1996) and an unpublished European calibration set of 46 large lakes.

5. The findings of this study suggest that Loweswater has experienced slight nutrient enrichment over the last century. The diatom record shows that Loweswater was less productive in the past with diatom assemblages indicative of relatively nutrient-poor waters and diatom-inferred TP (DI-TP) concentrations of around 10 μg TP l^{-1} from approximately 1300 until 1850 AD. There was a clear shift in the diatom community at around 1850 AD marked by a decline in the small, oligotrophic *Cyclotella* taxa and *Achnanthes minutissima*, and an increase in taxa typically associated with mesotrophic conditions. A further expansion of taxa such as *Tabellaria flocculosa*, *Cyclotella radiosa* and *Fragilaria crotonensis* since the 1950s indicates recent ecological change, most likely associated with nutrient enrichment. The DI-TP results indicate a 50% increase in TP concentrations since around 1850 AD with values increasing from around 10 to 15 μg TP l^{-1} over this period. These DI-TP results, however, are based on a prototype version of a training set that has yet to be expanded and fully validated, and currently includes only 46 lakes. A

refined model should be available later in 2000. Nevertheless, the recent DI-TP concentrations are in good agreement with current measured TP concentrations for Loweswater and therefore the values are likely to be in approximately the correct range.

6. The geochemical record also provides evidence of enrichment, most notably by the sharp increases in Si/Ti and P above 20 cm (around 1750 AD) which are most easily explained by increased productivity of the lake. The start of the enrichment, however, appears to be somewhat earlier than that suggested by the diatom record. Nevertheless, in accordance with the diatom data the most marked changes in the geochemistry profiles occurred in the uppermost part of the core, representing approximately the last fifty years. Recent changes in both the diatom and geochemical records are coincident with the timing of the increase in sediment accumulation rates, which have increased steadily since the 1950s.

7. Additionally, the geochemical record provided strong evidence for heavy metal pollution which is most likely associated with lead and zinc mining in the area, possibly dating from one thousand years ago and most notably since around 1750 AD.

8. In conclusion, the palaeolimnological study clearly demonstrates that Loweswater is not currently in a pristine state. The data indicate that changes in water quality have occurred since around 1850 AD (perhaps a little earlier) which most likely marks the impact of the Industrial and Agricultural Revolutions upon land-use in the district. This was followed by further pronounced changes since around 1950 which are indicative of increased productivity.

9. The exact causes of the inferred enrichment are not clear and there is limited documentary data on land use changes and historical events in the catchment to aid interpretation of the findings. Potential nutrient sources include bird populations, changes in the fish community, agriculture and sewage effluent. Of these, only the latter two are likely to be significant. There is evidence from the published water chemistry data of a long term increase in nitrate concentrations in Loweswater from the late 1920s with more recent chemical changes since the 1950s including a significant increase in both potassium and nitrate. These nutrients are present in fertilisers and the available data, therefore, lend support to the hypothesis that agricultural activity may be a source of nutrients to the lake. Information on agricultural statistics would be required to interpret the links between changes in land-cover/management and eutrophication with any confidence.

10. The contribution of sewage inputs to the nutrient load entering the lake has not been quantified. There are no sewage treatment works in the Loweswater catchment and, therefore, any effluent entering the lake will be derived from diffuse septic tank sources. Given the low population density in the catchment, it is unlikely that sewage effluent is the main cause of the increased nutrient levels in the lake although its role as a contributory factor cannot be ruled out.

11. The role of weather was also considered. Whilst the inferred enrichment is likely to be an important contributing factor to the observed recent trend in blue-green algal bloom incidents, the occurrence of blooms is also likely to be associated with low flushing and warm, calm weather conditions. Such conditions have been experienced more frequently in recent years (late 1980s and 1990s).

12. The study highlights the need for further monitoring of the water quality of Loweswater. A series of recommendations are outlined including water quality monitoring, construction of a nutrient budget, modelling of phosphorus pathways, phytoplankton monitoring, macrophyte surveys and trophic interaction studies. Finally, a number of management options are given with regard to control of nutrient inputs from agriculture and sewage. However, the necessity for their implementation will depend on the outcome of subsequent monitoring and identification of the major nutrient sources to the lake.

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1.1 Study rationale

The recently published West Cumbria LEAP (Environment Agency, 1999) identifies the need for further information on the algal blooms in Loweswater as one of its specific issues for the area. Currently, there are only limited data on the water quality of Loweswater. Nevertheless, these data reveal several features that are not compatible with the local geology and catchment. The algal blooms are now occurring throughout the year and not only in the summer months and, therefore, there is growing concern over the water quality of the lake. Much of the evidence, however, is anecdotal and the true water quality status is unknown. Of the eight major lakes in the West Cumbrian LEAP, only Loweswater and Thirlmere are not designated Sites of Special Scientific Interest and this may explain why Loweswater has received relatively little attention until now.

1.2 Objectives

The objectives of the current study were to:

i) Review the published, and available unpublished, literature on the water quality and catchment of Loweswater.
ii) To take a sediment core from the lake, and analyse for evidence of eutrophication.
iii) To assess the severity of eutrophication over recent years, identify the dominant causes and suggest management options for resolving the water quality issue at Loweswater.

2. LITERATURE REVIEW

2.1 Eutrophication and blue-green algal blooms

2.1.1 Defining eutrophication, its causes and consequences

Eutrophication is the enrichment of waters by plant nutrients. The definition adopted by the Environment Agency (1998a) is:

“The enrichment of waters by inorganic plant nutrients which results in the stimulation of an array of symptomatic changes. These include the increased production of algae and/or other aquatic plants, affecting the quality of the water and disturbing the balance of organisms present within it. Such changes may be undesirable and interfere with water uses.”

The nutrient levels in a given lake are determined by the nutrient supply from its catchment, which is a function of the geology. Therefore, a large range in the natural nutrient status of waters can be found. Lakes can enrich naturally over long timescales as nutrient-rich material from the catchment slowly accumulates in the system. Since the 1970s, however, the word eutrophication has been increasingly used in the sense of artificial addition of plant nutrients to waters, that is human-induced enrichment or “cultural eutrophication” as it is often referred to.

The nutrient most commonly considered to be the major cause of eutrophication in temperate freshwaters is phosphorus (P), although nitrogen (N) may also be important in some cases. This is because P is usually the key limiting nutrient for growth, meaning that its demand often exceeds its supply and thus it tends to control productivity of the system. Increasing amounts of these nutrients have entered lakes in recent decades from agriculture, forestry, domestic sewage, industrial effluent and urban storm water. The nutrient sources can be divided into two major types: i) point sources in which the nature of the pollution source can be easily identified, for example waste water treatment and raw sewage discharge; and ii) diffuse sources in which the nature and exact source of the pollution are harder to recognise, for example run-off from agricultural land (fertilisers and animal slurry) and roads.

When nutrients are in short supply, algal growth is restricted but as concentrations increase, algal and other plant growth is encouraged. Early indications of eutrophication include the growth of blanketing algae at the edges of lakes with suspended microscopic algae (phytoplankton) in the open water (Moss *et al.,* 1996). This enhanced plant growth causes fluctuations in dissolved oxygen levels between day and night with de-oxygenation of the deeper waters. At the extreme, this can result in the death of invertebrates and fish and a subsequent change in the fish community. Algal decay can further reduce the oxygen content of the water. Eutrophication reduces the biological diversity of the aquatic ecosystem by increasing the dominance of nutrient tolerant plant and algal species. For example, nutrient-rich lakes can become choked with higher plants such as Canadian pondweed (*Elodea canadensis*) or by macro-algae (e.g. *Cladophora*). The more sensitive plant species with higher conservation value are often out-competed.

In many nutrient-rich shallow lakes, eutrophication has resulted in the complete loss of submerged macrophytes. The most famous examples of these in the UK are the Norfolk Broads (e.g. Moss *et al.*, 1996).These systems became dominated by algae, largely owing initially to P enrichment from sewage effluent, so that sufficient light was no longer available for the survival of submerged plants. In the absence of macrophytes, it has proved difficult to restore these lakes back to their previous clearwater, macrophyte-dominated states because the macrophytes themselves play an important role in maintaining a healthy, diverse system. For instance, plants act as refuges for large water fleas (zooplankton) that graze on the algae but once these refuges are lost, fish can easily remove the zooplankton so that grazing pressure on the algae is minimal (Moss *et al*., 1996). Furthermore, plants are important in stabilising lake sediments so that in their absence, sediment disturbance by wind is more likely and the lake water becomes turbid with resuspended particles, and thus light penetration is further reduced. There are a series of mechanisms which stabilise aquatic plant-dominated and algal-dominated communities in shallow lakes and when these are overcome the system can switch from one of these "alternative stable states" to the other (Moss *et al*., 1996). Biomanipulation is often required to restore such lakes.

A conceptual diagram of the eutrophication process is shown in Figure 1.

Figure 1 Conceptual diagram of the eutrophication process
Taken from Environment Agency (1998a)

2.1.2 Blue-green algae

A further consequence of eutrophication is the occurrence of blue-green algal blooms, often referred to as cyanobacteria. Although such blooms are known to have occurred for at least 3500 years (National Rivers Authority, 1990), in recent years there has been a dramatic increase in their incidence. The growth and development of blue-green algal blooms depend on a variety of interacting physical, chemical and biological conditions (Reynolds, 1991). Their abundance is strongly correlated with nutrient availability and, therefore, the increase in nutrient loading to lakes, combined with milder winters and warmer summers in recent decades is thought to have been responsible for their increased frequency (Reynolds, 1991). In a survey of 3,000 freshwater bodies (predominantly standing waters) during 1989–1997, approximately two thirds had blue-green algae as their dominant species (Environment Agency, 1998a). Many common blue-green algae produce toxins which can seriously affect domestic animal and human health as well as wildlife. In 1989, 68% of blooms tested in the UK had produced toxins (National Rivers Authority, 1990). In Britain the problem of toxic algal blooms was highlighted by the deaths of 20 sheep and 15 dogs in Rutland Water in the summer of 1989. As a result of these events, the National Rivers Authority (now the Environment Agency), set up a 'Toxic Algae Task Group' to act as an advisory body and to increase public awareness of the risks of blue-green algae (National Rivers Authority, 1990). The Environment Agency has since established a monitoring programme for affected sites and has identified warning thresholds and guidelines based on the probability of toxicity (Environment Agency, 1998a). A series of site-specific 'Action Plans' have been developed, which involve an holistic and detailed study of lake physical, biological, chemical and economic characteristics.

2.1.3 Eutrophication trends

The problem of eutrophication in the UK has increased significantly since the Second World War, becoming a widespread and significant environmental problem in recent decades (Carvalho & Moss, 1995). A study of 76 lakes, in which current water quality was compared with that in the 1930s inferred from export-coefficient models, indicated that many lakes had become enriched (Environment Agency, 1998b). The conservation value of many of Britain's pristine lakes is thought to be affected by eutrophication. This was revealed in a recent survey in which 102 Special Sites of Scientific Interest (SSSI) were examined to determine their nutrient status, with 84% of these displaying symptoms of eutrophication (Carvalho & Moss, 1995).

With regard to point sources of P, there was evidence of a reduction in loadings from sewage treatment works during the 1990s, and reductions in river phosphate concentrations have also been observed since this time (Environment Agency, 1998a). However, despite declines in both fertiliser use and cattle numbers in England and Wales in recent decades, the contribution of agricultural P losses to the eutrophication of waters is of growing concern. Modern farming methods cause excess P to accumulate in soils, in turn leading to transportation of P from diffuse agricultural sources to waterbodies. Soils could continue to contribute to elevated P concentrations in receiving waters for many years even if farming practices were changed (Environment Agency, 1998a). Nitrate enrichment from agricultural sources is also of concern in the UK with the use of N-based fertilisers increasing by around 50% between 1970 and the early 1990s (Environment Agency, 1998a). Approximately 70% of the N input to inland surface waters originates from diffuse pollution (Environment Agency, 1998a). A further source of nutrients which can contribute to the maintenance of high lake nutrient concentrations is the lake sediments. Nutrients can build up in the sediments deposited in the bottom of lakes and thereby act as an internal nutrient load. Under anoxic conditions in summer, P can be released back into the water column and become available for algal growth. Release of P from sediments has resulted in a delayed recovery in many lakes despite reductions in external nutrient loads from the catchment, for example at Esthwaite Water, Cumbria (Talling & Heaney, 1988).

2.1.4 Trophic classification of lakes

Lake trophic status as a criterion for lake classification was first attempted by Thienemann (1909, 1915) and Naumann (1917, 1932) who recognised two types of lakes: nutrient-poor, "oligotrophic" lakes in Alpine regions and nutrient-rich, "eutrophic" lakes in lowland regions. Other studies identified, however, that lakes demonstrated a continuous range of variation in their characteristics and that they could not be easily categorised (e.g. Pearsall, 1921). Despite these findings, waters have long been classified into defined, though arbitrary, trophic states ranging from ultra-oligotrophic through oligotrophic, mesotrophic and eutrophic to hypertrophic. The most widely used trophic classification scheme is based on the work of the Organisation for Economic and Co-operative Development (OECD, 1982). This scheme aimed to relate classical trophic terminology (i.e. oligotrophic, mesotrophic, eutrophic) to specific levels of water quality variables to provide a widely applicable scheme. This system still acts as the standard lake trophic classification scheme today. The boundaries for the five classes of waters identified in the scheme are shown in Table 1. Initially it was based on these fixed boundaries but was later adapted to provide an open boundary scheme, incorporating the probability of the lake falling into each category.

Table 1 OECD trophic classification scheme (OECD, 1982)

Trophic category	*Mean Total Phosphorus ($\mu g\ l^{-1}$)*	*Mean Chlorophyll a ($mg\ m^{-3}$)*	*Maximum Chlorophyll a ($mg\ m^{-3}$)*	*Mean Secchi depth (m)*	*Minimum Secchi depth (m)*
Ultra-oligotrophic	< 4	< 1	<2.5	> 12	> 6
Oligotrophic	4-10	1-2.5	2.5-8	12-6	6-3
Mesotrophic	10-35	2.5-8	8-25	6-3	3-1.5
Eutrophic	35-100	8-25	25-75	3-1.5	1.5-0.7
Hypertrophic	> 100	> 25	>75	< 1.5	< 0.7

It has also long been recognised that water chemistry is extremely important in the determination of the aquatic macrophyte species composition of a waterbody and thus aquatic macrophytes have also been used to classify lake systems. Perhaps the most sophisticated of such schemes, and the one employed widely in the UK, is that developed by Palmer *et al*. (1992). This scheme identifies ten standing water types on the basis of floating and submerged macrophyte species. The approach is based on indicator species analysis of over one thousand fresh and brackish waters in the British Isles. The ten classes were related to alkalinity, pH, conductivity, substrate and to a lesser extent location, to produce a physio-chemical category for each type and can also be used to provide a Trophic Ranking Score (TRS) (Palmer *et al*., 1992).

The TRS scheme demonstrates the role of nutrients as well as pH in explaining the variation in aquatic macrophyte species composition. However, this work also demonstrated that these properties co-vary markedly with each other and with other physical properties of lakes such as substrate type and geographical location. For example, a classic, oligotrophic lake type gives rise to a particular aquatic macrophyte assemblage where the emergent flora is severely restricted by the coarse nature of the shoreline substrates and directly by wind and sometimes ice. The submerged flora in such lakes is adapted to low nutrient availability and low temperatures, and due to the absence of phytoplankton is able to penetrate to considerable depths. In this case, plants with an isoetid growth-form such as *Isoetes lacustris* and *Lobelia dortmana* tend to dominate. Categories of lake described as mesotrophic, or indeed eutrophic, however, contain a much broader range of site type characteristics and hence potential biological assemblages. Although some consistent macrophyte species can be identified within these trophic categories, the exact species composition will be dependent on factors such as basin morphometry, substrate, extent of shelter and influence of phytoplankton blooms and, therefore, no two lakes will be alike.

2.1.5 Managing eutrophication in the UK

Eutrophication has had a high profile as a water quality issue since the late-1980s. In a recent report on the state of the freshwater environment in England and Wales, nutrient enrichment was identified as one of ten priority issues which must be addressed to provide a sustainable balance between societies needs and the health of freshwater ecosystems (Environment Agency, 1998c). This balance is hoped to be achieved through a partnership and catchment based approach within a wider national framework for management. With this in mind, the Environment Agency has produced a proposed management strategy for aquatic eutrophication in England and Wales (Environment Agency, 1998a). This emphasises that co-operation and consultation by all parties, including the major regulatory bodies, stakeholders and local land users, is necessary to successfully reduce eutrophication and its effects.

2.2 The regional context: the English Lake District

There has been growing concern in recent decades about environmental impacts upon the English Lake District, most notably eutrophication of surface waters via sewage effluent, introduction of P-rich detergents, increasing tourism and recreation on the lakes, application of artificial fertilisers for agriculture and forestry in the catchments, and urban storm run-off from roads (e.g. Carvalho & Moss, 1995). The decline of rare species of high conservation status, such as arctic charr and vendace, has been of particular concern. Consequently, there is increasing awareness amongst agencies concerned with lake management and conservation that lake classification and management strategies should not be based solely on the current status of the waters but should also incorporate the concept of degree of change.

For a number of the lakes there exist long term monitoring datasets (generated by the Institute of Freshwater Ecology, Windermere since the 1940s) of physico-chemical variables and phytoplankton populations, notably Windermere, Esthwaite Water, Blelham Tarn and Grasmere (e.g. Sutcliffe *et al*., 1982; Talling & Heaney, 1988; George *et al*., 1990; Talling, 1993); a summary is given in Elliott (1990). These data provide a rare opportunity to assess the degree of environmental change since the start of data collection. For other lakes in the region, however, historical data are limited. Kadiri & Reynolds (1993) compared phytoplankton data collected in 1978 and 1984 with those collected by Gorham *et al*. (1974) in the 1950s-1960s to explore changes in lake water quality over a period of two to three decades for a group of twenty lakes. In the absence of historical data, one way to assess the degree of change in lakes over long timescales (decades, centuries) is to use the sediment record, a science known as palaeolimnology (Smol, 1992). A number of palaeolimnological studies have been undertaken at selected sites in the English Lake District to assess changes in the plant and animal communities; for example Blelham Tarn (Haworth, 1980; Pennington & Lishman, 1984), Windermere (Pennington, 1973; Sabater & Haworth, 1995) and Esthwaite Water (Round, 1961) and a review of palaeolimnology in the district is given in Pennington (1991). Diatoms (*Bacillariophyceae*), single-celled siliceous algae, are particularly good indicators of past lake conditions and have formed the focus of many of the above studies.

A number of the Cumbrian lakes are now showing evidence of eutrophication and it is becoming increasingly recognised that in-lake surveys are required to understand the dynamics of eutrophication (Carvalho & Moss, 1995; Zinger-Gize *et al*., 1999). In 1994, in response to concern over deterioration of water quality in the district, a monitoring programme was established by the National Rivers Authority (now the Environment Agency) on eight key lakes: Bassenthwaite Lake, Derwent Water, Ullswater, Brotherswater, Esthwaite Water, Coniston Water, Elterwater and Grasmere. Various chemical measurements and planktonic data were collected at these sites. The monitoring scheme has recently been extended to include Loweswater, Buttermere and Ennerdale Water. The Agency has produced a series of Local Environment Agency Plans (LEAPs) that identify key environmental issues, including those affecting the lakes. Where issues are identified, action plans are formulated to address these and to protect the long term sustainability of these waters.

2.3 Loweswater

2.3.1 Introduction

Loweswater, National Grid Reference NY 126 217, lies on the Ordovician Skiddaw Slates in the northern region of the Lake District (Figure 2). These are the oldest rocks, marine in origin and are comprised of mudstones, flags, grits and shales (Sutcliffe, 1998). The lake lies in a catchment which contains both lowland areas (< 150 m), particularly at the north and southern ends of the lake, and upland features with a maximum altitude of 540 m (see Plate 1). Loweswater lies within the "Quieter Areas Policy" of the Lake District National Park where the aim is to conserve the special character of remoteness and quiet naturalness, and therefore there is no commercial recreation on the lake. Use of the lake is confined to fishing. There is a large area of woodland on the western side of the lake, known as Holme Wood (see Plate 2). The wood is dominated by conifer plantation but approximately 20% of the canopy cover is of native broad-leaved woodland. There is a footpath along the lake edge through part of Holme Wood. Loweswater is one of the smaller, shallower lakes of the district, with a maximum depth of 16 m, as shown by the depth contour map taken from the survey of Ramsbottom (1976) in Figure 3. The major physical characteristics of the lake and its catchment are summarised in Table 2. Figure 4 shows the catchment boundary of Loweswater and the major inflow and outflow streams.

Land use in the Loweswater catchment is summarised in Table 3 which gives percentage cover of the different land use types for both 1972 and 1988 (Lake District National Park data) along with an indication of change over this period. The only notable differences between the two years were that improved pasture increased by 5% whilst rough pasture decreased by around 4%. A map of the 1988 land use data is shown in Figure 5 which illustrates that the catchment is predominantly upland grass moorland and improved pasture with sizeable areas of woodland and bracken. Both the tabulated and mapped data relate only to the part of the Loweswater catchment that lies within the Lake District National Park. Approximately 12.5% (c.1 km^2) of the catchment falls outside the National Park boundary and no data were available for this area.

Table 2 The major physical characteristics of Loweswater lake and catchment.

Altitude m	Lake area km^2	Lake length km	Lake max depth m	Lake mean depth m	Volume m^{3}x10^6	Catchment area km^2	Cultivated area %	Approx. retention days	Littoral zone %	Pop per 10^6 m^3 lake vol.
121	0.64	1.8	16	8.4	5.4	8.9	39	150	59	13

Data from Ramsbottom (1976) and Kadiri & Reynolds (1993); human pop per 10^6 m^3 lake volume from 1971 census (Jones *et al*., 1979); potential littoral zone (0-10 m depth) from Henson (1993).

Table 3 Land use data for the Loweswater catchment within the Lake District National Park (source: Lake District National Park)

Land cover type	*% in 1972*	*% in 1988*	*Change*
Broad leaved woodland	1.2	1.1	None
Coniferous woodland	4.4	4.7	None
Mixed woodland	2.6	2.6	None
Scrub	0.3	0.3	None
Clear felled/newly planted	0.02	0.8	Small increase
Upland grass moor	40.1	40.0	None
Blanket peat grass moor	3.4	3.4	None
Bracken	10.7	8.9	Small decrease
Improved pasture	22.8	27.8	Increase
Rough pasture	5.6	1.3	Decrease
Open water (lake)	8.2	8.2	None
Bare rock	0.5	0.5	None
Isolated farmstead	0.3	0.3	None

Figure 2 Location map of Loweswater showing the main geological features of the district

Figure 3 Depth contour map of Loweswater from Ramsbottom (1976)

Figure 4 Map of Loweswater showing the catchment boundary and the major inflow and outflow streams.
(The maximum water depth of 16 m is indicated by the star)

Figure 5 Map of the land use in the Loweswater catchment

2.3.2 Loweswater in the context of the English Lake District series

A number of studies have been undertaken which have attempted to order the lakes of the English Lake District into a series. The findings of the key studies in relation to Loweswater are summarised in Table 4 and are described briefly below.

Table 4 Classification of Loweswater within the English Lake District series

Pearsall (1930, 1932)	Round (1957a)	Gorham *et al*. (1974)	Kadiri & Reynolds (1993)	Talling (1971)	Jones (1972)
Group IIa high hardness, high nitrate, circumneutral pH, low phosphate and silica.	Group I, low nutrient status based on sediments.	Group 3, productive.	Eutrophied, larger lakes.	Limited light penetration.	Productive based on phosphatase activity.

Pearsall (1921) arranged the lakes of the English Lake District into a series according to the percentage of cultivated land in the catchment, the rockiness of the littoral zone and water transparency. At one extreme were “rocky”, unproductive lakes occupying steep-sided, uncultivated valleys such as Wastwater, Ennerdale Water, Buttermere and Crummock Water while, at the other, were “silted”, relatively productive lakes adjacent to gentler, soil-covered slopes supporting woodland and improved pasture. The latter included Coniston Water, Windermere, Ullswater and Esthwaite Water. A category of intermediate lakes between the two extremes included Derwentwater, Bassenthwaite Lake, and Hawes Water. The study concluded that the differences between the lakes could be attributed to variations in the rates of erosion and sedimentation of the lake basins resulting from the variation in the durability of the underlying rocks. Therefore, the distinction between rocky and silted lakes equates to a contrast between “relatively primitive” and “more highly evolved” lakes, and the series illustrates the stages in the post-glacial development of a typical rock basin lake. Unfortunately, this study was based on only eleven lakes and Loweswater was not included. Pearsall (1930, 1932), however, went on to discern differences in the quantities of dissolved substances in the lake waters and related these to differences in the abundance and composition of their planktonic algal assemblages. Loweswater was one of nine lakes in these studies and was described as a silted lake with plankton dominated by blue-green algae and diatoms, along with Esthwaite Water and Windermere (Pearsall, 1930). It was classed as a Group IIa site, with relatively high carbonate hardness, circumneutral pH, and low phosphate and silica (Pearsall, 1930). A correlation was observed between nitrate content, pH and carbonate hardness with degree of soil in the lake catchments and shores. Loweswater was one of the lakes with relatively high nitrate, pH and hardness owing to its relatively high proportion of soil in the catchment and silt in the lake. In this study, Loweswater was described as one of the peaty sites.

The series has been reinforced by subsequent studies, notably Gorham *et al*. (1974) who classified the lakes into three groups of low, intermediate and high fertility on the basis of algal standing crop, sedimented plant pigments and water chemistry in the 1950s and 1960s. Loweswater was classified into Group 3, the productive lakes, and ranked 14 out of 20 in order of increasing fertility. Kadiri & Reynolds (1993) undertook a similar study based on data from the late 1970s to early 1980s and classed Loweswater as one of the “eutrophied larger lakes”, ranked 17 out of 20 in order of trophic status. Round (1957a) produced a classification based on sediment characteristics and their algal associations, which placed Loweswater into Group 1, low nutrient status, ranking 8 out of 21 in order of increasing productivity. Round (1957a) concluded that his sediment-based groupings agreed well with the lakewater-based Pearsall classification (1930), although in contrast to Round’s classification of the lake as “low nutrient status”, the studies of Gorham *et al*. (1974) and Kadiri & Reynolds (1993) suggested that Loweswater was amongst the more productive lakes in the region.

Talling (1971) ordered photoelectric data for 16 Cumbrian lakes according to decreasing light penetration and Loweswater was ranked 15, indicating that underwater light penetration was limited. A study of phosphatase activity in the English Lake District by Jones (1972), ranked Loweswater as 12 out of 16 lakes in terms of increasing degree of enrichment, illustrating that it is one of the more productive lakes in the series. More recently, studies on the major ions in surface waters (Sutcliffe *et al.*, 1982) have supported Pearsall's contentions about the links between land use, water chemistry and biological responses. In summary, there are clearly important gradations in chemistry and algal biomass which reflect differences in the structure and land-use of the individual catchments, as well as the changing patterns and intensities of human settlement and exploitation of resources within those catchments. Further detail on the findings of the above studies in relation to Loweswater is given in the following sections.

2.3.3 Chemistry

2.3.3.1 Early lake chemical surveys

The earliest chemical survey of Loweswater reported in the literature was in 1928 (Pearsall, 1930). Table 5 summarises the data from this survey. The data show that at that time Loweswater had relatively high nitrate, pH and hardness, and low phosphate and silica concentrations.

Table 5 Summary of chemical data for Loweswater from a survey in 1928 (Pearsall, 1930)

Determinand	*Units*	*Minimum*	*Maximum*	*Summer mean*
Iron	$\mu g\ l^{-1}$	10	120	17
Phosphate	$\mu g\ l^{-1}$	0.7	9	1.1
Nitrate	$\mu g\ l^{-1}$	22	120	54
Carbonate hardness as $CaCO_3$	$\mu g\ l^{-1}$	4000	7500	6200
Silica as SiO_2	$\mu g\ l^{-1}$	300	1800	400
pH	pH units	6.6	7.2	6.9

A number of other chemical surveys of the lake are summarised in Table 6. The first column of data is taken from Jones *et al*. (1979). Loweswater was ranked as 12 out of 16 lakes in terms of increasing degree of enrichment and was clearly one of the more productive lakes in the series (Jones, 1972; Jones *et al*., 1979). The study reported hypolimnetic deoxygenation in Loweswater (and in other productive lakes, Rydal Water, Grasmere, Esthwaite Water and Blelham Tarn). Phosphatase activity was found to be significantly correlated with total P (TP) concentrations. The richer lakes contained more TP and were capable of supporting a larger biomass of micro-organisms (Jones, 1972).

Sutcliffe *et al.* (1982) measured major ions in 24 surface waters over the period 1974-1978 and these data are summarised for Loweswater in Table 6. Loweswater was ranked 19 in terms of total anions. Sutcliffe *et al.*, (1982) compared their 1970s data with those from 1955-56 (Carrick & Sutcliffe, 1982). This showed that the sum total for means of cations had increased in most of the Cumbrian lakes, with an increase of 20-25% in Loweswater. Sodium and chloride also increased in almost all lakes including Loweswater. Interestingly there was remarkably little change in potassium in most lakes but there was a significant increase in Loweswater from 15 to 20 $\mu eq\ l^{-1}$. Also of note was a threefold increase in nitrate from 9 to 30 $\mu eq\ l^{-1}$ over this period. The sum of calcium and magnesium increased by 28% in Loweswater and Sutcliffe *et al*. (1982) observed that, in general, the biggest changes in mean calcium plus magnesium occurred in the more productive lakes. They believed that the principal sources were probably the rocks and soils in the catchments, sewage and *in situ* biological production. Agricultural use of lime in Westmorland reached its peak in the 1950s and may have leached into the lakes. However, the sources of alkalinity were not investigated on a site by site basis in this study and, therefore, the exact sources of these ions for Loweswater were not established.

Table 6 Summary of chemical data (mean values) for Loweswater from Jones *et al*. (1979), Sutcliffe *et al*. (1982), and Sutcliffe (1998).

Determinand	*Units*	*Jones et al., 1979*	*Sutcliffe et al., 1982*	*Sutcliffe, 1998*
pH	pH units		6.9	6.9
alkalinity	μeq l^{-1}		175	149
Ca^{2+}	μeq l^{-1}		291	307
Na^{+}	μeq l^{-1}		266	271
K^{+}	μeq l^{-1}		21	18
Mg^{2+}	μeq l^{-1}		130	128
Cl^{-}	μeq l^{-1}		321	359
Na/Cl			0.83	0.75
SO_4^{2-}	μeq l^{-1}		171	166
NO_3N	μeq l^{-1}		30	40
NO_3N	μg l^{-1}	404	420	
TP	μg l^{-1}	7	8	
SRP	μg l^{-1}	1.3		
Chlorophyll a	μg l^{-1}	6.8		
Depth of 1% light penetration	m	8.5		

2.3.3.2 Recent lake chemical surveys

Sutcliffe (1998) re-examined the ionic composition of surface waters in the region and these data are given for Loweswater in Table 6. The major ion concentrations and pH values were very similar to those recorded by Sutcliffe *et al*. (1982). Indeed, pH appears to have remained unchanged since Pearsall's survey in 1928 (see Table 5). Sutcliffe (1998) observed that Loweswater is different to other lakes on the Skiddaw Slates. Its chloride concentration is as high as that of Esthwaite Water which lies on Silurian Slates in the south of the Lake District. The ratio of Na/Cl is low, similar to that of the southern lakes. The ionic composition suggests some underground connection with groundwater in the Carboniferous rocks to the west, which could be a source of nutrients to Loweswater. The lake is also unusual for the northern lakes because it is relatively productive, ranked 17 amongst 20 major lakes by Kadiri & Reynolds (1993). Sutcliffe (1998) suggests that enrichment may be due to local agricultural activity but he cites the work of Pennington (1981) as showing a history of summer anoxia and high algal productivity extending back to medieval times (see section 2.3.7 for details of this study).

Recent water chemistry data for October 1998 to January 2000 collected by the Environment Agency from the sampling station at the deepest point in the lake are summarised in Table 7. The mean values must be interpreted with caution as these were calculated based only on values above the detection limit of the analytical method and will, therefore, tend to overestimate the true mean. The minimum value in each case is also based only on values that exceeded the detection limit and so for some variables, the true minimum value will be lower. The TP values were below the detection limit of 20 µg TP l^{-1} during most months, but higher concentrations were recorded at various times through the year with a maximum value of c. 1000 µg TP l^{-1} recorded in July 1999. The mean of all values above the detection limit was 115 µg TP l^{-1} but this was influenced by the one high value as all other measured values were below 70 µg TP l^{-1}. The SRP concentrations over this period were generally below the detection limit of 1 µg l^{-1}. The mean value was 2.65 µg l^{-1} but this was largely due to one high value of 6.3 µg l^{-1} measured in January 1999. These data suggest that P is the limiting nutrient in Loweswater. Nitrate values were relatively high, ranging from 200-1630 µg l^{-1} with a mean of 488 µg l^{-1}. Chlorophyll *a* concentrations were also quite high and ranged from 4-14 µg l^{-1} with a mean of c. 8 µg l^{-1}.

Table 7 Summary of chemical data for Loweswater (9-10-1998 to 31-01-2000) from the Environment Agency sampling station in the deepest point of the lake

SAMPLE DATE	Sld Sus mg/l	Chlorophylls ug/l	PhaeophytnAB ug/l	Alkalinity mg/l	Tot P ug/l	N Oxidised ug/l	Nitrate - N ug/l	Nitrite - N ug/l	Ammonia - N ug/l	Orthph-P ug/l	SiO2 ug/l
09/10/1998	<3			11.3	<20	391	386	7.8	66.6	<1	802
28/10/1998	<3	8.48									
12/11/1998	<3	11.1									
24/11/1998	<3	8.03	6.95			442	422	19.7	13.7	<1	1070
17/12/1998	<3	9.19	8.32	8.8	20	668	661	6.6	24.1	<1	1500
07/01/1999	<3	6.6	2.7	8.5	<20	494	490	3.9	31.7	6.3	1270
04/02/1999	<3	5.71	3.11	9.1	<20	621	617	4.2	20.5	1.1	480
18/02/1999	<3	8.84	7.32	73.8	29	1630	1630	3.8	8.8	<1	1900
01/03/1999	<3	7.85	7.42	8.7	62	728	726	2.4	10.5	<1	1780
31/03/1999	3	12.9	7.48	8.5	<20	675	673	1.5	23.3	1.8	1350
14/04/1999	<3	12	9.4	8.9	<20	565	562	3	19.2	<1	1230
23/04/1999	<3	6.25	6.03	11.3	18	393	391	1.8	18.9	<1	1760
29/04/1999	5	12	5.93	9.2	32	652	650	2.2	11.3	<1	1130
17/05/1999	<3	5.62	4.54	9.7	<20	366	363	2.5	9.4	<1	840
02/06/1999	<3	4.64	3.56	10.2	24			2.2	9.5	<1	394
16/06/1999	3	6.25		10.4	<20	388.4	386.4	2	21.3	<1	58.6
01/07/1999	<3			16.3	55	356.6	353.6	3	5.9	<1	279.4
13/07/1999	<3	14.4	11.6	16.8	70	204	198.7	5.3	7.9	<1	417
30/07/1999	<3	4.73	3.65	10.3	1002	298.9	291.7	7.2	5.4	<1	643
13/08/1999	3	4.02	3.59	10.5	<20	272.3	266.1	6.2	35.1	<1	719
25/08/1999	<3	10.3	8.56	10.3	20	267.3	262.9	4.4	19	<1	596.3
13/09/1999	<3	5.98	4.24	11.1	31	241.1	236.2	4.9	50.1	<1	645.7
05/10/1999	3	4.11	3.24	13.2	20	270.4	260.4	10	127.8	1.4	1435
25/10/1999	3	4.28	2.54	<15		434	366	68.7	5.1	<1	1720
30/11/1999	<3	7.5	7.28	9.5		555	553	2.2	8.7	<1	1890
20/01/2000	3	7.68	1.82	8.6		773		2.9	22.1	<1	2080
31/01/2000	5	8.21	6.69	8.5		701		2.1	17	<1	2040
Mean	3.57	7.87	5.73	13.20	115.25	516.13	488.45	7.22	23.72	2.65	1121.20
Minimum	3	4.02	1.82	8.5	18	204	198.7	1.5	5.1	1.1	58.6
Maximum	5	14.4	11.6	73.8	1002	1630	1630	68.7	127.8	6.3	2080
Det. limit	<3				<20					<1.0	

Mean = arithmetic mean based on values above the detection limit only; *Minimum* and *Maximum* = the minimum and maximum value recorded, respectively, excluding those below detection limit. *Det. limit* = detection limit of analytical method.

2.3.3.3 Continuous lake water quality monitoring data

Loweswater is one of a number of lakes in the English Lake District in which continuous water quality monitoring equipment have been installed since the mid-1990s. The system measures dissolved oxygen, pH, conductivity, temperature and chlorophyll *a*. The instruments were installed in Loweswater in June 1996 in the deep basin at 2 m, 9.5 m and 13 m water depths, with a logger at the surface for recording chlorophyll *a* concentrations. Some difficulties resulting in data loss have been encountered at Loweswater because of presence of mite eggs on the surface instrument during the summer months (Environment Agency, 1997). The general patterns for the period 1997-1999 are described below. The temperature profiles show strong thermal stratification in the lake from late April to early September, leading to very low hypolimnetic dissolved oxygen levels (<20%) during July and August (Figure 1, Appendix 1). The surface dissolved oxygen remained high throughout the year, with around 100% saturation during the spring. The pH values were in accordance with earlier surveys with surface pH usually in the range 6.5-7.5, and surface conductivity ranged from 65-85 µS cm^{-1}. In 1997 and 1998, the chlorophyll *a* data showed peak levels in spring of 30-90 µg l^{-1} and summer levels in the range 10-20 µg l^{-1}, falling to steady concentrations less than 10 µg l^{-1} after the turnover in September (Figure 2, Appendix 1). In 1999, chlorophyll *a* concentrations were lower and showed peak levels in summer of 25-34 µg l^{-1} (Figure 3, Appendix 1). Annual mean chlorophyll *a* concentrations for 1997 and 1998 were 11 µg l^{-1} and 12 µg l^{-1} respectively, placing the lake just into the eutrophic category according to the OECD (1982) classification scheme. The 1997 chlorophyll *a* conditions were reported to be indicative of high productivity and were considered to be unexpected given that the lake lies in a relatively upland catchment (Environment Agency, 1997). Secchi disk results for 1997 gave a minimum value of 1.2 m and an average value of 2.9 m.

2.3.3.4 Trends in nutrient concentrations in Loweswater

Lambert (1991) reviewed the nitrate data for Loweswater and reported that the values recorded by Carrick & Sutcliffe (1982) in 1955-56 were higher than those measured by Pearsall (1930) in 1928. When Sutcliffe *et al.*, (1982) compared their 1970s nitrate data for Loweswater with those from 1955-56 (Carrick & Sutcliffe, 1982), they concluded that there had been a threefold increase in mean nitrate concentrations from 9 to 30 µeq l^{-1} over this period. In the later survey by Sutcliffe (1998), mean nitrate concentrations had increased still further to 40 µeq l^{-1}. Sutcliffe *et al*. (1982) also recorded nitrate data expressed in µg l^{-1} and reported a mean nitrate value of 420 µg l^{-1} over the period 1974-78. A comparison of this value with a mean of 488 µg l^{-1} from the recent Environment Agency dataset also suggests that there has been an increase. Therefore, there is evidence of a long term increase in nitrate concentrations in Loweswater from the late 1920s to the present day.

There are insufficient data to examine long term trends in P in the lake but comparison of the patchy available data with the recent data indicate that TP concentrations may have increased slightly in recent years. For instance, Jones *et al*. (1979) and Sutcliffe *et al*. (1982) recorded mean TP values of 7-8 µg TP l^{-1}, and a survey by Lambert (1991) during summer 1991 recorded TP and SRP concentrations at or below the detection limit of 10 µg l^{-1}. However, in the Environment Agency dataset for 1998-2000, TP has exceeded 20 µg TP l^{-1} on a number of occasions. There does not appear to have been a marked change in SRP concentrations in Loweswater with reasonably low concentrations both in the past and currently. For example, Pearsall (1930) recorded values in the range 0.7-9 µg l^{-1} with a summer mean of 1.1 µg l^{-1}, Jones *et al*. (1979) reported a mean of 1.3 µg l^{-1}, and in the recent Environment Agency dataset, SRP ranges from < 1.0-6.3 µg l^{-1}. The current chlorophyll *a* values are higher than those recorded by Lambert (1991) (0.4 to 3.6 µg l^{-1}) or by Jones *et al*. (1979) (mean 6.8 µg l^{-1}). A comparison of the Secchi disk results for 1997 (minimum=1.2 m, mean=2.9 m), with those of Lambert (1991) who recorded a Secchi depth measurement of 3.5 m on 8/8/1991, indicates a deterioration in light penetration.

Importantly, however, a direct comparison of the current data with previous surveys is problematic owing to the different sampling and analytical methods, and the variable timing and frequencies of measurements in the studies. Therefore, all of the trends described above must be interpreted with caution.

2.3.3.5 Recent chemical surveys of the inflows and outflow

The four main inflows, Dub Beck, Crabtree Beck, Holme Beck and Black Beck, and the outflow, Dub Beck, (see Figure 4) were sampled for water chemistry on a monthly basis between April 1992 and December 1997 by the Environment Agency. However, Dub

Beck, the main inflow, was only sampled from April 1994 to December 1997. Spot chemistry data for December 1999 for all five streams was also available.

Nutrient concentrations were below the detection limit of the analytical methods for most inflows for much of the sampling period. In general, this reflects the insensitivity of the analytical methods employed and does not necessarily represent low nutrient concentrations. Dub Beck is the exception to this, where, nutrient concentrations were frequently higher and above the analytical detection limit. More sensitive analytical methods appear to have been introduced for orthophosphate in late 1996 and for the other nutrient variables for the spot chemistry in December 1999.

This limited dataset does, however, reveal that of all the inflows, Dub Beck clearly has the highest nitrogen and phosphorus concentrations. Total oxidised nitrogen concentrations are frequently below the analytical detection limit (0.5 mg l^{-1}) in Crabtree Beck, Holme Beck and Black Beck, whereas in most months they are typically >1.0 mg l^{-1} in Dub Beck (peak of 3.70 mg l^{-1} in February 1997). A comparison of orthophosphate-phosphorus concentrations can be made of the four main inflows using the data measured between November 1996-October 1997. Dub Beck had an annual mean concentration of 13 µg l^{-1} compared with 2 µg l^{-1}, 6 µg l^{-1}, and 5 µg l^{-1} for Holme Beck, Black Beck and Crabtree Beck, respectively. Assuming that Dub Beck also has the greatest flow, it can be concluded that this inflow provides the most significant contribution to the nutrient load of Loweswater. These findings are in agreement with those of Lambert (1991) who chemically examined the inflow streams and found Dub Beck to have the highest levels of nutrients of all the inflows, although the concentrations were not particularly high. In accordance with these data, the Environment Agency General Quality Assessment (GQA) scheme for monitoring river water quality identified the major inflow stream to Loweswater (Dub Beck) as class A "good", based on chemical data (Environment Agency, 1998d).

Nutrient pollution 'episodes' are also apparent in the monitoring data. The most distinct event occurred in Dub Beck in September 1996, when extremely high orthophosphate-phosphorus concentrations (295 µg l^{-1}) occurred alongside very high ammonia-nitrogen concentrations (0.78 mg l^{-1}). These could be associated with periods of high rainfall which tend to result in enhanced nutrient run-off from agriculture. Other high phosphorus concentrations such as in Holme Beck in January 1995 (126 µg l^{-1}) and Black Beck in October 1995 (153 µg l^{-1}) could simply be analytical error, as there are no simultaneous increases in other chemical variables, as might be expected.

It is imperative that future monitoring schemes employ much more sensitive analytical methods (e.g. 1 µg l^{-1} orthophosphate-phosphorus, 10 µg l^{-1} oxidised nitrogen) to provide a more complete picture of nutrient status of the main inflows. It is also essential that flow data for the inflow and outflow streams are collected alongside the chemistry samples to enable nutrient loadings to be compared and budgets to be constructed.

2.3.3.6 Summary of lake trophic status

Using the recent secchi disk and chlorophyll *a* measurements independently, Loweswater would be classified as eutrophic under the OECD system (1982) and according to the criteria of the Urban Waste Water Treatment Directive (UWWTD). However, the nutrient concentrations are relatively low with TP concentrations generally below 20 µg TP l^{-1} which places Loweswater into the mesotrophic category. On the basis of current water chemistry, therefore, it is very difficult to classify Loweswater using existing schemes. In a regional context, it is one of the more productive lakes of the English Lake District, largely because it has a higher percentage of cultivated catchment than many of the other lakes (cf. Pearsall, 1921). In a national context, however, Loweswater is less productive than many UK lakes, especially those in predominantly lowland catchments, such as the Cheshire and Shropshire Meres and the Norfolk Broads (e.g. Carvalho & Moss, 1995).

2.3.4 Phytoplankton

2.3.4.1 Blue-green algae

The earliest phytoplankton surveys reported in the literature are those conducted by Pearsall in the 1920s and even at that time there was evidence of blue-green algae in Loweswater, although apparently not in bloom densities. Pearsall & Pearsall (1929) observed *Anabaena flos-aquae* in Loweswater and Esthwaite Water, and *Microcystis incerta* in Loweswater and other productive lakes of the district. Pearsall (1930, 1932) studied the phytoplankton and dissolved substances of nine lakes and noted that a maximum cyanobacteria population occurred in Loweswater during August in the 1928 dataset. *Coelosphaerium kutzingianum* was the dominant species and *Anabaena* spp. was also present. Loweswater was one of the lakes with considerable blue-green algae in their plankton, becoming abundant at a strikingly early period owing to high organic matter in the lake at that time. Pearsall (1932) concluded that these algae show a general correlation with high organic matter in the study lakes. He also noted a high ratio of desmids to green colonial algae in Loweswater which he suggested might be due to the fact that desmids are often associated with peaty water.

Gorham *et al.* (1974) collected algal data during 1949-1951, 1955-1956, and 1961-1963. Loweswater was one of six lakes with the largest numbers of algae and high algal biomass (Group 3, productive lakes). The approximate percentage contributions of µ-algae, large algae and blue-green algae were 2, 85 and 12, respectively. Large algae made up a high proportion in all of the productive Group 3 lakes. Loweswater had one of the highest proportions of blue-green algae, with only Esthwaite Water having a higher value of 16%. The authors observed a strong correlation between algal standing crop and degree of agricultural activity in the lake catchments suggesting that the environmental factors favouring agricultural productivity also lead to high aquatic productivity. Their data also indicated that autochthonous sources of sedimentary organic matter are important in these more productive sites (i.e. burial of aquatic plant detritus rich in pigments in the anaerobic sediments). In Loweswater, numbers of blue-green algae had increased in 1961-1963 compared with 1949-1951. Lambert (1991) collated meteorological data for the district and showed that the summer of 1962 was warmer than that of 1951 and the autumn rainfall was lower in the former. Therefore, these patterns could be related to different weather conditions in those years rather than any change in lake productivity.

Kadiri & Reynolds (1993) compared phytoplankton data collected in 1978 and 1984 with those collected by Gorham *et al*. (1974) to explore changes in lake water quality over a period of two to three decades for a group of twenty lakes. In their survey, Loweswater supported a substantial population of *Oscillatoria agardhii* var. *isothrix* (now called *Planktothrix mougeotii*) throughout 1978. The same blue-greens were present in 1984. Their data placed Loweswater into the same group as the Gorham *et al*. (1974) study with the lake falling into Group 3. These were the mildly to strongly eutrophic lakes with abundant algal biomass and relatively abundant blue-green algal populations. The main difference between their data and the earlier study is that there was a two-fold increase in the large algal biomass which the authors suggested may be an indication of enrichment. They noted that this trend could be corroborated by the increasing frequency of reported blue-green algal blooms in recent years. The biomass of blue-green algae was, however, lower than in the Gorham *et al*., (1974) study. Kadiri

& Reynolds (1993) concluded that the lake required more intensive study. A summary of the phytoplankton biomass in the two studies is given in Table 8. It is difficult with studies such as these to differentiate between the influence of weather conditions during the study years and any real nutrient change. For instance, 1978 was a cool, wet summer whilst 1984 was a warm, dry summer and this could account for the greater cell count numbers of phytoplankton in 1984 compared to 1978 in the study by Kadiri & Reynolds (1993). Different weather conditions between years have been shown to have a marked influence on the composition and biomass of phytoplankton in other lakes (e.g. Reynolds & Bellinger, 1992). In Loch Leven in Scotland, weather conditions have been shown to be important in determining the size and duration of algal blooms. The most severe blooms occur in the loch during periods of warm weather and low flushing (Armstrong *et al*., 1994). Under favourable conditions, many planktonic algae can double in numbers within a few days. Therefore, the formation of a blue-green algal scum or bloom could depend on ideal weather conditions lasting for just a few hours longer than usual (Bailey-Watts, 1994).

Table 8 Summary of phytoplankton data for Loweswater from the surveys of Gorham *et al*. (1974) and Kadiri & Reynolds (1993).
All values are in µg l^{-1}.

	µ-algae	*large algae*	*blue-green algae*	*total*
Gorham *et al*. (1974)	48	1842	260	2150
Kadiri & Reynolds (1993)	31	4050	49	4130

There are a number of reports of blue-green algae blooms and scums dating from 1989. These are summarised in Table 9 and show that the years 1989, 1993, 1994 and 1996 were the most significant. NRA data collected in 1990-1991 showed that *Oscillatoria* was abundant in May 1990 and *Anabaena* was abundant in May 1991, although densities of blue-green algae in summer 1991 were very low and no blooms occurred. Annual phytoplankton surveys were conducted by the NRA in September 1992 and July 1993 (see Table 10), and revealed relatively low algal productivity with mixed populations of diatoms, flagellates and blue-green algae. The dominant blue-green alga varies between years with generally either *Oscillatoria* or *Anabaena*, and less frequently *Aphanizomenon*, occurring in the greatest concentrations. This variability may be dependent on ambient conditions because *Oscillatoria* competes well in light-deficient conditions and may be at an advantage when the water column is deeply mixed (Lambert, 1991). More recently, the Environment Agency received a report of blue-green algae on the north-east shore of the lake (NY 125 200) on 17/01/2000. The area covered was quite large. The dominant alga was *Limnothrix redeckei* which was present at bloom densities. Other taxa present in low numbers were *Gomphosphaeria* sp. and *Coelosphaerium* sp. Lambert (1991) suggests that blooms are a relatively recent phenomenon in Loweswater as high abundances of blue-green algae were not recorded in earlier surveys (e.g. Pearsall, 1932; Gorham *et al*., 1974). However, trends in blue-green algae abundance are difficult to discern because of different methodologies employed in the previous studies. Many of the early studies tended to be based on net collections where large algae, such as blue-greens, would often be exaggerated at the expense of small algae.

Table 9 Summary of blue-green algal bloom reports in Loweswater

Date	***Location***	***Species present***	***Description of bloom***
19/05/1989	NY028217 and by outflow	*Oscillatoria agardhii.*	Bloom in deeper water shown by green colouration. Scum had developed in sheltered areas.
28/11/1989		*Oscillatoria* dominant, *Anabaena* present.	Bloom was reported to cover the entire lake surface and was the third reported outbreak in 1989.
19/10/1993	NY124214	*Oscillatoria* dominant, *Coelosphaerium* and *Anabaena* significant.	Very productive bloom.
17/06/1994	NY124124	*Oscillatoria* (110 units), *Anabaena* (100 units), *Gomphospheria* / *Coleosphaerium* (18 units).	Scum was present on the surface of the sample.
13/05/1996		*Anabaena* dominant, *Aphanizomenon* present.	Bloom densities recorded.
04/09/1996	NY125213	*Anabaena* dominant.	Bloom densities recorded.

Table 10 NRA annual phytoplankton surveys of Loweswater sampled by the Marine and Special Projects in 1992 & 1993 (by D. Scott & K.J. Rouen)

Date	*Phytoplankton Productivity*	*Phytoplankton composition*	*Blue-green algae composition*	*Summary*
17/09/1992	Productivity was relatively low. Mean chl *a* value=1.67 $\mu g\ l^{-1}$ & max=2.76 $\mu g\ l^{-1}$. Group 1 of the I.F.E. Lakes series (i.e. chl *a* < 5 $\mu g\ l^{-1}$).	Diverse -14 taxa. Chrysophyte / cryptophyte flagellates, planktonic diatoms (especially *Asterionella formosa*) and blue-green algae dominant.	*Aphanizomenon flos-aquae* dominant (density 3.62 - 12.69 filaments ml^{-1}). *Coelosphaerium* / *Gomphosphaeria, Oscillatoria*, *Microcystis* and *Anabaena* present (0 -7.25 colonies / filaments ml^{-1}).	Composition similar to August / October 1991 I.F.E. Lakes Tour. Composition indicative of mesotrophic conditions.
28/07/1993	Productivity was relatively low. Mean chl *a* value=2.55 $\mu g\ l^{-1}$ & max=3.6 $\mu g\ l^{-1}$. Group 1 of the I.F.E. Lakes series (i.e. chl *a* < 5 $\mu g\ l^{-1}$).	Diverse – 15 taxa. Diatoms (especially *Tabellaria flocculosa*, *Aulacoseira*, *Fragilaria crotonensis* and centric diatoms) dominant. Blue-green algae present in small densities. Green and yellow flagellates (Cryptophyceae) and dinoflagellates present in low densities.	*Anabaena* was present in the greatest quantities at station 10 (202.4 cells ml^{-1}). *Oscillatoria* was found at all stations and had a low maximum density of 35.17 cells ml^{-1}. Additional species found in low abundance included *Aphanizomenon flos-aquae, Microcystis* and *Chroococcus*.	Composition similar to 1992 data. Composition indicative of mesotrophic conditions. The only positive result from the NRA blue-green reactive monitoring on Loweswater for 1993 was in October (*Oscillatoria*, *Anabaena*, & *Coelosphaerium*) – see Table 9.

2.3.4.2 Diatoms

There have been a number of diatom studies on Loweswater. In Pearsall's early phytoplankton studies (Pearsall, 1932), *Melosira granulata* (now called *Aulacoseira granulata*) was abundant from January to March (characteristic of all the study lakes possessing abundant blue-green algae) with *Tabellaria fenestrata* as a second characteristic diatom in May (also seen at this time in Crummock Water). The disappearance of spring diatoms in Loweswater was associated with a fall in silica to below about 0.5 mg l^{-1}. Knudson (1955) examined the distribution of *Tabellaria* in the English Lake District and observed this genus in Loweswater. Talling (1955, 1957) used clones of *Asterionella formosa* isolated from Loweswater in culture experiments so clearly this taxon was present in the lake at that time.

Round (1957a,b) in his study of the sediments of 21 lakes showed that diatoms comprised a large proportion of the total algal population on the sediments of these lakes. Loweswater was noteworthy for its high production of *Gyrosigma*, a diatom that prefers sediments with high calcium content. Other frequent genera were *Neidium, Diploneis, Caloneis, Navicula, Amphora, Nitzschia*, and *Pinnularia*. Round observed a similarity between the algal composition of Loweswater sediments and those of some of the more productive lakes of the southern region. However, the small fluctuation in the seasonal cycle of these epipelic diatoms was more similar to the pattern observed in the rocky lakes, with small growth peaks in May and July, followed by a minimum in October. There was a fall in silica and nitrate during the spring growth phase, as seen by Pearsall (1932). Round (1960) suggested that growth of blue-green algae may also contribute to the removal of N during spring.

NRA phytoplankton data from September 1992 report that the diatoms were represented by *Nitzschia* sp., *Tabellaria flocculosa*, *Melosira* (now *Aulacoseira*) sp., *Cyclotella* sp. and *Stephanodiscus* sp. at low densities. *Asterionella formosa* was the dominant taxon, although it is typically at its most abundant in early to late spring. A second comparable survey in July 1993 reported moderate densities of *Tabellaria flocculosa* and low densities of *Aulacoseira* spp., *Fragilaria crotonensis* and centric diatoms. It is difficult to detect any changes in diatom species composition from the data presented in the above studies, although it is clear that *Asterionella*, *Aulacoseira* (formerly *Melosira*) and *Tabellaria* have been important components of the diatom plankton in Loweswater for at least 70 years. Based on the known ecology of these taxa, Loweswater appears to have a diatom flora typical of mesotrophic lakes.

2.3.5 Macrophytes

Stokoe (1983) undertook plant surveys of Loweswater on several occasions over the period 1975-1980 and the species list is given in Table 11. Using this list, Loweswater is classified as a "Type 3", oligotrophic lake (Palmer *et al*., 1992), largely owing to the presence of the oligotrophic species *Callitriche hamulata*, *Isoetes lacustris*, *Juncus bulbosus*, *Littorella uniflora* and *Myriophyllum alterniflorum*.

When the Palmer *et al*. (1992) trophic ranking score (TRS) is calculated, the mean TRS for Loweswater is 6.8, as shown in Table 11, also indicating that the lake is oligotrophic. Individual plant TRS scores, however, range from 3.7 (oligotrophic) to 10 (eutrophic) suggesting that a diverse range of chemical conditions exist in Loweswater. For example, the lake supports oligotrophic species such as those listed above as well as species more commonly associated with mesotrophic to eutrophic lakes, such as *Potamogeton pusillus* and *Elodea nuttallii*. Consequently, the average TRS is not particularly useful in this case. The macrophyte diversity could be explained by variation in substrates and the existence of both shallow (e.g. north and south ends) and deep (e.g. west side) marginal zones around the lake but in the absence of a vegetation distribution map, it is not possible to explore these factors further.

Table 11 Aquatic macrophyte species list for Loweswater (Stokoe, 1983), showing trophic ranking scores (TRS) where available.

Species Name	
Achillea ptarmica	*Mentha species*
Agrostis stolonifera	*Menyanthes trifoliata*
Callitriche hamulata 6.3	*Montia fontana*
Callitriche stagnalis	*Myosotis caespitosa*
Callitriche hermaphroditica 8.5	*Myriophyllum alterniflorum 6.7*
Caltha palustris	*Nitella opaca 6.7*
Carex rostrata	*Nuphar lutea 8.5*
Elatine hexandra 7.0	*Nymphaea alba 6.7*
Eleocharis palustris	*Oenanthe crocata*
Elodea canadensis 7.3	*Phalaris arundinacea*
Elodea nuttallii 10.0	*Polygonum minus*
Equisetum fluviatile	*Potamogeton alpinus 5.5*
Fontinalis species	*Potamogeton berchtoldii 7.3*
Galium palustre	*Potamogeton crispus 9.0*
Glyceria fluitans 7.0	*Potamogeton polygonifolius 3.7*
Hydrocotyle vulgaris	*Potamogeton pusillus 9.0*
Isoetes lacustris 5.0	*Potentilla palustris*
Juncus acutiflorus	*Ranunculus flammula*
Juncus articulatus	*Ranunculus omiophyllus*
Juncus bulbosus 5.3	*Ranunculus peltatus 7.0*
Juncus conglomeratus	*Scirpus lacustris*
Juncus effusus	*Senecio aquaticus*
Littorella uniflora 6.7	*Sparganium angustifolium 4.0*
Lobelia dortmanna 5.0	*Sparganium erectum*
Lythum salicaria	***MEAN TRS = 6.8***

A macrophyte survey was carried out as part of the National Trust Biological Survey in 1981 (National Trust, 1981). *Myriophyllum* sp. was recorded as the chief submerged plant, with presence of *Isoetes lacustris*. Species recorded included *Schoenoplectus lacustris*, *Carex rostrata*, *Equisetum fluviatile*, *Nuphar lutea* and *Phalaris arundinacea*. All of these species were recorded by Stokoe (1983), with the exception of *Schoenoplectus lacustris***.** Unfortunately, there were no recent macrophyte survey data available for comparison with the above studies and, therefore, changes in the macrophyte flora of the lake could not be explored**.**

2.3.6 Aquatic fauna and fish populations

There are no known species of fish or invertebrates of particular conservation interest at Loweswater (National Trust, 1981). The National Trust Biological Survey in 1981 (National Trust, 1981) reported that the aquatic fauna of the lake is moderately rich. The bottom fauna, and the leeches and oligochaetes are indicative of a productive lake. However, the presence of corixid water bugs, *Sigara scotti* and *Sigara dorsalis*, are more typical of an unproductive waterbody, and the characteristic nutrient-rich species *Sigara falleni* and *Sigara fossorum* were not recorded.

The Environment Agency General Quality Assessment (GQA) scheme for monitoring river water quality identified the major inflow stream to Loweswater as class A, very good, based on biological (invertebrate) data (Environment Agency, 1998d). Their water classification scheme, known as the River Ecosystem Classification scheme which sets targets for river quality maintenance and improvement, identified the major inflow to

Loweswater as class 1, water of very good quality suitable for all fish species, compliant with the River Ecosystem targets (Environment Agency, 1998d). The inflow streams of Loweswater have low nutrient concentrations which comply with EC standards for Salmonid waters. Even Dub Beck, with the highest nutrient load of all the inflows, supports trout (*Salmo trutta*) and salmon (*Salmo salar*). Pike, perch, minnow, bullhead and eel are also present in the lake (National Trust, 1981).

2.3.7 Sediment studies

There have been a number of studies on the surface sediments of the Cumbrian lakes. A geochemical study (Dean *et al*., 1988) showed that differences in the chemical concentrations of 35 elements mainly reflected the diverse regional geology. Loweswater was grouped with other lakes in the northeast part of the district which are all underlain by Skiddaw Slates. A microbiological study of the sediments of 16 Cumbrian lakes (Jones *et al*., 1979) showed that the lakes formed a series from oligotrophic to eutrophic. Loweswater was one of the more productive sites whose sediments contained more organic matter, were more reduced and where the overlying water was deoxygenated to a greater degree than in the rocky lakes. These results were correlated with greater microbial activity, biomass and numbers in the sediments of the richer lakes. Round (1957a, b, 1960, 1961) studied chemical features of the sediments in relation to benthic algal production of 21 lakes in the district. He devised a nutrient index based on all of the measured nutrient characteristics of the sediment and subsequently divided the study lakes into Group 1, low nutrient status and Group 2, high nutrient status. Loweswater was in Group 1. Round noted very high manganese content and low iron content in the sediments of Loweswater which he suggested might explain the absence of chlorophyceae in the lake.

The only sediment core study of Loweswater that was found in the literature was by Pennington (1981). She studied sediment cores from 17 Cumbrian lakes and recorded a present rate of sedimentation for Loweswater (core taken in 1972) of 0.5 cm yr^{-1}, faster than some of the larger, northern lakes (e.g. Wastwater, Buttermere) but slower than the productive lakes of Windermere North Basin, Blelham Tarn and Esthwaite Water. The core exhibited a clear ^{137}Cs profile, with first appearance (1954) at c. 15 cm and a peak (1963) at c. 3 cm, indicating that the record was complete. Pennington (1981) observed an horizon of extensive forest clearance at the onset of ^{14}C dating anomalies in the sediments of Loweswater as well as in those of Esthwaite Water, Blelham Tarn, Windermere, Ennerdale Water and Brotherswater at c. 1000 years ago. This marks the point at which large changes occurred in these ecosystems. There was evidence of increased input of humic substances, a diatom flora associated with productive waters and a midge fauna tolerant of seasonal deoxygenation coincident with this horizon in Esthwaite Water and Blelham Tarn (Pennington, 1981). It appeared that the shallow lakes were more vulnerable to change during this period than the deeper lakes. The cores also showed a layer of mineral sediment associated with mediaeval cultivation of cereals and crops, and thereby increased erosion. Pennington (1981) noted that in Loweswater, this thick mineral deposit reduced the hypolimnion to a level where anaerobic conditions were present during summer, despite the lake having an apparently low rate of primary production. She concludes that from around 1000 years ago, these shallow lakes have been seasonally anoxic with a flora and fauna distinct from those of the deeper lakes, though not fully “eutrophic”.

2.3.8 Potential sources of nutrients/causes of blooms in Loweswater

2.3.8.1 Agriculture

Lambert (1991) conducted a survey of farms in the Loweswater catchment and concluded that organic loading from agricultural sources was of low significance. The farms are typically small-scale and extensive with low stock numbers of mostly beef or sheep. Fangs Brow Farm (pasture 85 acres, fell 85 acres), Ireland Place (pasture 60 acres, fell 100 acres), Askill Farm (pasture 65 acres, fell 135 acres), and Hudson Place (pasture 81 acres, fell 71 acres) are all situated at the north of the lake near Waterend. There are two farms at the south end of the lake which lie partly in the Loweswater catchment: High Nook Farm which includes areas of both permanent pasture and of relatively unimproved hay meadows, and Watergate Farm which consists primarily of improved grasslands with light stock-grazing (National Trust, 1991). At the time of the Lambert (1991) survey, silage effluent did not appear to pose a serious pollution threat. Livestock have access to the lake edge and, therefore, animal excreta could be a nutrient source but this is likely to be relatively minor.

The use of artificial fertilisers is on a small scale with very low application rates, for example High Nook farm applies approximately 12 cwt of NPK fertiliser to each major field (National Trust file notes). Lund (1972) noted that the use of nitrogenous fertilisers in Westmorland was ten times greater than it was fifteen years previously (i.e. the 1950s) and that this could explain the rise in nitrate nitrogen in lake waters. Agricultural use of lime in the area reached its peak in the 1950s and may have leached into the lakes (Sutcliffe *et al*., 1982) but lime does not appear to have been applied in the Loweswater catchment for at least the last 20 years (National Trust file notes). The sources of alkalinity have not been investigated on a site by site basis, however, and the extent and impact of liming at Loweswater has not been fully established. Sutcliffe (1998) suggests that enrichment in Loweswater may be due to local agricultural activity but he cites the work of Pennington (1981) as showing a history of summer anoxia and high algal productivity extending back to medieval times.

2.3.8.2 Geology

Sutcliffe (1998) noted that the ionic composition of the lake suggests the existence of some underground connection with groundwater in the Carboniferous rocks to the west, which could be a source of nutrients to Loweswater. However, geological sources of nutrients are unlikely to have changed over short timescales.

2.3.8.3 Birds

Loweswater is an important lake in a regional context for waterbirds, owing to its relatively nutrient-rich and shallow nature, and the locally well-developed emergent and shore-line vegetation. The water supports large numbers of waterfowl, including pochard, tufted duck, widgeon, 40-50 teal, 200 coot, and over-wintering greylag geese (National Trust, 1991). The fields in the Watergate Farm area are a key feeding area for the geese and wintering numbers reach 40 pairs (National Trust, 1991). Birds contribute to the nutrient loading via excreta but are unlikely to be significant unless thousands of individuals are present.

2.3.8.4 Sewage effluent

There are no sewage works in the Loweswater catchment and, therefore, the lake does not have the large and obvious point source, nutrient inputs that contributed to the eutrophication of Windermere and Esthwaite Water (e.g. Talling & Heaney, 1988). The only possible source of sewage effluent is from diffuse septic tank sources. There are a number of properties, including a hotel, at Waterend at the northern end of the lake. Pearsall (1930) suggested that the lake may be slightly affected by sewage effluents because of its relatively high nitrate and albuminoid ammonia concentrations, although no evidence of sewage inputs was presented by the author. In contrast some fifty years later, Pennington (1981) noted that the lake was as yet free from sewage enrichment.

2.3.8.5 Changes in trophic structure
Since the early 1970s, large numbers of perch (*Perca fluvitilis*) were removed from Loweswater to restock the lake with trout. Large perch are piscivorous and can, in abundance, therefore reduce algal biomass through impacts on zooplanktivorous fish. Their removal could consequently increase algal biomass. The impact of such changes in the fish community of Loweswater are not clear, and further work on trophic interactions is required to establish whether there is any link between these changes and algal blooms in the lake.

2.3.8.6 Changes in weather patterns
Changes in weather patterns more favourable to blue-green algae development may be important. There has been a recent tendency towards milder winters and warmer summer temperatures, along with higher rainfall. In the UK, blue-green algal blooms were particularly severe in summer 1989 which was exceptionally warm and was preceded by a mild winter. Indeed three separate bloom incidents were reported in Loweswater during that year (see Table 9). In contrast, the blooms were less well developed in the lake during summer 1991 when conditions were generally cold and damp (Lambert, 1991). High rainfall episodes tend to result in enhanced nutrient run-off from agricultural land (e.g. Bailey-Watts *et al*., 1990) and could be responsible for the observed peaks in nutrient concentrations in both the inflows and the lake itself.

2.3.9 Literature review summary
A summary of the key variables and classification schemes presented in this chapter and the definition of Loweswater's current trophic status according to these, is shown in Table 12. This demonstrates that it is difficult to describe the trophic status of Loweswater because certain aspects of the water chemistry indicate mesotrophic conditions (e.g. nutrient concentrations) whilst others suggest slightly eutrophic ones (e.g. secchi disk depth and chlorophyll *a*). The inflow streams have low nutrient concentrations and support trout and salmon which suggests good water quality. A straight-forward definition based on the biological data is also not possible. The phytoplankton survey data indicate that Loweswater is a mesotrophic to mildly eutrophic lake with abundant algal biomass and relatively abundant blue-green algal populations. The current diatom flora is typical of mesotrophic lakes. However, the diverse macrophyte flora indicates that a range of chemical conditions exist in Loweswater, supporting species typical of oligotrophic, mesotrophic and eutrophic waters. Furthermore, the bottom fauna and the leeches and oligochaetes are indicative of a productive lake, whilst the dominant corixid water bugs species are more typical of an unproductive waterbody. Classification of lakes into a single trophic category, such as those used by the OECD (1982), in such situations is clearly problematic.

Table 12 Summary of Loweswater trophic classification according to criteria described in the literature review

Variable **(scheme or dataset)**	**Trophic status**
Total phosphorus (OECD, 1982)	Mesotrophic
Chlorophyll a (OECD, 1982)	Mesotrophic to Eutrophic
Secchi depth (OECD, 1982)	Eutrophic
Phytoplankton composition (Gorham et al., 1974; Kadiri & Reynolds, 1993)	Eutrophic
Phytoplankton composition (NRA, 1992/1993)	Mesotrophic
Diatom flora (current study)	Mesotrophic
Macrophyte flora (Stokoe, 1983 using Palmer et al., 1992)	Oligotrophic but including species associated with a range of trophic states
Aquatic bottom fauna (National Trust, 1981)	Productive

Water bug fauna (National Trust, 1981)	Unproductive

From the available data, there is perhaps some evidence of slight enrichment during the last century. The lake is naturally more fertile than many others in the English Lake District, because of its relatively lowland catchment with well developed soils. Nutrient data are sporadic but nitrate concentrations appear to have risen, probably associated with diffuse inputs from agricultural intensification. Phosphorus concentrations appear to remain generally low with occasional high values of TP in recent years which could be cause for concern, although data are scarce and need augmenting. Available monitoring data indicates that the main inflow of Dub Beck provides the most significant contribution to the nutrient load of Loweswater. The recent trend in blue-green algal bloom incidents is most likely also associated with optimal weather conditions. Long term data on higher aquatic plants are lacking and the impact of any chemical change on the macrophyte community could not be established. The literature review supports the need for further investigation into the water quality and ecology of the lake.

3. METHODS

3.1 Coring and lithostratigraphic analyses

A 98 cm sediment core (LOWS1) was taken on 20 October 1999 in a water depth of 15.5m in the deepest part of the lake using a mini-Mackereth piston corer (Mackereth, 1969). The coring location is indicated by the star in Figure 4. The core was extruded in the laboratory at 0.5 cm intervals from 0-50 cm and thereafter at 1.0 cm intervals to the core base. The main characteristics of the sediment and any stratigraphic changes were noted. The percentage dry weight (%dw) which gives a measure of the water content of the sediment, and percentage loss on ignition (%loi) which gives a measure of the organic matter content, were determined for each sample using standard techniques (Dean, 1974). The wet density of the sediment was measured by weighing an empty 2 cm^3 capacity brass phial to four decimal places and then filling it with wet sediment. The phial was re-weighed and the weight of the sediment divided by two to determine the density in g cm^{-3}. Variations in density downcore indicate fluctuations in sediment composition. These measurements are further explained in the results section 4.1.

3.2 Radiometric dating

A reliable method of establishing a chronology for sediment cores is to use radiometric dating techniques. ^{210}Pb occurs naturally in lake sediments as one of the radioisotopes in the ^{238}U decay series. It has a half-life of 22.26 years, making it suitable for dating sediments laid down over the past 100-150 years. The total ^{210}Pb activity in sediments comprises supported and unsupported ^{210}Pb (Oldfield & Appleby, 1984). In most samples the supported ^{210}Pb can be assumed to be in radioactive equilibrium with ^{226}Ra and the unsupported activity at any level of a core is obtained by subtracting the ^{226}Ra activity from the total ^{210}Pb.

^{210}Pb dates for sediment cores can be calculated using both the constant rate of ^{210}Pb supply (CRS) model and the constant initial ^{210}Pb concentration (CIC) model (Appleby & Oldfield, 1978). The CRS model is most widely accepted; it assumes that the ^{210}Pb supply is dominated by direct atmospheric fallout, resulting in a constant rate of supply of ^{210}Pb from the lake waters to the sediments irrespective of net dry mass accumulation rate changes. If there are interruptions to the ^{210}Pb supply, for example sediment focusing, dates are calculated either by the CIC model or by using a composite of both models. The factors controlling the choice of model are described in full in Appleby & Oldfield (1983), and Oldfield & Appleby (1984). ^{137}Cs activity in sediments prior to the 1986 Chernobyl nuclear accident derives mainly from nuclear weapons testing fallout. Where this isotope is strongly adsorbed on to sediments, the activity versus depth profile is presumed to reflect varying fallout rate and useful chronological markers are provided by the onset of ^{137}Cs fallout in 1954, and peak fallout in 1963.

Sediment samples from core LOWS1 were analysed for ^{210}Pb, ^{226}Ra, ^{137}Cs and ^{241}Am by direct gamma assay in the Liverpool University Environmental Radioactivity Laboratory, using Ortec HPGe GWL series well-type coaxial low background intrinsic germanium detectors (Appleby *et al*., 1986). ^{210}Pb was determined via its gamma emissions at 46.5keV, and ^{226}Ra by the 295keV and 352keV γ-rays emitted by its daughter isotope ^{214}Pb following three weeks storage in sealed containers to allow radioactive equilibration. ^{137}Cs and ^{241}Am were measured by their emissions at 662keV and 59.5keV. The absolute efficiencies of the detectors were determined using calibrated sources and sediment samples of known activity. Corrections were made for the effect of self absorption of low energy γ-rays within the sample (Appleby *et al.,* 1992).

3.3 Geochemical analyses

All geochemical analyses were undertaken at the Geography Department, University of Liverpool. Total cadmium (Cd), cobalt (Co), copper (Cu), lead (Pb), manganese (Mn), nickel (Ni), zinc (Zn) and phosphorus (P) were determined by total digestion using a cocktail of perchloric, nitric, and hydrofluoric acids (Allen *et al.*, 1974). The above metals were determined using flame atomic absorption with a Unicam 939 Atomic Absorption Spectrophotometer (AAS) and P was determined colorimetrically. Total silicon (Si), titanium (Ti), calcium (Ca), potassium (K), iron (Fe), sulphur (S), and zirconium (Zr) were determined by isotope source X-ray fluorescence analysis with an X-ray fluorescence spectrometer (XRF) (Boyle, 2000). Accuracy was determined using NIST SRM 2704 Buffalo River Sediment. Labile P was extracted using Olsens reagent, and determined colorimetrically. The list of determinands measured with the analytical method used and detection limit for each is given in Table 13.

Table 13 Geochemical determinands measured on LOWS1 indicating analytical method used and detection limits

Determinand	*Method*	*Detection limit*
Total P & Labile P	Colorimetric	0.01 mg g^{-1}
Ca	XRF	1.2 mg g^{-1}
Fe	XRF	0.6 mg g^{-1}
K	XRF	1.7 mg g^{-1}
Ti	XRF	0.2 mg g^{-1}
S	XRF	1.2 mg g^{-1}
Si	XRF	6.0 mg g^{-1}
Zr	XRF	26 µg g^{-1}
Cu	AAS	0.6 µg g^{-1}
Pb	AAS	1.2 µg g^{-1}
Zn	AAS	0.02 µg g^{-1}
Cd	AAS	0.05 µg g^{-1}
Ni	AAS	2 µg g^{-1}
Co	AAS	2 µg g^{-1}
Mn	AAS	0.01 mg g^{-1}

3.4 Diatom analyses

In the absence of long-term historical water chemistry data, the sediment accumulated in lakes can provide a record of past events and past chemical conditions (e.g. Smol, 1992). Diatoms (*Bacillariophyceae*) are unicellular, siliceous algae and their silica valves are generally well preserved in most lake sediments. Diatoms are sensitive to water quality changes and are, therefore, good indicators of past lake conditions such as lake pH, nutrient concentrations and salinity.

Twenty-five sub-samples from the Loweswater core, selected to cover the period of interest, were prepared and analysed for diatoms using standard techniques (Battarbee, 1986). At least 300 valves were counted from each sample using a Leitz research quality microscope with a 100 x oil immersion objective (magnification x 1000) and phase contrast. Principal floras used in identification were Krammer & Lange-Bertalot (1986, 1988, 1991a, b). All slides are archived at the ECRC. Cluster analysis was performed on the diatom core data to identify the major zones in the diatom profile using CONISS (Grimm, 1987), implemented by TILIA and TILIAGRAPH (Grimm, 1991).

CONISS is a program for stratigraphically constrained cluster analysis by the method of incremental sum of squares.

3.5 Diatom transfer functions

In recent years, the technique of weighted averaging (WA) regression and calibration, developed by ter Braak (e.g. ter Braak & van Dam, 1989), has become a standard technique in palaeolimnology for reconstructing past environmental variables. A predictive equation known as a transfer function is generated that enables the inference of a selected environmental variable from fossil diatom assemblages, based on the relationship between modern surface-sediment diatom assemblages and contemporary environmental data for a large training (or calibration) set of lakes. This approach has been successfully employed to quantitatively infer lake pH (e.g. Birks *et al.*, 1990) and lake total phosphorus (TP) concentrations (e.g. Anderson *et al*., 1993; Bennion, 1994; Bennion *et al*., 1996a), whereby modern diatom pH and TP optima and tolerances are calculated for each taxon based on their distribution in the training set, and then past pH and TP concentrations are derived from the weighted average of the optima of all diatoms present in a given fossil sample. The methodology and the advantages of WA over other methods of regression and calibration are well documented (e.g. ter Braak & van Dam, 1989).

More recently the technique has been improved by extension to a method called WA partial least squares (WA-PLS) (ter Braak & Juggins, 1993). This method overcomes some of the limitations of simple WA by using the residual correlation in the diatom data to improve the estimates of the taxa 'optima' or regression coefficients, as shown by Bennion *et al*. (1996a). WA-PLS can, however, result in over-fitting and the various advantages and problems of the technique are fully discussed by Birks (1998). The transfer function approach is able to provide estimates of baseline pH and TP concentrations in lakes, and coupled with dating of sediment cores enables the timing, rates and possible causes of acidification and enrichment to be assessed for a particular site. This information is useful for designing more informative lake classification systems and can be incorporated into lake management and conservation programmes (e.g. Bennion *et al*., 1996b).

Diatom transfer functions were applied to the Loweswater diatom data, following taxonomic harmonization between the training sets and the fossil data. Quantitative reconstructions of TP were produced using three different models, as follows:

i) Model 1: based on an unpublished training set of 46 relatively large, deep, unproductive European lakes (> 10 m maximum depth), with annual mean TP concentrations ranging from 1-130 µg TP l^{-1}, and a median value for the dataset of 17 µg TP l^{-1}. This model used WA with tolerance downweighting (i.e. taxa with high tolerances are considered to be less important than those with low tolerances) and un-transformed TP data.

ii) Model 2: based on the same training set as Model 1 but using WA partial least squares component one model (WA-PLS1), which is equivalent to simple WA with inverse deshrinking, and $\log_{10}$-transformed TP data in an attempt to improve upon Model 1.

iii) Model 3: based on a northwest European training set of 152 relatively small, shallow, productive lakes (< 10 m maximum depth), with annual mean TP concentrations ranging from 5-1200 µg TP l^{-1}, and a median value for the dataset of 104 µg TP l^{-1} (Bennion *et al.*, 1996a). This model used the WA partial least squares component 2 model (WA-PLS2) and $\log_{10}$-transformed TP data, which has been shown to maximise the predictive power and reduce the bias in the residuals for this training set (Bennion *et al.*, 1996a).

The strength of the relationship between diatom-inferred (DI-TP) and measured TP is described by the coefficient of determination known as r^2. This is calculated for the original training set of lakes (the apparent r^2) and is also calculated by a computer intensive cross-validation procedure known as jack-knifing, which is based on a test set of lakes generated from the original training set and, therefore, provides a more realistic estimate (the predicted r^2). The r^2 value can range from 0 (no fit) to 1 (perfect fit) and, therefore, the higher the value the better the model is considered to be. The errors of the models are described by the root mean square error (RMSE) which essentially summarises the differences between the measured TP values for the training set of lakes and the DI-TP values generated by the model. As for the r^2 values, these can be calculated based on the original training set (the apparent RMSE) and more realistically on the cross-validated, jack-knifed test set (the RMSE of prediction or RMSEP). The RMSE and RMSEP are measured in the units of TP used in the model (i.e. µg TP l^{-1} or $\log_{10}$ µg TP l^{-1}). The lower the error, the better the model performs. These statistics are given for the three models in Table 14. All reconstructions were implemented using CALIBRATE (Juggins & ter Braak, 1993).

Table 14 Performance statistics of the three diatom models used for reconstructing total phosphorus of Loweswater

	MODEL 1	*MODEL 2*	*MODEL 3*
Number of lakes	46	46	152
Number of diatom taxa	148	148	298
TP range	1-130 µg TP l^{-1}	0.001-2.11 $\log_{10}$ µg TP l^{-1}	0.7-3.08 $\log_{10}$ µg TP l^{-1}
Median TP	17 µg TP l^{-1}	1.23 $\log_{10}$ µg TP l^{-1}	2.02 $\log_{10}$ µg TP l^{-1}
Apparent r^2	0.63	0.72	0.91
Predicted r^2 (Jack-knifed)	0.34	0.51	0.82
Apparent RMSE	19 µg TP l^{-1}	0.25 $\log_{10}$ µg TP l^{-1}	0.15 $\log_{10}$ µg TP l^{-1}
Predicted RMSEP	26 µg TP l^{-1}	0.33 $\log_{10}$ µg TP l^{-1}	0.21 $\log_{10}$ µg TP l^{-1}

4. RESULTS

4.1 Core description

A 98 cm sediment core (LOWS1) was taken on 20 October 1999 in a water depth of 15.5m in the deepest part of the lake. The upper 40 cm of the core was a dark brown, organic mud. From 40-55 cm the sediment was slightly paler, greyish-brown in colour and the lowermost part of the core was grey and was more consolidated with a higher clay content.

The %dw, %loi and wet density profiles (Figure 6) exhibit marked changes throughout the core. Both %dw and wet density decrease gradually from the core base to the surface. High %dw values of approximately 50% and relatively high wet density values of c. 1.5 g cm^{-3} at the core base indicate dense, consolidated sediments with low moisture content. In contrast, low %dw values of less than 10% and correspondingly low wet density values 1.1 g cm^{-3} at the surface indicate sediments with high water content. The very low %dw values in the uppermost 10 cm of the core reflect very high water content because of the unconsolidated nature of the uppermost sediments. Conversely as expected, the %loi values increase steadily from very low values of around 5% at the bottom of the core to somewhat higher values of around 20% at the top. These changes indicate that the sediments in the lower core section (particularly below c. 40 cm) have a high mineral content whilst the uppermost part of the core is relatively organic.

Figure 6 Lithostratigraphic data for Loweswater core LOWS1 showing % dry weight, % loss on ignition and wet density

4.2 Radiometric dating

The results of the radiometric analyses are given in Table 15 and are shown graphically in Figure 7.

4.2.1 Lead-210 activity

Equilibrium between total ^{210}Pb activity and the supporting ^{226}Ra, corresponding to c.120 years accumulation, was reached at a depth of about 15 cm (Figure 7a). The unsupported ^{210}Pb activity (Figure 7b), calculated by subtracting the ^{226}Ra from the total ^{210}Pb activity, has a maximum value at 2.25 cm but thereafter declines more or less exponentially with depth. These results suggest relatively uniform rates of accumulation except possibly in the more recent sediments.

4.2.2 Artificial fallout radionuclides

The ^{137}Cs activity versus depth profile (Figure 7c) has a well defined sub-surface peak at a depth of 4.25 cm. Since traces of ^{241}Am, characterising fallout from the atmospheric testing of nuclear weapons in the 1950s and 1960s (Appleby *et al.,* 1991), were detected at 8.25 cm, the 4.25 cm ^{137}Cs peak almost certainly records fallout in 1986 from the Chernobyl reactor fire. This is confirmed by the very high ^{137}Cs inventory in the core (13370 ± 320 Bq m^{-2}). A core collected in 1972 (Pennington, 1981) had a weapons fallout inventory of 2700 Bq m^{-2}, equivalent to just 1820 Bq m^{-2} in 1999. It would appear that downwards migration of Chernobyl ^{137}Cs via porewater diffusion has masked the earlier 1963 ^{137}Cs peak. In the 1972 core, the peak ^{137}Cs concentration (recording the 1963 weapons fallout maximum) occurred at a depth of 2.5 cm. After a further 28 years accumulation it is expected that this will now be at c.7-8 cm, supporting the inference from the ^{241}Am record of a 1960s date for sediments of this depth. We can thus fairly confidently place 1986 at a depth of 4.25 ± 1.5 cm and 1963 at 8.25 ± 1.5 cm.

4.2.3 Core chronology

The ^{210}Pb dates calculated using the CRS and CIC dating models (Appleby & Oldfield, 1978) are shown in Figure 8, together with the 1963 and 1986 depths estimated from the ^{137}Cs and ^{241}Am stratigraphy. The CRS model dates are in good agreement with the 1986 ^{137}Cs date. The discrepancy with the CIC model dates can be attributed to the non-monotonic variations in ^{210}Pb activity in the near surface sediments. The CRS model calculations place 1963 at a depth of 7.25 cm, a little above the depth at which ^{241}Am was detected. In view of the results from the 1972 core, and since the 7.25 cm section of the present core was not analysed and the discrepancy with the stratigraphic date is within the levels of uncertainty, it appears overall that the CRS model gives the most reliable dates. The detailed chronology determined by this method is given in Table 16. The results suggest a uniform sedimentation rate of 0.022 g cm^{-2} yr^{-1} (0.08 cm yr^{-1}) from the late 19th century through to the mid 20th century. Since the 1950s there has been a steady acceleration in sedimentation rates and the contemporary value is estimated to be c. 0.046 g cm^{-2} yr^{-1} (0.43 cm yr^{-1}).

The data can be extrapolated to give an approximate date for the bottom of the core at 95 cm and for the start of the diatom record at a depth of 50 cm. Using the average sediment accumulation rate of 0.022 g cm^{-2} yr^{-1} for the lower part of the dated section, this would date the 95 cm sample to c. 1935 years BP (= before present) or c. 60 AD, and the 50 cm sample to c.690 years BP or c.1300 AD. In reality, the core may be a little older because the accumulation rate may have been slower than this in the past. An accumulation rate of 0.022 g cm^{-2} yr^{-1} is fairly fast and pristine sites often have values less than 0.01 g cm^{-2} yr^{-1} (P. Appleby, pers. comm.)

Table 15 Fallout Radionuclide Concentrations in Loweswater core LOWS1

Depth		^{210}Pb Total		^{210}Pb Unsupported		^{210}Pb Supported		^{137}Cs		^{241}Am	
cm	g cm^{-2}	Bq kg^{-1}	±	Bq kg^{-1}	±	Bq kg^{-1}	±	Bq kg^{-1}	±	Bq kg^{-1}	±
0.25	0.02	384.4	27.3	298.2	27.8	86.3	5.4	588.0	9.8	0.0	0.0
2.25	0.24	447.4	45.6	362.8	46.7	84.5	10.3	882.5	19.8	0.0	0.0
4.25	0.58	407.3	34.6	327.7	35.3	79.6	6.9	1104.1	17.1	0.0	0.0
6.25	0.96	331.6	28.1	256.1	29.0	75.5	7.3	688.1	16.4	0.0	0.0
8.25	1.37	208.1	10.7	158.6	11.0	49.5	2.3	352.9	4.8	4.5	1.3
10.25	1.86	175.5	20.7	110.2	21.3	65.2	5.2	141.2	6.3	0.0	0.0
15.25	3.22	79.7	10.8	10.6	11.3	69.1	3.4	29.8	3.7	0.0	0.0
20.25	4.41	54.5	10.3	6.2	10.7	48.3	2.6	11.5	1.7	0.0	0.0
25.25	6.04	61.6	31.8	-12.2	32.6	73.8	7.1	0.0	0.0	0.0	0.0
30.25	7.97	46.2	18.8	-16.5	19.3	62.7	4.3	0.5	3.3	0.0	0.0

Table 16 CRS ^{210}Pb chronology of Loweswater core LOWS1

Depth		Chronology			Sedimentation Rate		
		Date	Age				
cm	g cm^{-2}	AD	yr	±	g cm^{-2} yr^{-1}	cm yr^{-1}	± (%)
0.0	0.00	1999	0				
1.0	0.10	1997	2	2	0.046	0.43	16.7
2.0	0.21	1994	5	2	0.039	0.29	18.5
3.0	0.37	1990	9	3	0.035	0.22	19.8
4.0	0.54	1985	14	4	0.032	0.18	21.1
5.0	0.72	1979	20	5	0.029	0.16	24.4
6.0	0.91	1972	27	6	0.028	0.14	28.4
7.0	1.11	1965	34	9	0.026	0.13	35.1
8.0	1.32	1957	42	12	0.025	0.11	42.8
9.0	1.56	1947	52	17	0.023	0.10	48.9
10.0	1.80	1936	63	23	0.022	0.09	54.4
11.0	2.06	1924	75	25	0.022	0.08	
12.0	2.34	1912	87	27	0.022	0.08	
13.0	2.63	1899	100	29	0.022	0.08	
14.0	2.90	1887	112	32	0.022	0.08	
15.0	3.16	1875	124	33	0.022	0.08	

Extrapolated dates:
25 cm=c. 1750 AD; 50 cm=c. 1300 AD; 60 cm=c. 1100 AD; 95 cm = c. 60 AD.

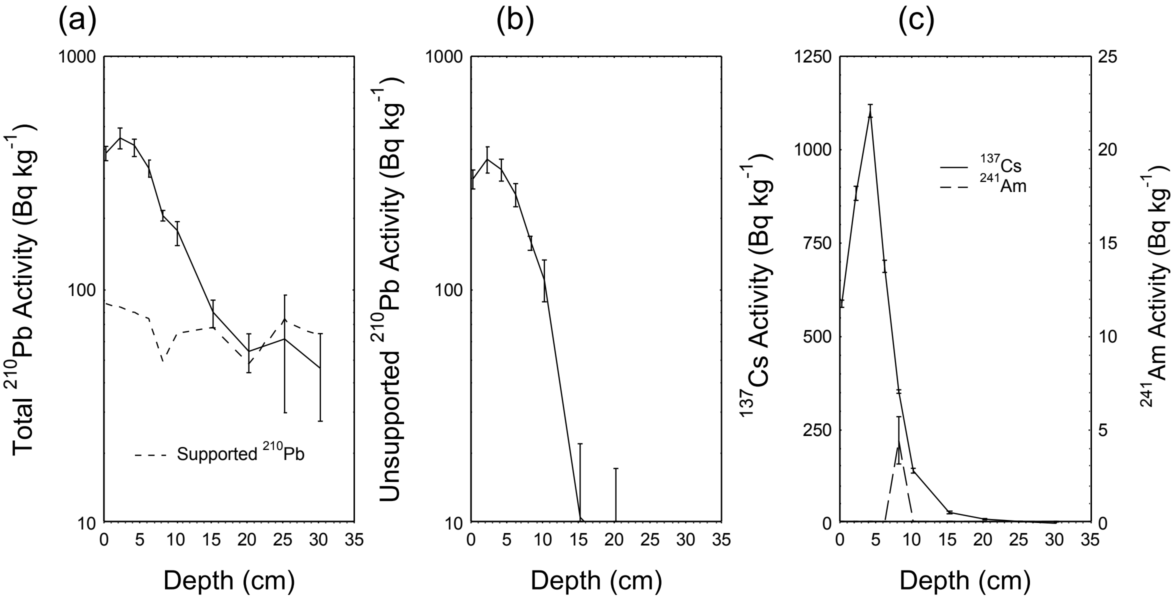

Figure 7 Fallout radionuclide concentrations versus depth in Loweswater core LOWS1 showing (a) total and supported ^{210}Pb, (b) unsupported ^{210}Pb, (c) ^{137}Cs and ^{241}Am.

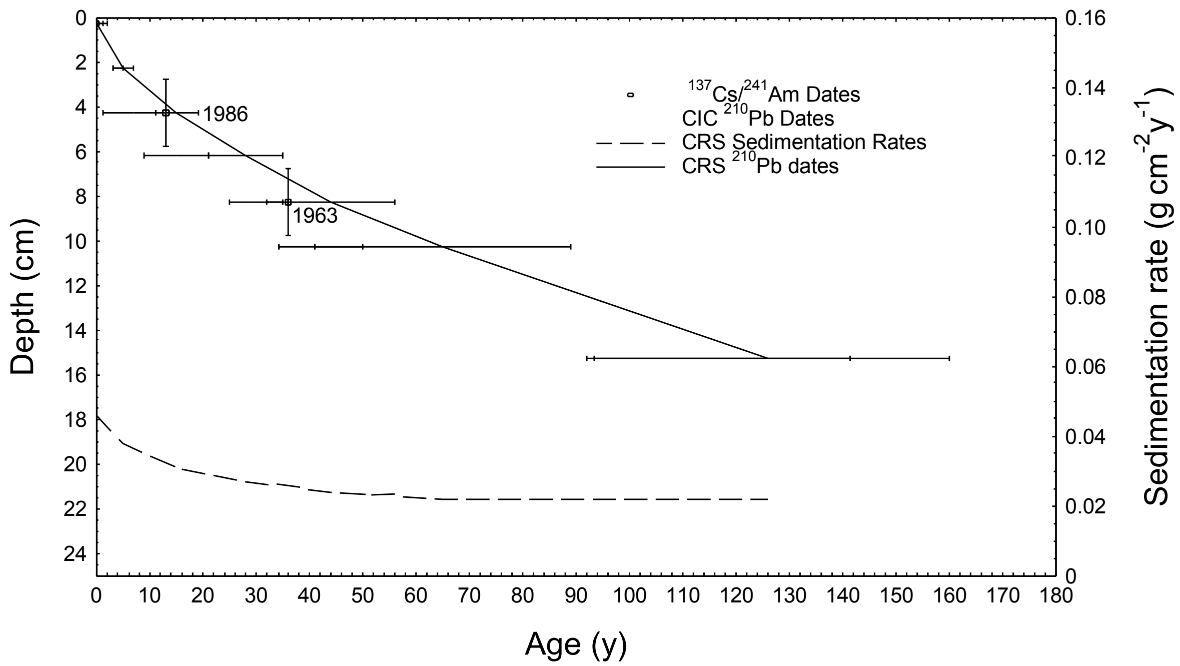

Figure 8 Radiometric chronology of Loweswater core LOWS1
showing the CRS and CIC model ^{210}Pb dates together with the 1963 and 1986 depths estimated from the ^{137}Cs and ^{241}Am stratigraphy. Also shown are sedimentation rates calculated using the CRS model.

4.3 Geochemical analyses

Concentration-depth profiles for all elements analysed in the Loweswater core are shown in Figure 9. To aid interpretation of the heavy metal profiles, a correction for the natural baseline values has been made using the method of Norton & Kahl (1987). In this, the baseline value for metal *M* at depth *d* is given by the expression:

$M_d = k_M Ti_d$ where $k_M = M_{basal}/Ti_{basal}$

Basal values for M and Ti are taken to be all values deeper than 60 cm. The baseline profiles for the heavy metals are shown in Figure 9 and the baseline corrected concentration profiles for these are shown in Figure 10.

The element concentration record can be subdivided into a number of stages according to the change with depth, and patterns of inter-element relationships. These are described below.

95 to 60 cm (c. 60-1100 AD)
This interval shows the least variation with depth. The mean concentrations of the elements in this section of the core, the concentrations in a typical shale and the ratio of the Loweswater concentrations to those in average shale are given in Table 17. In terms of major element concentrations this is a fairly typical minerogenic lake sediment, rich in the lithogenic elements except Ca. This is shown by the ratio of the Loweswater concentrations to those in average shale being close to 1.00 for most elements. The concentrations suggest a predominance of clay minerals. The low Ca values can be explained by the low Ca levels in the Skiddaw Slates bedrock which are typically around 3 mg g^{-1} (British Geological Survey, 1992). The only unusual feature of the major element concentrations is the high Fe concentration with an exceptionally high mean value of approximately 60 mg g^{-1}. The trace element composition is, however, very unusual. The Pb concentration is ten times greater than in average shale, suggesting that a significant source of Pb enrichment is present in the catchment. Furthermore, Zn is double the average shale values.

Table 17 Comparison of mean element concentrations in Loweswater basal sediments with average shale concentrations

Element	*Units*	*Mean basal concentration*	*Average shale concentration*	*Ratio to average shale*
Si	mg g^{-1}	251	238	1.05
Ti	mg g^{-1}	7.0	4.5	1.56
Ca	mg g^{-1}	3.3	25	0.13
K	mg g^{-1}	26.0	25	1.04
Fe	mg g^{-1}	63.6	47	1.35
Mn	mg g^{-1}	1.3	0.85	1.53
P	mg g^{-1}	0.99	0.75	1.32
S	mg g^{-1}	0.5	2.5	0.20
Cd	µg g^{-1}	0.29	0.3	0.97
Co	µg g^{-1}	34	20	1.70
Cu	µg g^{-1}	50	50	1.00
Pb	µg g^{-1}	164	20	8.20
Ni	µg g^{-1}	51	80	0.64
Zn	µg g^{-1}	197	90	2.19
Zr	µg g^{-1}	194	180	1.08

60 to 25 cm (c. 1100-1750 AD)

In terms of the major compositional components, this interval is characterised by increasing organic matter and S, and falling lithogenic elements (Si, Ti, K, etc.). To some extent this can be seen as an acceleration of a trend that is possibly present in the basal sediment. Si behaves slightly differently in detail, as is shown by the Si/Ti ratio. This shows a broad bulge over this depth interval, which can be interpreted as increased importance of biogenic Si. Ca shows little response; it is likely to be present in both the biogenic and lithogenic fractions. There is no change over this interval in the concentration of P.

The Fe concentration is the same as in the underlying sediment, except for a large peak at 35 cm, which coincides with peaks in a number of other elements. In order of decreasing magnitude, these are P, organic matter, Cu, S and Cd. There are also corresponding minima in the lithogenic elements. The composition is compatible with a iron-rich oxidation crust. Burial and preservation of such features is not uncommon.

The trace elements show a number of different patterns. Zr, in keeping with its lithogenic character, follows the other mineral-related major elements. Co and Cu show a gradual increase, more or less similar to the organic matter trend. Cd, Ni and Zn all show a stepped increase in concentration at 55 cm, and then maintain a fairly constant concentration, before increasing again at 20 cm. Pb is different; it may be showing an underlying trend rather like Cu and Co, but has a sharp peak at 45 cm. No other elements show a corresponding peak.

Baseline corrected heavy metal profiles (Figure 10) show a steeper rise over this interval. The heavy element patterns indicate a source of gradually increasing magnitude. Mining or smelting activities are a possible explanation.

25 to 9 cm (c. 1750-1950 AD)
This interval is characterised by steep concentration gradients, and a sharp peak in many of the elements. The lithogenic elements continue to decline, and show a depressed value between 16 and 19 cm. Si does not decline over the interval, and instead shows a slight increase. The sharp increase in the Si/Ti ratio suggests a rapidly growing importance of biogenic silica. Both S and organic matter rise steeply through the interval, and S shows a strong, sharp peak at 19 cm. Total P rises steadily through the interval, showing a depth profile very similar to organic matter.

The trace elements show a range of patterns though this interval. Cd and Zn are very similar to S, showing both a steep increase, and a sharp peak at 19 cm. Pb resembles this pattern, but with a particularly strong peak at 19 cm. The metal/S ratios for Cd and Zn are constant through this interval, while for Pb the ratio increases steeply. Co and Mn both show a peak at 19 cm, but decrease through the interval. Cu and Ni show least variation, and in terms of the uncorrected concentrations they are different from the other elements. Though they both show slight concentration increases, they have depth profiles similar to organic matter. However, if we look at their baseline corrected concentrations (Figure 10), these are more similar to the other heavy metals.

The patterns through this interval suggest growing biogenic inputs, and steady increases in the supply of Cd, Cu, Ni, Pb and Zn. At 19 cm there is a sediment supply event of very different composition. It is extremely rich in Pb, and also enriched in S, Cd, Co, Mn and Zn.

9 to 0 cm (c. 1950-1999 AD)
This interval is characterised by an accelerating concentration of biogenic matter: organic matter, P and Si/Ti all rise steeply, and the lithogenic elements fall to a

corresponding degree. However, the trend is not simply a continuation of that seen in the underlying sediment, where S broadly covaries with organic matter. Over the top 9 cm S falls; sharply at first, but then levelling off at 5 cm. Fe also falls steeply at the base of this interval, before rising to a sharp subsurface peak (1 cm). Mn rises steadily at the base of the interval, and then exponentially through the top 3 cm to a very strong surface peak.

The trace elements again show a number of different patterns. Cd and Ni both show a broad peak between 2-5 cm, and then fall at the surface. Co follows Mn in showing a sharp surface peak. Zn gradually rises through the interval, while Pb falls steadily to a value comparable to the basal concentrations. Cu shows a strong surface enrichment, but not the steady increase of Mn or Co; it is rather more like the pattern of organic matter.

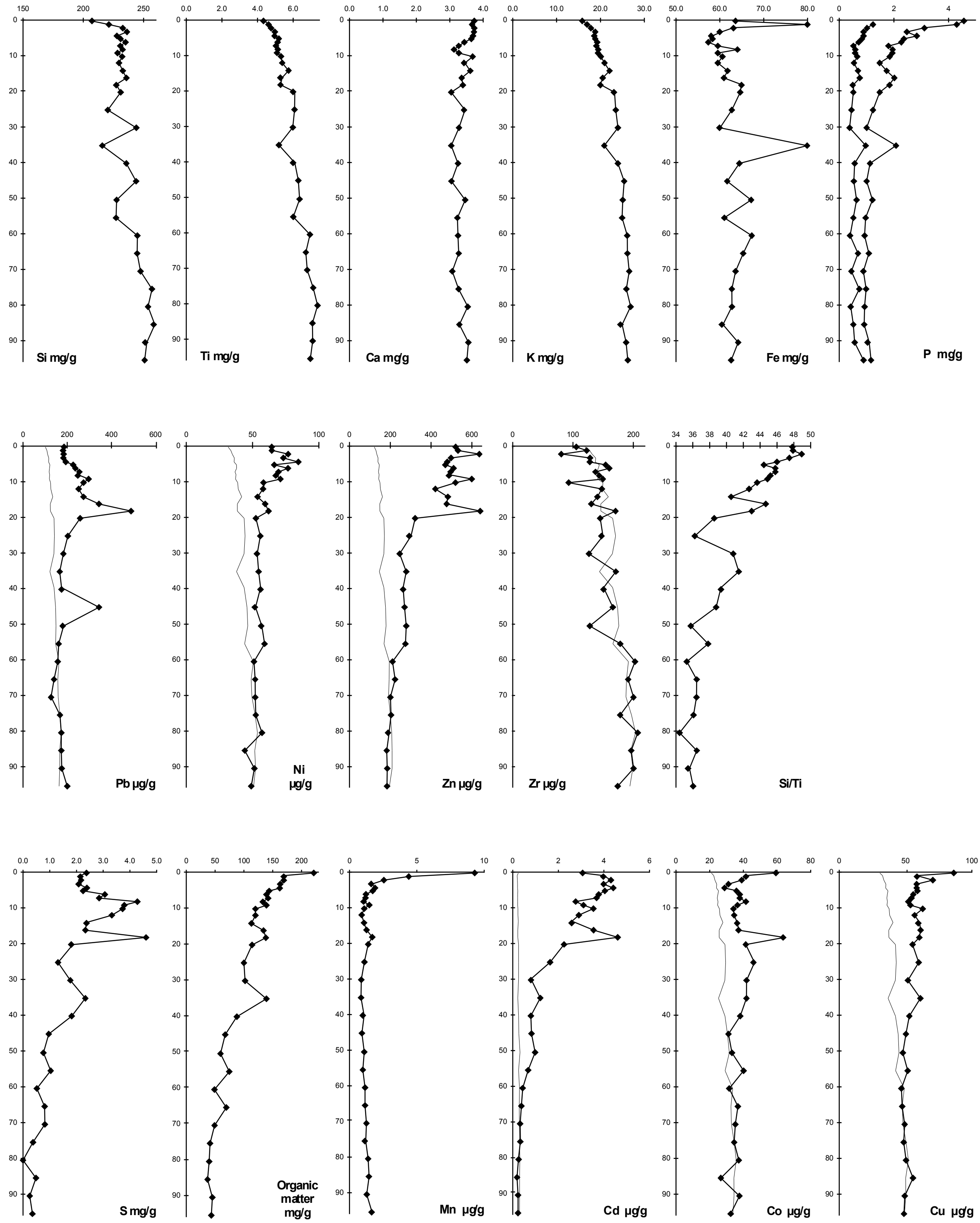

Figure 9 Total element concentrations in the Loweswater core LOWS1

Sodium hydroxide extractable, labile P is also shown as the lower of the two sets of values on the P graph. For the trace elements (Pb, Ni, Zn, Zr, Cd, Co, Cu), predicted natural baseline values have been calculated using the method of Norton & Kahl (1987) and are shown by the lines without markers.

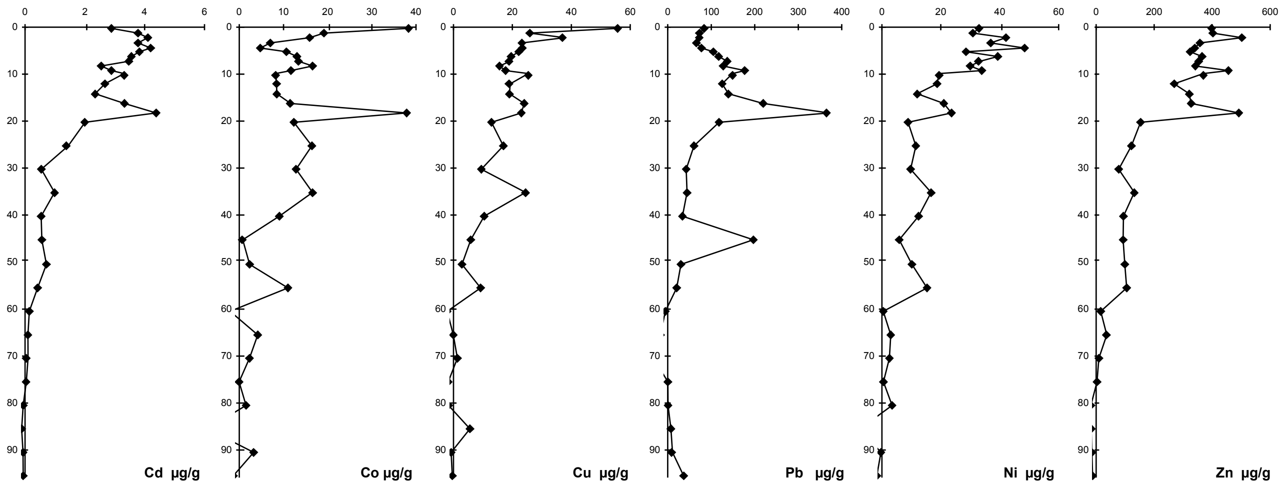

Figure 10 Baseline corrected concentrations for the trace metals in the Loweswater core LOWS1

4.4 Diatom analyses and phosphorus reconstruction

4.4.1 Diatom species shifts

The percentage relative frequencies of diatom species in twenty-five levels of the sediment core LOWS1 were calculated. A total of 125 taxa was observed and Figure 11 illustrates the results for the major taxa (> 2% abundance and > 1 occurrence). The complete list of diatom taxa observed in the core is given in Appendix 2. Diatom preservation was good throughout the upper 40 cm of the core but deteriorated below this level to the extent that samples below 50 cm could not be analysed. This was most likely due to the very inorganic nature of the sediment in the lower core section. Cluster analysis identified three major zones in the upper 50 cm section of the core. Extrapolation of the radiometric dating results given in Table 16 suggests that this section represents the period from approximately 1300 to 1999 AD.

Zone 1 (50-17 cm; c.1300-1850 AD) was co-dominated by the planktonic diatom *Aulacoseira subarctica* (20-30%), *Achnanthes minutissima* (20-30%) a non-planktonic diatom with a preference for well oxygenated water, and small, planktonic *Cyclotella* taxa (15-25%), especially *Cyclotella comensis* and *Cyclotella gordonensis*, species generally found in oligotrophic waters. Other relatively abundant taxa were the planktonic diatoms *Asterionella formosa* and *Tabellaria flocculosa. Cyclotella radiosa* was relatively important in the samples below 35 cm (5%) but this may be a function of preferential preservation given that the diatoms generally appeared to be poorly preserved and badly dissolved in this lower core section. This diatom is highly silicified and can, therefore, preserve better than some of the finer diatoms and the lightly-silicified small *Cyclotella* spp. The samples in this zone were quite diverse with an average of 40 species observed in counts of approximately 300 valves, although most of these taxa were present in very small relative percentages. Minor taxa (2-5%) included *Eunotia incisa*, *Gomphonema gracile* and *Brachysira vitrea*, all non-planktonic species generally associated with nutrient-poor waters.

Zone 2 (17-8.5 cm; c. 1850-1950 AD) saw a rise in the importance *of Aulacoseira subarctica* (c. 40%) and a consequent decline in *Cyclotella comensis*, *Cyclotella gordonensis*, and *Achnanthes minutissima*. Other notable species shifts included a marked increase in *Cyclotella glomerata* and the increasing importance of *Cyclotella radiosa* and *Tabellaria flocculosa*. Furthermore, *Fragilaria crotonensis*, a planktonic diatom often associated with lake enrichment, appeared for the first time in the sediment record. The relative abundance of *Asterionella formosa* remained similar to that in Zone 1.

Zone 3 (8.5-0 cm; c. 1950-1999 AD) saw a further decline in *Achnanthes minutissima* and the two oligotrophic *Cyclotella* taxa. Indeed *Cyclotella gordonensis* disappeared from the diatom assemblages. In contrast *Tabellaria flocculosa*, *Cyclotella radiosa* and *Fragilaria crotonensis* continued to expand. *Cyclotella glomerata*, *Asterionella formosa* and *Aulacoseira subarctica* remained important taxa in this zone. The assemblages were generally less diverse than those in Zone 1, with an average of 30 species observed in the counts of approximately 300 valves. Some of the minor taxa of Zone 1 declined in importance in Zone 3 (e.g. *Eunotia incisa*).

Figure 11 Summary diatom diagram for Loweswater core LOWS1
(showing all taxa > 2% abundance and > 1 occurrence, and zoned according to CONISS - see text. The chronology is shown on the left side of the diagram; ext = extrapolated dates - see text)

4.4.2 Diatom-phosphorus reconstructions

The diatom inferred-total phosphorus (DI-TP) reconstructions for Loweswater using the three different models are shown in Figure 12. Species analogues were good with greater than 93% of the fossil assemblage in every sample being used in the TP reconstructions in all models. This indicates that all of the major taxa present in the Loweswater core were also present in the training sets.

The results for Model 1 show that DI-TP was around 10 µg TP l^{-1} in Zone 1 with values ranging from 6-13 µg TP l^{-1}. The slightly higher values for the 30 cm and 35 cm samples were due to the lower relative abundances of the two oligotrophic *Cyclotella* taxa (*C. comensis* and *C. glomerata*) which have low TP optima, and the higher relative abundances of *Aulacoseira subarctica* which has a higher TP optimum. The DI-TP values increased slightly in the lower part of Zone 2 to around 14 µg TP l^{-1}, and then increased a little further in the upper part of Zone 2 to around 16-18 µg TP l^{-1}. There was a noticeable dip to 9 µg TP l^{-1} at the 12 cm level, owing to small peaks of *C. comensis*, *C. glomerata* and *Achnanthes minutissima* and a relative decline in *A. subarctica*. The DI-TP values in Zone 3 remained similar to those in Zone 2 falling in the range 13-16 µg TP l^{-1}. There appeared to be a slight decline in DI-TP in the uppermost part of Zone 3, due to the increase in *Tabellaria flocculosa* which has a low TP optimum. Given the relatively high errors associated with Model 1, however, these small changes in DI-TP must be interpreted with caution (see section 5 below).

The results using Model 2 were similar to those generated by Model 1. In Zone 1, DI-TP values were around 10 µg TP l^{-1}, with slightly elevated values for the 30 cm and 35 cm samples for the reasons described above. As for Model 1, DI-TP increased at the start of Zone 2 and remained similar through most of Zone 3 with values fluctuating between 13 and 20 µg TP l^{-1}. The upper two samples, however, had slightly lower DI-TP values of 11-12 µg TP l^{-1}. Model 2 seemed to be largely driven by the abundance of *Aulacoseira subarctica*, as the peaks and troughs in DI-TP values very closely matched those of this taxon.

The results of Model 3, derived from the larger north-west European dataset, were different from those of Models 1 and 2. There was no observed trend in DI-TP, with values in Zone 1 similar to those in the upper two zones. DI-TP values fluctuated between 13 and 19 µg TP l^{-1} throughout the core.

Figure 12 Diatom-total phosphorus reconstructions for Loweswater using the three different models

[The primary scale on the y-axis is age (Date AD) and the secondary scale on the y-axis is depth (cm). All TP values have been back transformed to µg TP l^{-1}. See Table 14 for the errors associated with each model. The diatom assemblage zones are shown on the right]

5. DISCUSSION

5.1 Evidence for changes in trophic status from the geochemical record

The decrease in lithogenic elements, and corresponding increase in organic matter up the core would classically be regarded as an indication of reduction in catchment erosion rates. It may, however, indicate an increase in biogenic supply or eutrophication and there is circumstantial evidence to support the latter. The sharp increases in Si/Ti and P above 20 cm are most easily explained by increased productivity of the lake.

A trend of growing eutrophication is likely, therefore, starting at a depth of 25 cm which represents around 1750 AD. The surface peak in P, and subsurface peak at 35 cm, cannot however, be interpreted the same way. While they may reflect productivity, they are at least as likely to reflect diagenetic migration of Fe and P.

5.2 Evidence for changes in redox conditions from the geochemical record

The surface enrichment in Co, Mn and Fe are all characteristic of redox remobilisation. In the case of Mn the mobility in the sediment is such that probably no features of the original depositional stratigraphy remain. However, for both Fe and Co this remobilisation is only partial. The falling trend in Co and Fe from 25 cm to 4 cm (1750-1985 AD) may indicate increased redox dissolution, and hence, more highly reducing conditions after 1750 AD. However, with the number of other possible supply factor effects, this is only speculative. There is no evidence for redox remobilisation of Ni.

5.3 Evidence for heavy metal pollution from the geochemical record

There is ample evidence for heavy metal pollution, which has a fairly complex history.

Stage 1: Before 55-60 cm (c. pre-1100 AD)
Loweswater is already strongly enriched in Pb at the base of the core, and Zn values are also high. It is not possible to establish whether these are natural concentrations or whether they are due to disturbance from early mining or smelting activities. Similarly high values were found in the bedrock of the nearby Bassenthwaite Lake catchment, which also lies on the Skiddaw Slates, e.g. Zn > 300 $\mu g\ g^{-1}$, Pb c. 150 $\mu g\ g^{-1}$ (British Geological Survey, 1992). Rollinson (1967) reports that mining development (chiefly Pb and Zn) did not occur on a major scale in the English Lake District until the 16th century which post-dates this section of the core by some 400 years. However, small scale activities may have occurred earlier.

Stage 2: 55-60 cm to 25 cm (c.1100-1750 AD)
A stepped increase in the concentration of Cd and Zn, and less distinct increases in Cu and Ni (most notably in the baseline corrected concentrations) show a change in the supply of heavy metals. There is some sign that Pb also increases, and a sharp peak in Pb at 45 cm can only be anthropogenic in origin because there is no natural fractionation method that is likely to concentrate Pb whilst not affecting any other elements. This stage may reflect relatively indirect supply from the catchment, perhaps related to disturbance. Certainly, there is no clear evidence to link these metals directly to sulphides, as can be done in the next stage.

Stage 3: 25 to 9 cm (c.1750 –1950 AD)

From 25 cm there is a steep rise in the concentrations of Cd, Pb and Zn, which continues to at least 9 cm. Baseline corrected Cu and Ni also increase, though less steeply. These increases all coincide with an increase in the concentration of S. A direct input of heavy metal sulphides is a reasonable explanation for this stage. Co shows no such effect. Mid way through this interval, at 19 cm, the sharp spike in S is accompanied by peaks in Pb, Cd, Co, Zn and, arguably, Mn. The composition of any sulphide related to this is quite different, being much more Pb-rich, and also has distinct Co enrichment. This stage is plausibly associated with heavy metal sulphide pollution related to mining or smelting.

A geochemical study of a core from nearby Bassenthwaite Lake also showed elevated concentrations of Pb and Zn, peaking around the 1880s (Morrison, 1997). The author of the study concluded that this was most likely associated with mining in the catchment. The literature indicates that metalliferous mines were present in the Loweswater and Buttermere valleys (Moon & Wildridge, 1970), thus supporting the hypothesis that the rise in Pb and Zn levels in the Loweswater core is attributed to mining practices in the catchment. If the assumption is made that these metals are delivered as sulphides, it is possible to calculate the composition of the undiluted sulphide end-members. These are shown in Table 18. It would be interesting to compare these figures with known ore from the area.

Table 18 Calculated sulphide end-member compositions
Heavy metals and S are made to total 100%.

Element	*Interval 25-9 cm, excluding the 19 cm peak (wt %)*	*19 cm peak (wt %)*
S	83	82
Cd	0.10	0.07
Co	0.0	0.8
Cu	0.35	<0.16
Pb	3.2	7.6
Ni	0.33	0.38
Zn	12.6	9.4
Fe	<39*	0

* Fe not included in the sum due to the very great uncertainty.

Stage 4: 9 to 0 cm (c. 1950-1999 AD)
S concentrations fall sharply, followed by Pb, and initially, Zn. Conversely, Ni, Cu and Cd concentration continue to rise. Clearly, this last stage of heavy metal contamination is once again not directly associated with sulphides. By comparison with the other stages, this period shows lower Pb, and higher Cd, Ni and Cu. A decline in the levels of Pb and Zn in the uppermost sediments was also observed in the Bassenthwaite Lake study (Morrison, 1997), although concentrations remained high. In this case, mining activity ceased in the 1920s and, therefore, the persistent high values were attributed to contamination from dis-used mine debris.

5.4 Evidence for eutrophication from the diatom record

The species shifts in the diatom record suggest that there has been a progression towards mesotrophy in Loweswater. Prior to around 1850, the diatom assemblages were typical of relatively nutrient-poor waters, containing many taxa commonly observed in oligotrophic systems, such as those in Wastwater (Bennion *et al.*, in press) and pre-Alpine European lakes (e.g. Wunsam & Schmidt, 1995). A number of taxa indicative of more intermediate nutrient concentrations started to increase from the mid-1800s but the most pronounced changes have occurred since the 1950s with a further expansion of *Tabellaria flocculosa* and *Cyclotella radiosa,* and the appearance of *Fragilaria crotonensis*. Numerous palaeolimnological studies from lakes in Europe and the United States have demonstrated the gradual replacement of the small *Cyclotella* spp. by *Aulacoseira* spp., *Tabellaria flocculosa* or *Fragilaria crotonensis*, which in turn are replaced by the small *Stephanodiscus* taxa as the process of nutrient enrichment takes place (e.g. Bradbury, 1975; Battarbee, 1978; Bennion, 1994; Anderson, 1997).

The oligotrophic *Cyclotella* taxa and *A. minutissima* were also abundant in the lower part of sediment cores from nearby Bassenthwaite Lake and Esthwaite Water, prior to the expansion of more nutrient tolerant taxa, suggesting that other Cumbrian lakes were relatively nutrient-poor in the past compared to recent decades (Bennion *et al.*, in press). Unlike atmospheric pollution, eutrophication is a site specific problem and there is no fixed point in time when standing waters in the UK began to show signs of change in trophic status. A comparison of the Loweswater data with those from Bassenthwaite Lake and Esthwaite Water (Bennion *et al*., in press), and Windermere South Basin (Sabater & Haworth, 1995) shows, however, that changes were first observed in the diatom assemblages of all four lakes from the mid-1800s. The coincident timing of these changes most likely marks the impact of the Industrial and Agricultural Revolutions upon the land-use in the district. The introduction of artificial fertilisers around the turn of the century may have been important. Loweswater exhibited very little change in diatom composition throughout the first half of the twentieth century, the next point of major change occurring from the 1950s. Again the timing is coincident with changes observed in other Cumbrian lakes, such as Blelham Tarn (Haworth, 1980), Windermere South Basin (Talling & Heaney, 1988; Sabater & Haworth, 1995), and Bassenthwaite Lake and Esthwaite Water (Bennion *et al*., in press). A shift towards more nutrient-rich conditions was seen in all of these lakes from the mid twentieth century. Since the 1970s, more pronounced changes have been observed in these waters largely associated with increasing P inputs from sewage effluent, including the appearance of *Stephanodiscus* spp. associated with eutrophic lakes. Importantly, *Stephanodiscus* spp. have not yet been found in the diatom record of Loweswater which suggests that it has not experienced enrichment to the same extent as these lakes. This is most likely due to the absence of major sewage inputs to the lake.

The timing of the increase in sediment accumulation rates, based on the radiometric dating results, is coincident with the change in the diatom assemblages. The ^{210}Pb profile suggested a steady acceleration in sediment accumulation rates since the 1950s, following a long period of more uniform accumulation dating back to the late nineteenth century (an increase from 0.08 to 0.43 cm yr^{-1}). This increase could be due to a combination of allochthonous inputs from the catchment such as soil inwash and agricultural runoff, and autochthonous sources of sedimentary organic matter from algal production.

The fossil diatom data agree well with the species lists produced by phytoplankton surveys of Loweswater. Talling (1955, 1957) reported the presence of *Asterionella formosa* in the lake and the fossil data show that this taxon has been present since at least 1300 AD. More recently, the phytoplankton surveys carried out by the NRA in 1992 and 1993 reported *Asterionella formosa*, *Fragilaria crotonensis*, *Tabellaria flocculosa*, *Aulacoseira* spp. and *Cyclotella* spp., and these were the dominant taxa in the recent fossil assemblages of the sediment core. The palaeolimnological data illustrate that there has been variability in diatom species composition throughout the diatom record. This is particularly evident in the upper two zones of the core where data resolution is highest, for example abundances of *Aulacoseira subarctica*, *Asterionella formosa* and *Cyclotella glomerata* vary considerably from one sample to another. The two former diatoms are typically spring-blooming taxa (Heaney *et al*., 1992) and may compete with each other for nutrients. The fluctuations in relative abundances of the major taxa could be explained simply by natural variation in the phytoplankton populations. Factors such as the timing of the onset of stratification and the flushing rate from year to year are important in controlling the composition of the phytoplankton in lakes (Talling, 1993). Palaeolimnological investigations of this kind are valuable for assessing natural versus cultural variability of lake ecosystems over long time-scales, and in the case of Loweswater the data suggest that the diatom populations have experienced variation for hundreds of years. There is no evidence of increased ecosystem instability in recent decades, despite the tendency to more nutrient-rich conditions.

5.5 Evidence for eutrophication from the diatom-phosphorus reconstructions

The DI-TP reconstructions using Models 1 and 2 support the enrichment process inferred from an ecological interpretation of the species shifts in the core. If the baseline DI-TP values (c. 10 µg TP l^{-1}) are compared with average Zone 3 DI-TP values (c. 15 µg TP l^{-1}), Loweswater has experienced a 50% increase in nutrient concentrations since around 1850. The recent DI-TP concentrations are in good agreement with current measured TP concentrations for Loweswater (see Table 7) although a direct comparison cannot be made because many of the recorded TP values are below the detection limit of 20 µg TP l^{-1}. Nevertheless, the DI-TP reconstructed values appear to be in approximately the correct range.

The results for Model 3 do not follow the pattern of enrichment inferred by Models 1 and 2. This is most likely because Model 3 is based on a training set containing many shallow, lowland lakes with annual mean TP concentrations in excess of 40 µg TP l^{-1} and is, therefore, less appropriate for application to Loweswater than the training set of large, less productive lakes used to generate Models 1 and 2. Unfortunately, however, the predictive power of Models 1 and 2 is not as high as that for Model 3, and the errors of the two former models are rather high (see Table 14). This is because the training set for the large, less productive lakes has not been fully developed and validated and currently includes only 46 lakes. Work is in progress to expand and improve the training set for oligotrophic to mesotrophic lakes, and a refined model should be available later in 2000.

The accuracy of all model results must be considered in relation to the natural variability of P in the system. The model is able to provide an estimate of annual mean TP concentrations only, whereas it is well documented that TP can be highly variable both intra-annually and inter-annually and thus it is a difficult parameter for a simple model to predict (e.g. Gibson, Foy & Bailey-Watts, 1996).

6. CONCLUSIONS

The findings of this study suggest that Loweswater has experienced slight nutrient enrichment over the last century. According to the changes in the diatom assemblages, the onset of the enrichment process occurred at around 1850 AD with a further increase in productivity since the 1950s. This is in good agreement with the geochemical record which also provides evidence of enrichment. The start of the increase in productivity, however, appears to be somewhat earlier according to the geochemical record, from around 1750 AD. Nevertheless, in accordance with the diatom data the most marked changes have occurred in the uppermost part of the core, representing approximately the last fifty years. These recent changes in both the diatom and geochemical records are coincident with the timing of the increase in sediment accumulation rates, which have increased steadily since the 1950s. Additionally, the geochemical record has provided strong evidence for heavy metal pollution which is most likely associated with mining in the area, possibly dating from one thousand years ago and most notably since around 1750 AD.

Our data, therefore, clearly indicate that Loweswater is not currently in a pristine state. Anthropogenic activity has been present in the English Lake District since at least Neolithic times (Pearsall & Pennington, 1947) and the sediment core study by Pennington (1981) demonstrated that substantial changes occurred in Loweswater at around 1000 years ago owing to forest clearance for agriculture. One would, therefore, have to analyse a much longer sediment sequence to establish true, pre-anthropogenic baselines for the lake. Clearly, our 1300 AD baselines cannot be assumed to be the "natural" lake conditions. Importantly, however, our data indicate that changes in water quality have occurred since around 1850 AD which probably marks the impact of the Industrial and Agricultural Revolutions upon land-use in the district, followed by further changes since around 1950. The findings of the study are generally in agreement with the available data on Loweswater which was reviewed in chapter 2. The lake is naturally more fertile than many others in the English Lake District because of its relatively lowland catchment with well developed soils but the literature review provided some evidence of slight enrichment during the last century.

The exact cause of the inferred enrichment is not clear and there is limited documentary data on land use changes and historical events in the catchment to aid interpretation of the findings. Of the potential sources of nutrients listed in section 2.3.8, there is insufficient data to assess the importance of bird populations and changes in the fish community, and nutrients from geological sources are unlikely to have changed over such short timescales. The contributions of the two other potential sources, agriculture and sewage effluent, will be discussed in turn below.

In terms of agricultural nutrient sources, the introduction of artificial fertilisers around the early 1900s may have been important. A ten fold increase in the use of nitrogen fertilisers in the region was reported between the 1950s and the 1970s (Lund, 1972) although no specific data exist for the Loweswater catchment. There is evidence from the published water chemistry data of a long term increase in nitrate concentrations in Loweswater from the late 1920s with more recent chemical changes since the 1950s including a significant increase in both potassium and nitrate. Potassium and nitrate are present in fertilisers and these data, therefore, lend support to the hypothesis that agricultural activity may be a source of nutrients to the lake. Unfortunately, the trends in phosphorus concentrations over the last century are less clear and the data are too sporadic to draw any firm conclusions. Historical land cover data over long time periods were not available but comparison of recent data for 1972 and 1988 showed that the general patterns of land cover had changed relatively little over this period. The data indicated conversion of around 5% of unimproved grasslands to improved pasture. Changes in land management such as intensification of practices, however, can be of more significance than simply the land cover or land use. The significance of this is unclear and further information on agricultural statistics would be required to interpret the links between changes in land-cover/management and eutrophication with any confidence.

In the absence of nutrient budget data or information on numbers of dwellings, it is not possible to quantify the contribution of sewage inputs to the nutrient load entering the lake. There are no sewage treatment works in the Loweswater catchment and, therefore, any effluent entering the lake will be derived from septic tanks. Given the low population density in the catchment, it is unlikely that sewage effluent is the main cause of the increased nutrient levels in the lake although its role as a contributory factor cannot be ruled out. Dub Beck, the main inflow to Loweswater, has the highest nutrient concentrations of all the inflows and as it flows through the most populated part of the catchment, it is possible that some of these nutrients are derived from septic tank discharge.

The inferred enrichment is likely to be a contributing factor to the observed recent trend in blue-green algal bloom incidents since nutrients are required to support the algal populations. The occurrence of blooms, however, is likely also to be associated with optimal weather conditions such as mild winters, periods of low flushing and warm, calm summers that have been experienced more frequently in recent years (late 1980s and 1990s).

7. RECOMMENDATIONS FOR MONITORING AND MANAGEMENT

7.1 Lake water quality monitoring

The ongoing lake water quality surveys are highly valuable and the present monthly water sampling of the lake (at the buoy location in the deepest point) should continue. The current detection limit of < 20 µg l^{-1} for the TP analysis, however, is rather high for Loweswater, and prevents any trends and seasonality in concentrations below this level from being established. We would, therefore, recommend that a more sensitive analytical method is adopted. The monthly sampling frequency is adequate for detecting overall patterns but intensive lake and stream sampling could be undertaken following rainfall events to establish whether there is enhanced nutrient runoff from the catchment at these times.

The continuous monitoring programme of the lake should also be continued. Clearly there is a need to continue to monitor the extent of summer deoxygenation in order to assess impacts on the fish community. The continuous chlorophyll *a* data is also essential for identifying the magnitude and frequency of algal blooms. We recommend that these data are compared with meteorological records to establish whether there is any relationship between weather patterns and occurrence of algal blooms in Loweswater.

7.2 Construction of a nutrient budget

The Environment Agency have already recognised the need for further investigations to determine the major nutrient sources to Loweswater and this should now be given priority. There is a need to gather more data via intensive and focused monitoring of the inflows and outflow streams. This is particularly important for the major inflow, Dub Beck, because available data indicates that this is the major source of nutrient loading to the lake. As discussed in 7.1 above, it is essential that future monitoring schemes employ much more sensitive analytical methods (e.g. 1 µg l^{-1} orthophosphate-phosphorus, 10 µg l^{-1} oxidised nitrogen). It is also essential that flow data for the inflow and outflow streams are collected alongside the chemistry samples to enable nutrient loadings to be compared and budgets to be constructed. An understanding of nutrient loading over the whole catchment will allow an assessment of the relative importance of nutrient sources to the lake to be made, and will ultimately permit the development of informed management strategies.

The sampling requirements for nutrient budgets are slightly different to those for general water quality monitoring and the following sampling methodology is recommended:

- The key sampling sites should be all major inflows at the point where they enter the lake, the lake itself and the outflow. It is also recommended that tributaries within the Dub Beck sub-catchment are sampled to identify where the nutrient load from this sub-catchment is originating. The co-ordinates of the sampling sites should be recorded.
- Lake level data, and flow data for all inflows and outflow are required.
- The sampling should be at regular intervals and the frequency should preferably be a minimum of once per month. The sampling period should cover a minimum of one whole year.

- Nutrient budgets should be calculated on "total" nutrient concentrations, TP and TN, because entering compounds may exit the lake as different compounds; for example ammonium may exit as particulate nitrogen (algae) or nitrate may exit as organic N.
- Chloride can be a useful element to analyse for nutrient budgets as it is very conservative and can provide information on unknown additional sources or sinks. We recommend that chloride is measured in all the inflows, the outflow and in the lake.
- Precipitation and evaporation figures for the region are also useful in interpreting budget calculations.

It is possible that both N and P loadings have increased to Loweswater and the effect of any changes on the N:P loading ratio and any subsequent effect on algal species succession should be explored.

7.3 Modelling of phosphorus pathways

If the nutrient budget calculations reveal that internal nutrient sources might be important in Loweswater (e.g the budget is unbalanced), modelling of the various P pathways through the lake could be undertaken. This should include modelling of sedimentation processes, which would identify the importance of the internal P load stored in the lake sediments, and modelling the uptake of P by phytoplankton and macrophytes.

7.4 Phytoplankton monitoring

Regular and frequent monitoring of phytoplankton abundance and species composition is recommended to determine patterns in biomass and species succession. The frequency of algal blooms and scums along shorelines and in open water areas should be monitored.

7.5 Macrophyte surveys

Long term data on higher aquatic plants are lacking and, therefore, the impact of any chemical change and the occurrence of algal blooms on the macrophyte community cannot be established at present. New surveys should be undertaken for comparison with those of Stokoe (1983) to establish whether there have been any species losses or gains in recent decades. We recommend a standard survey methodology involving inspection of lake perimeters by walking and wading, and trawling of a double headed rake grapnel operated from a boat in a direction perpendicular to the shoreline along a number of transects. We recommend the use of a bathyscope (under-water viewer) for viewing plants in deepwater locations.

7.6 Trophic interaction studies

Since the early 1970s, large numbers of perch (*Perca fluvitilis*) were removed from Loweswater to restock the lake with trout. The impact of such changes in the fish community of Loweswater are not clear, and further work on trophic interactions is required to establish whether there is any link between these changes and algal blooms in the lake.

7.7 Management options

Given that insufficient data are available at present to identify the major sources of nutrients to Loweswater, it is not possible to make specific recommendations for reducing nutrient inputs to the lake. However, a number of general recommendations are given below, which may or may not be necessary to adopt depending on the outcome of subsequent monitoring and nutrient budget calculations.

7.7.1 Agriculture

It is important to ensure that farmers in the catchment are aware of the water quality issues facing Loweswater and the role that agriculture may play as a source of nutrients to the lake. Topics such as informed management (storage and application) of inorganic and organic (silage and slurry) fertilisers should be addressed to ensure that their impacts on water quality are minimised. The codes of good agricultural practice issued by MAFF (1998) give practical guidance to farmers on these issues. Fertiliser application prior to a heavy rainfall event increases the risk of nutrient runoff (McGarrigle & Champ, 1999). These issues are particularly important given the recent concern that P loadings from modern agricultural practices may play an increasing role in the eutrophication of aquatic systems in future decades. There is substantial evidence that sustained application of fertilisers and slurry in excess of requirements has resulted in increased P surplus in agricultural soils to the extent that considerable amounts of P are transported to waters (e.g. Heathwaite *et al*., 1996; Tunney *et al*., 1997). A Nutrient Management Plan (NMP) could be implemented if deemed necessary to ensure that nutrients supplied to a crop match the demand as closely as possible.

The fencing of stream banks and lakeshores is recommended in order to limit animal access, as animal dung and urine can increase the nutrient load to the lake and provide significant BOD loading to the streams. Even a small number of cattle can cause disruption via grazing and trampling of vegetation, resulting in destabilisation of banks and enhanced erosion.

7.7.2 Sewage effluent

Although there are no sewage treatment works in the Loweswater catchment, there are a number of rural dwellings, farmhouses and a hotel, most of which are situated at Waterend to the north of the lake near the inflow. Septic tanks serving these properties could be a potential source of nutrients to the lake and, therefore, it is important that direct discharges from septic tanks to all catchment watercourses should be prevented where possible. Septic tanks should be well maintained and appropriate for current population usage. The use of phosphate-free detergents should also be encouraged.

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10. GLOSSARY

Algae	General term applied to photosynthetic organisms that are generally aquatic, may be microscopic or very large (e.g. seaweeds), and may be floating or attached.
Algal bloom	Surface scums of algae in fresh and saline waters which accumulate under calm weather conditions from populations that were previously distributed throughout the water. Decomposing dead cells consume large quantities of oxygen in the water which may result in the waters becoming anaerobic. Some blooms may produce toxins.
Benthic	Associated with the bed of a waterbody.
Biodiversity	(Biological diversity) - the variety of life, as indicated by the number of species present.
Biogenic elements	Elements that are predominantly concentrated in material formed by living organisms. Carbon and Sulphur are commonly mainly of biogenic origin. Elements such as Calcium or Silica may be either biogenic or lithogenic.
Biomanipulation	Deliberate alteration of the biological community to achieve a desirable and planned change in environmental conditions.
Biomass	Total quantity or weight of organisms in a given area or volume - e.g. algal biomass.
Blue-green algae (cyanobacteria)	Group of largely microscopic, photosynthetic organisms with a bacterial structure, but containing chlorophyll a and a photosynthetic biochemistry unlike other bacteria but similar to that of other algae and higher plants.
Buffer zone	Area of natural, semi-natural or uncultivated vegetation bordering a waterbody which is managed to intercept and trap certain diffuse pollutants, and/or to provide desired habitat types.
Catchment	Area drained by a river or a river system.
Chlorophyll a	The major photosynthetic pigment of algae and plants.
Codes of Good Agricultural Practice	A set of three codes, produced by MAFF and WOAD, giving practical guidance, that helps farmers to avoid causing water, soil or air pollution.
De-stratification	Process of mixing, or inducing circulation, in a waterbody that might otherwise settle into layers of different water quality. During calm summer periods, this artificial turbulence reduces the time which planktonic algae can spend in the well-illuminated, upper part of the water column, and so inhibits their ability to form blooms.
Diagenesis	This refers to changes in the composition of the sediment that take place after the particles have accumulated on the lake bed. Dissolution of Manganese at depth, and re-precipitation near the sediment surface is an example of diagenetic change.
Diatoms	Group of brown or yellow coloured algae commonly found in natural waters. The cell wall is made of polymerised silicate which is readily preserved in sediments when the organic part of the organism decays.
Diffuse source	Supplies of nutrients or other pollutants that come from a myriad of small-sized locations.
Directive	Legislation issued by the European Community which requires a member state to implement its requirements, for example to achieve specified environmental standards.
Eutrophic	A description of water which is rich in nutrients and is highly productive.
Eutrophication	The enrichment of water, by inorganic plant nutrients, which results in the stimulation of an array of symptomatic changes. These include the increased production of algae and/or other aquatic plants, affecting the quality of the water and disturbing the balance of organisms present within it. Such changes may be undesirable and interfere with water uses.

Export co-efficient model	Technique for calculating nutrient loadings and concentrations in a stream or lake from a knowledge of land use, numbers of stock and number of people in the catchment, stream discharge and the rates at which the nutrients are leached or excreted from the various sources.
General Quality Assessment (GQA)	The Agency's method of placing waters in categories according to assessments of different aspects ("windows") of water quality. The scheme provides a means of assessing and reporting environmental water quality in a nationally consistent and objective way.
Hindcasting	The process of estimating the state of a waterbody at a given point in the past, in the absence of appropriate baseline water quality data. This is achieved by considering natural catchment characteristics e.g. morphology, geology and soil type, and historic records of land-use, or by the assessment of past diatom communities preserved in the lake sediment.
Hypertrophic	A description of water which is extremely nutrient-enriched, and typically affected by heavy growth of algae and other water plants.
Invertebrates	Animals which lack a vertebral column (backbone)– used for biological classification. Especially aquatic macro-invertebrates (animals of sufficient size to be retained in a net with a specified mesh size).
Limiting Nutrient	Nutrient in an ecosystem which is in short supply relative to demand, and can thus inhibit efficient and productive ecological development.
Lithogenic elements	Elements that are predominantly concentrated in rock forming minerals. e.g. Potassium, Titanium and Zirconium.
Local Environment Agency Plan	The process by which the Agency plans to meet all the environmental issues in a catchment. A consultation plan is published followed by an action plan which is reviewed at five year intervals.
Macrophyte	Any plants large enough to be seen with the naked eye, including all higher aquatic plants, together with some algal species.
Mesotrophic	A description of water which is of medium nutrient status and medium biological productivity (between oligotrophic and eutrophic).
Nutrient	Substance providing nourishment for plants (or animals) e.g nitrogen, phosphorus, silicon, potassium, etc.
Nutrient export coefficient	A measure of the nutrient loss from a specific land use, typically measured as kg $ha^{-1}yr^{-1}$
Nutrient Management Plan (NMP)	A plan, often resulting from on-farm nutrient budget calculations and environmental audits, to ensure that the nutrients supplied to a crop, from the soil reserve, chemical fertilizers, manures and other wastes, closely match the need of that crop. Minimising the surplus reduces nutrient losses to water.
Oligotrophic	A description of water which has a low nutrient status and low biological productivity.
Orthophosphate	A fraction of phosphorus, often approximately equated to Soluble Reactive Phosphorus, as measured by the molybdenum blue assay on a filtered sample. (If the determination is carried out on an unfiltered sample, the fraction measured is Total Reactive Phosphorus).
Partnership Approach	Co-operation and collaboration between the Agency, its customers, and all relevant sectors of society in order to achieve a coherent and consistent framework for environmental protection and enhancement.
Phytoplankton	Community of largely microscopic algae suspended or floating in natural waters. Most species are denser than water and tend to sink, but are maintained in suspension by wind-generated water currents.
Piscivore	Animal that eats fish

Point source	Supplies of nutrients or other pollutants that come from discrete, identifiable, comparatively large origins (e.g sewage treatment works).
Pristine state	Nature of an ecosystem that is not influenced by any human activity, or at least by technologically sophisticated activity.
Redox	All environments can be classified in terms of the relative abundance of oxidising and reducing agents. Reactions that are sensitive to this balance are termed redox reactions. High productivity tends to lead to consumption of reducing conditions.
Scum	The surface debris that can result from blooms of algae and other plants.
Site of Special Scientific Interest (SSSI)	An area given a statutory designation by English Nature or the Countryside Council for Wales because it is particularly important, on account of its nature conservation value.
Total Phosphorus	The sum of dissolved and particulate phosphorus fractions.
Transfer function	A predictive equation based on the relationship between modern biological assemblages and contemporary environmental data for a set of lakes. It is used to infer a selected environmental variable from fossil assemblages in sediment cores.
Trophic state	The category of a water in relation to the process of eutrophication, typically assessed on the basis of nutrient and chlorophyll concentrations, and transparency. Waters have traditionally been classified into five trophic states: ultra-oligotrophic; oligotrophic; mesotrophic; eutrophic; and hypertrophic (see individual definitions).
Ultra-oligotrophic	A description of water with extremely low nutrient availability for the growth of algae or other plants (sometimes referred to as dystrophic).
Zooplankton	Animal community free swimming or suspended in the open water.

APPENDIX 1

Data from the Environment Agency continuous water sampling programme for Loweswater

Figure 1

Figure 2

Figure 3

APPENDIX 2

Full diatom species list for Loweswater (core LOWS1) with authorities

DIATOM NAME AND AUTHORITY	DIATOM NAME AND AUTHORITY
Achnanthes lanceolata (Breb. ex Kutz.) Grun. in Cleve & Grun. 1880	Frustulia rhomboides rhomboides (Ehrenb.) De Toni 1891
Achnanthes clevei clevei Grun. in Cleve & Grun. 1880	Gomphonema olivaceum (Hornemann) Breb. 1838
Achnanthes minutissima minutissima Kutz. 1833	Gomphonema olivaceum minutissimum Hust.
Achnanthes flexella (Kutz.) Brun 1880	Gomphonema angustatum angustatum (Kutz.) Rabenh. 1864
Achnanthes pusilla pusilla Grun. in Cleve & Grun. 1880	Gomphonema gracile Ehrenb. 1838
Achnanthes levanderi Hust. 1933	Gomphonema acuminatum acuminatum Ehrenb. 1832
Achnanthes subatomoides (Hust.) Lange-Bertalot & Archibald in K&LB 1985	Gomphonema bohemicum Reichelt & Fricke in A. Schmidt 1902
Achnanthes impexa Lange-Bertalot 1989	Gomphonema constrictum Ehrenb. ex Kutz. 1844
Achnanthes sp.	Gomphonema parvulum parvulum (Kutz.) Kutz. 1849
Amphora veneta veneta Kutz. 1844	Gomphonema minutum (Ag.) Ag. 1831
Amphora libyca Ehr.	Gomphonema [cf. pseudotenellum] LOWS (HB) 2000
Amphora pediculus (Kutz.) Grun.	Gomphonema sp.
Amphora sp.	Gyrosigma acuminatum (Kutz.) Rabenh. 1853
Amphipleura pellucida (Kutz.) Kutz. 1844	Hannaea arcus arcus (Ehrenb.) Patr. in Patr. & Reimer 1966
Asterionella formosa formosa Hassall 1850	Meridion circulare circulare (Grev.) Ag. 1831
Aulacoseira subarctica (O.Mull.) Haworth	Navicula radiosa radiosa Kutz. 1844
Brachysira vitrea (Grun.) R. Ross in Hartley 1986	Navicula hungarica Grun. 1860
Caloneis silicula (Ehrenb.) Cleve 1894	Navicula seminulum Grun. 1860
Caloneis sp.	Navicula cryptocephala cryptocephala Kutz. 1844
Cymbella sinuata sinuata Greg. 1856	Navicula rhynchocephala rhyncocephala Kutz. 1844
Cymbella microcephala microcephala Grun. in Van Heurck 1880	Navicula lanceolata (Agardh) Kutz.
Cymbella cesatii cesatii (Rabenh.) Grun. in A. Schmidt 1881	Navicula pseudoscutiformis Hust. 1930
Cymbella gracilis (Rabenh.) Cleve 1894	Navicula gregaria Donk. 1861
Cymbella ehrenbergii Kutz. 1844	Navicula cocconeiformis cocconeiformis Greg. ex Greville 1855
Cymbella minuta minuta Hilse ex Rabenh. 1862	Navicula minima minima Grun. in Van Heurck 1880
Cymbella silesiaca Bleisch ex Rabenh. 1864	Navicula subcostulata Hust. 1934
Cymbella sp.	Navicula pseudolanceolata Lange-Bertalot 1980
Cocconeis placentula placentula Ehrenb. 1838	Navicula atomus (Kutz.) Grun. 1860
Cocconeis diminuta Pant. 1902	Navicula vitabunda Hust. 1930
Cyclotella stelligera (Cleve & Grun. in Cleve) Van Heurck 1882	Navicula muraliformis Hust. ex Brendemuhl 1949
Cyclotella glomerata Bachm. 1911	Navicula [cf. seminulum] NJA+HB, Eutrophic sites 1992
Cyclotella comensis Grun. in Van Heurck 1882	Navicula sp.
Cyclotella radiosa (Grunow) Lemmerman 1900	Neidium sp.
Cyclotella distinguenda unipunctata (Hustedt) Hakansson & Carter 1990	Nitzschia fonticola Grun. in Van Heurck 1881
Cyclotella gordonensis Kling & Hakansson 1988	Nitzschia palea palea (Kutz.) W. Sm. 1856
Denticula tenuis tenuis Kutz. 1844	Nitzschia commutata Grun. in Cleve & Grun. 1880
Diploneis ovalis (Hilse) Cleve 1894	Nitzschia amphibia amphibia Grun. 1862
Diploneis oblongella oblongella (Naegeli ex Kutz.) R. Ross 1947	Nitzschia dissipata (Kutz.) Grun. 1862
Diploneis parma Cleve 1891	Nitzschia gracilis Hantzsch 1860
Diatoma hyemale hyemale (Roth) Heib. 1863	Nitzschia angustata angustata (W. Sm.) Grun. in Cleve & Grun. 1880
Epithemia adnata adnata (Kutz.) Rabenh. 1853	Nitzschia recta Hantzsch ex Rabenh. 1861
Epithemia sp.	Nitzschia linearis linearis W. Sm. 1853
Eunotia pectinalis pectinalis (O.F. Mull.) Rabenh. 1864	Nitzschia lacuum Lange-Bertalot 1980
Eunotia exigua exigua (Breb. ex Kutz.) Rabenh. 1864	Nitzschia graciliformis Lange-Bertalot & Simonsen 1978
Eunotia formica Ehrenb. 1843	Nitzschia sp.
Eunotia incisa W. Sm. ex Greg. 1854	Pinnularia interrupta W. Smith
Eunotia bilunaris (Ehrenb.) F.W. Mills 1934	Pinnularia major major (Kutz.) W. Sm. 1853
Eunotia subarcuoides Alles, Norpel, Lange-Bertalot 1991	Pinnularia viridis viridis (Nitzsch) Ehrenb. 1843
Eunotia sp.	Pinnularia sp.
Fragilaria pinnata pinnata Ehrenb. 1843	Stephanodiscus sp.
Fragilaria construens construens (Ehrenb.) Grun. 1862	Surirella linearis linearis W. Sm. 1853
Fragilaria construens venter (Ehrenb.) Grun. in Van Heurck 1881	Surirella sp.
Fragilaria virescens exigua Grun. in Van Heurck 1881	Synedra ulna ulna (Nitzsch) Ehrenb. 1836
Fragilaria brevistriata brevistriata Grun. in Van Heurck 1885	Synedra rumpens rumpens Kutz. 1844
Fragilaria crotonensis Kitton 1869	Synedra acus acus Kutz. 1844
Fragilaria capucina capucina Desm. 1825	Synedra acus angustissima (Grun. in Van Heurck) Van Heurck 1885
Fragilaria capucina gracilis (Oestrup) Hustedt 1950	Synedra nana/tenera
Fragilaria elliptica Schum. 1867	Synedra minuscula Grun. in Van Heurck 1881
Fragilaria intermedia Grun. in Van Heurck 1881	Synedra sp.
Fragilaria parasitica (W. Sm.) Grun. in Van Heurck 1881	Tabellaria flocculosa flocculosa (Roth) Kutz. 1844
Fragilaria parasitica subconstricta Grun. in Van Heurck 1881	Tabellaria sp.
Fragilaria pseudoconstruens Marciniak 1982	Tetracyclus sp.
Fragilaria sp.	