THE UNIVERSITY OF HULL

Understanding good ecological status: a palaeolimnological approach

being a Thesis submitted for the Degree of Doctor of Philosophy

in the University of Hull

by

Karen Tracy Yates-Sabren, B.Sc. (Hons)

January (2009)

Summary of Thesis submitted for Doctor of Philosophy degree

by

Karen Tracy Yates-Sabren

on

Understanding good ecological status: a palaeolimnological approach

The current study uses multiproxy palaeolimnological analysis of sediment cores to reconstruct the long-term (10^2 to 10^3 years) nutrient status of Upper Talley and Llyn Pencarreg, two closely adjacent, moderately nutrient-enriched lakes in Carmarthenshire, South Wales. The use of diatoms, *Pediastrum* and ephippia allows for the analysis of changes in ecosystem dynamics in response to nutrient enrichment, whilst pollen, charcoal analysis and archaeology enables the degree to which limnological change was driven by human impact to be examined. Reconstruction of long-term ecosystem status shows that despite the similarity of environmental parameters, Upper Talley and Llyn Pencarreg are very different lakes. Upper Talley (max. depth; 4m) demonstrates constant fluctuations in ecosystem dynamics, typical of shallow lakes with a high sensitivity to environmental change, whilst Llyn Pencarreg (max. depth 10m) illustrates a high degree of stability with gradual changes in limnological status that are indicative of deeper lakes with fewer functional groups. Within the context of the EUWFD's aim to restore lakes to good ecological status, the results indicate the need for a more detailed methodology that accounts for the complexity of long-term ecosystem dynamics in response to human impact, examines changes at different levels of the food chain using a range of proxy indicators and locates realistic baseline states based on individual lake ecology of pre-impact conditions. In this respect the ultimate aim of restoring lakes to good ecological status in a range of lake types is more likely to be achieved.

TABLE OF CONTENTS

CHAPTER 1: INTRODUCTION AND RESEARCH FOCUS

CHAPTER 2: THE PROXY INDICATORS

CHAPTER 3: THE STUDY REGION AND DESCRIPTION OF THE STUDY SITES

CHAPTER 4: METHODS AND TECHNIQUES

CHAPTER 5: RESULTS FROM THE UPPER TALLEY CORE (UT2/94)

CHAPTER 6: RESULTS FROM THE LLYN PENCARREG CORE (LPC/05)

CHAPTER 7: INTERPRETING RESULTS FROM UPPER TALLEY

CHAPTER 8: INTERPRETING RESULTS FROM LLYN PENCARREG

CHAPTER 9: RECONSTRUCTION OF THE LONG-TERM PALAEOLIMNOLOGY AND PALAEOECOLOGY OF UPPER TALLEY AND LLYN PENCARREG

CHAPTER 10: COMPARISON OF THE LONG-TERM CHANGES IN ECOSYSTEM DYNAMICS AT UPPER TALLEY AND LLYN PENCARREG

CHAPTER 11: IMPLICATIONS FOR THE EU WATER FRAMEWORK DIRECTIVE

CHAPTER 12: CONCLUSIONS

LIST OF FIGURES

LIST OF TABLES

ACKNOWLEDGEMENTS

I am indebted to my supervisors Dr. Jane Reed, Dr. Jane Bunting and Dr. Malcolm Lillie for their advice and encouragement in the completion of this PhD thesis. I would like to thank the Department of Geography at the University of Hull for financially supporting this research and for funding field trips to South Wales to extract the Llyn Pencarreg core and collect water chemistry measurements from the study sites. Thanks also to Dr. Henry Lamb from the University of Aberystwyth for the Upper Talley core (and to his family for their hospitality) and to Wendy Johnson for the pollen data from her PhD thesis.

I would particularly like to thank the Department of Geography at the University of Birmingham for the loan of their coring platform and Livingstone corer and to Dr. Warren Eastwood and Mike Marshall for their expert help with repairing the Livingston corer on site and collecting the Llyn Pencarreg core sediments. Their skill and good company kept us going despite the dreadful Welsh weather! Thank you also to the land owners, particularly Tony Wainright and John Goddard-Fenwick, for granting access to the Llyn Pencarreg site. My thanks are also extended to Professor Melanie Leng from the NERC Isotope Geosciences Laboratory in Nottingham for stable isotope analysis and to Andy Moss (University of Birmingham) for his meticulous and obsessive approach to pollen analysis. Also to John Garner for compiling the Upper Talley and Llyn Pencarreg site maps, to Dr. Graham Ferrier for ARC GIS instruction and to Dyfed Archaeology for coffee, maps and documents.

Finally, I would like to thank my family for their support both emotionally and financially. Without their encouragement this thesis would have been an impossible task.

CHAPTER 1

INTRODUCTION AND RESEARCH AIMS

1.1 Background

Water quality is a key environmental issue of global concern. Inland waters are subject to multiple pressures, including nutrient enrichment, acidification, bulk organic and toxic metal pollution, and hydro-morphological impact (e.g. Abel, 1996). Anthropogenic nutrient enrichment, or eutrophication (Greek *eutrophus*; well fed), is recognised as the most common cause of poor water quality throughout northern Europe today, particularly within many of the lowland lakes of Britain (Moss *et al.*, 1997).

The European Union Water Framework Directive (EUWFD; Directive 2000/60/EC) aims to address issues of poor water quality, by ordering the restoration of inland water bodies across Europe to *good ecological status* by 2015. The emphasis on ecological, rather than simply physico-chemical, status has had major implications in terms of water quality monitoring. There has been an expansion in the development of biotic indices and associated predictive models for invertebrates (e.g. Boon, 2000; Wright *et al.*, 2000) and, more recently, for diatoms (Kelly *et al.*, 2008). The instigation of coherent regional and international monitoring programmes has also received much attention (e.g. Boix *et al.*, 2005; Diekmann *et al.*, 2005; Gassner *et al.*, 2005; Nyman and Korhola, 2005). In brief, following the classification of water bodies (rivers, lakes, transitional waters and groundwater) into *ecotypes* based on a suite of relevant geographic, catchment, physico-chemical and biological parameters, a key concept is the definition of the *baseline* or *reference* state for each type of water body. For a particular type of river or lake, the baseline condition is taken to represent its natural state prior to significant human impact. Through a process of screening, examples of *pristine* sites have been identified for each type of water body (e.g. Bennion *et al.*, 2004 for Scottish lochs; Leira *et al.*, 2006 for lakes in Eire). The ecological status may then be defined as the degree of ecological deviation between the pre-industrial and present states. The reference state also acts as the target for restoration, in order to achieve long-term sustainability (Smol, 1992; Battarbee, 1999).

An obvious drawback in this approach is that non-impacted reference sites may be difficult to locate for some types of water body, such as lowland, fertile lakes in the UK, the catchments of which tend to be heavily populated. Following the screening of UK river and stream sites for the purposes of diatom monitoring (e.g. Kelly *et al.*, 2008), the lack of pristine reference sites in England is notable compared to the less densely populated areas of the UK (Figure 1.1). The common assumption that permanent limnological impact has occurred only over the last *ca.* 150 - 50 years as a result of intensified industrial activity and agricultural expansion is thus uncertain, particularly at sites with a long-term history of anthropogenic activity extending back beyond *ca.* AD 1850.

Figure 1.1 Map showing pristine reference sites for rivers and streams in the UK (not including N. Ireland). Note the paucity of pristine reference sites compared to the more densely populated areas of the UK (taken from Kelly *et al.*, 2008).

In terms of water quality monitoring, lakes are a special case. Unlike rivers, groundwater and coastal waters, sediment accumulates over time in the lake basin, providing the potential for high resolution proxy data on past environmental change (Smol, 2002). A range of biological, geochemical and mineralogical remains may be preserved in sediment cores, and multi-proxy palaeolimnological analysis of lake sediments can offer a powerful means by which to reconstruct past water chemistry and ecosystem dynamics. This potential has been exploited in the context of EUWFD research and palaeolimnological data from core samples dated to *ca.* AD 1850 (pre-Industrial Revolution) provide information for the reference state of lakes prior to major impact (Bennion *et al.*, 2004; Leira *et al.*, 2006). Relying on two samples per site, the approach has the major logistical advantage that analytical time is minimised. On the other hand, it is acknowledged (e.g. Bennion and Battarbee, 2007) that its reliability rests on a set of assumptions, one of the most important being that conditions prior to *ca.* AD 1850 represent long-term stability and minimal human impact. In addition, with some exceptions (e.g. Leira *et al.*, 2006), the potential for associated palynological reconstruction of the cause-and-effect relationship between human activities and eutrophication is rarely used for devising management plans.

As outlined below (Section 1.10), the primary focus of this study is to investigate long-term variability in eutrophic lake ecosystems, using palaeolimnological techniques to improve our understanding of long-term ecosystem dynamics and response to human impact. By doing so, the validity of the EUWFD approach to lake classification can be tested and realistic reference states can be defined.

1.2 Eutrophication

As water drains over the surface of lake catchments it collects dissolved organic substances, major ions and key nutrients from catchment soils and transports them via runoff to the aquatic environment (Wetzel, 2001). The concentration of key nutrients in freshwater is generally low; therefore increases in phosphorus (P) and nitrogen (N) can be the precursor to a rise in lake productivity (Moss, 1988). The extreme result of this process is *eutrophication*; a condition represented initially by increased macrophyte growth, which is dominated by canopy forming species (Scheffer, 2004). This is followed by a rise in algal production, which is characterised by species indicative of enrichment, and increased turbidity causing loss of light to the benthic zone. Deoxygenation of the hypolimnion, due to an increase in the supply of organic material, which promotes bacterial activity, may then lead to the reduction, structural simplification and eventual loss of biodiversity at all trophic levels. In extreme cases these changes are accompanied by the development of toxic, unicellular blue-green algae (e.g. *Anabaena* and *Aphanizomenon)* causing serious conservation and water quality problems (Harris, 1994).

In some cases, particularly shallow, polymictic lakes with high baseline, aquatic macrophyte biomass, competition for light and nutrients following nutrient enrichment can result in significant increases in algal biomass and the complete loss of plants. This shift in lake conditions from one state to another is referred to as *alternative stable states* (Philips and Moss, 1994; Breukers *et al.*, 1997; Karst and Smol, 2000; Scheffer and Carpenter, 2003; McGowan *et al.*, 2005) and highlights the importance of maintaining a healthy plant community. Aquatic macrophytes keep algal biomass low by shading and preventing photosynthesis and by supporting a healthy zooplankton population, which exploits the algal community and keeps primary production under control (Scheffer, 2004). Phytoplankton population expansion is prevented by the uptake of nutrients from the water column by aquatic macrophytes and clear water is maintained because macrophyte architecture and root networks reduce sediment re-suspension (James and Barko, 1990). A clear, plant-dominated state gives rise to maximum habitat diversity and a rich assemblage of species at all trophic levels (Moss *et al.*, 1997; Smol, 1992; Lau and Lane, 2001) thus producing high conservation value.

The switch to algal domination and a concomitant decrease in aquatic macrophyte abundance can lead to the loss of zooplankton refuge and the subsequent collapse of many predator–prey relationships. Consequently, the shift from one state to another is relevant to ecosystem stability. Spawning and feeding grounds for piscivorous fish may be lost whilst predation pressure from zooplanktivorous fish tends to increase (Lammens *et al.*, 1997). This reduces the number of zooplankton available to graze on algae and allows algal blooms to expand and water turbidity to increase thus encouraging the development of filamentous algal mats (Thoms *et al.*, 1999; Balaya and Moss, 2003). Periphyton growth may increase as fish such as *Tinca tinca* (Tench) and *Rutilis rutilis* (Roach) increase grazing pressure on benthic animals such as snails, which feed on periphyton (Phillips *et al.,* 1978; Phillips *et al.*, 1996). The lack of protection that accompanies a reduction in aquatic macrophytes can increase wind fetch, causing the re-suspension of internally loaded sedimentary P into the water column and perpetuating algal production even further (Smol, 2002).

Recent research has placed a great deal of emphasis on locating the exact point of change (temporally and spatially) between the clear water, macrophyte-dominated state and the turbid, algal-dominated state (McGowan *et al.*, 2005; Sheffer *et al.*, 1993; Didham and Norton, 2006). Although evidence suggests that the vegetation-turbidity relationship is controlled by changes in nutrient concentrations (Moss *et al.*, 1997), the precise nutrient levels at which the threshold can be crossed varies between lakes depending on long-term trends in anthropogenic disturbances and individual patterns of stability. This suggests that the threshold is not controlled by one linear overriding process, but by a range of complex and unpredictable feedback mechanisms (Figure 1.2) (Scheffer, 2004). Information explaining the parameters controlling ecosystem response and therefore how the combination of conditions defining ecological thresholds may vary from lake to lake is still extremely limited. An improved understanding of these mechanisms in a range of lake types would increase our ability to define realistic and sustainable baseline targets for restoration.

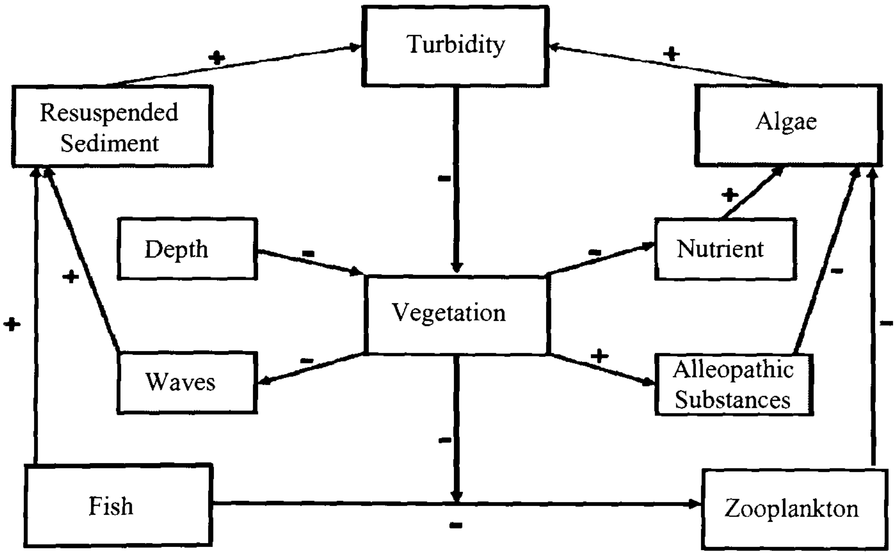


Figure 1.2 Diagram showing the positive and negative feedback systems involved in the switch from one alternative state to another (adapted from Scheffer, 2004). Each route can be evaluated by multiplying signs along the chosen path.

1.3 Catchment Characteristics and Water Chemistry

A few long-term nutrient dynamic studies have shown that before the stabilisation of catchment vegetation in the early Holocene, late-glacial lake development involved a process of *natural* eutrophication, particularly in areas of carboniferous bedrock where nutrients were easily leached (Edwards and Deevey, 1984; Delcourt and Delcourt, 1991). As nutrient rich run-off found its way into local water bodies, these lakes became increasingly more productive. However, it appears that lake development was not always characterised by increased enrichment. Renberg *et al.* (1993) and Engstrom *et al.* (2000), for example, found that the ontogeny of lakes in sparsely vegetated, non-calcareous catchments in Sweden produced an opposite trend towards reduced productivity. This was found to be the result of low nutrient availability within the more acidic catchments and the subsequent reduced movement of base cations to water bodies.

The above studies indicate that bedrock type is a major influence on natural baseline nutrient loads. To test this theory Kamenik *et al.* (2001) examined the effect of catchment characteristics on the water chemistry of lakes in the Austrian Alps along an altitudinal gradient of 1502 - 2309m a.s.l. Stepwise linear regression and multivariate statistical analysis revealed that bedrock mineralogy and the chemical weathering of carbonate minerals from the lake catchments did indeed exhibit the greatest influence on water quality status. The majority of high mountain lakes are naturally acidic due to bedrock with low calcium carbonate ($CaCO_3$) content and sparsely vegetated catchments. However, some of these lakes were unexpectedly alkaline. Kamenik *et al.* (2001) found that this was due to the additional influence of large, steep catchments, high exposure and thick soils, which encouraged the physical weathering of carbonate minerals from a catchment containing very little limestone. The study therefore also indicated that the trophic status of a lake and its natural susceptibility to eutrophication are controlled by a wide range of environmental characteristics that in combination influence its development trajectory and govern its limnological status (Smol, 2002).

1.4 The Role of Phosphorus in Lake Enrichment

During the late 1960's and early 1970's, when lake enrichment was first recognised as a problem, scientists began to take an empirical approach towards examining the cause of the increase in algal growth in lakes throughout the USA and Europe. These studies supported *Leibig's Law of the Minimum* (which states that the growth of an organism is controlled by the availability of one specific environmental factor) by showing a strong positive correlation between algal growth and the nutrient P (Vollenweider, 1968; Brylinsky and Mann, 1973; Schindler, 1974; Dillon and Rigler, 1974). The most notable of these studies was by Schindler (1974), who fertilised half of oligotrophic Lake 226, in the Experimental Lakes Area (ELA) of north-western Ontario, with P, nitrogen (N) and carbon (C) and the other half with just N and C. Within two months algal blooms appeared in the P-enriched half, whilst the other half remained oligotrophic (low productivity with P concentrations of >8 $\mu g\ l^{-1}$). From this he concluded that although P is present naturally at lower concentrations than N, it is in fact the most important limiting nutrient to algal growth.

Although some studies do stress the role of N in the process of nutrient enrichment (Penczak *et al.*, 1992; Goldman *et al.*, 1990), many examples, such as Lake Washington, Seattle, USA (Edmondson, 1991), demonstrated a significant correlation between a direct improvement in lake transparency and a reduction in P. At Lough Augher, Co. Tyrone, Northern Ireland (Anderson *et al.*, 1990) the redirection of P-rich, untreated effluent from a local creamery also reduced lake enrichment by stabilising P concentrations. This remedial action led to the eradication of algal blooms and caused the lake to return to its previous mesotrophic (medium productivity with P concentrations of >25 $\mu g\ l^{-1}$) status.

Initially, catchment characteristics, such as bedrock, exposure, dilution capacity and wind induced sediment re-suspension (Sheffer, 2004), influence the amount of P that enters surface waters leading to differences in the magnitude of P enrichment between lakes. Human impact in the catchments of shallow, lowland lakes, where P concentrations are often already high due to carbonate bedrock geology (e.g. English Lake District; Moss, 1988) may exacerbate natural enrichment. Changes in vegetation disturbance associated with changing catchment land-use can also alter hydrology patterns, causing increased run-off, and as a consequence the gross transfer of dissolved P ions from soils to lake may also be increased (Likens *et al.*, 1977). Studies in Ireland (Neal *et al.*, 2005; Jordan *et al.*, 2005) also found that the precipitation and reactive transfer of P from non-calcareous soils during periods of anthropogenic activity were lower than from soils with a high $CaCO_3$ content due to differences in hydrological properties and P-absorption, even following identical disturbance events. Drainage area size and natural flow regimes were also found to influence P inputs (Lazzarotto *et al.*, 2005). These differences indicate that even lakes located closely together within a small geographical area may have different thresholds during the enrichment process.

1.5 Temporal Patterns of Epilimnetic Phosphorus Loading

To date, the main focus for eutrophication studies has been a relatively short time span of 150 to 200 years, during which period an exponential increase in P concentrations has been observed at many sites (e.g. Anderson, 1994; Haworth *et al.*, 1998; Lotter, 1998). Since the 1960's significant quantities of nutrient rich waste from sources such as untreated sewage, storm drainage, industrial effluents, detergents, intensive stock husbandry and agricultural fertilisers have entered water-bodies causing serious water quality problems (Moss, 1988; Moss, 1996; Moss *et al.*, 1997). A recent report by DEFRA (2004) on the contribution of agriculture to the diffuse pollution of surface waters states that between 1961 and 1991 P loads from cereal cultivation alone rose by 418%, whilst loads from cattle and sheep waste increased by 95% and 135%, respectively. During this period poor nutrient management led to a doubling of losses of P from land to water from 0.62kg ha^{-1} in 1931 to 1.24kg ha^{-1} in 1991.

Catchments dominated by arable cultivation have also lost up to 100 tonnes of sediment per hectare every year compared to 5 tonnes yr^{-1} in uncultivated catchments, of which at least 25% is delivered to surface waters. DEFRA (2004) discuss case studies from Esthwaite Water, the River Avon and Slapton Ley, which indicate that these increases in sediment and P loads have led to incidents of high turbidity and a reduction in the abundance of aquatic macrophytes accompanied by the loss of some fish communities. The occurrence of toxic blooms resulting from these increases in P have also caused the contamination of potable water supplies and have compromised the amenity value and ecological status of many lowland water-bodies, making eutrophication a key conservation issue (Moss, 1996).

These short-term studies assume that modern cultural eutrophication has occurred against a long-term background of relative stasis, which represents the baseline required for restoration targets. This approach does not take into account the possibility that natural, long-term variability and/or the influence of prehistoric and historic land-use activities may have had a significant effect on water quality status, which in turn may have led to permanent shifts in water quality status and ecosystem dynamics. The few studies that link long-term changes in vegetation patterns and catchment hydrology with shifts in limnology show that pulses of prehistoric and pre-AD 1850 historic human activity and land-use change correlate closely with inferred increases in total phosphorus (TP) and changes in aquatic ecology. Fritz (1989), for example, found that moderate limnological response to prehistoric land-use at Diss Mere, Norfolk, UK, began in the Neolithic and Bronze Ages as a result of widespread deforestation to provide land for farming. Additionally, although the lake appeared to recover from these catchment events, further episodes of forest disturbance and agricultural practices within the lake catchment prevented its complete return to a pre-agricultural baseline conditions.

A similar pattern was observed at Crawford Lake, Ontario, Canada. Ekdahl (2004) found that lake productivity increased between the 13th and 15th centuries when Iroquoian Indians settled in the area. Again, permanent limnological impact followed despite over 400 years of minimal nutrient inputs following abandonment of the settlement area. Håkansson and Regnéll (1993) found similar results over the last 6000 years at Lake Bussjösjön in Southern Sweden, whilst at Dallund Sø in Denmark changes in farming practices during the medieval period released high amounts of soil nutrient ions, which contributed significantly to the lake's long-term eutrophic history (Bradshaw, 2001).

It can be seen from these studies that although some lakes appear to exhibit a certain degree of recovery following major impact events, limnological impact may be permanent, even when anthropogenic influences are reduced or removed. More importantly it appears that in cases like these, lakes may have been primed by earlier disturbances for a rapid return to eutrophic conditions as P inputs increased in the last 150-200 years. The possibility of permanent, long-term changes over recent millennia could mean that the natural, pre-impact state aimed at by restoration ecologists may now be extremely difficult to define and ultimately that the future sustainability of waterbodies may be jeopardised by choosing inappropriate restoration targets. The provision of long-term palaeolimnological proxy data is essential to reconstruct lake history, identify natural patterns of ecosystem behaviour and elucidate rates of recovery associated with ecosystem stress. Only with such data can the validity of assumptions implicit in the concept of a natural, stable baseline state be tested, and the point at which the system began to change from its natural trajectory, be defined.

1.6 Lake Restoration and Reducing P Concentrations

Studies at Lake Washington, Seattle, USA (Edmondson, 1991) and Lough Augher, Co. Tyrone, N. Ireland (Anderson *et al.*, 1990; Ripley, 1983) indicate that eutrophication from artificially-induced increases in nutrient loads may be reversed by a reduction in P loads. The success of this approach has also been demonstrated more recently at Lake Apopka, Florida, USA (Coveney *et al.*, 2005) and Lake Müggelsee, Berlin, Germany (Köhler *et al.,* 2005) where reductions in external loading led to improvements in water chemistry status and in the functioning of ecosystem dynamics.

One of the most significant studies highlighting the response of lake systems to reduced nutrient loading incorporated 35 lakes from North America and Europe (Jeppersen *et al.*, 2005). This study considered a wide range of lake types and environmental variables, which could influence and explain the observed patterns of eutrophication and identified rates of recovery. Lakes were selected to represent a trophic spectrum from oligotrophic to hypereutrophic (with mean summer TP concentrations between 7.5 $\mu g\ l^{-1}$ and 3500 $\mu g\ l^{-1}$), a range of altitudes from lowland to upland (altitude up to 481 m), climatic regimes from sub-tropical to temperate (latitude 28 to 65°) and shallow (mean depth < 5 m) to deep (mean depth up to 177m) morphometry. In the lakes that responded to a reduction in P loading, recovery was characterised by reductions in phytoplankton biomass, enhanced zooplankton grazing, the return of fish populations with an emphasis on piscivores, and an increase in the abundance of aquatic plants, particularly in shallow lakes, along with the return of annual fluctuations in plant species composition.

However, this study also demonstrated that strategies for successful restoration are not as simple as just reducing nutrient loads. As reported by Moss (1988), Scheffer (2004) and Moss *et al.* (2005), unpredictable changes in predator-prey relationships along with apparently random shifts in water chemistry have prevented the complete recovery of some lakes. The most common problem preventing sustainable lake restoration appeared to be the delayed release of sedimentary P, which can accumulate during enrichment and be released decades later into the hypolimnion from the water sediment interface by factors such as wave action (Aalderink *et al.*, 1985), benthic foraging (Meijer, *et al.*, 1989) or conditions of anoxia (Wetzel, 2001). P-resuspension can maintain a state of enrichment despite the reduction of P-input via runoff and may be most significant in shallow lakes where sedimentary P loads are already high due to previous phases of eutrophication (Søndergaard *et al.*, 2003; Welch and Cooke, 2005). Producing a viable restoration programme appears to depend on the natural balance of a system before enrichment and its inherent ability to deal with external and internal shifts in P concentration. To date little attention has been paid to developing an understanding of the thresholds of response, which lead to the turbid, algal dominated state.

1.7 UK Water Legislation with Reference to the European Water Framework Directive (EUWFD)

Since 1945 the regulation of water quality in Britain has been fragmented, with a distinct lack of emphasis on sustainable restoration and conservation of water supplies and the surrounding terrestrial environment to which they are directly linked (Table 1.1).

Legislation	**Aim**
1945 Water Act	To develop water companies and boards
1973 Water Act	To create Water Authorities
1974 Control of Pollution Act	Water Authorities responsible for issuing discharge consent for sewage effluent
1980 Water Act *	To provide clean water for domestic purposes
1985 Food and Environment Protection Act	To provide controls for disposal of waste at sea under licence
1989 Water Act	Privatisation with economic regulation.
1990 Environment Act	To adopt an integrated approach to pollution release into air, water and land
1991 Water Industry Act	To amalgamate sewage legislation
1991 Water Resource Act	To control pollution discharge to surface water
1991 EU Urban Waste Water Dir.	To sufficiently treat waste-water before disposal
1992 Private Water Supply Reg. *	To define wholesomeness by improving private supplies
1995 Environment Act	EA to assume responsibility for the NRA's and pollution.
1996 Surface Water Act *	To set standards for improving public water supplies.
1998 Groundwater Regulation	To protect groundwater supplies
1999 Water Industry Act	To control disposal of industrial and domestic effluent.
2000 Water Framework Directive	Most significant piece of water legislation so far, which integrates all previous water Acts

Table 1.1 Table outlining water legislation in Britain between 1945 and 2000 (* Scotland only) (Compiled from the Scottish Parliament Environment Office; scot.gov.uk, 2006)

Increasing ecosystem degradation and associated demands for the ecological and chemical improvement of surface water quality status, has led the European Union to issue a directive to ensure the sustainable management of water resources throughout Europe. These constitutional changes are presented under the EUWFD (Directive

2000/60/EC) (European Union, 2000), which aims to encourage public participation and reduce the social and economic impacts of pollution, whilst preventing further deterioration of water resources by promoting sustainable water consumption and improving aquatic habitats. When coupled with conservation legislation such as the EC Wild Birds Directive (Directive 79/409/EEC) (DEFRA, 2005) and the EC Habitats Directive (Directive 92/43/EC) (DEFRA, 2005), which are intended to restore, maintain and protect flora and fauna in their natural habitats, the EUWFD has great potential to improve the long-term regional management of water supplies and their associated catchments.

The fundamental objective of the EUWFD is to achieve at least *good ecological status* of standing waters throughout Europe by 2015, and achieving *good ecological potential* of heavily-modified waters (UKTAG, 2008). Initiated in 1997, the program aimed to establish environmental water monitoring networks to enable the classification of reference conditions by 2006. In Europe, this led to the development of the first large-scale, integrated monitoring programmes designed to assess water quality status and investigate the causes and effects of water pollution. Monitoring techniques included new biotic indices based on the relative abundance of microcrustaceans (Boix *et al.*, 2005; Nyman and Korhola, 2005), the comparison of present day, habitat-specific fish communities (Diekmann *et al.*, 2005) and reconstructed historical fish communities (Gassner *et al.*, 2005), general biodiversity assessment (Oertli *et al.*, 2005), and the classification of eco-regions based on minor differences in climate (Moos *et al.*, 2004; Kolada, *et al.*, 2005).

As noted above, a key concept is the non-impacted reference state by which to measure ecological impact at polluted sites, and from which to define targets for restoration. As part of this, the adoption of palaeolimnological techniques to provide snapshot proxy palaeolimnological data on the reference state prior to major recent human impact (*ca.* AD 1850), against a background of assumed *naturalness* (Bennion, *et al.*, 2005; Anderson and Odgaard, 1994; Bradshaw 2001; Amsinck *et al.*, 2003; Bradshaw and Rasmussen, 2004), was noted. By 2008 member states were required to present draft management plans to individual governments with an emphasis on the classification of surface water, based on the developed reference conditions, which will then be used as a guideline for identifying appropriate monitoring strategies and defining *good ecological status*. Management plans will be finalised by 2010 and should be in full operation by 2015 with environmental objectives attained by 2021.

In Britain it is the task of the UK Technical Advisory Group (UKTAG, 2008), made up of experts from a range of conservation agencies, specialist task teams and steering groups, to coordinate the appropriate research and development required to implement the intended changes. It is also the job of UKTAG to provide technical advice to, and receive policy advice from, The Department for Environment, Food and Rural Affairs, The Welsh Assembly Government, The Scottish Government and The Department of Environment, Northern Ireland, and to develop policy consistency pertaining to water quality in the UK. Using current knowledge of hydrology and physico-chemical and biological water quality, UKTAG has developed a classification scheme and devised a framework for monitoring based on environmental standards that support healthy communities of aquatic plants and animals. The result is an 18 class, ecotype classification system for lakes that first addresses alkalinity (low, medium, high, marl, peat, brackish) and depth (very shallow, shallow, deep), then considers local factors such as geology, altitude and type-specific features such as water residency time (UKTAG, 2008).

However, there is still little consensus as to what exactly is considered good ecological status and if conditions prior to any permanent human impact are considered to represent the baseline reference state, then little work has been done to identify when the ecological consequences of impact in a range of lake types first occurred. Examining and quantifying the differences in limnological stability and ecosystem behaviour during phases of anthropogenic stress for different lake types is therefore a subject that needs attention. Some lakes, for instance, may have a long history of human impact but the ability to return quickly to a pre-impact state, whereas others, with a history of natural susceptibility to eutrophication, may not.

In a study of the ecological succession of algae, following impact events at Lough Augher, Co. Tyrone, N. Ireland, for example, Anderson *et al.* (1990) found that the degree and duration of change depends on the natural condition of the lake before impact and the extent of damage caused by the impact event. During recovery, indicator species re-appeared only when their optimal conditions were achieved and, though patterns of eutrophication can be fairly predictable in this sense, the recovery trajectory varied following different impact events. The variations were explained in terms of changes in ecosystem structure, individual water chemistry thresholds, water residence times and the availability of certain nutrients such as P and Si (Anderson *et al.*, 1990).

Ideally, effective restoration requires long-term data to provide information on ecosystem health, natural variability and natural productivity patterns without the effects of human impact. Generating this information from a range of sites would highlight how a range of factors interact in different lake types in order to enable the definition of a realistic, attainable baseline and facilitate the maintenance of good quality status. Knowledge of individual site histories can also help to identify the exact timing of change and the possible reasons for changes in limnological conditions (Smol, 1992). A long-term palaeolimnological approach has clear potential to produce this information, which can then contribute towards developing improved restoration plans for reversing change in complex ecosystems. In contrast, short-term approaches share the assumption that systems were in a 'state of good health' prior to the Industrial Revolution or immediately before the intensification of land-use during the 1950's (Anderson *et al.,* 1990; Anderson and Rippey, 1994; Anderson, 1997; Bennion, *et al.*, 2000; Bennion *et al.*, 2002). This does not take into account any earlier phases of human impact and assumes that significant deleterious changes in water chemistry and ecosystem health have only occurred in recent decades.

Studies in Scotland (Bennion *et al.*, 2005), and Denmark (Amsinck *et al.*, 2003) used palaeolimnology to identify baseline reference conditions, but their data only covered the previous 150-200 years, and in both cases this was shown to be insufficient due to a background of long-term nutrient pollution before the nineteenth century. It was concluded that the definition of baseline conditions was unreliable. Fritz (1989), Håkansson and Regnéll (1993; 2000), Ekdahl *et al.*, (2004) and Bradshaw (2001) have already shown that earlier events can have significant and long-lasting effects.

Although it is not logistically feasible to study the long-term record for every site needing management the assumption that ecosystem change has occurred only recently needs to be tested further. Long-term, site-specific research across a range of ecosystem types and impact scenarios is needed before definition of the development of realistic reference conditions for good ecological status is attempted.

1.8 The Value of a Palaeolimnological Multi-proxy Approach

In the absence of long-term, water quality data, shifts in species presence and changes in the abundance of certain biotic indicators (e.g. diatoms, Cladocera and pollen) in lake sediment sequences can be used to infer changes in water quality and link this to climate change and human activity within the surrounding catchment for the time period represented by the core. Knowledge of the sensitivity of proxy indicators to environmental variables in the modern environment allows interpretation of changes within the fossil community and enables inferences to be made about past environmental conditions (Battarbee, 1986; Stoermer and Smol, 1999).

Important advances in the field of quantitative palaeolimnology have enabled the estimation of water chemistry variables from biological sedimentary records using weighted averaging (WA) regression and calibration techniques (ter Braak and van Dam, 1989). Surface sediment calibration sets of biological and water quality data from a large dataset of lakes spanning the environmental gradient of interest (e.g., TP, pH, salinity) have been analysed using regression techniques to generate quantitative estimates of species optima, which are used to construct transfer functions. Application of the resultant transfer function to fossil assemblage data by calibration enables quantitative estimates to be generated for key water chemistry variables and produces a picture of change over time (Smol, 2002). Palaeolimnological reconstruction can provide the information needed to examine the timing and rate of change in ecosystem stability prior to pre-pollution background conditions, and assess whether a stable baseline existed in the past and if so how modern environmental conditions deviate from this.

The physical and chemical parameters of a lake ecosystem (e.g. geology, light, temperature, climate, nutrient availability and water residency time) determine its biotic composition and structure. Because these components are closely integrated, both within- and between-trophic level changes in one parameter will automatically have an effect on the whole community (Whittaker, 1975). Ideally therefore, sustainable management and restoration plans should be based on and monitored via the examination of interactions within a system in order to incorporate the complexity of, often non-linear, relationships between components. The same principle applies to palaeolimnological reconstruction; although the analysis of one strong proxy indicator can yield valuable information about water quality status, it does not reflect the real complexities of food web interactions and changes within the immediate environment. With limited information, the natural patterns of past ecosystem behaviour and the possible links between biotic components at different trophic levels may be obscured.

A wide range of physical, chemical and biological remains can be preserved in lake sediments and the use of several of these as indicators to produce complimentary sets of data can strengthen interpretation (Battarbee, 1999) and help to evaluate a range of explanations (Birks *et al.*, 2000). The results of a multi-proxy approach can thus provide the evidence necessary to explore temporal variations in natural stability, vulnerability and general ecosystem health and produce a clearer picture of the causes and consequences of nutrient enrichment. More specifically, this approach can be used to explore interactions between trophic groups during the eutrophication process and to test rates and levels of response to varying degrees of impact. Multi-proxy reconstruction can also compensate for gaps in the sedimentary record of one key indicator where poor preservation is a problem (Lowe and Walker, 1999), although the confines of time and finance often prevent the use of many indicators (Lotter, 1998).

Each fossil reflects different aspects of the overall food web. Diatoms, which occur at the bottom of the food chain, can reflect past changes in water chemistry and rates of photosynthesis (Smol, 1990). A range of micro-crustacea (e.g. Cladocera) can respond to community changes at the zooplankton level (Hann, 1990), whilst ephippia production can indicate stress within the wider ecosystem (Sarmaja-Korjonen, 2004). Chironomids, whose head shields preserve well in lake sediments, can provide a biological indicator for both the health of an aquatic ecosystem (Walker, *et al.*, 1991) and for changes in climatic conditions. Changes in land-use, which might be the cause of limnological change, can be indicated by the presence of stenotypic species of Coleoptera (Kenward, 1975) and by pollen and plant macrofossil analysis (Faegri and Iversen, 1989). The pollen record reflects changes in both local catchment vegetation, such as phases of deforestation and agricultural activity, and the aquatic macrophyte community, which can highlight shifts in trophic state (Warner, 1990), whilst plant macrofossils mainly reflect changes in the aquatic macrophyte community.

Examination of the chemical and physical properties of the sediment matrix can also reveal information concerning shifts in the allochthonous and autochthonous organic and mineral components with regards to limnological and catchment processes (Engstrom and Wright, 1984). In this respect, palaeolimnology can meet the requirements of the EUWFD by providing long-term information on ecosystem dynamics, reconstructing changes during phases of human impact and locating the lakes baseline reference state. Additionally, this provides a compatible approach to the monitoring of modern water quality using biotic indicators such as diatoms (Yang *et al.,* 2005; Kelly *et al.*, 2008)

1.9 Theories of Ecosystem Dynamics and Ecological Functioning

1.9.1 Stability or Fluctuation?

A common assumption in lake conservation is that high ecosystem diversity is a reflection of stability and good ecosystem health. A central tenet is that a wide range of species at different trophic levels will protect the system against collapse during phases of environmental stress (Odum, 1953; MacArthur, 1955; Elton, 1958; May, 1973; Pimm and Lawton, 1978; Tilman, 1996; Naeem and Li, 1997; Naeem, 1998). In addition, Yodzis (1981) suggests that the higher the degree of interaction between trophic levels and the stronger the links in community structure, the more ecosystem integrity will be protected during disturbance. Work by Persson *et al.* (2001) supports this hypothesis by showing that during nutrient enrichment the species that are more vulnerable to predation are replaced by invulnerable prey, which in turn stabilise trophic dynamics. However, McCann (2000) points out that the removal of just one species during perturbation (known as Keystone) can lead to significant changes in community composition and in some cases may cause complete community collapse. *Daphnia* is a prime example of this as its disappearance can cause disturbance to the entire aquatic food web (Moss *et al.,* 1994).

Many of the studies based on diversity and ecosystem health are based on theoretical debate or field studies focusing on changes in terrestrial environments. Aquatic studies also show a positive correlation between diversity and stability, but reveal that the factors governing ecosystem health are far more complex than just a measure of diversity (Scheffer, 2004). The multiple interactions between trophic levels mean that freshwater systems are often unpredictable and factors such as disease or seasonal changes in temperature produce baseline states that are naturally dynamic rather than static. These long-term shifts in ecosystem dynamics, however, can produce internal stabilising mechanisms that become inherent to the system and are instrumental in retaining stability and building up ecosystem resilience by a process of reorganisation (Holling, 1973; Carpenter *et al.*, 1985; Levin, 1999). Additionally, Elmqvist *et al.* (2003) found that community expansion also increases the diversity of *responses* amongst species contributing to the same ecosystem, which is critical to ecosystem resilience, renewal and reorganisation associated with future environmental change.

In their discussion of resilience in lake ecosystems, Dent *et al.* (2002) refer to these regime shifts as *multiple states* and explain that each condition represents the response of internal processes to external perturbation and protect the system against future episodes of eutrophication. Within each state individual species regulate ecological processes and biogeochemical cycles, which help to maintain equilibrium so that ecosystem functioning and integrity is not lost during phases of impact (Peterson *et al.*, 1998). This confirms that rather than just being a function of high diversity, ecosystem stability depends on the presence of keystone species to maintain ecological function (Lawton, 1994). Peterson *et al.*, (1998) also proposed that although there are great similarities in ecosystem organisation at the trophic scale, the stability of an individual ecosystem depends on the ecological history of its catchment and the evolutionary history of interacting species.

For many lakes the presence of a healthy aquatic macrophyte community is the key to good ecosystem health. Fringing aquatic plants have the benefit of controlling algal growth by the release of allopathic exudates and maintaining water clarity by the uptake of nutrients (Lau and Lane, 2001). Additionally, Howard-Williams (1981) found that some aquatic plants are associated with certain filamentous algae that act as nutrient filters by helping to absorb excess nutrient loads during high concentrations. Filter-feeding cladocerans, which essentially keep algal growth in balance, are provided with refuge from fish predation and thus help to control algal biomass (Scheffer, 2004). Water velocity is impeded, whilst root growth prevents sediment re-suspension and nutrient release (Carpenter *et al.*, 1983). The chances of water turbidity and sedimentary P release are therefore reduced (Schiemer and Prosser, 1976) and a firm substrate is provided for plant germination (Lau and Lane, 2001). A healthy plant community also provides refuge and spawning grounds and encourages a community structure of invertebrates (Engel, 1988), macroinvertebrates such as dragonflies, stoneflies and mayflies (UK Biodiversity, 2001), zooplankton (Timms and Moss, 1994), fish (Lammens, 1989) and birds (Hargeby *et al.*, 1994).

It has been suggested that mesotrophic lakes represent the most stable of lake ecosystems due to their narrow range of nutrient loads (between 10 and 25 μg l^{-1}) and high diversity (UK Biodiversity, 2001). One of the most important features is that they support the highest macrophyte diversity of any lake type and are known to produce the highest proportion of nationally scarce and rare aquatic plants. Characteristic plants include White lotus (*Nymphaea alba*), Yellow water lily (*Nuphar lutea*) and several species of pondweed; *Potamogeton gramineus, P. filiformis, P. obtusifolius* and *P. perfoliatus* (Fossitt, 2000). The enhanced water clarity of mesotrophic lakes encourages habitat availability in the pelagic, littoral and benthic zones, which ultimately gives rise to high diversity at all trophic levels.

Naturally mesotrophic lakes are now rare due to anthropogenic eutrophication, and many have high conservation status to protect them against further degradation. Lough Neagh in Northern Ireland (Foy *et al.*, 2003) underwent a marked increase in nutrient loads from 20 μg l^{-1} in 1981 to 145 μg l^{-1} in 1999, which corresponded with a combined increase in point (sewage) and diffuse (agricultural) inputs from the lake catchment. Poor water quality was indicated by the increase in nutrient status, and despite a nutrient reduction programme in four of Lough Neagh's river catchments, trophic status has remained high. Many similar examples can be found in the literature (e.g., the Bosherston Lakes, Wales, Rees *et al.,* 1991; the Anglesey Lakes, Wales, Haworth *et al.*, 1996; Lake Windermere, England, Pickering, 2001; Lough Melvin, Ireland, Girvan and Foy, 2006; lakes in the Somerset Levels, England, Carvalho and Moss, 2006).

In these lakes the high diversity and strong connectivity between functional groups enabled them to deal with phases of impact and although nutrient enrichment has heavily affected ecosystem dynamics at these lakes, resilience mechanisms based on species richness may also help them to recover once the source of impact is removed. In contrast, the low productivity of oligotrophic lakes means that functional groups are reduced and there are fewer trophic interactions with weaker bonds, which can lead to destabilisation and collapse (McCann, 2000; Parker and Schindler, 2006; Rellstab *et al.*, 2007).

1.9.2 Ecosystem Resilience and Thresholds of Response

Schindler (1990) and Smol (2002) propose that when a system is under stress from external or internal drivers, shifts in ecosystem health are reflected as behaviour outside of the natural range of variability. Although some systems may respond slowly to environmental change, in other cases the evidence suggests the existence of thresholds that mark the border between one set of feedback mechanisms and another which, if crossed, can cause dramatic changes in state that have catastrophic effects on the entire system (Carpenter *et al.*, 1999; Scheffer *et al.*, 2001; Folke *et al.*, 2004). The paradox of enrichment proposed by Rosenzweig in 1971, for example, shows that when these thresholds are crossed normal patterns of behaviour become redundant. As enrichment at the algal level amplifies, population cycles also increase and although there is a strong relationship between high diversity and good ecosystem health, there is a point where populations are taken beyond the systems carrying capacity and community structure becomes unstable. The patterns that follow depend on the individual system but if enrichment persists the collapse of trophic interactions will gradually bring both predator and prey populations nearer to zero. A threshold may then be crossed to a regime state that can devastate ecosystem structure and functioning (Carpenter *et al.*, 1985) and it is clear that there is an increased chance of this when human activities erode the system's ability to repair and nutrient loads increase beyond a manageable capacity. However, it is also possible that the multiple states model described by Dent *et al.* (2002) can maintain a system and enable it to return to a steady cyclic state following perturbation.

From studies based on experimental work and field observation designed to understand ecosystem stability in freshwater systems, an important concept has arisen that highlights thresholds of ecological response and the role of multiple states in preventing complete ecosystem collapse. The *alternative stable states* model involves a shift from macrophyte dominated, clear water to a turbid state with high phytoplankton concentrations (Timms and Moss, 1984; Blindow, *et al.*, 1993, Moss *et al.*, 1996; Karst and Smol, 2000). As turbidity increases, plant macrophyte abundance decreases and the zooplankton that are using plants for daytime refuge to escape fish predation become vulnerable (Phillips *et al.*, 1996). The decrease in zooplankton grazing may then result in negative feedback allowing phytoplankton crops to expand and be dominated by the inedible taxa (e.g. *Aulacoseira granulata*, *Aphanizomenon*) normally characteristic of turbid conditions (Lau and Lane, 2001). The point where a system begins to destabilise is usually marked by an eventual reduction in edible prey at the algal level (Roy and Chattopadhyay, 2007).

Many studies describe shifts from plant to algal domination in response to anthropogenic stress, whilst some have even observed both states in the same lake

(Sheffer *et al.*, 1994; Telesh *et al.*, 1999). In all cases the shift to algal domination is regarded as representative of poor water quality. However, algal domination can provide a protective mechanism against complete collapse during phases of eutrophication, by maintaining photosynthesis at the phytoplankton level and protecting the more vulnerable species (Holling, 1973; Levin, 1992). Holling (1973) describes this as resilience theory and explains that ecological resilience is the amount of reorganisation it takes to transform a system from one stable state to another and to return to a steady or cyclic state following perturbation. As noted, a healthy aquatic plant community is essential for good ecosystem health and the ability to locate the boundary between these states may provide a key to maintaining good quality status (Scheffer, 1990; Scheffer *et al.*, 1993; Janse, 1997; Coops, 2002; van Ness *et al.*, 2002; Morris *et al.*, 2003; Scheffer, 2004).

1.10 Rationale for the Research

Most inland European waters are subject to human-induced pollution impacts, of which eutrophication is the most serious. In order to address the problem, the dictates of the EUWFD have led to a major expansion of regional European water quality and biomonitoring programs over the last decade. For lakes, which preserve a proxy record of past change in the sediment record, regional monitoring is increasingly being supported by large-scale palaeolimnological programmes, to provide a temporal perspective, which is otherwise lacking.

The EUWFD research methodology focuses on producing regional data-sets for ecological impact assessment and for defining the reference state of water bodies as a target for restoration based on conditions of minimal impact. For lakes, modern lake type is classified by a suite of physico-chemical parameters (e.g. geology, alkalinity, water depth), to produce regional and local typology groups. Under the typology classification, 'pristine' sites have been selected for risk assessment with the intention of examining the degree of recent change in diatom assemblages and analogue matching the results with lakes from other typology groups to compare degrees of impact and prioritise restoration plans. Using rapid palaeolimnological techniques, a statistical measure of the compositional change between 'top' (present) and 'bottom' (*ca.* AD1850) samples is estimated by detrended correspondence analysis (DCA) of proxy data-sets such as diatoms, whilst inferred ecological status in the bottom, *ca.* AD 1850 sample provides the reference state for restoration targets based on conditions of minimal impact.

Although DCA provides a simple and rapid estimate of recent (last 150 yrs.) compositional change in response to human impact, the application of the 'top-bottom' approach is based on a number of important assumptions. The first assumption is that the comparison of two single samples is a valid technique for summarising patterns of ecological change over the last *ca.* 150 years. However, this is least likely to be valid in the most complex ecosystems such as shallow, eutrophic lakes. In defining the reference state, assumption two is that the pre- industrial state represents a stable, equilibrium prior to significant human impact, even at lakes with a long-term history of human activity. In classifying lakes, a third assumption is that the 'pristine' state defined from a lake with a given set of physico-chemical characteristics may provide a useful tool for the restoration of other lakes with similar, non-biological characteristics.

Longer-term palaeolimnological research has the potential to test these assumptions by increasing our understanding of long-term ecosystem dynamics at a range of lake types. Due to their complexity, meso to eutrophic lakes are an appropriate focus for this study. To date, most high-resolution palaeolimnological research has focused on relatively short-term environmental change (*ca.* 150 yrs. BP) (Anderson and Odgaard, 1994; Bennion and Appleby, 1999; Bennion, 2001; Bennion *et al.*, 2001). While these studies are valuable for testing the degree to which the 'top and bottom' approach is a reliable reflection of change over recent timescales (assumption 1), they do not address the issue of long-term ecosystem stability and human impact (assumption 2), and the validity of lake classification (assumption 3).

The top-bottom approach has focused on diatoms, which are well recognised as sensitive proxy indicators for key water chemistry parameters such as total phosphorus (TP), pH and salinity. In attempting to improve our understanding of the ecosystem as a whole, it is also well recognised that the multi-proxy approach, combining a range of palaeolimnological indicators, may strengthen interpretation of past environmental change.

An additional drawback of much palaeolimnological pollution-impact research on the recent timescale is that it does not take advantage of changes in catchment dynamics through analysis of proxies such as pollen and charcoal, which may be transported from the catchment to the lake sediment. Similarly, while many long-term pollen studies do focus on the effects of human impact within terrestrial environments (e.g. Gaillard *et al,* 1992; Walker, 1993; Dumayne and Barber, 1994; Harmata, 1995; Stevenson *et al,* 2001), there are still few studies that link these changes to corresponding impacts on the aquatic environment (Bradshaw, 2001; Fritz, 1989; Håkansson and Regnéll, 1993). Long-term human activity within lake catchments and its direct effect on ecosystem health is therefore still poorly understood.

Although there is no doubt that 20th century human activities have led to accelerated eutrophication, permanent limnological change due to human processes may have first occurred many centuries ago. The long-term ecology of the natural baseline state needs to be explored in order to provide realistic ideas about sustainable restoration targets and the future ecology of surface waters. It is only by examining long-term ecosystem dynamics that the validity of the assumptions surrounding definition of the reference state may be tested. To date, relatively few studies have concentrated on the dynamics of long-term (Holocene) eutrophication. Those that have tend to focus on lakes which are currently hypereutrophic, and results have shown that permanent changes in ecosystem status have often taken place well before this date (Amsink *et al.*, 2003; Bradshaw and Rasmussen, 2004; Bennion *et al.*, 2005).

In this project, a long-term multi-proxy palaeolimnological approach is adopted to reconstruct the dynamics of changing water quality and ecosystem status in two moderately shallow eutrophic lakes, and to examine the underlying patterns of ecosystem response to external impact. The concept of a stable baseline state is tested and rates of change during periods of natural and anthropogenic environmental change are assessed. To achieve this, qualitative and quantitative palaeolimnological, multi-proxy data are compared to Holocene proxy data for natural and anthropogenic impact over millennia rather than decades or centuries. More specifically, the concept of long-term equilibrium prior to major (post-AD 1850) human impact is tested (Smol, 1992; Birks *et al.*, 2000). The study will thus produce data for determining whether the concept of the pre-1850 AD reference state is valid. By analysing impacts and response in two closely-adjacent sites of highly similar physico-chemical limnological characteristics, the validity of lake EUWFD classification systems may also be tested.

1.11 Aims

1 In two moderately nutrient-enriched sites which would be classified as highly similar in the context of WFD guidelines, the aim is to reconstruct mid- to long-term ecosystem dynamics over the long (10^2 to 10^3 years) timescale, since the early- to mid-Holocene,

2 To explore the degree to which observed limnological change has been driven by human activities rather than natural (ontogeny or climate-induced) change prior to the documented intensification of water pollution following the Industrial Revolution,

3 To test the validity of the single-sample pre-1850 AD definition of the reference state (assumption 1) and the concept of a stable equilibrium baseline prior to *ca.* AD 1850 (assumption 2),

4 To test the validity of site classification (assumption 3) by comparing and contrasting thresholds of ecosystem response, stress and resilience between the two sites during the process of eutrophication,

5 To use the results to inform the definition of realistic sustainable plans for maintaining good ecological status, as outlined by the dictates of the EUWFD,

1.12 Objectives

1 To select two closely-adjacent study sites of similar catchment and limnological characteristics, with similar modern catchment land-use characteristics,

2 To obtain a long (>6m) sediment core from at least one of the chosen sites using an appropriate combination of coring techniques,

3 Within the confines of a PhD, to use an appropriate range of proxy indicators to reconstruct:

   a. physical, chemical and biological change within the lake ecosystem,
   b. different trophic levels in the past ecosystem, and
   c. changes in human land-use activity in the catchment.

4 To collate extant local archaeological and documentary data on the chosen sites as an independent record of past human land-use within the catchment.

5 To carry out integrated data analysis in order to achieve Aims 1-5, above.

CHAPTER 2

THE PROXY INDICATORS

2.1 Introduction

A range of indicators have been chosen to achieve the aims and objectives outlined in Chapter 1. In combination, they have the potential to provide strong proxy data on changes in the lake catchment and associated changes in water quality and ecosystem status, at both the long and short time scales. Pollen and charcoal have been selected to identify impact within the lake catchment and to evaluate lake productivity status, whilst a range of biological indicators are chosen to represent the plant macrophyte, phytoplankton and zooplankton levels of the food chain and to define ecosystem complexity and the patterns of ecosystem dynamics related to the process of eutrophication.

2.2 Stratigraphic Description

The Munsell Soil Colour Chart (MDKI, 1994), first developed by the US Soil Conservation Service to classify soil types, provides a rapid and consistent means of describing changes in basic sediment colour and composition. The Troels-Smith (1955) classification scheme, designed for the rapid field analysis of peat, is used for detailed description of sediment structure. Results can be summarised using a system of standardised symbols representing physical features, such as colour and stratification, humicity and sediment structural composition, including the origins of organic components. However, the scheme can be complex and difficult to translate for non-users. Digital photography can also provide a permanent record of any stratigraphic changes throughout the core before sediments are sub-sampled for analysis.

2.3 Magnetic susceptibility

This non-invasive method is used to infer relative changes in ferruginous mineral concentrations in sediment sequences. Since the magnetic qualities of deposited minerals remain unaltered following deposition, their magnetic susceptibility can be quantified, providing an indication of relative changes in catchment erosion over time (Dearing, 1999). Results are presented as susceptibility, which is a measure of the magnetic receptivity of a sample (Nowaczyk, 2001). One of the first successful uses of this technique was at Lough Neagh in Northern Ireland (Thompson *et al.*, 1975), where higher values correlated closely with evidence of land clearance and increased sediment accumulation rates. Although the integrity of these results means that the method has become widely used to identify changes in land-use and catchment inwash (Dearing and Flower, 1982; Anderson and Rippey, 1988; Snowball and Thompson, 1988; Dearing, 1991; Maher and Thompson, 1995), the method does have some limitations.

Blakemore (1975) discovered that fine-grained magnetite was internally metabolised by a certain type of soil bacteria, which used it for navigation. This process produced high magnetic susceptibility values in samples containing the bacteria, which has the potential for being interpreted as increased catchment erosion. Additionally, the post-depositional dissolution of magnetite can cause a reduction in magnetic susceptibility, thus producing an underestimation of the erosion signal (Hilton and Lishman, 1985). Nevertheless, magnetic susceptibility is a widely used, non-destructive, cost effective proxy and results can be produced relatively quickly.

2.4 Loss on Ignition

The organic content of sediment cores is composed of autochthonous material from aquatic plants and phytoplankton, and allochthonous material from the inwash of catchment vegetation (Meyers and Teranes, 2001). Combustion of sub-samples at 550°C can provide an assessment of the total percentage content of organic carbon (Dean, 1974). Further combustion at 850°C provides a measure of inorganic carbonate (Dean, 1974) by molar conversion of CO_2 loss. The residue then comprises non-carbonate clay minerals, feldspar minerals and quartz, etc. (Smol, 2002). In theory, a more robust interpretation of catchment erosion can be produced when the results are compared to the magnetic susceptibility profile. Both techniques provide a fast, inexpensive and relatively accurate means of quantifying sedimentary composition, which compares well with other, more complex approaches (Dean, 1974).

The limitations of these methods are reviewed in Sutherland (1998), Heiri *et al.* (2001) and Boyle (2004), whilst Smol (2002) points out that because LOI measurements are expressed as a percentage, an increase in one component will always lead to a decrease in the other so that when lake productivity *and* catchment erosion are both high a true reflection of sediment composition is not given. Although many studies have shown that organic measurements are an important component of down core studies (Gaillard *et al.,* 1991; Meyers and Ishiwatari, 1993; Anderson and Rippey, 1994; Dean, 1999), LOI does not identify the primary source of organic production. This means that the dominant depositional process is not identified and periods of high autochthonous productivity during periods of eutrophication cannot be separated from periods of high organic inwash from catchment activities such as deforestation.

2.5 Carbon/Nitrogen Ratio (C/N)

Changes in C/N can provide proxy data on the origin of sedimentary organic biomass and identify changes in the sources of sedimentary organic material (Hollander *et al.,* 1992; Hodell and Schelske, 1998; Meyers and Lallier-Vergès, 1999; Sampei and Matsumoto, 2001; Fellerhoff *et al.*, 2003). Aquatic and terrestrial plants have different ratios of carbon to nitrogen (algae contains a higher proportion of nitrogen than land plants), which is usually measured as a value representing the weight ratio between the two elements (Meyers and Teranes, 2001). Thus, C/N values falling between 4 and 10 typically represent algal production, whereas land plants tend to provide ratio values of over 18. Ratios that fall between 10 and 18 are more common in lake sediments, indicating a mixture of both allochthonous and autochthonous plant matter and provide a baseline from which to interpret shifts in either direction.

Sarazin *et al.* (1992) found that the C/N of algal biomass can increase as nitrogen-rich proteins become degraded, especially in highly productive lakes such as Aydat Lake, France, where C/N values increased as a result of down-core de-nitrification.

Conversely, Meyers *et al.* (1995) found that the selective degradation of carbon-rich sugars and lipids in plant matter caused the C/N ratio of sedimentary wood to be lower than in fresh wood. However, any changes in the C/N due to post-depositional changes in the elemental composition of organic material are usually small compared to the stratigraphic shifts displayed by terrestrial inwash (Kaushal and Binford, 1999) and primary production (Brenner *et al.*, 1999). Additionally, Boutton (1991) found that C/N analysis was inadequate for making specific inferences about shifts from a macrophyte to an algal dominated system as freshwater macrophytes display such a wide range of C/N values.

2.6 Carbon Isotope ($\delta^{13}C$) Ratio

Changes in the isotopic carbon ratio, generally written as $\delta^{13}C$ or $^{13}C/^{12}C$, can provide a proxy for shifts in palaeo-productivity. As the sedimentation of C_3 algal material removes ^{12}C from dissolved organic carbon (DIC) in surface waters, the heavier ^{13}C remains and deposited organic matter subsequently displays a lighter $^{13}C/^{12}C$ ratio than the DIC source (Wolfe *et. al.*, 2001). During periods of increased algal sedimentation the $\delta^{13}C$ value of the remaining inorganic carbon available for algal growth rises, producing higher $\delta^{13}C$ values in newly produced material. Thus, increases or decreases in the $\delta^{13}C$ of sedimentary organic matter are positively correlated with changes in algal productivity (Meyers and Teranes, 2001).

When studying eutrophication at subtropical lakes in Florida, USA, Brenner *et al.* (1999) found that changes in diatom assemblages associated with raised productivity and measurements of total phosphorus (TP) corresponded closely with increases in

$\delta^{13}C$. Similarly, palaeolimnological reconstruction, also of lakes in Florida (Gu *et al.*, 1996), revealed a strong relationship between high planktonic $\delta^{13}C$, raised nutrient levels and increased Chl *a* (chlorophyll *a*) in surface sediments during periods of increased productivity suggesting that $\delta^{13}C$ analysis may be a reliable proxy for phases of enrichment.

During the process of oxidation, however, the release of ^{12}C from the decay of organic carbon can also produce high $\delta^{13}C$ values, whilst the anoxic conditions indicative of hypertrophic waters can produce relatively low values (Meyers and Teranes, 2001). Additionally, $^{13}C/^{12}C$ of DIC in lake-water is controlled by factors such as the exchange of atmospheric CO_2, rates of photosynthesis and respiration, carbonate weathering and the decomposition rate of different types of plant matter (Brenner *et al.*, 1999). As the sedimentation of C_3 algal material removes ^{12}C from DIC, changes in any of these parameters could potentially result in shifts in the $\delta^{13}C$ profile that may be interpreted as changes in rates of primary production.

Analysis of the $\delta^{13}C$ profiles for biota from four Canadian Shield lakes (France, 1995) shows that broad differences between the $\delta^{13}C$ signatures of littoral and pelagic primary producers may also be misleading in terms of lake productivity. It is thought that the pelagic-littoral differentiation of carbon flow between the two groups is due to the relationship between the diffusion resistance of CO_2 in water and water turbulence, i.e. under conditions of high turbulence the diffusive boundary layer resistance of algae is lower (Osmond *et al.*, 1981). Thus, in the benthic zone where water turbulence is generally low compared to the pelagic zone, primary producers will become enriched in ^{13}C relative to ^{12}C. Changes in the carbon isotope ratio may therefore reflect shifts in the availability of benthic/pelagic habitats rather than changes in nutrient enrichment.

Ultimately, care must be taken when interpreting the data and comparisons should be made between the C/N profile and multiproxy indicators in order to increase the confidence with which one may use these techniques to infer phases of eutrophication.

2.7 Diatoms (Bacillariophyceae)

Diatoms are unicellular algae, whose amorphous, hydrated, siliceous frustules often preserve well in lake sediments. These centric or pennate frustules are characterised by perforations and reticulations which, when observed in detail under a high-powered microscope, enable them to be identified, in most cases, to species level (Barber, 1981). The preference of each species for specific environmental variables (e.g., water chemistry and light availability) and habitat availability (e.g., benthic, pelagic or littoral) make them powerful environmental indicators.

Past limnological conditions can be reconstructed quantitatively by using transfer functions. This approach has been used to great effect to investigate a wide range of research questions including lake acidification (Battarbee, 1984; Charles *et al.,* 1987), changes in sea level (Palmer and Abbot, 1986; Zong and Tooley, 1996) and shifts in salinity as a reflection of climate change (Reed *et al.*, 2001). Phosphorus (P) is the most important limiting nutrient to algal growth in many lakes and has major significance in the eutrophication process. Past epilimnetic P-concentrations can be reconstructed using a diatom-inferred total-phosphorus (DI-TP) transfer function and changes in reconstructed DI-TP may then be used to infer changes in P loads. With strong chronological control, this information can be compared with catchment history to trace the possible causes of change in the nutrient budget.

Modern datasets of lake chemistry and diatom assemblages have been combined and harmonised in the Environmental Diatom Database Initiative (EDDI) (Battarbee *et al.*, 2000). Transfer functions derived from individual training sets have been validated for a number of lakes. Anderson (1997), for example, used the Northern Irish transfer function to reconstruct DI-TP in six rural lakes in Northern Ireland, whilst Bennion (1995) reconstructed the nutrient histories of three hypereutrophic lakes in Anglesey using the Welsh transfer function. Likewise, Marchetto and Bettinetti (1995) successfully utilised the combined European transfer function to reconstruct the phosphorus histories of two deep sub-alpine lakes in Italy. Although not infallible, Hall *et al.* (1997) have shown that reconstructed environmental parameters can compare well with modern water chemistry records.

Whilst DI-TP models have made considerable progress in recent years, they are constantly re-evaluated and refined in order to improve their performance. Birks (1998), for example, reviews the need for robust statistical methods to construct the non-linear unimodal relationship between species and their environment with reliable estimations of the standard errors of prediction, whilst Rippey *et al.* (1997) discuss the need for further validation of transfer functions, and suggest that increasing the number of lakes within modern calibration datasets would improve species representation and lengthen the variable gradients. Fritz *et al.* (1993) suggest that care should be taken when studying long time periods where minor, yet important, fluctuations in phosphorus concentrations can be lost. From a taphonomic point of view, poor preservation of diatom frustules can destroy the record and bias the reconstruction. Dissolution may occur under highly alkaline conditions, in which case the fossil assemblage may favour the stronger, heavily silicified species (Round, 1964; Flower, 1993). Problems may also occur with the identification of individual species that are visually very similar in structure, yet have different water chemistry preferences.

2.8 *Pediastrum*

The genus, *Pediastrum*, is a non-motile, radially symmetrical, coccal green algae, the flattened cells of which form star-like colonies in numbers genetically determined for each species. Cell wall structures and morphological modifications are specific to each species enabling taxonomic identification to species level (Komárek and Jankovská, 2001). The exine of *Pediastrum* is composed of resistant sporopollenin, which protects colonies against desiccation and microbial attack, enabling their long-term preservation in lake sediments and allowing them to survive harsh pollen preparation techniques (Lowe and Walker, 1999). The remains of *Pediastrum* colonies can be abundant in limnic sediments, often surpassing the concentration of pollen grains (Jankovská and Komárek, 2000), and providing an additional indicator of phytoplankton composition.

In recent years palynologists have begun to recognise the potential of *Pediastrum* for palaeoecological reconstruction (van Geel, 1986). Although few studies have used *Pediastrum* as an indicator for eutrophication both Huber (1996) and Eilers *et al.*

(2004) suggest that their presence in exceptionally large numbers could be an indication of changes in productivity. However, *Pediastrum* analysis is still in its infancy and there are only two algological identification keys available to separate *Pediastrum* into its different species and provide descriptions of the habitat niches and indicative environmental tolerances (Jankovská and Komárek, 2000; Komárek and Jankovská, 2001).

2.9 Cladocera Ephippia

The presence and/or absence of Cladoceran ephippia (egg sacs), which preserve well in lake sediments due to their protective, chitinous carapace, can be used as a qualitative proxy of stress within in the zooplanktonic component of the food chain. Under normal conditions Cladocera reproduce by parthenogenesis and eggs are kept in the brood chamber of the parent individual where they grow and eventually hatch (Langdon Brooks, 1959). During times of stress (such as food shortage, lake-level change, climate change and possibly eutrophication) ephippia are produced by gamogenesis (Frey, 1982; Fryer, 1972). The eggs are produced and fertilised sexually in the brood chamber then become surrounded by a hardy protective carapace, the ephippia, which separates from the female during seasonal moulting. These resting eggs become part of the sedimentary sequence, where they are capable of dormancy over long time periods until circumstances are more suitable for their continued development (Pennack, 1989).

It is common to find a small number of ephippia in most temperate lake sediments as seasonal temperature changes cause ecosystem stress. However, the increases in abundance during phases of stress caused by human impact or long-term climate change are usually far greater. In sedimentary records from lakes Kaksoislammi and Rutikka in southern Finland (Sarmaja-Korjonen, 2004) and Bølling Sø in Denmark (Bennike *et al.*, 2004), increases in the abundance of ephippia correlated well with known phases of extreme cold, whereas, during the rapid warming of the early Holocene ephippia were absent. At lakes Kaksoislammi and Rutikka, however, additional sporadic increases in ephippia, indicative of significant changes in predator-prey relationships, were most likely to be explained by anthropogenic eutrophication.

This method therefore has potential for identifying periods of ecosystem stress associated with human impact, although observed changes in ephippia abundance during eutrophication reflect changes in predator-prey relationships resulting from shifts in trophic status rather than being a direct response to trophic change (Slusarczyk, 2001). Although this method is not as useful as the taxonomic analysis of Cladoceran body parts, which can indicate changes in habitat availability as a result of changes in water chemistry status, it does provide rapid evidence of biological response at the zooplankton level of the food web. However, little research has been done to compare modern analogues with the presence and abundance of fossilised ephippia, so their application as a proxy indicator of environmental change in palaeolimnology is currently in its early stages.

2.10 Pollen

Pollen analysis (palynology) is a well-established technique for reconstructing environmental change within lake catchments, and can be a strong proxy for human occupation and changes in land-use. In addition, regionally synchronous events such as the *Ulmus* decline (Davis, 1981) can be used as biostratigraphic markers for indirect dating. The outer exine of each pollen grain consists of tough, waxy sporopollenin (Lowe and Walker, 1999), which allows unfertilised grains to preserve in sediment layers. Each taxon has unique morphological and structural features such as germinal apertures and bladder sacs (Moore *et. al.,* 1991), which enable identification by comparison with pollen keys, photographs and reference slides.

Based on uniformitarian principles, fossil pollen assemblages can be interpreted in terms of climate change (Faegri and Iversen, 1989), deforestation (Walker, 1993) and agricultural practices (Behre, 1986). Additionally, patterns of aquatic macrophyte growth, which can change in abundance and composition with eutrophication, can be examined and assessed in terms of ecosystem health (Boutton, 1991).

Apart from the differential preservation of pollen grains, the most prominent limitation to the interpretation of pollen diagrams is that of differential pollen production between different taxa. Self-pollinating (autogamous) plants with cleistogamous flowers, which rarely or never open, release very little pollen and are therefore under-represented in pollen diagrams (Lowe and Walker, 1999). Conversely, some wind-pollinated species can be over-represented resulting in interpretations that suggest a higher number of trees than reality (Andersen, 1973).

However, correction factors have been developed to overcome the marked differences in the amount of pollen produced by different plants (e.g. Andersen, 1973; Sugita *et al.*, 1999: Broström *et al.*, 2005)

2.11 Charcoal

Charcoal analysis can provide a valuable complement to palynological inferences of human activity. Analysis assumes that charcoal particles are carried into the air by radiated heat as carbonaceous material is burnt. Particles of varying size become dispersed and can be carried to a lake basin by water or wind, depending on distance of the fire from the lake and meteorological conditions (Tolonen, 1986). Charcoal particle abundance can thus be used as a proxy indicator for the presence of fire, which is often a secondary indicator of local habitation (Burden *et al.,* 1986).

Although particles can sometimes become crushed during the sieving process (Bennett *et al.,* 1990) charcoal generally survives the pollen preparation techniques. When particles are counted and the results are compared to the pollen data, peaks in charcoal abundance and corresponding reductions in arboreal pollen can represent human induced forest burning (Davies *et al.,* 1971; Tolonen, 1978). Smoother curves in the charcoal record can indicate small-scale burning (Burden *et al.,* 1986), human habitation (Tsukada *et al.*, 1986), the accumulation of airborne charcoal from the larger regional catchment, or the burning of small lakeside fires (Griffin and Goldberg, 1979). More recently the presence of carbonaceous particles in lake sediment cores has been explored as a method for examining the onset and effects of industrial activity and the combustion of fossil fuels, though the methods for preparation and analysis are different from charcoal (Wik *et al.*, 1986).

The use of charcoal as an indicator of fire was first carried out at Ordrup Mose in Denmark (Iversen, 1941). Charcoal analysis was widely used by the 1960's (Hutchinson and Goulden, 1966; Tsukada and Deevey, 1967; Davies, 1967); by the 1970's, Waddington's (1969) work at Minnesota encouraged the development of more quantitative approaches (Hope and Peterson, 1976; Byrne *et. al.*, 1977; Tolonen, 1978; Clark, 1982). Clark's (1982) point count estimation is the most accurate, simple and rapid method, although the main limitation is that the area scanned on the slide is relatively small compared to the sample volume, and results are extrapolated to the whole sample. Additionally, micro-charcoal data can include particles from a wide source area (Clark, 1990).

2.12 Sediment Accumulation Rates

The sediment accumulation rate (SAR) is the degree at which allochthonous and autochthonous sediments from numerous sources accumulate at the lakebed within a given time (e.g. cm^2 yr^{-1}). Changes in the SAR can be observed in relation to increases in catchment erosion during climate change (Molnar, 2004) and human activity (Colman *et al.*, 2000), and increases in the deposition of material from within-lake productivity during phases of eutrophication (Sapota *et al.*, 2006). Fluctuations in accumulation rates can be compared to other multiproxy data to infer the components responsible for observed changes. Values are usually derived from the age-depth curve, although this assumes constant rates of accumulation between dates and when only a few dates are available the estimation of changes in SAR may not be reliable.

2.13 Radiocarbon Dating

Radiocarbon dating provides a temporal framework to enable the comparison of shifts in nutrient status and rates of limnological change with known phases of climate change and human occupation, in relation to the pollen record and documented archaeological and historical events. Two radiocarbon dating methods are available:

a) The decay counting of ^{14}C in bulk samples, using gas proportional or liquid scintillation counting methods, which measures the departure of $^{12}C/^{14}C$ from an equilibrium value (Smol, 2002). This method requires 5-10g dry sediment, which due to its size typically contains organic matter from a number of sources.

b) Accelerator Mass Spectrometry (AMS), based on counting to estimate the $^{12}C:^{14}C$ ratio, can produce age estimates from samples as small as a few milligrams, reducing analytical time to only a few hours compared to weeks. This has increased the range of dateable materials to include items such as conifer needles (Zbinden, et. al., 1989), insect assemblages (Hedges *et. al.,* 1989; Elias and Toolin, 1990) and pollen grains (Brown *et al.,* 1989; Regnéll, 1992), although the technique is expensive.

There are several problems associated with the radiocarbon dating of lake sediments. The in-wash of terrestrial material, which often contains a mixture of old and young organic matter or humic acids, can produce ages older or younger than expected (Olsson, 1968; O'Sulivan *et. al.,* 1973; Björck and Håkansson, 1982; Hedenström and Risberg, 1999). Bioturbation can pull younger sediment down into the lower layers, disturbing the sequence (Björck and Wohlfarth, 2001), whilst root penetration can also affect the overall age of bulk samples (Kaland *et al.*, 1984). A lake reservoir effect may occur if the $^{14}C/^{12}C$ in aquatic plant tissues is lower than the $^{14}C/^{12}C$ of atmospheric CO_2, or where carbon dissolved from carbonate rich bedrock becomes incorporated into the lake water and thus into submerged aquatic plants during photosynthesis (Peglar *et al.,* 1989).

To produce an age-depth model a few dates are usually obtained for each core and supplemented by indirect dating *via.* the pollen record, or biostratigraphical correlation with cores either from the same lake or from lakes in the same region (Telford *et al.,* 2004). Age estimates for depths where dates are not available are usually estimated by linear interpolation. However, this method assumes a constant sediment accumulation rate between points, which may not be the case (Bennett, 1994a). In cases where few dates are available the use of stratigraphic markers such as the Elm decline (Huntley and Birks, 1983) can strengthen the chronology.

CHAPTER 3

THE STUDY REGION AND DESCRIPTION OF THE STUDY SITES

3.1 Introduction

Selection of the study area was based primarily on the availability of the Upper Talley core from Aberystwyth (University of Wales), which potentially provided a full Post-glacial and Holocene sequence for the Carmarthenshire region of Wales. The lack of recent palaeolimnological research in South Wales when compared to the north was also noted. The second study site, Llyn Pencarreg, was chosen due to its proximity to Upper Talley (10k north), which provided the opportunity to standardise environmental variables between the two sites. There was also a distinct lack of previous palaeoecological and palaeolimnological research at Llyn Pencarreg. Upper Talley and Llyn Pencarreg therefore provided the ideal opportunity for the environmental reconstruction and ecological comparison of two sites with apparently similar environmental characteristics, and also to expand palaeolimnological research for this part of Wales.

This chapter first describes geology and glacial landforms for Wales, and present an overview of changes in Holocene climate, landscape and human occupation for the British Isles with particular reference to studies from Wales. This is followed by individual descriptions of the selected study sites with an overview of their catchment character, lake size, and the archaeological evidence for human-landscape interactions in the catchment. The archaeological record is considered at the catchment level in

Chapters 6 and 7 in relation to direct human impact and potential causes of limnological change at the study sites.

3.2 Geology

Differences in the bedrock geology of the UK have produced a range of lake types that respond differently to changes in environmental conditions depending on their specific characteristics. As water runs over the Earth's surface, ionic crystals pass into solution by binding with the ions in water (Summerfield, 1999), which are then carried to surface water bodies. In this respect bedrock type can influence the nature of lake water and define its physical character. In England, for example, lakes in limestone catchments (e.g. parts of the Lake District, Cumbria) are more prone to enrichment than lakes in granitic catchments, due to the increased levels of carbon dioxide, produced by the dissolution of $CaCO_3$ from bedrock (Moss, 1988). These lakes often become hypereutrophic due to heavy catchment activity and constant long-term use (Moss *et al*, 1997). Scotland on the other hand is dominated by hard rock, granitic geology (e.g. the uplands of Galloway) and here lake waters are naturally prone to acidification, which has been exacerbated by acid precipitation (Battarbee *et al*, 1990).

Wales has a complicated geology composed of both hard and soft bedrock that represent several successive geological stages (Figure 3.1). South Wales is mainly composed of calc-alkaline type, Pre-Cambrian, volcanic and plutonic rock that correspond with the Uriconian Volcanic Series. These extend as far east as Norfolk, suggesting an extensive arc of volcanic activity across the south of Wales and Central

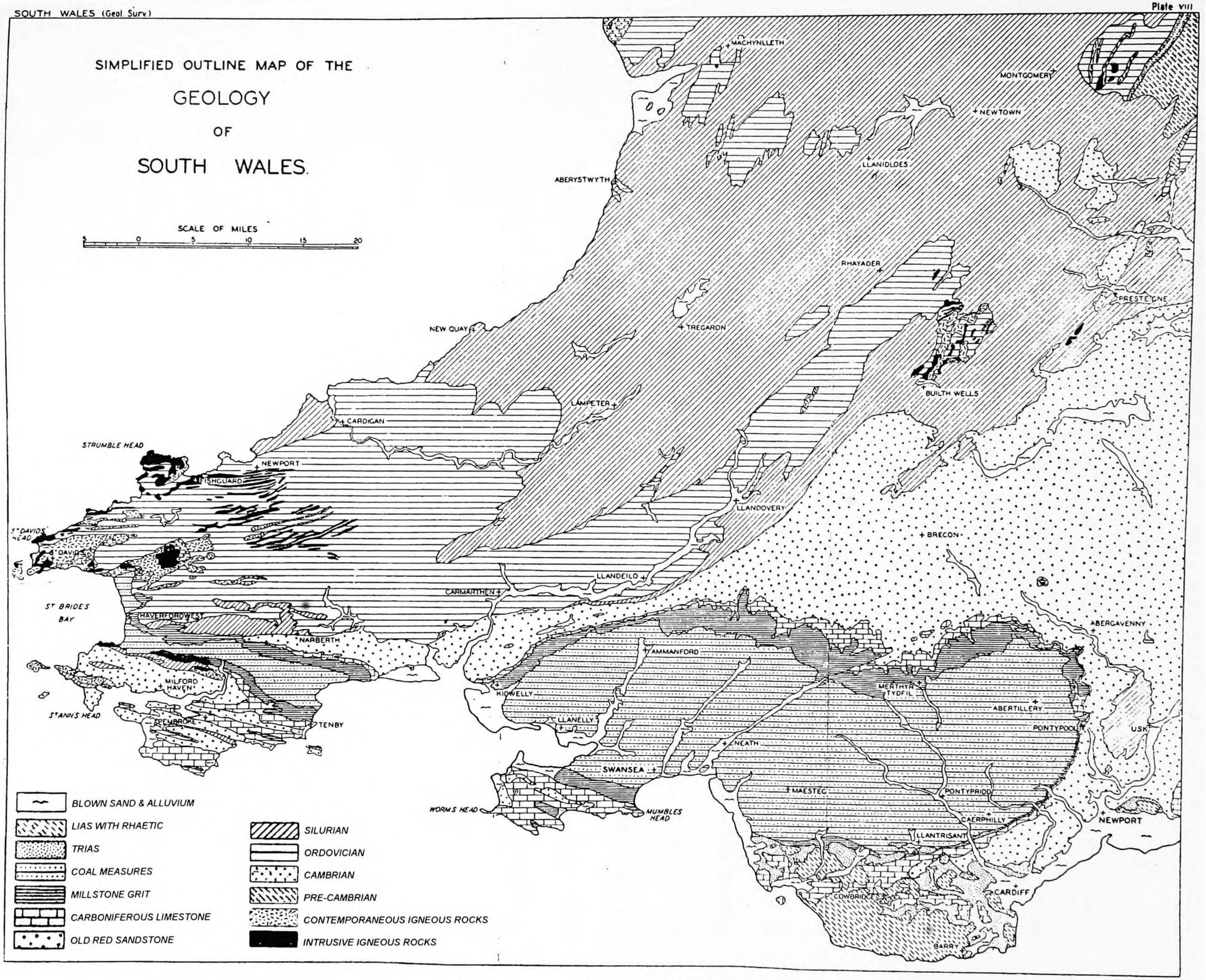


Figure 3.1 Diagram showing the geological stages of rock formation in South Wales.

England *ca.* 6 - 7 million years ago (Piasecki *et al.*, 1980). The overlying Cambrian sequence, which contains the earliest records of marine life in Wales, was formed *ca.* 5mya when the sea extended its boundaries across parts of western Wales. This material now appears as rock outcrops through the more recent Ordovician and Silurian muds and silts. Above this, Devonian Old Red Sandstone features such as the Gower Peninsula and the Brecon Beacons are followed by the Millstone Grits and Coal Measures of the Lower Carboniferous (Pringle and George, 1948).

Repeated regional oscillations and compressions have greatly modified the original patterns of geological deposition throughout the whole of South Wales and as a consequence the rock has produced stratigraphic sequences that are individual to each area. These regional differences have given rise to a wide range of lake types including oligo-mesotrophic lakes. The latter lake type can be naturally eutrophic but are often situated in rural catchments with relatively low population densities. As a consequence they provide sites ideal for the investigation of ecosystem stability at locations where human impact is not particularly extreme and possibly temporary in nature.

3.3 Glacial Landforms

According to ice margin maps, during the height of the Devensian glaciation (*ca.* 20 k cal. yrs. BP) south Wales was at the southernmost limit of the ice sheet (Bowen, 1978). Radiocarbon dated glacial deposits and oxygen isotope evidence from continental margins and marine cores show that during the Late Devensian (between *ca.* 20 and 12 k cal. yrs. BP) the British-Irish ice sheet and Welsh Ice Cap, which covered most of Wales, fluctuated several times due to minor climatic oscillations (Bowen *et al.*, 2002). These oscillations appear to correspond with a series of Heinrich events (i.e. periods of climatic warming when large icebergs broke off from glaciers and traversed the North Atlantic; Alley and MacAyeal, 1994), followed by the re-advancement of ice during the Younger Dryas (*ca.* 12.8 to 11.5 k cal. yrs. BP) and subsequent retreat as temperatures finally increased during the climatic amelioration of the Late-Glacial.

These climatic oscillations caused a high degree of glacial and periglacial activity in southern Wales, resulting in the development of a number of glacial landforms. The prominent south Wales end-moraine, which extends across the south coast (Charlesworth, 1929), for example, marks the southern limit of the Welsh Ice Sheet. Lateral melt-water channels and terminal and recessional moraines reflect several ice margin retreats (Ellis-Gruffydd, 1977; Lewis and Thomas, 2005) that correspond with the glacial fluctuations suggested by Bowen *et al.* (2002). In addition, major troughs caused by at least four ice streams mark the retreat of the Welsh Ice Cap from this area (Jansson and Glasser, 2005). On the south west coast sediment sequences from a number of proglacial lakes indicate their development in dammed river valleys as the Irish Ice Sheet re-advanced into Wales during the Younger Dryas (e.g Llyn Teifi; Etienne *et al,* 2006). The formation of glacial kettle lakes further inland (the focus of this study), are the result of large areas of dead ice breaking away from the edge of the Welsh Ice Cap during its final retreat (Eyles and McCabe, 1989).

3.4 Post-glacial/Holocene Climate Change

The Blytt-Sernander classification scheme divides Holocene climate change in north-western Europe into four distinct units; Boreal (10.0 k – 7.5 k cal. yrs. BP), Atlantic (7.5 – 5.0 k cal. yrs. BP), Sub-Boreal (5.0 – 2.5 k cal. yrs. BP) and Sub-Atlantic (2.5 k cal. yrs. BP – present). Based on observed changes in the colour and texture of sediment layers in Danish peat bogs, Blytt (1876) proposed that the dark layers containing macrofossils were produced during dry, continental periods, whilst the lighter layers with low organic content were produced during wet oceanic periods. With the addition of a firm chronology and comparison with detailed paleoclimatic records (e.g. mire stratigraphies and lake level studies; Sernander, 1908) and ice margin reconstructions from varved clays in Scandanavia (De Geer, 1912) the system is recognised as the standard Post-glacial sequence for climate change in north-west Europe (Table 3.1). The additional biostratigraphic pollen divisions constructed for Sweden (von Post, 1916) and Denmark (Iverson, 1954), identify the major vegetation type for each unit according to the prevailing climatic conditions.

Holocene palaeoclimate studies focusing on the British Isles tend to support these general, pre-determined divisions; despite regional differences in vegetation type that compare with present-day differences, the reconstruction of post-glacial forest succession for the British Isles (e.g. Hibbert and Switsur, 1976; Godwin, 1940) display broad synchroneity with the Scandanavian model. In Wales pollen profiles from Snowdonia, (Ince, 1983), Llanilid, Glamorgan (Walker and Harkness, 1990), the Elan Valley, Cardiganshire (Moore, 1970) and Gors Geuallt, Caernarvonshire (Crabtree, 1972), for example, all indicate that at the end of the Younger Dryas

Radiocarbon Years BP		Blytt-Sernander Climatic Period	Climatic Description	Dominant Vegetation
1000- 2000-	Post-glacial/Holocene	**SUB-ATLANTIC** (*ca.*2500 cal. yrs. BP)	Cool and humid	Spread of grasses and *Fagus*
3000- 4000- 5000-		**SUB-BOREAL** (*ca.* 5000 – 2500 cal. yrs. BP)	Warm temperatures and moist conditions due to changes in atmospheric circulation.	Decrease in *Ulmus* at 5k cal. yrs. BP and increase in *Quercus*, *Tilia* and *Alnus*.
6000- 7000-		**ATLANTIC** (*ca.* 7500 – 5000 cal. yrs. BP)	Summer temperatures increased to the maximum of the entire Holocene (Climatic Optimum). Mild winter temperatures with high rainfall.	*Quercus*, *Tilia* and *Alnus*. Initial formation of raised bogs.
8000- 9000- 10000-		**BOREAL** (*ca.* 10000 – 750 cal. yrs. BP)	Rapid climatic amelioration and increased atmospheric moisture content.	*Corylus* and *Pinus*
11000	Late - glacial	**YOUNGER DRYAS** (*ca.* 11000 - 10000 cal. yrs. BP)	Cold and dry	*Juniperus* scrub and *Betula.*

Table 3.1 Table showing the Blytt-Sernander climate model for North-Western Europe from *ca.* 11000 cal. yrs. BP to present including the corresponding pollen zones (adapted from Caseldine, 1990; Lowe and Walker, 2000)

pioneering grassland communities typical of open habitats (e.g. Common Sorrel; *Rumex acetosa*, Nettle; *Urtica,* Meadowsweet; *Filipendula,* Crowbwrry; *Empetrum*) dominated the landscape. Juniper (*Juniperus*) and Birch (*Betula*) were generally established by *ca.* 10.0 k cal. yrs. BP followed by Hazel (*Corylus*) and Pine (*Pinus*), which spread in response to the rising Boreal temperatures. The wet Atlantic period was marked by the arrival of Alder (*Alnus*) and Willow (*Salix*) and the spread of Oak (*Quercus*) and Lime (*Tilia*), whilst the onset of the Sub-Boreal was characterised by peat formation and a decline in Elm (*Ulmus*). The latter decline corresponds at sites across Wales at *ca.* 5.0 k cal. yrs. BP (Hibbert and Switsur, 1976) and has been controversially linked to climate change, the Dutch Elm pathogen (*Ophiostoma ulmi*), ecological competition and human impact (Peglar and Birks, 1993). The links between climate, vegetation and human impact suggest that the cause is probably a combination of all these factors. Finally, the cooler climate of the Sub-Atlantic has encouraged the establishment of Beech (*Fagus*) and the return of grassland communities to the Welsh landscape.

Corresponding wet and dry palaeoclimatic phases have also been observed in the development of raised bogs at sites in England (e.g.; Hughes *et al.*, 2000; Barber *et al.*, 2000; 2003; Hughes and Barber, 2003; Dark, 2006) and north Wales (e.g. Chambers, 1983a; Ellis and Tallis, 2001) from *ca.* 7.0 k cal. yrs. BP and again at *ca* 5.0 cal. yrs. BP in response to the moist Sub-Boreal climate. These findings are strengthened by the observed relationship between past phases of river flooding (reflected as abrupt changes in sedimentation rates and gravel deposition on river floodplains) and climatic deterioration marked by periods of increased rainfall (Coulthard and Macklin, 2001; Macklin and Lewin, 2003).

Faunal studies from sites in England and Wales also provide corresponding evidence for Holocene climate change. Molluscan and ostracod analysis on lake marl sediments from Llangorse Lake, south Wales (Walker *et al.*, 1993) reflect shallow water conditions and highlight the relatively dry climate of the Late Younger Dryas/Early Boreal. Evidence from stenotypic sub-fossil Coleoptera (beetles) from the Glanllynau kettlehole in north Wales (Coope and Brophy, 1972) and Goldcliff in the Severn Estury (Smith *et al.*, 2000), support the suggestion of rising temperatures throughout the Boreal period; changes in the relative abundance of thermophilous taxa (e.g. *Lixus elongates, Sibinia variata, Cyphocleonus trisulcatus*) from *ca.* 9.5 k cal. yrs. BP indicate that the increase in Holocene temperature and rainfall was rapid, whilst taxonomic shifts between 7.0 and 5.0 k cal. yrs. BP support Lamb's (1985) hypothesis of a postglacial 'climatic optimum' during the Medieval period .

During the Sub-Boreal there appears to be a gap in the available faunal data to explain climate change in Britain. However, changes in atmospheric moisture during this period were recognised in the analysis of sub-fossil chironomids and their habitat preferences at rivers in northern France (Gandouin, 2005; 2006). Rather than the stable moist conditions postulated by Blytt (1876) however, the results of these studies indicate that this was a period of shifting weather patterns. These shifts caused changes in the flow rates of watercourses, which was reflected in shifts in the relative abundance of lentic and lotic species (e.g. *Microtendipes* spp. and *Rheocrictopus* spp. respectively). Finally, the cool conditions of the sub-Atlantic period are validated for Britain by high resolution, multiproxy reconstructions of climate change using diatoms, chironomids and pollen at Lochnagar, north-east Scotland (Dalton *et al.*, 2005), where thermophilous taxa decreased in response to climatic cooling. This was particularly marked during the 'Little Ice Age' (between *ca.* 1450 and 1890 AD) when temperatures were *ca.* 1.5°C lower than today (Jacoby and D'Arrigo, 1989).

Although regional studies indicate differences in climate change at the local scale, controlled by factors such as changes in oceanic and atmospheric circulation (Hoerling *et al.*, 2001), coastal proximity (Burn, 1997) and micro-topography (Guisan and Theurillat, 2000), the above discussion indicates that the traditional Blytt-Sernander model for north-western Europe can be applied, with caution, to the British Isles in order to produce a broad overview of climate change throughout the Holocene period. The present study will therefore adopt the system to compare and evaluate the influence of climate change on the core data.

3.5 Human Impact and Changes to the Holocene Landscape

Evidence for human impact during the Holocene can be found in pollen records, which reflect deforestation, changes in agriculture, and soil erosion due to the introduction of new technology and subsequent increases in land-use. This information can be compared to archaeological evidence for human settlements and burial sites to highlight direct catchment occupation and potential impact. Many studies show that settlement sites are particularly common in lowland lake catchments where sheltered positions and good quality soils encouraged human occupation (e.g. Fritz, 1989; David *et al.*, 1998; Bradshaw, 2002). In fact, the following section indicates that significant changes in climate can also have a great deal of influence over patterns of human migration. Welsh palaeoecological research has had a strong focus on palynological studies aimed at reconstructing past anthropogenic activity (Caseldine, 1990) and although there are regional differences in the degree of impact, when pollen data is compared to the local archaeology, a picture of landscape change due to human occupation can be developed. The following section addresses Holocene human occupation of the British Isles (with particular reference to Wales) and explores the potential of each archeological period to function as a major cause of environmental change.

3.5.1 Late Upper Palaeolithic (*ca.* 11 to 10.0 k cal. yrs. BP; Barton, 1999)

The return to glacial conditions in the high latitudes of the northern hemisphere during the Younger Dryas (*ca.* 11.0 k cal. yrs. BP) meant that Britain was still largely unpopulated (Jacobi, 1979). By *ca* 10.0 k cal. yrs. BP, however, climatic improvement, coupled with low sea levels encouraged Palaeolithic groups to return from the continent to continue a hunter-gatherer lifestyle and reoccupy many of the caves inhabited during the Upper Palaeolithic (*ca.* 40.0 k cal. yrs. BP) (Wymer, 1981; Wymer, 1999; Barton, 1999). Although archaeological evidence for human occupation has been found at several coastal sites around the River Severn (e.g. Tidenham near to Gloucester, Caldicot in Gwent, and Roath and Lavernock in South Glamorgan; Wymer, 1999), the south coast (e.g. Priory Farm Cave, Eel Point, Bacon Hole, Bosco's Den, Cathole Cave, Paviland Cave; Jacobi, 1979) and Carmarthen Bay (Coygan Cave; Clegg, 1970 and Little Hoyle Cave; McBurney, 1959) and in the north at Pontnewydd and Tremeirchion (Green, 1986), human impact on the landscape appears to have been minimal and difficult to separate from natural changes in the pollen record. By constrast, there is a significant paucity of environmental evidence for occupation further inland during this period (e.g. Traeth Mawr and Bwlch Owen, Carmarthenshire; Caseldine, 1990), possibly resulting from the extensive glaciation of inland and upland Wales. This may have restricted human habitation to the coastal areas where temperatures were warmer and flora and fauna more abundant.

3.5.2 Mesolithic Period (*ca.* 10.0 to 6.0 k cal. yrs. BP; Mithen, 1999)

According to Jacobi (1976) and Mithen (1999) climatic improvement during the Boreal period (from *ca.* 10.0 k cal. yrs. BP) and the development of mixed deciduous woodland encouraged humans to spread further inland in pursuit of migratory animals (e.g. roe deer, red deer, elk, brown bear, Arctic hare). In addition, charcoal records from the north Yorkshire Moors, England, show evidence of forest burning during the late-Mesolithic (*ca.* 6.0 – 5.0 k cal. yrs. BP) (Simmons and Innes 1996), suggesting movement to the north and human interaction with the immediate environment. However, it should be noted that during the earlier Mesolithic, human groups followed a food extraction strategy comprising hunting, fishing and gathering, and as a consequence anthropogenic impacts on the Welsh landscape during the Early Mesolithic were minimal.

In Wales charcoal evidence at upland sites in locations such as Moel y Gerddi (Chambers and Price, 1985), The Preseli Mountains (Seymour, 1985), Coed Taf (Chambers, 1983b) and The Black Mountains (Cloutman, 1983; Smith and Cloutman, 1988), also suggest deliberate intervention (and potentially incipient management of resources) by *ca.* 6.0 k cal. yrs BP. The discovery of lowland flint knapping sites at Craig-Y-Llyn (Savory, 1961) and Waun-Fignen-Fellen (Berridge, 1980) also suggest a pattern of logistical movement from the lowlands during the winter to the highlands in the summer in order to exploit local resources and take advantage of seasonal changes in the local flora and fauna.

Whilst hunting and the exploitation of plant resources were still important in the later Mesolithic, coastal sites such as Prestatyn (Clark, 1938), the Gower Peninsula (Lacaille and Grimes, 1955) and Goldcliff East (Ingrem, 2000) have all produced evidence for both shallow and deep sea fishing. Further inland, the archaeological evidence suggests that lowland, wetland areas were also accessed for hunting and fishing and in some areas environmental conditions encouraged a certain degree of settlement (e.g. Freshwater West; Lambert, 1963). The lack of pollen evidence for deforestation in these areas, however, may indicate that activities were centered on the use of natural clearings (Brown, 2002) or that subtle forest clearance is missed by low resolution studies (e.g. Star Carr; Day, 1993).

3.5.3 Neolithic Period (*ca.* 6.0 to 4.5 k cal. yrs. BP; Whittle, 1999)

Although the archaeological record for the Neolithic period in England suggests that Neolithic communities had interacted with Europe for some time, the abrupt spread of settlements, pottery, monumental funerary structures and polished tools dates the Mesolithic/Neolithic transition to *ca.* 6.0 k cal. yrs. BP (Whittle, 1999). The discovery of flint axeheads and wooden handles from this period may relate to woodland clearance, which links to the provision of open land for grazing domesticated animals and growing edible plants (Edmonds, 1995). Although changes to the landscape were generally still minor and localized, pollen studies from sites in the north of Wales (e.g. Llyn Mire; Moore, 1978, Trum Felen; Bostock, 1980 and Nant Helen; Chambers and Price, 1988) and in the south (e.g. the Brecon Beacons; Chambers, 1982, Llangorse; Jones *et al.*, 1978 and the Black Mountains; Price, 1981) show that decreases in arboreal pollen were increasingly accompanied by the appearance of ruderal (e.g. *Plantago lanceolata*) and cereal pollen (e.g. *Hordeum* and *Triticum dicoccum*), demonstrating the development of a mixed farming economy. It has been suggested that the diversity in farming practices during the Neolithic was promoted by the high rainfall and temperatures of the late Atlantic and early Sub-Boreal periods (Bonsall *et al*, 2002), which encouraged a more sedentary lifestyle marked by cultivation and animal husbandry.

The construction of communal tombs also grew in importance throughout the Neolithic (Whittle, 1999). Although styles varied from region to region, in Wales small chambered monuments were usually constructed from stone slabs enclosed within a cairn, that were used to mark land boundaries and provide a final resting place for ancestors and a focus for ritual (e.g. Pant y Saer on Anglesey in the north and Carreg Coetan in the south; Williams, 2003). Throughout Britain there is also evidence for an increase in ditched and causewayed enclosures (e.g. Windmill Hill, Haddenham, Edmonds, 1993) and cursus monuments (e.g. Cleaven Dyke Perthshire, Barclay, 1997) indicating the expansion of both domestic and ritual landscapes. In Wales, for example, evidence of an Early Neolithic settlement has been excavated at Llandygai, 2km south of the Menai Straits (Lynch and Musson, 2001) and included eight postholes from an early Neolithic house, ceremonial monuments, two henges and a cursus. 'Irish Sea' pottery found at the site dates the settlement to *ca.* 6.5 k cal. yrs. BP, thus producing some of the earliest structural evidence of permanent human occupation in Wales.

3.5.4 Bronze Age (*ca.* 4.5 k to 2.8 k cal. yrs. BP; Parker Pearson, 1999)

The Early Bronze Age in Britain is marked by the first occurrence of metal artifacts such as the flat axe found at the base of the ditch at Mount Pleasant (*ca.* 2300 cal BC). The only secure calendar date for the use of metal tools, however, comes from the Corlea trackway in Ireland (2264-2134 cal. BC), not from the tools themselves (Parker Pearson, 1999). Although there is evidence of settlement continuity at many sites in Britain, during the Early Bronze Age the landscape became more open as major land clearance allowed for large scale colonization of previously untouched areas (Smith, 1984). The warm, dry conditions of this period encouraged an increase in agricultural activity. At many sites in Wales agricultural expansion in the lowlands and a pastorally dominated landscape in the highlands is represented at many sites by the first major increase in cereal, grass and *Plantago lanceolata* pollen (Caseldine, 1990). Table 3.2 shows that in Wales the palynological shifts correlate well in areas where the development of mixed farming was common.

Recent research shows, however, that these changes were non-sustainable, particularly on the poorer upland soils where sustained deforestation, increased run-off, soil saturation and the formation of blanket bog, which began during the middle Bronze Age, caused a population shift to the lowland areas where soils were in better condition (Caseldine, 1990; Champion, 1999). It is also possible that these factors were encouraged by a phase of climatic deterioration at the end of the Sub-Boreal period (van Geel *et al.,* 1996), which may have been triggered by major volcanic activity; the Santorini eruption occurred at the late 17th century BC, 40 miles north of

Site	**Author**	**First Record of Pollen Indicators of Agricultural Activity (cal. years BP)**
Cefn Graeanog, North Wales	(Chambers, 1983)	3,000
Nant Helen, West Glamorgan	(Chambers *et al.*, 1988)	3, 310
Llanllwch, near Carmarthen	(Thomas, 1965)	3, 200
Llangorse Lake, Carmarthenshire	(Chambers, 1999)	3, 240
Machynys, Llanelli, Carmarthenshire	(Lillie *et al.*, 2000)	3,700

Table 3.2 Table showing the date of first occurrence of agricultural indicators in pollen records from a selection of sites in Wales.

Crete inducing tephra deposition in the Nothern Hemisphere at least (Manning *et al.*, 2006).

On the other hand, the environmental evidence from lowland sites in Wales suggests that by *ca.* 3.0 k cal. yrs BP the intensive exploitation of farmland caused major increases in open heathland, accompanied by shifts in hydrology, changes in soil quality and the first significant evidence of soil inwash to lake systems due to the permanent removal of woodland (e.g. Bryn y Castell, Snowdonia, Mighall and Chambers, 1995; Coed Taf, Chambers, 1983a; Nant Helen, Chambers and Price 1988; Llangorse, Chambers, 1999; Tregaron and Whixall Mosses, Turner, 1964). Local archaeology indicates that although the main focus of land clearance was to increase the amount of available farmland, forest cover was also removed for the mining of metal ores and the production of Bronze (e.g. Great Orme, Llandudno; Champion, 1999), the construction of funerary monuments (e.g Penrhyncoch, Smith and Taylor, 1969 and Moel Goedog, Conway and Younis, 1984) and settlement enclosures

(Champion, 1999). The result of these combined factors was that by the end of the Bronze Age both upland and lowland soil quality had seriously declined and there was a contraction of available land with which to produce food for a growing population.

3.5.5 Iron Age (*ca.* 2.8 k to 2.0 k cal. yrs BP; Haselgrove, 1999)

Climatic deterioration continued into the Iron Age and there was a general movement towards the warmer foothills, lakesides and margins of river valleys (Mytum, 1988; Caseldine, 1990). On the whole, the construction of enclosed farmsteads, suggesting tactical defense, was increasingly important, along with an emphasis on land clearance for grazing and small scale farming (e.g. Penycoed, Walker, 1985). At some sites, however, the archaeological evidence demonstrates that the traditional shift from Bronze to Iron Age, which Haselgrove (1999) places at 2.7 k cal. yrs. BP, was actually less distinctive. In fact, prolonged woodland clearance, along with an significant increase in both arable and pastoral farming indicates that there was also a high degree of settlement continuity at many sites (e.g. Welsh Marches [Savory, 1980]; Llangorse [Jones *et al.*, 1985]; Carneddau [Walker, 1993]).

Excavation of sites in Britain shows that during the Early Iron Age enclosed farmsteads of 1 to 2 hectares were common, though the shape of the domestic buildings (e.g. D-shape, rectangular or circular) and settlement layout appear to be regional features (Hingley, 1992). Towards the end of the Iron Age closed settlements in areas of political stability, such as south Wales, were replaced with an increased number of aggregated, open settlements (Haselgrove, 1999), whilst at many sites in mid-Wales deforestation correlates with marked peaks in *Plantago lanceolata* suggesting that pastoral farming and general animal husbandry increased in importance, particularly on the higher grounds (Hogg and King, 1963; Moore and Chater, 1969). This pattern is also apparent in south Wales, where a phase of settlement and agricultural expansion at *ca.* 2.2 k cal. yrs. BP is reflected as deforestation, soil deterioration and soil inwash in pollen records from Llangorse (Jones *et al.*, 1985), the Brecon Beacons (Chambers, 1982) and the Black Mountains (Price and Moore, 1984). At some sites (e.g. Castell Henllys [Mytum, 1988]; Castell Ditches [Parkinson, 1976]), bone evidence from the butchery of cows, sheep, goat and pigs also indicate the importance of domesticated animals and the consumption of meat as a part of the Iron Age diet.

Large scale deforestation during the Iron Age in Wales also coincides with the exploitation of woodland to produce charcoal for iron smelting (e.g. Braich y Dinas; Hughes, 1922) and for the building of hillforts (e.g. Moel-y-Gaer [Guilbert, 1975]; Bryn y Castell [Mighall and Chambers, 1995]). Although common across large areas of Britain, the latter defended settlements appear to be particularly clustered in the south of Wales, with the smaller forts (>3 acres) extending along the west coast to Anglesey and the larger structures (<15 acres) placed along the English border to the east (Cunliffe, 2005). Intensive deforestation for the construction of forts, made possible with the adoption of more effective iron tools, is reflected in pollen records. Changes to the landscape are often accompanied by the construction of funerary monuments and Late Iron Age funerary deposits (e.g. Harding's Down [Crampton, 1973]; Llyn Cerrig Bach [Fox, 1946]).

3.5.6 The Roman Period (*ca.* 2.0 k cal. yrs BP to *ca.* 400 AD; Cleary 1999)

It is understood that long-term settlement expansion and agricultural intensification by local populations during the Roman period probably caused great impact on the British landscape, though this differs regionally depending on the resistance of local populations (Cleary, 1999). The development of towns with administration centers, amphitheatres and bath-houses, etc., on old settlement sites was common throughout England, producing the nuclei of many major English cities today (e.g. York, Leicester, London). Mining of mineral ores such as copper, gold, iron, lead and silver was also an important component of the Roman economy and may even have been one of the primary reasons for the Roman conquest in Britain, for regardless of previous mineral exploitation the scale of production and the effect on woodlands increased dramatically following Roman occupation (Cleary, 1999).

Although the Welsh tribes fiercely resisted Roman occupation until 74 AD, after this date the high degree of chronological and cultural overlap with Iron age communities suggests that the Romans generally settled in areas that were already occupied (Hanson, 1999). Following this date, on the basis of the remains of Roman forts and bathhouses (Manning, 2001) and the large-scale.manufacturing of pottery in Wales (Blackmoor, 2002), it is suggested that the evidence indicates the development of a strong Roman economy. Carbonized grains in the plant macrophyte record (e.g. *Triticum dicoccum*, *T. spelta* and species of *Avena* accompanied by arable taxa such as *Chenopodium* and *Urtica*) and evidence of palaeosoils at Colfryn (Keeley, 1986) and Stackpole Warren (Benson *et al.*, 1990) provide evidence of ancient field systems, plough marks and even podzolisation from the first century AD, suggesting a concurrent increase in land-clearance and arable cultivation (Caseldine, 1999). A number of waterlogged plant remains from Roman wells at sites in Wales also produce valuable information of changing crops and Romano-British diets. Exotic plants such as *Anethum graviolens* (dill), *Coriandrum sativum* (coriander), *Figus carica* (fig) and *Vitis vinifera* (grape), along with indigenous *Corylus avellana* (hazelnut), *Sambucus* (elderberry), *Rubus rubus* (blackberry) and *Rubus idaeus* (raspberry) have been discovered in deposits from Caernarvon (Hillman, 1985), Carmarthen (Hillman, 1978) and Whitton (Wilson, 1981), indicating a variety of crops based on both local and imported food types.

There is also evidence that underground mining activities changed the Welsh landscape. The most outstanding example of Roman mining in Carmarthenshire is the Dolaucothi goldmines in the River Cothi valley where half a million tons of rock and earth were removed leaving a series of hills and humps in the Carmarthenshire landscape (Shepherd, 1993). New roads were also constructed to transport the mined metals and metal products to the coast for export (Lewis and Jones, 1969). It seems that these activities also caused significant air pollution. Atmospheric pollution in the geochemical record of peat layers, detected in samples taken from the former metal mining area of Ystwyth valley, Dyfed (Mighall *et al,* 2002a), for example, show that lead processing caused significant lead enrichment during the Roman Period, whilst samples from Copa Hill, Cwmystwyth, in mid-Wales indicate that copper production, which continued into the Roman Period, caused serious soil contamination (Mighall *et al,* 2002b).

3.5.7 Post-Roman Period and the Scandinavian Presence (*ca.* 400 to 1050 AD; Hills, 1999)

Following the Roman retreat, direct and continuous contact with Ireland to the west, Scandinavia to the north-east, Germany to the south-east and France to the south maintained an outside influence on the British Isles (Hills, 1999). Although this was a period of settlement continuity (e.g. Dark, 1996), there is little evidence to explain how these changes affected the cultural and rural landscape outside of England. So far, it is clear that Post-Roman Wales was characterized by a high concentration of scattered lowland farmsteads (Caseldine, 1990) with a distinct lack of upland settlements, accompanied by forest regeneration on the higher grounds. The desertion of upland areas is likely to reflect the Sub-Atlantic climatic deterioration reported by Lamb (1977) when populations in general moved to the warmer lowland areas. The economy was mixed and appears to have varied at a regional scale; at some sites (e.g. Dinas Emrys; Seddon, 1960) pollen studies for this period show evidence for an increase in pastoralism on the lower slopes, whereas others reflect an arable landscape in the lowland areas dominated by the cultivation of *Triticum spelta*, *T. dicoccum* and *Hordeum vulgare* and supplemented by the rearing of domestic cattle (e.g. Cledemutha, Tomlinson, 1987).

From 800 AD, monuments, settlements and farmsteads in the British Isles had a strong Scandinavian influence dominated by rural hamlets consisting of large timbered halls, enclosure ditches and small family buildings (Richards, 1999). In Wales the influence of 'Celtic' populations remained dominant in the north, whilst in the south (e.g. Gwent, Dyfedd and Glamorgan) evidence of Scandinavian incursions and general migration patterns are reflected in the style of buildings and burial sites and the distribution of Nordic family and place names (e.g. Skomer and Swansea) (Weale *et al.*, 2002). On the whole, large and prosperous settlements from this period are rare and Scandinavian populations appear to have concentrated mainly on the trading of goods, such as horses, fur, honey, wine, whale oil and dairy produce, with the existing Welsh communities (Charles, 1934).

3.5.8 The Medieval Period (*ca.* 1050 to 1500 AD; Schofield, 1999)

During the Medieval period climate was the main factor influencing human migration and patterns of agricultural activity. At the beginning of the 11th century improved climatic conditions (i.e. the 'Medieval Warm Period'), coupled with a population increase, encouraged rural communities to move to higher altitudes where upland areas were exploited and a hafod and hendre system of seasonal transhumance agriculture was common (Caseldine, 1990). In the 14th century, however, climate change in the northern hemisphere (i.e. the 'Little Ice Age') influenced a second shift back to the warmer, lower grounds (Schofield, 1999).

In his study of the documentary evidence of peasant agriculture in Gwynedd, Thomas (1975) describes the economic and environmental changes in rural Medieval Wales, which appear to correspond with these significant changes in climate. In Phase 1 (1000 to 1200 AD) populations practiced a 'natural' barter economy where every plant had its prescribed value for making agricultural and domestic implements, building materials, food and medicine. Small lowland hamlets contained 'wattle and daub' huts with modest arable plots and incentives were given to clear upland woodland for common pasture where cattle were grazed and utilized for meat, dairy produce and skins. During Phase 2 (1200 to 1350 AD), though deforestation was still limited, large areas of pasture were utilized, often more than 1000 feet above sea level, with the main nuclei of village life found in the semi-lowland and sheltered upland areas where crops such as oats, corn and barley were grown. By the late 13th century, however, many communities had left the upland slopes. Although actual figures are unknown due the absence of census data, by the mid-14th century the Black Death had wiped out whole communities; the county of Gwynedd, for example, contained *ca.* 120 townships with sparse population densities of between 1 and 6 families per square kilometer.

During Phase 3 (1350 to 1500 AD) uneven mortality rates due to further outbreaks of the Black Death and a money based economy under a new administrative structure coupled with climatic deterioration caused a shift back to the lowland areas. At Col Fryn (Jones and Milles, 1989) and Ty Mawr (Caseldine, 1990) plant macrofossil evidence shows that *Secale cereale* (rye), *Hordeum vulgare* (barley), *Triticum spelta* (spelt wheat), *T. aestivo-compactum* (club wheat) and *T. dicoccum* (emmer wheat) crops were grown, whilst pollen evidence from Llyn Mire (Seymour, 1985) and Llyn Cororion (Watkins, 1990) indicate that *Cannabis sativa* cultivation became increasingly important. In fact, high sediment accumulation rates at Llyn Mire (Seymour, 1985) and Llyn Berwin (Caseldine, 1990), probably caused by deep ploughing, and permanent deforestation at Plynlimon (Moore, 1968) and Bwlch-Y-Fign (Taylor, 1973) indicate that, although land divisions and similar farming techniques remained (Hills, 1999), this was a period of intense agricultural activity.

Ecclesiastical establishments constructed during the early medieval period became the focal point of the parish with churches serving small nucleated villages and monasteries forming the center of towns and cities with each diocese centered within monastic or secular cathedrals. One of the finest examples of Cistercian Abbeys in Wales is Strata Florida (Robinson and Platt, 1998), the building of which completely changed the immediate landscape. Deforestation was followed by large-scale land drainage and the digging of water channels to accommodate the Abbey, which along with additional alms houses, gatehouses and agricultural buildings was completed in 1184 AD. These building not only served a religious function but also became the regional centers of administration and trade; evidence of a growing Cistercian wool industry from the middle of the 13th century, for example, is supported by the high concentrations of sheep/goat bones at rural sites like Strata Florida, Tintern Abbey (Parkes and Webster, 1976) and Valle Crucis Abbey (Barker, 1976).

3.6 Palaeolimnological Research in Wales

Palaeolimnological research in Wales is far less extensive than studies that focus on reconstructing the effects of climate and human impact on the landscape. Figure 3.2 shows that detailed studies that highlight long-term palaeolimnological change are centered in the highlands of north Wales and include the Late Quaternary diatom stratigraphy of Cors Geuallt (Crabtree, 1965; 1969; 1972), the environmental history of high mountain, oligotrophic tarns in Snowdonia (e.g. Llyn Glas and Llyn Clyd) using pollen and diatoms (Evans and Walker's, 1977) and a reconstruction of the evolution and subsequent histories of Llyn Padern and Llyn Peris at Llanberis Pass,

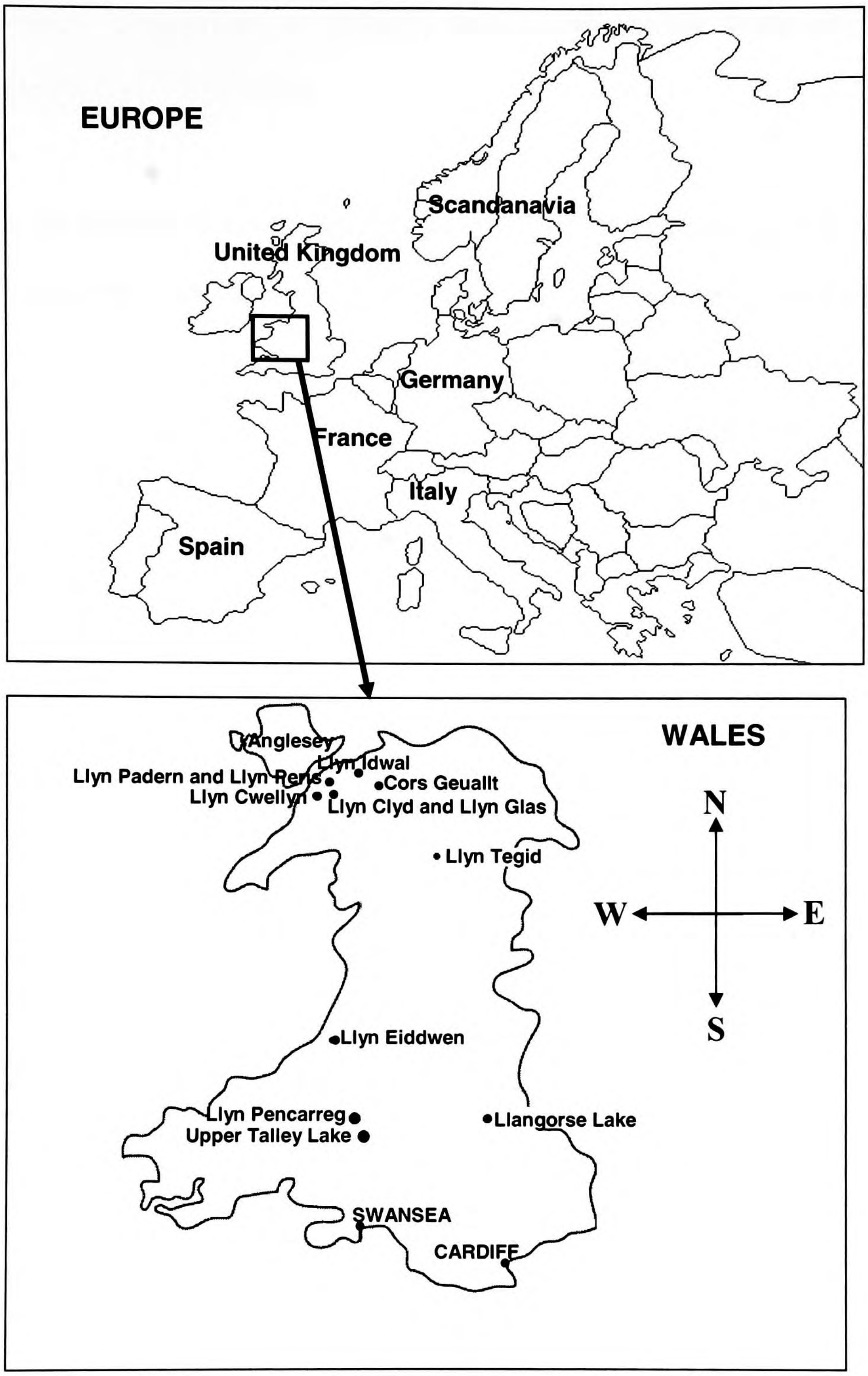


Figure 3.2 Map showing the location of sites in Wales where previous palaeolimnological reconstructions have been carried out in relation to the study sites selected for this project (Upper Talley and Llyn Pencarreg).

North Wales, using pollen, chrysophytes, diatoms and chemical stratigraphy (Elner and Happey-Wood, 1978; 1980).

To date the majority of diatom-based studies from Wales contribute to EDDI training sets for the development of total phosphorus (TP) transfer functions (Battarbee *et al.*, 2000). Although the Welsh transfer function has been applied to reconstruct past nutrient concentrations at individual lakes (Figure 3.2), the studies are short-term only. Bennion (1995) and Bennion *et al.*, (1996a), for example, reconstructed the trophic history of three lakes in Anglesey over the last 150 years and recent environmental impact has been investigated at Llyn Tegid, Gwynedd (Bennion *et al.*, 2002), Llyn Idwal and Llyn Cwellyn, Snowdonia (Bennion *et al.*, 1997) and Llyn Eiddwen and Llyn Fanod, Ceredigion (Bennion *et al.*, 1998).

Llangorse Lake (SO131262), the largest glacial lake in South Wales, is a naturally eutrophic site within the Llynfi river valley in the Brecon Beacons National Park, which has been subject to the most extensive scientific investigation. Recently designated a Special Area of Conservation (SAC) due to its high biodiversity, algal blooms were first recorded in the 12th and 16th centuries (Griffiths, 1939) and blooms occurred during the 1970's as a result of sewage effluent from the adjacent village (Brecon Beacons, 2006). A diatom inferred TP (DI-TP) reconstruction (Bennion and Appleby, 1999) indicates a rise in nutrient status which appears to be linked to agricultural intensification in the recent past.

These studies show that there is a clear lack of long-term palaeolimnological research in Wales as a whole, but particularly in the south where studies focus mainly on changes in the diatom record due to recent human impact (*ca.* past 200 years). The present study therefore aims to address this gap in current research by selecting two sites in the south of Wales for palaeolimnological investigation.

3.7 Selection of Study the Sites

To test whether two very similar lakes may show significant differences in response to environmental change, closely adjacent sites (10 km apart in Carmarthenshire, South Wales; Figure 3.2) were selected in catchments with similar geological characteristics. Sites were chosen with pH in the circumneutral range in order to avoid complications with limnological response to acid precipitation (Battarbee *et al.*, 1990). The limnological similarity of the lakes was assessed in March, 2005 by basic water chemistry measurements and field observation of physical similarities (Table 3.3).

Upper Talley is higher in nutrient status and has a larger catchment (calculated using ARC-GIS), whilst Llyn Pencarreg has a greater maximum depth, but otherwise the two lakes are similar in all respects. There is documentary evidence for similar patterns of long-term human impact at both sites particularly during the medieval period, which gave rise to associated changes in land-use.

Environmental Parameters	**Upper Talley**	**Llyn Pencarreg**
Origin	Glacial	Glacial
Geology	Non-calcareous Silurian Wenlock bedding, with Old Red Sandstone layers	Non-calcareous Silurian Wenlock bedding, with Old Red Sandstone layers
Soil Type	Well-drained, loamy brown earth	Well-drained, loamy brown earth
Catchment Size (m^2)	180	60
Height OD (m)	105	110
Surface Area (km2)	0.07	0.09
Max. Depth (m)	4.3	10
Spring TP ($\mu g\ l^{-1}$)	100	36
DOC (%)	93.3	92.8
pH	6.6	6.3
Conductivity ($\mu S\ cm^{-}1$)	91	82

Table 3.3 A comparison of environmental parameters between Upper Talley and Llyn Pencarreg including origin, geology, soil type, catchment size, height above sea level (OD), surface area, maximum depth, Spring total phosphorus (TP) concentration, dissolved oxygen content (DOC), acidity (pH) and conductivity.

3.7.1 Upper Talley, Carmarthenshire, South Wales

3.7.1.1 The Lake and its Immediate Surroundings

Upper Talley Lake (NGR SN633330) (Figure 3.3) is situated at approximately 105m OD. It has a surface area of 7 ha (0.07 km^2) and a maximum water depth of 4.3 m at the centre. Catchment geology is predominantly non-calcareous Silurian Wenlock bedding overlaid by Old Red Sandstone layers, which outcrops throughout the Llandeilo area (Pringle and George, 1948), and the catchment size is 180 m^2.

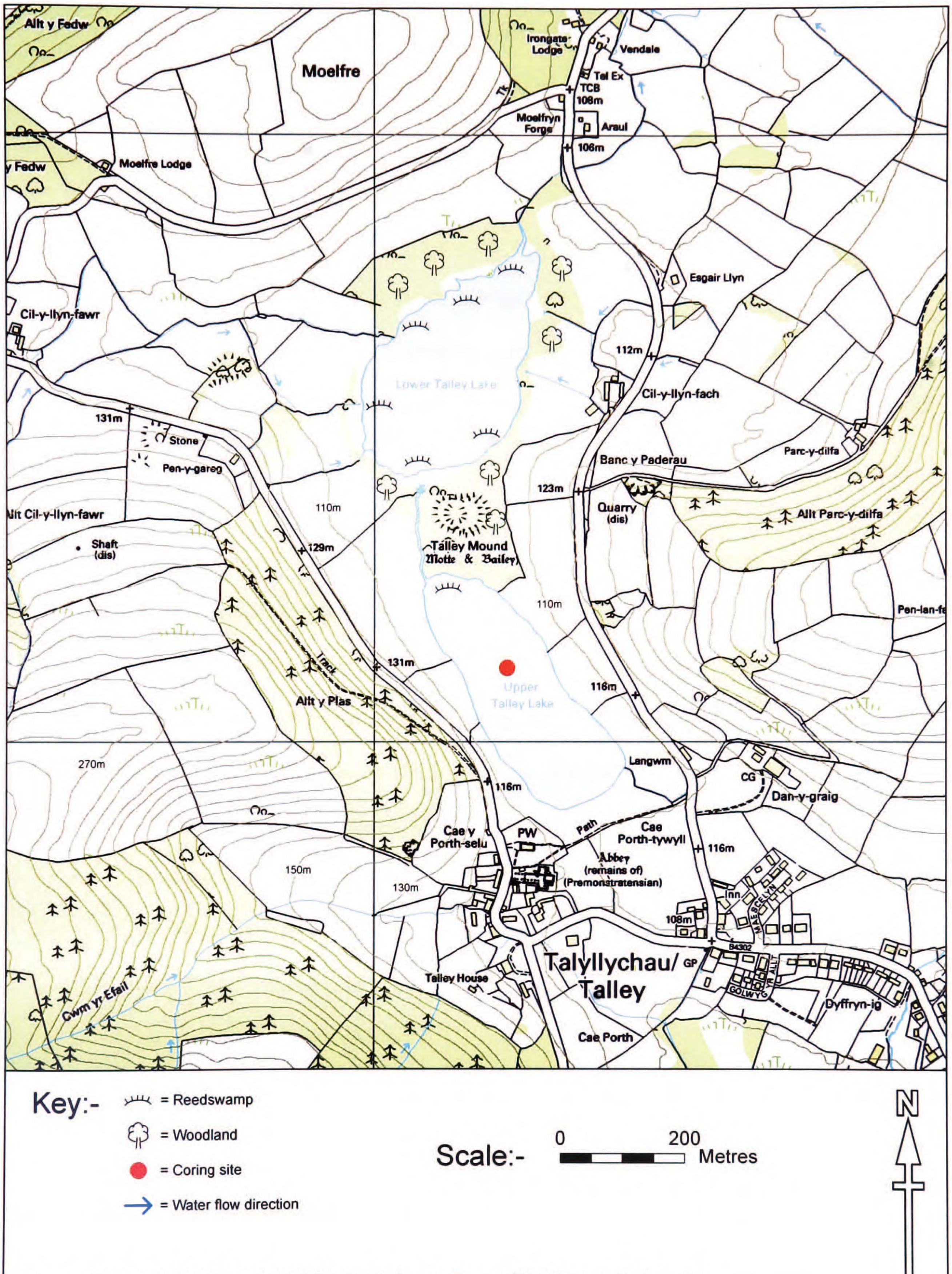


Figure 3.3 Map showing the Upper Talley Lake, Carmarthenshire, South Wales.

The lake basin itself was originally formed during the Late Devensian either by erosion of a lithological weakness, sub-glacial melt-water or as a kettle hole (Pacey, 1993); the glacial origin is supported by the presence of glacial material at the base of sediment cores (Butler, 1984). Soils are now mostly well-drained, loamy brown earths apart from areas where clays are present (Davies, 1982). In addition to surface precipitation, hydrological input appears to be mainly generated from run-off from hills situated to the north (Moelfre), east (Mynydd Llansadwrn) and west (Mynydd Cynros) and by input from three subterranean springs, which provide the lake with mineral-rich water (Johnson, 1998).

The land directly surrounding the lake forms a steady incline to the water's edge and is predominantly pasture dedicated to low intensity sheep grazing with small wooded areas of *Quercus*, *Alnus* and *Corylus*, whilst in the littoral zone the modern lake supports a low abundance and diversity of aquatic macrophytes. Upper Talley is one of two lakes linked by means of a narrow channel; the second lake, Lower Talley, is bounded by a well-marked hydroseral transition from mature alder-carr to reed swamp to open water with a littoral zone rich in aquatic plants. The mound separating the two lakes, interpreted by Butler (1984) as a terminal moraine, now supports a number of different tree species including *Quercus, Fagus, Larix, Ficus* and *Corylus* and a range of other woodland plants.

Due to the high conservation value of the Talley Lakes a notification of Site of Special Scientific Interest (SSSI) has been allocated under the Wildlife and Country Act of 1981. The Carmarthenshire Bird Trust reports sightings of the rare ring-neck duck, whilst in winter the lake supports a wide variety of wildfowl including

Goldeneye, Pochard, Tufted Duck, Teal, Mute and Whooper Swans, Pintail, Scaup and Smew. In summer the Great Crested Grebe and Kingfisher visit the lake and otters have been spotted hunting at its north end (Forster, 1993). One of Upper Talley's most important features is the presence of the only known population of the medicinal leech (*Hurudo medicinalis*) in Wales (Lloyd, 1998). This species is listed on Appendix II of The Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES), Appendix III of the Bern Convention and Annex V of the Habitats Directive. The International Union for the Conservation of Nature and Natural Resources (IUCN) lists *Hurudo medicinalis* as 'Vulnerable' due to a decline in its natural habitat and host species and the adverse effects of agricultural pollutants, which may have serious implications for its long-term survival.

3.7.1.2 Archaeology in the Upper Talley Region

Figure 3.4 shows the known archaeology within 5 km of the study sites. The map indicates that there was human activity in the Upper Talley area from as early as the Neolithic period. Although there is no direct evidence for settlements during the Bronze Age, activity is highly concentrated around the Talley lakes and consists of standing stones and round barrows. Their presence on higher ground suggests that they may relate to funerary and ritual sites away from any main settlement area, whilst their composition, mainly of stone and covered with turf, suggests that they were constructed from cairn material typical of prehistoric monuments and probably found in the immediate vicinity the site. On the mound between Upper and Lower Talley, archaeological remains of what is thought to be a Norman Motte and Bailey Castle have been located by aerial photography (Hogg and King, 1963). Hillfort sites from

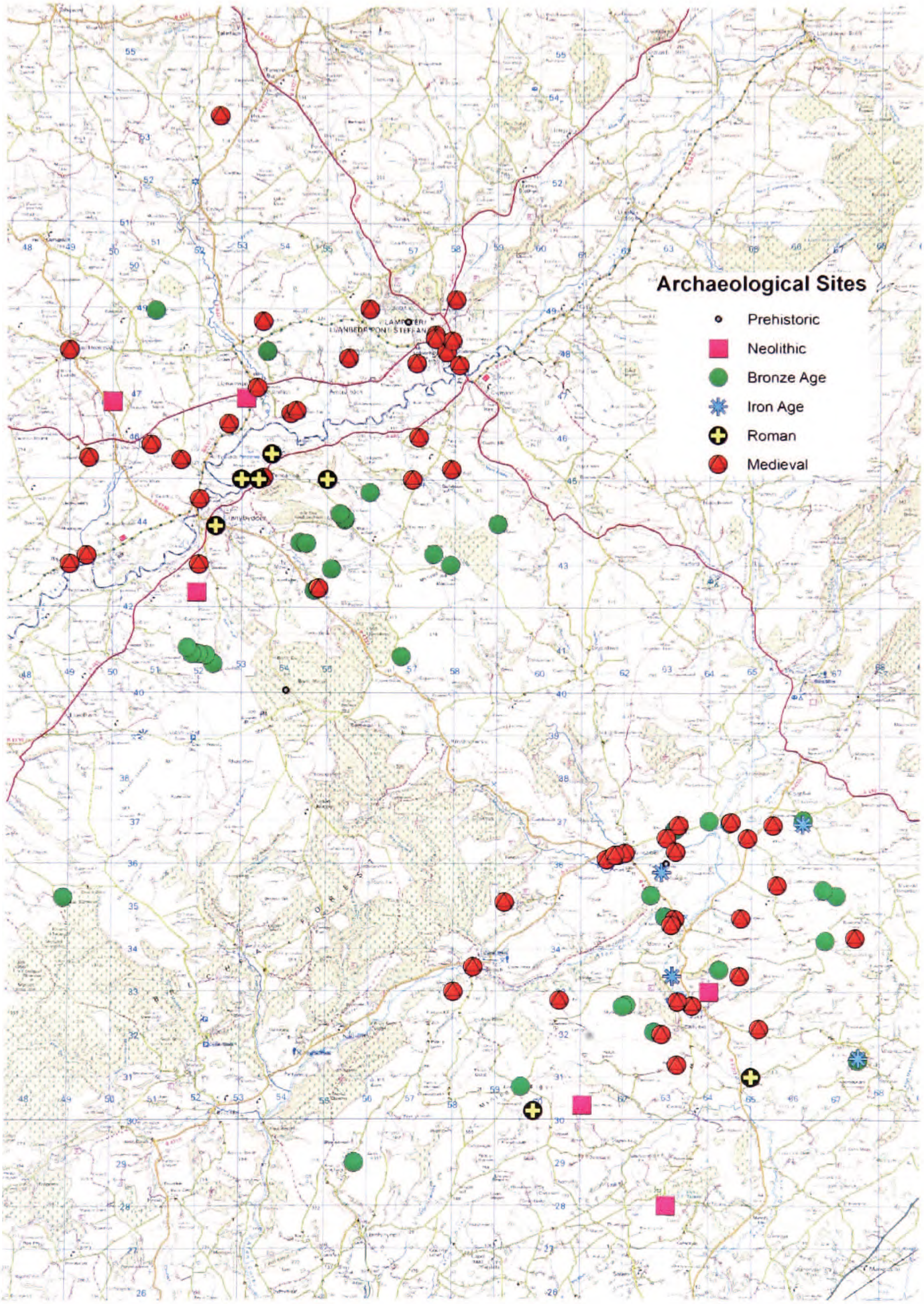


Figure 3.4 Diagram showing known archaeology within 5 km of the study sites.

the Iron Age have also been found within a 4.5 km radius of Talley, the nearest being at Pen-y-Ddinas, which lies 2.0 km to the north of the study site, though Roman archaeology is far less common.

Evidence of Medieval settlements and farmsteads are profuse in the Talley region, but most importantly the site provides an excellent example of medieval occupation and subsequent abandonment, with the remains of one of Britain's largest 12th century Premonstratensian Abbey, which lies on the south bank of the Upper Talley lake (Price, 1894; Owen, 1893, 1894; Robinson and Platt, 1998). Founded in 1185 AD by Rhys ap Gruffydd (ruler of Dehevbarth, south Wales), Talley Abbey housed an order of white cannon Cistercian monks. The widespread influence of ecclesiastical settlement on woodland clearance and agricultural activity is indicated in pollen records in mid to south Wales at around 1200 AD (e.g., Tregaron: Turner, 1964; Preseli: Seymour, 1985). However, this is the first study to explore the possibility of associated limnological impact.

In an account of his journey through Wales (*Itinerarium Cambriae*, 1191 AD), Giraldus Cambrensis (1146-1223 AD) describes the Talley Lakes as a 'rough and sterile spot, surrounded by woods and trees on every side', suggesting that the monks may have settled in an area showing little indication of any previous human occupation and that changes to the landscape due to the building of the Abbey were minimal. Apart from the subsequent evolution of the village of Talyllychau (translated as 'the head of two lakes') to the south of the Abbey (present population 530; Carmarthenshire, 2007), no settlements have been established since the monastery was abandoned. The site therefore provides an excellent opportunity for assessment of the long-term impact of early settlements on water quality between the 12th and 16th centuries.

3.7.1.3 Previous Research at Upper Talley

An undated sediment core from Upper Talley was analysed for pollen by Butler (1984) at a resolution of 20-25cm. Inferred postglacial shifts in vegetation were attributed to early Holocene climate change, whilst phases of woodland clearance and agricultural activity were tentatively linked to documented local human occupation. Although absolute dates were not produced due to the lack of material suitable for dating, Butler (1984) reasons that a significant clearance phase beginning soon after the *Tilia* decline could be the result of Bronze Age activity. Butler also suggested that a subsequent phase of clearance, reflected as a decline in *Alnus* and accompanied by a three-fold increase in agricultural and pastoral herbs, could represent activities on the valley floor during the building and occupation of the Abbey.

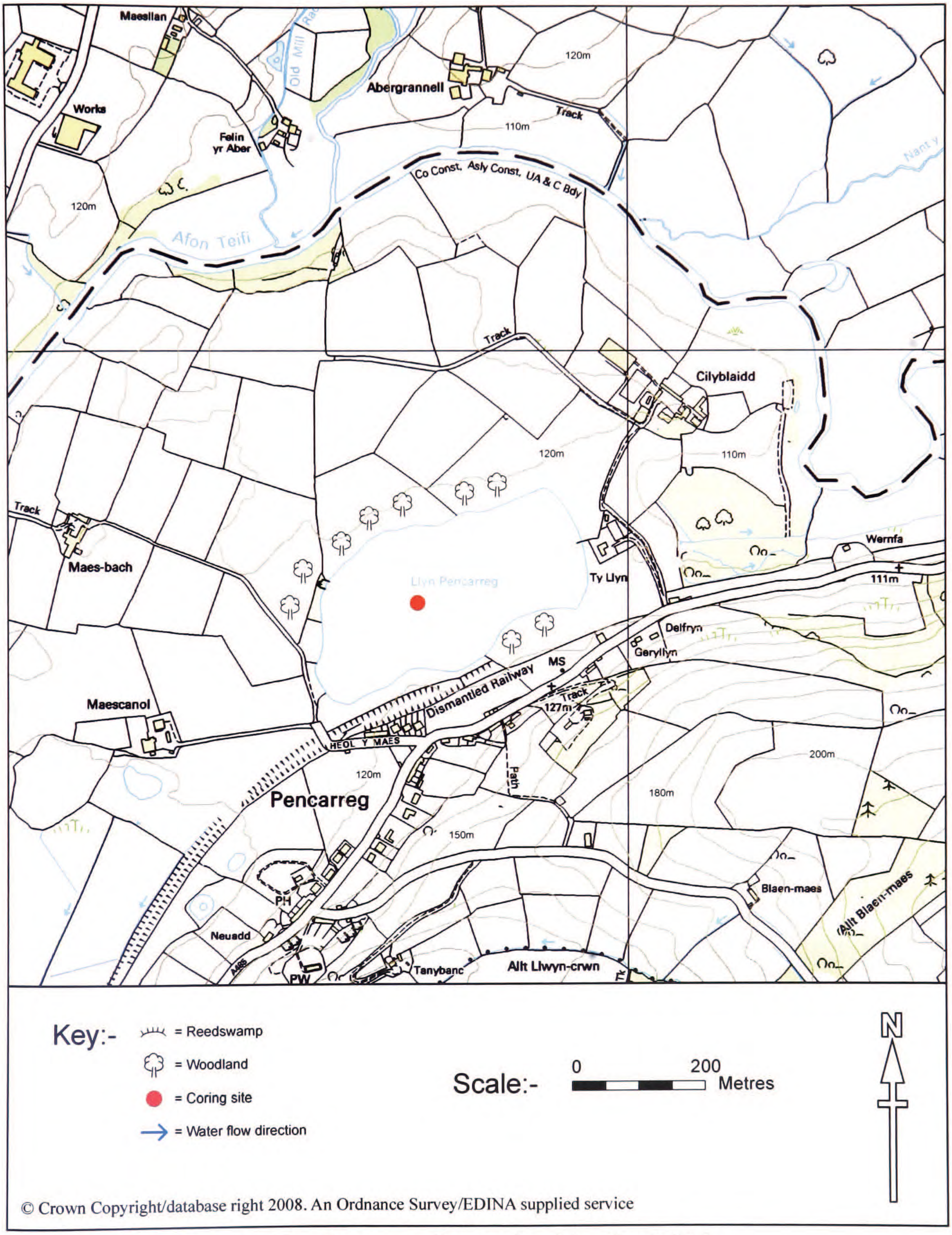
Fourteen years later, a higher resolution pollen reconstruction was carried out by Johnson (1998) on the same core used for the present study. Palynological correlation with a range of radiocarbon-dated, biostratigraphic horizons from the Lower Talley Lake and linear interpolation provided an age-depth chronology. Johnson (1998) identified minor forest disturbance during the late Mesolithic period (the cause of which was difficult to evaluate) and found evidence for major clearance during the Bronze Age and Medieval periods, as did Butler (1984). These major shifts were characterised by increases in cereal, pastoral herbs, hemp and flax pollen suggesting major agricultural changes in the Talley landscape during this period. Changes in the aquatic pollen record also suggested a shift in lake productivity, with a loss of aquatics during the medieval period and an increase in the diversity of aquatic plants immediately following this phase. *Urtica* also appeared, which both Moore (1978) and Heinselman (1973) argue is indicative of the addition of phosphorus to the nutrient budgets of the local catchment.

3.7.2 Llyn Pencarreg, Carmarthenshire, South Wales

3.7.2.1 The Lake and its Immediate Surroundings

Llyn Pencarreg (NGR SN538457) (Figure 3.5) is situated 10km to the north of the Talley Lakes and 6km from Lampeter at approximately 110m OD. It has a surface area of 9 hectares (0.09km^2) and a maximum water depth of 10m at the centre - almost 6m deeper than Upper Talley. It has been suggested that the lake is also glacial in origin, formed in the late Devensian period during glacial retreat, but to date there is no evidence of this. Bedrock geology at Llyn Pencarreg is the same as at Upper Talley and the main catchment soils are also well-drained, loamy brown earths. Catchment size, however, is only 60 km, which is a third of the size of Upper Talley.

Apart from a wave-cut platform on the west bank and a steady incline to the east, the north and south lakesides are steep, forming sills that overlook the lake. Beyond this to the north are hills (Allt Llwyn), which presumably contribute to the hydrological budget by surface run-off. Like Upper Talley the land immediately around the lake, to the north, west and east, is predominantly pasture with patches of *Corylus* and *Quercus*, and is used for low intensity sheep grazing and stud farming. Aquatic



© Crown Copyright/database right 2008. An Ordnance Survey/EDINA supplied service

Figure 3.5 Map showing Llyn Pencarreg, Carmarthenshire, South Wales.

vegetation in the littoral zone of the lake is dominated by *Potamogeton* spp. and *Littorella uniflora*, which grow on the gravely, western marginal shelf down to a water depth of about 3 m. The village of Pencarreg (population 1120; Carmarthenshire, 2007) is situated on the high granite sill (160m) at the southern edge of the lake.

Llyn Pencarreg lies just within the river valley of Afon Teifi, which, along with ten of its tributaries, has been designated a Special Area of Conservation (SAC) and Site of Special Scientific Interest (SSSI) for its significant conservation and landscape value. The Carmarthenshire Bird Club, for example, report sightings of the rare Whooper Swan and migrants such as Black Tern, Sandwich Tern and Sabine's Gull. In the winter months the lake supports a community of waterfowl including Teal, Widgeon, Pochard, Goldeneye and Coot (Forster, 1993).

3.7.2.2 Archaeology in the Llyn Pencarreg Region

Figure 3.4 indicates human activity close to Upper Talley from as early as the Neolithic period with the discovery of several axe pieces within a 4.0 km radius of the site indicating interaction with the immediate environment. As with Upper Talley, Bronze Age round barrows and standing stones are profuse on the hills surrounding Llyn Pencarreg. However, the Pencarreg record also lists a Bronze Age well at Banc-y-Garn (2.0 km southeast of Pencarreg) and several remnants of pottery, suggesting that there was also some settlement activity in the area.

The remains of a defended enclosure at Blain-Maes (1.0 km from Pencarreg) provides the closest archaeological evidence of Iron Age settlement, whilst hillforts can be seen at Dolgwn Isaf (1.5km to the east of Pencarreg) and Ty Neydd, immediately northwest of Dolgwm Isaf (Dyfed Archaeological Trust, *pers. comm*). Finds from the Roman period are more common at Pencarreg than at Talley and are discussed in more detail in Section 8.3. There is also a main road that runs through the village of Pencarreg, which is situated on the sill directly overlooking the lake (the A485, now linking Carmarthen to Aberystwyth). The route is recorded as a Roman road, which may have been built to transport gold from the Dolaucothi goldmines to the south coast.

The medieval period is marked by the discovery of archaeology directly associated with settlement activities. These include several holy wells within a 4.0 km radius of Pencarreg and two water mills and a corn mill at Rhyd Y Bont (2.0 km south of Pencarreg), Rhyddlan (4.0 km southwest of Pencarreg) and Tirnewydd (5km northwest of Pencarreg) respectively. As with Talley there are several medieval finds in the village of Pencarreg itself suggesting a high degree of activity within the lake catchment during this period. The parish church and graveyard are also medieval in origin, indicating a well-established village community from the 13th century onwards. However, the church may not have acted as a focus for a nucleated settlement as during the 12th and 13th centuries it was customary for the Welsh to live in dispersed communities.

A disused railway is situated on the south bank of the lake below the sill, part of the Lampeter to Carmarthen line, which was built in 1866 and extended to Aberystwyth in 1867, thereby providing a link between these major towns until 1965 (Lampeter, 2005). Due to its proximity to the lake its construction may have had some influence on limnological conditions. To date, there has been no limnological or palaeolimnological research at Llyn Pencarreg.

CHAPTER 4

METHODS AND TECHNIQUES

4.1 Field Techniques

4.1.1 Collection of the Long Cores

In March 1994, two 11m parallel sediment cores were collected by Wendy Johnson and team, Aberystwyth University, from closely adjacent boreholes in the deepest central water of the Upper Talley Lake, at a water depth of 4m (UT2/94a). A single 7m sediment core was collected in March 2005 from the deepest waters of Llyn Pencarreg at a depth of 10m (LPC/05a) by a team from the Universities of Hull, Birmingham and Aberystwyth. In both cases a rod-driven Livingstone piston corer (Livingstone, 1955) and anchored platform were used.

At Llyn Pencarreg a coring platform attached with rope to two inflatable boats (Figure 4.1) was stabilised with three anchors to ensure that a constant position was maintained. Water depth was measured with a hand-held echo sounder. The distance between the water surface and platform surface was calculated to determine the number of rods needed for each drive and the drive depth was marked on the rods before the corer was inserted through the central hole of the coring platform for each drive.

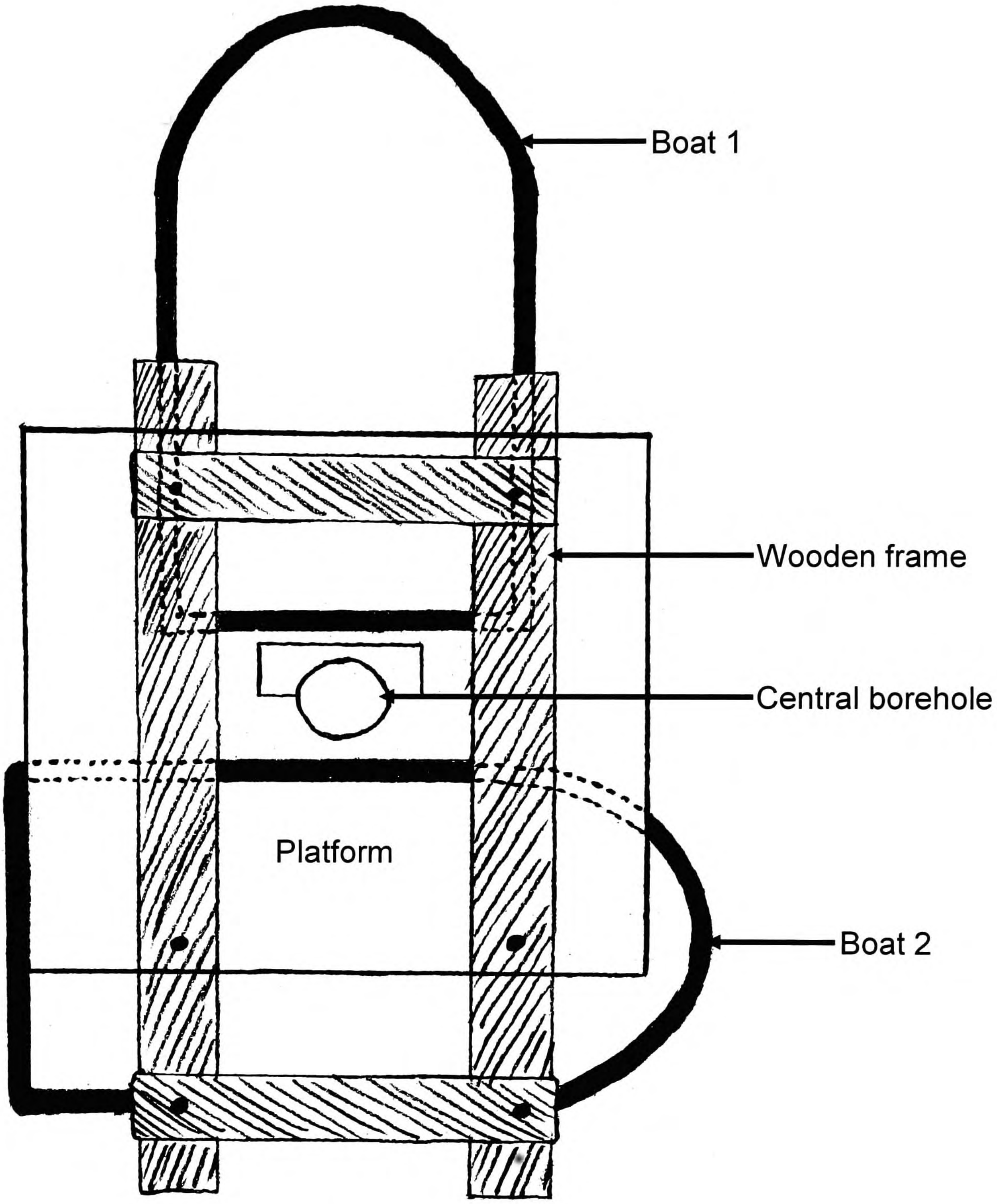


Figure 4.1 Schematic diagram showing the coring platform, raft and boats used to support the Livingstone corer. The raft is attached to the boats with luggage straps before fully inflated. (Modified from drawing provided by J. Hope, UCL)

When coring reached a depth of 4m, 8m of 12" drainpipe casing was inserted into the central coring hole to ensure that the corer extracted sections from the same borehole and to prevent distortion of the extension poles. A tripod and hoist provided mechanical aid to recovering sections as the drives became deeper and the force of suction on the corer increased. To prevent contamination of the core sediments the core barrel interior was cleaned following each drive by immersing the device in water and pulling the piston up to draw water through. The sediment sections were extruded into labelled 3" drainpipe, which had previously been cut in half lengthways, and each section was tightly wrapped in cling-film, with tissue blocking each end, to avoid drying (Wright, 1980) and bacterial fixation of CO_2 from the surrounding atmosphere, which can lead to younger radiocarbon age estimates (Geyh *et. al.*, 1974). The sections were then sealed with waterproof tape.

4.1.2 Collection of the Sediment Water Interface

At Upper Talley a shorter surface core of 70cm (UT2/94b), which encompassed the sediment-water interface, was collected from a water depth of 4m using a clear perspex tube fitted with a piston (Johnson, 1998). At Llyn Pencarreg a short surface core of 70cm (LPC/05b) was collected with a Glew messenger-operated gravity corer (Glew, 1991) from a water depth of 10m. In both cases the sediments were extruded in the field at a resolution of 0.5cm for the top 1cm and at a resolution of 1cm thereafter, and sub-samples were stored in labelled, sterile whirl-pack bags.

The core sections and surface samples from Upper Talley were stored in the University of

Aberystwyth cold store at a temperature of 4°C and Dr. Henry Lamb provided access to this material in March 2003. Core sections and surface samples from Llyn Pencarreg were kept in the University of Hull cold store at a temperature of 4°C.

4.1.3 Water Chemistry

Water samples were collected in acid-washed, sterile bottles, which were rinsed with lake water immediately before use. Samples were collected from the southern littoral zone of the Upper Talley on 22nd March 2005 and from the western littoral zone of Llyn Pencarreg on 24th March 2005. Dissolved oxygen (DO) and temperature were measured in situ on the same dates, using a VWR DO 200 probe, and pH using a VWR pH probe. Conductivity was measured with a HI 9033 conductivity meter. Unfiltered water samples were analysed for total phosphorus (TP) concentration within twelve hours of collection, using a Palintest Photometer 5000 with tube-test heater.

4.1.4 Collection of Modern Plant Samples

Modern plant samples were collected from the lake edge and local catchments of Upper Talley and Llyn Pencarreg, placed in labelled envelopes and wrapped in newspaper to prevent bacterial decomposition. Envelopes were kept in the University of Hull cold store at a temperature of 4°C and sent to the NERC (Natural Environment Research Council) Isotope Geosciences Laboratory (NIGL) in April 2006, for standard carbon isotope analysis (see Section 4.2.5) by Prof. MJ Leng. Results were compared to Meyers and Teranes (2001) classification of bulk organic matter from lacustrine algae and C_3 land plants.

4.2 Laboratory Techniques

4.2.1 Stratigraphic Description

The surface of cores UT2/94a and LPC/05a were cleaned by scraping horizontally with a scalpel to avoid vertical contamination. Colour and texture were described using a Munsell soil colour chart (MKDI, 1994) and the Troels-Smith (1955) classification scheme. Following visual examination the cores were photographed digitally for future reference before sub-sampling.

4.2.2 Magnetic Susceptibility

A type M. S. 2 Bartington magnetic susceptibility meter and core-scanning loop were used to measure 2 cm interval profiles of uncalibrated volume susceptibility at the Universities of Aberystwyth (UT2/94) and Birmingham (LPC/05). In each case the core sections were passed continuously through the scanning loop and drift was calculated automatically by blank (air) measurements at the beginning and end of each run. The Upper Talley magnetic susceptibility data extended to a depth of 11.50m, the top 9.90m are employed in this study to match the multi-proxy profile.

4.2.3 Sub-sampling

All equipment was thoroughly washed after taking each sub-sample to avoid contamination. UT2/94a was sliced into 2cm sub-samples, put into labelled, sterile whirl-pack bags and stored in the University of Hull cold store at 4°C. Subsample levels were correlated where possible with Johnson's (1998) pollen sequence. However, there was a 39cm gap in the sequence between 4.34 and 3.95m.

LPC/05a was sent to The University of Birmingham for sub-sampling for pollen analysis under the supervision of Dr. Warren Eastwood. On return to the Hull University it was noted that some drying and shrinkage had occurred due to careless re-wrapping of the core sections following pollen analysis. Correction factors were calculated for each core section by dividing the original section length (measured in the field) by the length following shrinkage. The correction values were used as a replacement measurement for each centimetre of core sediment before shrinkage (Table 4.1). Where excessive drying and shrinkage had occurred at the top and bottom of some core sections the pollen sub-sample points and core log data were used to find the original length and separate correction factors were calculated. Most of the pollen sub-sampling points were clearly visible so good correlation between subsamples was established with confidence. The correction lengths were measured out on each individual section before sub-sampling. Core sections were sliced into the equivalent of 2cm thickness and stored in sterile whirl-pack bags in the University of Hull cold store at a temperature of 4°C.

Drive;	Length of section in field (cm)	Length of section after shrinkage (cm)	Correction equation (cm)	Correction factor (cm)	Correction equation (cm) for desiccated ends	Correction factor (cm) for desiccated ends
I	90	84	84/90	0.93		
II	92	86	86/92	0.94		
III	86	75	75/86	0.87	62/86 (At basal end)	0.72
IV	80	70	70/80	0.87	55/80 (At basal end)	0.68
V	58	50	50/58	0.86	37/58 (At basal end)	0.63
VI	94	80	80/94	0.85		

Table 4.1 Table showing the correction factors applied to each section of core LPC/05a to overcome the problem of shrinkage.

Additionally, on return from the University of Birmingham the Glew core samples were missing. A second surface-core, which encompassed the sediment-water interface, was collected with a Glew messenger-operated gravity corer (Glew, 1991) from a water depth of 10m in July, 2005 by Dr. Warren Eastwood and team from the University of Birmingham. The second core, however, was only 22cm long, thus for the final analysis there was a gap in core sediments between 22cm and 1.22m at the top of the Livingston core

4.2.4 Loss on Ignition (LOI)

Sub-samples of approximately 1g were taken at a resolution of 4cm and dried overnight at 100°C. Each sample was weighed and heated by furnace combustion at 550°C for two hours in pre-weighed ceramic crucibles. Samples were cooled in a desiccator, to avoid the re-absorption of moisture, and weighed again. Each crucible weight was subtracted from the total weight of each sample and the remaining percentage weight difference was calculated using equation 1 (Heiri *et al.*, 2001).

$$LOI_{550}=((DW_{100}-DW_{550})/DW_{100})x100 \qquad (1)$$

Where;

LOI_{550} = percentage loss of dry weight following ignition at 550°C

DW_{100} = dry weight of the sample before combustion

DW_{550} = dry weight of the sample after heating at 550°C

The same samples were combusted at 850°C for a further two hours and cooled in a desiccator. The percentage weight difference, before and after combustion, was calculated for each sample to estimate CO_2 loss, using equation 2 (Heiri *et. al.*, 2001).

$$LOI_{850}=((DW_{550}-DW_{850})/DW_{100})x100 \qquad (2)$$

Where;

LOI_{850} = percentage loss of dry weight following ignition at 850°C

DW_{550} = dry weight of the sample after combustion of organic matter at 550°C

DW_{850} = dry weight of the sample after heating at 550°C

DW_{100} = dry weight of the sample before organic combustion

To convert CO_2 loss to carbonate content, this figure was divided by a molar conversion factor of 0.44 (Dean, 1974; Bengtsson and Enell, 1986). This assumes that all the carbonate present is calcium carbonate, which may not be the case.

4.2.5 Carbon Nitrogen (C/N) and Carbon Isotope ($\delta^{13}C$) Analysis

Sub-samples of 1cm^3 were taken at a resolution of 4cm, from the same depths as the LOI sub-samples, and stored in glass vials at a temperature of 4°C. In June 2005 samples were analysed by Prof. MJ Leng at NERC (Natural Environment Research Council) Isotope Geosciences Laboratory (NIGL). Samples were loaded into tin capsules, placed into the carousel of a Carlo Erba NA 1500 and dropped sequentially, in a continuous flow of helium carrier gas, into a 1020°C furnace. A pulse of oxygen gas was used to promote exothermal flash oxidation of each tin, ensuring further combustion of the sample, and the product gases were further oxidised by chromium and cobalt oxides in the lower part of the furnace. After the removal of excess oxygen and water, by passing through hot copper and magnesium perchlorate, the remaining N_2 and CO_2 were passed through a GC column and a thermal conductivity detector, which generates an electrical signal proportional to the concentrations of N_2 and CO_2 present in the helium stream. The data were evaluated by the Costech EAS software station with a Costech chromatography software package (EAS 1.7) to report %N and %C data for each sample. The weight percentages were calibrated against an Acetanilide standard and multiplied by the C and N atomic weight ratio (1.167) to produce the C/N atomic ratio.

Meanwhile, the helium stream carried the N_2 and CO_2 through a trap at 90°C (for the complete removal of water), before reaching a VG Triple Trap held at -196°C. Here the CO_2 was frozen allowing the N_2 and helium to vent into the atmosphere. The Triple Trap was then evacuated before warming the CO_2 trap and expanding the sample into the inlet of the Optima mass spectrometer, which has triple collectors allowing for the simultaneous monitoring of CO_2 ion beams at m/e = 44, 45 and 46; and a duel-inlet allowing the rapid comparison of sample CO_2 with reference CO_2. The 45/44 ion beam ratios were converted to $^{13}C/^{12}C$ ratios after correction for common ion effects ('Craig' correction). In each run approximately ten replicates of the laboratory standard (BROC1) were analysed and from knowledge of the laboratory standard's $\delta^{13}C$ value versus VPDB (belemnite carbon shell from the Cretaceous Peedee Formation of South Carolina) the $^{13}C/^{12}C$ ratios of the samples were converted to $\delta^{13}C$ value *versus* VPDB (Prof. M.J. Leng, *pers. comm.*).

The organic data (lithology, magnetic susceptibility, LOI and carbon isotope) were presented using TILIA (Grimm, 1991) and TG View (Grimm, 2004) apart from magnetic susceptibility from Upper Talley, which was scanned from Johnson (1998) and analogue matched by depth with the remaining organic data. Zone boundaries were defined by eye and labelled UTOZ (Upper Talley Organic Zones) and LPOZ (Llyn Pencarreg Organic Zones).

4.2.6 Diatoms (Bacillariophyceae)

Sub-samples were taken at a resolution of 8cm throughout both cores, increased to 4cm at key horizons, and dried in an oven overnight. 0.025g of sediment was measured from each dry sample and prepared following standard techniques (Battarbee, 1986). Organic matter was oxidised in hot 30% hydrogen peroxide (H_2O_2) by placing samples in beakers on a hotplate at a temperature of 90°C. A few drops of concentrated hydrochloric acid (HCl) were added to each sample following oxidation, to remove carbonates. Samples were then transferred to test tubes and washed with distilled water in a Centaur 2 centrifuge at a speed of 1200rpm for five minutes to allow the clay particles to form a suspension while the diatoms and other silt-sized particles formed a pellet at the base of the test tubes. The water was decanted after each wash leaving the pellet intact. This process was repeated until all clays had been removed (i.e., when supernatant ceased to be cloudy). Individually numbered plastic pipettes were used to mix each sample to avoid cross contamination. Samples were then topped up with distilled water to 10ml.

A microsphere solution (supplied by UCL Geography Department) for calculating absolute abundance of diatom frustules (Battarbee, 1982) was mixed to a concentration of 6.81×10^7 microspheres ml^{-1} and agitated in a sonic bath before use to prevent clumping.

Using an Eppendorf pipette, 1ml aliquot measures of the solution were added to each test-tube immediately prior to use. Sample residues were dried overnight on 40mm round cover slips then mounted onto microscope slides using Naphrax®, which has a suitable refractive index (Cooper, 1999). Slides were heated on a hotplate in the fume cupboard at a temperature of 90°C for ten minutes, to evaporate the toluene from the Naphrax®, and then cooled before counting.

Taxonomic identification was made using a Leica DMLS under oil immersion at a magnification of x 1000 and determined to species level using Sims (1996), Barber (1981), Germain (1981) and Krammer and Lange-Bertalot (1986, 1988, 1991a, b). Approximately 500 valves were counted per slide along transects to provide a statistically reliable estimate of species composition and a tally was made of the number of microspheres observed per count. The diatom data were displayed using TILIA (Grimm, 1991) and TG View (Grimm, 2004). Biostratigraphic zone boundaries were defined by application of the constrained incremental sum of squares clustering technique, using CONISS (Grimm, 1987) on square root transformed data and zones were referred to as UTAZ (Upper Talley Algal Zone) and LPAZ (Llyn Pencarreg Algal Zone). CONISS (Grimm, 1987) was not displayed on the final diagram, due to lack of space. Species' TP optima were taken from the European Diatom Database Initiative (EDDI) Combined European TP transfer function (Battarbee *et al.*, 2000) and trophic classification was taken from Vollenweider and Kerekes (1981) model, which is based on the relationship between inflow P-loads, water residence time and the resultant P and chlorophyll *a* concentrations.

4.2.6.1 Data Analysis

The concentration of diatoms in each sample was calculated using equation 3.

$$X = (YxN_1/N_2)x40 \qquad (3)$$

Where;

X = diatom concentration

Y = the number of microspheres initially introduced

N_1 = the number of diatoms counted

N_2 = the number of microspheres counted

40 = the correction factor used to enable the final data to be expressed as microsphere concentration per gram of dry sediment originally prepared.

Diatom concentrations were expressed as the number of valves per 1g of dry sediment and presented with the percentage diatom data under the defined Algal Zones. To test whether fluctuations in diatom concentrations were the result of changes in sediment accumulation rate, diatom concentration (DC) was divided by pollen concentration (PC) and compared graphically to the original DC trend (after Bennett *et al.*, 1990). This approach assumes that pollen influx is constant and that variations in PC are solely due to variations in sediment accumulation at the core location. Although this works best when the pollen signal indicates a fully forested landscape (with arboreal pollen at > 90%), work by Broström *et al.* (2005) indicates that the method also works in more open landscapes. An alternative approach is to divide sediment accumulation rates (SAR) by DC to convert DC into diatom accumulation rates (DAR). However, the lack of dates at sample depths means that changes in the profile will be abrupt and constant rates will be assumed between dates.

Diatom-inferred total-phosphorus (DI-TP) was reconstructed by application of the EDDI Combined European TP transfer function (Battarbee *et al.*, 2000). The combined TP dataset was chosen as this was developed from, and validated for, a broad range of sites across Europe and so provided a long TP gradient. To derive the transfer function from the raw data the species optima of the modern training set were estimated by regression using C2 version 1.4.2 (Juggins, unpublished) and diatom inferred TP (DI-TP) values were estimated from the fossil diatom data using weighted averaging with tolerance down-weighting and inverse de-shrinking (WATOL _ Inv) calibration methods, as this produced the lowest prediction error. DI-TP values were back-transformed to $\mu g\ l^{-1}$ from $\log_{10}$ values and presented as part of the algal stratigraphy.

The sample specific errors of prediction and confidence intervals were estimated by bootstrapping, which removes random groups of samples from the dataset and re-examines the performance of the reconstruction. Prediction errors are given as root mean square errors of prediction (RMSEP (boot)). The 95% confidence intervals were displayed against the reconstruction at each sample depth to assess predictive ability. The Hills N_2 Diversity Index for each fossil sample, calculated in C2, was used as a measure of species diversity of fossil samples.

To further explore species variation, detrended correspondence analysis (DCA) was applied to the percentage diatom data using the program CANOCO version 4.5 (ter Braak and van Dam, 1989). This technique assumes a unimodal response, which is appropriate for large data-sets with many zero values. For both sites the gradient length of Axis 1 was >2.5 SD units (2.7 SD units for UT; 3.1 SD units for LP), indicating that DCA rather than correspondence analysis was the most appropriate technique (Jongman *et al.* 1995). Analysis was performed by detrending by segments and non-linear rescaling on 62 samples and 114 species for Upper Talley and 69 samples and 47 species for Llyn Pencarreg. Sample scores were scaled as weighted mean species scores. Results were displayed graphically using the program C2 version 1.4.2 (Juggins, unpublished). Species and sample distribution was then interpreted based on a knowledge of contemporary diatom ecology.

4.2.7 Cladocera Ephippia

The study intended to use cladocera analysis to complement diatom analysis, but the high organic content of samples made the cleaning process slow, intricate and impractical within the given timeframe. Tests were made to find the most appropriate heating times and sieve sizes but problems were encountered with loosing body parts in the process of sediment removal or body parts being obscured by organic material under gentler treatment, particularly at the base of the Llyn Pencarreg core. The same problems arose when preparing samples for ephippia analysis and it was therefore decided that counting in dry sediment was appropriate.

Alternate 2cm slices (providing *ca.* 20g of sediment per sample) were wet sieved at 150μm under warm running water. Samples were dried overnight in petri dishes and residues examined under a microscope at a magnification of x 63 by sorting using a metal seeker. Although ephippia can be identified to species level (Sarmaja-Korjonen, 2004) time was a limiting factor so the total ephippial sum was counted instead. A note was made of the frequency of double yolked ephippia, which appeared at various points of the Upper Talley sequence. Total abundance for each sample was divided by 20 and presented as number of ephippia per g^{-1} dry sediment, displayed using TILIA (Grimm, 1991) and TG View (Grimm, 2004).

4.2.8 Pollen, *Pediastrum* and Charcoal

Sub-samples of $0.5cm^3$ were taken at a resolution of 8cm to a depth of 10m for UT2/94 and 6.80m for LPC/05. A higher resolution of 4cm was employed at key horizons. Sub-samples were taken from both Glew cores at a resolution of 1cm and every 0.5cm for the top cm of each core

Samples were prepared using standard pollen techniques (Moore *et al.*, 1991) at the University of Aberystwyth (pollen) and Hull University (*Pediastrum* and charcoal) for core UT2/94 and The University of Birmingham (pollen) and Hull University (*Pediastrum* and Charcoal) for core LPC/05. In all cases preparation included the addition of one *Lycopodium* tablet, dissolved in 10% hydrochloric acid (HCl), a sodium hydroxide (KOH) treatment to remove humic acids, sieving with a 150μm mesh to remove large organics, washing with hot sodium pyrophosphate to remove clay particles and a hydrofluoric acid (HF) treatment to remove silica. This was followed by acetolysis to remove lignin and cellulose. A few drops of 0.2% aqueous safranin solution were added to each sample, to enhance the characteristics of pollen grains and *Pediastrum* colonies, before washing with tertiary-butyl alcohol (TBA) to dehydrate the samples. The residues were mounted on microscope slides with glycerine jelly (UT2/94) or silicon oil (LPC/05). Pollen identification was carried out by Johnson (1998) to a depth of 7.50m for the UT2/94 sequence and by Andrew Moss (University of Birmingham) to a depth of 6.90m for LPC/05. The Upper Talley data were presented by Johnson (1998) as part of a palynological study that also incorporeted a second core from the Lower Talley lake, which lies adjacent to the Upper Talley lake.

The percentage pollen data were displayed using TILIA (Grimm, 1991) and TG View (Grimm, 2004). Biostratigraphic zone boundaries were defined by application of the constrained incremental sum of squares clustering technique, using CONISS (Grimm, 1987) on square root transformed data and zones were referred to as UTPZ (Upper Talley Pollen Zone) and LPPZ (Llyn Pencarreg Pollen Zone). Land pollen data was limited to human indicators to highlight human impact within the lake catchments (e.g. deforestation and agricultural activity), whilst non arboreal pollen (NAP) was expressed as a percentage of total land pollen (TLP) and presented graphically to provide an estimate of changes in land-use for the length of the core. The percentage aquatic pollen and spore data were also presented with the limnological data in the final multiproxy analysis to enable the comparison of changes in habitat availability with the process of eutrophication.

4.2.8.1 Data Analysis

Charcoal particles were counted using the Clark (1982) point count method under a Leica DMLS at a magnification of x 400. The systematic approach involved counting the number of charcoal 'hits' on fifty random fields of view per slide or continuing until at least ten *Lycopodium* grains were observed. Charcoal concentration was calculated using equation 4 (Clark, 1982).

$$C = \frac{h.Af.L.W}{p.l} \qquad (4)$$

Where;

h = number of hits

Af = area of field of view (0.00233 cm^2)

L = number of *Lycopodium* spores added to each sample (10,679 per tablet)

l = number of *Lycopodium* spores counted in each sample

p = number of points on the graticule (202)

W = volume of sediment processed (0.5cm^3)

The resulting charcoal data, expressed in cm^2 cm^{-3}, were presented using TILIA (Grimm, 1991) and TG View (Grimm, 2004) by comparison with selected pollen taxa. Major stratigraphic boundaries were defined by eye based on changes in abundance.

For *Pediastrum* analysis twenty-four traverses were made per slide using a Leica DMLS at a magnification of x 400. A tally of *Lycopodium* grains was made at the same time and *Pediastrum* abundance was estimated using equation 5.

$$P = ((Lt / Lc)xPo) \times V \tag{5}$$

Where;

P = *Pediastrum* abundance,

Lt = Number of *Lycopodium* spores in each tablet added to a single sample,

Lc = Number of *Lycopodium* spores counted for each sample,

Po = *Pediastrum* colonies observed on the same slide,

V = Volume of sediment processed.

Concentration was expressed as the number of colonies present per cm^3 of sediment originally prepared. Identification was made to species level using Komárek and Jankovská (2001) and specific environmental preferences were taken from Jankovská and Komárek (2000) and Komárek and Jankovská (2001). Results were displayed using TILIA (Grimm, 1991) and TG View (Grimm, 2004) and presented with the diatom data. The concentration of pollen grains in each sample (pollen grains/ cm^{-3}) was calculated using equation 6.

$$\mathrm{N} = (\frac{p}{e} \times \mathrm{X}) \times V \tag{6}$$

Where;

N = Number of pollen grains per cm^{-3}

p = Number of pollen grains counted in each sample

e = Number of *Lycopodium* counted in each sample

X = Number of *Lycopodium* in each sample

V = Volume of sediment processed

The concentration for each sample was divided by the original sample volume to produce the total number of pollen grains per cm^3.

To test whether Upper Talley exhibited evidence for alternate stable states, the aquatic pollen sum (APS), determined by calculating the total percentage abundance of aquatic pollen every 4-8cm, was compared to diatom concentrations (DC) for the same depths, to identify shifts between algal (DC) and plant (APS) dominance. Major phases of human impact were also marked for comparison.

The Arable/ Pastoral Index, developed by Turner (1964) to reflect contrasts in modern agricultural land use at different sites, was used to summarise catchment farming patterns in each sequence. The index was derived by expressing the number of *Plantago* grains as a percentage of the combined number of pollen grains from *Plantago*, *Artemesia*, Chenopodiaceae, Compositae (Asteraceae) and Cruciferae (Brassicaceae) (Turner, 1964). Results were displayed graphically and presented with the percentage NAP curve to assess changes in land use. Turner's (1964) boundaries for arable (> 15%) and pastoral (<50%) regions were used to assess agricultural land-use and values between 15% and 50% were taken to represent a combination of both types. Phases of human occupation were marked to examine the relationship between human activity and changes in land-use and allow for between-site comparison of agricultural patterns.

4.3 Sediment Accumulation Rates

Sediment Accumulation Rates (SAR) were estimated using 'psimpoll' (Bennett, 1994b), which produces values using individual radiocarbon dates, assuming constant rates of sediment accumulation between dated sample depths.

4.4 Radiocarbon Dates

4.4.1 AMS Dates: Upper Talley and Single Sample from Llyn Pencarreg

The sampling strategy for ^{14}C radiocarbon dating aimed to provide maximum chronological resolution within the constraints of limited funding. Alternate 2cm slices from UT2/94 were wet sieved at 250μm under warm running water. Residues were dried overnight in petri dishes and examined under a microscope at a magnification of x 63. Specimens suitable for radiocarbon analysis were removed using a fine, dampened plastic bristled paintbrush, cleaned with distilled water and put into individual glass vials then stored in the University of Hull cold store at 4°C. Table 4.2 gives a list of materials extracted from core UT2/94. Plant macrofossils were stored in a dry environment to avoid bacterial and fungal activity and the absorption of CO_2 (Wohlfarth *et al.*, 1998).

In December 2004, the Upper Talley samples marked * in Table 4.2 (1 x *Corylus avellana*, 1 x wood fragment, 1 x charcoal taken from 3 samples between 2.52m and 2.62m and 1 x *Potamogeton* seeds) were submitted to the SUERC Dating Laboratory in East Kilbride for AMS radiocarbon dating. Emphasis was given to terrestrial organics weighing between 2-10mg in order to provide enough carbon for analysis. For this reason the charcoal from M63a, M64a and M65a were submitted as one sample. Justification for the selected horizons is given in Table 4.3.

From Llyn Pencarreg, a sample of well preserved wood from the base of LPC-05 was submitted to SUERC for AMS radiocarbon dating in June 2005, to provide initial chronological assessment.

4.4.2 Bulk AMS Dates: Llyn Pencarreg

Apart from the piece of wood no other material suitable for AMS dating was found in core LPC/05 so a chronology was obtained instead by bulk sediment matrix dating. This was not ideal as it can incorporate a mixture of organic matter from a number of sources and could produce erroneous dates (O'Sullivan *et al.*, 1973). 4g samples were taken using a metal spatula, which was cleaned in distilled water between samples, and placed into plastic whirl pack bags. Samples were stored in the University of Hull cold store at 4°C. Justification for the selected horizons is given in Table 4.4.

4.4.3 Age-depth Model

Age-depth curves were constructed for both cores using linear interpolation (Telford *et al.*, 2004). When referred to in the text all inferred ages were rounded off to the nearest 100 years, i.e. estimated time resolution of the century scale. Although this made it difficult to date phases of limnological change exactly, a major aim of the study was to reconstruct ecosystem dynamics during the process of eutrophication, thus the timeframe produced enough certainty to locate major phases of occupation (e.g. Bronze Age, Medieval period) as reasons for long-term limnological change.

Depth (m)	**Sample No.**	**Material**	**Weight (g)**
0.44-0.46	M11a	8 x *Potamogeton* seeds	0.0082
0.92-0.94	M23a	6 x *Potamogeton* seeds	0.0045
0.96-0.98	M24a	3 x *Potamogeton* seeds	0.002
1.04-1.06	M26a*	7 x *Potamogeton* seeds	0.0067
1.16-1.18	M29a	3 x *Potamogeton* seeds	0.002
2.52-2.54	M63a*	Charcoal	0.0027
2.56-2.58	M64a*	Charcoal	0.0026
2.60-2.62	M65a*	Charcoal	0.0018
4.60-4.62	M115a	Pine bud	0.006
5.24-5.26	M131a*	Wood	0.0063
6.12-6.14	M153a	2 x *Potamogeton*	0.0033
7.28-7.30	M182a	Wood	0.0085
7.32-7.34	M183a	*Alnus* cone	0.0127
7.96-7.98	M199a	Burnt organics	0.0078
8.16-8.18	M204a	Wood	0.04
9.28-9.30	M232a	Degraded seeds	0.0121
9.36-9.38	M234a	Degraded seeds	0.0079
9.62-9.64	M242a*	Nut (*Corylus avellana*)	1.06

Table 4.2 Table showing the materials extracted from core UT2/94 for radiocarbon dating.

Sample Depth (m)	**Material**	**Justification**
1.04-1.06	*Potamogeton* seeds	To check for a possible carbon reservoir effect (Peglar *et al.*, 1989) and provide a chronology for the top of the core.
2.52-2.62	Charcoal	Carbon isotope results indicate a significant increase in algal biomass and an increase in rates of primary production at this point. Increase in diatom abundance and peak in ephippia. Decrease in arboreal pollen and introduction of agricultural indicators.
5.24-5.26	Wood	Extremely well preserved specimen.
9.62-9.64	Nut (*Corylus avellana*)	To fix chronology for the core base. Extremely well preserved specimen.

Table 4.3 Table justifying the horizons selected for radiocarbon dating for the Upper Talley core (UT2/94).

Sample Depth (m)	**Justification**
0.02 – 0.03	To fix chronology for the top of the core
1.36 – 1.38	Peak in *Aulacoseira granulata*, DI-TP, diatom concentration and ephippia. Decrease in tree pollen.
3.40 – 3.42	Decrease in *Tabellaria flocculosa* and diatom concentration and increase in *Fragilaria exigua* and *F. virescens*. Decrease in grass and herb pollen and increase in arboreal pollen.
5.44 – 5.46	Major shift in planktonic and benthic diatom taxa, decrease in DI-TP and increase in diatom concentration. Decrease in tree and shrub pollen, expansion of grass and herb pollen and increase in aquatic macrophyte pollen.
6.72 – 6.73	To compare with the AMS date produced from a layer of wood and fix chronology for the core base.

Table 4.4 Table justifying the horizons selected for radiocarbon dating for the Llyn Pencarreg core (LPC/05).

CHAPTER 5

RESULTS FROM THE UPPER TALLEY CORE (UT2/94)

The results from Upper Talley are presented in the following chapter. Chronological data are reported based on the age-depth model in Section 5.18, at the end of the chapter.

5.1 Water Chemistry

A set of basic water chemistry measurements taken in March 2005 indicated that Upper Talley is circumneutral (pH 6.6), and hypereutrophic, with a total phosphorus (TP) concentration of 100 $\mu g\ l^{-1}$. In spite of this, the waters are well oxygenated (DO; 93.3%), with low conductivity (91 $\mu S\ cm^{-1}$).

5.2 Lithology

The lithology description (Table 5.1) indicates that from the core base the sediments became increasingly organic rich, with a gradual transition from light to very dark brown, consolidated, diatomaceous lake mud, which was at its darkest between 7.00m and 6.75m. Plant macrofossils comprising seeds, leaves and twigs were present, with a well-preserved hazelnut (*Corylus avellana*) at a depth of 9.62m. Between 6.75m and 6.00m a gradual transition to dry, friable, homogenous lake mud occurred with the appearance of

Depth (m)	Boundary Depths (m)	Laboratory Sediment Description	Munsell Description (MDKI, 1994)	Troel-Smith Notation (Troels-Smith, 1955)
0 – 1.00 -		Fine organics with traces of silt and clay. Insect remains and charcoal present.		
2.00 – 3.00 –	2.54 - 2.50	Gradual transition to soft, brown/green, homogeneous gyttja.	Olive brown; 2.5YR 4/4	Ld.3, As.1, Ag+, nig.2, strf.0, sicc.2
4.00 – 5.00 – 6.00 –	4.50 - 4.30	Gradual transition to dark brown, fibrous, organic rich, homogeneous mud.	Dark brown; 10YR 3/3	Lso.2, As.2, Ag+, nig.3, strf.0, sicc.2, humo.2
6.50 – 7.00 –	7.00 – 6.50	Gradual transition to dry, friable lake mud with patches of clay.	Brown; 10YR 5/3	Lso.2, As.2, Ag+, nig.3, strf.0, sicc.2, humo.1, Lf.2
8.00 – 9.00 –		Plant macrofossils present with well-preserved hazelnut at 9.62m.	Very dark brown; 10YR 2.5/2	Lso.2, As.2, Ag+, nig.3, strf.0, sicc.4, elas.2, humo.2
10.00-		Diatomaceous lake-mud becoming increasingly dark and organic rich towards the upper zone boundary.	Brown; 10YR 5/3	Lso.3, As.1, Ag+, nig.2, strf.0, sicc.4, elas.3

Table 5.1 A detailed lithology describing the changes in colour and texture of core UT2/94

light brown patches of clay between 6.50m and 6.00m. Between 4.50m and 4.30m the sediments became increasingly dark brown and organic rich with a fibrous texture. A final, gradual transition produced soft, green-brown, homogenous gyttja between 2.54m and 2.50m, which continued to the top of the core. Traces of silt and sand were also found in the upper sequence along with the appearance of macro-charcoal and insect remains.

5.3 Magnetic Susceptibility

The magnetic susceptibility results for the core sections from 9.90m to the sediment-water interface are given in Figure 5.1 (taken from Johnson, 1998). Apart from high values for the basal *ca.* 50cm of the sequence, magnetic susceptibility values were relatively stable. A minor increase began at a depth of 3.75m, which peaked at a depth of 2.20m. This was followed by a brief fall then a second increase, which began at 1.60m and continued to the upper zone boundary where values decreased sharply.

5.4 Organic Content

In UTOZ-1 (Figure 5.1) organic content was low at the core base (20%), rising to 54% at a depth of 9.66m then falling sharply to 35% at the top of the zone. A consistent rising trend subsequently continued throughout UTOZ-2 to a depth of 6.22m in UTOZ-3 where a maximum percentage of 67% was produced. Organic content then decreased sharply to 34% at a depth of 5.95m, followed by an increase to 61% at 5.74m.

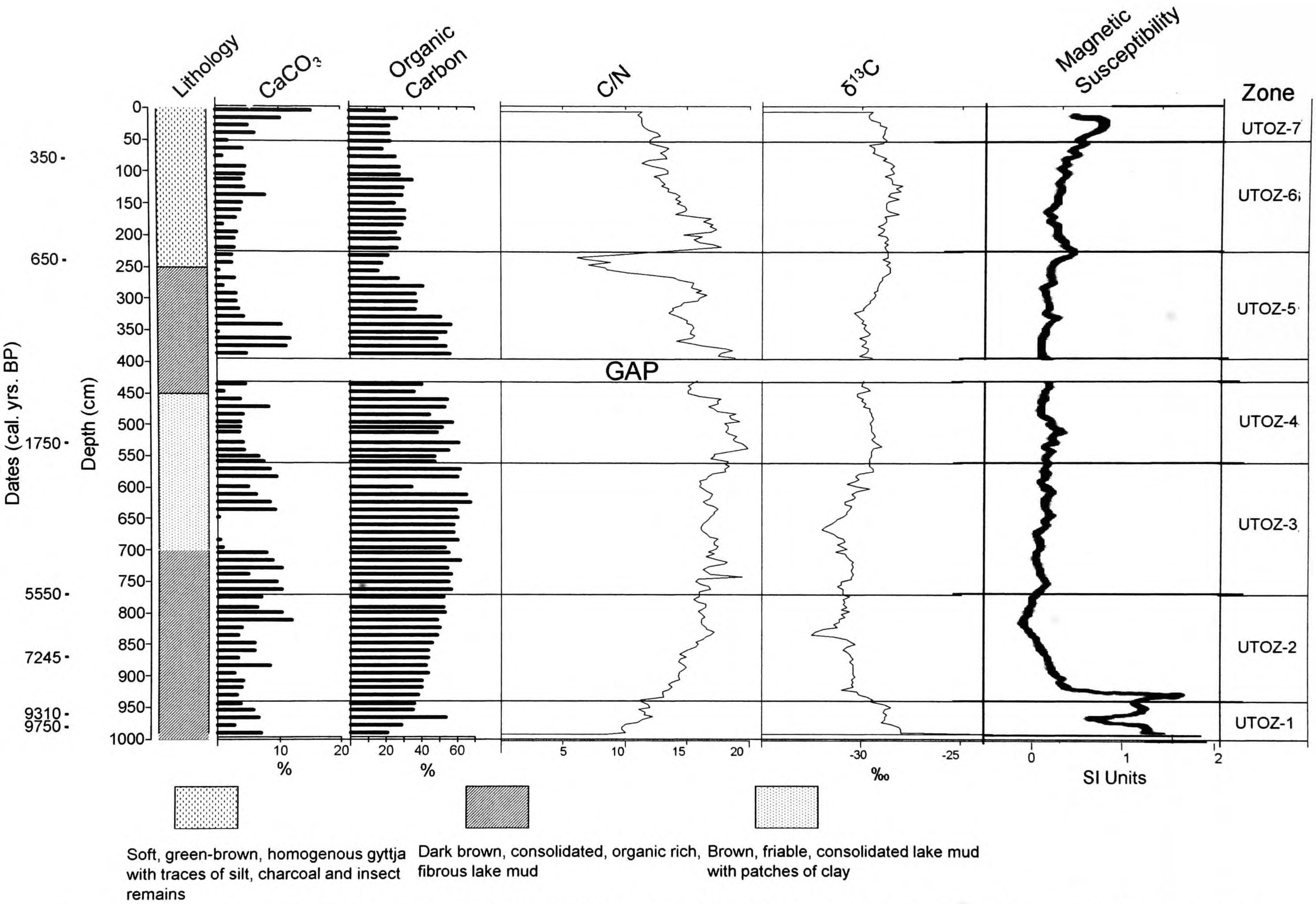


Figure 5.1 Diagram showing the lithological results for core UT2/94 including sediment lithology, % $CaCO_3$ (calculated from LOI at 850°C), % organic carbon (LOI at 550°C), C/N ratios , $\delta^{13}C$ and magnetic susceptibility values (taken from Johnson, 1998).

In UTOZ-4 percentages fluctuated significantly between a maximum of 60% and a minimum of 35% with an overall decreasing trend to 34% (4.34m). A general decreasing trend continued throughout UTOZ-5, from 45% at a depth of 3.95m to 16% at a depth of 2.50m, producing the minimum percentage of the entire sequence. Values immediately increased to 28% at the top of the zone (2.3m), followed by a general increasing trend to 35% at 1.10m in UTOZ-6. Finally, above 1.10m and throughout UTOZ-7, organic content fluctuated but remained relatively low between a maximum of 26% and a minimum of 20% at the top of the core.

5.5 Calcium Carbonate ($CaCO_3$)

$CaCO_3$ content fluctuated for most of the sequence (Figure 5.1). In UTOZ-1 a base percentage of 7.0% marked the start of a decreasing trend which continued into UTOZ-2, culminating in a value of 3% at 8.20m. This was followed by values fluctuating between 5-12% to a depth of 7.0m in UTOZ-3. Between 7.0m and 6.30m values decreased to the minimum of the core (1%) followed by a sharp increase to 9% at 6.40m. Percentages in the upper part of UTOZ-3 fluctuated between 5% and 10%. In UTOZ-4 $CaCO_3$ was relatively stable with an overall decreasing trend to 2% at a depth of 4.50m. Following an increase to 12% at 3.60m, low values (mean = 4%) continued into UTOZ-6 with an increase to 8% at 1.30m. Finally, in UTOZ-7, values increased from 2% at the zone base to 15% at the top of the core, producing the maximum of the entire sequence (15%)

5.6 Carbon: Nitrogen (C/N) Ratio

In UTOZ-1 (Figure 5.1) a C/N ratio of 10.0 is observed at the core base followed by a steady increase to a value of 17.0 at a depth of 8.30m in UTOZ-2. Values are then relatively stable varying between 15.7 and 19.9 with a mean of 17.0. This continued into UTOZ-3 with an increase to 17.5 at the top of the zone. In UTOZ-4 C/N ratios decreased from a value of 19.5 (the maximum value of the core) at 5.30m to 16.7 at a depth of 4.34m. From 3.95m in UTOZ-5, C/N ratios decreased from 18.7 to 14.1 at a depth of 3.20m, then increased to 17.0 at 3.00m, followed by a sharp decrease to 6.1 (2.30m). This, the minimum recorded for the sequence, was immediately followed by a sharp increase to 16.2 at the base of UTOZ-6. In UTOZ-6 and UTOZ-7 values formed a decreasing trend to11.3 at the top of the core.

5.7 Stable Carbon Isotopes ($\delta^{13}C$)

The results of $\delta^{13}C$ analysis are given in Figure 5.1. In UTOZ-1 an initial value of –22.0 ‰ at the core base decreased sharply to –29.0 ‰ at a depth of 9.70m then more gradually to –30.5 ‰ at a depth of 9.20m in UTOZ-2. Between 9.00 and 6.75m relatively stable values were recorded between a minimum of –29.0 ‰ and a maximum of –31.0 ‰, apart from a decrease to the minimum value of the entire sequence (-32.5) at 8.50m (UTOZ-2). In the upper part of UTOZ-3, $\delta^{13}C$ values increased to –29.0 ‰ at a depth of 5.60m. UTOZ-4 displayed a gradual decreasing trend, which continued into UTOZ-5 where a value of –30.4 ‰ was produced at 3.18m. A sudden increase to –28.6 ‰ at 2.54m was followed by relatively stable values around a mean of –28.7 ‰ which were observed for the remainder of UTOZ-5 and into UTOZ-6 to a depth of 1.26m. Above this latter depth, $\delta^{13}C$ values decreased to –28.1 ‰ at the top of the zone. In UTOZ-7 values finally decrease from –28.5 ‰ at the zone base to a value of –29.5 ‰ at the top of the core.

5.8 Carbon Isotope Analysis of Modern Plant Samples

The results of carbon isotope analysis of modern vegetation (Table 5.2 and Figure 5.2) indicated that the $\delta^{13}C$ values of individual species fell within the expected range (–25.0 ‰ to –39.0 ‰) (Meyers and Teranes, 2001). C/N ratios, however, did not clearly represent the expected signatures for algae or C_3 land plants; algae produced a high C/N ratio of 13.6, whilst terrestrial *Urtica* was lower than expected (6.9). Aquatic macrophytes fell within a wide range of values predicted by Boutton (1991), with *Sparganium* producing a low C/N ratio of 6.6 and *Phragmites* producing the highest value (37.7).

Plant Type and date of collection	**$\delta^{13}C$ (‰)**	**C/N Ratio**
Sparganium (22/03/05)	-28.2	6.6
Urtica (22/03/05)	-30.2	6.9
Moss from inlet (22/03/05)	-30.3	7.3
Algae from lake edge (22/03/05)	-27.8	13.6
Moss from lake edge (24/03/05)	-28.9	19.4
Poaceae (24/03/05)	-30.1	8.2
Sphagnum (24/03/05)	-29.2	22.1
Juncaceae (22/03/05)	-30.4	21.9
Phragmites (24/03/05)	-29.5	37.7

Table 5.2 Table showing the $\delta^{13}C$ (‰) and C/N composition of modern vegetation samples from Upper Talley lake and its immediate catchment.

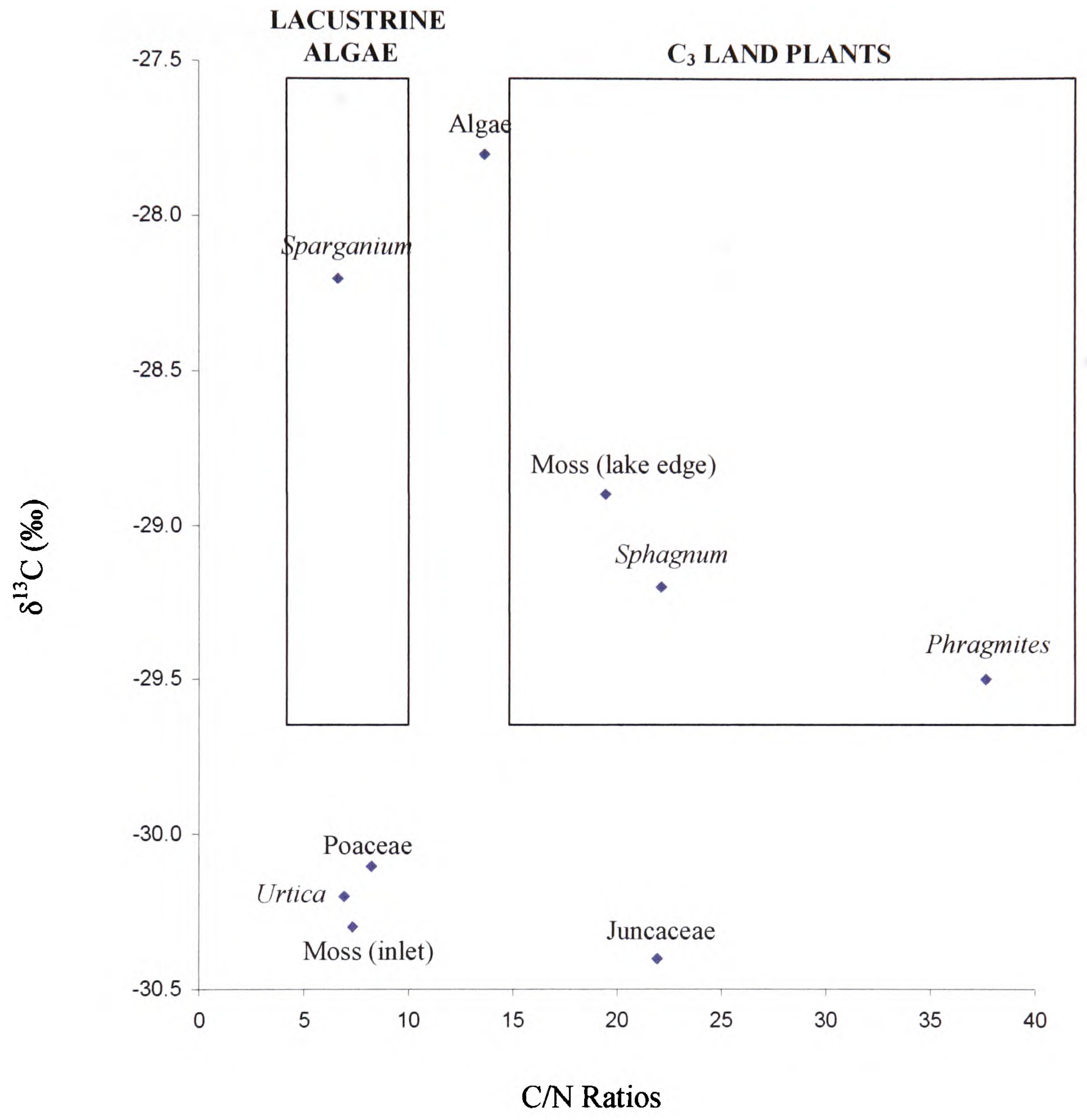


Figure 5.2 Diagram showing the elemental and carbon isotope compositions of organic matter in modern vegetation samples from Upper Talley lake and the local catchment compared to Meyers and Teranes (2001) classification of bulk organic matter from lacustrine algae and C_3 land plants.

5.9 Diatom Analysis

Diatom preservation was excellent throughout, with a total of 114 taxa present. 31 taxa were present at >4% in at least one sample (Table 5.3). The results of diatom analysis, expressed as percentage counts for taxa occurring at >4%, are given in Figure 5.3. Zones defined by CONISS were named Upper Talley Algal Zones (UTAZ).

UTAZ-1: 9.90 – 9.80m (*ca.* 9.7 - 9.3 k cal. yrs. BP)

The diatom assemblage at the base of the core mainly comprised benthic taxa. These included *Sellaphora pupula*, *Navicula rhyncocephala*, *Navicula radiosa*, *Gomphonema parvulum*, *Gomphonema gracile*, *Eunotia bilunaris*, *Cymbella silesiaca*, *Cymbella elginensis*, *Anomoeoneis vitrea*, *Achnanthes minutissima* and *Cocconeis placentula*. A range of benthic Fragilariales (*Fragilaria construens* var *venter*, *F. pinnata*, *F. ulna*, *F. vaucheriae*, *F. virescens* and *F. exigua*) were also present. Of the planktonic taxa, *Tabellaria flocculosa* was the most common at 20%.

UTAZ-2: 9.80 – 8.50m (*ca.* 9.3 - 6.9 k cal. yrs. BP)

A marked and abrupt transition occurred at the zone boundary to an assemblage dominated by planktonic *Aulacoseira granulata*, *Stephanodiscus minutulus* and *Cyclotella stelligera*. The relative abundance of *A. granulata* increased at the expense of *C. stelligera*, reaching a maximum of 80% at a depth of 9.20m.

Species Name	**EDDI Code**	**TP Optima ($\mu g\ l^{-1}$) transformed from $\log_{10}$**	**Trophic Preference**	**Life Form**
Achnanthes minutissima	XXG997	24.5	Mesotrophic	Benthic
A. lanceolata	XXG978	123.0	Eutrophic	Benthic
A. subatomoides				
Anomoeoneis vitrea	AN9999	N/A	Meso-eutrophic	Benthic
Asterionella formosa	AS001A	44.0	Meso-eutrophic	Planktonic
Aulacoseira distans	XXG956	12.5	Mesotrophic	Planktonic
A. granulata	XXG988	112.2	Hypereutrophic	Planktonic
Cocconeis placentula	XXG977	132.0	Hypereutrophic	Benthic
Cyclotella distinguenda var. *unipunctata*	XXG992	13.5	Meso- eutrophic	Planktonic
C. meneghiniana	CY003A	251.0	Hypereutrophic	Planktonic
C. ocellata	XXG991	34.0	Eutrophic	Planktonic
C. pseudostelligera	XXG971	40.0	Eutrophic	Planktonic
C. stelligera	XXG968	19.0	Eutrophic	Planktonic
Cymbella silesiaca	CM103A	16.2	Meso-eutrophic	Benthic
C. ventricosa	CM001A	91.2	Eutrophic	Benthic
Eunotia bilunaris	XXG949	15.5	Mesotrophic	Benthic
E. pectinalis	XXG946	33.5	Meso-eutrophic	Benthic
Fragilaria capucina var. *gracilis*	N/A	N/A	Meso-eutrophic	Planktonic/ Tycoplanktonic
F. construens var. *venter*	FR002C	51.50	Meso-eutrophic	Benthic
F. crotonensis	XXG985	30.0	Meso-eutrophic	Benthic
F. exigua	XXG961	16.0	Mesotrophic	Benthic
F. pinnata	XXG995	34.0	Meso-eutrophic	Benthic
F. ulna	XXC917	N/A	Meso-eutrophic	Benthic
F. vaucheriae	FR007A	53.0	Eutrophic	Benthic
F. virecens	XXG961	16.0	Mesotrophic	Benthic
Gomphonema angustatum	XXG978	N/A	Meso-eutrophic	Benthic
G. parvulum	XXG938	64.5	Meso-eutrophic	Benthic
Navicula radiosa	NA003A	30.2	Mesotrophic	Benthic
N. rhyncocephala	NA008A	85.8	Eutrophic	Benthic
Stephanodiscus minutulus	XXG974	40.0	Eutrophic	Planktonic
Tabellaria flocculosa	XXG981	15.0	Mesotrophic	Planktonic

Table 5.3 Diatom taxa recorded at >4% total diatom abundance in at least one sample of core UT2/94 with their EDDI codes (Battarbee *et al.*, 2001), TP optima (Battarbee *et al.*, 2001), trophic preference (Vollenweider and Kerekes, 1981) and ecological life form (Barber, 1981; Germain, 1981; Sims, 1996).

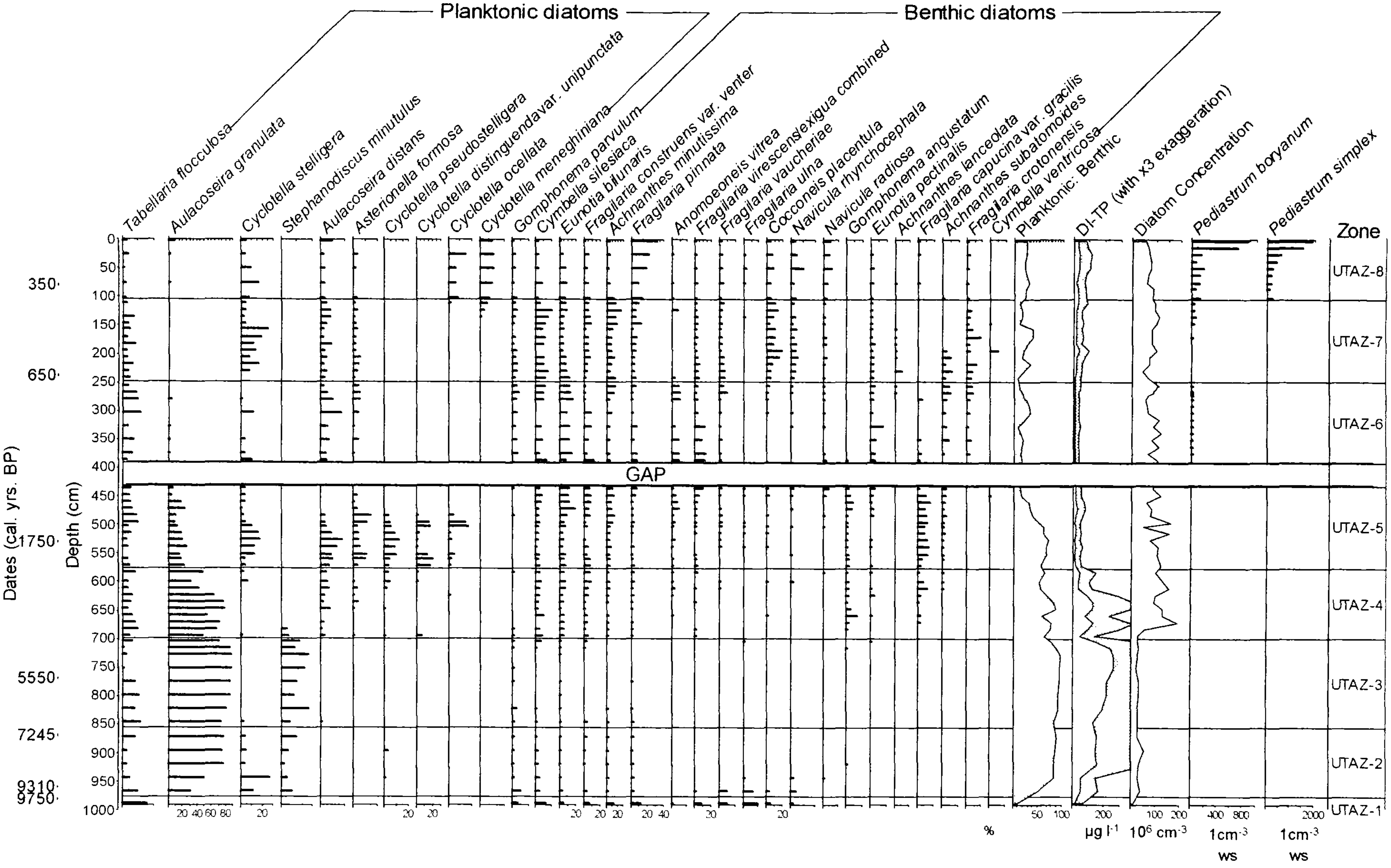


Figure 5.3 Summary diatom diagram for core UT2/94 showing diatom taxa present at >4%, the planktonic:benthic ratio, diatom-inferred total phosphorus (DI-TP), diatom concentration and *Pediastrum* spp. Biostratigraphic zones were defined using CONISS.

UTAZ-3: 8.50 – 7.00 m (*ca.* 6.9 - 4.4 k cal. yrs. BP)

This zone was dominated throughout by *A. granulata* with the significant co-occurrence of *S. minutulus*, which increased to 15% at 7.15m. Benthic taxa were very rare, with *G. parvulum* being the only benthic taxon with any significant presence, resulting in a zone of extremely low diversity.

UTAZ-4: 7.00 – 5.70m (*ca.* 4.4 - 3.3 cal. yrs. BP)

A. granulata continued to dominate in this zone, although a notable reduction in abundance at the zone base was followed by an increase to 75% at 6.50m and a reduction to 30% at 6.00m. The decrease in *A. granulata* and loss of *S. minutulus* at 6.80m was matched by a corresponding increase in the relative abundance of centric, planktonic taxa characterised by *C. stelligera*, *C. distinguenda* var. *unipunctata*, *C. pseudostelligera*, *C. ocellata*, *Asterionella formosa* and *A. distans*. In the benthic group *C. silesiaca* made a notable reappearance at the zone base rising to 10% at 6.10m, whilst *Gomphonema angustatum*, *Fragilaria capucina* var. *gracilis* and *Eunotia pectinalis* appeared for the first time.

UTAZ-5: 5.70 – 4.20m (*ca.* 3.3 - 1.2 k cal. yrs. BP)

From the zone base the planktonic taxa recorded in Zone 4 increased in abundance at the expense of *A. granulata*. At a depth of 4.80m, however, *A. distans, C. distinguenda* var.

unipunctata. *C. ocellata* and *C. pseudostelligera* disappeared, and *C. stelligera*, *A. formosa* decreased markedly in relative abundance.

UTAZ-6: 3.95 – 2.50m (*ca.* 1.1 - 0.7 k cal. yrs. BP)

As *A. granulata* abundance decreased, *T. flocculosa, A. distans, A. formosa* and *C. stelligera* increased in abundance. This was accompanied by a increase in the relative abundance of benthic taxa (*F. pinnata, F. vaucheriae, E. bilunaris, C. silesiaca, A. minutissima, A. vitrea, A. subatomoides* and *A. lanceolata*) and the introduction of new species (*Fragilaria crotonensis* and *Navicula stroemii*). Zone 6 was thus a high diversity zone with a general increase in the relative abundance of both planktonic and benthic taxa, although planktonic species abundance decreased significantly at the top of the zone.

UTAZ-7: 2.50 – 1.00m (*ca.* 0.7 – 0.4 k cal. yrs. BP)

Zone 7 was marked by a clear initial increase in the relative abundance of *C. stelligera*, which dominated to a depth of 1.50m then decreased sharply. This was matched by an increase in *A.distans* and *C. meneghiniana*, which appeared for the first time at 1.30m and increased markedly thereafter. *C. ocellata* reappeared at the zone base at low abundance and increased at the upper zone boundary. The latter increase in planktonic taxa was accompanied by a trend towards increased relative abundance of many benthic taxa (eg. *A. minutissima, C. silesiaca, F. construens* var. *venter, F. crotonensis, F.*

pinnata, C. placentula, G. parvulum and *E. pectinalis*) at the expense of *A. vitrea, F. capucina* var. *gracilis* and *A. subatomoides*.

UTAZ-8: 1.00 – 0m (*ca.* 0.4 – 0.0 k cal. yrs. BP)

The relative abundance of benthic *F. pinnata* increased from 10% at the zone base to 30% at the top of the core, dominating Zone 8 and accompanied by notable increases in the planktonic, centric species *C. stelligera, C. ocellata* and *C. meneghiniana. A. granulata* reappeared at the zone base and increased in abundance towards the top of the core, whilst benthic species either decreased at the zone base (e.g., *G. parvulum, A. minutissima, F. crotonensis*) or increased towards the midzone then decreased at the top of the core (eg. *E. bilunaris, C. placentula, N. radiosa, N. rhyncocephala*).

5.10 DI-TP Reconstruction

The results of the DI-TP reconstruction are given in Figure 5.3. Good representativity of fossil taxa in the modern dataset (99%) indicates that the reconstruction is likely to be reliable. The sample specific errors of prediction and confidence intervals estimated by bootstrapped coefficient of determination are given in Figure 5.4 with an $\mathrm{RMSE}_{\mathrm{boot}}$ value of 0.27.

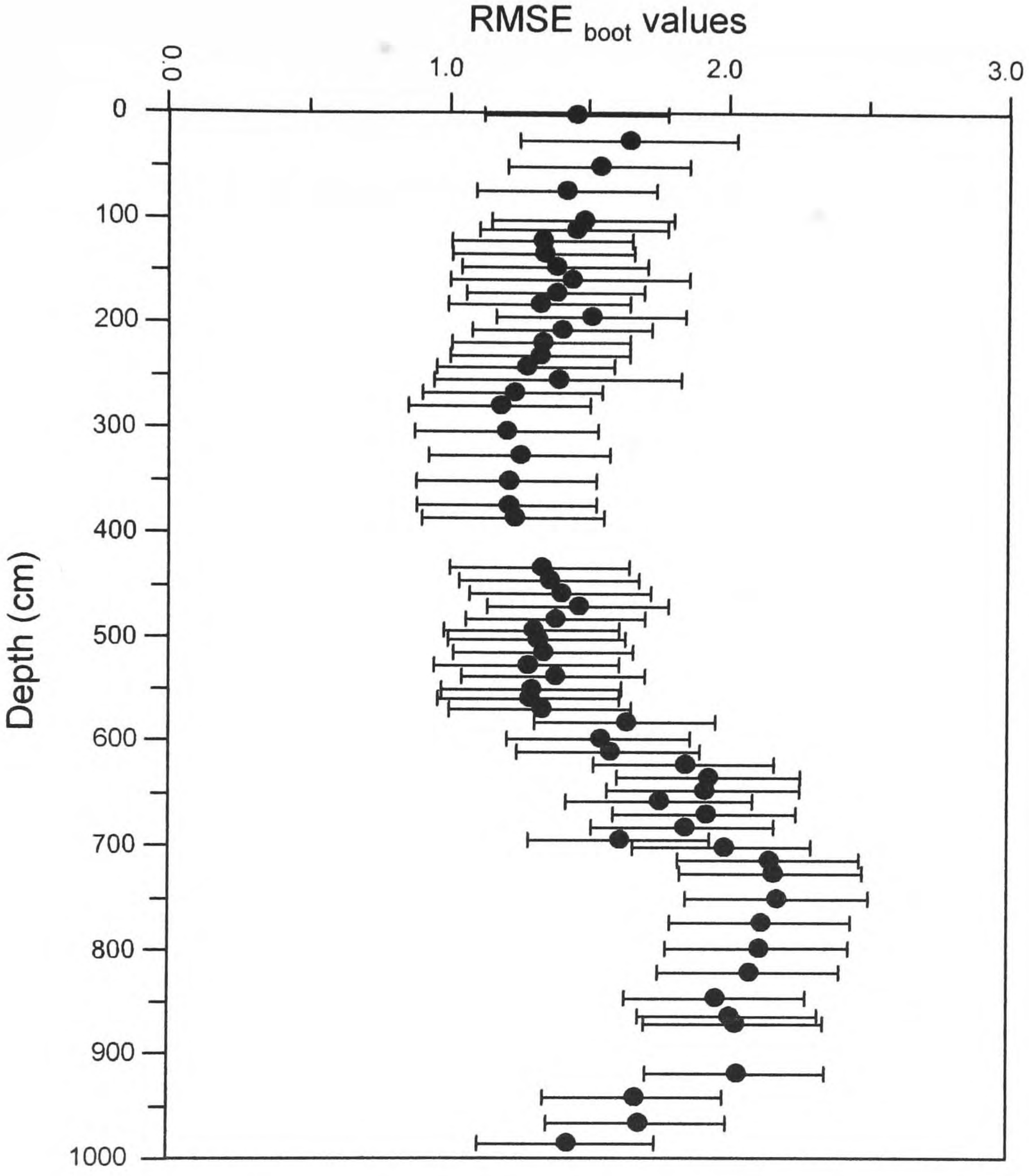


Figure 5.4 Scatter plot of the sample specific errors of prediction for Upper Talley (UT2/94) estimated by bootstrapping and presented with error bars

The overlap in error bars indicates that fluctuations in DI-TP are not statistically significant. The dominant taxa are both abundant and well represented in the training set, but they are all common, morphologically variable taxa which are merged in the EDDI training set and have broad tolerance ranges (e.g. *A. granulata* $N_2 = 32.3$; Tol = 0.43 $\log_{10}$ μg l^{-1}; *C.stelligera* $N_2 = 24.7$; Tol = 0.35 $\log_{10}$ μg l^{-1}; *F. pinnata* $N_2 = 53.9$; Tol = 0.50 $\log_{10}$ μg l^{-1}. Although they include classic eutrophic species, statistically, they are not strong indicator taxa. The results are therefore presented, but interpreted with caution. In UTAZ-1 to UTAZ-4 the trend in DI-TP values was controlled largely by fluctuations in the relative abundance of the eutrophic taxa *A. granulata* and *S. minutulus*.

In UTAZ-1 (a single, outlier sample), the DI-TP estimate was 25.0 μg l^{-1}. This increased in UTAZ-2 to 50.0 μg l^{-1} at a depth of 9.75m and again to 150.0 μg l^{-1} at 9.25m, remaining high and stable for the rest of the zone. In UTAZ-3 a further increase to the maximum of the entire sequence (256.0 μg l^{-1}) at 7.50m was followed by an abrupt decrease to 45.8 μg l^{-1} at the top of the zone. In UTAZ-4 two marked peaks in DI-TP to 127.3 μg l^{-1} and 126.6 μg l^{-1} at 6.70m and 6.34m respectively, were followed by a decrease to 15.0μg l^{-1}. DI-TP was relatively stable for the whole of UTAZ-5, UTAZ-6 and into UTAZ-7, producing a mean value of 16.0 μg l^{-1}, then increased gradually to a peak of 33.1 μg l^{-1} at 1.94m. A decrease to 19.0 μg l^{-1} was followed by an increase to 28.0 μg l^{-1}. Finally, in UTAZ-8, DI-TP increased to 33.8 μg l^{-1}, and decreased to 26.4 μg l^{-1} at the top of the core.

5.11 Diatom Concentrations

At the core base (Figure 5.3) the diatom concentration was the minimum for the sequence ($1.1x10^7$ cm^{-3}), increasing thereafter to stable, higher values with a mean of $3.0x10^7cm^{-3}$, from the base of UTAZ-2 to the top of UTAZ-3. In UTAZ-4 values increased sharply and to $1.87x10^8cm^{-3}$ (6.80m), the maximum for the sequence. A general decreasing, although fluctuating, trend followed in UTAZ-5 and -6 to a value of $7.0x10^7cm^{-3}$ at 3.00m depth. At the top of UTAZ-6 concentration increased sharply to $9.2x10^7cm^{-3}$. In UTAZ-7 an increase from $5.1x10^7cm^{-3}$ (2.30m) to $1.12x10^8cm^{-3}$ (1.25m) was followed by a general decreasing trend to $5.3x10^7cm^{-3}$ at the top of the core.

5.12 Data Analysis

5.12.1 Detrended Correspondance Analysis (DCA)

DCA axis 1 ($\lambda_1 = 0.582$) and DCA axis 2 ($\lambda_2 = 0.131$) explained 22% and 18% of the variance in the diatom percentage data, respectively, indicating that a relatively high proportion of the total percentage variance was summarised in the first two axis. A scatter plot of Axis 1 against Axis 2 (Figure 5.5) of the species scores indicates that the first axis may represent a nutrient gradient and thus that P could have had a major influence on species composition and distribution. *A. granulata* (XXG988) and *S. minutulus* (XXG974) plot closely together and have high scores on Axis 1. These taxa are characteristic of high epilimnetic P and turbid conditions.

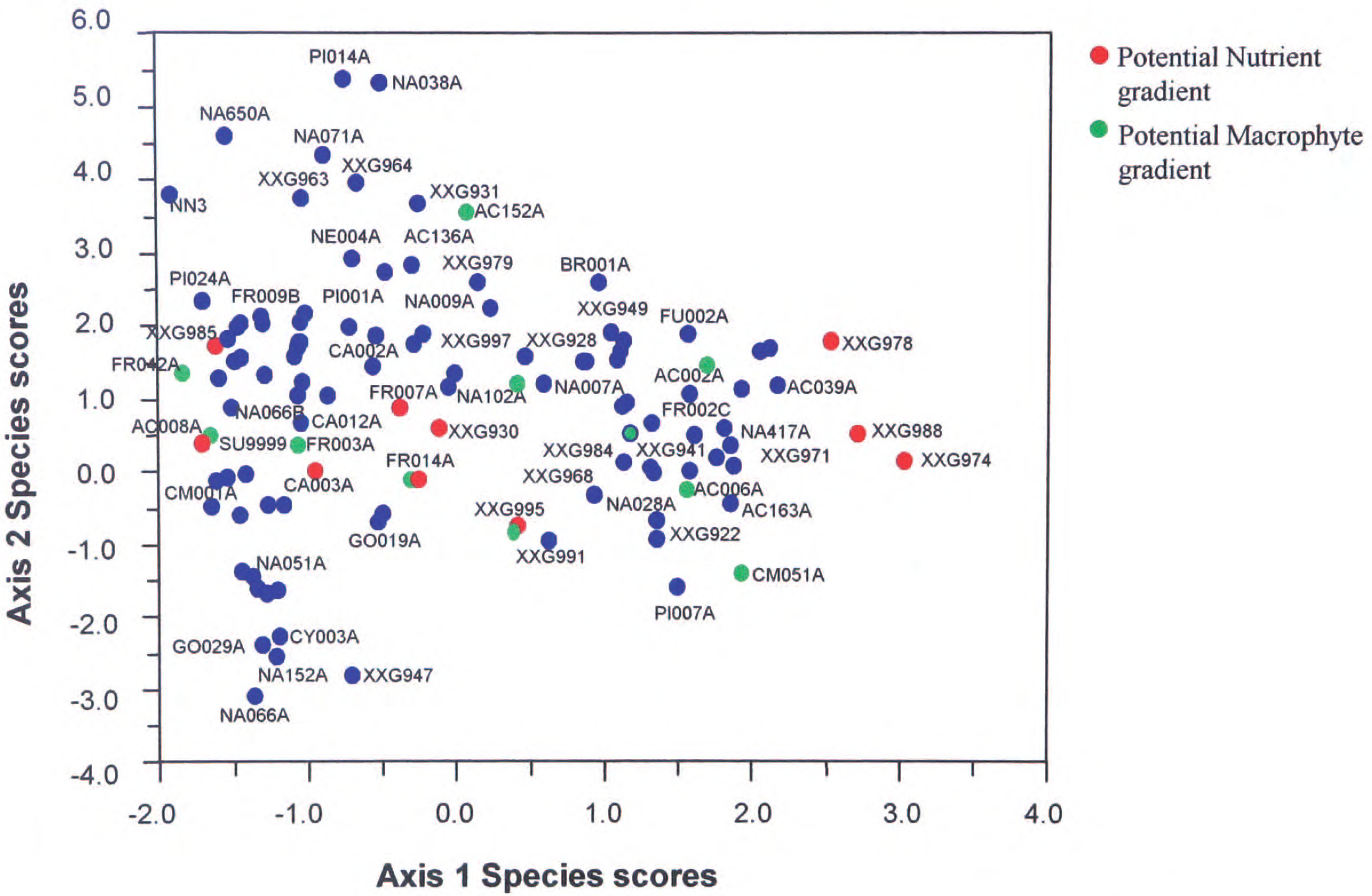


Figure 5.5 Scatter plot of the DCA species scores for Axis 1 against Axis 2 for the Upper Talley core (UT2/94) based on114 taxa.

Nutrient tolerant *G. angustatum* (XXG978) can also be found in the same ordination space to the right of Axis 1. Meso-eutrophic diatoms such as *F. vaucheriae* (FR007A), *F. pinnata* (XXG995), *F. leptostauron* (FR014A) and *N. pupula* (XXG930) with TP optima of 53.0 μg l^{-1}, 34.0 μg l^{-1}, 57.5 μg l^{-1} and 47.0 μg l^{-1}, respectively, plot in the centre of the range. The underlying pattern was less clear for oligo- to mesotrophic taxa. Taxa with low scores on Axis 1 included mesotrophic taxa such as *A. exigua* (AC008A), *Caloneis silicula* (CA003A) and *F.crotonensis* (XXG985), which have TP optima of 48.0 μg l^{-1}, 32.0 μg l^{-1} and 30.0 μg l^{-1} respectively. These plotted in the same ordination space as *F. nitzchoides* (FR042A), *C. ventricosa* (CM001A), which have high TP optima of 115 μg l^{-1} and 91.2 μg l^{-1}.

Although TP is probably the major floristic influence, the lack of a clear gradient may be due to the influence of other limnological variables such as pH (e.g. *A. carissima*; AC152A, *A. lapponica*; AC038A and *A. rupestris*; AC118A). Changes in pH could also be associated with the process of eutrophication or, of course, acidification, producing a more complex nutrient gradient than expected. The spread of species scores on DCA1, for example, may also suggest a gradient associated with the presence of aquatic macrophytes, which can change in abundance with trophic status particularly in shallow, eutrophic lakes where an increase in turbidity can cause a shift from benthic taxa characteristic of the clear water, macrophyte dominated state to planktonic taxa associated with organic pollution and high light attenuation. Taxa supporting this interpretation include *Achnanthes clevei* (AC006A), *A. linearis* (AC002A) and *Cymbella elginensis* (CM051A) at the high end of Axis 1, *A. carissima* (AC152A) and *Fragilaria* *leptostauron* (FR014A) in the middle range and *A. exigua* (AC008A) and *F. nitzschoides* (FR024A) at the low end. The gradient may also be associated with changes in lake level, which can cause a shift in the planktonic:benthic ratio as the depth of the photic zone changes with lake depth.

A plot of Axis 1 scores against depth (Figure 5.6) followed the same pattern as the diatom based phosphorus reconstruction (DI-TP) and the abundance of taxa associated with enrichment, particularly *A. granulata* (Figure 5.2). This shows that the diatom assemblages are tracking changes that are related to P concentrations and, although other environmental variables or interactions between variables may be influencing changes in the diatom flora, nutrient availability has a highly significant effect on species distribution.

5.12.2 Hills Diversity Index (N_2)

Figure 5.7 shows that species diversity displayed an approximately inverse relationship with the DI-TP reconstruction and the abundance of *A. granulata*. The results support the idea that an increase in P concentrations and productivity are often accompanied by a decrease in species richness and abundance (Scheffer, 2004).

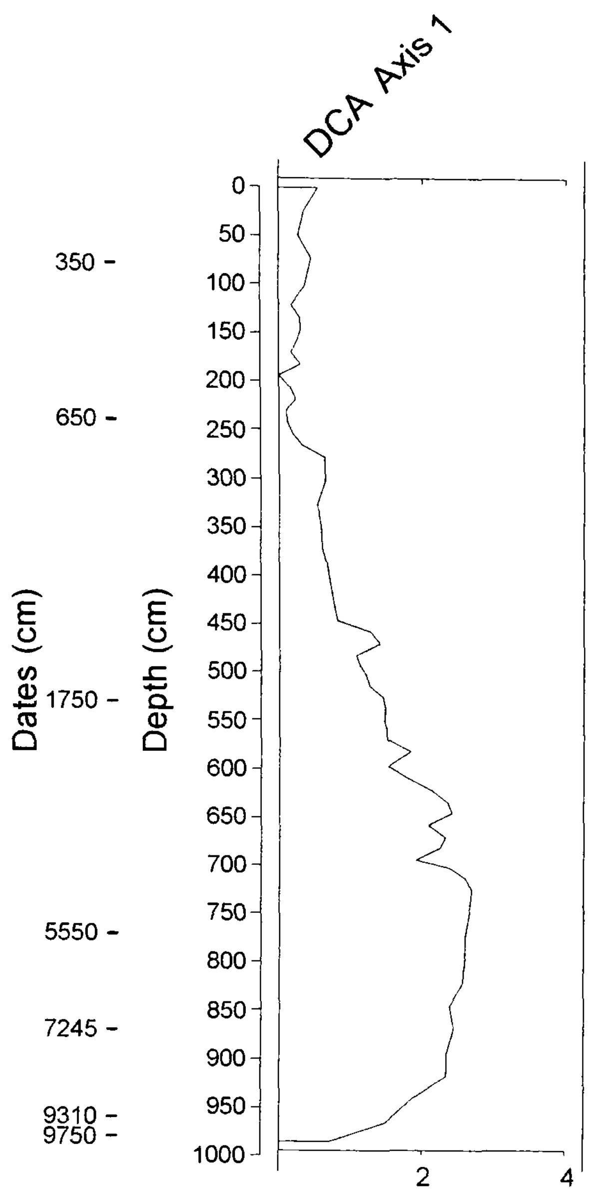


Figure 5.6 Diagram showing the temporal variation in Axis 1 of the sample scores data for fossil diatom assemblages in core UT2/94 based on 62 samples.

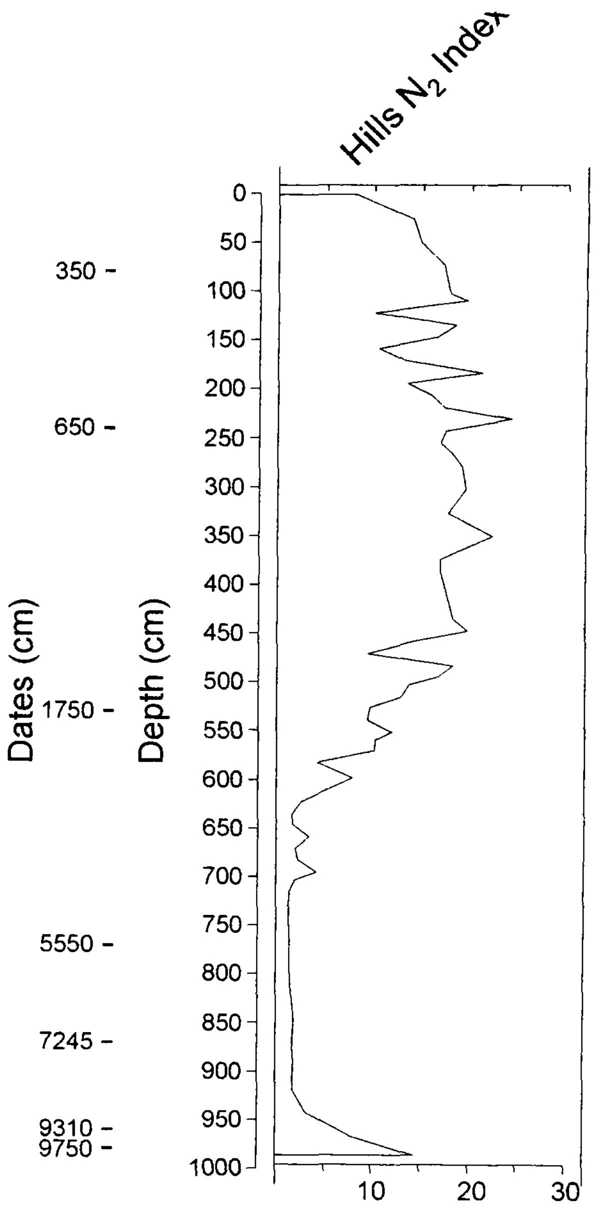


Figure 5.7 Diagram showing the Hills N_2 Diversity Index values for fossil diatoms in core UT2/94 based on 62 samples.

5.13 Cladocera Ephippia

In UTEPHZ-1 (Figure 5.8) ephippia abundance increased from 0.05 g^{-1} ds at the core base to a peak of 0.9 g^{-1} ds at a depth of 8.13m. A decrease to 0.05 g^{-1} ds at a depth of

6.93m followed. In UTEPHZ-2 ephippia presence immediately increased to 0. 9 g^{-1} ds at a depth of 6.53m followed by a final decrease to 0.05 g^{-1} ds at 4.60m. At the base of UTEPHZ-3 ephippia were mainly absent, then abundance increased sharply and significantly from 0.1 g^{-1} ds at 3.00m to 1.45 g^{-1} ds at 2.53m. This was followed by a return to low abundance at the upper zone boundary. Finally, in UTEPHZ-4 ephippia presence was low (*ca.* 0.05 g^{-1} ds) to a depth of 29cm then increased rapidly to the maximum of the entire sequence (1.5 g^{-1} ds) at the top of the core.

5.14 *Pediastrum*

Only two species were observed, *Pediastrum boryanum* and *P. simplex*. As can be seen in Figure 5.3, three main phases of *P. boryanum* deposition occurred. The first was in UTAZ-5 between 5.80 and 5.20m, the second in UTAZ-6 and 7 between 3.75 and 2.50m and the third in UTAZ-7 and 8. The latter occurrence began at a depth of 1.80m and continued to the top of the core where 72196 cm^{-3} was the maximum abundance recorded in the entire sequence. *P. simplex* appeared at a depth of 1.00m in UTAZ-8 and increased from 235 cm^{-3} to 1504 cm^{-3} at the top of the core.

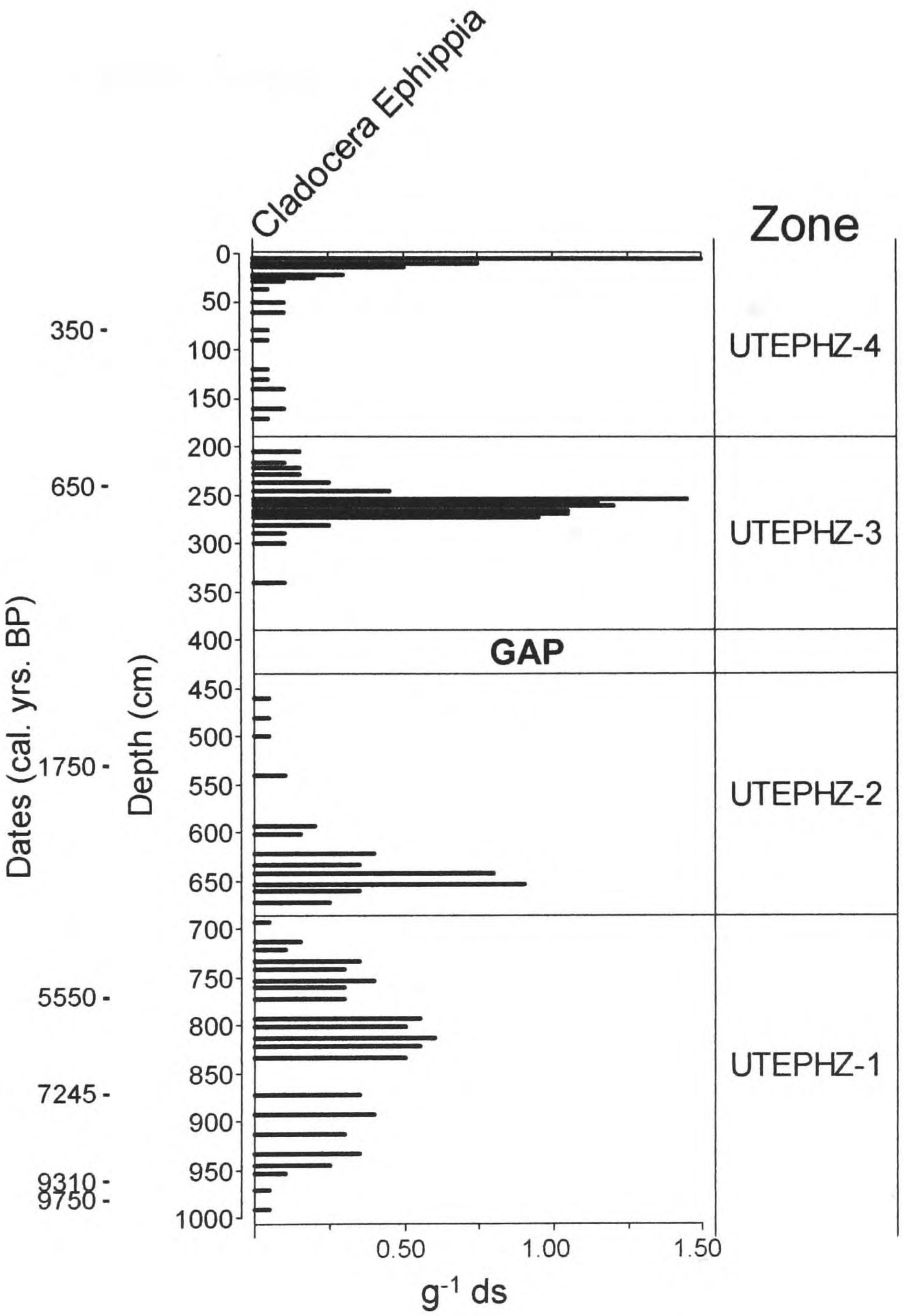


Figure 5.8 Diagram showing the abundance of cladocera ephippia in core UT2/94

5.15 Pollen Analysis

Figure 5.9 shows a summary pollen diagram of selected trees, shrubs, grasses, herbs and aquatic pollen, which highlight changes in vegetation patterns that may be the result of human activity in the Upper Talley catchment. A more detailed description of the pollen data from the entire core is presented in Johnson (1998).

UTPZ-1: 7.75 - 6.40m *(ca.* 5.6 – 3.5 k cal. yrs. BP)

Zone 1 was dominated by arboreal pollen types, which comprised 80% of the sum. Following a decrease at 7.25m, Birch (*Betula*) pollen percentages increased towards the upper zone boundary accompanied by increases in the abundance of Oak (*Quercus*) and Hazel (*Corylus avellana*) pollen, whilst Alder (*Alnus glutinosa*) decreased in relative abundance from 7.25m. Plantain (*Plantago*) and Sorrel (*Rumex acetosa*) and common grass (Poaceae) pollen were recorded for the first time. A general increase in aquatic macrophyte pollen was also observed characterised by the significant presence of pondweed (*Potamogeton*) and Sedge (Cyperaceae) pollen.

UTPZ-2: 6.40 – 4.50m (*ca.* 3.5 - 1.2 k cal. yrs. BP)

A general decrease in arboreal pollen percentages was observed, characterised by reductions in the relative abundance of *Quercus*, *Betula* and *C. avellana* pollen. The relative abundance of *A. glutinosa*, however, displayed a general increase from 55 % at

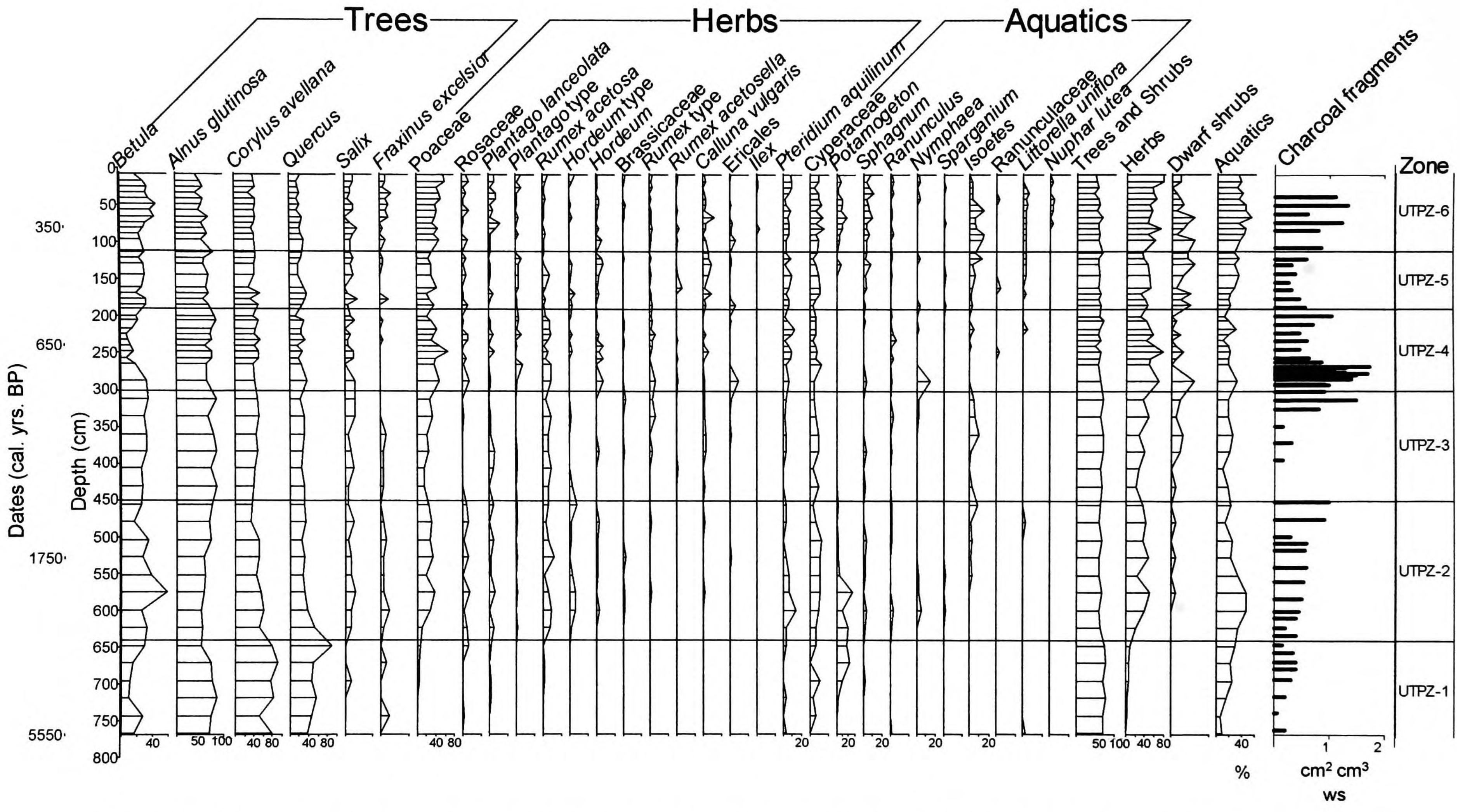


Figure 5.9 Diagram showing the abundance of selected pollen taxa and groups for core UT2/94 (adapted from Johnson, 1998) presented with charcoal abundance.

the zone base to 75% at the upper zone boundary. A general increase in herb pollen, from 30% at the zone base to 55% at 5.75m, was characterised by a significant increase in Poaceae and *Rumex acetosa*. Other taxa recorded were Barley (*Hordeum*), *Plantago*, Rose (Rosaceae) and Cabbage/Mustard (Brassicaceae). The increase in aquatic pollen percentages noted in the previous zone continued to a depth of 5.75m before decreasing to the top of the zone. This was mainly controlled by an increase in pondweed (*Potamogeton*), minor increases in Bur-reed (*Sparganium*), Buttercup (*Ranunculus*) and Shore weed) *Littorella uniflora*.

UTPZ-3: 4.50 – 3.00m (*ca*. 1.2 – 0.8 k cal. yrs. BP)

In Zone 3, although arboreal pollen abundance increased at the zone base, the relative abundance of individual taxa was relatively stable with the exception of Ash (*Fraxinus excelsior*), which disappeared from the record at 3.50m. Poaceae pollen percentages increased from 15% at 4.30m to 30% at the top of the zone, whilst Heather (*Calluna vulgaris*) percentages also increased from a depth of 3.80m. Little change was observed in aquatic macrophyte pollen abundance, apart from a minor increase in Quillwort (*Isoetes*) at 3.10m

UTPZ-4: 3.00 – 2.40m (*ca*. 0.8 – 0.6 k cal. yrs. BP)

Marked changes were observed in this zone, which was characterised by the reappearance of *F. excelsior* at 2.25m and a reduction in the relative abundance of *A. glutinosa* and *C.*

avellana followed by a considerable reduction in *Betula* from 42% at the zone base to 11% at 2.40m. This decrease in *Betula* abundance was followed by an increase to 20% at the top of the zone. At a depth of 2.50m Poaceae pollen increased to the maximum percentage of the entire core (67%), followed by an immediate decrease to 21% at the upper zone boundary. The increase in Poaceae pollen was matched by a significant increase in the abundance of Bracken (*Pteridium aquilinum*) to 12% at a depth of 2.20m. A concurrent increase in the proportion of herbs and cereals was reflected in a general increase to 70% at a depth of 2.40 although this was followed by a decrease towards the upper zone boundary.

The percentage of aquatic macrophyte pollen displayed a shifting trend in this zone, characterised first by a decrease towards the mid-zone (from 32% at the zone base to 9%), followed by an increase to 31% at 2.10m and finally a decrease to 7% at the top of the zone. Changes in aquatic pollen were mainly the result of shifts in the percentage of Cyperaceae pollen.

UTPZ-5: 2.40m – 1.10m (*ca.* 0.6 – 0.44 k cal. yrs. BP)

In this zone the relative abundance of *Betula* pollen increased to 36% at 1.40m, whilst *Quercus* pollen percentages displayed a general decrease towards the top of the zone. These changes were accompanied by a marked increase in the abundance of pollen from dwarf shrubs. Aquatic pollen increased in relative abundance to 38% at the top of the zone and was mainly characterised by *Isoetes*, White water lily (*Nymphaea alba*) and *Littorella uniflora* and the reintroduction of *Potamogeton* at 1.25m.

UTPZ-6: 1.10 – 0m (*ca.* 0.4 – 0.0 k cal. yrs. BP)

In Zone 6 major changes in arboreal pollen proportions were matched by a corresponding increase in the abundance of grasses and herbs. The relative abundance of *Betula* pollen initially increased from 25% at the zone base to 40% at 50cm, followed by a decrease to 20% at the top of the core. *Quercus* pollen also decreased in abundance from 35% at the zone base to 12% at the top of the core, accompanied by a decrease in *A. glutinosa* and *C. avellana* pollen proportions. Corresponding with these changes Poaceae pollen abundance increased from 35% at the zone base to 60% at the top of the core, whilst overall herb pollen percentages also increased towards the upper zone boundary, characterised by increases in *P. lanceolata* and Brassicaceae. At 0.60cm aquatic pollen increased to its maximum percentage of the entire core (57%), characterised by increases in Cyperaceae, *Potamogeton*, *Isoetes* and Moss (*Sphagnum*) and, to a lesser extent, *Ranunculus*, Yellow pond lily (*Nuphar*) and *Nymphaea alba*. This latter peak in aquatic pollen percentages was followed by an immediate decrease to 37% at the top of the core.

5.15.1 Alternate Stable States

Comparison of diatom concentration (DC) and aquatic pollen sum (APS) is given in Figure 5.10. Relatively stable DC values (mean = $25\text{x}10^6$ cm^{-3}) between 7.75m and 7.00m were followed by a sharp increase to 190 $\text{x}10^6$ $\text{cm}^{-}3$ at a depth of 6.70m. DC then displayed a general decreasing trend for most of the core, with marked peaks at 6.00m and 2.50m.

APS displayed an initial increase from 8% at the core base to 45% at 5.75m followed by a decreasing trend to 5% at 4.00m. Percentages then increased again to 30% at 2.90m followed by a sharp decrease to 8% at 2.30m. The APS then displayed a sustained increase to 54% at 60cm, which was followed by an immediate decrease to 32% at the top of the core. Apart from at a depth of 2.50m, there were no other obvious shifts between the two plant groups. With each phase of human impact, however, APS reached a limit reflecting a threshold in the growth of aquatic macrophytes. This was immediately followed by a decrease suggesting deterioration of the plant community.

5.15.2 Diatom and Pollen Concentrations

The diatom concentration divided by pollen concentration (DC/PC) and the original diatom concentration (DC) and for each sample are plotted in Figure 5.11. This analysis removes the affect of sediment accumulation from the DC data so that changes observed in the DC curve are controlled by factors other than sedimentation. From the core base to

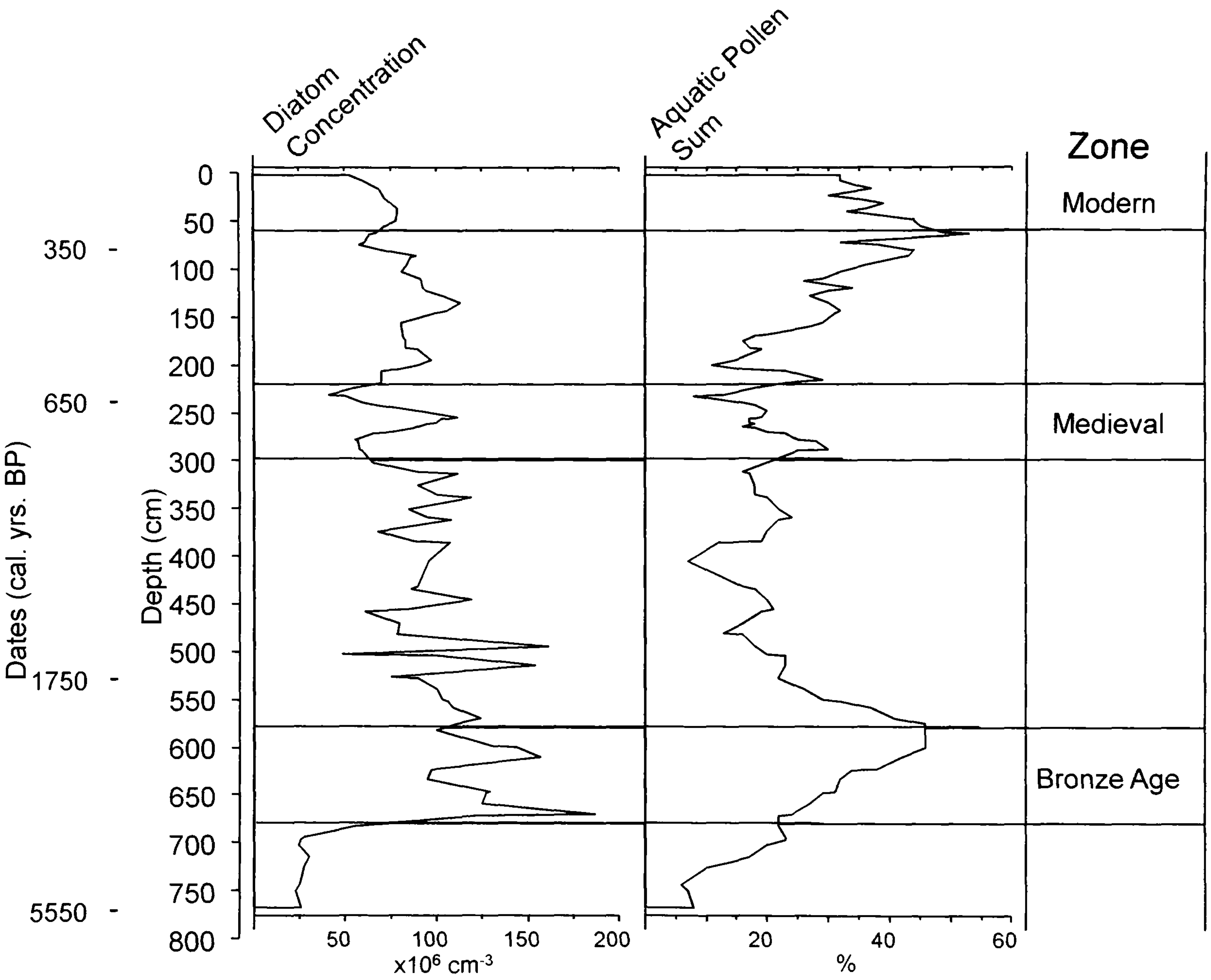
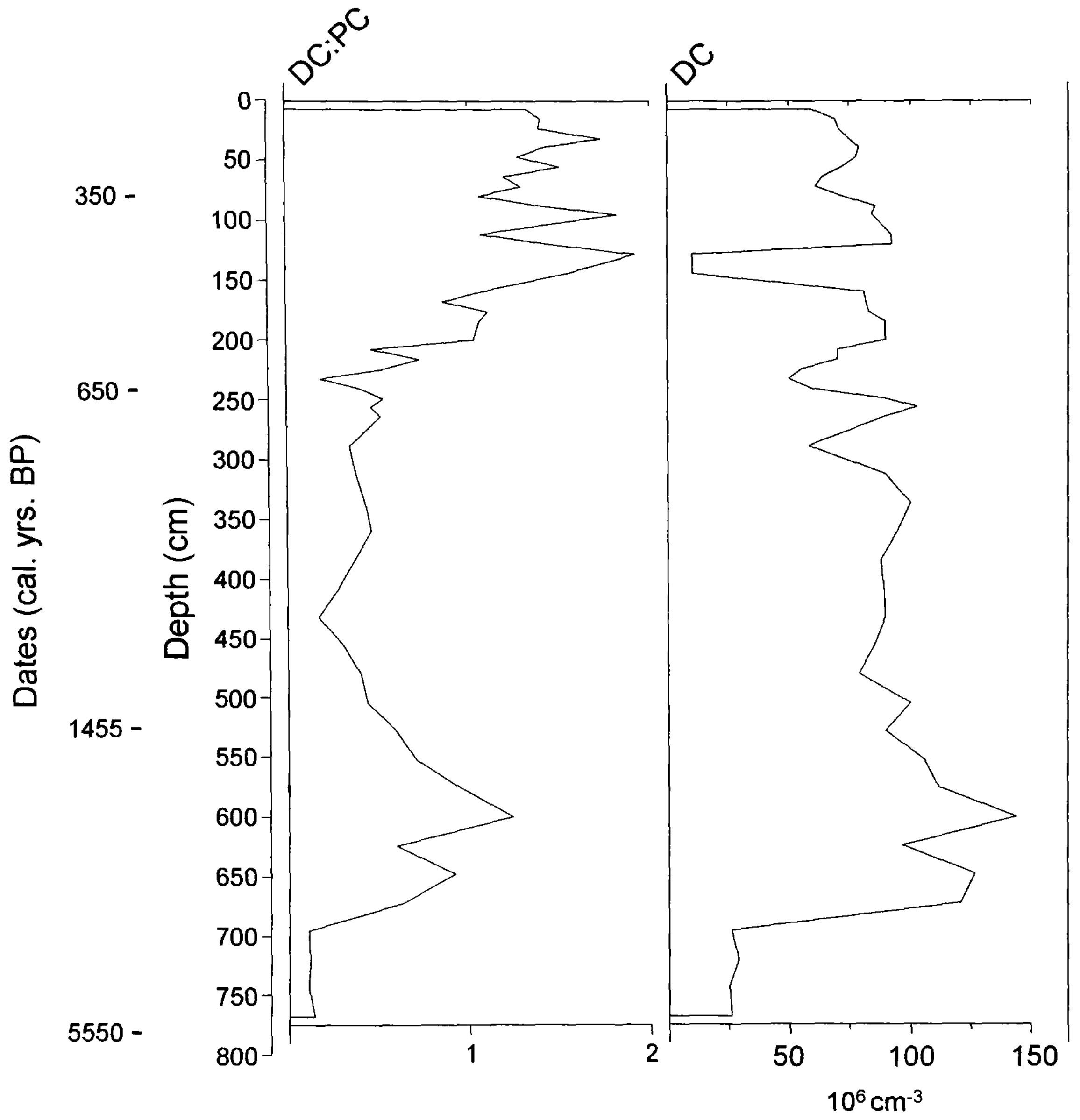


Figure 5.10 Diagram comparing diatom concentration (DC) with the aquatic pollen sum (APS) for the Upper Talley core (UT2/94)



5.11 Diagram comparing the diatom to pollen concentration ratio (DC:PC) with the diatom concentration (DC) for the Upper Talley core (UT2/94).

a depth of *ca.* 5.30m, removal of the sediment accumulation factor (DC/PC) shows that changes in DC can be attributed purely to changes in diatom abundance. The major influences on this process are increased diatom production during phases of enrichment and changes in silica (Si) concentrations (Anderson, 1989). From a depth of 5.30m the DC and DC/PC graphs diverge indicating that changes in the DC trend may have been influenced by a combination of sediment accumulation and changes in in-lake diatom abundance.

5.15.3 Land-clearance and the Arable/Pastoral Index

The results for Non-arboreal Pollen (NAP) and the Arable/ Pastoral Index (API) are given in Figure 5.12. Between 7.68m and 6.70m NAP is *ca.* 1% reflecting a predominantly forested catchment. This was accompanied by a sudden peak in API to 50% at 7.20m. At the beginning of the Bronze Age (6.70m) an increase in NAP from 1% to 23% at a depth of 6.00m indicates an increase in open ground, followed by a fluctuating trend (between 10% and 20%) to a depth of 4.30m. This was accompanied by a significant decrease in API to 3% at 4.80m indicating that deforestation was accompanied by a sustained increase in arable land-use. From 4.30m NAP decreased to 36% at 2.48m, whilst the API for this period (medieval) reflects predominantly arable farming with values fluctuating between 3% and 10%. From a depth of 2.00m to the top of the core NAP indicates a sustained increase from 13% to 48% reflecting an increase in deforestation and open ground, whilst the API for this period reflected values between 3% and 10% and a peak to 17% at 72 cm.

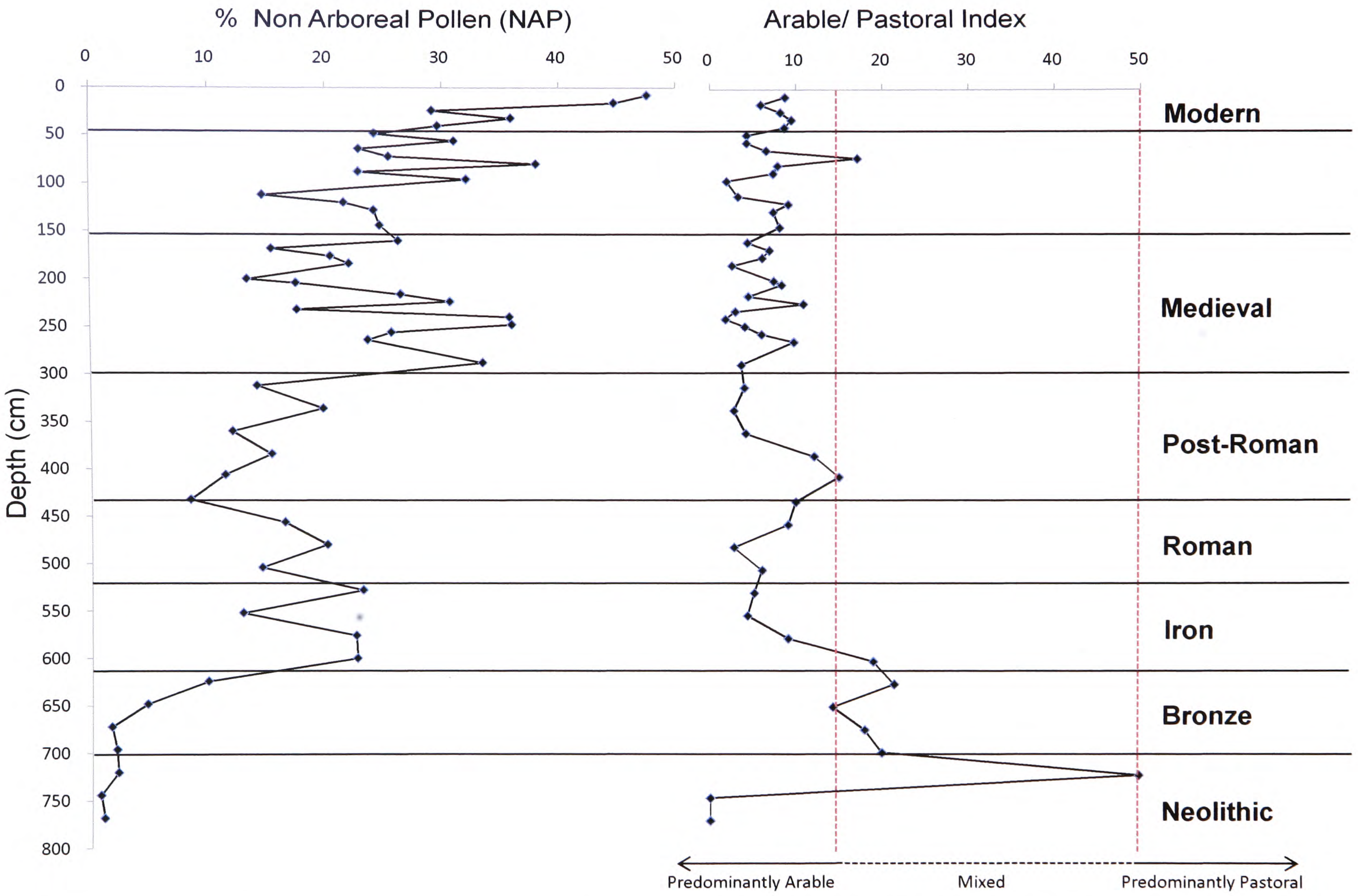


Figure 5.12 Diagram showing a comparison of the percentage changes in non-arboreal pollen a(NAP) and the Arable/ Pastoral Index (API) for Upper Talley (UT2/94)

5.16 Charcoal Analysis

The results for charcoal analysis are given in Figure 5.9 and are presented with selected pollen taxa to enable comparison with shifts in vegetation patterns. Charcoal particles were present throughout UTPZ-1 and UTPZ-2, increasing steadily from $2cm^2\ cm^{-3}$ at the core base to $8cm^2\ cm^{-3}$ at a depth of 4.46m. In UTPZ-3 charcoal abundance decreased significantly then increased to $12cm^2\ cm^{-3}$ at the top of the zone. In UTCZ-4 a maximum of $15cm^2\ cm^{-3}$ at a depth of 2.54m was followed by a significantly decrease in UTCZ-5, apart from an increase to $9cm^2\ cm^{-3}$ at 1.82m. Finally, in UTCZ-6, charcoal particles steadily increased from $6cm^2\ cm^{-3}$ at the zone base to $13cm^2\ cm^{-3}$ at the top of the core.

5.17 Sediment Accumulation Rates

Sediment Accumulation Rates (SAR) for the sequence are given in Figure 5.13 and summarised in Table 5.4. Although rather imprecise due to lack of chronological control, the observed changes indicate clear increases in SAR during phases of human activity.

Depth (m)	**SAR (yr cm^{-1})**
9.97 – 8.85	9.4690
8.77 – 7.09	7.6608
6.93 – 5.49	13.4969
5.33 – 3.25	5.9916
2.77 – 0.85	6.7991
0.69 – 0.05	9.0066

Table 5.4 Table showing a summary of SAR for the Upper Talley core (UT2/94)

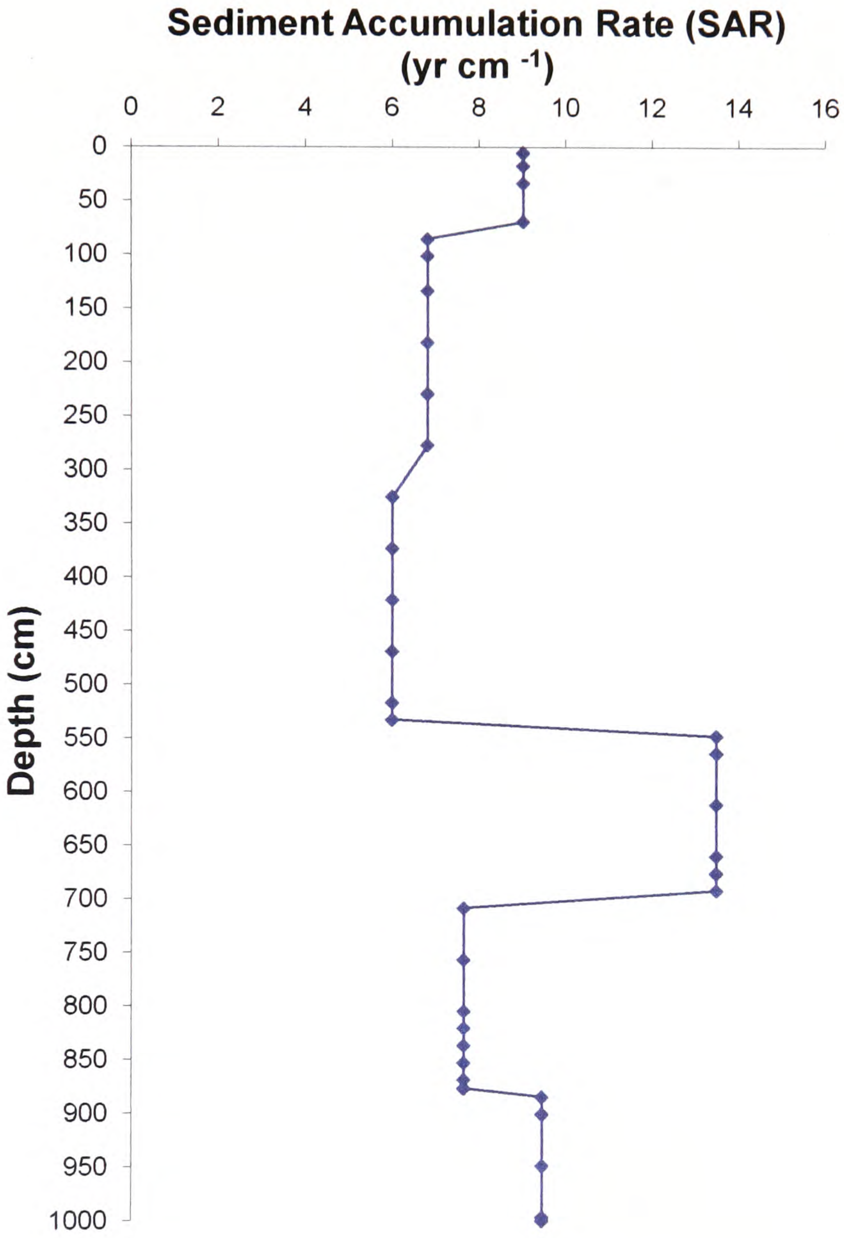


Figure 5.13 Diagram showing approximate sediment accumulation rates (SAR) for the Upper Talley core (UT2/94) based on linear interpolation between dates.

5.18 Radiocarbon Dates and Chronology

The results of AMS ^{14}C dates (calibrated using the University of Oxford Radiocarbon Accelerator Unit calibration program, OxCal3; Bronk Ramsey, 1995) are given in Table 5.5. Figure 5.14 gives a comparison of these dates with those from Johnson's (1998) radiocarbon, cross-correlated horizons from the Lower Talley (Table 5.6). Both sets of dates displayed extremely good correlation between 9.30m and 5.25m providing a secure chronology. However, above 5.25m the dates from this study were older than the dates from Johnson (1998) placing the most prominent synchronous shift of the entire core at 1365±40 cal. yrs. BP compared to 650 cal. yrs. BP in Johnsons (1998) study. This may suggest a dating error possibly from the reworking of sediments or the inwash of older/newer carbon from the lake catchment (O'Sullivan *et. al.*, 1973). Additionally, the date at 2.57m was obtained by AMS analysis of charcoal, which by definition can produce older dates as it is derived from wood from older arboreal species.

In light of these dates, there may be a possible chronological error in terms of absolute dating in the upper part of the Upper Talley sequence, above *ca.* 2.5m, of up to *ca.* >700 radiocarbon years. As such, this part of the sequence should be treated with caution until further, higher resolution dating evidence is available. However, when these dates are compared to catchment history at Upper Talley, the date of *ca.* 0.65 k cal. yrs. BP corresponds with a well-documented phase of medieval occupation, which included the construction of an Abbey and additional ecclesiastical buildings directly on the lake shore. Additionally, a phase of limnological impact at *ca.* 1.3 k cal. yrs. BP suggests

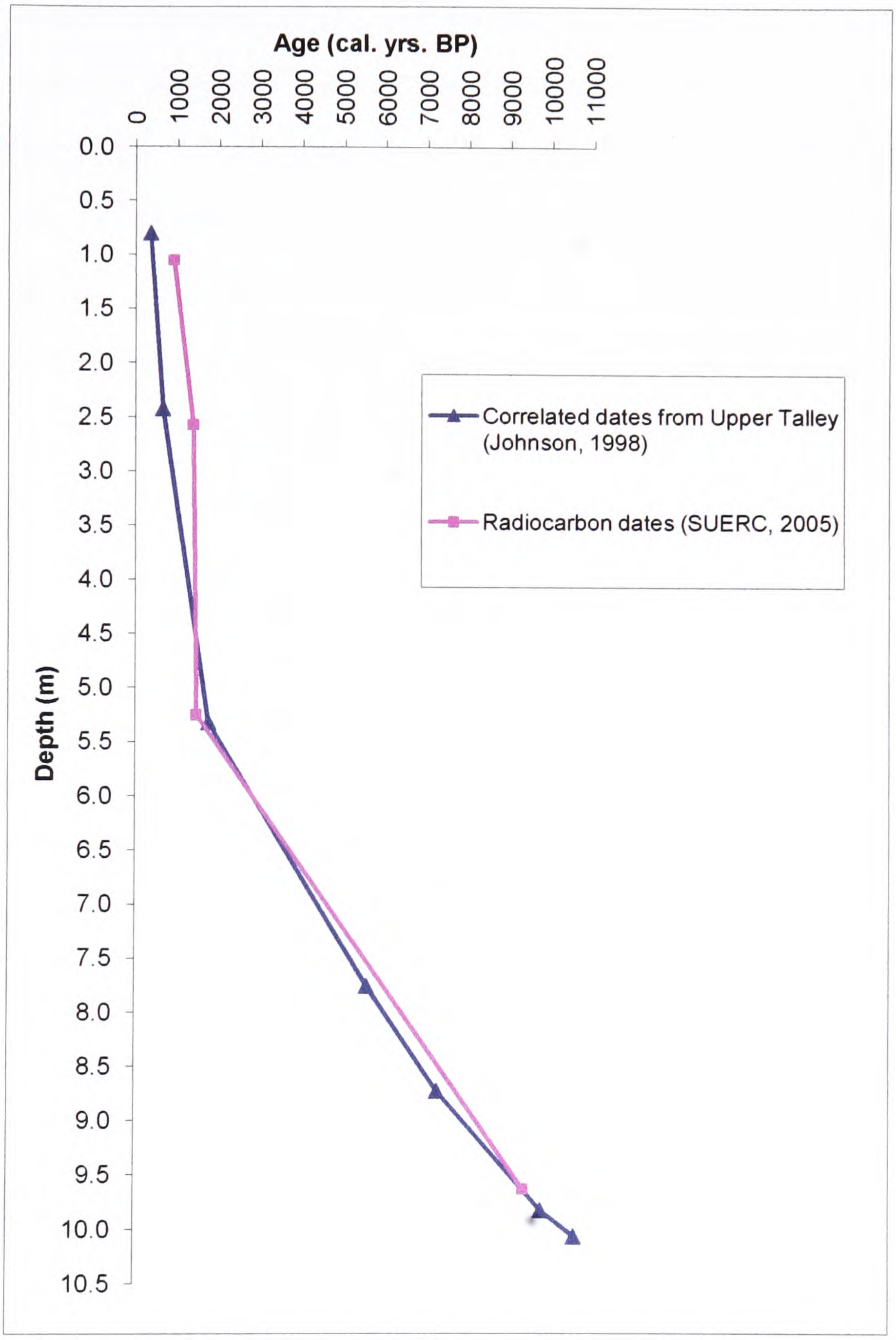


Figure 5.14 Diagram showing the comparison of age depth curves from SUERC (2005) and from Johnson (1998)

significant human activity around Upper Talley during the 8th century AD. Post-Roman Wales, however, was characterized by a distinct lack of upland settlements, accompanied by forest regeneration on the higher grounds possibly reflecting human migration associated with the Sub-Atlantic climatic deterioration (Lamb, 1977). A major phase of impact during this period would also contradict the statement by Giraldus Cambrensis (1146-1223 AD) who in the 12th century describes the Talley Lakes as a 'rough and sterile spot, surrounded by woods and trees on every side' suggesting little previous impact to the immediate landscape.

Although the more recent dates from SUERC were discarded based on the above assessment of documented catchment conditions, the chronology should still be treated with caution. Figure 5.15 gives the age-depth curve used for the length of the core based on the selected dates. Extrapolation of dates for the remainder of the core allowed for comparison with documented phases of human activity within the lake catchment as potential causes of limnological change.

Publication Code	**Sample Depth (m)**	**Measured Age (^{14}C yrs BP ±1 σ)**	**^{13}C ‰**
SUERC-4382	1.04-1.06	895±40	-23.4
SUERC-4410	2.52-2.62	1365±40	-30.1
SUERC-4383	5.24-5.26	1455±35	-23.7
SUERC-4384	9.62-9.64	9310±45	-27.7

Table 5.5 Table showing calibrated ^{14}C ages for samples from the Upper Talley core (UT2/94).

Bio-stratigraphic Horizon	**Depth (m) at Lower Talley**	**Depth (m) at Upper Talley**	**Correlated Radiocarbon Dates (cal. yrs. BP)**
Cannabis peak	1.40	0.8	350
Magnetic susceptibility peak	2.56	2.42	650
LOI (550°C) dip	6.16	5.32	1, 750
Ulmus decline	7.08-6.92	7.68-7.92	5, 550
Magnetic susceptibility peak	9.16	10.06	10, 550
Magnetic susceptibility peak	10.04	11.20	12,550

Table 5.6 Table showing the bio-stratigraphic markers and magnetic susceptibility horizons cross-correlated from the Lower to the Upper Talley core (Johnson, 1998).

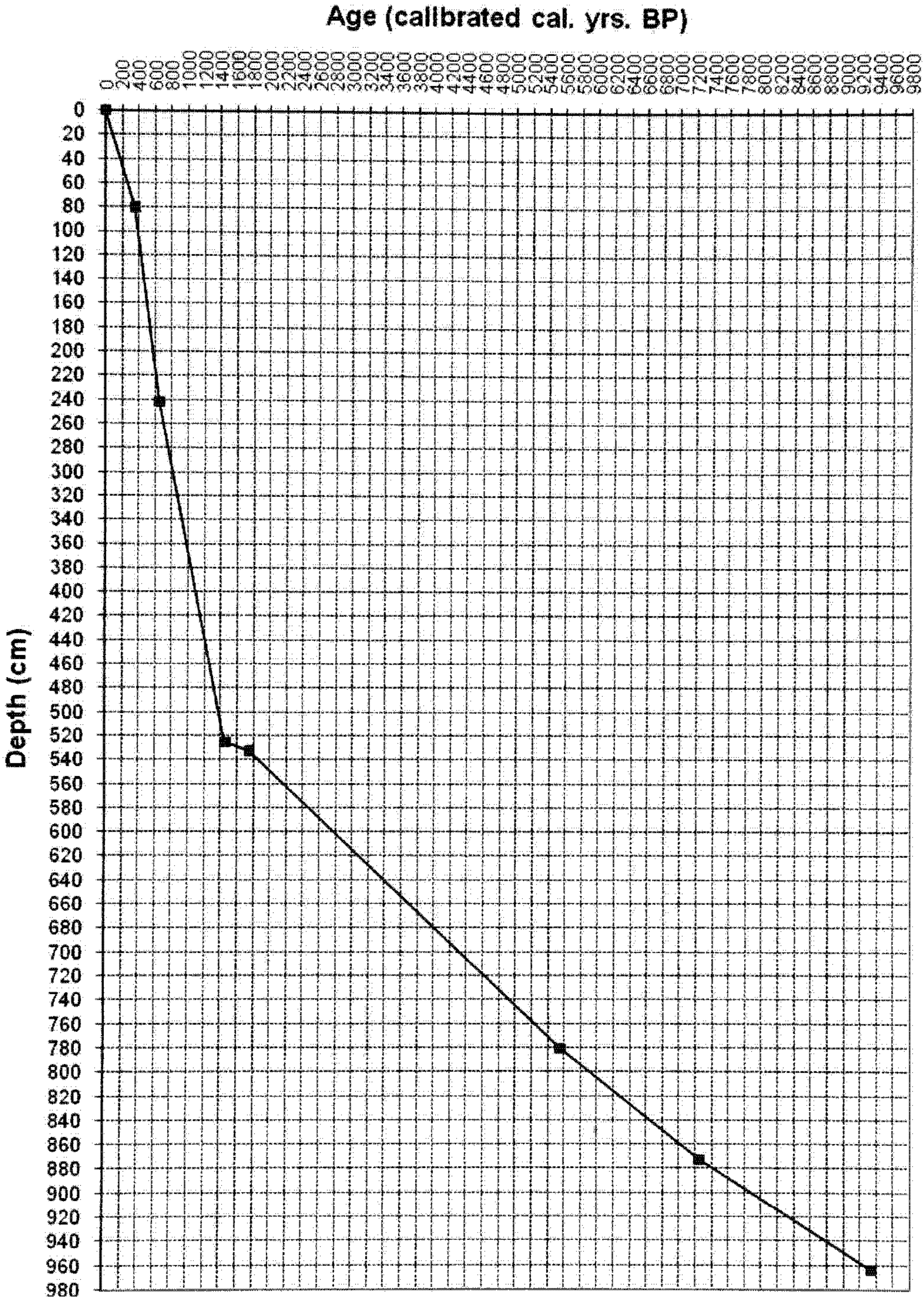


Figure 5.15 Table showing the final age-depth model for UT2/94 constructed from a combination of dates from Table 5.4 and Johnson (1998).

CHAPTER 6

RESULTS FROM THE LLYN PENCARREG CORE (LPC/05)

The results from Llyn Pencarreg are presented in the following chapter and, as with Upper Talley, the chronological data are reported based on the age-depth model in Section 6.18.

6.1 Water Chemistry

A set of basic water chemistry measurements taken in March 2005 indicated that Llyn Pencarreg is circumneutral (pH 6.6) and eutrophic (DI-TP; 31 $\mu g\ l^{-1}$). Waters were well oxygenated (DO; 93.3%) with a low conductivity value of 72 $\mu S\ cm^{-1}$.

6.2 Lithology

The core lithology (Table 6.1) indicated that the basal sediments were composed of a firm consolidated, very dark brown, fibrous lake-mud, which continued to a depth of 5.80m. Plant macrofossils, such as leaves, were present in high abundance between 7.00m and 5.70m and a layer of well-preserved wood, which inhibited core recovery, was found between 6.56 and 6.54m. A gradual transition to a less consolidated, brown, homogenous lake gyttja occurred between 5.80m and 5.00m and a second gradual

Depth (m)	Boundary Depths (m)	Laboratory Sediment Description	Munsell Description (MDKI, 1994)	Troel-Smith Notation (Troels-Smith, 1955)
0-				
1.00-				
2.00- 3.00- 4.00-	2.00 – 1.50	Gradual transition to soft, green-brown, organic rich, unconsolidated gyttja.	Olive brown; 2.5YR 4/4	Ld.2, As.2, Ag+, nig.1, strf. 0, sicc.2, elas 2-3, Ld.2-3, Dh.2
5.00- 6.00-	5.80 – 5.00 6.56 – 6.54	Gradual transition to brown homogenous lake mud Layer of wood.	Brown; 10YR 5/3	Lso.2, As.2, Ag+, nig.2, strf.), sicc.2, elas.2-3, Ld.2-3, Dh.2
7.00-		Firm consolidated, very dark brown lake mud with plant macrofossils, which remain throughout the core.	Very dark brown; 10YR 2.5/2	Lso.2, As.2, Ag+, nig.3, strf.0, sicc.2, elas.3, Ld.2-3, Dh.2

Table 6.1 A detailed lithology describing changes in the colour and texture of core LPC/05

transition to soft, green-brown, organic-rich unconsolidated gyttja occurred between 2.00m and 1.50m. This unit continued to the top of the core.

6.3 Magnetic Susceptibility

The results for magnetic susceptibility are given in Figure 6.1. In LPOZ-1 magnetic susceptibility was initially stable (mean –0.7) with a maximum value of -0.2 at 6.10m. An initial decrease in LPOZ-2 was followed by a sharp increase to 0.2 at 5.70m and a further decrease to -0.2 at the upper zone boundary. In LPOZ-3 relatively stable values were followed by a decrease from 0.2 at 4.50m to -1.0 at 4.00m, producing the minimum of the sequence. In LPOZ-4 magnetic susceptibility was relatively stable (mean –0.5) to a depth of 2.50m, then displayed a sustained increase, which continued to 1.30m in LPOZ-5 where a value of 1.1 was recorded. The latter increase was followed by a sharp decrease to –0.5 at the upper zone boundary. Finally, in UTOZ-6 values initially fluctuated around a mean of 0.25 then, from 10cm, increased sharply to the top of the core producing the maximum of the entire sequence (1.5).

6.4 Organic Content

In LPOZ-1 (Figure 6.1) organic content remained exceptionally high, falling between 84% and 92%, with the maximum of the entire sequence (92%) at the core base. Above a depth of 5.28m (LPOZ-2) organic content decreased suddenly and sharply to 37%. Values then remained relatively stable (mean 44%) to a depth of 3.50m (LPOZ-3) followed by an increase to 63% at 3.28m. An immediate decrease to 50% in LPOZ-4 was followed by stable values to the top of the zone (mean 55%). In LPOZ-5, organic content produced a peak (68%) at 1.75m then decreased to 30% at the top of the zone. Finally in LPOZ-6, organic content decreased sharply to 15% at 3.5cm, producing the minimum of the entire sequence, followed by a final increase to 27% at the top of the core.

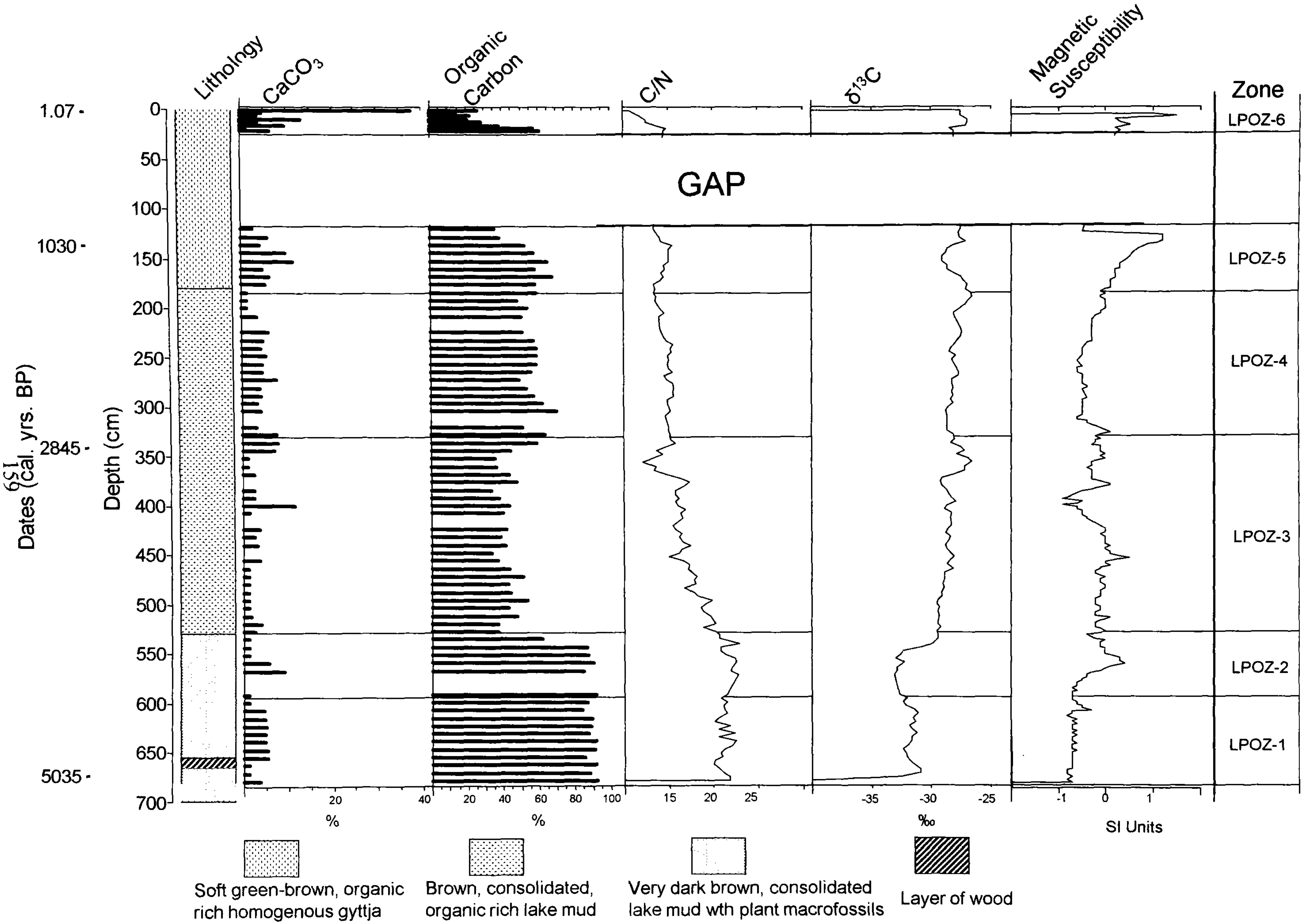


Figure 6.1 Diagram showing the lithological results for core LPC/05 including sediment lithology, % $CaCO_3$ (calculated from LOI at 850°C), % organic carbon (LOI at 550°C), C/N ratios, $\delta^{13}C$ and magnetic susceptibility values.

6.5 Calcium Carbonate ($CaCO_3$)

In LPOZ-1 (Figure 6.1) sedimentary $CaCO_3$ content fluctuated between 1% and 5% (mean 3%) followed by an increase to 9% at 5.20m in LPOZ-2. Percentages decreased to a mean of 2% in the upper part of LPOZ-2, which continued into LPOZ-3 to a depth of 4.00m, where a value of 15% was produced. Lower values (*ca.* 2%) were maintained to the upper zone boundary where percentages increased to 8% (3.95m). An initial decrease at the base of LPOZ-4 was followed by relatively stable values (mean 6%) and a decrease to 1% at a depth of 2.00m. In UTOZ-5 an increase to 12% at 1.55m was followed by a decrease to 4% at the top of the zone. Finally in UTOZ-6, estimated $CaCO_3$ content increased towards the upper zone boundary, fluctuating sharply between a minimum of 1% and maximum of 13% then increased to the maximum of the entire sequence (28%) at1.5cm.

6.6 Carbon Nitrogen (C/N) Ratio

The C/N profile (Figure 6.1) displayed a general decreasing trend towards the top of the core. A basal value of 21.8 in LPOZ-1 was followed by a fluctuating trend with a value of 22.4 at the top of the LPOZ-2 (5.40m). Ratios decreased for the whole of LPOZ-3, with a sudden decrease to 12.5 at a depth of 3.60m, followed by a sharp increase to 16.6 at the upper zone boundary. In LPOZ-4 ratios formed a gradual decreasing trend to 13.6 at the upper zone boundary then in LPOZ-5 values increased to a peak of 15.5 at a depth of 1.36m before decreasing back to 13.0 at the top of the zone. Finally, in UTOZ-6 values decreased sharply from 14.5 at the zone base to 10.7 at the top of the zone, producing the minimum value recorded in the entire sequence.

6.7 Stable Carbon Isotopes ($\delta^{13}C$)

In LPOZ-1 (Figure 6.1) a basal value of –32.4 ‰ immediately increased to –30.9 ‰ at 6.68m. Although $\delta^{13}C$ fluctuated, a general decreasing trend followed to a value of –33.0 ‰ at a depth of 5.60m (LPOZ-2). At the top of UTOZ-2, $\delta^{13}C$ increased sharply to –29.4 ‰ (5.36m). In UTOZ-3 values fluctuated between a maximum of –28.0 ‰ and a minimum of –29.0 ‰ then increased sharply to –26.5 ‰ at 3.60m, producing the maximum ratio of the sequence. This was followed by a decrease to –28.6 ‰ at the top of the zone. A general increase in UTOZ-4 produced a value of -27.4 ‰ at the upper zone boundary and in LPOZ-5, a sharp decrease to –29.1 ‰ at 1.52m was followed by an increase to –27.1 ‰ (1.32m). Finally, C/N ratios in LPOZ-6 increased from –28.1 ‰ to –27.0 ‰, at 9.5m, followed by a decrease to –27 ‰ at the top of the core.

6.8 Carbon Isotope Analysis of modern plant samples

The results of carbon isotope analysis of modern vegetation (Table 6.2 and Figure 6.2) indicated that with the exception of *Urtica, Alnus glutinosa* and mosses, $\delta^{13}C$ values fell within the expected ranges for algae and C_3 land plants (–25 to –30). Although algae fell within the expected C/N range (7.2) (Meyers and Teranes, 2001), the values for terrestrial plants were not clearly represented. Figure 6.2, for example, shows that *Rumex*, Poaceae, mosses and *Urtica* produced C/N values, which fell within the algal range (4 – 10).

Plant Type and Date of Collection	**$\delta^{13}C$ (‰)**	**C/N Ratio**
Alnus glutinosa seeds (22/03/05)	-30.3	19.9
Catkins (22/03/05)	-29.6	33.5
Urtica (22/03/05)	-30.1	6.5
Rubus fruticosus leaves (22/03/05)	-29.3	20.1
Poaceae (22/03/05)	-30.0	8.6
Juncaceae (22/03/05)	-29.2	21.2
Rumex (22/03/05)	-29.6	8.4
Moss (22/03/05)	-30.3	7.3
Algae from lake edge (22/03/05)	-29.4	7.2
Phragmites (22/03/05)	-29.4	37.7
Sphagnum (22/03/05)	-29.3	20.1

Table 6.2 Table showing the $\delta^{13}C$ (‰) and C/N results of modern vegetation samples from Llyn Pencarreg and its local catchment.

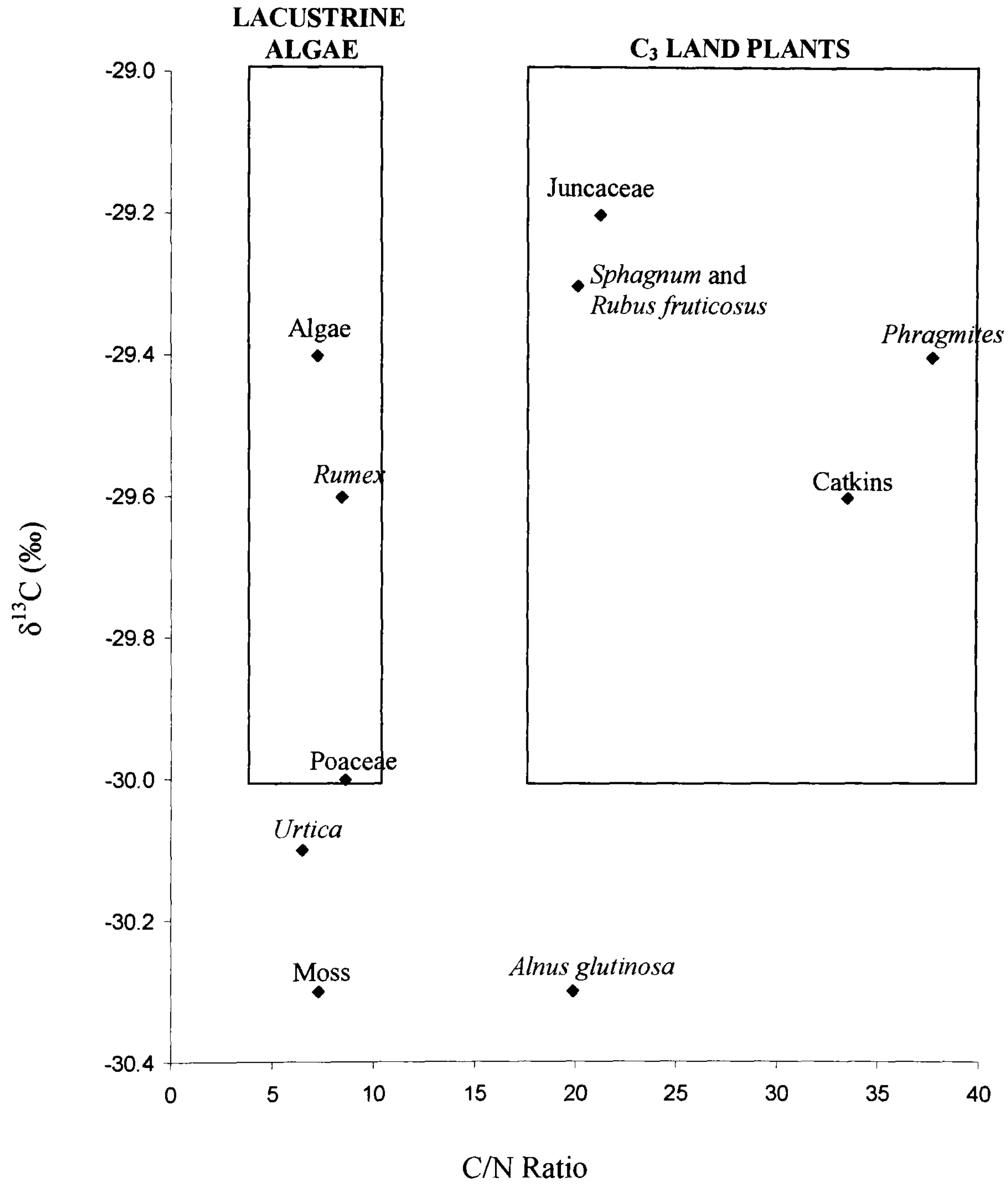


Figure 6.2 Diagram showing the elemental and carbon isotope composition of organic matter in modern samples from Llyn Pencarreg compared to the classification of bulk organic matter from lacustrine algae and C_3 land plants (Meyers and Teranes, 2001)

6.9 Diatom Analysis

The preservation of diatom frustules was poor towards the core base and diatoms were missing completely below 6.14m. Above this depth a total of 55 taxa were found to be present. 47 taxa were present at >2% in at least one sample (Table 6.3). The results of diatom analysis, expressed as percentage counts for taxa occurring at >2%, are given in Figure 6.3. The decision to display taxa occurring at >2% (rather than >4% as displayed at Upper Talley) was based on the presence of taxa in the bottom sample, which were significant to the overall interpretation and were excluded from the profile at >4%. Biostratigraphic boundaries were defined with the aid of CONISS (Grimm, 1987). Zones were named Llyn Pencarreg Algal Zones (LPAZ).

LPAZ-1: 6.90-5.95m (*ca.* 5.2 – 4.6 k cal. yrs. BP)

From a depth of 6.14m to the top of LPAZ-1 the diatom assemblage was composed of planktonic (*Aulacoseira granulata*, *Cyclotella stelligera*, *C. meneghiniana*, *Stephanodiscus minutulus*, *Asterionalla formosa* and *Tabellaria flocculosa*) and benthic (*Amphipleura pelucida*, *Navicula menisculus*, *N. recens*, *N. saxophila*, *N. cryptotenella*, *N. cryptocephala*, *Cymbella silesiaca*, *Achnanthes minutissima*, *A. lanceolata*, *Nitzschia palaea*, *Cocconeis placentula*, *Eunotia bilunaris*, *Fragilaria virescens*, *F. exigua*, *F. capucina* var. *gracilis*, *F. pinnata* and *Amphora libyca*) taxa. *A. granulata* was the most common with an abundance of *ca.* 40%.

Species Name	**EDDI Code**	**TP Optima ($\mu g\ l^{-1}$) transformed from EDDI $\log_{10}$ values**	**Trophic Status**	**Life Form**
Achnanthes lanceolata	XXG978	125.36	Hypereutrophic	Benthic
A. minutissima	XXG997	24.62	Mesotrophic	Benthic
A. subatomoides	AC136A	19.67	Mesotrophic	Benthic
Amphipleura pellucida	AP001A	24.05	Mesotrophic	Benthic
Amphora libyca	AM011A	74.25	Eutrophic	Benthic
Anomoeoneis brachyseira	AN9999			Benthic
Asterionella formosa	AS001A	44.01	Meso - eutrophic	Benthic
Aulacoseira distans				
Cocconeis placentula	XXG977	132.09	Hypereutrophic	Benthic
Cyclotella meneghiniana	CY003A	254.41	Hypereutrophic	Planktonic
C. ocellata	XXG991	34.49	Eutrophic	Planktonic
C. stelligera	XXG968	18.46	Mesotrophic	Planktonic
Cymbella affinis	CM022A	16.89	Mesotrophic	Benthic
C. silesiaca	CM103A	16.52	Mesotrophic	Benthic
Eunotia bilunaris	XXG949	15.26	Mesotrophic	Benthic
Fragilaria capucina	FR009A	22.06	Mesotrophic	Benthic
F. capucina var. *gracilis*	FR009H	50.91	Eutrophic	Benthic
F. capucina var. *perminuta*	FR009J			Benthic
F. exigua	XXG961	15.98	Mesotrophic	Benthic
F. pinnata	XXG995	34.62	Eutrophic	Benthic
F. vaucheriae	FR007A	52.84	Eutrophic	Benthic
F. virescens	XXG961	15.98	Mesotrophic	Benthic
Frustula rhomboides	FU002A	14.04	Mesotrophic	Benthic
Gomphonema angustatum	XXG940	49.82	Eutrophic	Benthic
G. gracile	GO004A	37.27	Eutrophic	Benthic
G. parvulum	XXG938	65.89	Eutrophic	Benthic
Navicula cryptocephala	NA007A	56.33	Eutrophic	Benthic

N. cryptotenella	XXG979	25.09	Eutrophic	Benthic
N. menisculus	XXG966	51.01	Eutrophic	Benthic
N. minima	NA042A	83.95	Eutrophic	Benthic
N. pupula	XXG930	47.82	Eutrophic	Benthic
N. recens				
N. rotunda	NA090A	26.08	Meso - eutrophic	Benthic
N. saxophila				
N. submeniscula	NA134A	39.99	Eutrophic	Benthic
Neidium alpinum	NE006A	12.18	Mesotrophic	Benthic
N. bisulcatum	NE004A			Benthic
Nitzschia bacillum	NI211A	14.11	Mesotrophic	Benthic
N. palea	XXG976	80.25	Eutrophic	Benthic
N. linearis	XXG927	109.64	Eutrophic	Benthic
Pinnularia divergens	PI008A			Benthic
P. microstauron	PI011A	25.65	Meso - eutrophic	Benthic
P. subcapitata	PI022A	58.10	Eutrophic	Benthic
Stauroneis anceps	SA001A	20.97	Mesotrophic	Benthic
Stephanodiscus minutulus	XXG974	40.12	Eutrophic	Planktonic
Tabellaria flocculosa	XXG981	14.81	Mesotrophic	Planktonic

Table 6.3 Diatom taxa recorded at >2% abundance in at least one sample of core LPC/05 with their EDDI codes (Battarbee *et al.*, 2001), TP optima (Battarbee *et al.*, 2001), trophic preference (Vollenweider and Kerekes, 1981) and ecological life forms (Barber, 1981; Germain, 1981; Sims, 1996).

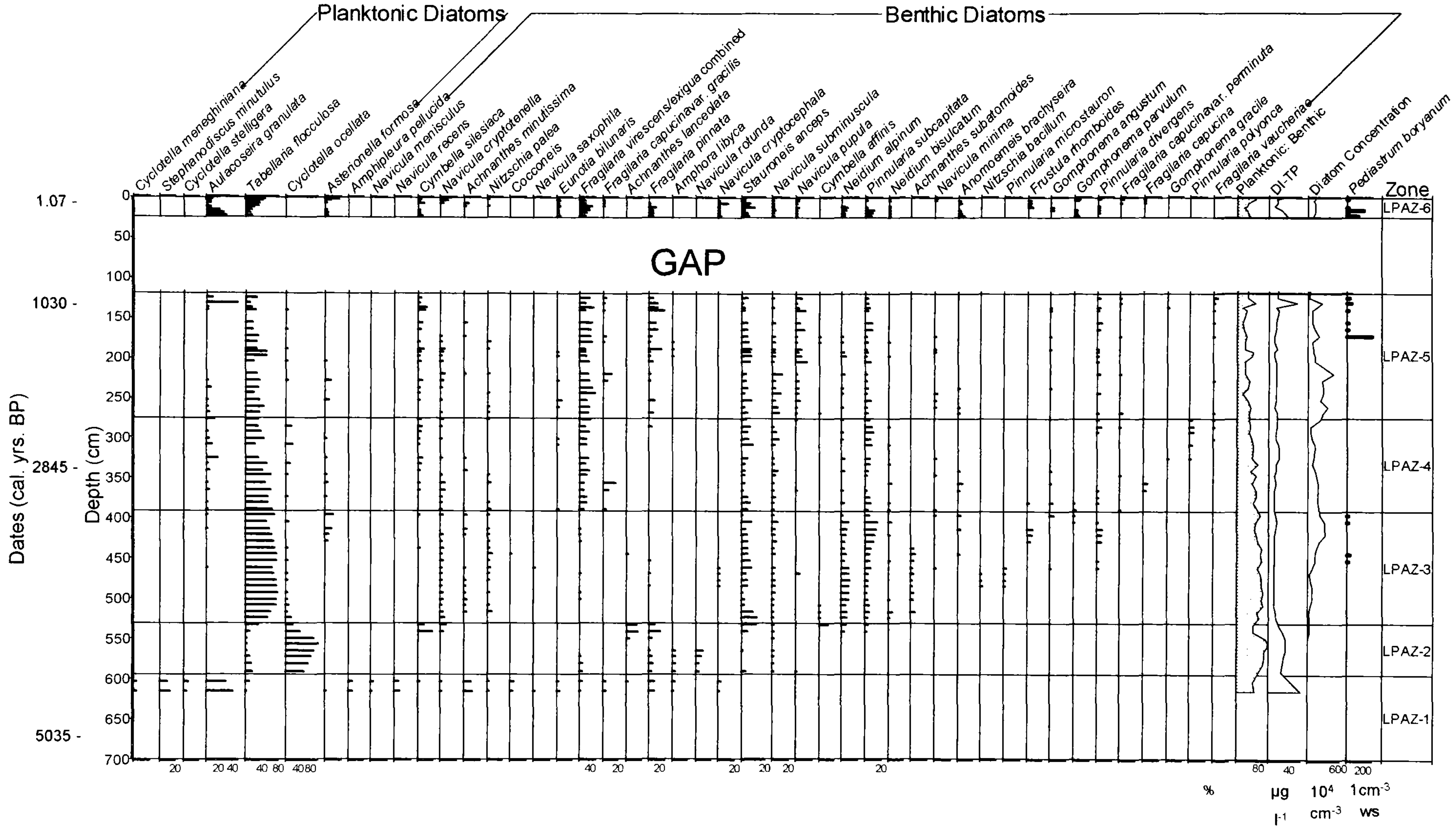


Figure 6.3 Summary diatom diagram for core LPC/05 showing diatom taxa present at> 2%, the planktonic:benthic ratio, diatom-inferred total phosphorus (DI-TP), diatom concentration and *Pediastrum*. Biostratigraphic zones were defined using CONISS.

LPAZ-2: 5.95-5.40m (*ca.* 4.6 – 4.2 k cal. yrs. BP)

A marked and abrupt transition occurred at the zone boundary characterized by the replacement of *A. granulata*, *C. stelligera*, *C. meneghiniana* and *S. minutulus* with *Stauroneis anceps* and *C. ocellata* and the loss of many benthic taxa. *C. ocellata* abundance immediately increased to 85% and dominated the zone. At a depth of 5.50m *T. flocculosa* plus a range of benthic taxa (*S. anceps*, *C. silesiaca, N. cryptotenella*, *A. lanceolata, Cymbella affinis, Neidium alpinum, Pinnularia subcapitata*) increased in relative abundance.

LPAZ-3: 5.40-3.90m (ca. 4.2 – 3.2 k cal. yrs. BP)

A second major transition occurred at the zone base marked by significant increases in *T. flocculosa* and *S. anceps* at the expense of *C. ocellata,* which decreased to 15% before disappearing from the record. *T. flocculosa* immediately increased to 60% relative abundance and dominated the entire zone. Planktonic *Asterionella formosa* and a range of benthic taxa reappeared or were recorded for the first time (*Anomoeoneis brachyseira, Frustulia rhomboides, Navicula alpinum, Pinnularia divergen*s and *P. subcapitata*) in this zone.

LPAZ-4: 3.90-2.75m (*ca.* 3.2 – 2.2 k cal. yrs. BP)

Although *T. flocculosa* continued to dominate assemblages in this zone it gradually decreased to 30% relative abundance at the upper zone boundary, partly in response to the increase in *A. granulata, C. ocellata, A. formosa* and benthic *S. anceps*. *F. virescens* and *F. exigua* increased in abundance at the zone base at the expense of *P. subcapitata* and *N. alpinum*, whilst the P:B ratio increased generally towards the top of the zone in response to the introduction of new benthic taxa (*Fragilaria capucina* var. *perminuta, F. capucina, F. vaucheriae* and *Pinnularia polyonca*). These trends produced a high diversity assemblage for the entire zone.

LPAZ-5: 2.75-1.40 (*ca.* 2.2 – 1.1 k cal. yrs. BP)

The co-dominance of *T. flocculosa* and *S. anceps* continued with both forming peaks of 50% and 10% respectively at a depth of 1.85m before decreasing towards the upper zone boundary. The zone base was marked by increases in the abundance of other benthic taxa including *N. palea, F. capucina* var. *gracilis, F. pinnata, N. pupula, N. minima* and *A. brachyseira*. *A. granulata* proportions increased significantly in one sample (to 50%) at a depth of 1.35m. Among the benthic taxa *F. exigua* and *F. virescens* continued to increase in relative abundance, peaking at 58%.

LPAZ-6: 0.22 - 0cm (*ca.* 0.1 – 0 k cal. yrs. BP)

Following a gap in the core sediments between 1.40m and 25cm, planktonic *A. granulata* abundance declined and was replaced by increases in *T. flocculosa* and *A. formosa* at the midzone where *C. ocellata* was also recorded. A wide range of benthic taxa continued to be present (*C. silesiaca, E. bilunaris, F. virescens, F. exigua, F. capucina* var. *gracilis, F.*

pinnata, F. capucina, N. cryptocephala, N. submeniscula, N. pupula, N. alpinum, N bisulcatum, P. subcapitata, A. brachyseira, G. parvulum, N. cryptotenella, A. minutissima, N. palea, N. minima, F. rhomboides, G. angustatum and *P. divergens*) and showed changes in abundance and species replacement at the mid zone.

6.10 DI-TP Reconstruction

The results of DI-TP reconstruction are given in Figure 6.3. Good representation of fossil taxa in the modern dataset (98%) indicates that the reconstruction is likely to be reliable. The basal DI-TP estimate of 59.6 μg l^{-1} was the maximum of the sequence and was followed by an immediate decrease to 26.0 μg l^{-1} at 5.90m in LPAZ-2. An increase to 35.0 μg l^{-1} (5.60m) was then followed by a decrease to 17.7 μg l^{-1} the upper zone boundary. In LPAZ-3, and to a depth of 3.20m in UTAZ-4, DI-TP estimates were extremely stable (mean 13.3 μg l^{-1}). An increase to *ca.* 19.0 μg l^{-1} persisted to a depth of 1.85m in LPAZ-5 and was followed by a further gradual increase to a sharp peak of 53 μg l^{-1} at 1.30m. Finally, in LPAZ-6 DI-TP decreased from 34.9 μg l^{-1} at the zone base to 11.5 μg l^{-1} at 10cm, followed by an increase to 25.3 μg l^{-1} at the top of the core.

The sample specific errors of prediction and confidence intervals estimated by the bootstrapped coefficient of determination are given in Figure 6.4 with an RMSE $_{boot}$ value of 0.34. The overlap of error bars indicates that fluctuations in DI-TP are not statistically significant. As in Upper Talley, the dominant taxa are both abundant and well represented in the training set, but they are all common, morphologically variable

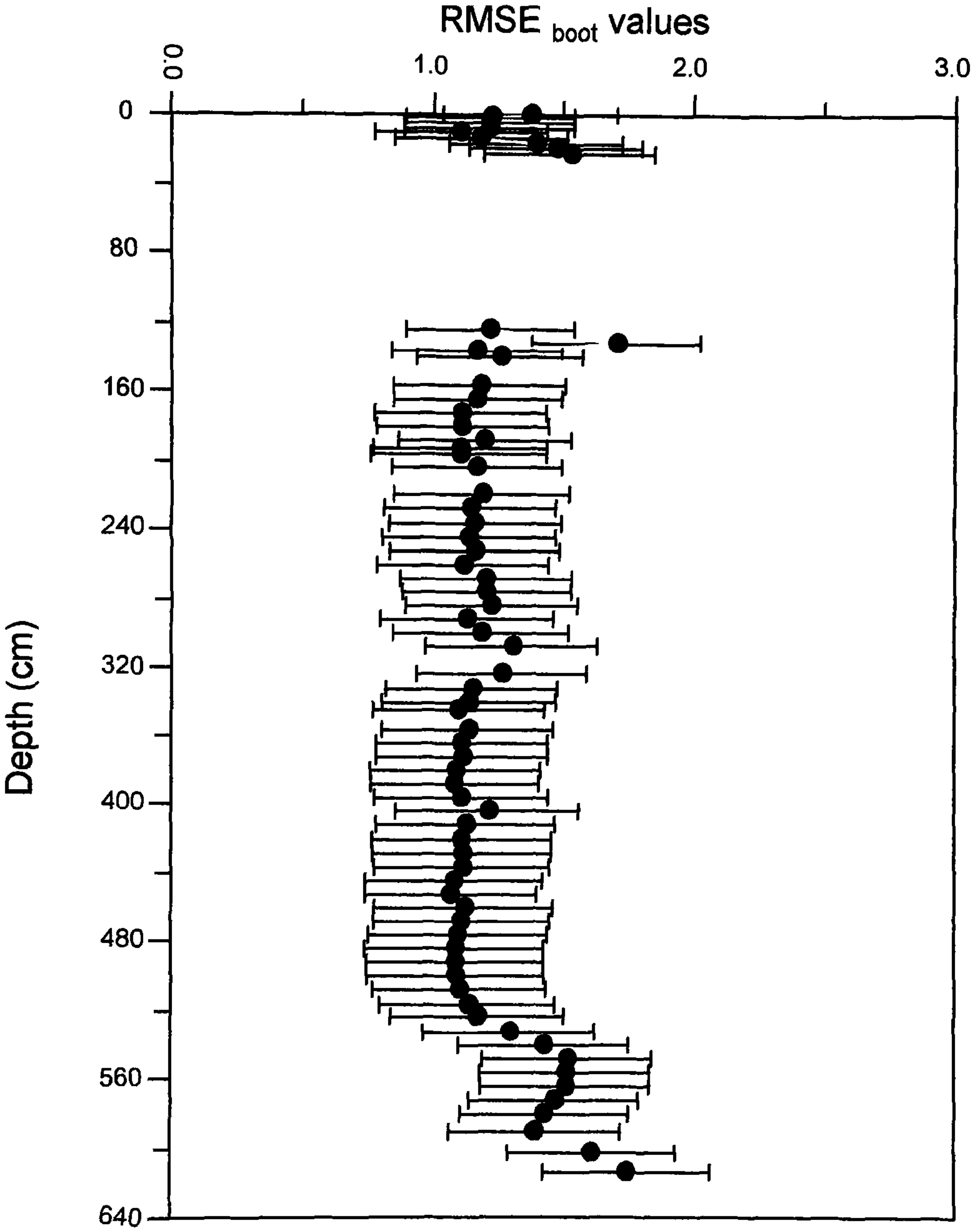


Figure 6.4 Scatter plot of the sample specific errors of prediction for Llyn Pencarreg (LPC/05) estimated by bootstrapping and presented with error bars

taxa which are merged in the EDDI training set and have broad tolerance ranges (*A. granulata* $N_2 = 32.3$; Tol = 0.43 $\log_{10}$ µg l^{-1}; *C. ocellata* $N_2 = 29.8$; Tol = 0.36 $\log_{10}$ µg l^{-1}; *T. flocculosa* $N_2 = 29.1$; Tol = 0.42 $\log_{10}$ µg l^{-1}). Although they include classic eutrophic species, statistically, they are not strong indicator taxa. The results are therefore presented, but interpreted with caution.

6.11 Diatom Concentration

In LPAZ-1 and -2 (Figure 6.3) the low abundance of diatom frustules and extremely high relative proportion of microspheres prevented the calculation of concentration densities. At the base of LPAZ-3 an estimated concentration of $2.4\text{x}10^5$ cm^{-3} gradually increased to $4.1\text{x}10^7$ cm^{-3} at 4.20m. A steady and sustained decrease followed and continued into LPAZ-4 where a concentration of $6.7\text{x}10^6\text{cm}^{-3}$ was produced at a depth of 2.84m. At the upper zone boundary diatom concentrations began to increase, rising to $6.2\text{x}10^7$ cm^{-3} at 2.20m. Following a decrease to $9.00\text{x}10^6$ cm^{-3} at 1.40m concentrations peaked sharply to an estimate of $3.5\text{x}10^7$ cm^{-3} (1.32m). Finally, in LPAZ-6, diatom concentrations increased from $1.5\text{x}10^7$ cm^{-3} at the zone base to $1.6\text{x}10^7$ cm^{-3} at a depth of 1cm.

6.12 Data Analysis

6.12.1 Detrended Correspondence Analysis (DCA)

DCA axis 1 (λ_1 = 0.572), DCA axis 2 (λ_2 = 0.209) and DCA Axis 3 (λ_3 = 0.057) explained 35%, 13% and 4% of the variance in the diatom percentage data, respectively, indicating that a moderately high proportion of the total percentage variance was summarised in the first two axes. Apart from samples from LPAZ-2, which form a cluster with high scores on Axis 1, the majority of Llyn Pencarreg samples cluster together with low scores on Axis 1. This reflects the fact that the most significant element of variance in the diatom data is between LPAZ-2, during the lake's early history. Species with high scores on Axis 1, which characterise LPAZ-2, include *C. ocellata* (CY009A) and *A. lanceolata* (AC001A). To explore variance over the rest of the sequence to the present day in more detail, a plot is given in Figure 6.5 of Axis 2 against Axis 3.

The distribution of diatom taxa is not clearly driven by nutrient status. Although mesotrophic *P. microstauron* (PI011A), *A. subatomoides* (AC136A) and *N. bacillum* (NI211A) (TP optima; 25.7 $\mu g\ l^{-1}$, 19.7 $\mu g\ l^{-1}$ and 14.1 $\mu g\ l^{-1}$ respectively) plot closely together in the same ordination space with high scores on Axis 2, species of low trophic status with low scores can also be found (e.g., *E. bilunaris*; XXG949, *C. silesiaca*; CM103A, *C. affinis*; CM022A and *F. exigua*; XXG961).

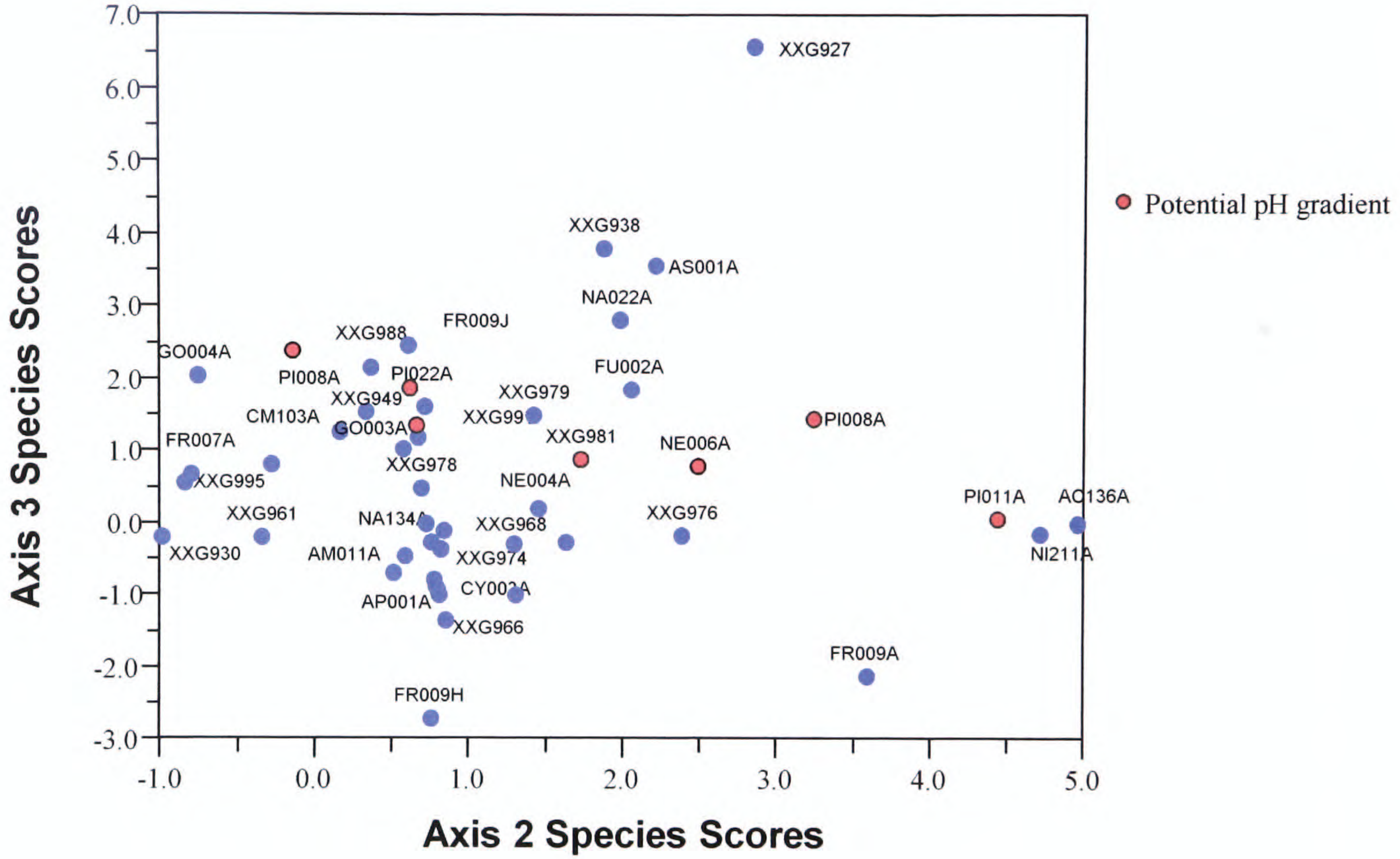


Figure 6.5 Scatter plot of the DCA species scores for Axis 1 against Axis 2 for the Llyn Pencarreg core (LPC/05) based on 55 taxa.

Taxa of high trophic status are similarly distributed with *N. linearis* (XXG927), *A. granulata* (XXG988) and *C. placentula* (CO9999), which have TP optima of 109.6 $\mu g\ l^{-1}$, 133.5 $\mu g\ l^{-1}$ and 132.1 $\mu g\ l^{-1}$ respectively, spread along Axis 2. The results suggest that the underlying pattern of species distribution is complex and may be influenced by multiple factors such as temperature and pH. For example, taxa with low pH (e.g. *P. subcapitata* (PI022A); pH 4.9 and *G angustatum* (GO003A); pH 4.0) plot to the left of the diagram, whereas species with relatively high pH optima (e.g. *T. flocculosa* (XXG981); pH 7.1, *P. microstauron* (PI011A); pH 6.8, *P. divergens* (PI008A); pH6.5) plot to the right (Battarbee, 2004). When Axis 1 is plotted against core depth (Figure 6.6) the resulting trend follows a similar pattern to the abundance of the dominant taxon, *T. flocculosa*.

6.12.2 Hills Diversity Index (N_2)

Figure 6.7 shows that species richness and abundance fluctuated significantly. To a depth of 3.25 m values mirrored the trend in benthic taxa. From the core base a shift from 6.1 (6.14m) to 2.0 (5.56m) was followed by an increase to 6.9 (5.32m). Diversity values immediately decreased again to 3.9 at 3.84m followed by a sustained increase to 5.4 at 3.32m. A gradual decreasing trend followed, with a value of 2.3 at 1.56m, which then increased to 5.7 at 1.32m. Diversity finally decreased from 3.6 at 22.5cm to 2.9 at the mid-zone (10cm) followed by a final increase to 5.56 at the top of the core.

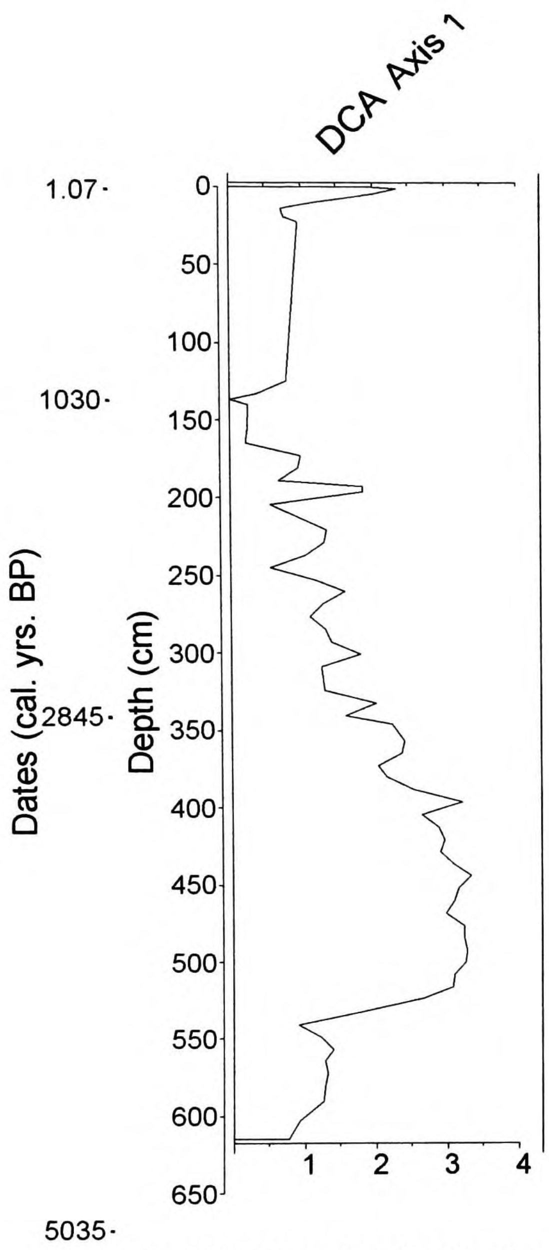


Figure 6.6 Diagram showing the temporal variation in Axis 1 of the sample scores data for fossil diatom assemblages in core LPC/05 based on 69 samples.

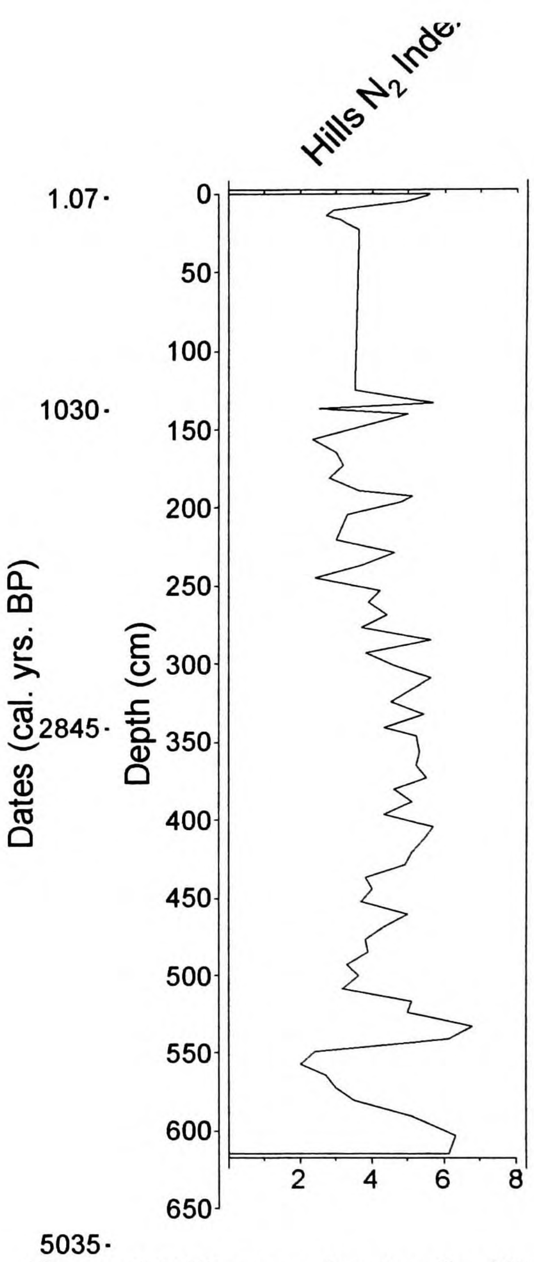


Figure 6.7 Diagram showing the Hills N_2 Diversity Index values for fossil diatoms in core LPC/05 based on 69 samples.

6.13 Cladocera Ephippia

Ephippia were absent for most of the sequence (Figure 6.8). Two major peaks in abundance were noted, the first in LPEPHZ-2 at a depth of 1.31m where 0.55 g^{-1} ds were observed producing the maximum of the sequence. The second was at 22cm where 0.3 g^{-1} ds were recorded followed by a decrease to 0.1 g^{-1} ds at a depth of 10cm.

6.14 *Pediastrum boryanum*

Only one species, *Pediastrum boranum*, was found (Figure 6.3). Apart from an increase to 500 cm^{-3} at a depth of 4.40m (LPAZ-3), *Pediastrum* colonies were recorded at low density in LPAZ-1 to LPAZ-4. In LPAZ-5 a marked peak at 1.75m was the maximum value for the sequence (3000 cm^{-3}). Finally, in LPAZ-6 *Pediastrum* abundance increased to 2500 cm^{-3} at 15cm followed by a decrease to 200 cm^{-3} at the top of the core.

6.15 Pollen Analysis

Results of pollen analysis are given in Figure 6.9, showing selected trees, shrubs, grasses, herbs and aquatic pollen. The results of CONISS (Grimm, 1987), based on the total species count, and resulting biostratigraphic zones are also displayed.

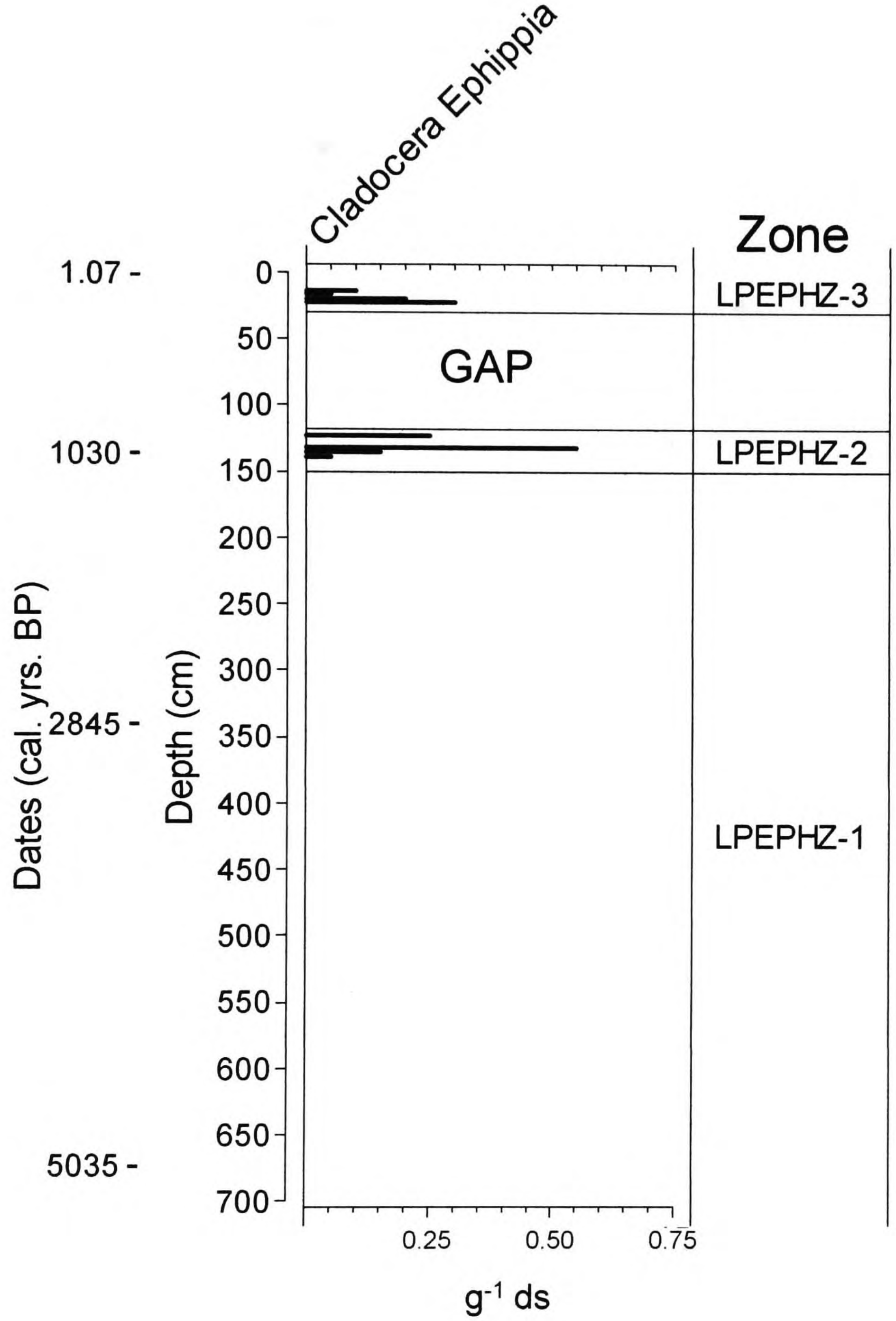


Figure 6.8 Diagram showing the abundance of cladocera ephippia in core LPC/94.

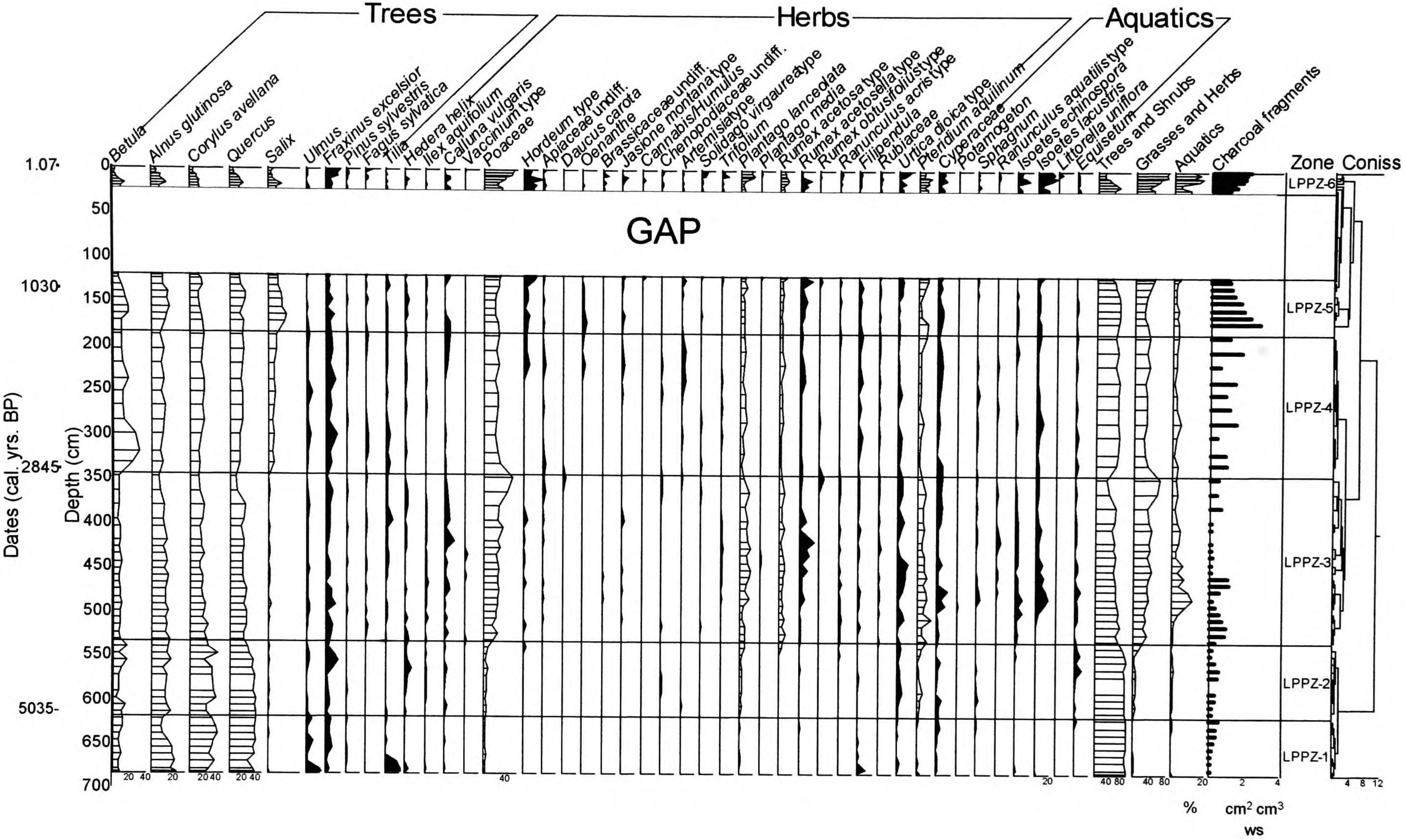


Figure 6.9 Diagram showing the abundance of selected pollen taxa and groups for core LPC/06 presented with charcoal abundance.

LPPZ-1: 6.90m – 6.20m (*ca.* 5.2 - 4.7 k cal. yrs. BP)

This zone was dominated by arboreal pollen (98%). At the core base *Quercus* pollen comprised 38%, *Corylus avallana* 22%, *Alnus glutinosa* 22% and *Betula* 8%. *Betula* and *Quercus* pollen percentages were relatively consistent for the entire zone, whilst *Corylus avellana* pollen percentages increased to 40% at the upper zone boundary. This was accompanied by a decrease in *Alnus glutinosa* pollen to 10% at the top of the zone. *Ulmus, Fraxinus excelsior, Pinus sylvestris* and *Tilia* pollen were also present from the core base and *Salix* pollen was first recorded at 6.55m. The non-arboreal land pollen component consisted of grasses and herbs (e.g. *Plantago lanceolata, Urtica, Pteridium aquilinum* and *Filipendula*).

The count of aquatic pollen was negligible and mainly characterized by *Cyperaceae* with occasional spores of *Isoetes lacrustis, I. echinospora* and *Equisetum*. *Sphagnum* spores were also present at the top of the zone.

LPPZ-2: 6.20m – 5.40m (ca. 4.7 – 4.2 k cal. yrs. BP)

Zone 2 was still dominated by arboreal pollen types. *Quercus* pollen percentages remained at *ca.* 38% to a depth of 5.60m then decreased to 22% at the top of the zone, whilst *Betula, Alnus glutinosa* and *Corylus avellana* pollen percentages fluctuated in abundance. *Ulmus* and *Fraxinus excelsior* pollen remained at low abundance and *Tilia* pollen disappeared from the record at the zone base followed by Pine (*Pinus sylvestris*) at a depth of 5.95m. Grass and herb pollen remained low at *ca.* 2%, although *Plantago lanceolata* pollen and *Pteridium aquilinum* spores percentages showed a minor increase towards the upper zone boundary, whilst *Hordeum*-type, *Rumex acetosa* and *Ranunculus acris*-type were first recorded at the top of the zone. No change was observed in the aquatic pollen and spore component.

LPPZ-3: 5.40m – 3.45m (*ca.* 4.2 – 2.9 k cal. yrs. BP)

Arboreal pollen percentages decreased from 63% of the land pollen sum at the zone base to 40% at the top of the zone. *Betula* pollen proportions remained constant throughout, whilst *Alnus glutinosa, Corylus avellana* and *Quercus* pollen displayed a steady decline in relative abundance, with the largest decrease at the top of the zone. *Fagus sylvatica* pollen was first recorded at the zone base and *Tilia* and *Pinus sylvestris* pollen were observed again from 5.15m and 5.00m respectively. Corresponding with these changes the proportion of herbs increased significantly and Poaceae pollen increased from 20% at the zone base to 50% at the top of the zone. *Plantago lanceolata* and *Rumex acetosa* pollen and *Pteridium* spores also increased in abundance at the zone base where other herb taxa associated with agricultural and pastoral activity were first recorded (eg. Apiaceae, Brassicaceae, *Jasione montana*, *Artemesia*-type, *Aster*-type, *Rumex acetosella* and *Trifolium*).

In this zone aquatic pollen and spores displayed a peak in relative abundance (12%) at a depth of 4.90m due to an increase in *I. lacrustis* spore abundance and the appearance of *Ranunculus aquatilis* pollen and *I. echinospora* and *Sphagnum* spores in the record.

Cyperaceae pollen continued to be present and *Potamogeton*-type pollen was recorded in just one sample at a depth of 5.00m. Following this peak, aquatic pollen and spores decreased in relative abundance to 5% at the top of the zone.

LPPZ-4: 3.45m – 1.85m (*ca.* 2.9 – 1.4 k cal. yrs. BP)

In LPPZ-4, the proportion of arboreal taxa increased to 70% at the zone base at the expense of grass and herb pollen taxa. This shift was sustained to the top of the zone. Although minor shifts were observed in the abundance of *Alnus glutinosa, Corylus avellana* and *Quercus* pollen, the increase in arboreal taxa was mainly the result of increases in the abundance of *Betula*, which increased to 30% at 3.75m, and *Salix*, which increased to 7% at the zone base. These increases were matched by a decrease in the relative abundance of Poaceae pollen to 20% at the zone base, whilst decreases were also observed in *Plantago lancoelata, Rumex acetosa* and *Pteridium aquilinum* abundances. *Oenanthe* pollen appeared in the record for the first time at a depth of 2.20m.

Aquatic pollen and spore abundance remained low at *ca.* 4% and was comprised mainly of *Isoetes lacustris* spores and Cyperaceae pollen.

LPPZ-5: 1.85m – 1.25m (*ca.* 1.4 – 0.9 k cal. yrs. BP)

Arboreal pollen maintained a mean abundance of 65% to the mid-zone then decreased to 43% at the upper zone boundary. This shift was due to decreases in *Betula, Alnus* *glutinosa, Corylus avellana, Quercus* and *Salix* pollen abundances following a significant increase of the latter at the zone base. Poaceae pollen proportions remained consistent at 20% to a depth of 1.60m then displayed a minor increase in relative abundance to 22% at the top of the zone. This was accompanied by increases in the percentages of *Plantago lanceolata*, *Hordeum*-type and *Rumex acetosa* pollen and the first records of *Cannabis sativa* and *Plantago media*. *Pteridium aquilinum* spores also displayed a significant increase in abundance towards the upper zone boundary. No change was observed in aquatic macrophyte pollen and spores, which remained constant at *ca.* 5% and were composed mainly of *Isoetes* spp. spores and Cyperaceae pollen.

LPPZ-6: 25cm – 0 (*ca.* 0.1 – 0 k cal. yrs. BP)

From the zone base to the top of the core a significant decrease in arboreal pollen was observed, including decreases in *Betula* percentages from 18% to 3%, *Alnus glutinosa* percentages from 13% to 2%, *Corylus avellana* percentages from 20% to 6%, *Quercus* percentages from 20% to 9% and *Salix* percentages from 5% to 2%. The major decrease in arboreal taxa was matched by an increase in Poaceae from 20% to 55%. *Hordeum*-type, *Rumex acetosa* pollen and *Pteridium aquilinum* spores were also present and a significant increase in the relative abundance of *Plantago lanceolata* percentages was recorded. Aquatic pollen and spore abundance expanded to its maximum abundance of the entire core (18%) at a depth of 7cm linked to a substantial increase in *Isoetes* spp. spore percentages.

6.15.1 Alternate Stable States

Comparison of the diatom concentration (DC) and aquatic pollen sum (APS) is given in Figure 6.10. Following a significant peak in APS at 5.00m, results indicated a steady decrease during the Bronze Age. The decrease correlated with an increase in DC, which peaked at 4.20m and may represent a shift from plant macrophyte to algal domination. APS remained low throughout the medieval period and DC peaked at 1.30m suggesting increased diatom production. Above the gap in core sediments DC remained constant, while APS fluctuated. A decrease to mid-zone was followed by a significant increase, a peak in production and a final decrease to the top of the core.

6.15.2 Diatom and Pollen Concentrations

Comparison of DC divided by PC with the original DC for each sample is given in Figure 6.11. Once the sediment accumulation factor is removed from the DC curve, the two graphs are comparable and suggest that variations in DC were not influenced by sediment accumulation rate. This suggests that within-lake abundance was directly influenced by factors such as increased diatom production during phases of enrichment and changes in silica (Si) concentrations (Anderson, 1989).

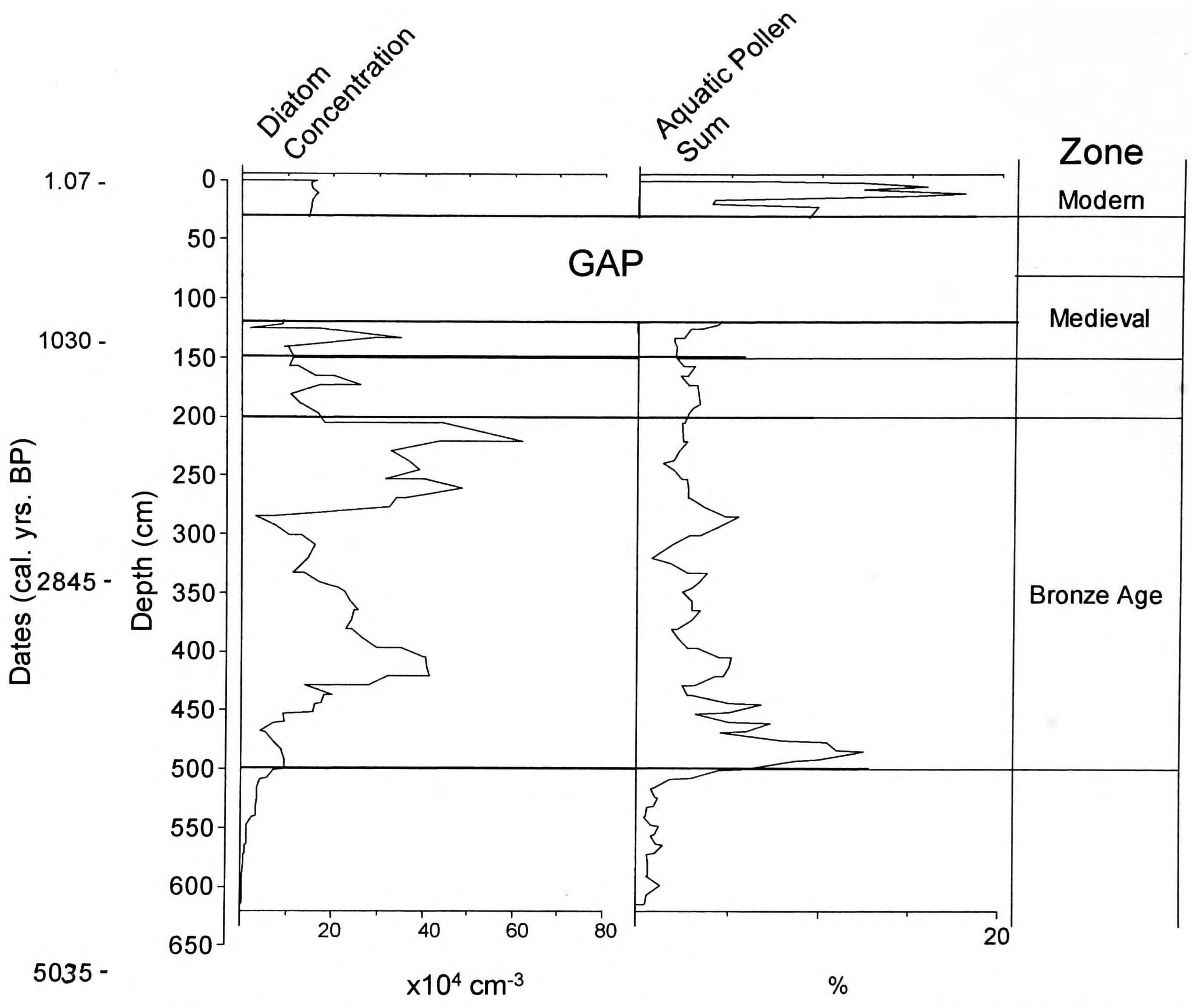


Figure 6.10 Diagram comparing diatom concentration (DC) with the aquatic pollen sum (APS) for the Llyn Pencarreg core (LPC/06)

185

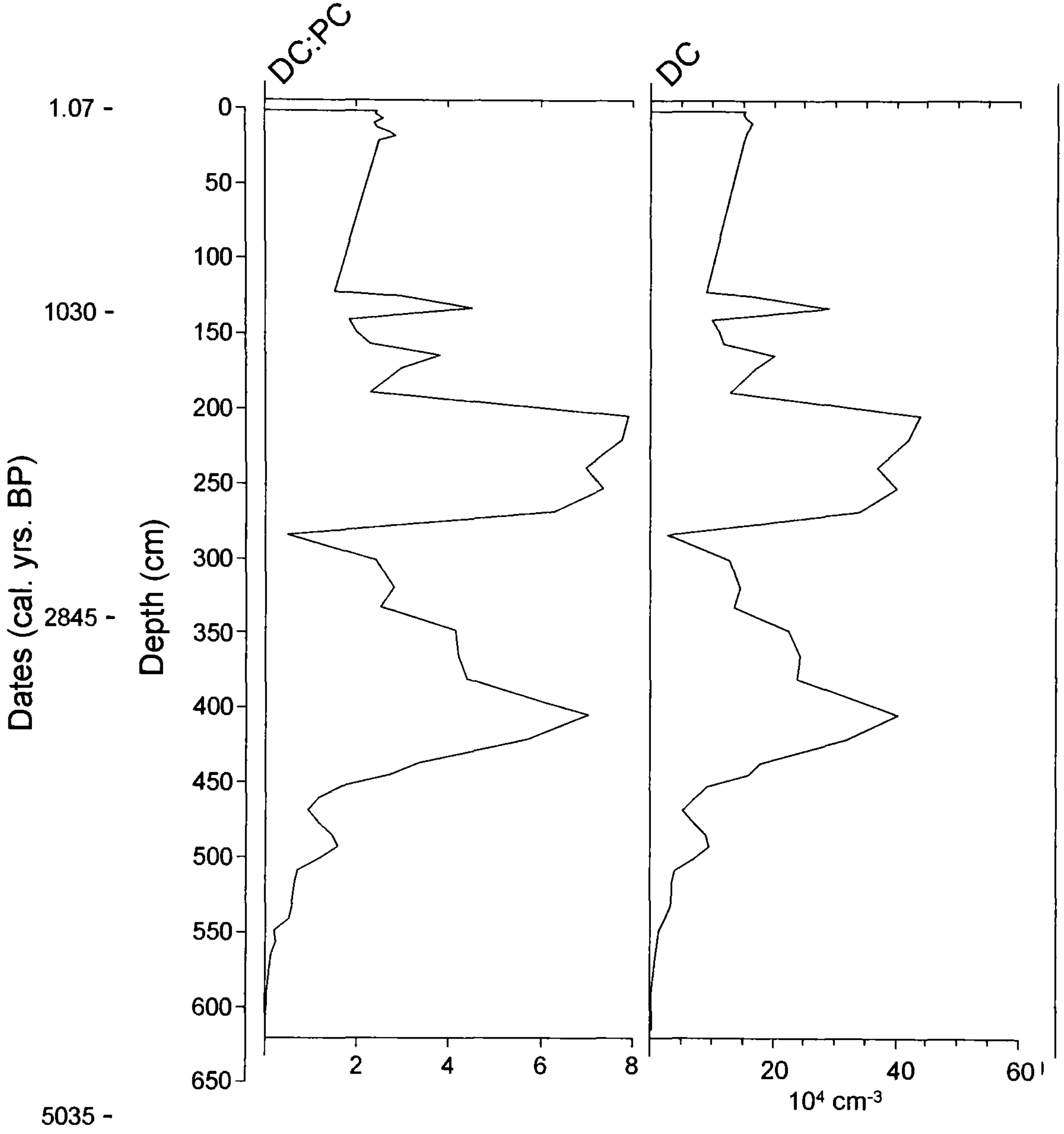


Figure 6.11 Diagram comparing the diatom to pollen concentration ratio (DC:PC) with the diatom concentration (DC) for the Llyn Pencarreg core (LPC/05).

6.15.3 Land-clearance and the Arable/Pastoral Index

Non-arboreal Pollen (NAP) and the Arable/Pastoral Index (API) are given in Figure 6.12. Between 6.80m and 5.50m (during the Neolithic period) API fluctuates considerably (max; 50% at 6.50m, min; 9% at 6.20m), suggesting variability in agricultural land-use, whilst a mean NAP value of *ca.* 1% indicates that farming activities took place in a predominantly forested catchment. A steady increase in NAP during the Bronze Age from 5% at 5.50m to 65% at 3.50m indicates a sustained increase in open ground. API shows that tree clearance was accompanied by a trend towards arable farming, which continues to a depth of 3.30m, where a value of 3% suggests predominantly arable land-use.

A decrease in NAP from 65% at 3.30 to 26% at 3.00m at the end of the Bronze Age suggests some forest regeneration, followed by relatively stable values throughout the Iron and Roman periods, and into the Post-Roman period to a depth of 1.55m (25%). API indicates that land-use remained predominantly arable, which continued into the medieval period to a depth. Medieval forest clearance, however, is indicated by the significant increase in NAP from 26% at 1.57m to 51% at 1.22m. Above the gap in core sediments NAP increases sharply from 32% at 22cm to a maximum value of 78% at the top of the core reflecting further increases in open ground. API displays an increase to 18% at the mid-zone followed by a decrease to 11% at the top of the core, though land-use remained predominantly arable.

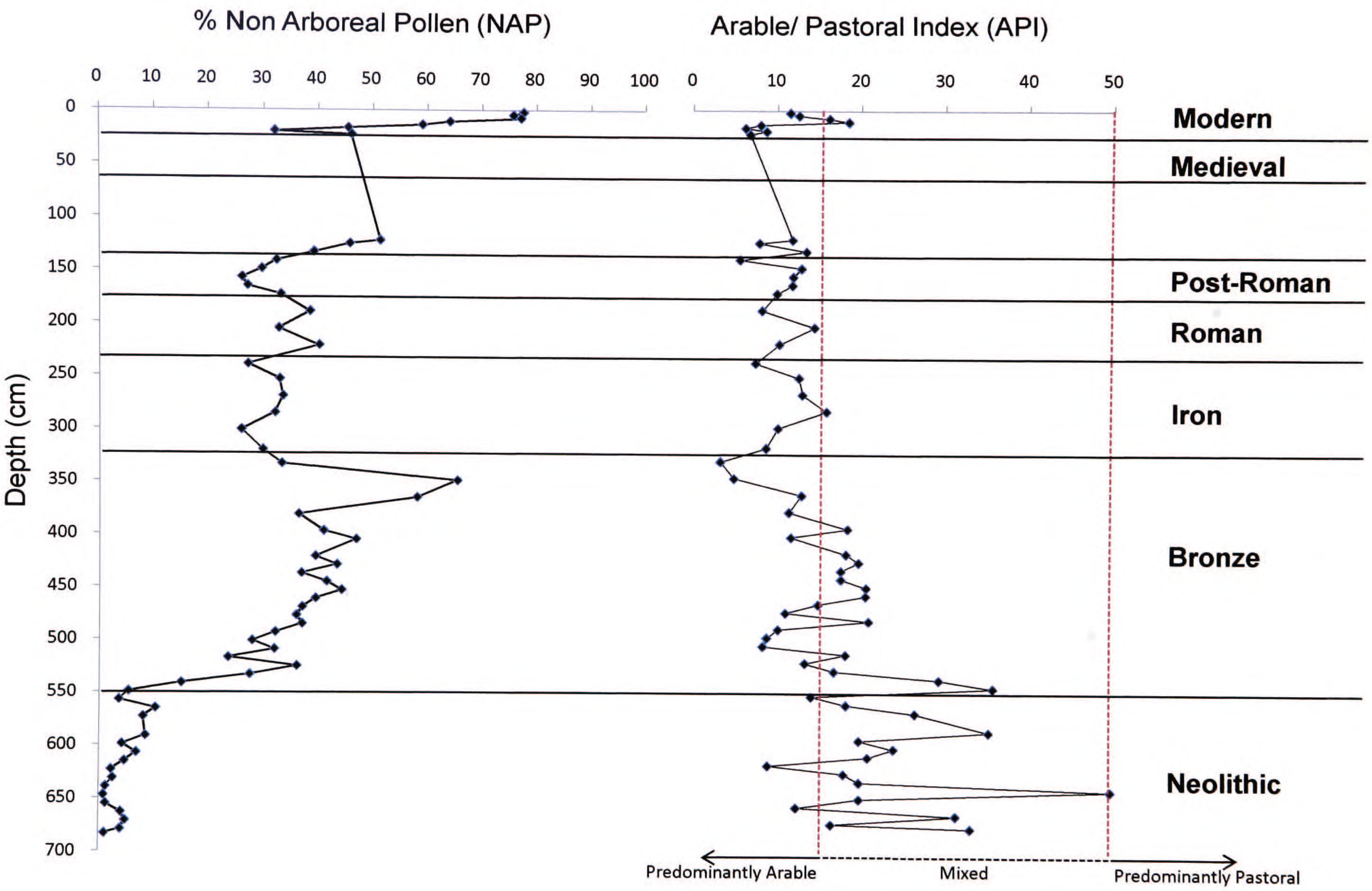


Figure 6.12 Diagram comparing the percentage change in a non-arboreal pollen (NAP) and the Arable/ Pastoral Index (API) for Llyn Pencarreg (LPC/05)

6.16 Charcoal Analysis

The results for charcoal analysis are presented along with selected pollen taxa to enable comparison with shifts in vegetation patterns (Figure 6.9). LPPZ-1, LPPZ-2 and LPPZ-3 have relatively low values with a mean of 0.23cm^2 cm^{-3}. In LPPZ-4 a gradual increase began, which continued into LPPZ-5 where a peak of 2.79cm^2 cm^{-3} at 1.72m was the maximum of the entire sequence. This was followed by a decrease towards the upper zone boundary. Finally, in LPPZ-6 charcoal abundance increase from 1.3cm^2 cm^{-3} at the zone base to 2.20cm^2 cm^{-3} at the top of the core.

6.17 Sediment Accumulation Rates

Approximate Sediment Accumulation Rates (SAR) are given in Figure 6.13 and summarised in Table 6.4. The observed changes indicate increases in SAR during phases of medieval and modern occupation, though the results are not definitive due to the poor chronological control.

Depth (m)	**SAR (yr cm $^{-1}$)**
6.81 – 6.57	6
6.49 – 1.45	3
1.37 – 0.9	8
0.1 - 0	1

Table 6.4 Table showing a summary of SAR for the Llyn Pencarreg Core (LPC/05)

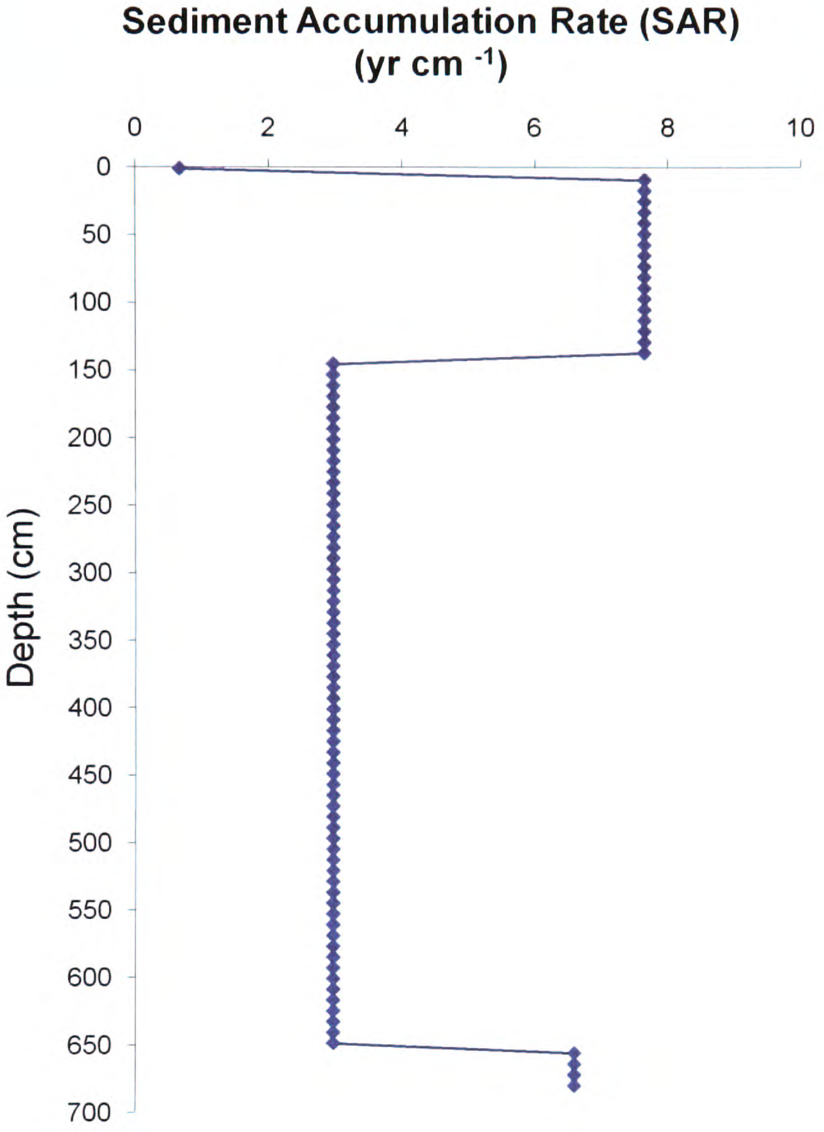


Figure 6.13 Diagram showing approximate sediment accumulation rates (SAR) for the Llyn Pencarreg core (LPC/05) based on linear interpolation between dates

6.18 Radiocarbon Dates and Chronology

The dating of lake sediments is notoriously difficult due to the combination of organic matter from a number of different sources that may include both 'old' and 'new' organic material, which can potentially contaminate the sample; realistically, a comparatively large number of radiocarbon dates are necessary to obtain a reliable sequence and reconstruct past sediment accumulation rates (Bennet, 1994). This was not possible in the present study and due to a lack of finance the age-depth model for Llyn Pencarreg was based on six dates of which two were considered unreliable. Table 6.5 shows that the radiocarbon age of a well-preserved piece of wood, taken from between 6.56m and 6.54m (SUERC-11315), was quoted as 2565 ± 35 cal. yrs. BP. The results of bulk sediment matrix AMS ^{14}C dates from SUERC (calibrated using OxCal13) are also given in Table 6.5 and as part of the age-depth model in Figure 6.14, which incorporates all the AMS dates and highlights the chronological problems associated with the Llyn Pencarreg core.

Date 1 (SUERC-13488), taken from a depth of 6.73cm, produced an age of 5035 ± 35 cal. yrs. BP for the sediments near the base of the core. At the very base of the core, *Ulmus* pollen percentages are slightly higher than those from the subsequent horizon. On this basis, it is tempting to suggest that the subsequent reduction, which occurs at the same depth as the AMS date, represents the '*Ulmus* decline', which first appears in British pollen records at *ca.* 5.2 – 5.0 cal. k yrs. BP (Williams, 2003). Significant reductions in *Ulmus* due to a combination of possible factors such as disease, deforestation and the provision of fodder for livestock, correspond with this date at several sites in Mid to South Wales, (e.g. Llangorse [70 km southeast], Jones *et al.*, 1985; Brecon Beacons [50 km east], Walker, 1982; Rhandirmwyn [40 km east], Chambers, 1982; Tregaron [22 km northeast] Turner, 1964). The subsequent increase in the relative abundance of *P. lanceolata* coupled with the absence of cereal pollen at Llyn Pencarreg is also consistent with patterns of changing vegetation during the Neolithic period (Caseldine, 1990). Although sparse, the presence of Neolithic archaeology in the form of several axe pieces does reflect human presence within a 4km radius of Llyn Pencarreg during this period. This date was therefore retained for the base of the sequence.

Date 2 (SUERC-11315) was the well-preserved piece of wood, which produced a date of 2565 ± 35 cal. yrs. BP for the sediments at 6.5m depth. Compared to the basal date (Table 6.5) this age would appear to be particularly young for this part of the sequence. Additionally, if this date is correct sediment accumulation rates for this part of the core would be exceptionally high at $137cm^2$ yr, which is not reflected in the low, stable magnetic susceptibility profile and, compared to other sites, is unrealistic for phases of human impact during the Neolithic period in a landscape with over 90% arboreal taxa (Figure 6.9). Considering that there was some resistance to the piston corer below *ca.* 5.0m, it is reasonable to assume that the wood may have originated from higher up in the sequence and was displaced by the corer during sediment retrieval. It is also possible that the weight of the wood caused it to sink to the deeper sediments over time, or additionally, the problems encountered with root penetration, which can lead to the production of younger dates, may also be a potential source of younger carbon (Kaland, 1984). For these reasons Date 2 was considered unreliable.

SUERC-13488 (Date 3) was taken from a major statigraphic horizon at 5.5 m, where significant changes in the diatom assemblage, a shift in DI-TP and lithological status, an increase in land clearance for arable farming accompanied by increased magnetic susceptibility values and a reduction in organic carbon, C/N and arboreal taxa indicate major changes in both the lake catchment and within the lake itself. This is followed by a significant increase in Poaceae and *P. lanceolata* and the appearance of cereal pollen, possibly reflecting the first signs of settled agriculture in the Llyn Pencarreg region. In pollen records from across Britain, arable farming first appears in the pollen record during the Bronze Age (Godwin, 1940).

However, whereas palynological evidence generally indicate an increase in cereal pollen and associated plants during the early Bronze Age (*ca.* 4.0 k cal. yrs. BP) in the south of Wales (e.g. Nant Helen [100 km southwest], Chambers *et al.*, 1988; Llanllwch [30km south], Thomas, 1965; Llangorse [70 km east], Chambers, 1995; Machynys [60km south], Lillie *et al*, 2000) and the mid- Bronze Age (from. 3.0 k cal. yrs. BP) in the north (e.g. Cefn Graeanog, Chambers, 1988; Llyn Cororion, Watkins 1990; Bryn y Castell, Mighall and Chambers, 1995) this date falls during the Iron Age. Considering that this is the first indication of arable farming at Llyn Pencarreg, an estimated age of 2585 ± 35 cal. yrs. BP would appear, given the evidence from elsewhere in Wales, to be too young for this part of the sequence. Whilst date 3 lies *ca.* 1m above the wood sample, and would appear to be from a securely stratified context, if this date were used to calculate sedimentation rates then an annual average sedimentation rate of *ca.* 0.2cm per year would be occurring, which would be unsupported in the current analysis due to the absence of an increase in LOI at this point in the sequence.

Although not as pronounced as the multiproxy evidence for change at 5.5m, Date 4 (SUERC-13487) marks a phase of limnological impact at a depth of 3.41m. The radiocarbon age of 2845 ± 35 cal. yrs. BP places these sediments during the late Bronze/ early Iron Age when changes in vegetation patterns were also related to land clearance for the mining of metal ores, the construction of fortifications, funerary monuments and settlement enclosures (Champion, 1999). The presence of both Bronze and Iron Age funerary monuments and Iron Age mottes around Llyn Pencarreg does indicate significant human activity in the region during this period. If this date is correct then an age of *ca.* 4.3 k cal. yrs. BP can be estimated for the major stratigraphic change at 5.5m, thus producing a date closer to those marked by major phases of Bronze Age activity and the introduction of arable farming in this part of Wales. Finally, in relation to date 3, it should also be borne in mind that the absolute dating of lake sediments is a complex process due to the myriad perturbations in sedimentation regimes, but in light of the available dates, there may be a possible chronological error in terms of absolute dating in the lower part of the Llyn Pencarreg sequences, below *ca.* 3.5m, of *ca.* >1000 radiocarbon years. As such, the lower part of the sequence should be treated with caution until further, higher resolution dating evidence is available for this location.

Dates 5 and 6 provide a chronology for the upper part of the Llyn Pencarreg sequence and place a significant phase of catchment impact during the early medieval period. Although it is difficult to validate these dates, if they are correct then the sedimentation rates for the upper part of the sequence would equate to a rate of *ca.* 0.1cm per year. This rate is commensurate with general sediment accumulation rates identified throughout lakes in Britain (e.g. Edwards and Whittington, 2001) and northwest Europe (e.g. Korhola *et al.*, 2000).

SUERC Code	**Depth(cm)**	**Estimated Radiocarbon Age (cal. yrs. BP)**
SUERC-13485 (GU-15021)	2 - 3	1.0761 ± 0.0045
SUERC-13486 (GU-15022)	136 - 138	1030 ± 35
SUERC-13487 (GU-15023)	340 - 342	2845 ± 35
SUERC-13488 (GU-15024)	544 - 546	2585 ± 35
SUERC-11315 (GU-14397)	656 - 654	2565 ± 35
SUERC-13489 (GU-15025)	672 - 674	5035 ± 35

Table 6.5 Table showing the results of bulk AMS ^{14}C dates for the Llyn Pencarreg core (LPC/05).

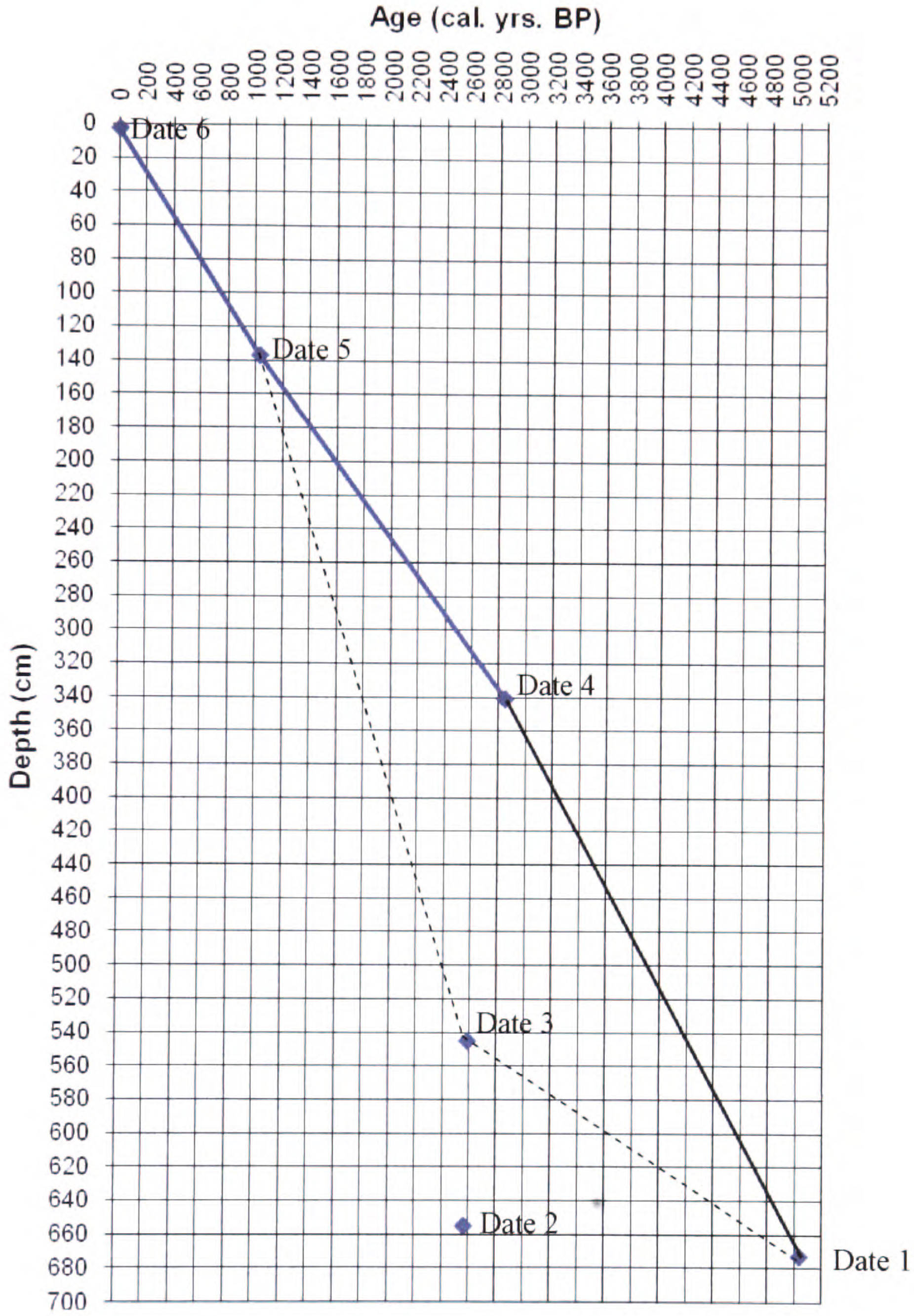


Figure 6.14 Diagram showing the age-depth curve for core LPC/05 based on bulk AMS dates (SUERC, 2006). Outlier dates (SUERC-13488 and SUERC 11315) are also shown.

CHAPTER 7

INTERPRETATION OF THE RESULTS FROM UPPER TALLEY

7.1 Introduction

To structure the interpretation of the results from each site, Chapters 7 and 8 are divided into three sections, 'Nutrient Status', 'Ecosystem Status' and 'Drivers of Limnological Change'. 'Nutrient Status' is interpreted mainly on the basis of diatom palaeoecology, supported by DI-TP. The $\delta^{13}C$ profile provides a potential estimate of changing rates of primary production and the C/N ratio is used to explore the source of organic matter and identify and separate changes in algal biomass from terrestrial in-wash. 'Ecosystem Status' is described on the basis of shifts in the diatom stratigraphy and diatom diversity (Hills N_2 Index). The appearance of *Pediastrum* taxa is taken to represent competition within the algal community and the presence of Cladocera ephippia indicates stress within the wider ecosystem. Changes in the relative abundance of aquatic macrophytes, reconstructed from changing percentages of pollen spores from aquatic plants, are also used to indicate ecosystem status. Drivers of Limnological Change' are identified from interpretation of the pollen data; inferred changes in catchment vegetation are compared to charcoal abundance, local archaeological data and evidence from documentary history to locate possible causes.

For the Interpretation of results from Upper Talley, a summary of key indicators is given in Figures 7.1a (limnology) and 7.1b (lithology) and provides strong evidence for ecosystem response to long-term limnological change. Zonation was undertaken visually in order to highlight correlations between significant shifts in the data and labelled UTMPZ-1 to UTMPZ-7 (i.e. Upper Talley Multi-Proxy Zones). Figure 7.2 outlines the potential limnological drivers and compares the predetermined zones with the anthropogenic and climatological influences discussed in Section 3.5. Finally, Figure 7.3 shows catchment archaeology at Upper Talley as potential evidence of direct anthropogenic impact.

7.2 Nutrient Status

UTMPZ-1: 9.90 – 9.40 m (*ca.* 9.9 – 8.8 k cal. yrs. BP)

During the Early Holocene Upper Talley was a relatively unproductive lake. Figure 7.1a shows that the diatom flora at the core base is dominated by oligotrophic and mesotrophic planktonic taxa (e.g. *Tabellaria flocculosa*; Figure 5.3) and a range of benthic taxa. DI-TP increased from mesotrophic (25.0 $\mu g\ l^{-1}$) at the core base, to eutrophic (50.0 $\mu g\ l^{-1}$) at the upper zone boundary, with an increase in relative abundance of eutrophic planktonic taxa (*Aulacoseira granulata* and *Cyclotella stelligera*) at the expense of mesotrophic planktonic and benthic taxa. The interpretation of relatively low productivity may be supported by low diatom concentrations and the subsequent increase in DI-TP is supported by a relative increase.

The isotope data cannot be interpreted simply in terms of productivity. Figure 7.1b shows that where C/N ratios increase from a basal value of 8.0 to 12.4 at the top of the zone, there is a significant decrease in $\delta^{13}C$ from –22.0‰ to –29 ‰. In respect to C/N

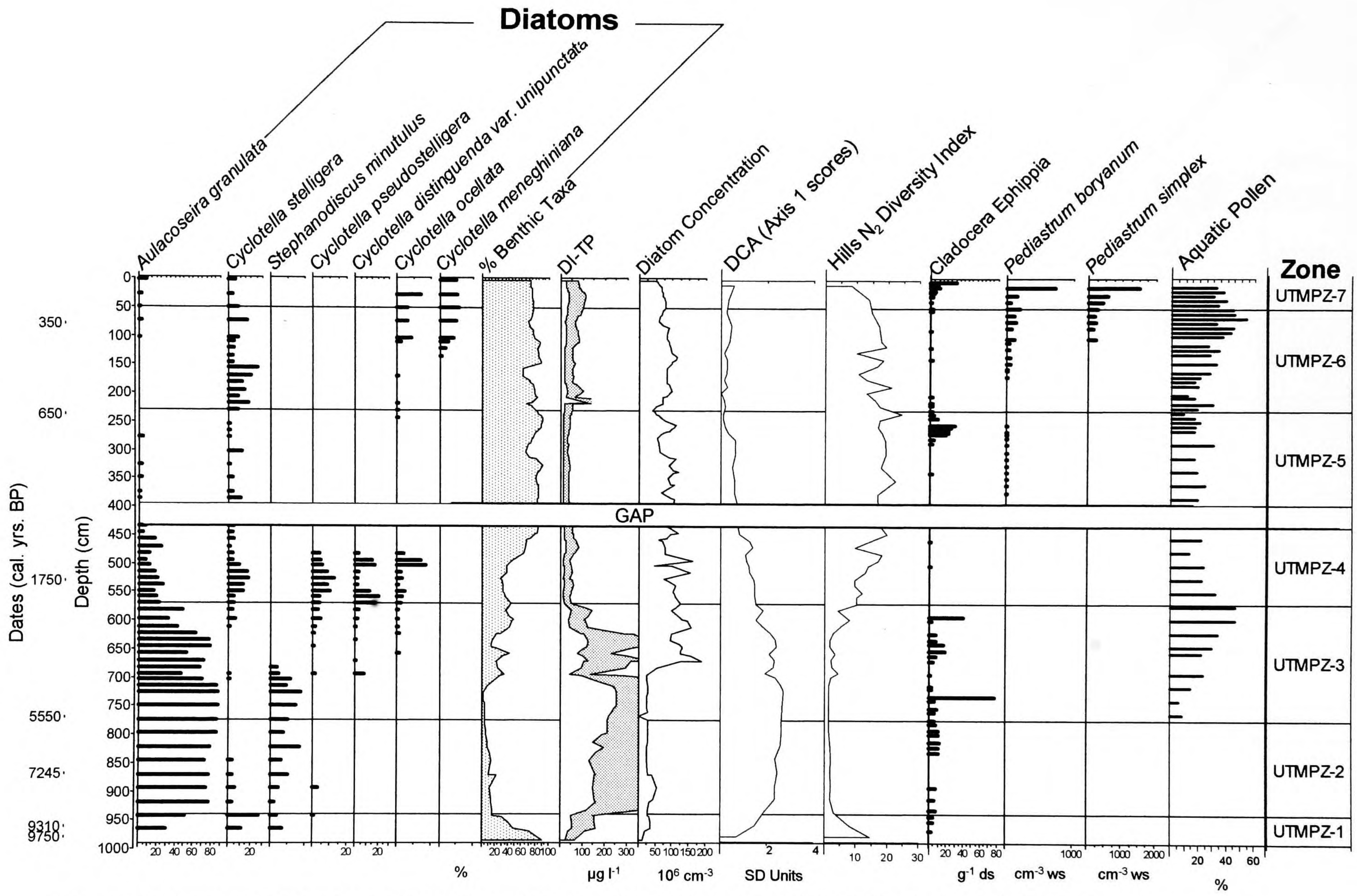


Figure 7.1a Summary diagram showing the multiproxy limnological results for core UT2/94 including dominant planktonic diatom taxa present at >4%, % benthic taxa, diatom-inferred total phosphorus (DI-TP), diatom concentration, DCA Axis 1 scores, Hill's N_2 Diversity Index, cladocera ephippia, *Pediastrum* spp. and aquatic pollen.

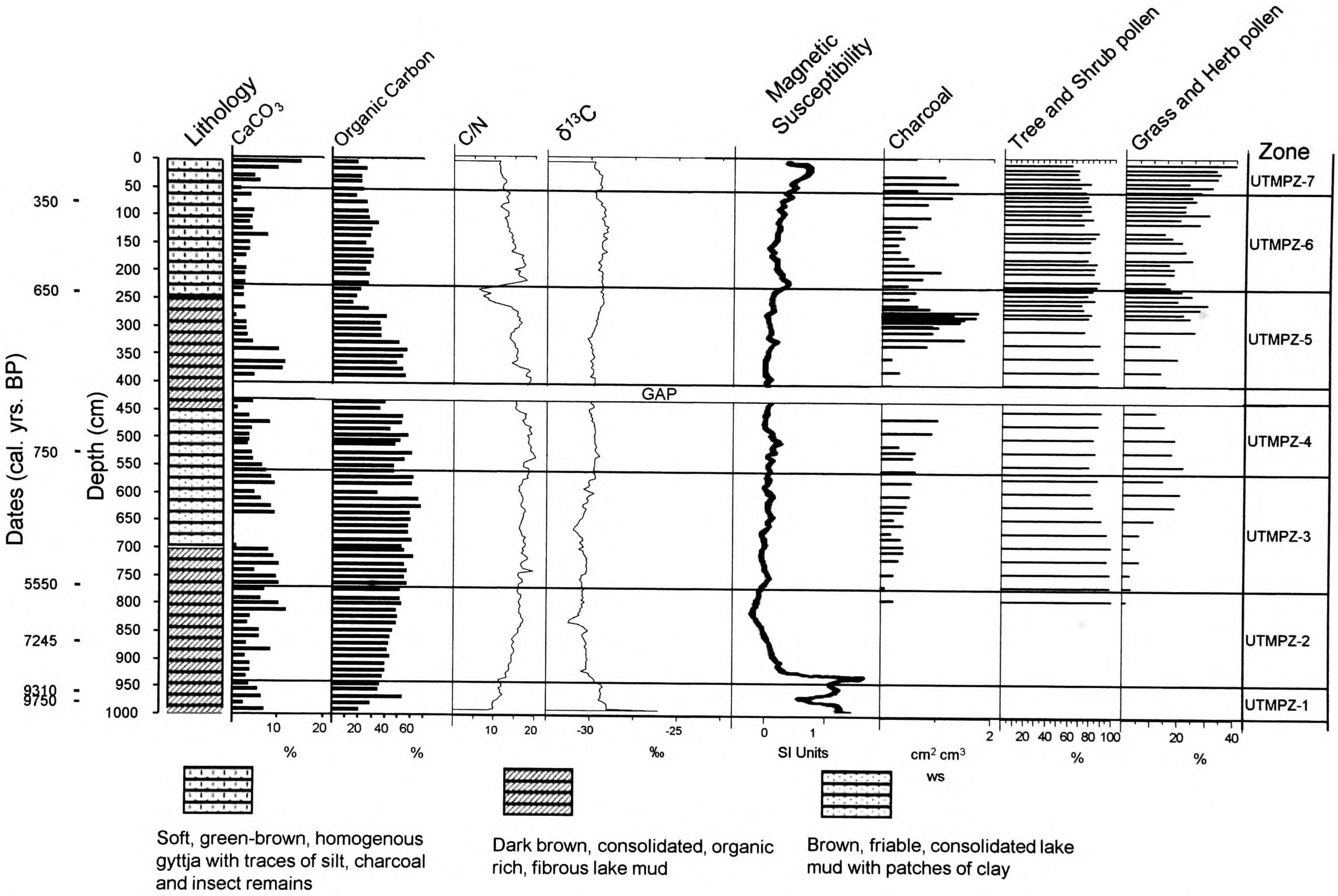


Figure 7.1b Summary diagram showing the multiproxy results for core UT2/94 including lithology, % CaCO$_3$ (calculated from LOI at 850°C), % organic carbon (LOI at 550°C), C/N ratios, δ^{13}C, magnetic susceptibility values (Johnson, 1998), charcoal abundance and selected pollen groups (adapted from Johnson, 1998).

analysis, which produces a mean ratio from the organic content within each sample, an underlying increase in algal material (C/N<10) may have been masked by a significant concomitant increase in terrestrial inwash (C/N>10), which is also reflected in an associated increase in organic carbon from 20% at the core base to 38% at the top of the zone. At the Florida Lakes, USA, Brenner *et al.* (1999) found that when terrestrial inwash increased, the signal for high algal biomass was concealed within a wide range of C/N values, thus preventing discrimination of plant type based on C/N. This interpretation of data from Upper Talley is supported by the analysis of modern algal and terrestrial plant material (Table 4.3), which also reflects a wide range of C/N values (between 6.9 and 22.1). Boutton (1991) argues further that the C/N signal can be complicated by the contribution from aquatic macrophytes, which also displayed a wide range of C/N values in this study, i.e. 6.6 to 37.7 for emergent species.

The apparent decrease in primary production ($\delta^{13}C$) is unlikely to be due to problems of progressive diagenesis of organic material, as changes due to microbial decomposition have been shown to be less than in the present study (Hodell and Schelske, 1998). The $\delta^{13}C$ results may have been affected by anoxic conditions, which can alter the relationship between algae and the uptake of dissolved carbon dioxide (CO_2). During nutrient enrichment, oxygen availability within the hypolimnion is low since algal blooms restrict oxygen exchange between the water and atmosphere, particularly at night when photosynthesis ceases (Mannion and Bowlby, 1992). When this process is coupled with the decay of algal blooms, anoxia can occur. Bacterial decomposition of organic matter, which releases light $^{12}CO_2$, may then decrease at the sediment-water interface, resulting in subsequent increases in the heavy isotopic composition of aqueous CO_2 and thus higher sedimentary $\delta^{13}C$ values (Wetzel, 1975).

UTMPZ-2: 9.40 – 7.75 m (8.8 – 5.5 k cal. yrs. BP)

During the Early to Mid Holocene the trophic status of Upper Talley moved into the hypereutrophic range. Figure 7.1a shows that this is marked by a rising trend in DI-TP to 125.0 μg l^{-1} at a depth of 9.25m (ca. 8.4 k cal. yrs. BP), which is driven by an associated shift in assemblage composition, with the progressive replacement of mesotrophic diatom taxa by those indicative of eutrophic to hypereutrophic waters (*Aulacoseira granulata* and *Stephanodiscus minutulus*). The low, stable diatom concentration may be a function of a turbidity-induced reduction in light penetration, which is not uncommon at the height of nutrient enrichment when only a few dominant species are present (Kamenir, *et al.,* 2004). Figure 5.9 indicates that aquatic pollen and spore percentages increased during this phase, suggesting that aquatic macrophytes coverage also increased. Figure 5.10 suggests that the low diatom concentrations may alternatively be associated with high rates of sediment deposition.

Figure 7.1b shows that in UTMPZ-2, the broad trend of increasing C/N and decreasing $\delta^{13}C$ values is the opposite of that expected during increasing eutrophication. The consistent increase in organic carbon and the presence of terrestrial plant remains in this zone provides strong support for the significant input of allochthonous material. As in UTMPZ-1, the lack of indication for algal biomass during this inferred phase of nutrient enrichment indicates that the C/N signal cannot always be interpreted simply in terms of algal or terrestrial input. Similarly, $\delta^{13}C$ values may have been affected by associated factors such as the inwash of soil humus, which has been shown to produce values similar to those representing low primary productivity (Jenny, 1980), and therefore may not be giving a clear signal of primary production within the lake.

UTMPZ-3: 7.75 - 5.70 m (*ca.* 5.5 – 2.3 k cal. yrs BP)

UTZ-3 in Figure 7.1a is marked by a final increase in DI-TP to 256 μg l^{-1} followed by a major decrease to 46.0 μg l^{-1} at 3.8 k cal. yrs. BP, implying that trophic status returned to eutrophic at *ca.* 4.0 k cal. yrs. BP. Unlike the stable values of UTZ- 2, DI-TP then fluctuated within the eutrophic range (90 μg l^{-1} - 120 μg l^{-1}). A sustained reduction in nutrient concentrations to 15μg l^{-1} at the upper zone boundary (*ca.* 2.3 k cal. yrs. BP) is driven by an increase in meso-eutrophic species such as *Cyclotella stelligera* and *C. pseudostelligera* (Battarbee *et al.,* 2005). The inference of lower productivity is also supported by the relative increase in benthic taxa.

The maximum sedimentary organic content for the sequence (65%) occurs at 6.22 m (Figure 7.1b). This does not correlate with high DI-TP and the data is again suggesting that the former is not simply related to algal productivity. The increase does, however, correlate with an increase in aquatic macrophyte pollen and spores suggesting that aquatic plant material, which can also display high $\delta^{13}C$ values, may have contributed greatly to the overall percentage of organic carbon. The increase in diatom concentrations in a zone of declining DI-TP is difficult to explain, but may reflect increased light penetration. An alternate explanation would be reduced sediment accumulation rates, which is indicated by the DC/PC ratio (Figure 5.10), but which is not supported by the radiocarbon chronology, which shows a major increase in sediment accumulation rate at some point after *ca.* 5000 cal. yr. BP (Figure 5.12). Both C/N and $\delta^{13}C$ values fluctuate slightly in this zone, with a subtle increasing trend in $\delta^{13}C$. These fluctuations may relate to the changing nutrient status inferred from

DI-TP, but given the inferred complexity of the signal in UTZ-1 and UTZ-2, interpretation is difficult.

UTMPZ-4: 5.70 - 4.34 m (*ca.* 2.3 - 1.2 k cal. yrs. BP)

Low, stable, DI-TP values (mean 16.0μg l^{-1}) occurred throughout the Late Holocene, driven mainly by a reduction in the relative abundance of *A. granulata* and maintained by a gradual increase in benthic taxa. However, Figure 7.1a shows that although the DI-TP profile is stable, there are major fluctuations in diatom species assemblage composition. These changes in the diatom community may be associated with changes in other environmental parameters, which are discussed below in terms of ecosystem stability and baseline states (Section 10.2).

UTMPZ-5: 3.95 – 2.30 m (*ca.* 1.1 – 0.6 k cal. yrs. BP)

UTZ- 5 is difficult to interpret in terms of nutrient status. Apart from a minor increase to 18 μg l^{-1} at *ca.* 0.7 k cal. yrs BP, a mean DI-TP value of *ca.* 16.0 μg l^{-1} was maintained during the eleventh to fourteenth centuries AD (Figure 7.1a). The low relative abundance of planktonic taxa supports an interpretation of mesotrophic status. Sedimentary organic content approaches the maximum for the sequence (50%) at the start of the zone, but then declines steadily to 16% at 0.7 k cal. yr. BP and is accompanied by a gradual transition to soft green-brown gyttja suggesting a decrease in allochthonous input. Whilst decreasing organic carbon in the sediment is often associated with low productivity, in shallow, eutrophic lakes the higher recycling rate of organic matter within the water column during increased algal respiration has been seen to produce a decrease in sedimentary organic carbon under conditions of high primary production (Meyers and Lallier-Vergés, 1999).

Although the measurements largely conflict with inferences of trophic status in this core, if interpretable in terms of algal biomass, the sharp decrease in C/N from 17.0 to 6.1 at the top of the zone (Figure 7.1b) may also support high algal productivity (Meyers and Teranes, 2001). It is also tempting to infer that the consistent trend towards higher $\delta^{13}C$ values from *ca.* 0.8 k cal. yrs. BP represents a reliable indication of increasing productivity, despite the problems of conflict encountered in the lower zones. It is possible that the lack of change in DI-TP is related to sedimentary P absorption, which can be tested by analysing the chemical composition of sediment matrix and has been found to produce low DI-TP measurements during phases of eutrophication (Brenner *et al.*, 1999).

UTMPZ-6: 2.30 – 0.50 m (*ca.* 0.6 - 0.2 k cal. yr. BP)

Figure 7.1a shows that during the fourteenth to eighteenth centuries AD a predominantly benthic diatom flora characteristic of meso- to eutrophic conditions continued to be present. At *ca.* 0.6 k cal. yrs. BP DI-TP values exhibited a short-lived increase to 33 $\mu g\ l^{-1}$, with a notable increase in the relative abundance of *Cocconeis placentula* and *Cymbella ventricosa* (Figure 5.3). Both taxa are known to replace mesotrophic species during phases of natural and artificial catchment fertilisation in a geographically broad range of countries including Turkey (Kilinç and Sivaci, 2001), Russia (Kravtsova *et al.*, 2006), Scotland (Eggilshaw and Shackley, 1971) and Wales

(Bennion, 1995; Bennion *et al.*, 1996a) and have high TP optima of 132 μg l^{-1} and 98 μg l^{-1} respectively.

If the interpretation of enrichment in the previous zone is correct, then the observed peak in DI-TP in this zone may represent a delayed response linked to the crossing of a sedimentary redox threshold where oxidation by a process of electron exchange with iron molecules has released sedimentary P into the overlying hypolimnion (Smol, 2002) (Figure 7.1a). This process is common during periods of anoxia in hypereutrophic lakes but can also result from lake-bed agitation, changes in water-pH, light and temperature (Engstrom and Wright, 1984; Wetzel, 2001) or changes in the growth pattern of aquatic macrophytes (Søndergaard *et al.*, 2003). The inferred time-lag between the height of nutrient enrichment (indicated by the C/N and $\delta^{13}C$ in Zone 5) and response peak in DI-TP at Upper Talley, however, is *ca.* 80 years, which exceeds the maximum period of 20 years observed by Welch and Cooke (2005).

DI-TP decreased to mesotrophic status thereafter, fluctuating around a mean of 19.0 μg l^{-1} and an increase to 28 μg l^{-1}, at *ca.* 0.4 cal. yrs. BP suggests a return to eutrophic status. These latter changes were marked by the introduction of planktonic taxa characteristic of nutrient-rich conditions, comprising *Cyclotella ocellata* and the first appearance of the highly pollution-tolerant *Cyclotella meneghiniana* (Hecky and Kilham, 1973; Izaguirre *et al.*, 1993; Romo and Miracle, 2004). DI-TP and $\delta^{13}C$ values remain high, supported by a corresponding steady decrease in C/N (Figure 7.1b) suggesting that sedimentary algal biomass increased as DI-TP and productivity increased.

UTMPZ-7: 0.5 – 0 m (0.25 - 0 cal. yrs. BP)

Over the last 250 years of the depositional sequence the record is dominated by diatoms indicative of eutrophic conditions. Figure 7.1a shows that eutrophication is reflected in the increase in DI-TP to 38.0 μg l^{-1} at *ca.* 1.0 cal. yrs. BP (*ca.* 1949 AD). Although a lower value of 26.4 μg l^{-1} was produced at 2.5 cm, spring measurements of TP are presently in the range of 100.0 μg l^{-1} suggesting that P loads have increased by at least 70.0 μg l^{-1} since the 17th century. It also suggests DI-TP estimates may be too low for a sample dominated by small *Fragilaria* taxa (discussed in Section 9.2.1). The decrease in sedimentary organic content to 20%, (Figure 7.1b) can be interpreted as the consequence of increased recycling of organic material in the water column associated with increased primary productivity (Meyers and Lallier-Verges, 1999). Decreasing C/N values can then be interpreted in terms of a dominance of phytoplankton production over terrestrial inwash. This interpretation, however, is not supported by the $\delta^{13}C$ profile, which fluctuates around a mean value of –29.0 ‰ with an overall trend towards slightly lower values suggesting an overall gradual decrease in primary production.

7.3 Ecosystem status

UTMPZ-1: 9.90 – 9.40 m (*ca.* 9.9 – 8.8 k cal. yrs. BP)

In the Early Holocene, significant trends in many of the proxy indicators at Upper Talley suggest relatively rapid lake ontogeny. At the core base, the co-dominance of a range of benthic taxa (Figure 7.1a) indicate that the lake was shallow and circumneutral. Benthic taxa were dominated by small *Fragilaria* spp., which are common opportunistic, pioneering species during the expansion of the aquatic environment (Rawlence, 1988), although examination of a longer core would be necessary to confirm this. Their broad habitat preferences and tolerance ranges makes palaeoenvironmental interpretation difficult. Hickman and Reasoner (1998), for example, relate their presence to the warmer temperate climate of the Early Holocene lakes, whilst Karst and Smol (2000) associate their presence to the colder conditions surrounding Arctic lakes.

A common point of these studies, however, is that these species occur in shallow (<6m) water, which enables them to photosynthesise on the lakebed. Under these conditions the growth of aquatic macrophytes is encouraged and in a study of a shallow, eutrophic lake, Sayer, *et al.* (1999) found the distribution of small *Fragilaria* taxa to be epiphytic, growing on and around the stems of *Phragmites*. This implies that some *Fragilaria* have the potential to increase in abundance with corresponding increases in the macrophyte community. Although the pollen record for Upper Talley is absent before *ca.* 5.5 k cal. yrs. BP, the presence of small *Fragilaria* and a range of other benthic taxa suggest that this may have occurred at Upper Talley. Clear water is supported by the inference of mesotrophic status, whilst the dominance of benthic over planktonic taxa suggests low diatom productivity in the open waters and that the lake was probably fairly shallow (Likens, 1985).

While not definitive, the absence of *Pediastrum* colonies in this zone (Figure 7.1a), which have been found to increase during phases of increased primary production in lakes throughout Europe (e.g. Bottema, 1974; Nielsen and Sørensen; 1992 Komárek and Jankovská, 2001) and disappear at the height of enrichment in favour of toxic algae (e.g., *Aphanizomenom flos-aquae*) (Eilers *et al.*, 2004), also supports low productivity. The low abundance of ephippia suggests that Cladocera were mainly reproducing parthenogenically and their presence could relate to low winter temperatures (Langdon Brookes, 1959). At Aitajärvi Lake in northern Finnish Lapland, Sarmaja-Korjorne (1999) found that gamogenesis was probably an effective survival tool against the extreme cold of the immediate Post-glacial period. Although temperatures at Upper Talley were unlikely to be as extreme as northern Finland, where seasonal ice coverage extended well into the Holocene, their presence may reflect small seasonal bursts of gamogenesis during the cold winter months (Langdon Brooks, 1959).

A subsequent marked trend towards increasing diatom planktonic dominance driven by *C. stelligera*, *A. granulata* and *S. minutulus* (Figure 7.1a) correlates with major increases in the percentage of organic carbon and C/N values, and a decrease in $\delta^{13}C$ (Figure 7.1b); however complex the latter indicators are to interpret, this does provide evidence for major limnological change. The abundance of *C. stelligera* is consistent with increasing climatic warming and productivity change in the Early Holocene (Platt *et al.*, 2004) and the influence of more recent warming trends in the Northern Hemisphere (Harris *et al.*, 2006). In this zone of increasing productivity Hills Index (N_2) indicates that diatom diversity was high at the core base (*ca.* 14.5) and decreased inversely as enrichment increased towards the upper zone boundary.

UTMPZ-2: 9.40 – 7.0 m (8.80 – 4.3 k cal. yrs. BP)

UTZ-2 exhibits remarkable long-term ecological stability. Figure 7.1a indicates that diatom assemblages are dominated by *A. granulata,* which is indicative of turbid conditions and low water transparency. *A. granulata* has been observed to increase considerably as a result of vertical mixing during the colder winter months at lakes in Brazil (da Silva *et al.*, 2005) and Israel (Zohary, 2004) and can form algal blooms which out-compete other species, thereby preventing a more diverse flora from developing (Kamenir *et al.*, 2004). Kilham (1986) and Kilham *et al.* (1996) suggest that this is enabled by its heavy silica frustules, which help keep it suspended and allow photosynthesis under conditions of low light availability. *S. minutulus*, present at a lower relative abundance, has also been found to grow successfully under conditions of low light availability to the extent of forming seasonal blooms (Marchetto *et al.*, 2004), particularly during spring turnover when waters become turbid due to mixing of the water column (Bradbury, 1986). In general, the low diatom diversity of this zone (Hills N_2 = 1.8) is a direct function of a hypereutrophic state and turbid conditions (Hynynen *et al.*, 2004).

At *ca.* 4.7 k cal. yrs. BP the peak in Cladoceran ephippia (Figure 7.1a), corresponding with the height of enrichment, might be significant to the diagnosis of ecosystem health, nutrient enrichment and associated turbidity. Slusarczyk (2001) found that ephippial production can be associated with ecosystem pressure caused by fish predation and that gamogenesis is more likely to occur in shallow lakes where the effectiveness of vertical migration and escape from predators is low. At Bølling Sø, Denmark, Bennike *et al.* (2004) found that changes in ephippia abundance were related to the development of predator-prey relationships, which had a direct influence on food supplies. Pennack (1989) also found that gamogenesis can occur in response to decreases in the amount of food available to female Cladocera. It has also been demonstrated that the heavily silicified frustules of the dominant diatom species, *A. granulata*, may render it inedible to Cladocera (Kilham *et al.*, 1976).

Low abundance of aquatic plants would also be expected in a hypereutrophic lake of this type and although a record of aquatic pollen and spores is missing until *ca.* 5.5 k cal. yrs. BP, the signal above this is initially low and dominated by Cyperaceae (which are typically found at wet margins rather than growing in any significant depth of water) (Figure 5.9). A lack of aquatic macrophytes is supported to some extent by the lack of plant macrofossils in the sediments from this zone (Table 4.2).

UTMPZ-3: 7.75 - 5.70 m (*ca.* 5.40 – 2.30 k cal. yrs BP)

The disappearance of *S. minutulus* from the diatom flora at *ca.* 4.0 k cal. yrs. BP correlates with an increase in the diversity of both planktonic and benthic taxa, and marked rises in diatom concentrations and in the abundance of aquatic pollen and spores (Figure 7.1a). The continued dominance by *A. granulata* to the mid-zone and the appearance of *F. pinnata, F. exigua, F. virescens* and *F. vaucheriae* (Figure 5.3) is consistent with a shallow, enriched lake environment (Kawecka and Olech, 1973). All of these factors are consistent with an increase in water clarity (Karst and Smol, 2000). However, trophic status remained high and some species present (e.g. *A. granulata*, ephippia) suggest that conditions at Upper Talley were still nutrient-rich, turbid and suffering from ecosystem stress.

Figure 5.10 shows that diatom concentration may have been influenced by factors other than sediment accumulation, namely, an increase in P and in silica (Si), which was inferred by Johnson (1998) from sediment chemistry analysis. The potential influence of P and Si on diatom concentrations is also demonstrated by Schelske *et al.* (1986). These chemical changes may have been encouraged by the long phase of turbidity in the previous zone, which distributes dissolved nutrients throughout the water column (Wetzel, 2001). Van Donk and Kilham (1990) found that *A. formosa* and species of *Fragilaria* are particularly good competitors for these elements.

The mid-zone (*ca.* 3.5 k cal.yrs. BP) marks the start of long-term dynamic in diatom species assemblage composition, which culminates in UTMP-4. This is paralleled by an increase in diatom diversity (Hills N_2 increased to 10.1 at the top of the zone), the disappearance of Cladoceran ephippia, the first appearance of *Pediastrum* colonies and a rise in aquatic pollen and spore percentages (Figure 7.1a). *A. distans* (Figure 5.3) is recorded in the diatom assemblage, a species that has an optimum TP of 182mg l^{-1} in southern England (Bennion, 1994; 1995), 289 $\mu g\ l^{-1}$ in the Cheshire, Shropshire and East Midland Meres (Bennion *et al.*, 1996b) and 145 $\mu g\ l^{-1}$ at lakes in Wales (Allott and Monteith, 1999), but which has also been observed in mesotrophic waters following eutrophication at lakes in France (St. Jacques, 2005). Likewise, species of *Cyclotella* have been seen to appear as the water column begins to stabilise following phases of turbidity, particularly *C. stelligera* (Marchetto *et al.*, 2004; Salmaso, 1996). Their presence at Upper Talley therefore supports the inferred decrease in nutrient availability and turbidity (indicated by a concurrent decrease in *A. granulata*) at the top of this zone.

Figure 7.1a also shows that the increase in *Cyperaceae* and *Potamogeton* pollen from the mid-zone supports the decrease in nutrient concentrations (Haslam *et al.*, 1975). More importantly, the abundance of *Cyperaceae* can be interpreted as the maturation of a fringing zone of emergent taxa in at least part of the littoral zone. This may have had a significant influence on wind-induced turbidity, and may explain in part the above transition to a lower relative abundance in *A. granulata*. *Potamogeton* seeds were also recorded (Table 3.2), providing strong evidence for macrophyte growth within a few meters of the core site (Zhao *et al.*, 2006). Although the presence of green algal *P. boryanum* colonies is often used as evidence for increased primary productivity, and they have been observed to form algal mats during phases of eutrophication (Janokovská and Komárek, 2000), at Upper Talley their first occurrence in this zone may indicate competition within the phytoplankton community as enrichment and the associated turbid conditions decreased.

UTMPZ-4: 5.70 - 4.34 m (*ca.* 2.3 - 1.2 k cal. yrs. BP)

The trends initiated in Zone 3 of Figure 7.1a culminated in a reduction to mesotrophic status in the Mid to Late Holocene, with the replacement of *A. granulata* dominance by a diverse diatom flora reflecting increased water clarity. Diatom species continue to represent a wide trophic range, whilst diatom diversity continued to increase (Hill's N_2; 18.1). Brenner *et al.* (1999a) found that at Orange Lake in Florida, USA, a similar switch in assemblage composition occurred as turbulent conditions subsided following the height of enrichment and the heavier *Aulacoseira* settled out of the water column allowing for the expansion of a range of planktonic taxa.

Increased phytoplankton diversity is also supported by the presence of *P. boryanum*, which is consistently present at low abundance. High diversity, which represents good ecosystem health and protects the community against dramatic change (McArthur, 1955; Smol, 1992; Moss *et al.*, 1997), may have been supported at Upper Talley by clear, mesotrophic waters and the high availability of habitats that accompany these conditions (Moss *et al.*, 1997). The low ephippia abundance suggests that Cladocera were reproducing by parthenogenesis and reflects a lack of stress within the wider ecosystem. The algal communities were relatively dynamic, suggesting that a system that appears to be stable is not necessarily static.

A transition in the upper quartile of UTMPZ-4 (after *ca.* 1.3 k cal. yrs. BP), characterised mainly by the loss of *Cyclotella* species, led to a decrease in the planktonic:benthic ratio in the upper zone. This could reflect either a decrease in lake depth or increase in light penetration with reduced turbidity (Scheffer, 2004). Figure 5.9 shows that the stability of the *Alnus glutinosa* pollen curve does not suggest infilling and an associated reduction of the limnic zone, and the concurrent reduction of *A. granulata* is certainly consistent with reduced turbidity, so the latter is the most likely explanation.

UTMPZ-5: 3.95 – 2.30 m (*ca.* 1.1 – 0.6 k cal. yrs. BP)

Following the gap in the sediment record, UTZ-5 of Figure 7.1a is characterised by the expansion of a range of meso to eutrophic, benthic diatom taxa, many of which were present at lower relative abundance in UTZ-4. *P. boryanum* was also present from the zone base at a higher frequency than in the previous zone, suggesting high phytoplankton diversity and thus relatively good ecosystem health.

Although $\delta^{13}C$ clearly increases, indicative of an increase in rates of primary production, changes in the diatom flora were subdued, perhaps indicating that a marked shift in diatom assemblage composition does not always occur during nutrient enrichment. Instead the increase in primary production and sedimentary algal biomass was matched by an increase in the relative abundance of diatom taxa. The shift between diatom concentration and aquatic macrophyte dominance, inferred in Figure 5.10 at *ca.* 0.6 k cal. yrs. BP, suggests that a shift in alternate states may have occurred in response to these changes. The decline in *P. boryanum* abundance at 0.6 k cal. yrs. BP and subsequent peak in Cladoceran ephippia immediately prior to the inferred height of enrichment may again reflect stress due to changes within the wider food chain and a loss of stability between trophic groups (Bennike, *et al.*, 2004).

UTZ-6: 2.30 – 0.50 m (*ca.* 0.6 - 0.2 k cal. yr. BP)

On the basis of reductions in the abundance of a range of eutrophic taxa (*A. distans, G. parvulum, E. bilunaris, E. pectinalis, A. minutissima, A. subatomoides* and *A. vitrea*), Figure 7.1a indicates that by *ca.* 0.6 k cal. yrs. BP, a degree of recovery is indicated by the diatom community. *C. stelligera*, which increased from the zone base, dominated the zone and peaked at *ca.* 0.50 k. cal. yrs. BP, accompanied by an increase in *C. placentula.* Both species are commonly found in productive waters (Huttunen and Meriläinen, 1983), although at Upper Talley the presence of *C. stelligera* appears to correlate with evidence for the increase and decrease in nutrient status before and after enrichment-related high-turbidity episodes, suggesting that it is a positive indicator of the threshold between eutrophic and hypereutrophic conditions. At Laguna Chica of San Pedro in Chile, Urrutia *et al.* (2000) found a similar pattern where *C. stelligera* increased in abundance as waters became less eutrophic following enrichment.

F. pinnata (Figure 5.3) also increased from the base of this zone, suggesting that this diatom was responding to enhanced enrichment. Similar responses of *F. pinnata* have been found at Lake St Luis (Reavie *et al.,* 1998), Canada, Llyn Clyd and Llyn Glas, Snowdonia, North Wales (Evans and Walker, 1977), and Crowfoot Lake, Alberta (Hickman and Reasoner, 1998) and may be related to the growth of certain aquatic macrophytes. This is consistent with evidence for a marked rise in the proportion of pollen and spores from aquatic taxa such as Cyperaceae, *Potamogeton, Isoetes, Sparganium, Nuphar lutea* and *Litorella uniflora* in mid-zone (Figure 5.9), indicative of a rich macrophyte flora both within and around the lake.

In addition to the return of *P. boryanum*, this zone marks the first appearance of *P. simplex*, at *ca.* 0.4 k cal. yrs. BP. The latter appears to be a positive indicator of both productive conditions and warm temperatures and has recently been described from many lakes throughout Europe (Komárek and Janovská, 2001). Goulden (1970), for example, reported it to be present at Lago di Monterosi during its recent history as climate became warmer and the lake became more eutrophic. However, *P. simplex* tends to occur mainly at the onset of enrichment and disappears as waters become increasingly turbid (Komárek and Janovská, 2001). From *ca.* 0.5 k cal. yrs. BP, an ongoing change is indicated by the first appearance of pollution-tolerant *Cyclotella meneghiniana* in particular. Cladoceran ephippia also appeared in low abundance at the same depth indicating that lack of stability was causing stress in the wider ecosystem.

UTMPZ-7: 0.5 – 0 m (*ca.* 0.2 – 0.0 k cal. yrs. BP)

Shifts in the proxy data over the past 200 years correlate well; in combination with the continued presence of 'new' taxa, there is strong evidence not only for recent eutrophication, but also for a major shift in ecosystem status (Figure 7.1a). Increased planktonic dominance suggests a reduction in water clarity. Under these conditions a switch can occur from plant- to algal-dominance. At Upper Talley a reduction in aquatic pollen and spore abundance at the lower zone boundary suggests that the balance between aquatic macrophytes and algae shifted in favour of the algal community. In contrast to Lower Talley, which supports a rich and diverse macrophyte flora, the modern lake of Upper Talley is now almost devoid of aquatic vegetation.

Figure 7.1a also shows that the decrease in relative abundance of aquatic pollen and spores correlates with an unprecedented increase *P. boryanum* and *P. simplex* colonies whose occurrence in a wide range of eutrophic waters during the Late Holocene must highlight their ability to adapt to the increasing eutrophication of water bodies throughout Europe (Alhonen and Ristiluoma, 1973; Bottema, 1975), America (Smith, 1950; Prescott. 1962; Jansen, 1968), Switzerland (Komárek and Jankovská, 2001), the Middle East (van Zeist and Bottema, 1977) and Russia (Jankovská and Panova, 1998). At Lake Karlsø, Denmark, Henning (2002) found that an increase in *Pediastrum* was particularly associated with high nutrient loads from sewage effluent and may therefore be linked with heavy cultural pollution.

Cladoceran ephippia also increased significantly at the top of the core, indicating pressure on the Cladocera population, probably in response to ecosystem stress occurring during the eutrophication process. The factors directly responsible are difficult to define. Additionally, this is a zone of decreasing diatom diversity (Hill's N2; from 17.07 at the zone base to 8.1 at the top of the core) in accordance with the inferred increase in enrichment towards the top of the core.

7.4 Drivers of Limnological Change

UTMPZ-1: 9.90 – 9.40 m (*ca.* 9.9 – 8.8 k cal. yrs. BP)

In UTMPZ-1 of Figure 7.1b, the transition to increased productivity and terrestrial inwash at the zone base is accompanied by high magnetic susceptibility values (Johnson, 1998) suggesting that the inferred catchment inwash initially contained a high mineral component. This is supported to some degree by the $CaCO_3$ profile, but this decreased towards the upper zone boundary. The increase in C/N values and the presence of brown, diatomaceous sediments containing terrestrial macrofossils such as leaves and twigs support the inference of increased terrestrial inwash. These sediments include a well-preserved hazelnut (*Corylus avellana*), which provided an AMS date of 9,310±45 cal. yrs. BP. This, and a second AMS date of 9,330 cal. yrs. BP (Johnson, 1998), indicated that sedimentation at this point of the core occurred during the early Post-glacial.

Although temperatures were still relatively low following the Younger Dryas, particularly during the winter months, Figure 7.2 indicates that this date falls during the Boreal; a time of rapid warming when catchment vegetation expanded, organic based allochthonous inwash was high (Walker, 1981; Rehakova, 1983: Hickman and Reasoner, 1998, Pennington, 1999; Birks *et al.*, 2000; Grölund and Kauppila, 2002). Sedimentary organic deposition containing a combination of plant debris and humic acid from developing catchments was particularly common in lakes in the northern hemisphere from *ca.*10.0 k cal. yrs. BP (Lowe and Walker, 1999) and has been observed in lake sequences at Scotland (Connoly and Dickson, 1969; Selby, 2004), England (Brown, 1977; Prartono and Wolff, 1998) and Wales (Ince, 1983; Watkins, 1990).

Pollen data are not available for this zone but other pollen studies (discussed in Section 3.4) demonstrate climatically driven patterns of vegetation change in South Wales, where Late Glacial instability is followed by the establishment of catchment vegetation and the stabilisation of catchment soils. At the upland site of Traeth Mawr (Walker, 1981) and at Llangorse Lake (Walker *et al.*, 1993), situated 35 km and 40 km to the east of Upper Talley respectively, a closed *Betula* canopy replaced open ground dwarf shrub and heathland communities with Juniper scrub at *ca.* 9.80 k cal. yrs. BP; *Alnus* became more widespread on the valley floor and, by 9.50 k cal. yrs. BP, *Corylus avellana*, *Quercus* and *Pinus* had expanded on the drier soils to form mixed deciduous woodland.

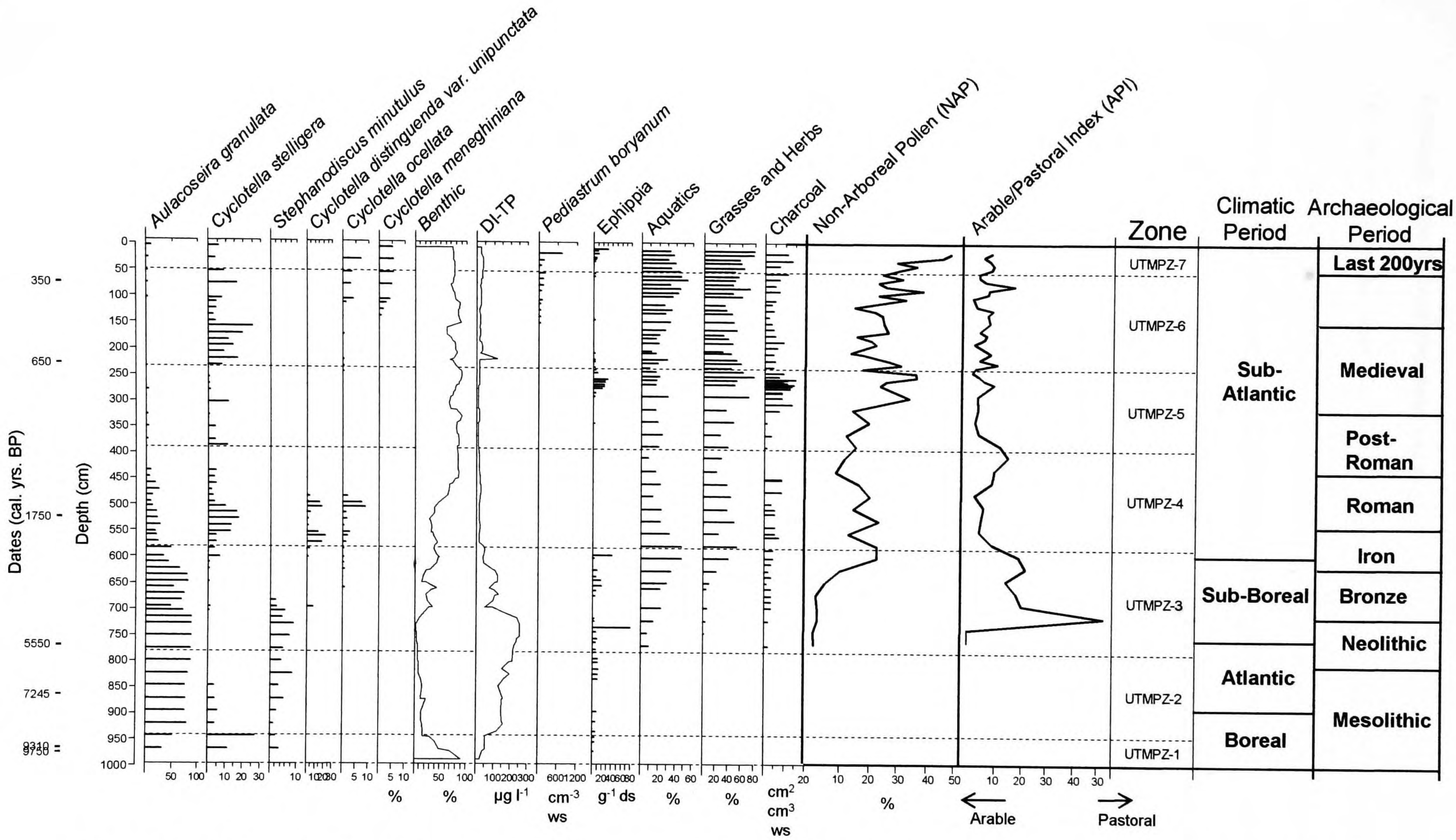


Figure 7.2 Diagram showing the factors potentially responsible for limnological change at Upper Talley including % benthic taxa, diatom-inferred total phosphorus (DI-TP), relative abundance of trees, shrubs, aquatic, grass and herb pollen, % organic carbon (LOI at 550°C), % $CaCO_3$ (calculated from LOI at 850°C), Magnetic susceptibility, C/N ratios, non-arboreal pollen (NAP) and the Arable/Pastoral Index (API)

Although there is overwhelming evidence for allochthonous organic material in the lake sediments at this point, it is also likely that internal productivity contributed to the accumulation of organic matter at Upper Talley. The warmer temperatures of the Post-glacial gave rise to a major phase of aquatic ecosystem development in temperate lakes of the northern hemisphere (Pickering, 2001), which Whiteside (1981) has described as lake ontogeny rather than eutrophication, since ontogeny suggests directional changes in water chemistry and community structure as opposed to a trend of increasing enrichment. At Kråkenes in Norway (Birks *et al.*, 2000), for example, ecosystem development led to the accumulation of sedimentary organic matter as primary production increased, chironomid, coleopteran and Cladoceran communities developed and aquatic and terrestrial vegetation expanded. Likewise, at Mirror Lake, New Hampshire, USA (Likens, 1985) concentrations of organic matter, chlorophyll *a*, diatoms, chrysophyte cysts and bosminids increased at a constant rate due to temperature-induced increases in productivity.

The inferred shallow status of Upper Talley (indicated by the high ratio of benthic taxa) may also be attributed to the Post Glacial, Boreal climate when regardless of the general increase in temperature and atmospheric moisture, precipitation was still low and inland lakes, which are sensitive to climatically induced changes in the local water budget, were still relatively shallow following the dryer Younger Dryas (Guiter *et al.*, 2005).

UTMPZ-2: 9.40 – 7.75 m (8.80 – 5.50 k cal. yrs. BP)

As at other sites in South Wales (Walker, 1981; 1993; Chambers, 1999), the reduction in magnetic susceptibility (Figure 7.1b) indicates that catchment soils at Upper Talley had stabilised by *ca.* 8.5 k cal. yrs. BP. This was a common pattern during the Boreal period as climatic improvement encouraged the development of mature woodland dominated by *Ulmus, Corylus avellana, Betula* and *Quercus*. However, while catchment vegetation and sedimentary organic carbon stabilised from *ca.* 9.0 k cal. yrs. BP at Traeth Mawr (Walker, 1981) and Llangorse (Walker *et al.*, 1993) giving rise to what Chambers (1999) describes as stable, baseline conditions, at Upper Talley the accumulation of organic matter continued.

This was accompanied by a sustained increase in C/N values (Figure 7.1b), a dark brown sediment matrix with visible terrestrial plant material and turbid, nutrient-rich conditions. Although the deposition of autochthonous organic material may have contributed to the organic record during the lake development process, evidence from both Birks *et al.* (2000) and Likens (1985) suggests that aquatic development and high deposition of internal organic matter only continued for *ca.* 2.0 k years following the Late Glacial and is therefore unlikely to contribute to the significantly long-term accumulation of sediments rich in organic carbon at Upper Talley. However, it is clear that the accumulation of organic carbon is directly associated with lake turbidity and the progressive deposition of organic material. Such long-term turbidity is rare and, although not analogous to the present study, Lakes Hora and Tilo (crater lakes in tropical Ethiopia on the western margin of the Rift Valley) were the only other early post-glacial comparisons which could be found in the literature (Telford and Lamb, 1999). Long-term turbidity contributed to lake development as the lake basin filled with hydrothermal groundwater and solute terrestrial runoff from a catchment composed mainly of grassland and woody scrub.

The sustained increase in organic carbon and the low benthic ratio may indicate that Upper Talley was extremely shallow (i.e. >1.5m), causing high nutrient concentrations and the resuspension of fine, unconsolidated sediments, which in turn can encourage the production of diatom taxa with superior light harvesting properties (e.g. *A. granulata*) (Reynolds *et al.*, 1994). However, meteorological changes at *ca.* 7.5 k cal. yrs. BP are argued to have brought warm onshore Atlantic winds and increases in precipitation to South Wales (Briffa and Atkinson, 1997; Walker *et al.*, 1993), which in turn caused a general increase in lake levels throughout northwestern Europe (Guiter *et al.*, 2995). It is possible that the high rainfall of this period caused an increase in catchment inwash (indicated by C/N values) and subsequent in-lake turbidity at Upper Talley, though this would be unusual during a period of maximum forest cover, which is more likely to prevent soil erosion (Schlosser and Karr, 1981). The possibility that the lake was sheltered by dense forest at the time also means that wind fetch across the lake is also unlikely to be the cause (van Eerden and Voslamber, 1995 suggest that wind induced resuspension requires wind speeds of up to 8km hr).

It is also possible that lake conditions were linked to human activity within the Upper Talley region. Figure 7.3 shows that there is a Neolithic chambered tomb at Cwm

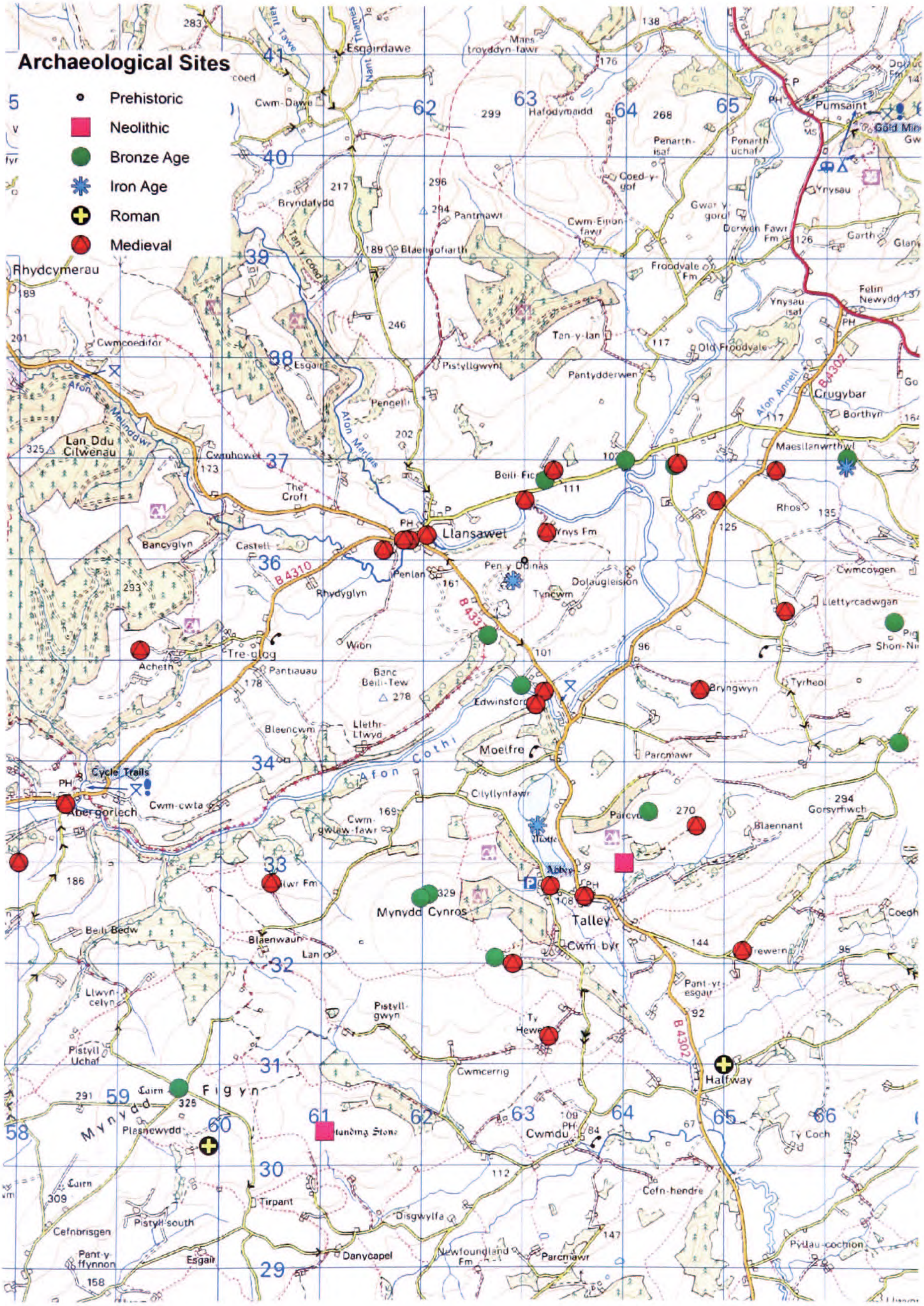


Figure 7.3 Diagram showing the presence of archaeology within and immediately around the Upper Talley lake catchment

Gwyddil (3.0 km southwest of Talley) and a stone circles at Llyn Taliaris (5.0 km directly south of Talley) and at Cefn Blaidd (0.4 km east of Talley and within the lake catchment). As discussed in Section 3.5.3, the labour intensive construction of chambered tombs and stone circles suggests a sizeable farming community with the free time necessary to build such structures that reflect ideas of land-ownership and territorial boundaries (Whittle, 1999). Although palaeoenvironmental evidence for human occupation in Wales during the Atlantic period indicates a pattern of seasonal occupation coupled with semi-permanent agriculture in small scale forest clearings (e.g. Chambers and Price, 1988), the pollen record is not available for this part of the Upper Talley core.

Although regional pollen studies often lack the spatial resolution necessary to identify the presence of temporary forest clearings for agricultural activity (Day, 1993), it is unlikely that deforestation was the cause of eutrophication to this extent; even clearance episodes regarded as large-scale during the Neolithic were small and temporary in comparison to later slash and burn events (Brown, 1999; 2002). Despite the cause of long-term turbidity being hard to identify, low diatom diversity and the presence of *double-yolked* ephippia from *ca.* 6.8 k cal. yrs. BP to the top of the zone indicate that Upper Talley was showing strong signs of ecosystem stress.

UTMPZ-3: 7.75 - 5.70 m (*ca.* 5.40 – 2.30 k cal. yrs BP)

Following the above phase of natural long-term enrichment, significant decreases in *Alnus glutinosa, Quercus* and *Corylus avellana* pollen percentages, a significant increase in Poaceae from 2% at the zone base to 40% at the top, and the introduction of *Hordeum*-type, *Rumex acetosa,* Brassicaceae and *Plantago* spp. (Figure 5.9), indicate a period of anthropogenic deforestation and agricultural intensification in the catchment from *ca.* 3.6 cal. yrs. BP. Increased human activity is also implied by the increase in charcoal towards the upper zone boundary, whilst the sustained increase in non-arboreal pollen (NAP) and decrease in the Arable/Pastoral Index (API) (Figure 7.2) suggest that changes in land-use were significant and predominantly arable. Figure 7.2 shows that the evidence for catchment activity occurred during the Sub-Boreal when mild climatic conditions encouraged an increase in agricultural activity at many sites in Wales (Table 3.2).

Figure 7.2 also shows that these inferred changes in land-use at Upper Talley took place during the Late Neolithic-Bronze Age and corresponds with Bronze Age archaeology on the higher ground overlooking the lake as indicated in Figure 7.3. Although this matches the general establishment of Bronze Age settlements in Wales from *ca.* 4 k cal. yrs. BP, it does not always provide direct evidence of settlement within the catchment itself; funerary practices during this period often took place away from the main settlement area (Parker Pearson, 1999). However, the barrows were constructed from stone suggesting that local soils were valued for agricultural purposes, which was typical of the warmer early Bronze Age; the presence of Bronze Age burial archaeology on the hills immediately west (Mynedd Cynros and Allt Hafod-Wen) and east (Cefn Blaidd) of Upper Talley, and significant changes in the pollen record provide strong evidence for Bronze Age occupation and a regime of settled agriculture within the Talley region. The NAP (Figure 7.2) suggests that land-clearance at Upper Talley increased during the Iron Age, though the only definitive evidence of Iron Age occupation are hillforts situated at least 2km north of the site (e.g. Pen Y Ddinas; Figure 7.3)

During this period at Upper Talley, *Potamogeton* and Cyperaceae pollen percentages increased from the zone base indicating the development of a fringing reed zone, which may have been encouraged by the generally warmer Sub-Boreal. temperatures The increase in macrophytes also suggests that the lake was shallow during this period, possibly causing expansion of the littoral zone (Rasmussen and Anderson, 2005), though Hannon and Gaillard (1997) suggest that it is unwise to make this assumption based on a single, central core. However, in northern Europe Gaillard (1984), Gaillard and Diggerfeldt (1991) and Borgmark (2005) report a climate driven decrease in lake levels at *ca.* 2845 cal. yrs. BP that caused the expansion of littoral macrophytes in shallow lakes.

Although the relationship between aquatic plants and nutrient levels is complex and dependent on factors such as lake depth, the magnitude and nature of nutrient loading, water residence times and the type of plants present (Melzer, 1999), the abundance of aquatic macrophytes also increases in response to an initial increase in nutrient loads (Brenner *et al.*, 1999). Regardless of the cause however, an increase in littoral plants also controls lake water nutrient levels (the importance of a healthy littoral zone in the definition of the baseline state is discussed in Section 1.9.2) and the expansion of aquatic plants at Upper Talley may have contributed to the increase in water clarity (indicated by the increase in benthic taxa) and reduction in nutrient loads during a period of increased human impact.

UTMPZ-4: 5.70 - 4.34 m (*ca.* 2.3 - 1.2 k cal. yrs. BP)

The limnological stability demonstrated in UTZ-4 of Figure 7.1a appears to correlate well with evidence for limited catchment impact. The pollen record is mainly characterised by steady values of *Alnus glutinosa*, whilst herb and grassland taxa are stable (Figure 5.9), indicating settled agriculture with constant cultivation (Bradshaw, 2001). There is a significant reduction in *Betula* percentages though the decrease in NAP (Figure 7.2) suggests that on the whole land-clearance was relatively minor compared to UTMPZ-3. The relatively consistent API values indicate that, despite the general increase in pastoral farming in Wales throughout pre-history (Caseldine, 1990), land-use was predominantly arable around the Talley lakes. The decrease in charcoal particles in this zone may also be associated with a decrease in human activity around the lake (Burden *et al.*, 1986; Tsukada *et al.*, 1986).

Some diatoms in this zone (e.g. *E. pectinalis*) are typical of shallow, acidic, oligotrophic waters (deNicola, 2000) (Figure 7.1a). Their presence could be a response to a lowering of lake water pH from decayed organic matter and leaf litter, which can follow deforestation (Håkansson and Regnéll, 1993), or the previously discussed spread of acid heathland, which became well established in South Wales following sustained Bronze Age land clearance and the prolonged farming of upland soils (Chambers, 1988). The presence of acidophilic diatom taxa is accompanied by an increase in *Calluna vulgaris* pollen percentages, a plant which has a preference for acidic podsolic soils and grows in areas where there is a lack of competition from taller plants (Tansley, 1939).

The bottom of UTMPZ- 4 marks the beginning of the Sub-Atlantic period (Figure 7.2) when climatic conditions became cool with increased rain and humidity and lake levels were generally higher than during the Sub-Boreal (Issar, 2003). At Upper Talley there is evidence of a lake deepening event during this period (between 2.3 k to 1.7 k cal. yrs BP). Littoral macrophyte pollen decreases in relative abundance suggesting retraction of the littoral zone, planktonic diatoms increase in abundance indicating deepening of the pelagic zone, while the decrease in inferred DI-TP may suggest nutrient dilution. In limnological terms the evidence suggests that a sudden deepening event at Upper Talley may have put a stop to long-term nutrient enrichment at Upper Talley and encouraged the expansion of planktonic *Cyclotella*. The sustained increase in benthic taxa and the ^{14}C inversion at 5.25 m indicate that rapid sediment infilling followed and that the lake became shallower from 1.6 k cal. yrs. BP. This may be a reflection of the dryer climate recorded at *ca.* 1.7 k cal. yrs. BP when lake levels in the Northern Hemisphere began to decrease (Rasmussen and Anderson, 2005).

This zone also crosses the Iron Age-Roman transition (Figure 7.2) and apart from the reduction in *Betula*, the palaeoenvironmental evidence on the whole indicates low catchment impact. The preference that *Betula* has for damp woodland (Archibold, 1995) suggests that its decrease may be associated with the building of a Motte on the wooded area between the lakes, rather than with land-clearance for agricultural purposes. Nevertheless, its construction and subsequent use appears to have caused minimal environmental and limnological impact. Although a Roman Fort was discovered in the eastern part of Dinefwr Park, Llandeilo in 2003 (12km north of Talley), there is no recorded archaeological evidence for Roman occupation within the

Upper Talley catchment; the nearest Roman find is a collection of glass beads from Cynwyl Gaeo, 2.5 km southeast of Talley (Figure 7.3). Cunliffe (2005) explains that in west and central Wales between the Rivers Teifi and Tywi extending down to Cardigan Bay there is a distinct lack of settlement archaeology with which to mark the Roman period. This may of course be due to inadequate survey and/or identification, though casual finds are also extremely rare. However, it is possible that the lack of Roman archaeology in the area could be explained to some extent by the continued Ordovician (the ruling tribe of the area) resistance to Roman occupation (Cunliffe, 2005). The intrinsic link between water bodies and the land directly around them thus indicates that this clear phase of catchment stability and evident lack of human impact may well explain the inference of relative limnological stability.

UTMPZ-5: 3.95 – 2.30 m (*ca.* 1.1 – 0.7 k cal. yrs. BP)

UTZ-5 of Figure 7.1a is characterised by a phase of inferred eutrophication, which peaked *ca.* 0.7 k cal. yrs BP (1300 AD), correlating well with evidence for anthropogenic impact in the catchment. Major limnological changes in diatom abundance, $\delta^{13}C$, organic carbon, C/N, *Pediastrum* and ephippia correspond with a gradual increase in mineral inwash towards the top of the zone (Johnson, 1998) (Figure 7.1b), suggesting unstable catchment soils. Shifts in the pollen record highlight human activity, with reductions in *Fraxinus excelsior, Betula*, *Quercus* and *Corylus avellana* percentages indicating forest disturbance on the elevated soils, and a decrease in *A. glutinosa* reflecting deforestation on the damper soils of the valley floor (Figure 5.9). Although the API (Figure 7.2) reflects continued arable land-use, the significant increase in NAP from the base of UTMPZ-5 (*ca.* 1.1 k cal. yrs. BP), characterised by the presence and increase in pollen from agricultural weed taxa (e.g., *Plantago* spp., *Rumex* spp. and *Ononis*; Figure 5.9), suggests changes in farming activity (Johnson, 1998). The presence of *Rhamnus* and *Viburnum* pollen is consistent with an increase in light penetration to the forest floor but also with the introduction of hedgerows for field division. The significant increase in charcoal concentration from *ca.* 0.9 k cal. yrs. BP could therefore indicate human habitation and local domestic activity on a larger scale than previously (Burden *et al.* 1986; Tsukada *et al.* 1986).

A date of 0.65 k cal. yrs. BP, correlated by magnetic susceptibility with the Lower Talley (Johnson, 1998), indicates that these landscape changes occurred during the Medieval period. This was a phase of widespread forest clearance, with the exploitation of richer catchment soils. Recent studies have started to demonstrate evidence for associated impacts on nutrient status in lakes across Northern Europe (Bradshaw, 2001; Gaillard *et al.*, 1991; Håkansson and Regnéll, 1993; Chambers, 1999) and it is believed that the observed increases in agricultural activity were driven by a population increase and made possible by advances in agricultural technology (Chavas and Bromley, 2005). On the one hand these advances broadened the amount and types of crops that could be grown, whilst on the other, they caused an increase in sediment accumulation rates to nearby lakes (Bradshaw, 2001; Jones *et al.*, 1978).

There is therefore indisputable evidence in the pollen record for major changes in human activity within the Talley catchment during the medieval period. Evidence of human occupation within the wider landscape can also be observed in the archaeological record (Figure 7.3); medieval excavations throughout the entire region provide the first evidence of permanent settlements. These include a grange at Cefn

Blaidd (to the west and just outside of the lake catchment), a longhouse at Lletty Leucu (3 km northeast), a mansion 1.0 km to the north in Edwinsford and settlements at Bryngwyn-Fawr (1.5 km northeast), Cilwr (3 km west), Cwmbleidd (3km northeast), Maes-y-Feundre (3km to the north), Trefywern (2 km northeast) and Ynystwilth (2.5 km north). Their presence is typical of the late thirteenth century when populations were forced, by climatic deterioration, to the lower foothills, and small nucleated settlements and farmsteads developed where agriculture became extremely important to local economies (Shofield, 1999).

Within the lake catchment itself, Talley Abbey, along with its chapel and cemetery (Figure 7.3), was constructed on the southwest bank of the Upper Talley Lake, approximately 50 m from the present lake-shore (Section 3.7.1.2). The Abbey administered 7167 acres of parish land; 200 acres of woodland and 3900 acres of arable with the remaining land providing rough pasture (Lewis, 1844). The high acreage of monastic arable land is reflected in the proportion of cultivated land suggested by changes in the pollen record (Figure 5.9). The Abbey lies on a watershed between two streams; one flowing north towards the River Cothi and one running off Mynydd Cynros to the south-west of Upper Talley and towards the River Tywi near Llandeilo and it has been suggested that along with the Upper and Lower Talley lakes these streams were likely to have provided the monks and surrounding settlements with a water supply (Robinson and Platt, 1998).

UTMPZ-6: 2.30 – 0.50 m (*ca.* 0.70 - 0.25 k cal. yr. BP)

The phase of catchment stability at the base of this zone (marked by a significant decrease in NAP) correlates with an increase in C/N, indicating reduced nutrient enrichment, and *C. stelligera*, suggesting improvements in water quality. Although there are chronological uncertainties, the climatic cooling, which followed the Medieval Warm Period and began with three years of torrential rain in *ca.* 1315 AD, changed settlement patterns as communities moved to the warmer lowland areas where crop practices were altered to adapt to the shorter, less reliable growing season (Jacoby and D'Arrigo, 1989). It is also tempting to infer that this phase was a function of abandonment linked firstly to the Black Death in 1350 AD, which had a profound effect on the running of monasteries and their influence on the Welsh countryside, and to the subsequent demise of Talley Abbey *ca.* 200 years later in AD1536 as King Henry VIII dissolved the smaller non-profitable abbeys (Robinson and Platt, 1998).

The gradual return to eutrophic status from *ca.* 0.4 k cal.yrs. BP (indicated by shifts in the diatom record, the increase in *P. boryanum* and aquatic plants) suggests that Talley was not abandoned completely; changes may be related to the growth of the village of Talyllychau, which developed from the agglomeration of cottages associated with the Abbey in direct response to the late-16th century population increase (Owen, 1893a; 1893b; R. Price, Dyfedd Archaeological Society, *pers. comm.*, 2005). Charcoal increased substantially from *ca.* 0.4 cal. yrs. BP, which could support the suggested increase in human habitation (Figure 7.3). The API (Figure 7.2) indicates that cereal cultivation remained constant, but the increase in NAP proportions, characterised by

Poaceae (Figure 5.9), reflect a phase of sustained deforestation and an increase in the amount of available meadow. This pattern was common in south Wales from the sixteenth century to provide pasture for sheep farming, which expanded considerably following the Black Death (Caseldine, 1990).

UTMPZ-7: 0.5 – 0 m (0.2 - 0 cal. yrs. BP)

The unprecedented increase in *Pediastrum* colonies, increases in DI-TP and Cladocera ephippia concentration, decrease in C/N and the first appearance of the pollution-tolerant diatom, *Cyclotella meneghiniana*, all provide strong evidence for the effects of accelerated anthropogenic impact in the last *ca.* 200 years. Sediment accumulation rates are also high (Figure 5.13), possibly reflecting increased deposition from both internal production and catchment erosion (Jones *et al.*, 1985). The steady decrease in the pollen and spore signal from submerged aquatics (e.g. *Potamogeton*, *Isoetes*) over (Figure 5.9) is consistent with a trend towards increasing algal domination and has culminated in a lake devoid of aquatic macrophytes, in contrast to its sister lake, Lower Talley. The low abundance of littoral plants has also left soils around the immediate lake edge bare and unstable, probably offering little protection from erosion and in-wash (Moss *et al.*, 1997).

Figure 5.9 does not contain palynological evidence that these changes are driven by an increase in cereal cultivation. Significant reductions in *Betula* percentages were accompanied by an increase in the *Poaceae* pollen signal and the types of herbs associated with grazed land (e.g. *Plantago lanceolata*). Since the early nineteenth century, a substantial portion of common land has been used for grazing sheep around the Talley Lakes (Lewis, 1833). This continues to date with intense grazing on the grassy banks that run directly to the water's edge. This type of sheep management is known to saturate soils with excessive nutrients, which arrive at the lake *via* run-off (Sharpley *et al.*, 1995). However, the API (Figure 7.2) also indicates continued arable land-use, which can also produce nutrient-rich runoff. Either process would be exacerbated by the lack of littoral macrophytes, which can protect the lake from excess run-off. The increase in micro-charcoal abundance in Figure 7.1b supports the idea of a population expansion around the lake and may reflect the modern increase in the use of fossil fuels (Wick *et al.*, 1986; Schmidt *et al.*, 1999). The cumulative effect of increased nutrient loads over the last 200 years, on a limnological background already primed for enrichment during the medieval period, appears therefore to have resulted in the more recent cultural eutrophication of Upper Talley.

7.5 Summary

Nutrient status and ecosystem dynamics at Upper Talley have fluctuated significantly over the last *ca.* 10 k cal. years in response to natural changes, changes in climate and human impact. During its development the lake became hypereutrophic characterised by a phase of long-term turbidity and a low diversity, planktonic flora (*A. granulata* and *S. minutulus*). Although increased productivity is usual during the first 2.0 k years of lake development (Likens, 1985; Birks *et al.*, 2000), enrichment at Upper Talley continued until *ca.* 4.4 k cal. yrs. BP. It is possible that the lake was very shallow at the time causing sediment resuspension, though the reason for such long-term enrichment is hard to evaluate. During this period the lake was receiving a high proportion of terrestrial inwash, which appears to have masked any possible signal for increased primary production.

As conditions began to improve a second increase in trophic status (DI-TP) and enrichment related turbidity corresponded with significant changes in land use during the Late Neolithic-Bronze Age and the presence of Bronze Age archaeology on the hilltops around the edges of the lake catchment. As light attenuation was probably high during this period the appearance of new benthic taxa may have been associated with the corresponding increase in aquatic plants and spread of the littoral zone. However, these changes may also be the result of the changes in climate during this period. Throughout the entire period ecosystem stress was indicated by the presence and increase in the abundance of Cladocera ephippia, probably associated with the loss of habitat and shifts in predator-prey relationships.

Following Bronze Age impact catchment conditions were relatively stable and the lake quickly became mesotrophic. Diversity began to increase and underlying shifts in the diatom record coupled with an increase in *P. boryanum* reflected changes in ecosystem dynamics probably associated with the gradual increase in water clarity following long-term eutrophication. The loss of planktonic centric taxa at *ca.* 1.3 k cal yrs BP may have been part of the recovery process as lake conditions improved, although a lake deepening event may also be indicated. During the following Medieval period limnological impact was reflected in the organic and isotope records but rather than displaying a corresponding shift in assemblage composition, the diatoms reflected an increase in the relative abundance of all taxa. The loss of *P. boryanum* corresponded with an increase in *A. distans* and ephippia indicating changes in ecosystem dynamics and stress within the wider foodweb. Although a gradual decreasing trend in diatom diversity corresponded with major deforestation in the lake catchment the lake remained mesotrophic, possibly due to sedimentary P retention, whilst a delayed peak in DI-TP at *ca.* 0.6 k cal. yrs. BP may indicate the later release of P into the hypolimnion. During this phase of increased land-use and limnological impact the wide presence of medieval settlement archaeology indicates a strong human presence in the Talley area, particularly within the lake catchment.

Although conditions appeared to revert to the pre-enriched state following medieval impact, rates of primary production remained high and were accompanied by a gradual increase in DI-TP suggesting that the lake may have been primed for a return to eutrophication by previous phases of impact. Though farming activities appear to have remained predominantly arable, sustained deforestation from *ca.* 0.6 k cal. yrs. BP indicate a consistent increase in land clearance. *P. boryanum* reappeared following the delayed release of sedimentary P and was accompanied by the appearance of *P. simplex*, both of which increased towards the top of the core. The marked increase in the relative abundance of *F. pinnata* and appearance of *C. meneghiniana* and Cladocera ephippia, from *ca.* 0.4 k cal. yrs. BP, accompanied by the reappearance of *A. granulata* at the top of the core, suggests that since the 17th century conditions at Upper Talley have become increasingly unstable and nutrient rich.

CHAPTER 8

INTERPRETING RESULTS FROM LLYN PENCARREG

8.1 Introduction

As in Chapter 7 the Llyn Pencarreg interpretation is divided into three sections; 'Nutrient Status', 'Ecosystem Status' and 'Drivers of Limnological Change'. A summary of key indicators is given in Figures 8.1a (limnological) and 8.1b (lithology and catchment) and provides strong evidence for ecosystem response to long-term natural and/or cultural eutrophication. Zonation again was undertaken visually in order, to highlight correlations between significant shifts in the data, and labelled LPMPZ-1 to LPMPZ-6 (i.e. Llyn Pencarreg Multi-Proxy Zones). As with Upper Talley, a comparison of these zones against the anthropogenic and climatological influences discussed in Section 3.5 are given in Figure 8.2. Figure 8.3 shows catchment archaeology at Llyn Pencarreg as potential evidence of anthropogenic impact.

8.2 Nutrient Status

LPMPZ-1: 6.80 m – 5.95 m (*ca.* 5.1 – 4.6 k cal. yrs. BP)

Diatoms are preserved from *ca.* 4.7 k cal. yrs. BP, during the mid-Holocene. At this point of the sequence, the DI-TP mean of 59.6μg l^{-1} in Figure 8.1a indicates eutrophic conditions. This is supported to some degree by the $\delta^{13}C$ values in Figure 8.1b, which

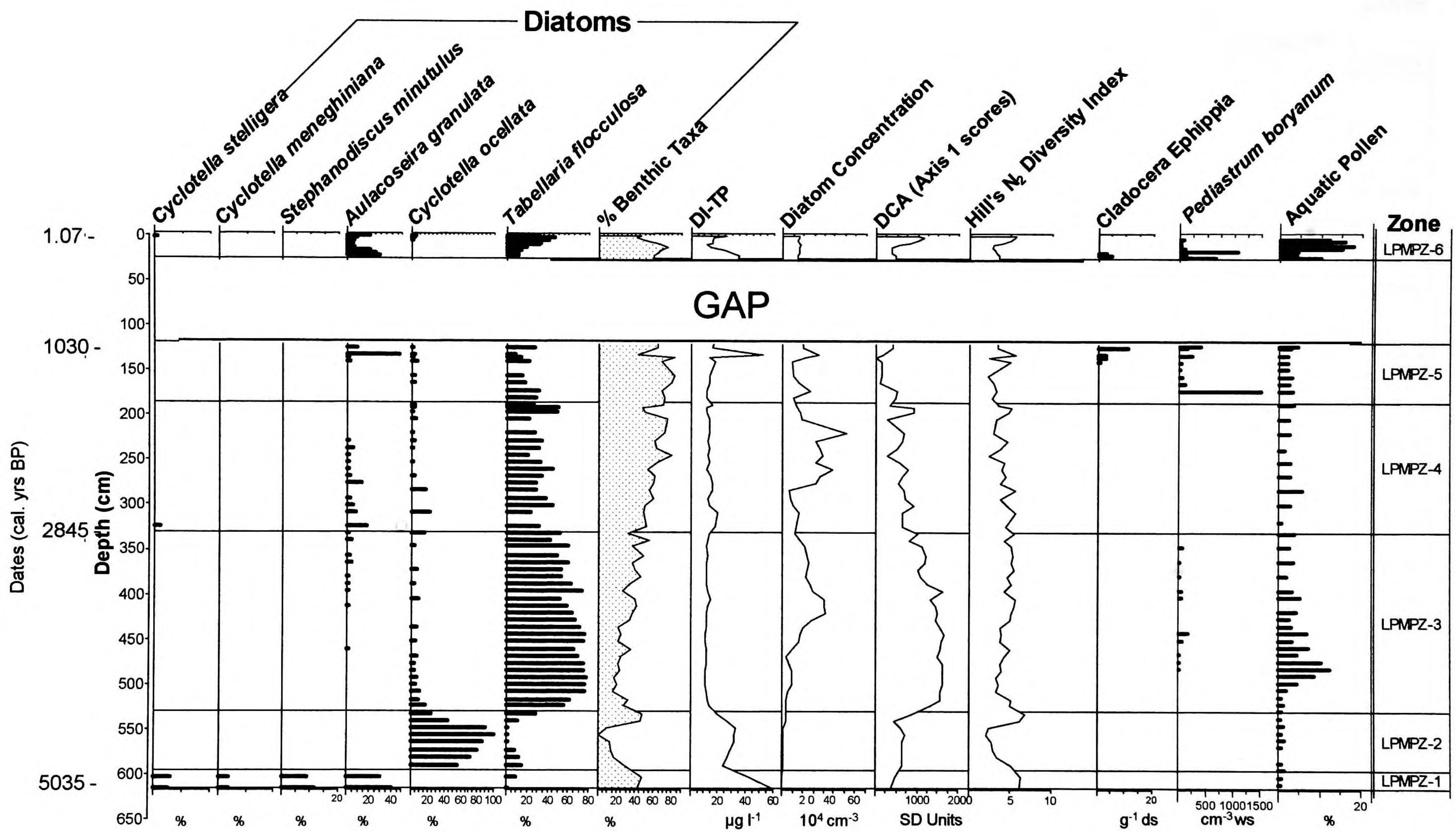


Figure 8.1a Summary diagram showing the multiproxy limnological results for core LPC/05 including dominant planktonic diatom taxa present at >2%, % benthic taxa, diatom-inferred total phosphorus (DI-TP), diatom concentration, DCA Axis 1 scores, Hill's Diversity N_2 Index, Cladocera ephippia, *Pediastrum* spp. and aquatic pollen.

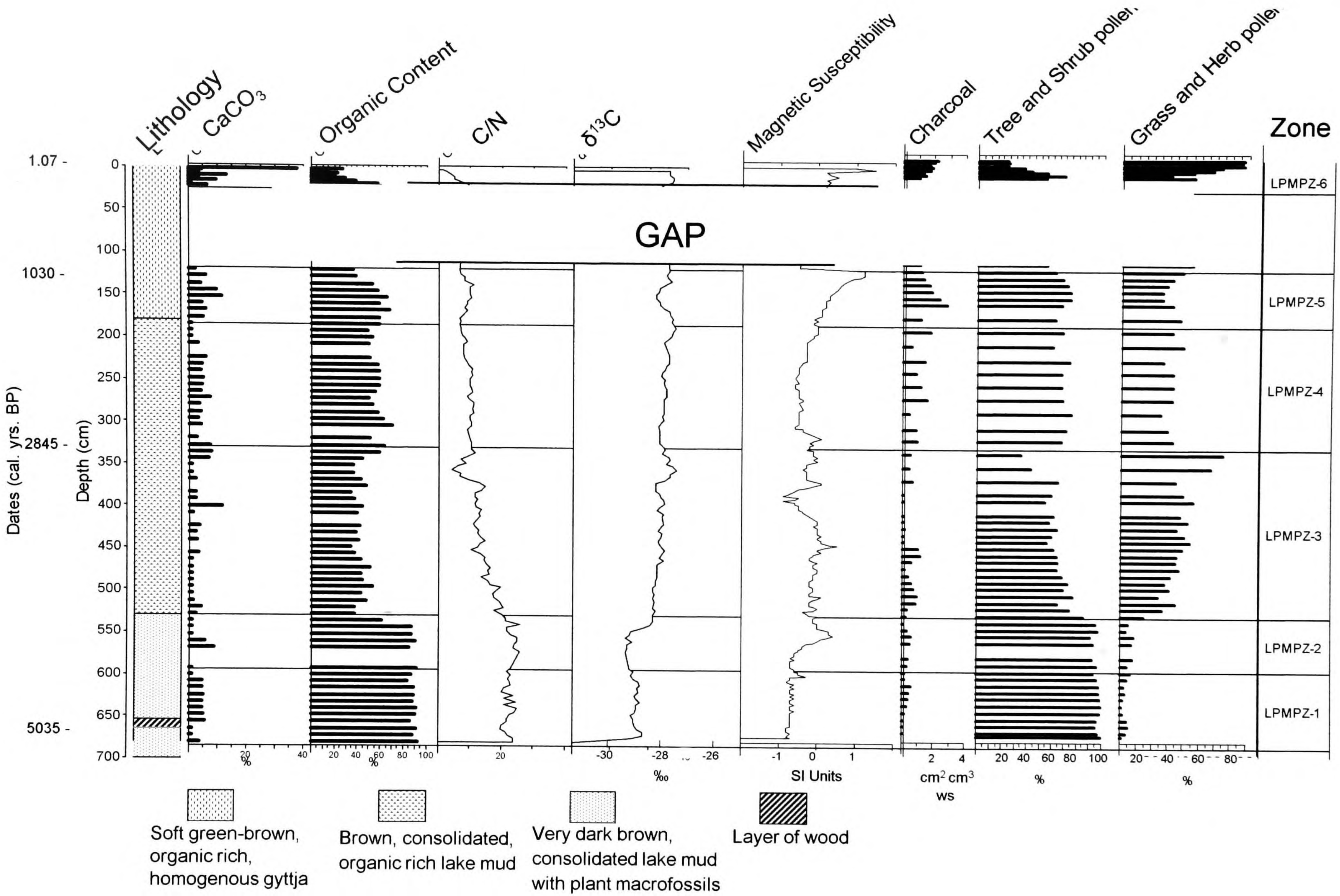


Figure 8.1b Summary diagram showing the multiproxy results for core LPC/06 including lithology, % $CaCO_3$ (calculated from LOI at 850°C), % organic carbon (LOI at 550°C), C/N ratios, $\delta^{13}C$, magnetic susceptibility values, selected pollen groups (adapted from Johnson, 1998) and charcoal abundance.

increased from –32.4‰ at the core base to –30.9‰ at 6.68 m (*ca.* 5.0 k cal. yrs. BP) reflecting an increase in primary production, but then decreased to –32.0 ‰ at the upper zone boundary suggesting a fall in productivity. The C/N profile has a mean value of *ca.* 20, which indicates low sedimentary algal biomass or high terrestrial inwash, though sedimentary organic carbon is extremely high (90%) and could indicate a combination of both.

The high organic carbon in this zone was reflected by the very dark brown and fibrous sediment matrix, which contained a high proportion of terrestrial material such as leaves, twigs and decomposed plant matter (Table 6.1). In terms of lake productivity, as discussed previously in section 7.1, a high proportion of organic carbon in the sedimentary record coupled with high C/N values can disguise the signal for algal biomass and any change in algal production may not be apparent (Brenner *et al.*, 1999; Sampei and Matsumoto, 2001). Kaushal and Binford (1999) show that a C/N value of 20 is indicative of sediments containing a high proportion of arboreal plant matter, particularly *Quercus*, which can produce C/N values of around 22 (Meyers and Lallier-Verges, 1999). Though the C/N ratio of *Quercus* was not analysed at Llyn Pencarreg, modern plant samples show that *A. glutinosa*, which produced a relative abundance of *ca.* 60% to 70 % in this zone (Figure 6.9), also has a high C/N value of 20 (Table 5.2).

LPMPZ-2: 5.95 m – 5.30 m (*ca.* 4.6 – 4.1 k cal. yrs. BP)

In this zone an abrupt and complete shift in the diatom flora to planktonic dominance by *Cyclotella ocellata*, and a decline in the proportion of benthic taxa to the mid-zone accompanies an inferred reduction in nutrient status to mesotrophic (24. 2μg l^{-1}) (Figure 8.1a). An estimated increase in DI-TP to a peak of 32.7 μg l^{-1} at 5.30 m (*ca.* 4.3 k cal. yrs. BP) is followed by a decline thereafter to a minimum of 17.7 μg l^{-1} at the upper zone boundary (*ca.* 4.1 k cal. yrs. BP), driven mainly by the appearance and dominance of *C. ocellata*. These shifts may indicate that the lake became deeper during this phase, in which case the initial decrease in DI-TP may be associated with the dilution of nutrient concentrations (Wolin and Duthie, 1999). The subsequent transitions may be a function of lake-ontogeny, with stabilisation of nutrient input following the initial unstable, turbid phase. This pattern is not mirrored exactly by the $\delta^{13}C$ data, which show an initial decrease followed by a sharp increase to –29.2‰ (*ca.* 4.3 k cal. yrs. BP) at the height of DI-TP inferred enrichment. Unlike DI-TP, values then remained high.

The trend in C/N data (Figure 8.1b) correlates well with other indicators during the inferred height of enrichment at the mid-zone, but this is followed by a marked decrease in C/N towards the upper zone boundary, which is not consistent with the inferred reduction in nutrient status. This apparent shift in the source of organic matter was accompanied by a decrease in organic carbon to 36% at the top of the zone and a gradual transition from very dark brown consolidated lake sediment to a light brown, homogenous, soft lake mud. However, the source of this organic component is not entirely clear; the less fibrous texture of sediments and loss of aggregate plant material suggest that the lower C/N values could also be driven by a decrease in terrestrial material influx.

LPMPZ-3: 5.30 m – 3.25 m (*ca.* 4.1 – 2.7 k cal. yrs. BP)

In the mid-late Holocene, diatom-based inferences of trophic status do not correlate well with other indicators. Following a second major change in diatom assemblage composition to one dominated by *T. flocculosa*, DI-TP values are complacent and low throughout the zone (mean 11.8 μg l^{-1}) and assemblages show a sustained trend towards increasing proportions of benthic taxa (Figure 8.1a). However, other indicators imply an opposite trend. Compared to the relative stability of LPMPZ-1 and LPMPZ-2, both C/N and $\delta^{13}C$ correlate well and exhibit marked and sustained trends towards decreasing and increasing values respectively, peaking between 3.80 m (*ca.* 3.1 k cal. yrs. BP) and 3.20m (*ca.* 2.7 k cal. yrs. BP) (Figure 8.1b). Above this point increases in magnetic susceptibility and $CaCO_3$ content and a later increase in organic carbon also indicate significant limnological change. Although not reflected in the DI-TP reconstruction, there is some qualitative diatom-based evidence that by *ca.* 3.4 k cal. yrs. BP the lake was displaying signs of enrichment. The presence of the classic eutrophic taxon, *A. granulata* above 4.60 m (*ca.* 3.7 k cal. yrs. BP) in Figure 8.1a is notable, and accompanies the introduction of *A. formosa* and a decrease in *T. flocculosa* abundance.

In the context of the patterns discussed above, a mean DI-TP estimate of 11.8 μg l^{-1} for the entire zone, reflecting a lake of stable mesotrophic status, is difficult to explain. Low DI-TP estimates for this zone are mainly influenced by the dominance of mesotrophic planktonic taxa, particularly *T. flocculosa* (mean DI-TP for the zone = 14.8 μg l^{-1}). The shift in planktonic diatom taxa at the zone base was also accompanied by the loss of benthic taxa present in LPZ-2 and the return of a wider range of mesotrophic, benthic species observed in LPZ-1. The sustained increase in total benthic proportions throughout the zone largely reflects the gradual introduction of new benthic species (Figure 6.3; *Achnanthes subatomoides, Anomoenoneis brachyseira, Frustulia rhomboides, Gomphonema angustatum, G. parvulum*), which may be related to patterns of aquatic plant growth and expansion of the submerged aquatic macrophyte habitat (Sayer *et al.*, 2007). This is supported by the peak in proportions of aquatic pollen types at *ca.* 3.8 k cal. yrs. BP. As aquatic pollen abundance decreases, diatom concentrations increase, which may suggest that nutrient levels were kept relatively low throughout the zone first by a dominance of aquatic macrophytes, then by an increase in diatom production (Scheffer, 2004).

LPMPZ-4: 3.25 m – 1.85 m (*ca.* 2.7 – 1.4 cal. yrs. BP)

During the Late Holocene, Figure 8.1a shows that the dominant planktonic diatom continued to be mesotrophic *T. flocculosa*, although it was present at a lower relative abundance than in LPZ-3. Benthic taxa continued a sustained increase from LPZ-3, influenced by increases in the relative abundance of some taxa (eg. *Stauroneis anceps*, *Cymbella silesiaca*, *Eunotia bilunaris*, *Navicula subminuscula*, *Fragilaria virescens* and *F. exigua*) and the introduction of others (eg. *F. pinnata, F. capucina, F. vaucheriae, N. pupula, Pinnularia polyonca*) (Figure 6.3). Low DI-TP estimates were maintained for the entire zone reflecting a lake of mesotrophic status, although an increase to 18.0μg l^{-1} at 3.10m (*ca.* 2.6 k cal. yrs BP) and a corresponding increase in *A. granulata* and *C. ocellata*, indicates minor enrichment. Estimates for the remainder of the zone have a slightly higher mean of 14.8 μg l^{-1}, compared to 11.8 μg l^{-1} in the previous zone.

Diatom concentrations displayed a second peak (*ca.* 1.8 k cal. yrs. BP) although a subsequent decrease suggests that changes in concentration are not directly associated with DI-TP and that diatom production may be influenced by some other environmental variable (Schelske *et al.*, 1986). Figure 6.11, which indicates almost perfect correlation between the DC and PC/DC trends, shows that the rate of sediment accumulation is also not an influencing factor. The increase in diatom concentration may therefore be related to changes in water chemistry and the availability of nutrients following the decrease in aquatic macrophytes (Sheffer, 2004), or it may be the result of increased silica concentrations, which can occur as a result of changes in ground water inputs providing the lake with mineral-rich water (Schelske *et al.*, 1986; van Donk and Kilham, 1990).

If the $\delta^{13}C$ curve in Figure 8.1b reflects rates of primary production, values in this zone imply a sustained increase, which correlates with the changes in DI-TP, reaching –26.7‰ upper zone boundary. C/N ratios also decreased to the top of the zone, where a value of 13.4 suggests increased importance of algal biomass in the lake sediments. Since these changes are occurring in a zone of increasing enrichment, a C/N ratio of 13.4 is probably reflecting a balance between terrestrial inwash and aquatic productivity.

LPMPZ-5: 1.85 m – 1.10 m (*ca.* 1.4 – 0.85 cal. yrs. BP)

Approximately parallel increases in DI-TP values (to 15.6 μg l^{-1} at the top of the zone) (Figure 8.1a), organic carbon (peak of 65% at 1.70 m; *ca.* 1.3 k cal. yrs. BP) and $CaCO_3$ (peak of 13% at 1.60 m; *ca.* 1.2 k cal. yrs. BP) (Figure 8.1b) are all consistent with a peak in the long-term trend of increasing lake productivity. This is marked in the diatom record by a continued decline in the abundance of *T. flocculosa*, the loss of some benthic taxa (e.g. *Nitzschia palea, Amphora libyca, N. minima, A. brachyseira*) and an overall increase in the total proportion of benthic taxa, mainly due to the rise in small *Fragilaria* (e.g. *F. virescens, F. exigua, F. pinnata*) (Figure 6.3). At 1.30m (*ca.* 1.0 k cal. yrs. BP) Figure 8.1a shows that *A. granulata* increased in proportion to 50% in just one sample generating an abrupt, though temporary, peak in DI-TP of 52.4 μg l^{-1}. Although diatom concentrations have shown little correlation with patterns of enrichment at Llyn Pencarreg, an increase to 3.5×10^7 cm^{-3} at the same depth may be associated with increased algal production

However, the C/N and $\delta^{13}C$ data are at odds with this interpretation (Figure 8.1b). A peak C/N value of 15.5 correlating with a peak in *A. granulata* at *ca.* 1.1 k cal. yrs. BP may be a response to land-use activities in the catchment rather than within-lake processes. If the expected increase in algal biomass is masked by a high proportion of terrestrial material, it is possible that the inferred increase in terrestrial in-wash may have delivered excess nutrient loads to the lake (Smol, 2002). The $\delta^{13}C$ profile in Figure 8.1b also shows shifts that are not typical of increased enrichment. However, changing $\delta^{13}C$ values can be interpreted in many different ways and the observed decrease in values may not necessarily imply a decrease in primary production.

Lighter sedimentary $\delta^{13}C$ values can also be influenced by a change in terrestrial plant type and the presence of C_3 land plants from extreme catchment inwash, compared to the heavier C_4 shrubs and grasses. This is supported to a certain degree by the pollen profile, which reflects an increase in pollen from C_3 plants such as *Hordeum*, *Plantago lanceolata* and *Rumex acetosella* (Figure 6.9).

LPMPZ-6: 22 cm – 0 cm (*ca.* 0.18 – 0 cal. yrs. BP)

Following a gap in the sequence, a eutrophic, turbid state is inferred from the initial dominance of the diatom assemblage by *A. granulata* and *A. Formosa* (Figure 8.1a). With the renewed dominance of *T. flocculosa* thereafter, DI-TP decreased from 34.9 $\mu g\ l^{-1}$ at the zone base to 16.5 $\mu g\ l^{-1}$ at 1.5 cm indicating a recent reduction in nutrient concentrations. At the top of the core, however, a higher DI-TP estimate of 25.9 $\mu g\ l^{-1}$ is associated with an increase in *A. granulata* abundance and closely matches the measured modern water TP measurement of 31.0 $\mu g\ l^{-1}$.

Although C/N analysis has demonstrated its lack of reliability as a proxy for trophic status, the decrease from 14.5 at the zone base to 10.8 at the top of the core suggests greater aquatic algal biomass and supports an increase in nutrient enrichment (Figure 8.1b). The overall trend from LPZ-2 to the top of the core is therefore one of increased sedimentary algal biomass indicating that over the last *ca.* 4.3 k cal. yrs. BP phytoplankton production at Llyn Pencarreg has increased steadily. There have been few long-term studies of C/N, since the technique has been used mainly to assess change and modern nutrient enrichment associated with human impact (Bernasconi *et al.*, 1997; Brenner *et al.*, 1996, 1999; Gu *et al.*, 1996). However, Likens *et al.*,

(1985) found that at Mirror Lake, New Hampshire, USA, autochthonous productivity gradually increased throughout the Holocene and was probably related to a gradual decrease in lake depth and an increase in algal productivity as lake metabolism increased. Additionally, at the top of the core they found that 79% of organic material was derived from autochthonous sources, of which photosynthesising primary producers provided 70% of all ecosystem source carbon.

The increase in $\delta^{13}C$ values from –28.4 ‰ at the zone base to –27.5 ‰ at the top of the core supports increases in productivity as lighter $\delta^{12}C$ is preferentially utilised (Figure 8.1b). Values then decrease towards the top of the core suggesting that factors other than primary production may be influencing the $\delta^{13}C$ curve.

8.3 Ecosystem status

LPMPZ-1: 6.80 m – 5.95 m (*ca.* 5.1 – 4.6 k cal. yrs. BP)

The diatom data in particular indicate that LPZ-1 is highly distinct. An initial DI-TP estimate of 59.6 $\mu g\ l^{-1}$ in Figure 8.1a reflects the presence of *C. meneghiniana* (optimum; 254.4 $\mu g\ l^{-1}$) and *S. minutissima* (optimum; 40.1 $\mu g\ l^{-1}$), which are common co-dominants in eutrophic lakes (e.g. Witkovski, 1994; Yangdong and Brugham, 1997). In this initial eutrophic phase the dominance of *A. granulata* (optimum; 133.5 $\mu g\ l^{-1}$) also indicates high turbidity and light attenuation (Kilham, 1986; Kilham *et al.*, 1996).

Mesotrophic *C. stelligera* (18.46 $\mu g\ l^{-1}$) is also present; its appearance is often associated with a decrease in the relative abundance of *A. granulata* prior to and following peaks in nutrient enrichment as a response to reduced turbidity (Harris *et al.*, 2006). Using correlation analysis, Köster and Pienitz (2006) found that the production of *C. stelligera* is strongly associated with a high percentage of sedimentary organic carbon due to human disturbance, a well-mixed water column, and high nitrate concentrations. Prolonged circulation within the water column, which Rautio *et al.* (2000) and Catalan *et al.* (2002) found to be characteristic of mesotrophic, subarctic and alpine lakes, may also explain the assemblage, and this is supported by the presence of *A. granulata*, although lack of frustules below 6.14 m means that this cannot be confirmed for that depth. It is also possible that nitrates are an important contributor to nutrient concentrations, in addition to total phosphorus (Bobbink *et al.*, 1993).

In this zone, diatom species diversity is high (Hill's N_2 = 6.3), reflecting benthic diversity that includes small *Fragilaria* (*F. virescens, F. exigua, F. capucina* var. *gracilis, F. pinnata*) and *Navicula* species (*N. menisculus, N. recens, N cryptotenella, N. saxophila*) with predominantly mesotrophic TP optima and a broad range of habitat preferences (Figure 6.3). This has produced a lower DI-TP estimate (59.6 $\mu g\ l^{-1}$) than would be expected for a lake showing signs of nutrient-related turbidity. Similarly, at Lake Mondsee, Austria, Dokulil and Teubner (2005) found that many benthic species displayed a broad tolerance to changes in water quality and responded at different rates to shifts in nutrient loads, affecting the reliability of DI-TP reconstruction. Benthic species do, however, provide important ecological information about habitat availability and their presence here suggests that the lake must have been shallow enough to allow for photosynthesis in the benthic zone.

The presence of *Amphipleura pelucida* suggests that not only was the lake shallow but possibly also muddy and high in silt (Kelly, 2000). This species is often found in shallow pools attached to filamentous algae (e.g. *Spirogyra, Anabaena*, *Oscillatoria*) during phases of enrichment and is particularly common during the summer months when blooms can appear due to the higher concentration of nutrients. The presence of *Cocconeis placentula* (optimum; 133.0 μg l^{-1}) in Figure 6.3 supports the presence of filamentous algae, particularly *Cladophora*, which increases in association with increased nutrient loads and high organic matter (Germain, 1981). In Brazil, O'Farrell (1994) observed *A. pellucida* as part of a eutrophic flora, dominated by *C. meneghiniana* and *A. granulata*, in rivers with a pH of >7, low transparency and a high degree of organic pollution. Surprisingly though *A. pelucida* has a low mesotrophic optimum of 24.5 μg l^{-1} (Battarbee *et al.*, 2000).

Pediastrum boryanum is present at *ca.* 4.8 k cal. yrs. BP (6.40m) in Figure 8.1a, although in just one sample. This species displays little ecological restriction but is abundant in shallow ponds and lakes where it competes well as part of the wider algal community particularly when nutrients are concentrated (Komárek and Jankovská, 2001). Aquatic pollen was also present, although at low abundance and was mainly composed of Cyperaceae, which typically grow in the littoral zone and are widely tolerant of a range of aquatic habitats and trophic states (Haslam *et al.*, 1975). Its presence is therefore difficult to interpret in terms of water quality, but its occurrence does indicate the development of a fringing reed bed. The low abundance of Cladocera ephippia suggests that regardless of the inferred high trophic status, Cladocera were reproducing under stress-free conditions favouring parthenogenesis (Langdon Brookes, 1959).

LPMPZ-2: 5.95 m – 5.30 m (ca. 4.6 – 4.1 k cal. yrs. BP)

During the mid Holocene, at the base of LPZ-2 the switch in diatom dominance to *C. ocellata* correlates with a trend towards increasing trophic status (DI-TP; 32.7 $\mu g\ l^{-1}$) (Figure 8.1a). Although *C. ocellata* is found in a wide range of environments and has a broad tolerance to temperature gradients and trophic regimes, studies of eutrophication show that its relative abundance can increase in accordance with increasing TP as part of a shift from a high diversity, benthic flora to a low diversity, plankton-dominated diatom assemblage (Cherapanova *et al.,* 2007). This is a direct consequence of the loss of light penetration to the benthic zone. The assemblage has a lower diversity than LPZ-1, with a Hills N_2 of 2.0 compared to 6.3 in the previous zone.

Rather than indicating a shift in productivity, it is possible that the dominance of plankton here may instead indicate an increase in lake level as the lake basin developed (Wolin, 1996). At Hamilton Harbour, Lake Ontario, Canada, Duthie *et al.* (1996) found that benthic and epiphytic taxa were replaced by euplanktonic species during flooding and lake deepening, *ca.* 4.0 k cal. yrs. BP, followed by a gradual increase in benthic taxa due to long-term infill. More specifically, Moos *et al.* (2005) used detrended correspondence analysis (DCA) to show that centric taxa, including *C. ocellata*, tend to increase in relative abundance as water depth reaches *ca.* 8.0 m, depending on water transparency. The diatom signal of changing lake level can become complicated in lakes where nutrient enrichment is occurring, as the shift in diatom assemblage described above can reflect either increasing water depth or increasing turbidity (Wolin, 1996). The continued presence of Cyperaceae at low abundance suggests a narrow reed bed typical of deeper waters with a restricted littoral zone (Davis, 1999).

From mid-zone (*ca.* 4.3 k cal. yrs. BP) *C. ocellata* abundance falls along with a renewed rise in benthic taxa and increase in overall diatom diversity (Hill's N_2; 6.9) (Figure 8.1a). This is matched by a decrease in DI-TP reflecting a return to mesotrophic status by *ca.* 4.2 k cal. yrs. BP. Figure 8.1b shows that these changes, however, are at odds with the corresponding peak in magnetic susceptibility, suggesting catchment erosion (Dearing, 1991; 1999), and the increase in $\delta^{13}C$ values and decreasing trend in C/N, are probably reflecting increased primary production and sedimentary algal biomass. It may be that the marked increase in magnetic susceptibility is associated with changes in land-use (Thompson *et al.*, 1975) coupled with the inferred development of Llyn Pencarreg, in which case the shift in trophic state may reflect the dilution of P concentrations as the lake deepened, whilst the increase in productivity may support ecosystem development as the lake expanded.

LPMPZ-3: 5.30 m – 3.25 m (*ca.* 4.1 – 2.7 cal. yrs. BP)

A major shift in the dominant diatom species from *C. ocellata* to *T. flocculosa* correlated with a further decrease in DI-TP to 17.7 $\mu g\ l^{-1}$ (Figure 8.1a) and a sustained decrease in organic carbon at the zone base from 90% to 37% (Figure 8.1b). The presence of *P. boryanum* colonies at higher abundance than LPZ-2 suggests a healthy phytoplankton community (Janokovská and Komárek, 2000) and a wide range of benthic diatom taxa produced an increase in species diversity (Hills N_2; 6.9). There is also an increase in the proportion of acidophilic taxa, indicating a drop in pH. In addition to *T. flocculosa*, these include *Pinnularia subcapitata, P. microstauron, P. divergens* and *Neidium alpinum* (Figure 6.3). These species are characteristic of high Arctic, tundra ponds (Canada; Antoniades *et al.*, 2005), lakes in the Rocky Mountain Foothills (Alberta, USA; Hickman and Schweger, 1991) and in the Adirondacks (northern New York, USA; Whitehead *et al.*, 1986), all of which are situated in catchments where water chemistry is influenced by low nutrient, acid based soils.

DCA results from Llyn Pencarreg (Figure 6.5) also indicate that TP concentrations are not the only factor influencing species distribution. There may be an underlying acidity gradient in the species data, and the introduction of these acidophilic species at Llyn Pencarreg suggests a decrease in pH over time in the early to mid-Holocene. Their appearance correlates with a decrease in DI-TP, which would be expected due to a decrease in the binding of P to organic acids (Prather and Hickman, 2000) in more acidic water conditions. The pH change may have been related to peat or heath formation in the hydrological catchment, which would release hydrogen ions (Turner *et al.,* 1989) and organic acids (Woods, 1989) into the inflowing water. The presence of *Sphagnum* spores and *Calluna vulgaris* pollen (Figure 6.9) support this interpretation to some degree, although there is no major change in the pollen signal in this zone.

The diatom assemblage also includes *Nitszchia palea*, *Navicula cryptotenella, N. cryptocephala* and *Eunotia bilunaris* (Figure 6.3), which also occur together in the Gravatai River, Brazil (Salomoni *et al.*, 2006) and have been described as typical of polluted waters (van Dam, 1994; Lobo *et al,* 1995). The inferred increase in rates of primary production ($\delta^{13}C$) and algal biomass (C/N) (Figure 8.1b) and the return of *A. granulata* (Figure 8.1a) in the diatom assemblage at *ca.* 3.6 k cal. yrs. BP is in conflict with the inferred low trophic status of the lake. As discussed in LPZ-1, the appearance of *A. granulata* may be related to turbidity and increased light attenuation due to eutrophication (Scheffer, 2004). Studies have also shown, however, that its presence is directly related to water column instability during vertical mixing (da Silva *et al.*, 2005; Zohary, 2004). Owen (1992) and Bradbury (1971) have shown that *Aulacoseira* species are characteristic of changes in water level due to the increase in turbulence, whilst Anderson and Odgaard (1994) found their presence to be associated with wind exposure and changes in thermal stratification. If the inference of an increase in water depth at the base of the previous zone is correct, then an increase in stratification would have led to the introduction of a mixing regime at Llyn Pencarreg.

By 3.4 k cal. yrs. BP. *A. formosa* had appeared in the diatom record (Figure 6.3). This species is common in temperate lakes (Bertrand *et al.,* 2003), and has been observed under a range of trophic conditions (Reynolds, 1998, 2000; Romo, 1998; Negro *et al.*, 2000). Experimental data have also shown that its growth patterns can be affected by stratification, temperature and an increase in Si reserves to >0.5 $\mu g\ l^{-1}$ (Bertrand *et al.,* 2003). Like *A. granulata, A. formosa* often out-competes smaller planktonic diatoms, such as *C. ocellata,* when Si availability is high (Kilham, 1986). An increase in Si reserves is highly possible with an increase in lake depth (Schelske *et al.,* 1986) and may be reflected in the increase in diatom concentrations at *ca.* 3.4 k cal. yrs. BP. The depletion of Si limits diatom production (Kilham, 1986) and the steady decrease in diatom concentrations from this point towards the top of the zone may be directly related to diminishing Si reserves as enrichment and diversity increased (Shelske *et al.*, 1986).

In this zone, aquatic macrophyte pollen percentages show a marked increase from stable, low values of *ca.* 1% to a peak of *ca.* 13% at 3.80 m (*ca.* 3.1 k cal. yrs. BP) (Figure 8.1a). This correlates with the reduction in DI-TP and reflects an increase in spores of mesotrophic *I. Lacustris* (Figure 6.9). The rest of the aquatic pollen assemblage was mainly composed of submerged macrophytes (e.g. *I. echinospora, Ranunculus aquatilis*) commonly found in oligo- to mesotrophic waters. *I. lacustris* in particular is found at sites with thin, peaty soils and both species are characteristic of remote highland lakes (Haslam *et al.*, 1975), which supports the inference of acidification.

LPMPZ-4: 3.25 m – 1.85 m (*ca.* 2.7 – 1.4 cal. yrs. BP)

During the Late Holocene the increase in relative abundance of *A. granulata* at *ca* 1.8 k cal. yrs. BP and the continuous presence of *A. formosa*, during an inferred phase of lower trophic status, indicates that stratification probably persisted at Llyn Pencarreg (Figure 8.1a). In a zone of gradually decreasing species diversity (from a Hill's Index of 5.6 at the core base to 3.4 at the top of the zone) the sustained rise in proportions of benthic taxa was driven by increased abundance of small *Fragilaria* species (eg., *F. exigua, F. virescens, F. pinnata*; Figure 6.3). Discussing lake-ontogeny, Wetzel (2001) stated that once a lake basin has filled with water, the deposition of allochthonous and autochthonous material will cause gradual infilling and shallowing over time and therefore benthic taxa will increase in relative abundance as light penetration and the availability of benthic habitats increases.

The reduction in relative abundance of aquatic pollen in this zone (compared to LPZ-3) suggests that the increase in benthic diatom taxa was not particularly associated with the presence of submerged macrophytes (Sayer *et al.*, 2007). The main pollen and spore taxa are the emergent Cyperaceae and submerged *I. Lacustris* (Figure 6.9), which is particularly associated with mesotrophic lakes up to 6 m deep (Haslam *et al.*, 1975). Its preference for low alkalinity means that *I. lacustris* is often found in catchments with peat formation and its presence therefore supports the previous inference of low pH (the increase in *Pinnularia* species also supports this). *Sphagnum* spores and *Calluna vulgaris* pollen are both present in the Llyn Pencarreg pollen record, indicating the continuing presence of acid-tolerant species in the lake catchment, albeit at slightly lower abundances than the previous zone.

LPMPZ-5: 1.85m – 1.10m (*ca.* 1.4 – 0.9 cal. yrs. BP)

Nutrient enrichment peaks at *ca.* 0.9 k cal. yrs. BP with an increase in both *A. granulata* and diatom concentrations (Figure 8.1a). Diatom production at Llyn Pencarreg appears to be influenced by multiple factors. The lack of change in aquatic plant abundance, however, suggests that these changes are not associated with changes in the littoral zone (Figure 6.9). Whitmore (2004) found that although diatom concentrations in surface sediments from lakes in Florida, USA, were positively correlated with limnetic chlorophyll *a*, other factors such as Si concentration and sediment accumulation rate influenced the trend. The peak in *A. granulata* abundance at *ca.* 0.9 k cal. yrs. BP supports an interpretation of high Si availability (Kilham, 1986) and indicates that the peak in nutrient availability was again accompanied by turbidity and increased light attenuation.

Figure 8.1a indicates that stress within the wider ecosystem was indicated by the presence of Cladoceran ephippia for the first time (*ca.* 1.0 cal. yrs. BP), suggesting that enrichment caused ecological change at various levels within the ecosystem. There is increasing evidence to support the idea that increasing nutrient loads destroy the integrity of aquatic systems and causes stress through changing food web structure and predator-prey relationships (Shovonlal and Chatopadhyay, 2007). It appears that following an initial increase in Cladocera (particularly *Daphnia*) as a response to increased nutrient concentrations and algal abundance, the production of ephippia at the height of eutrophication is a response to stress, and is important in the maintenance of ecosystem stability (Persson *et al.,* 1992; 2001).

The marked increase in *P. boryanum* colony abundance at the earlier date of *ca.* 1.3 k cal. yrs. BP may also be associated with enrichment (Figure 8.1a). Although its Holocene distribution is cosmopolitan, from lake and pond sediments to peat bogs, *P. boryanum* appears to be a taxon of altitudes below 600 m where it is often seen to increase in eutrophic lakes as both enrichment and temperature increase (Komárek, and Jankovská, 2001). Its disappearance at the height of eutrophication may also be associated with ecosystem status and may be reflecting changes in algal community dynamics. This interpretation is supported by an initial increase in diatom diversity (Hill's N_2 = 5.7), possibly related to the initial increase in productivity, followed by a reduction to a Hill's N_2 value of 3.8 at the height of enrichment.

LPMPZ-6: 22 cm – 0 cm (*ca.* 0.2 – 0.0 k cal. yrs. BP)

Over the past *ca.* 200 years correlated shifts in the proxy data indicate major changes in ecosystem status. These include changes in diatom assemblage composition, *P. boryanum* and Cladoceran ephippia abundance, associated with high nutrient concentrations and low diatom diversity (Hill's N_2 = 2.8) at the start of the zone, where DI-TP estimates decline before increasing again towards the top of the core (Figure 8.1a). Ecosystem status at the zone base, where *A. granulata* dominates the diatom flora, is similar to the peak at the upper boundary of LPZ-5, which is interpreted as a shallow, turbid, eutrophic lake. *P. boryanum* was also abundant, indicating competition in the algal community, although it is often replaced by toxic algae such as *Aphanizomenon* and *Anabaena* with extreme enrichment (Moss, 1988; Smol, 2002). Ecosystem stress is also indicated by the high Cladoceran ephippia concentrations at the zone base.

The reduction in DI-TP at the mid-zone (*ca.* 120 cal. yrs. BP) is associated with an increase in *T. flocculosa* and other diatom taxa at the expense of *A. granulata*. These changes correlate with higher species diversity (Hills N_2 = 5.5), a marked reduction in *P. boryanum* abundance and the absence of Cladoceran ephippia, suggesting good ecosystem health. However, non-biological proxies do not correlate with these changes; In Figure 8.1b the increase in magnetic susceptibility and $CaCO_3$, accompanied by increased inferred primary production (low C/N and high δ13C) are indicative of an increase in productivity.

In the very recent past the lake has returned to eutrophic status. This is supported by an increase in DI-TP to 25.3 $\mu g\ l^{-1}$ at the top of LPZ-6 in Figure 8.1a, accompanied by increases in planktonic *A. granulata* and *C. ocellata* and a reduction in the proportion of benthic diatom taxa. The increase in Hill's N_2 from 2.3 at the mid-zone to 5.5 at the top of the core, and the increase in relative abundance of aquatic pollen to 18%, appear to be an early response to increased nutrient loads and suggest a speedy return to eutrophication at Llyn Pencarreg in the past 100 years.

8.4 Drivers of Limnological Change

LPMPZ-1: 6.80 m – 5.95 m (*ca.* 5.1 – 4.6 k cal. yrs. BP)

At the base of this zone the palynological data indicate mature woodland with very high relative abundance of arboreal pollen (> 95%), composed mainly of *Betula, Corylus avellana*-type, *Alnus glutinosa* and *Quercus* (Figure 6.9). Underlying fluctuations in the relative abundance of *A. glutinosa* (e.g. at 4.9 k cal. yrs. BP) and *C. avellana*-type (e.g. at 4.7 k cal. yrs. BP) pollen may suggest phases of woodland disturbance associated with human activity; this is supported by the presence of *Pteridium aquilinum* spores, which usually increase in abundance with a loss of woodland cover (Tansley, 1939); at Llyn Pencarreg, changes in its relative abundance correlate closely with shifts in the relative abundance of *C. avellana*-type pollen in particular. The presence of *Plantago lanceolata* is not an uncommon feature of Neolithic pollen records from Wales and along with Poaceae is often interpreted as representing small areas of woodland meadow, maintained for grazing (Caseldine, 1990). The NAP curve (Figure 8.2) suggests that there may have been minor land-clearance activities from *ca.* 5.5 k cal. yrs. BP, whilst the API for this period indicates significant shifts in the amount of arable and pastoral farming, possibly reflecting the importance of small woodland clearings for grazing and minor cultivation (Brown, 1999; 2002).

Though not within the lake catchment itself, the discovery of two Neolithic axe flints from Glancyforiog (4.0 km southwest of Pencarreg) and Troed-Rhiw-Sion (3.5 km northwest of Pencarreg), and one extremely fine grained recrystallised crystal tuff handle of group VIII type along with two stone axes of polished flint (showing evidence of their latest use as sharpening stones) from Castell-Du (1.0 km north of Pencarreg) indicate human presence within 1.0 km of the site during this period (Figure 8.3). Although the presence of Neolithic axe remains is not at all definitive of human settlement, their importance as tools for woodland clearance supports the above inference of increased land-use. The absence of cereal pollen, low charcoal concentration (Figure 8.2) and low magnetic susceptibility values (Figure 8.1b) indicate that any cultivation, or possible human habitation, in the catchment was still small-scale (Greig, 1987; Webley; 1974). The NAP indicates that, regardless of fluctuations in some arboreal pollen taxa, the landscape was still predominantly woodland. This is supported by the abundance of leaves and twigs in a dark, fibrous sediment matrix. Again this may reflect the use of small woodland clearings for grazing and/or cultivation.

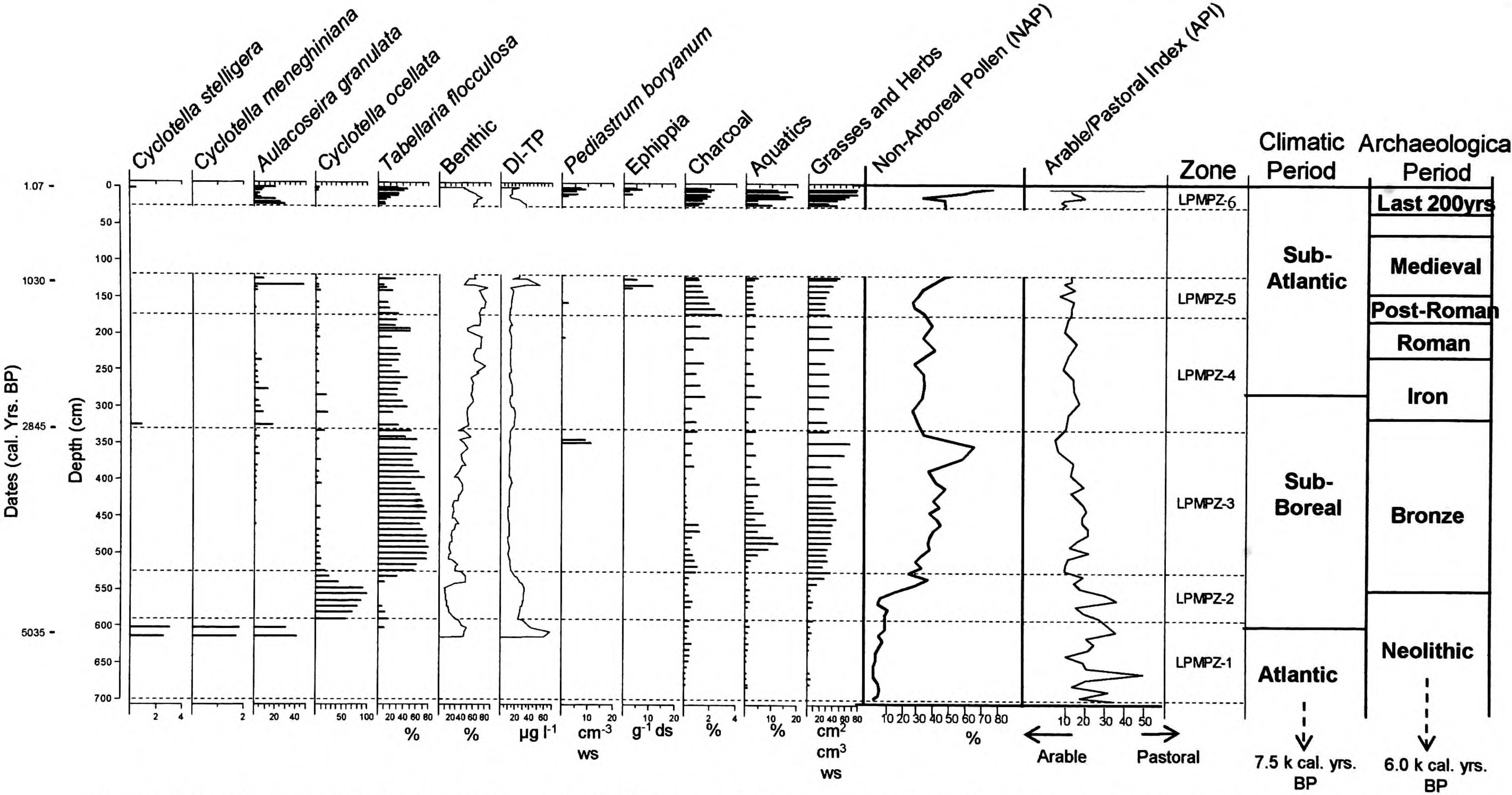


Figure 8.2 Diagram showing the factors potentially responsible for limnological change at Llyn Pencarreg including % benthic taxa, diatom-inferred total phosphorus (DI-TP), relative abundance of tree, shrub, aquatic, grass and herb pollen, % $CaCO_3$ (calculated from LOI at 850°C), % organic carbon (LOI at 550°C), C/N ratios, magnetic susceptibility, non-arboreal pollen (NAP) and the Arable/Pastoral Index (API)

Figure 8.3 Diagram showing the presence of archaeology within and immediately around the Llyn Pencarreg lake catchment

It is also possible that these minor changes in vegetation were climatically driven. The top of the zone marks the end of the Atlantic period when a cold episode caused the decline of thermophilous arboreal taxa and shifts in vegetation community dynamics throughout northwestern Europe (Lamb, 1997). Although it is known that shifts in climate influenced human migration, the causal link between the cultural and natural environments makes interpretation of the pollen record extremely complex (Berglund, 2003). However, evidence for Neolithic settlements and a transient, agricultural economy in Wales during this period suggests that human impact may have significantly changed the immediate environment (Caseldine, 1990; Whittle, 1999).

The basal sediments are devoid of diatoms (Figure 8.2) and may even represent peat or forest floor material prior to the filling of the lake basin; the high C/N ratios (mean; 22) of this zone (Figure 8.1b) could be due to the low organic N content of cellulose and lignin (Taylor *et al.*, 1989; Thain and Hickman, 1999). Likens (1985) noted that both P concentration and organic carbon are exceptionally high in forest floor soils, which is in accordance with the eutrophic status inferred for this part of the sequence. Diatom valves from 6.50 m were highly fragmented, although preservation improved in the upper two samples of this zone. The interpretation of sheltered woodland around Llyn Pencarreg suggests that wind fetch, which can disturb lake sediments and can lead to damaged diatom frustules, is probably not a significant factor at this stage. Flower (1993) suggests that frustule breakage can also be associated with high-energy environments such as rivers or streams, or in the littoral zone where energy is also high, particularly when aquatic vegetation cover is low. This supports an interpretation for the upper part of this zone of a very shallow, muddy environment;

the low abundance of aquatic pollen demonstrated in Figure 8.1a, restricted to Cyperaceae, may suggest varieties typical of damp forest floors and that specialist aquatic species have not yet colonised the water body (Haslam *et al.,* 1975).

The core was taken from what was thought to be the deepest part of the lake, though a complete bathometry was not possible due to ownership rights and lack of access to the whole lake. Although sediments were extracted from the centre of the lake, the presence of terrestrial conditions at the core base may suggest that this was not at the deepest point of the lake and that prior to this the lake-level may have fallen an unknown distance below the level of the coring site (Stager *et al*., 1986). This would explain first the lack of diatoms in the basal sediments, and secondly the presence of broken frustules, which may represent lake shore or shallow littoral sediments during an early phase of lake expansion. However, this phase also represents the end of the wet Atlantic period (*ca.* 5.0 k cal. yrs. BP) when it was more common for lake levels in north-eastern Europe to be high due to the increase in atmospheric moisture and precipitation, which started at *ca.* 7.5 cal. yrs. BP and was sustained over the next 2500 years (Almquist-Jacobson, 1995).

LPMPZ-2: 5.95 m – 5.30 m (*ca.* 4.6 – 4.1 k cal. yrs. BP)

In the mesotrophic lake environment inferred from LPZ-2, the presence of fine sand, silt (Table 6.1) and the reduction of organic material (Figure 8.1b) in the core sediments begins at *ca.* 4.4 k cal. yrs. BP and is typical of deep water deposition (Wolin and Duthie, 1999). The significant shift to an assemblage dominated by planktonic *C. ocellata* supports the inference of a deep water phase. These factors may also, in part, demonstrate increased erosion of the lake edge (indicated at Llyn Pencarreg by the peak in $CaCO_3$ at *ca.* 4.4 k cal. yrs. BP and steady increase in magnetic susceptibility from *ca.* 4.3 k cal. yrs. BP), which can inhibit plant growth around the lake edges causing a decrease in organic matter in core sediments (Hannon and Gaillard, 1997). Although Figure 6.9 shows that arboreal pollen abundance generally remained high, there are significant fluctuations in relative abundance of *Betula, Alnus glutinosa, Corylus avellana, Ulmus* and *Tilia* pollen throughout the zone with a marked decrease in *Quercus* at the top of the zone.

This zone falls during the beginning of the Sub-Boreal period (Figure 8.2) when the climate was warm and humid due to changes in atmospheric circulation and conditions were dryer than during the Atlantic causing changes in vegetation patterns. However, if the decline in species such as *Ulmus* and *Tilia* at Llyn Pecarreg are due to climatic influences (this corresponds with pollen records from across Wales at *ca.* 5.0 k cal. yrs. BP; Turner, 1964), then a marked cooling of climatic conditions is suggested by the reduction of these thermophilous species. Additionally, the observed fluctuations in *Alnus glutinosa*, which is commonly observed in pollen records covering the end of the Atlantic Period, is difficult to reconcile with the warm and humid interpretation suggested by Sub-Boreal peats at sites in Wales (Hughes *et al.*, 2000; Barber *et al.*, 2000; 2003; Hughes and Barber, 2003; Dark, 2006; Chambers, 1983a; Ellis and Tallis, 2001).

Increased erosion and fluctuations in arboreal pollen abundance can also indicate that impact from land-use change in the lake catchment was also affecting the lake sediments. Figure 8.2 shows that these inferred changes were associated first with the

Late Neolithic and may represent an expanding economy of shifting cultivation where temporary forest-openings were followed by arboreal regeneration (Brown, 1999). That *Alnus glutinosa*, *Tilia* and *Ulmus* grow on the more fertile catchment soils may suggest that these areas were selected for land-clearance and cultivation. This is also reflected in studies from Llyn Mire (Moore, 1978) and Trum Felen (Bostock, 1980), both east of Pencarreg and Moel y Gerddi to the west (Chambers and Price, 1988).

Secondly, the sharp increase in NAP (Figure 8.2) from *ca.* 4.2 k cal. yrs. BP (characterised by a decrease in *Quercus* and increases in Poaceae and *Plantago lanceolata*), which marks the first significant increase in open ground, corresponds with the Neolithic/Bronze Age transition. As discussed in section 3.5.4, the warm Boreal temperatures of the Early Bronze Age encouraged major land-clearance at sites across Britain (Smith, 1984). This is reflected in studies from Wales, where major forest clearance at many sites formed a predominantly open landscape and major increase in grassland by *ca.* 3.0 k cal. yrs. BP. At Llyn Pencarreg, the API curve (Figure 8.2) corresponds with these changes though a significant shift towards arable farming is indicated in the lake catchment possibly highlighting its lowland position compared to sites such as Waunwelen in the Preseli Mountains (Seymour, 1985) and Waun-Fignen-Fellen in the Black Mountains.

Although deforestation is often the cause of eutrophication, the opposite relationship seems to have occurred at Llyn Pencarreg. The deepening of the lake basin (which may itself in part be related to human-induced hydrological change; climate is unlikely to be responsible as the Sub-Boreal was a period of reduced rainfall) appears to have buffered the system (Figure 8.2). The proxy data in this zone therefore appear to indicate complex limnological change possibly resulting from a combination of catchment impact and lake development influences that are difficult to disentangle. The inferred human impact, however, may have been instrumental in directly causing a shift in lake level at Llyn Pencarreg. Evidence from Lake Texcoco, Mexico (Bradbury, 1971) indicates that the damming and channelling of waterways by human activity, particularly in association with deforestation, can cause rapid hydrological changes and increases in water level. It is also possible that activities such as dam building and pond creation by beavers, which can cause complex changes in hydrology, may have contributed to a shift in lake depth and changes in water flow (Johnson and Naiman, 1990).

However complex the latter changes are to interpret and separate into their cause and effect, this zone provides significant evidence of major limnological change.

LPMPZ-3: 5.30 m – 3.25 m (*ca.* 4.1 – 2.7 k cal. yrs. BP)

During the mid to late Holocene a second major shift in the diatom sequence correlates closely with palynological evidence for continued changes in land-use. A marked decrease in arboreal pollen percentages continues from the previous zone (Figure 8.1b) consisting of a reduction in the relative abundance of all major species (e.g. *Betula, C. avellana*, *A. glutinosa* and *Quercus*) and corresponding with fluctuations in magnetic susceptibility and a rise in charcoal particles. There is also a lithological shift from dark brown sediments with plant macrofossils to lighter sediments characterised by decomposed organics coupled with the sudden decrease in organic carbon and C/N at the zone base, indicating an increase in autochthonous organic deposition. It should be noted that charcoal concentrations throughout are low compared to those typically recorded during large-scale forest and vegetation removal (Eckmeier, 2007) and the observed changes may simply be associated with an increase in the use of wood as a household fuel or the burning of lakeside vegetation (Griffin and Goldberg, 1979).

As *P. aquilinum* often expands in parallel with the loss of tree cover (Tansley, 1959), it is likely that increasing spore percentages indicates the formation of open habitat as trees were cleared. The first appearance of many herb and cereal pollen taxa suggests that human impact at Llyn Pencarreg reflected an expanding pattern of settled agriculture. Increases in Poaceae, *Plantago lanceolata* and *Rumex acetosa* at *ca.* 4.1 k cal. yrs. BP and the appearance of *Jasione montana*, *Aster*-type and *Ranunculus acris* reflect the development of meadow and suggest an increase in grazing (Figure 6.9). *Hordeum*-type, *Trifolium*, Rubiaceae and Apiaceae pollen also appear in this zone along with *Daucus carota* and Brassicaceae indicating cultivation on a wider scale than in earlier zones. The NAP (Figure 8.2) also shows that there was a sustained increase in open ground within the lake catchment, whilst the API indicates that this was a period of agricultural expansion with a significant increase in arable farming.

A calibrated date of 2,845 cal. yrs BP at the top of the zone places these changes during the Bronze Age, when an upland pastoral economy in this part of Wales was accompanied by a significant increase in cereal cultivation (Caseldine, 1990). The most prominent changes, however, were observed at the zone base, which has an interpolated age of *ca.* 4.1 k cal. yrs. BP. The sharp increase in land-clearance in the previous zone and the Bronze Age agricultural expansion of this zone are somewhat early compared to other sites in Wales (Table 3.2), though Brück (1999) and Thomas (1999) argue that by *ca.* 4.0 k cal. yrs. BP, patterns of agriculture at some sites in Wales did reflect sedentary settlement and expansion, which was encouraged by the warm conditions of the Sub-Boreal. Throughout this zone, the intense agricultural activities that occurred in upland Wales during the Bronze Age (Section 3.5.4) may be the cause of the inferred increase in heathland and gradual change in diatom assemblage to one dominated by acidophilic taxa. This is matched at *ca.* 2.9 cal. yrs. BP by a sudden increase in *Betula*, which tends to prefer well-developed acid based soils (Archibold, 1995).

Compared to Upper Talley there is a distinct lack of Bronze Age archaeology within the Llyn Pencarreg catchment and its immediate surroundings (Figure 8.3). The majority of sites from this period are concentrated to the south of Pencarreg and, as with Upper Talley, are limited mainly to standing stones and round barrows. The discovery of pottery shards at Chwibanogl (3.5 km southeast of Pencarreg), a Bronze Age well (discovered at Banc y Garn during the summer drought of 1976 in attempts to find water) and a decorated, Pygmy cup with bevelled edges from Neudd-Fawr (2km directly north of Pencarreg) also provide evidence of human presence in the Llyn Pencarreg region. The time invested in the technological advancement of using metal alloys and the making of pottery items such as those above suggest an agriculturally driven society with time to spend on making cultural items such as pottery, jewellery and weapons (Parker Pearson, 1999).

LPMPZ-4: 3.25 m – 1.85 m (*ca.* 2.7 – 1.4 k cal. yrs. BP)

During the late Holocene the pollen data indicated a general increase in the relative abundance of arboreal taxa, which then remained stable at *ca.* 70% to 80%, and a notable decrease in grasses and herbs (*Poaceae, Plantago lanceolata, Rumex acetosa* and *R. obtusifolius*) (Figure 6.9). *Salix* pollen increased significantly in relative abundance for the first time, possibly reflecting its growth on the damper soils around the lake; its presence may be influenced by the heavy rainfall of the cool, wet Sub-Atlantic period, which began at *ca.* 2.5 k cal. yrs. BP (Lamb, 1997). *Corylus avellana*-type and *Quercus* pollen percentages also increased towards the top of the zone, suggesting their expansion on the higher, dryer soils and may reflect regrowth following widespread deforestation during the Bronze Age. Although an increase in woodland canopy is supported by the notable reduction in *P. aquilinum* spore abundance, the API (Figure 8.2) indicates that agricultural activity continued in the lake catchment during this period and farming patterns remained predominantly arable

An estimated date of *ca.* 2.7 k cal. yrs. BP at the zone base falls at the Bronze-Iron Age transition, where the diatom record indicates minor increases in nutrient concentrations and turbidity. Although the pollen evidence indicates a general expansion of agricultural and pastoral activities in South Wales from *ca.* 3.7 k cal. yrs. BP (Turner, 1964, 1965; Taylor, 1973; Walker and Taylor, 1976), the colder Sub-Atlantic climate at the end of the Bronze Age drove many communities away from the defended upland sites to small dispersed farmsteads in the foothills. This produced regional differences in settlement patterns depending on local topography, some of which had little impact on the landscape due to their temporary nature (Caseldine, 1990). Despite the inference of catchment stability in the pollen record, the increase in charcoal and magnetic susceptibility from *ca.* 2.0 k cal. yrs. BP and the minor peak in $CaCO_3$ at ca. 2.2 k cal. yrs. BP (Figure 8.1b), accompanied by a significant reduction in *Betula* (*ca.* 2.3 k cal. yrs. BP) (Figure 6.9), suggest human activity.

Both the Iron Age and Roman occupation of Wales are also represented by sediments in this zone. Archaeological evidence for Iron Age activity in the area consists mainly of structures and buildings found within a 2 km radius of Llyn Pencarreg and smaller finds from human habitation, such as the quern stones (used by both Iron Age and Roman populations) discovered at Sunny Hill (2.0km southwest of Pencarreg) and Cellan (0.5km north of Llyn Pencarreg), the latter being the only evidence of Iron Age activity within the Llyn Pencarreg catchment so far (Figure 8.2). Closer to Llyn Pencarreg and within the village itself, Roman finds are more common and include a gold coin from the Arcadius period (383-408 AD), found exposed in a garden in the village of Pencarreg, and a macehead found at the bed of Llyn Pencarreg itself. These items indicate Roman activity close to the site, which may be linked to the building of the road on the ledge above the lake that is known to be Roman in origin; members of the Roman army were often employed to do heavy manual/construction work (Pearson, 1999). Surprisingly however, these latter activities caused very little limnological impact.

LPZ-5: 1.85 m – 1.10 m (*ca.* 1.4 – 0.8 cal. yrs. BP)

Following the Roman period a sustained increase in magnetic susceptibility and a peak in $CaCO_3$ (Figure 8.1b) correlate with the inferred peak in enrichment at *ca.* 0.9 k cal. yrs. BP (Figure 8.1a), providing clear evidence for human impact. Limnological change is also reflected in the transition from brown consolidated lake mud to soft, green-brown gyttja, which typically contains plant and mineral based terrestrial material (Wetzel, 1983). In the pollen record (Figure 6.9) what appears to be an initial phase of woodland regrowth in the lake catchment is followed by decreases in *Fraxinus excelsior*, *Pinus sylvestris*, *Salix*, *Alnus glutinosa*, *Quercus*, *C. avellana*-type and *Betula*, suggesting wide scale deforestation from *ca.* 1.2 k cal. yrs. BP. This is mirrored by the NAP (Figure 6.12), which indicates a significant increase in open ground from *ca.* 1.1 k cal. yrs. BP and is supported by the increase in *Pteridium aquilinum* spores indicating increased light penetration to the forest floor. A predominantly arable economy is indicated (Figure 8.2), though the relative abundance of cereal pollen remained constant suggesting little change in the types of crops being grown. Anthropogenic activity at Llyn Pencarreg may also be indicated by the first significant increase in charcoal at the zone base.

A date of *ca.* 1.4 k cal. yrs. BP at the zone base falls within the 7th century AD when, regardless of the Viking invasion of Britain, the paucity of Scandinavian archaeology highlights the Vikings' failed attempts at colonising Wales (Schofield, 1999). The top of the zone, however, represents sediment accumulation during the early medieval period (*ca.* 0.8 cal. yrs. BP) when extensive woodland clearance is known to have taken place in South Wales to accommodate the increase in population (Seymour, 1985). An increase in upland farming was also encouraged by the warmer climate of the early Medieval period, which encouraged populations to move to higher ground where small farmsteads became the norm (Schofield, 1999).

From the mid-zone (*ca.* 1.2 k cal. yrs. BP; Figure 8.1b) the decrease in arboreal taxa described above is accompanied by an increase in *Hordeum* and the first record of *Cannabis sativa* pollen, reflecting the use of hemp during the medieval period for medicinal purposes and as source of fibre for canvas and rope (Mercuri *et al.*, 2002). Evidence for local *Cannabis* cultivation at Llyn Pencarreg compares well with other sites in Wales including Carningli (Seymour, 1985) and Llyn Mire (French and Moor, 1985) where high concentrations of *Cannabis* pollen in lake sediments suggest that lakes were often used for retting, which is known to cause eutrophication.

Medieval occupation within the Llyn Pencarreg district is represented by a wide range of archaeology that may provide evidence of clustered, agriculturally influenced, settlements protected by a regional defence network (Figure 8.3). The presence of several Granges (e.g. Mynachlog Glyn Du, 3.0 km east of Pencarreg and Rhuddlan Deifi, 5.0 km southwest), for example, suggests the existence of enclosed farming communities (Schofield, 1999). The excavation of water mills (e.g. at Rhyd Y Bont, 0.5 km southwest of Pencarreg and Rhyddlan, 2.5 km west) and fulling mills at Tirnewydd (3.5 km northwest) and Pandy (3.5 km southwest) and the discovery of silver coins at Lowtre (1.0 km northwest) and within the village of Pencarreg also suggest human presence from at least 1200 AD when the warmer climate enabled large areas of upland to be utilized for pasture and the semi-lowland and sheltered upland areas provided arable land for crops such as oats, corn and barley and the use of money for trading became more common (Thomas, 1975).

Additionally, the discovery of a network of Mottes around Pencarreg (e.g. Lampeter to the northeast, Castell Du to the north and Castell Santesau to the southwest) indicate that the area may have been tactically defended (Spurgeon, 1987); this may be supported by the presence of a tower-house at Pantscawen, 2.0 km northeast of Pencarreg. However, the Mottes are positioned along what are now major routeways, suggesting that they may have been travel outposts positioned along tracks that linked the individual small holdings within a larger farming community (Creighton, 2005). The presence of several medieval chapels (e.g. Capel Iago, 3.0 km east; Llanfechan, 2.0 km west), churches (e.g. Pencarreg and Lampeter, 3.5 km northeast) and holy wells (e.g. Ffynnon Fair, 0.5 km north; Ffynnon Bedr, 3.0 km northeast; Ffynnon Rhydderch, 3.5 km west) may have formed community focal points serving the small nucleated farming hamlets of the Pencarreg region (Schofield, 1999).

LPZ-6: 22 cm – 0 cm (*ca.* 0.2 – 0.0 k cal. yrs. BP)

Although environmental change between *ca.* 0.8 and 0.2 k cal. yrs. BP cannot be interpreted due to the missing sediments, Zone 6 provides strong evidence of enrichment at the zone base, followed by a recovery at the mid-zone and a final increase in productivity at the top of the core. Changes in the aquatic pollen record also appear to reflect shifts in nutrient dynamics (Figure 8.1a) and may highlight a transition from algal to plant domination (Scheffer, 2004). As enrichment decreases at the mid-zone, for example, aquatic plant pollen proportions display a significant increase, which continues to the top of the core as nutrient loads increase again. The corresponding increase in magnetic susceptibility and $CaCO_3$ in Figure 8.1b suggest that limnological change is associated with changes in land-use and the subsequent in-wash of mineral-rich soil from the lake catchment. Sediment accumulation rates are also at their highest for the entire core. As would be expected, these changes are accompanied by a decrease in organic carbon, increased rates of primary production ($\delta^{13}C$) and an increase in algal biomass (C/N).

Catchment disturbance is supported by a major decrease in the abundance of arboreal pollen (e.g. *Salix, Fagus sylvatica, Corylus avellana*-type, *Betula* and *Alnus glutinosa*) (Figure 6.9), a significant increase in open ground (Figure 8.2) and an increase in *P. aquilinum* spores. A general increase in the proportions of grass and herb pollen types (e.g. Poaceae, *Hordeum*-type, *P. lanceolata, Rosaceae acetosa* and *Trifolium*) indicates that tree removal is accompanied by an increase in cereal production and the expansion of meadow. Charcoal concentration also displays a minor increase towards the top of the core, which may reflect local population expansion over the last 200 years. Although national population figures have dropped steadily during this period, Pencarreg parish records provide evidence of a general population increase during the 19th century.

Unfortunately the missing sediments span the period between *ca.* 1200 and 1800 AD, so ecosystem status before 1800 AD cannot be examined. Considering that the chronology is somewhat uncertain, it is possible, however, that the inferred phase of enrichment at the zone base is partly a response to the impact of the construction of the Lampeter to Carmarthen railway line, which was built in 1866. Its extension to

Aberystwyth in 1867 provided a major communication link between the north and south of Wales until its closure in 1973 (Lampeter, 2005). Limnological impact during its construction and subsequent operation is extremely likely considering its proximity to Llyn Pencarreg and the use of coal fired engines. Over the last *ca.* 50 - 100 years nutrient status at Llyn Pencarreg has rapidly increased again possibly primed by previous impact and encouraged by the well-documented expansion of agricultural technology from the mid-20th century onwards.

8.5 Summary

At Llyn Pencarreg mesotrophic status was maintained for most of the 5,000 year sequence. Complex changes at the core base, however, included complete shifts in diatom assemblage composition at *ca.* 4.5 and at 4.2 k cal. yrs. BP. These changes are difficult interpret and may be associated with the limnetic changes of the Sub-Boreal or deforestation and changes in patterns of hydrology as the lake developed from what may have been a muddy woodland pool or stream with a high relative abundance of *P. boryanum* and surprisingly high diatom diversity, to a larger water body initially displaying a low diversity assemblage. Changes in agricultural activity from a predominantly transient pastoral economy to one dominated by arable farming were also reflected in the increase in open ground and presence of *P. lanceolata*. Mesotrophic status was established by *ca.* 4.1 k cal. yrs. BP and was marked by an increase in diatom diversity and a sustained increase in the introduction of new benthic taxa, which continued for the next 3,000 years and may represent a process of infilling at Llyn Pencarreg.

The diatom flora contained many acidophilic taxa, particularly *T. flocculosa*, which dominate the diatom assemblage and display decreases in relative abundance in correspondence with increases in DI-TP. Between *ca.* 3.1 and 2.7 k cal. yrs. BP, for example, a sustained decrease in the relative abundance of this species corresponded with changes in land-use during the Bronze Age. However, although catchment disturbance and limnological impact were highlighted, DI-TP remained stable and mesotrophic. The sustained increase in magnetic susceptibility from *ca.* 2.5 k cal yrs. BP followed a period of relative stability and corresponded with Iron Age activity within the Pencarreg district followed by Roman occupation within the catchment itself and the construction of a road through the village of Pencarreg. The decrease in productivity and increase in *T. flocculosa* followed by a peak in *P. boryanum* may have been associated with these changes, although limnological change was not highlighted by any of the remaining indicators.

During the medieval period human impact and significant land clearance led to a temporary increase to eutrophic status at *ca.* 1.0 k cal. yrs. BP, characterised by a peak in *A. granulata* and DI-TP and accompanied by an increase in ephippia and a decrease in diatom diversity. Local archaeology for this period is profuse and suggests a wide spread of nucleated settlements that were centred around several holy establishments with the development of a local economy based on a combination of pastoral and arable farming. The concurrent increase in deforestation and catchment erosion at Llyn Pencarreg during this period suggests that an increase in agricultural intensity was the cause of increased lake productivity. Unfortunately a significant gap in core sediments means that conditions between 1200 and 1800 AD could not be examined. However, though the chronology at the top of the core is tentative, ecosystem status at *ca.* 200 cal. yrs. BP - the EUWFD's choice of baseline conditions recommended for restoration – indicates nutrient enrichment with high algal diversity and increased productivity with signs of ecosystem stress. Although a significant degree of recovery from this latter state was reflected at the mid-zone of Zone 6, shifts in limnological conditions were complex and changes in land-use from *ca.* 0.1 k cal. yrs. BP once again appears to have influenced an increase in trophic status.

CHAPTER 9

RECONSTRUCTION OF THE LONG-TERM PALAEOLIMNOLOGY AND PALAEOECOLOGY OF UPPER TALLEY AND LLYN PENCARREG

9.1 Introduction

In Chapters 7 and 8, long-term palaeolimnological and palaeoecological change was reconstructed from a range of proxy indicators chosen to reflect ecosystem status at different trophic levels and changes in limnology due to external forcing (e.g. human impact and climate). The multiproxy data appear demonstrate marked differences in patterns of change in ecosystem status between Upper Talley and Llyn Pencarreg. Not all changes, however, were completely synchronous. This, on one hand, may indicate differences in the timing of response and, by implication, in the response thresholds of different proxy indicators in different parts of the system. However, on the other hand problems with the ability of the proxy indicators to realistically reflect past changes in ecosystem response may be demonstrated. This chapter examines the strength of the chosen proxy indicators to reconstruct long-term changes in ecosystem dynamics, identify the potential drivers of limnological change and to examine and compare the rates and magnitude of these changes at two sites with similar environmental characteristics. Ways in which the study could have been enhanced given sufficient time and finances are also discussed.

9.2 Palaeolimnological Interpretation of Long-Term Ecosystem Dynamics

The following section examines the ability of the selected indicators to effectively reconstruct shifts in ecosystem dynamics in response to changes in productivity. The discussion is divided into 4 sections; 'The Phytoplankton' (Diatoms, DI-TP and Other Algal Indicators), 'The Zooplankton', 'Aquatic Macrophytes' and 'Carbon and Nitrogen Isotope Ratios'. The strengths and weaknesses of each indicator are assessed and evaluated in terms of their ability to represent shifts in nutrient status and corresponding changes at each trophic level.

9.2.1 The Phytoplankton

9.2.1.1 Diatoms

During the process of eutrophication it is usual for the diatom population to respond to shifts in water quality. Although the dominant indicator species may vary from site to site, there is often a significant degree of predictability in the given shifts in species assemblage composition. Several authors have discussed the association between lake trophic status and expected sequences of algal production (Hutchinson, 1967; Moss 1988; Wetzel, 2001; Scheffer, 2004). These studies show that recognition of these patterns in the palaeolimnological record are of great value in locating the natural, pre-enriched, baseline state prior to the onset of nutrient enrichment, and also in defining a state of impact prior to recent acceleration.

In their discussion of changes in the diatom assemblage composition observed at Ammersee and Starnberger See, Germany, Alefs and Müller (1999) noted that several Alpine foreland lakes displayed similar sequences in response to nutrient loading (e.g. Lake Holzmaar and Chiemsee, Germany; Soppensee and Zürichsee, Switzerland). In each case the dominant species formed what Alefs and Müller (1999) referred to as the '*basic species sequence of eutrophication*' where four stages of eutrophication were characterised by shifts in the relative abundance of planktonic species indicative of certain degrees of P enrichment. Due to their predictive ability during eutrophication, Philbert and Prairie (2002) also suggest the assessment of changes in the dominant open-water planktonic taxa to obtain clear inference of nutrient dynamics. Although lake fertility is not quantified by these associations, these studies show that sequences of species presence often track changes in productivity well, providing information on changing lake status and a reliable warning of changes in trophic status.

Recognition of similar eutrophication patterns at the study sites helped to identify the different stages of nutrient enrichment and may enable between-site comparisons of temporal variation in the dominant eutrophic species, ultimately enabling the location of baseline states. At Upper Talley changes in diatom species assemblage composition during long-term eutrophication displayed a pattern of marked fluctuations in response to shifts in nutrient concentrations (Table 9.1). Figure 9.1 shows that at the onset of eutrophication (Stage 1) there was a marked increase in *C. stelligera* accompanied by a decrease in benthic taxa and diatom diversity. Stage 2 indicates a reduction in species diversity to planktonic taxa able to withstand high nutrient loads and increased turbidity (*A. granulata* and *S. minutulus*).

Stage of Sequence	**Diatom Dynamics; Upper Talley**	**Diatom Dynamics; Llyn Pencarreg**	**Other Significant Features**
1 Increase in Eutrophication	• Increase in *C. stellligera* • Increase in *C. meneghiniana* in modern pollution		• Increase in *Pediastrum*
2 Height of Eutrophication	•Dominance by taxa able to **withstand** high nutrient loads (*A. granulata* and *S. minutulus*) • Decrease/ loss of *C. stelligera*	• Peak in *A. granulata* • Temporary appearance of *C. stelligera*	•Presence of ephippia • Low diversity of benthic taxa
3 Decrease in Eutrophication	• Increase in C. stelligera •Decrease in eutrophic taxa		
4 Clear Water Phase	• Decrease in *C. stelligera*		High diversity of benthic taxa particularly small *Fragilaria* and *Navicula* spp.

Table 9.1 Table showing the inferred4-stage sequence of eutrophication at Upper Talley and Llyn Pencarreg highlighting changes in diatom assemblage composition and other proxy indicators

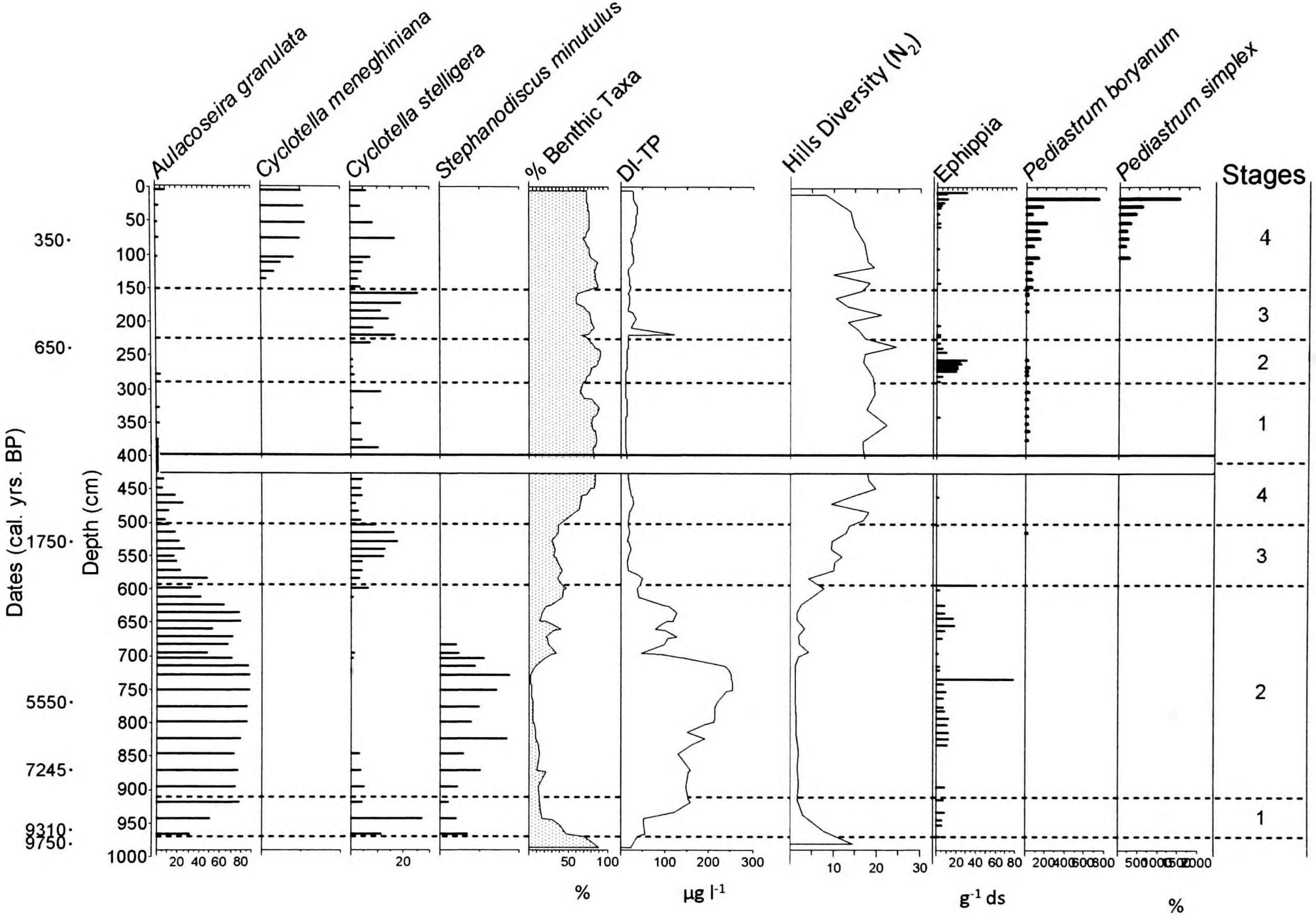


Figure 9.1 Diagram showing the inferred 4-stage sequence of eutrophication at Upper Talley.

The subsequent reappearance of *C. stelligera* and increase in benthic taxa (Stage 3) suggest improvements in water quality. Finally, Stage 4 shows the loss of *A. granulata* and a sustained increase in benthic taxa reflecting clear water, which is indicative of good ecosystem health.

During the medieval period the changes associated with Stage 2 (the height of enrichment), were not exhibited and instead benthic species increased in relative abundance from *ca.* 1.2 k cal. yrs. BP. This may be an alternate response to nutrient enrichment although lake-level decrease in freshwater lakes can also cause an increase in the relative abundance of benthic taxa (Round, 1964). However, the diatom assemblage at this point of the core is characterised by planktonic taxa usually associated with a more eutrophic assemblage (Reynolds, 1973). Following medieval impact, Stage 4 of the sequence is characterised by the first appearance of *C. meneghiniana* indicating poor water quality and possibly reflecting a change in the type of polluting activities in the lake catchments (El-Bestawy, 2000).

In contrast, limnological change in response to human impact at Llyn Pencarreg was marked by an increase in *A. granulata* and the brief appearance of *C. stelligera* at the height of enrichment (Table 9.1). The sequence in general is characterised by gradual, long-term trends of change; for example, the change in relative abundance of *A. granulata* in response to changing nutrient status during the Bronze Age was slow and prolonged (Figure 8.1a). Medieval and modern (i.e. last 150 – 200 years) enrichment produced much sharper changes in response to nutrient dynamics, although this may in part be a consequence of a natural reduction in water depth by lake infilling; the steady increase in benthic taxa and reduction in *T. flocculosa* are consistent with the effects of lake infilling, which may have increased the sensitivity of biotic communities to environmental change as the lake became shallower (Moss *et al.*, 1997). Alternately it is possible that resilience pathways may have been eroded, which has been observed in deep lakes as a result of the fewer functional groups (Carpenter *et al.*, 1985; Scheffer *et al.*, 2001).

Despite the variations in response to catchment impact at Upper Talley and Llyn Pencarreg, *A. granulata* and *C. stelligera* appear to be the strongest indicators of increased nutrient enrichment. The difference in behaviour of *C. stelligera* between the two sites (i.e. at Upper Talley it appears either side of the height of enrichment and at Llyn Pencarreg at the height of enrichment) may be associated with its preference for meso-eutrophic conditions and the relatively lower nutrient status of Llyn Pencarreg.

9.2.1.2 DI-TP

The predictive ability of the Combined European diatom-based total phosphorus transfer function has been validated for a range of lakes and climates across Europe (Battarbee *et al.*, 2000). Although inferences of nutrient status are strengthened by the reconstruction of quantitative nutrient dynamics, in this study the dominant taxa were common, though morphologically variable and are thus merged taxonomically in the EDDI training set, resulting in broad tolerance ranges. This is reflected in the sample specific errors of prediction and confidence intervals estimated by bootstrapped coefficient of determination. For Upper Talley, an RMSE $_{\text{boot}}$ value of

0.27 and a RMSE $_{boot}$ value of 0.34 at Llyn Pencarreg indicate that fluctuations in DI-TP are not statistically significant and although the dataset includes classic eutrophic species, statistically, they are not strong indicator taxa (e.g. *A. granulata* $N_2 = 32.3$; Tol = 0.43 $\log_{10}$ µg l^{-1}; *C. ocellata* $N_2 = 29.8$; Tol = 0.36 $\log_{10}$ µg l^{-1}; *C.stelligera* $N_2 = 24.7$; Tol = 0.35 $\log_{10}$ µg l^{-1}; *F. pinnata* $N_2 = 53.9$; Tol = 0.50 $\log_{10}$ µg l^{-1}; *T. flocculosa* N2 = 29.1; Tol = 0.42 log10 µg l^{-1}). The results are therefore presented, but interpreted with caution.

The increase in small *Fragilaria* and *Navicula* spp., which occurred from *ca.* 1.2 k cal. yrs. BP at Upper Talley and *ca.* 3.1 k cal. yrs. BP at Llyn Pencarreg (Figure 9.2), may also have affected the reliability of DI-TP reconstruction. Recent research suggests that assemblages with a high ratio of small benthic *Fragilaria* and/or *Navicula* may not be totally appropriate for DI-TP reconstruction as their affinity with benthic habitats and increased photosynthesis at the lakebed may reduce the reliability of the calibration (Bennion, 1994; Sayer, 2001; Philbert and Prairie, 2002). Empirical evidence from Kitner and Pouliěková (2003), for example, indicates that modern diatom samples from a range of benthic substrates produce different estimates of trophic status in the same water body. The presence of small *Fragilaria* and *Navicula* spp. with low TP optima may thus have caused an under estimation of DI-TP (Figure 9.2). However, omitting them from the transfer function, as suggested by Sayer (2001) and Bennion (1994), was not feasible as the high percentage (maximum; *ca.* 50%) would mean recounting slides to provide a statistically reliable estimate of species composition and the time limitations of the study did not allow for this.

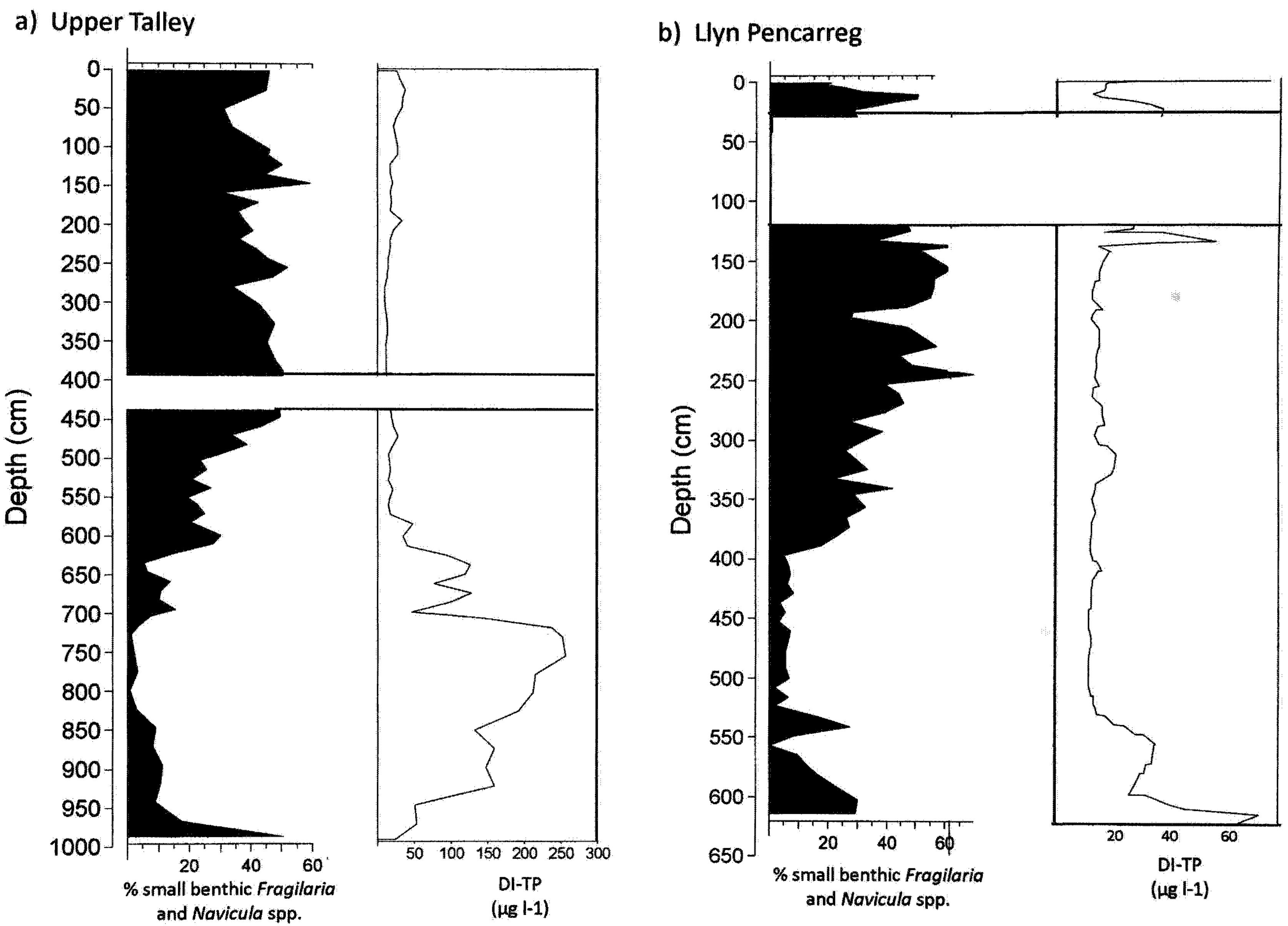


Figure 9.2 Diagram showing the relative abundance of small benthic *Fragilaria* and *Navicula* spp. (as a % of the total diatom sum) in relation to changing DI-TP at a) Upper Talley and b) Llyn Pencarreg.

The presence of a wide range of benthic taxa has, however, provided useful qualitative proxy data on changes in substrate/habitat availability, water clarity and the relative abundance of plants during the different stages of eutrophication (Scheffer, 2004). This has been explored in more detail by Stenger-Kovács *et al.* (2007) who developed the idea of using littoral and benthic diatoms to reflect patterns of substrate availability as an indication of ecosystem health. Sayer (2001) and Bennion *et al.* (2005) expand on this idea by suggesting the construction of transfer functions to reconstruct shifts in habitat availability. When used in conjunction with a nutrient reconstruction, the changes in benthic taxa tend to compensate for the lack of reliability in DI-TP and may help locate natural baseline states in terms of habitat availability and ecosystem health.

9.2.1.3 Other Phytoplanktonic Indicators

Although *P. boryanum* colonies display a cosmopolitan relationship with water chemistry parameters (Komárek and Jankovská, 2001), at Upper Talley and Llyn Pencarreg the presence and absence of colonies provided additional proxy data for phytoplankton dynamics and competition in the wider algal community. Whilst Upper Talley emphasises its general lack of ability to predict eutrophic conditions (Figure 7.1a), Llyn Pencarreg displays changes in the abundance of *P. boryanum* in response to eutrophication by the increasing immediately prior to increases in *A. granulata* (Figure 8.1a). As with other sites in Europe, modern eutrophication has produced a significant increase in *P. boryanum* colonies (and more recently by *P. simplex* at Upper Talley) possibly due to the combined influence of changes in climate and human impact (Komárek and Jankovská, 2001).

Diatoms and *Pediastrum* preserve well in lake sediments, but form only part of the algal community. The high diversity of phytoplankton in a healthy lake can encourage competition for nutrients and light availability so that even the baseline state is variable and changes in algal populations are not easy to predict without the complete long-term record (Cottingham, 1996; Reynolds, 1971). During the height of eutrophication the loss of light and increase in turbidity, which appears to have encouraged increases in *A. granulata* at the study sites, can also cause blooms of filamentous, toxic, blue-green algae (eg., *Oscillatoria, Anacystis*) accompanied by poor water quality and deterioration of ecosystem status (Dokulil and Skolout, 1986). Scheffer (2004) refers to this phase as the third alternate stable state and suggests that the threshold between algal dominance and cyanobacteria occurs at different nutrient levels, depending on individual lake characteristics.

The analysis of fossil pigments, which uses carotenoids (eg., myxoxanthophyll, aphanizophyll and oscillaxanthin) as a proxy indicator of the presence of cyanobacteria (Hall *et al.*, 1997; Lotter, 2001), would have allowed a more complete assessment of long-term phytoplankton dynamics. It may have strengthened reconstruction of shifts in the dominant plant groups that can take place at the height of enrichment. In addition, low DI-TP estimates at times of inferred enrichment may also have been explained, particularly during the medieval period at Upper Talley when diatom response to nutrient enrichment was subtle and the peak in DI-TP was delayed when compared to changes in the other proxy indicators. Charophyte analysis also records changes in community structure during enrichment episodes by reflecting changes in benthic and pelagic habitats (McGowan *et al.*, 2005). Although Charophytes are filamentous green algae, they are more closely related to the Plantae and tend to reflect the behaviour of aquatic macrophytes (Graham, 1993). Examination of changes in the relative abundance of Charophytes may thus have provided the present study with an additional tool with which to assess shifts in the wider algal community during phases of enrichment.

As an indication of limnological change charaphytes may be replaced by *Zannichellia palustris, Callitriche truncata* and species of *Potamogeton* during phases of catchment impact (Davidson *et al.*, 2005). Additionally, the appearance of Charophyte meadows at the onset of lake enrichment can provide a nutrient trap. This can have a positive effect on water clarity by reducing nutrient loads and turbidity, whilst delivering oxygen to the sediments via their roots, thereby enhancing denitrification and preventing the release of iron bound phosphorus to the epilimnion (Van den Berg *et al.*, 1999; Kufel and Kufel, 2002). Coops (2002) has shown that this process is usually reversed at the height of enrichment as Charophyte abundance decreases in line with the loss of aquatic macrophytes.

9.2.2 The Zooplankton

At both sites the presence of Cladocera egg sacs (ephippia) correlated with diatom inferred phases of increased nutrient status, producing clear evidence of ecosystem stress during phases of eutrophication (Table 9.1). At Upper Talley each phase of inferred eutrophication produced Cladocera ephippia (Figure 7.1a), with the additional presence of the double-yolked variety occurring during the initial phase of long-term, natural enrichment (between *ca.* 9.0 and 2.0 k cal. yrs. BP), supporting the inference that this was a period of intense ecosystem stress. Cladocera ephippia were absent from most of the Llyn Pencarreg sequence, but made an appearance during the medieval period and although a gap in the core prevented continuous analysis, egg sacs were still present above the gap (*ca.* 0.2 k cal. yrs. BP) (Figure 8.1a). This suggests either long-term ecosystem disturbance or a second shift in ecosystem dynamics following medieval impact, and indicates changes in community structure as an indirect response to changes in nutrient status.

The presence of double-yolked ephippia at Upper Talley appears to be particularly significant; Fryer and Frey (1981) suggest that the production of ephippia with double yolks reflects environmental instability when evolutionary trade-off can shift from K- to R-selection (Chapman and Reiss, 1997). During K-selection a single egg is produced in a watertight carapace to protect it against predation during diapause. Although the protective covering of the two-yolked ephippium tends to gape at the anterior end offering less protection, the provision of more yolk increases the chances of species survival. Double-yolked ephippia, identified to species level, have also been observed in lakes of the English Lake District, the Yorkshire Pennines and North America (*Daphnia thorata, Chydorus ovalis* and *C. sphaericus*; Fryer and Frey, 1981), South America (*Camprocerus macrurus*; Weismann, 1977) and the Isle of Rhum (*Daphnia longispina*; Fryer and Frey, 1981). In each case their presence indicate a phase of ecosystem stress.

In this study the use of Cladocera ephippia provided a quick and useful tool that helped to identify phases of stress and contribute to the understanding of ecosystem status at a higher trophic level during phases of nutrient enrichment. Studies show, however, that the appearance of ephippia is not simply a direct cause and effect result of eutrophication. McCarthy *et al.* (2006), for example, have shown that although the ephippia of some Cladocera do respond to changes in P concentrations, *Daphnia* ephippia are particularly sensitive to changes in the availability of N, and their presence may be linked to the dominance of inedible cyanophytes during phases of low N. This suggests that different species of Cladocera may respond to different ecological parameters. Although the present study was restricted to rapid analysis of changes in abundance, identification to species level may have provided the study with additional information on changes in ecosystem dynamics. Although in its early stages, work by Sarmaja-Korjonen (1999; 2003; 2004) shows that when certain habitats are under stress specific members of the Cladocera community will produce resting eggs. At Rutikka Lake in Finland (Sarmaja-Korjonen, 1999), for example, a threshold was crossed, which correlated with changes in prehistoric cultivation, and gave rise to an increase in the ephippia of just one species, *Alona affinis* (suggesting ecosystem change in the benthic and littoral zones; Hann, 1990).

The study of ecosystem dynamics would also have been improved by incorporating the analysis of Cladocera species composition. Their chitinous body parts (e.g., head shields, claws, post-abdomen) can usually be identified to species level (Hann, 1990), and analysis provides useful information on changes in the Cladoceran community. As keystone species their loss can lead to shifts in predator-prey relationships throughout the food chain (Frey, 1960; Whiteside, 1970; Crisman, 1980; Kerfoot, 1981; Brugham and Speziale, 1983). Their sensitivity to limnological change means that shifts within the Cladoceran community can be used as a tool to accurately reconstruct changes in habitat availability during eutrophication; the loss of aquatic macrophytes that can accompany increases in trophic status, for example, often results in the replacement of *Bosmina longispina* with *B. Longirostris* (Crisman and Whitehead, 1978).

However, as noted in the methods of the present study (Section 4.2.8) the use of Cladocera was unfeasible due to time constraints and the high organic content of core sediments. Studies by Simpson (2000, 2001) at acid lakes in Scotland met with more success, probably due to the lower sedimentary organic content.

9.2.3 Aquatic Macrophyte Pollen

Changes in macrophyte abundance can reflect both changes in lake depth, which controls the size of the littoral zone (Rasmussen and Anderson, 2005), and shifts in productivity status due to competition for nutrients between the major plant groups (i.e. macrophytes and algae) (Sheffer *et al.*, 2002). In the present study the aquatic pollen sum (APS), as a proxy for aquatic macrophyte community composition, was compared to changes in diatom concentrations (DC) in order to examine their strength as indicators for shifts in alternate states by representing changes in dominance between the two plant groups. The results suggest that apart from high DC values and low APS abundance during the medieval period at Upper Talley (Figure 5.10) and the replacement of high APS with high DC during the Bronze Age at Llyn Pencarreg (Figure 6.10), DC showed a general lack of change during phases of inferred nutrient enrichment. Changes in the abundance of aquatic pollen and spores, on the other hand, generally provide good representation of shifts in the aquatic macrophyte community during phases of external impact at both Upper Talley and Llyn Pencarreg. Again, the analysis of fossil pigments could be used to examine changes in the wider algal community and shifts between algal and plant domination could also be assessed.

The approach could also be tested more rigorously by analysis of plant macrofossils in the sediment record. Birks and Birks (2000) suggest that all pollen studies should be supplemented by macrofossil analysis to overcome the problems associated with limited taxonomic resolution, particularly for aquatic taxa that tend to reproduce vegetatively (asexually). Plant macrofossils display great potential for reconstructing community dynamics (Davidson *et al.*, 2005). Zhao *et al.* (2006) found that remains were dispersed unevenly with high concentrations close to source, particularly *Potamogeton* seeds. This provided an additional spatial estimation of the dominant taxa and information on species richness, which could be used to map changes at different points of the lake throughout the enrichment process.

Both terrestrial and aquatic macrofossils were rare in the Llyn Pencarreg sediments though *Potamogeton* seeds were common at Upper Talley (Table 4.2). Their presence correlated with peaks in aquatic pollen abundance, which supports the reliability of the pollen record, and also as an additional indicator of aquatic plant abundance and strengthens evidence for aquatic plant succession. This approach can also be used to locate the baseline vegetation before human impact, which can then be linked to lake restoration plans. Although other unidentified aquatic macrofossils were present sporadically at low abundance they were not quantified and analysed as an indicator of changes in vegetation, again due to lack of time. In studies where this method is adopted, the picking and analysis of Charaphyte oospores could be incorporated as an additional indicator of changes in ecosystem status.

9.2.4 Carbon and Nitrogen Isotope Ratios

At Upper Talley and Llyn Pencarreg changes in the carbon isotope ratio correlate with some phases of enrichment, though changing values often appear to be in response to factors other than rates of primary production. At Upper Talley, for example, the phase of long-term enrichment between *ca.* 9.0 k and 2.0 k cal. yrs. BP, characterised by the domination by *A. granulata*, high DI-TP, low diatom diversity and the presence of Cladocera ephippia, is accompanied by a contrasting decrease in $\delta^{13}C$ values implying low primary production, whilst C/N indicate a marked increase in terrestrial in-wash as opposed to algal biomass (Figure 9.3). Llyn Pencarreg also shows that the carbon isotope profile did not always change in parallel with eutrophication inferred from *A. granulata* and *C. stelligera* (Figure 9.4).

The possibility of the significant influence of environmental variables other than TP (eg. pH, temperature, respiration, anoxia) suggests that $\delta^{13}C$ may sometimes not be entirely appropriate for assessing trophic conditions (Meyers and Teranes, 2001) and the location of pre-impact baseline states using $\delta^{13}C$ analysis is therefore questionable. At several of the Florida Lakes, Gu *et al.* (1996) and Brenner *et al.* (1999) found that the pre-impact $\delta^{13}C$ values of sedimentary organic matter displayed notable between-lake differences, even within a small geographical area. Bernasconi *et al.* (1997) suggest that the range of differences may be due to between lake variations in the isotopic composition of primary producers during the annual cycle and is probably related to the complexity of the carbon cycle in aquatic systems. They also suggest that pre-impact $\delta^{13}C$ values may control the extent of change with future episodes of nutrient enrichment.

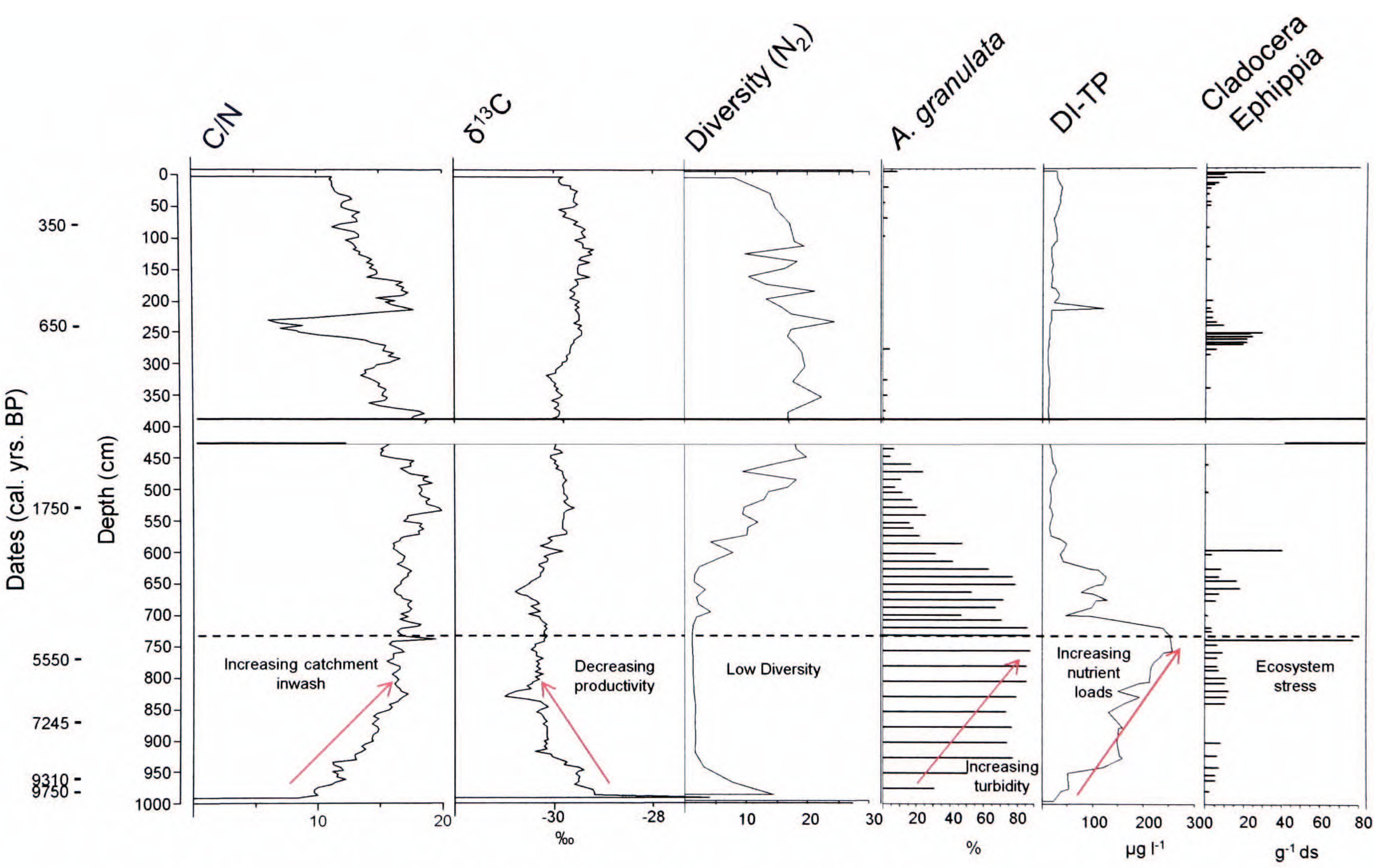


Figure 9.3 Diagram showing the failure of C/N and $\delta^{13}C$ to measure changes in trophic status at Upper Talley

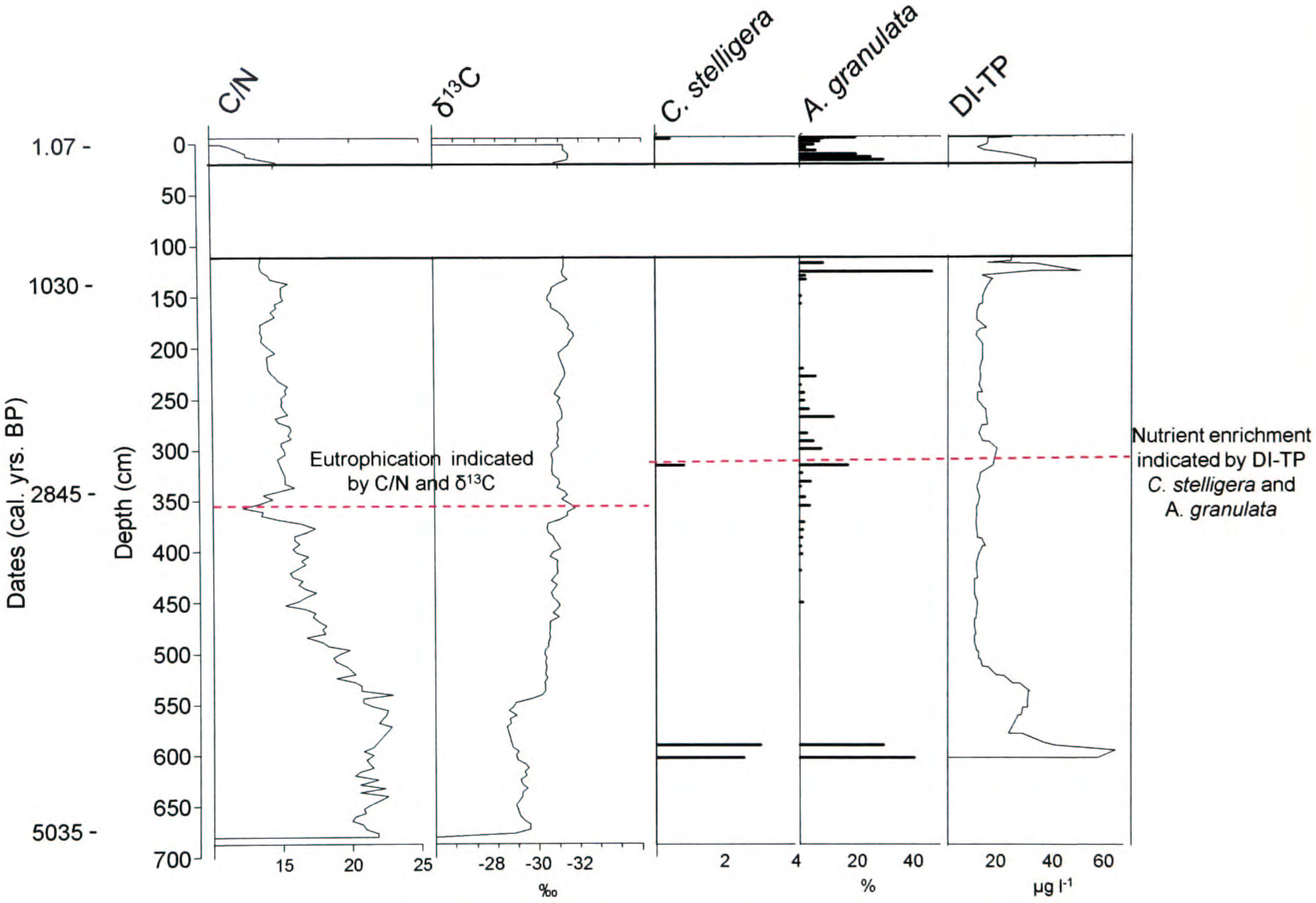


Figure 9.4 Diagram showing the failure of C/N and $\delta^{13}C$ to measure changes in trophic status at Llyn Pencarreg

Thus shifts in primary production may not be a true reflection of the magnitude of enrichment. Although a reliable indicator for changes in the source of organic matter during phases of human impact, the C/N signal lacks a clear response to high algal biomass during phases of eutrophication when catchment inwash is also high. Reasons for this are discussed in Section 7.1. As an alternative, C/N analysis in lakes where terrestrial inwash is high shows a strong potential for examining changes in the dominant terrestrial vegetation by inferring the composition of lacustrine organic matter (Lamb *et al.*, 2004).

9.3 Palaeoecological Interpretation of Long-Term Human Impact

The following section examines the ability of the selected indicators to effectively reconstruct changes in the external factors that may be responsible for initiating limnological change. The discussion is divided into 4 sections; 'Pollen', 'Charcoal', 'Archaeology' and 'Other Potential Drivers'. As with the limnological indicators, strengths and weaknesses are assessed and evaluated in terms of their ability to identify long-term changes in catchment conditions and, when compared to the limnological data, can help to determine the potential cause of corresponding changes in lake status.

9.3.1 Pollen

The use of a single core for all proxy analyses allows direct comparison of the pollen record with biological indicators to assess the causes of limnological change with a high degree of certainty (Birks *et al.*, 2000). Figures 9.5 and 9.6 show that changes in arboreal and herb pollen taxa provide representation of changes in catchment vegetation in response to human activity at Upper Talley and Llyn Pencarreg. Major deforestation phases during the Bronze Age, medieval period and the last 200 years were characterised by tree clearance and an increase in the types of grass and herb pollen associated with open ground (e.g. Poaceae, *Plantago lanceolata*, *Pteridium aquilinum*) and arable farming (e.g *Rumex acetosa, Hordeum*). These changes in vegetation patterns compare well with other pollen studies in Wales (e.g. Thomas, 1965; Chambers, 1983; Chambers, 1999; Lillie *et al.*, 2000).

Additionally, the Non-Arboreal Pollen (NAP) curve has provided the study with a summary of changes in the pollen record that represent changes in the availability of open land (Figures 7.1a and 8.1a). This provides a simple basis for comparing the timing and magnitude of land clearance in the lake catchments and evaluating the relationship between human impact and the extent of limnological change. Although work by Sugita *et al.* (1999) has shown that in different regions of Europe catchments with the same amount of open ground in the source area produce different NAP signals due to differences in the contribution of background pollen, the pollen signal is generally distance weighed and vegetation in the immediate area, close to the coring site (Sugita, 1993), will usually contribute more pollen to the record than vegetation from the wider landscape.

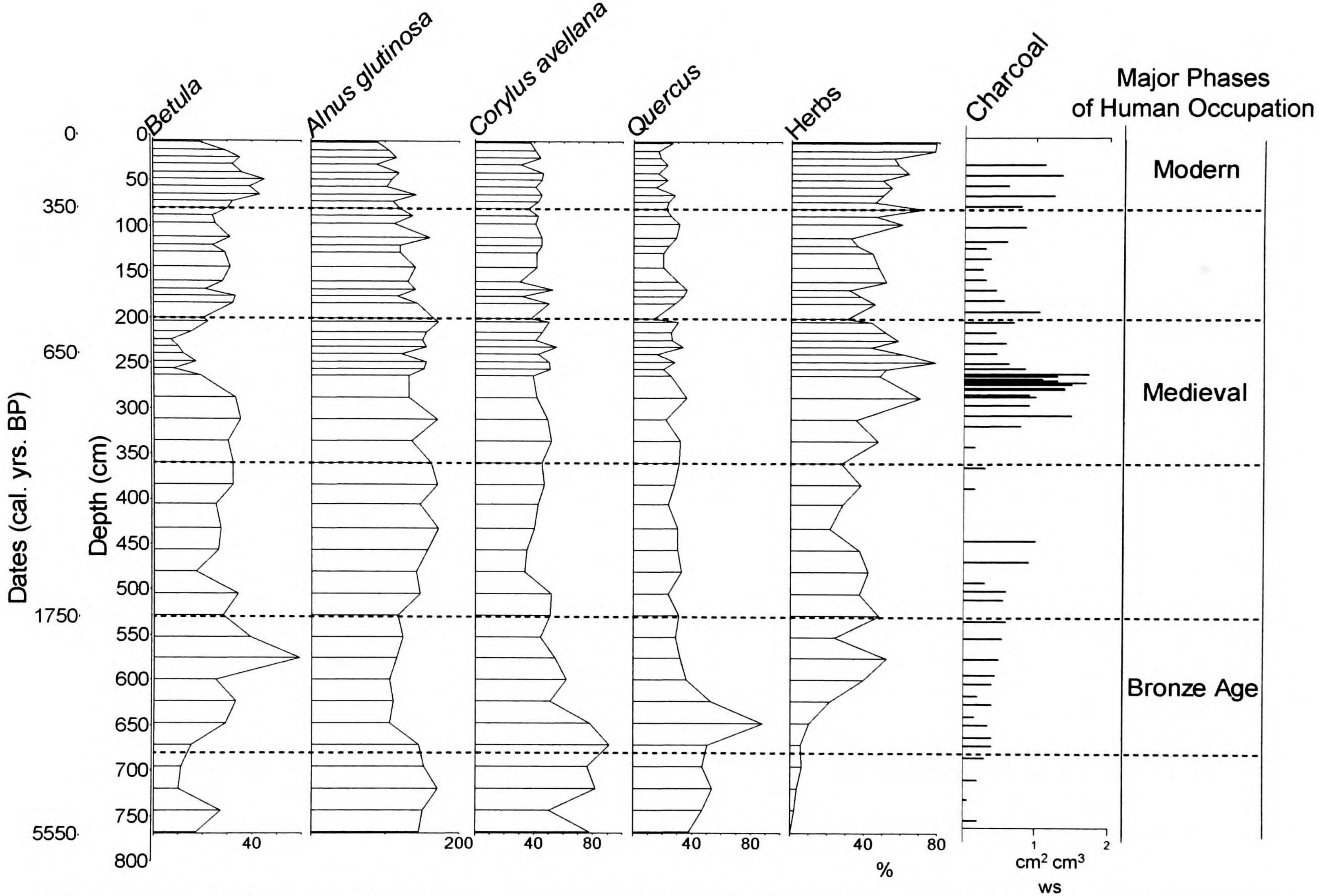


Figure 9.5 Diagram showing the changes in relative abundance of pollen taxa and charcoal during documented phases of human occupation in the Upper Talley region

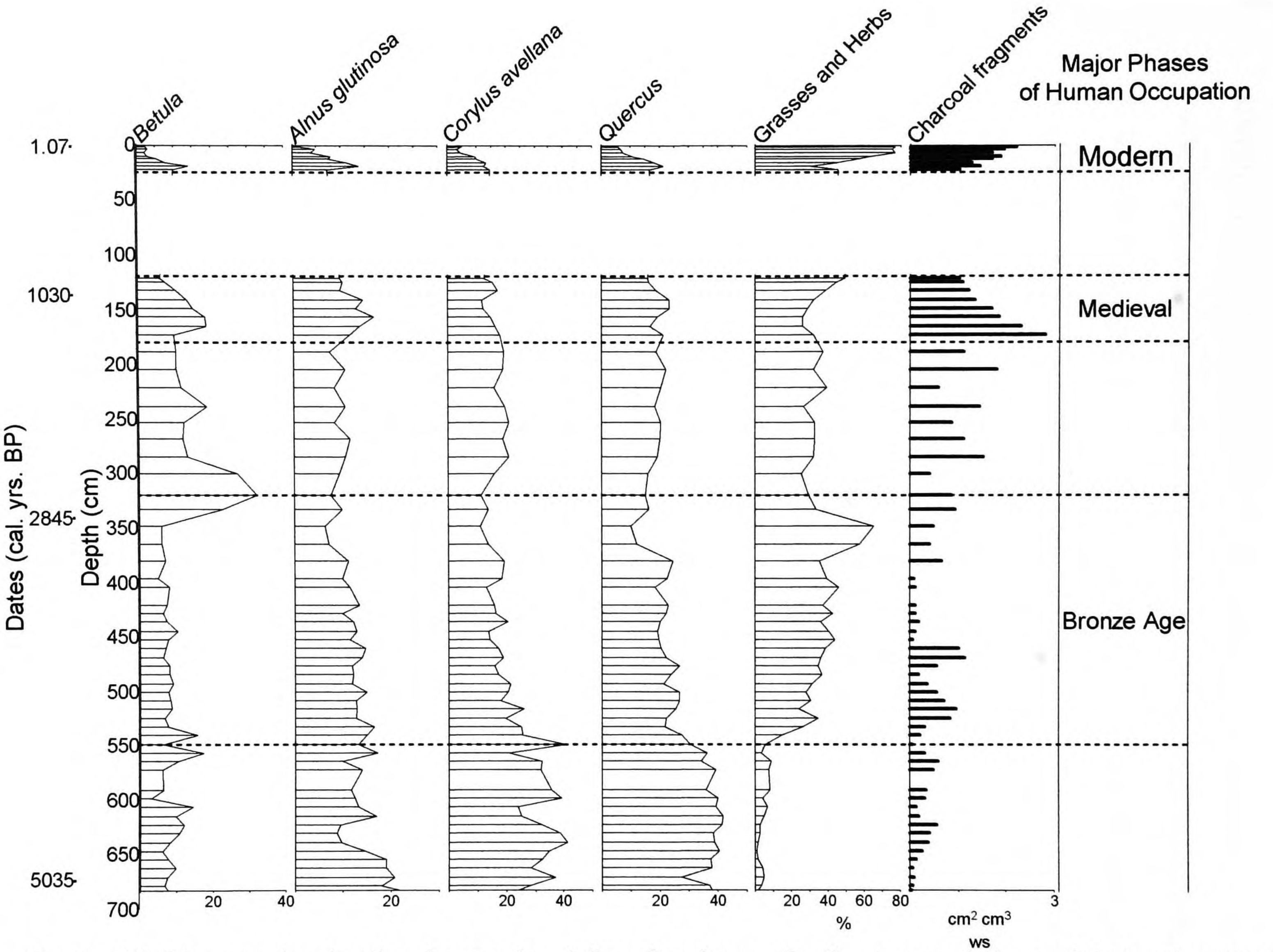


Figure 9.6 Diagram showing the changes in relative abundance of pollen taxa and charcoal during documented phases of human occupation in the Llyn Pencarreg region

As discussed in Chapter 3, Upper Talley and Llyn Pencarreg are situated in the same climatic and topographic region and it can therefore be assumed that background conditions are consistent and between-site differences in NAP reflect local differences in the extent of catchment land-clearance. Subject to similar land-use practices both sites also show evidence for major deforestation during the Bronze Age, medieval period and more recently (*ca.* past 400 years). Most importantly, in spite of showing a subdued pattern of eutrophication, the NAP data suggests that it is Llyn Pencarreg rather than Upper Talley, which shows the higher degree of human impact.

The Arable/Pastoral Index (API; Turner, 1964) (Figures 7.2 and 8.2) provides a simple measure of the character of human impact within a lake catchment that compliments the NAP data by characterising the type of land-use change. This allows for the comparison of phases of land clearance with types of agricultural land-use and an overall comparison of the degree of human impact between sites, thus producing a more detailed evaluation of human activity in the lake catchments. Upper Talley and Llyn Pencarreg have produced long-term records of broadly similar temporal changes in the type of agricultural activity taking place in the lake catchments. Prior to the onset of major deforestation at both sites API values suggest a mixed agricultural economy, though significant fluctuations at Llyn Pencarreg reflect a higher degree of change during this period. At both sites the most significant shift in the API occurred during the Bronze Age, when an increase in open land was characterised by values that suggest a gradual though major transition to a predominantly arable farming regime during this period. Although NAP at both sites indicates intensified land clearance during the medieval period and in the last *ca.* 400 years, API suggests that since *ca.* 0.2 k cal. yrs. BP at Upper Talley and *ca.* 0.1 k cal. yrs. BP at Llyn Pencarreg agricultural land-use has remained predominantly arable.

This is contradicted by the current status of land-use at Upper Talley where intense sheep grazing takes place on the grassy banks that run directly to the water edge. This has continued since at least the early nineteenth century when Lewis (1833) noted a substantial portion of common land was used for grazing sheep around the Talley Lakes. It is possible however, that the pollen types used in the API (Turner, 1964) do not fully represent the finer grasses and weed taxa found in modern improved pasture, thus underestimating the importance of pastoral farming. This would indicate that the herb spectrum needs to be studied in more detail to examine the possible differences in agricultural taxa between now and then (Broström *et al.*, 2004). In spite of this, when examined in combination with the NAP curve the API provides strong evidence for changes in land-use that correspond with shifts in ecological status.

9.3.2 Charcoal

Charcoal analysis provides an additional proxy indicator for human occupation within the lake catchments. Clark (1988) suggests that as with pollen analysis, it can be assumed that the background signal is constant and that under natural conditions lakes within the same climatic and topographic region will receive similar amounts of charcoal. As it is well established that Upper Talley and Llyn Pencarreg are in similar catchments, this assumption again allows for the comparison and evaluation of charcoal particles during periods of increased human activity.

At both sites increases in charcoal correspond with deforestation and a general increase in grasses and herbs during the Bronze Age, medieval period and more recently (past *ca.* 400 years) (Figures 9.5 and 9.6). Although Hörnberg *et al.* (2006) found that the subtle effect of hunter-gatherer populations on changing vegetation patterns through the use of fire was difficult to evaluate due to the re-deposition of sedimentary material, some studies do indicate large peaks in the charcoal data as a result of forest burning during the Bronze and Iron Ages (Wilcox, 1974; Millspaugh and Whitlock, 1995; Pitkänen, 1999).

At Upper Talley and Llyn Pencarreg however, the smooth curves and low values of the charcoal data, compared to those produced by natural fires or slash and burn farming, are more likely to indicate the small scale burning of lakeside or wetland vegetation (as recorded in a recent high resolution study of Mesolithic sediments from County Cork, south-west Ireland; Mighall *et al.*, 2008), or the burning of wood by increased human habitation within the lake catchments (e.g. Burden *et al.*, 1986; Tsukada *et al.*, 1986). It is assumed, therefore, that the comparison of charcoal abundance at the study sites represents differences in the degree of human occupation. Apart from during the Bronze Age the charcoal record is in general agreement with the pollen data and NAP curve, indicating that human occupation has generally been more intense at Llyn Pencarreg.

9.3.3 Archaeology

The presence of archaeological structures and artefacts was used as an additional indicator of anthropogenic impact both around Upper Talley and Llyn Pencarreg and within the lake catchments. Although this provided the study with the opportunity to spatially examine the presence of archaeology for each major archaeological period (Figure 3.4), knowledge of ancient settlements and changes to their immediate environment is still severely limited; the evidence found is always less than has actually survived, which in turn is far less than the total amount of evidence existing during the time of deposition (Barker, 1993). The interpretation of evidence in relation to the existence of prehistoric settlements was therefore extremely cautious and the presence of prehistoric axes, standing stones or funerary barrows was not applied significantly in the evaluation of human *occupation*. Nevertheless, the discovery of archaeology within a 5km radius of Upper Talley and Llyn Pencarreg does provide strong evidence for the *presence* of humans during the Neolithic, Bronze Age, Iron Age, Roman occupation and medieval period.

When the archaeological record is compared to changes in the abundance of pollen and charcoal, the occurrence of both Bronze Age and Medieval archaeology in the Upper Talley and Llyn Pencarreg region corresponds with major changes in the pollen record. These changes include a significant increase in the amount of open land, an increase in deforestation and the appearance of agricultural indicators, suggesting major changes to the landscape. Although the use of this technique to reinforce the case for human impact was tentative in the present study, at sites in northwestern France Gaudin *et al.* (2006) showed that the method was successful when the spatial analysis of archaeology using GIS was compared to 218 regional pollen datasets. In this respect, not only did the presence of ancient structures and artefacts indicate human occupation, but the pollen data showed that agricultural activity was confined to the lowlands whereas the upland areas were utilised for grazing.

9.4 The Influence of Other Variables

9.4.1 Climate

Although wet and dry palaeoclimatic phases are clearly seen in raised bogs throughout upland Britain (e.g. Ellis and Tallis, 2001; Barber *et al.*, 2000; 2003; Hughes and Barber, 2003; Dark, 2006), the complexity of attempting to disentangle the more subtle influence of climate during phases of limnological change from human impact is problematic. Some studies, however, show that there may be an influence of climate on lake levels in the northern hemisphere as a result of changes in precipitation regimes (Battarbee *et al.*, 2002; Hinzman *et al.*, 2005). By comparing the limnological and pollen data to the Blytt-Sernander model of climate change in northern Europe, the present study was able to produce a broad summary of the potential influence of climate on inferred changes in lake depth at Upper Talley and Llyn Pencarreg.

Application of this technique to the core data enabled changes in climate to be evaluated as a potential cause of limnological change. The inferred increase in lake depth at Upper Talley at *ca.* 2.3 k cal. yrs. BP (e.g. a decrease in DI-TP, an increase in planktonic *Cyclotella* and a decrease in aquatic macrophytes), for example, falls at the beginning of the wet Sub-Atlantic; a markedly wet period characterised by a sudden rise in lake levels (Issar, 2003). At Llyn Pencarreg, on the other hand, the sharp inferred increase in lake depth at *ca.* 4.5 k cal. yrs. BP falls during the Sub-Boreal when dry conditions between *ca.* 5.0 and 2.9 k cal yrs BP led to a phase of exceptionally low water levels between *ca.* 4.6 and 4.0 k cal yrs. BP (Diggerfeldt, 1988). A major strength of the present research is that climate is controlled by the proximity of Upper Talley and Llyn Pencarreg, and any major change in weather patterns should be consistent at both sites.

9.4.2 Other Potential Limnological Drivers

The diatom-based DCA species scores for both sites (Figures 5.6; Upper Talley, and 6.6; Llyn Pencarreg) summarise the major compositional changes in the diatom flora over time and, when compared to the multiproxy data, help to identify the major drivers of change. As noted in Chapters 5 and 6, the distribution along Axis 1 was related clearly to a nutrient/turbidity gradient in Upper Talley, but at Llyn Pencarreg other variables such as changes in lake depth and complex changes in macrophyte abundance and composition may also have had an influence on the diatom flora. The scatter plot of Axis 2 and Axis 3 (Figure 6.5) indicated that pH changes may have influenced the Llyn Pencarreg diatom community (Section 6.12.1). This may have been explored further by application of a diatom-based pH transfer function (Battarbee *et al.*, 2000). However acidification was not the focus of the study and Llyn Pencarreg produced adequate information to suggest changes in ecosystem dynamics in response to eutrophication.

9.5 Summary

The study demonstrates the value of a multi-proxy approach in reconstructing the timing and magnitude of ecosystem change in response to long-term enrichment, and for evaluating the contribution of external forces to the eutrophication process. The limnological indicators provided a record of long-term changes in productivity (DI-TP, aquatic plants, $\delta^{13}C$ and C/N) and evidence of shifts in ecosystem status. In addition, the pollen and charcoal data, the presence of archaeology and the Blytt-Sernander, north-eastern European model for Holocene climate change produced a framework of possible influences to changes in lake status.

Although some problems were encountered, on the whole the selected indicators performed well in the long-term reconstruction of lake conditions at Upper Talley and Llyn Pencarreg. The DI-TP did not perform as well as expected considering that the good representation of diatoms in the modern dataset (i.e. 99% at Upper Talley and 98% at Llyn Pencarreg). This may have been due to the influence of a high proportion of benthic and littoral diatoms, which tend to reflect habitat availability rather than shifts in nutrient status. At the primary trophic level, however, the diatom stratigraphy reflected shifts in abundance and assemblage composition in response to environmental change. Variations in the abundance and diversity of aquatic macrophytes also provided information that helped to identify eutrophic patterns, whilst *Pediastrum* reflected shifts in the wider algal community in response to variations in productivity status. However, a general description of changes in the phytoplankton community would have been strengthened greatly by the use of fossil pigment analysis and Chara oospores; these indicators may have also helped to highlight shifts between the main plant groups during the process of eutrophication.

Although it is possible that different species of ephippia reflect a range of water quality problems, when compared to the multiproxy record the presence of ephippia during phases of enrichment undoubtedly reflected ecosystem stress at both Upper

Talley and Llyn Pencarreg. This assessment may have been strengthened by the analysis of Cladocera body parts, which can reflect habitat availability and changes in nutrient status. Whereas the $\delta^{13}C$ values appeared to be influenced by a range of environmental factors other than productivity, and therefore lacked the ability to reflect changes in productivity, the C/N ratios did meet with more success. However, the C/N profile was more useful as an indicator for long-term terrestrial inwash and appeared to reflect changes in the immediate catchment better than shifts in productivity.

Catchment impact was supported by changes in the pollen record, particularly during the Bronze Age and medieval periods when an increase in deforestation and open ground corresponded with the introduction of agricultural indicators and an increase in arable farming. Although the occurrence of archaeology reflects human presence in the area, it would be overconfident to suggest that this indicates the existence of past human settlements within the lake catchments. The presence of Talley Abbey, and the availability of ecclesiastical records for the early 13th century, however, is indisputable evidence for human occupation within the lake catchment during the medieval period. Finally, the influence of climate and other variables are examined as alternative causes of limnological change. This emphasised the importance of considering that a range of drivers can be ultimately responsible for changes in lake conditions.

CHAPTER 10

COMPARISON OF THE LONG-TERM CHANGES IN ECOSYSTEM DYNAMICS AT UPPER TALLEY AND LLYN PENCARREG

10.1 Introduction

The selection of closely adjacent sites with similar catchment characteristics allowed geology (non-calcareous Silurian Wenlock bedding, with Old Red Sandstone layers), soil type (well drained, loamy, brown earth), vegetation (i.e. temperate deciduous) and climate at Upper Talley and Llyn Pencarreg to be treated as constants. Apart from catchment size and lake depth all other parameters are also comparable (Section 3.7). The two sites would thus be classified within the same category of lake-type under the rules of the EUWFD, which in turn enables the efficacy of the EUWFD methodology for classification, impact assessment and definition of the reference state to be tested (Chapter 11).

The focus of Chapter 10 is to compare long-term ecological and limnological reconstruction between Upper Talley and Llyn Pencarreg in order to explore the contrast in response mechanisms and magnitude of enrichment between two seemingly similar lakes (Aim 3). This will ultimately allow for the evaluation of ecological characteristics, which may differ between the two lakes despite the similarity of environmental parameters. The multiproxy data are interpreted in terms of changing ecosystem dynamics, as discussed in section 1.9, and the associated influence of changes in land-use are assessed by comparison with the pollen, NAP and API data. Figures 10.1 and 10.2 place the sites on a common timescale to allow for the direct comparison of both catchment and limnological indicators during phases of impact.

10.2 Comparison of Ecosystem Response to External Impact During the Major Archaeological Phases at Two Similar Lakes

10.2.1 Mesolithic (*ca.* 10.0 to 6.0 k cal. yrs. BP)

Sediments from the Mesolithic period are missing from the Llyn Pencarreg core due to on-site coring problems, which may have prevented the retrieval of a full Holocene sequence. It may be that the initial origin of the lake was earlier (i.e. below 7.0.m), in which case the basal sediments may represent a lake shallowing event. Alternately, the sequence may be almost complete and the woody sediments may represent its earliest stage of development. The latter would indicate that the lake is not of glacial origin (Section 10.2.2) and evolved more recently as a result of changes in catchment hydrology. Upper Talley, on the other hand, evolved from a shallow lake with a dominance of benthic taxa and low inferred nutrient levels (indicating shallow, clear mesotrophic status prior to *ca.* 9.7 k cal. yrs. BP) to one displaying a low proportion of benthic taxa, which were replaced with planktonic species indicative of eutrophic waters. The presence of double yolked ephippia from *ca.* 9.2 k cal yrs. BP provides very clear evidence of intense environmental pressure that may be habitat-linked; as discussed in Section 2.8, changes in the reproduction techniques of Cladocera are usually related to unpredictable shifts in the wider ecosystem in response to limnological change (Sarmaja-Korjonen, 2004).

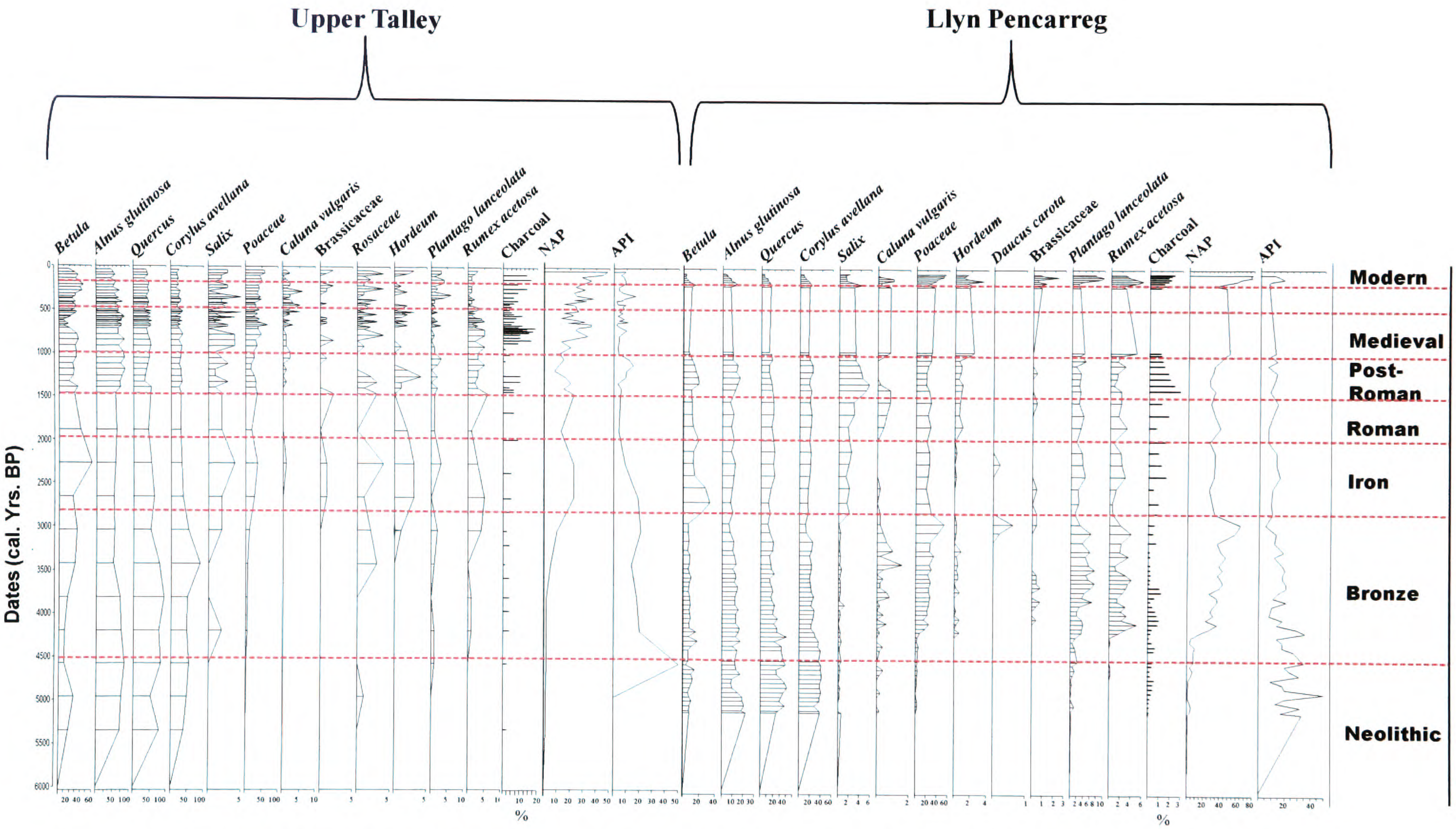


Figure 10.1 Diagram showing Upper Talley and Llyn Pencarreg on a common time-scale to allow for the direct comparison of changes in catchment characteristics.

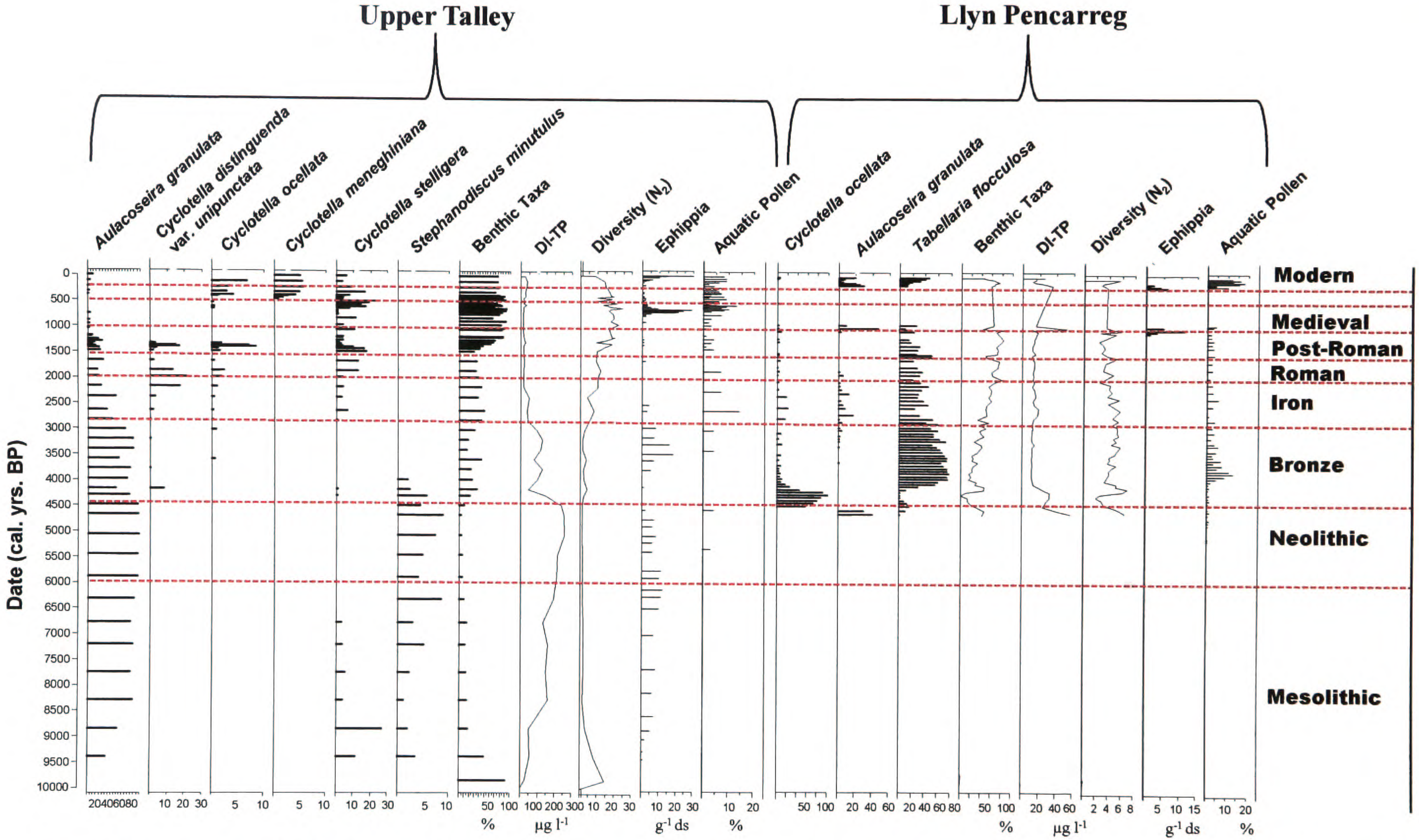


Figure 10.2 Diagram showing Upper Talley and Llyn Pencarreg on a common time-scale to allow for the direct comparison of changes in limnology.

The likely causes of limnological change are discussed in Chapter 7.3 and indicate that in addition to human impact, other factors including climate and lake depth may have had an influence. However, it is likely that limnological stress at Upper Talley was encouraged by the exceptionally high nutrient concentrations and turbid conditions, from *ca.* 8.4 k cal. yrs. BP, which pushed the ecosystem to a point where a resilience threshold was crossed to a new state that was maintained until the start of the Bronze Age (*ca.* 4.5 k cal. yrs. BP). A low diversity planktonic assemblage dominated by obligate meso- to eutrophic diatoms (*A. granulata, C. stelligera* and *S. minutulus*) (Stage 1; see section 9.2.1.1) was followed by further changes in diatom assemblage composition whereby *A. granulata* dominated the record demonstrating its ability to be able to withstand extreme turbidity (Stage 2).

As the robust silica frustules of *A. granulata* enable them to stay buoyant during phases of turbidity (Kamenir *et al.*, 2004), it is likely that along with filamentous blue-green algae (Dokulil and Skolout, 1986) its presence at Upper Talley maintained photosynthesis at the base of the food chain. The presence of ephippia may therefore be associated with a reduction in available prey; it is possible that the heavier silica frustules of *A. granulata* also rendered it inedible (Kamenir *et al.*, 2004). Furthermore, if the evidence from sedimentary fossil pigments is correct then the algal community during this period will also be limited to the presence of inedible toxic green algae (Hall *et al.*, 1997; Lotter, 2001). It is therefore likely that conditions at Upper Talley during the Mesolithic period offered little food opportunities for secondary consumers.

During this period conditions at Upper Talley are in accordance with the 'multiple states' model suggested by Dent *et al.* (2002) that describes ecological status during phases of extreme environmental pressure. The model predicts a process of species reorganisation during phases of environmental change whereby a shift in community structure to one with species more suited to withstand the new conditions enables ecological stability to be maintained (Holling, 1973; Levin, 1999). It is also consistent with the concept that the dominance of a few robust taxa can protect the more vulnerable species from extinction (Persson *et al.*, 2001). Although many authors (e.g. Odum, 1953; May, 1973; Yodis and Innes, 1992) suggest that stability is maintained by high diversity during times of stress, conditions at Upper Talley indicate that the new, low diversity diatom assemblage prevented community collapse by maintaining ecological stability and preserving ecosystem integrity at the phytoplankton level during extreme eutrophication.

Regardless of the traditional views of ecosystem stability (stated in the previous paragraph) that depend on strong community interactions, conditions at Upper Talley show that long-term stability can be maintained at the phytoplankton level alone. Recent theoretical analysis suggests that the persistence of large numbers of competing phytoplankton communities that can survive on a minimal number of limiting resources, are able to produce a range of ecological responses to environmental change. This promotes stability without the need for a high density community (McCann, 2000). Again, the concept of algal competition could have been tested further with the analysis of fossil pigments to examine the shifts in dominant species that can take place during the nutrient enrichment process (Hall *et al.*, 1997; Lotter, 2001).

10.2.2 Neolithic (*ca.* 6.0 to 4.5 k cal. yrs. BP)

Although nutrient loads at the study sites were high during the Neolithic period, human activity was generally of a temporary nature (Caseldine, 1990) and therefore likely to cause low level environmental impact within lake catchments. This is reflected at both sites by the presence of Neolithic archaeology in the region (Figure 3.4), situated in a landscape still dominated by arboreal taxa (Figure 10.1) and woodland plants (Figure 6.9, e.g. *Filipendula, Hedera helix* and *Pteridium aquilinum*). Reductions in the relative abundance of *Betula* at Upper Talley and *Alnus glutinosa* at Llyn Pencarreg, however, suggest that if these changes were the result of human impact then they probably occurred in different areas of the catchments; the shift in *Betula* suggests tree removal on the dryer slopes at Upper Talley, whilst at Llyn Pencarreg the reduction in *Alnus glutinosa* during the wet Atlantic period is likely to indicate deforestation of the valley floor. These differences in the location of tree removal may also be reflected in the peak in pastoral activity at Upper Talley (Figure 10.1), possibly indicating grazing on the higher catchment slopes.

Considering that human impact within the catchments appears to have been fairly subtle, corresponding limnological change is displayed at both sites. However, the shifts in limnology were very different in nature, possibly reflecting differences within the lake catchments. The peak in ephippia at Upper Talley indicates that the highest nutrient loads of the entire core led to an increase in ecosystem stress, which extended from the Mesolithic period. The presence of ephippia may be associated with the loss of specific habitats (e.g. aquatic macrophytes; Jeppesen, 2001), a reduction in available prey (Pennack, 1989) or direct changes in water chemistry parameters (Binford, 1986). Whatever the reason for limnological pressure, Upper Talley continued to display characteristics of the low diversity multiple states model, referred to in the previous section (Dent *et al*, 2002) that is created to protect ecosystem integrity during phases of extreme ecological pressure.

As noted, at Llyn Pencarreg the inferred changes in ecosystem status at the Neolithic/Bronze Age interface, were associated with what may be the origin and early development of the lake. Changes in nutrient status and species composition appear to reflect a hydrological shift from a nutrient-rich muddy, woodland pool to a larger water body. The corresponding decrease in nutrient loads, benthic taxa and planktonic taxa associated with high organic matter (e.g. *C. meneghiniana* and *A. granulata*) and the shift to an assemblage dominated by *C. ocellata* (Figure 10.2) may therefore be associated with the deepening and expansion of Llyn Pencarreg (Rühland *et al.*, 2003).

10.2.3 Bronze Age

The timing and extent of Bronze Age impact at the study sites was distinct and clearly comparable, though land-clearance began later at Upper Talley (*ca.* 3.8 k cal. yrs BP) compared to *ca.* 4.2 k cal. yrs. BP at Llyn Pencarreg (Figure 10.1). The concurrent and significant shift from a predominantly pastoral to a largely arable landscape was characterised by an increase in *Hordeum*, and ruderals such as *Plantago lanceolata* and a significant rise in Poaceae, which follows a similar pattern to other sites in Wales (Thomas, 1965; Caseldine, 1990; Mighall and Chambers, 1995; Chambers, 1983; 1988; 1999; Lillie *et al.*, 2000). *Rumex acetosa* also increases, suggesting that despite the shift to mainly arable farming, pastoral activities continued to a lesser degree. These changes in catchment conditions correspond with the presence of Bronze Age funerary archaeology on the hills surrounding both Upper Talley and Llyn Pencarreg (Figure 3.4). In both cases the cairns were constructed from stone, which will have cleared the land for ploughing and suggests that local soils were valued for agricultural purposes. Although not definitive evidence for anthropogenic impact around the lakes, the concurrent increase in charcoal (Figure 10.1) at both sites also suggests a major increase in human impact.

In spite of these between-site similarities, the magnitude of human impact was significantly different and land-clearance was notably higher at Llyn Pencarreg (compare the NAP profiles in Figure 10.1) suggesting that the inferred changes in land-use were more intense at this latter site. The difference in the degree of impact, however, was not reflected in the magnitude of limnological change and the associated increase in nutrient status was considerably higher at Upper Talley (compare DI-TP values in Figure 10.2). Llyn Pencarreg exhibited a slow response to the changes in land-use with gradual trends of change rather than a sudden threshold shift. Even though *A. granulata* appeared in response to land clearance and increased algal competition is indicated by the temporary peak in *P. boryanum* and increase in diatom diversity, there were no signs of ecosystem stress and the lake remained within the mesotrophic range. In contrast, nutrient loads at Upper Talley increased sharply and significantly in accordance with the changes in land-use, whilst poor water quality and ecosystem stress was indicated once more by a second major increase in *A. granulata* and a peak in ephippial production.

The changing aquatic macrophyte communities also exhibited significant differences. At Upper Talley the increase in aquatic macrophyte abundance at *ca.* 3.4 k cal. yrs. BP corresponded with a decrease in ephippial production suggesting that there may be an association between the two indicators. However, the overall diversity of aquatic macrophytes remained low and was dominated by the submerged plants Cyperaceae and *Potamogeton*. Dominance of these taxa at several lakes around the World has been found to reflect shallow, damp peaty conditions, particularly in the littoral zone (e.g. Mid-Wales, Handa, 1976; North America, Lauderback, 2008; Tibet, Shen *et al*, 2008). These conditions are also suggested by the introduction of *Salix* at *ca.* 3.4 k cal. yrs. BP.

At Llyn Pencarreg a peak in aquatic macrophyte abundance occurred at the onset of land-use change, possibly reflecting the higher degree of land-clearance. The development of a more diverse plant community, represented by Cyperaceae and a range of emergent, mesotrophic species (e.g. *Isoetes lacustis, I. echinospora* and *Ranunculus aquatilis*) (Figure 6.9) may reflect deeper water (Yuan, 2007). Their presence would have provided increased habitat availability and may be instrumental in the concurrent increase in diatom diversity at Llyn Pencarreg, which suggests good ecosystem health (van Nes *et al.*, 2002) compared to Upper Talley. As changes in land-use intensified aquatic macrophyte abundance rose to a sharp peak (*ca.* 3.9 cal. yrs. BP), followed by an immediate decline. The subsequent decrease in relative abundance corresponded with the appearance of *A. granulata*, which peaked at the height of land-clearance. Its appearance is turbidity-related and may be associated with the reduction in macrophytes, which can cause sediment instability in the littoral zone (Schiemer and Prosser, 1976). These changes are typical of the classic shift from plant to algal domination during phases of catchment impact (Figure 10.3) (Mason, 2002).

Considering that land-clearance was more intense at Llyn Pencarreg, its lower inferred nutrient status can only be described by the major difference in magnitude of eutrophication and the high trophic status of Upper Talley at the onset of Bronze Age land-clearance (mean; 46 $\mu g\ l^{-1}$). This may have primed the lake for a rapid return to hypereutrophic status, poor water quality and ecosystem stress. As noted, in contrast to Upper Talley, Bronze Age impact at Llyn Pencarreg took place against a background of relatively low nutrient loads and it is possible that this helped to maintain ecosystem stability. Additionally it is likely that lake water was deeper at Llyn Pencarreg, which may buffer the lake from the effects of human impact and prevent phases of eutrophication (Plater *et al*, 2006). Although a complex combination of natural and human influences caused a high degree of limnological change at Llyn Pencarreg during the first *ca.* 500 years of development, the transition to a mesotrophic status dominated by *T. flocculosa* appears to have produced a 'stable' state.

Conditions at Upper Talley and Llyn Pencarreg during the Bronze Age suggests that although large scale land-clearance and a shift to arable farming took place within the lake catchments, changes in lake trophic status during this period were predetermined by differences in nutrient concentrations before deforestation began rather than by catchment impact itself. Upper Talley shows that high background nutrient loads at the onset of human impact can quickly produce a threshold of poor ecosystem health, whilst results from Llyn Pencarreg suggest that low nutrient concentrations and a

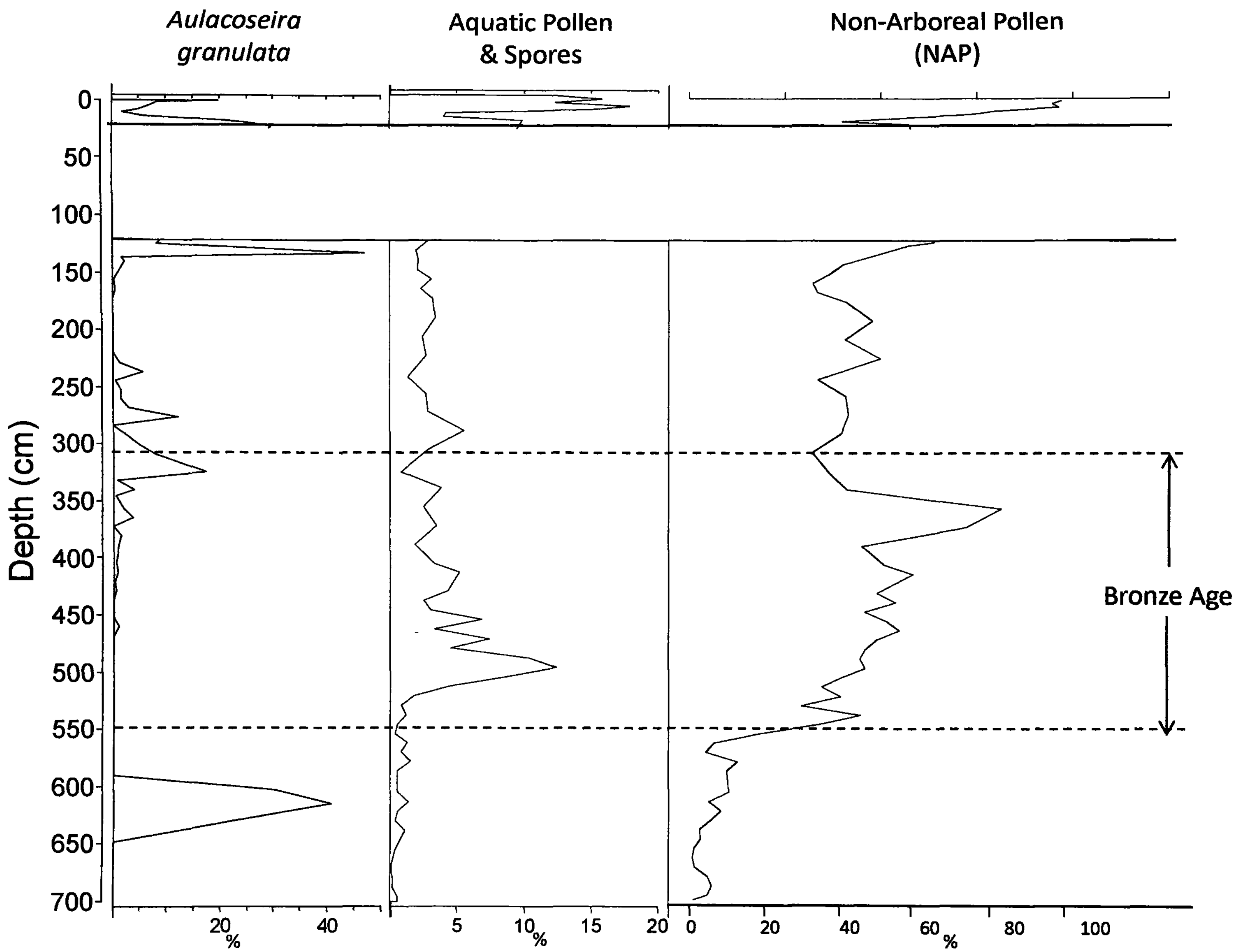


Figure 10.3 Diagram showing the shift between aquatic plant abundance (aquatic pollen and spores) and *A. granulata* during Bronze Age land-clearance (NAP) at Llyn Pencarreg.

deeper water depth can protect a system from extreme limnological change, despite the higher degree of land-clearance. Evidence of these significant differences in two sites with similar physico-chemical characteristics highlights the need for site-specific assessment to identify the main influences on long-term nutrient status, and, to evaluate their importance in setting baseline states (Chapter 11).

10.2.4 Iron Age

Archaeological evidence at both Upper Talley and Llyn Pencarreg indicates continued human presence in the region during the Iron Age (Figure 3.4). Although an increase in *Betula* at *ca.* 2.7 k cal. yrs. BP suggests re-growth on the dryer slopes, the pollen records indicate that both the timing and magnitude of human impact at the study sites was different. Figure 10.1 indicates that changes in land-use continued at Upper Talley and reflected the highest degree of land-clearance at the site so far at *ca.* 2.6 k cal. yrs. BP. The concurrent shift towards an arable farming regime (see API profile) is accompanied by the appearance of cultivated plants (e.g. Brassicaceae, *Daucus carota*), increases in Poaceae, *P. lanceolata* and *Hordeum* and a decrease in *R. acetosa*, reflecting a decrease in pastoral activities.

A pattern of prolonged land-clearance from the Bronze Age appears to show a phase of settlement continuity at the beginning of the Iron Age. This was also common at other sites in Wales (e.g. Welsh Marches [Savory, 1980]; Llangorse [Jones *et al.*, 1985]; Carneddau [Walker, 1993]) as populations remained on the warmer foothills, lakesides and margins of river valleys in order to compensate for climatic deterioration that extended from the late Bronze Age onwards (Mytum, 1988).

Although land-use remained primarily arable, at the end of the Iron Age (from *ca.* 2.2 cal. yrs. BP) the magnitude of land-clearance decreased and was accompanied by a concurrent decrease in *P. lanceolata* and *Hordeum*; possibly reflecting movement away from lake catchments at the beginning of the wet Sub-Atlantic period.

Temporal patterns of land-use at Llyn Pencarreg, however, are in complete contrast with those at Upper Talley. Compared to the increase in land-use at the latter site during the early Iron Age, there was a significant reduction in land-clearance at Llyn Pencarreg (Figure 10.1; NAP profile). This began at *ca.* 2.8 k cal. yrs. BP, and was characterized by a decrease in both Poaceae and *Hordeum.* Although the increase in *Salix* is later at Llyn Pencarreg (*ca.* 2.8 k cal. yrs. BP), compared to *ca.* 3.4 k cal. yrs. BP at Upper Talley, its presence at both sites may reflect the damp conditions of the Sub-Atlantic. This is supported at Llyn Pencarreg by the loss of *Caluna vulgaris*, though its corresponding increase at Upper Talley could reflect differences in local conditions, which may have been a contributing factor to spatial patterns of land-use.

At *ca.* 2.2 k cal. yrs. BP the minor increase in land-clearance at Llyn Pencarreg, characterised by *P. lanceolata* and *R. acetosa* (Figure 10.1), suggests renewed land-use, although not to the magnitude reflected at Upper Talley during the early Iron Age. At many sites in Wales (e.g. Llangorse (Jones *et al.*, 1985), the Brecon Beacons (Chambers, 1982) and the Black Mountains (Price and Moore, 1984) evidence of settlement expansion at *ca.* 2.2 k cal. yrs. BP is accompanied by deforestation, renewed agricultural and pastoral activity, soil deterioration and soil inwash, particularly in the dryer upland areas (Hogg and King, 1963; Moore and Chater, 1969).

In spite of long-term catchment activity, limnological conditions at Upper Talley improved throughout the Iron Age. The reduction in DI-TP to meso-eutrophic status and increase in a range of planktonic *Cyclotella* species accompanied by a decrease in *A. granulata* reflect recovery from long-term hyper-eutrophication. The increase in diatom diversity is concurrent with this interpretation and can reflect improvements in ecosystem health (Das *et al*, 2008). This development is difficult to explain in a lake showing evidence of intense human activity, though the increase in *Cyclotella* species could be directly linked to a lake deepening event, which as discussed in the previous section, can cause a decrease in enrichment, and also in *A. granulata* abundance (Plater *et al*, 2006), and may 'absorb' the usual limnological impact that is indicative of intensified land-use (Mason, 2002). Additionally, the peak in aquatic macrophytes corresponds with lower nutrient loads inferred from other indicators at the beginning of the Iron Age. Their subsequent decrease may reflect the reduction in littoral area that is characteristic of lake deepening events (Rasmussen and Anderson, 2005), thus strengthening the evidence for the potential of internal influences effecting limnological status.

At Llyn Pencarreg the peak in *A. granulata* at the start of the Iron Age may reflect slow limnological response to intensified land-use during the Bronze Age. As turbidity decreased and water clarity increased (indicated by the sustained increase in benthic species and reduction in DI-TP and planktonic *T. flocculosa*), aquatic macrophytes increased and peaked at *ca.* 2.3 k cal. yrs. BP. This suggests that there is a direct relationship between the algal and macrophyte plant groups at Llyn Pencarreg that may have helped to balance nutrient loads. If this is the case, renewed expansion of aquatic plants may have buffered the lake from the effects of limnological impact during the phase of land-use at *ca.* 2.2 k cal. yrs. BP.

Despite their similarities and proximity, not only do Upper Talley and Llyn Pencarreg follow different temporal patterns of land-use during the Iron Age, but again indicate that although land-clearance took place within the lake catchments, changes in lake ecology and the capacity of each system to buffer itself from limnological impact was dictated by individual lake history. Lake conditions at the study sites during the Iron Age therefore suggest that maintaining ecosystem integrity relies on factors such as changes in lake depth and the response of aquatic macrophytes to changing nutrient loads, which can keep lakes in a 'stable' state rather than demonstrating a catastrophic threshold shift. As discussed in Chapter 11, the greater depth of Llyn Pencarreg, in spite of being classified as 'shallow', may be significant.

10.2.5 Roman and Post-Roman Occupation

Again contrasts in temporal patterns of land-use were indicated at the study sites during the Roman period, although there are many similarities during the post-Roman period. At Upper Talley the lack of change in arboreal pollen and the reduction in Poaceae, *Hordeum* and *P. lanceolata* during the Roman period suggest catchment stability and may reflect the strong Ordovician resistance to Roman occupation and the reappearance of localised farming traditions. Cunliffe (2005) explains that this was common in west and central Wales, extending down to Cardigan Bay; which may also explain the lack of settlement archaeology at many sites. The archaeological record shows that this may also true of Upper Talley; the only evidence for Roman occupation is a Roman fort that was discovered in the eastern part of Dinefwr Park, Llandeilo, 12km north of Talley, and at Pencarreg, 10km north.

In contrast, the archaeological record at Llyn Pencarreg indicates that this was a site of Roman occupation (e.g. a gold coin from the Arcadius period (383-408 AD), quern stones and a macehead found at the bed of Llyn Pencarreg itself), which may be linked to the building of the road through the village of Pencarreg. However, although Figure 10.1 indicates an increase in *Hordeum* pollen, suggesting local cultivation from *ca.* 1.7 k cal. yrs. BP, there is surprisingly little evidence of deforestation around the lake. This on the one hand may be due to the temporary nature of occupation, although on the other hand major clearance during the Iron Age may have provided enough land for these changes to take place without further tree removal.

Following an initial decrease in land-use during the post-Roman period, Figure 10.1 indicates that land-clearance at both Upper Talley and Llyn Pencarreg began to increase again from *ca.* 1.3 k cal. yrs BP (NAP). However, the changes in land-use appear to vary between the sites. At Upper Talley a peak in cereal (*Hordeum*) production at *ca.* 1.3 k cal. yrs. BP is followed by a significant resurgence in pastoral activities (API) characterised by an increase in *R. acetosa*. In contrast, at Llyn Pencarreg the consistent presence of cereals and herbs in the pollen record indicates that although land-clearance increased, land-use type remained the same. Hills (1999) explains that during this period populations were driven to sheltered lowland areas, in response to the cool, Sub-Atlantic climate, where pollen records reveal a mixed economy of pastoral and arable farming that varied at a regional scale.

In spite of these changes to the landscape, limnological response at both sites was mild, indicating that by *ca.* 2.0 k cal. yrs. BP both Upper Talley and Llyn Pencarreg demonstrated the ability to withstand phases of low or temporary human impact in their immediate catchments. Shifts in the Upper Talley diatom record appear to relate to internally driven changes that began during the Iron Age in response to either changes in lake depth or improved water clarity following 6,000 years of natural eutrophication. In fact, regardless of the strong evidence for catchment impact during the post-Roman period, the increase in benthic taxa and diatom diversity indicates that water quality at Upper Talley improved even further, whilst nutrient concentrations remained mesotrophic. This may be related to the relatively high percentage of aquatic macrophytes during the Roman period, which may have maintained nutrient concentrations. Their subsequent reduction however, led to a stable aquatic community, which was established during the post-Roman period.

At Llyn Pencarreg the presence of *A. granulata* suggests that the turbid conditions represented during the Iron Age continued until *ca.* 1.8 k cal. yrs. BP. Although an increase in DI-TP was noted during the post-Roman period from *ca.* 1.3 k cal. yrs BP, nutrient status stayed within the mesotrophic range. Aquatic macrophyte growth also remained stable between *ca.* 2.0 and 1.0 k cal. yrs. BP, dominated by Cyperaceae and *Isoetes lacustris* indicating clear, mesotrophic waters (Haslam, 1975). Changes at the study site thus indicate that nutrient loads can remain stable during temporary phases of human impact. At the shallower Upper Talley, minor shifts in limnology may reflect the lake's natural ability to deal with environmental change (Scheffer, 2004), whilst the lack of change at Llyn Pencarreg again supports the hypothesis that deeper lakes can absorb the effects of catchment impact (Carpenter *et al*, 1999).

10.2.6 Medieval

Changes to the Medieval landscape at Upper Talley and Llyn Pencarreg were significant and palynological evidence for land-clearance (which continued from the Post-Roman period) was accompanied by an increase in cereal production (Figure 10.1; NAP). An arable regime was maintained at both sites (API), and thet intensified land-use was comparable with other sites across Northern Europe during this period (Bradshaw, 2001; Gaillard *et al.*, 1991; Håkansson and Regnéll, 1993; Chambers, 1999). These changes reflect the changes in Medieval farming technology that enabled increased cultivation and agricultural production to meet the needs of a growing population (Chavas and Bromley, 2005).

In Figure 3.4 the presence of Medieval archaeology (e.g longhouses, mills, silver coins, mottes, chapels and holy wells) at the study sites provides the first evidence of permanent occupation and may reflect the small nucleated settlements and farmsteads that developed in the semi-lowland and sheltered upland areas and on rich catchment soils as a result of the climatic improvements (The Medieval Warm Period) of the 11th century (Thomas, 1975). The Abbey at Upper Talley was also constructed during this period (completed 1185 AD; Robinson and Platt, 1998) and, as indicated by both documentary and pollen evidence, the high proportion of monastic arable (3900 acres) and pastoral land (3000 acres) (Lewis, 1844) produced the most significant changes to the local landscape since the Bronze Age. At Llyn Pencarreg the network of Mottes suggests tactical defence of the area (Spurgeon, 1987) and trade-links between the small nucleated villages and farming communities (Creighton, 2005), suggesting the presence of well-established communities based on a strong rural economy.

Considering the previously mesotrophic status of both lakes (Figure 10.2), limnological response to changes in land-use (from *ca.* 1.0 k cal. yrs. BP at Upper Talley and 1.2 k cal. yrs. BP at Llyn Pencarreg; Figure 10.1) was sudden and in both cases produced evidence for ecosystem stress. At Upper Talley a threshold of ecosystem stress at *ca.* 0.7 k cal. yrs. BP is indicated by the appearance and significant increase of Cladocera ephippia (Figure 10.2), an increase in inferred primary production and a sharp increase in algal biomass; all positive indicators of eutrophication. Changes in the diatom community, however, were subtle and limited to an increase in the relative abundance of benthic taxa. This change may on the one hand have caused an underestimation of nutrient concentrations, but on the other their increase may reflect maintenance of the algal community. At the highly productive Lake Kinneret (Israel), for example, Kamenir *et al.* (2004) found that increases in the abundance of individual species was conducive with maintaining community stability by preventing significant increases in species diversity and the potential for community collapse (the 'paradox of enrichment'; Rozentzweig, 1971). The increase in species abundance also suggests a resilience mechanism that assures photosynthesis at the base of the food chain and the continuous inflow of energy during a period of perturbation (Brinkhurst, 1974).

The presence of ephippia at Upper Talley may be linked to underlying changes in the aquatic macrophyte community and the associated shifts in predator-prey relationships. *A. granulata* abundance was low during this period (Figure 10.2) suggesting that turbidity was also low and it may be possible that nutrient loads were maintained by a process of sedimentary P-absorption. This mechanism removes P from the main water body and maintains epilimnetic nutrient concentrations during periods of high external loading, particularly at shallow lakes (Søndergaard *et al.*, 1992; 2003; Broström, 1998). As sedimentary P is potentially mobile, the peak in nutrient concentrations at *ca.* 0.6 k cal. yrs BP may be linked to phosphorus release from wind stress or sediment mixing (Wetzel, 2001).

In contrast, at Llyn Pencarreg land-clearance corresponded with the appearance and subsequent peak in *A. granulata* and a reduction in benthic taxa (Figure 10.2), indicating a strong response to changes in land-use, albeit of a different nature to changes at Upper Talley. The decrease in diatom diversity and the first appearance of ephippia at Llyn Pencarreg also reflect ecosystem stress and poor ecosystem health. The response to nutrient enrichment was therefore much faster than the gradual changes observed during the Bronze Age when catchment impact was also substantial. Studies have shown that in lakes deeper than 10m, where there are fewer functional groups (which may be reflected in the lower diatom diversity values of the entire core compared to those at Upper Talley) and resistance pathways can be eroded, a gradual loss of resilience may cause a sudden shift to a state of limnological stress during phases of environmental impact (Carpenter *et al.*, 1995; Scheffer *et al.*, 2001). However, despite marked limnological impact at Llyn Pencarreg the changes were mostly temporary; although the pollen record indicates that deforestation continued, the majority of indicators returned to equilibrium following the height of enrichment. This was accompanied by an increase in aquatic macrophytes, reflecting improvements in ecosystem health (Figure 10.2).

The data from the medieval period therefore suggest that resilience pathways are controlled by the innate ability of each lake to resist these phases of impact, which are different at each site. Additionally, although nutrient loads returned to a pre-impact state (mesotrophic) at both sites, the proxy indicators give a mixed impression of rates and degrees of recovery. At Upper Talley, although ephippia decreased and *C. stelligera* increased in abundance indicating ecosystem recovery (Stage 3 of the eutrophic sequence), the sharp inferred increase in algal biomass (Figure 7.1b) and the appearance and increase in relative abundance of *C. placentula* (Figure 10.2), which tends to be characteristic of lakes with floating vegetation during phases of eutrophication (Germain, 1981), indicates that primary production remained high. At Llyn Pencarreg rates of primary production and algal biomass also remained high (Figure 8.1b), whilst the presence of ephippia suggests that species of Cladocera from specific habitat types were still under pressure. The results may represent a lag between improvements in water quality and ecosystem status or subtle changes in ecosystem health that may not be immediately obvious (Bondavalli *et al*, 2006). This could have significant bearing on the location of baseline states based on the simple top-bottom approach (Chapter 11) as the presence of certain species may give the impression of good ecosystem health, when this is not in fact the case.

10.2.7 Modern (*ca.* 0.4 cal. yrs BP to present)

In spite of the relatively low levels of human impact recorded at rural sites in Britain, when compared to regions of high population density such as south east England (e.g. Bennion, 1994), modern cultural eutrophication has caused a pronounced deterioration in water quality and ecosystem health at both Llyn Pencarreg and Upper Talley. Intensified land clearance is indicated at both sites (NAP; Figure 10.1). It is important to note that although the degree of impact was higher at Llyn Pencarreg, Upper Talley again showed evidence for higher nutrient status, whilst the presence of Cladocera ephippia indicate a state of stress in the recent sediments, again demonstrating the different response pathways of the two sites with similar typology. Despite the earlier recognition of eutrophic sequences at both study sites, modern cultural enrichment appears to have caused an increase in the abundance of the majority of indicators and the appearance of new species at Upper Talley.

At Upper Talley an increase in nutrient concentrations as long ago as *ca.* 0.4 k cal. yrs. BP, caused a shift to an algal assemblage containing eutrophic *C. meneghiniana*, a gradual decrease in diatom diversity (Figure 10.2), and the first appearance of *P. simplex*. A peak in aquatic macrophyte growth was followed by a subsequent decrease, and corresponded with a gradual increase in turbidity, which may have been encouraged by the destabilisation of sediments in the littoral zone. As nutrient loads increased, *C. stelligera* displayed a reduction that is typical of its behaviour at Upper Talley (Stage 1 of the eutrophic sequence). The presence of ephippia reflects marked ecosystem stress and corresponds with the threshold in plant growth, again suggesting that there may be a relationship between these indicators and the loss of aquatic habitat.

At Llyn Pencarreg, the gap in the core sediments means that conditions between *ca.* 0.9 k and 0.2 k cal. yrs. BP cannot be analysed, and consequently the cause and extent of change in ecosystem status for this period remains elusive. From the approximate chronology, Figure 10.2 shows that poor water quality was apparent from as early as the 19th century; it is likely that the building of the Lampeter to Aberystwyth railway in 1866, which runs directly through the lake catchment, contributed to the inferred poor water quality indicated immediately above the gap. However, a subsequent trend towards mesotrophic conditions immediately followed, demonstrating the lake's resilience capacity and ability to return to equilibrium. Improvements in ecosystem health were demonstrated by the major decrease in *A. granulata* and the disappearance of Cladocera ephippia, indicating decreased light attenuation, a reduction in ecosystem stress and improvements in ecosystem health. Benthic taxa expanded and *P. boryanum* was present at low abundance, suggesting an increase in habitat availability and community competition.

About 50 – 100 years ago Llyn Pencarreg showed the strongest shift of the entire sequence with an abrupt return to eutrophic conditions (Figure 10.2). The increase in macrophytes was sudden compared to earlier trends, and although this implies that aquatic plants took advantage of the increase in nutrient concentrations, if eutrophication continues to intensify the plant community is likely to follow previous patterns by reaching a threshold of collapse. Compared to the rate of change during the Bronze Age and the medieval period, the recent changes were very rapid (Figure 6.12) and possibly reflect agricultural intensification and an increase in the use of silage.

The study therefore adds weight to the concept that differences between two 'identical' lakes can be due to natural behaviour and individual response mechanisms rather than simply being a direct function of external impact. It is possible however, that the inferred deterioration in water quality and ecological status towards the top of each core may reflect, in part, different types of catchment pollution. Intensified sheep grazing on improved pasture, which can produce surplus nitrogen that easily mobilises in soils and leaches into aquatic ecosystems causing damage at all trophic levels (Carpenter *et al.*, 1998), can clearly be observed on the banks of Upper Talley, whilst Llyn Pencarreg is located in what appears to be a predominantly agricultural landscape. Additionally, the contrasting histories of human impact at the sites has caused different enrichment trajectories according to the timing of change in the lake catchments

10.3 Summary

The use of data taken from a single core from each site has allowed for the comparative reconstruction of ecosystem dynamics over time ($10^2 - 10^3$ years) in two similar lakes and evaluation of the direct contribution of human impact to the eutrophication process. The multiproxy indicators were selected to represent different trophic levels and produce a realistic picture of ecosystem change. Higher up the food chain, although fish are not represented in this study, other studies show that changes at the phytoplankton level are representative of the amount of energy entering the system and therefore algal productivity is usually a robust indicator of changes in the wider ecosystem (Brinkhurst, 1974; Carpenter *et al.,* 1985). The results of the present study support this.

Examination of the trophic histories of Upper Talley and Llyn Pencarreg demonstrates that despite the uniformity of environmental parameters (e.g. climate, geology, depth, size and water chemistry), which leads to their similar clasification under the EUWFD, eutrophication can differ significantly from lake to lake due to differences in lake development, individual environmental experience and natural differences in ecosystem composition, population dynamics and ecosystem health (Holling, 1973; Peterson *et al.,* 1998; Levin, 1999). This supports the idea that, despite their similarities, each lake is not purely a product of a fixed set of environmental factors, and that individual components produce discrete pathways of ecosystem change and ecological thresholds of response that can cause unpredictable changes in community composition (Scheffer, 2004). The present study also supports the idea that differences in lake-depth can particularly influence lake response to phases of human impact (Moss *et al.*, 2003) This may have serious implications for the location of baseline states if lakes continue to be categorised under such a narrow classification system (i.e. ≥3.0 m and <3.0m). This is discussed further in Chapter 11.

Upper Talley and Llyn Pencarreg have also shown that the timing and extent of ecosystem change in response to human impact can vary with each phase of nutrient enrichment at the same site. This may be related to progressive increases in land-use within the lake catchments, the gradual erosion of resilience mechanisms; which can occur in deep lakes (Llyn Pencarreg) (Carpenter *et al.,* 1995; Scheffer *et al,* 2001), or increased lake sensitivity; which can take place in shallow lakes (Upper Talley) (Hakala *et al.* 2004). When the divergence in development trajectories between Llyn Pencarreg and Upper Talley is set in context with other sites, it is clear that although both lakes display a common capacity to return to a state of equilibrium when the source of impact is reduced or removed, lake character is strongly influenced by individual history, which is unique at each site and central to the problem of defining the relatively undisturbed reference state. This has important implications for applying the concept of a common reference state for lakes in the same region (see Chapter 11).

These findings appear to be supported by research, which focuses on different lake types in other regions. At Glacier Bay, Alaska, Engstrom and Fritz (2006), for example, demonstrated that the significant biochemical couplings between ecosystem development and catchment events are usually site specific. Likewise, at the shallow, calcium-rich, Lake Krankesjön and Lake Tåkern in southern Sweden (Blindow *et al.*, 1993), to maximise light availability submerged aquatics accessed one of a number of random feedback mechanisms depending on minor differences in water level and community composition at each site. In whole lake experiments Schindler (1990; 1998) found that changes in the biotic community due to ecosystem stress can vary significantly between lakes as a result of differences in baseline nutrient concentrations, productivity and community composition. Furthermore, Reynolds *et al.* (1998) found that the development of effective groups is not simply a function of increasing nutrient loads and that changes also depend on the arrival and establishment of new species, which is a function of time, individual development and catchment history.

CHAPTER 11

THE EU WATER FRAMEWORK DIRECTIVE; IMPLICATIONS OF SHORT-TERM (LAST 200YRS) ANALYSIS ON A BACKGROUND OF LONG-TERM (HOLOCENE) CHANGE

11.1 Introduction

In the following section, reconstruction of the long-term dynamics of eutrophication and shifts in ecosystem status at two closely adjacent lakes (Upper Talley and Llyn Pencarreg) classified similarly under the EUWFD, provide a means to test the efficacy of the 'top-bottom' approach for shallow, meso- to eutrophic lakes (Section 1.11, Aims 3-5). As outlined in Section 1.8, in the absence of long-term monitoring data, palaeolimnology has the potential to provide valuable information on ecosystem status prior to accelerated human impact during the 19th and 20th centuries. In recent years the top-bottom approach, based on the analysis of diatom surface sediment samples and 'pre-impact' samples (*ca.* AD 1850 in UK studies), has become an accepted approach for regional palaeolimnological research to measure limnological deviation due to the effects of acidification (Battarbee *et al.*, 2005) and recent climate change (Clark *et al.*, 2005). The method has also become an integral part of the EUWFD, in providing a measure of limnological impact, expressed as the magnitude of ecological deviation from the AD 1850 state, which may also be used to define reference conditions and restoration targets for different 'types' of lake (UKTAG, 2008; Bennion *et al.*, 2005; Leira *et al.*, 2006).

These studies, however, do acknowledge potential limitations. In a regional diatom-based European study of recent ecological deviation in mountain lakes, Clarke *et al.* (2005), for example, note the inherent limitations of the top-bottom approach in that it '*only compares two reference points within a core and cannot account for changes occurring throughout the record'*. This statement is reinforced by Bennion *et al.* (2005) who found the analogue matching of pristine sites within the same region difficult due to individual differences in site histories. Similarly, although Bennion and Battarbee (2007) suggest that the approach has the advantage that it can smooth annual variations in a sediment record, they also acknowledge that the method may fail to produce a realistic indication of ecological change.

There is currently a lack of detailed, long-term research on moderately shallow, meso- to eutrophic lakes with which to test the assumptions that the approach is based on. The most important assumptions comprise (1) that single top and bottom samples are adequate to model ecological deviation over the last *ca.* 150 years, (2) that a stable state of equilibrium existed prior to the putative reference state of *ca.* AD 1850, and (3) that lakes which are classed as similar in terms of physico-chemical characteristics in a given geographic region will have similar ecosystem characteristics, allowing definition of restoration targets by comparison with the reference state of another, similar lake. In the context of the results of the present study, the following chapter discusses the implications of the EUWFD and examines the dictates approach (Aims 3 - 5) to defining realistic sustainable plans for restoration based on the concept of a reference state.

11.2 Testing the Single-sample, Top-bottom Approach to Defining Recent (last *ca.* 150 years) Change

A 'top-bottom' approach, using diatoms to indicate changes in water quality, has been adopted at sites in the UK in order to prioritise restoration targets based on the degree of deterioration between a point of minimal impact and present lake status (Bennion and Battarbee, 2007; UKTAG, 2008). The method is relatively inexpensive and allows for rapid analysis of a regional data-set. As noted (assumption 1), the technique assumes firstly that the 'top and bottom' approach is a reliable reflection of change over the recent timescale (last *ca.* 150 years). This section aims to examine the validity of taking single samples to represent the degree of recent limnological change between present day (top sample) conditions and those of *ca.* AD 1850 (bottom sample).

To identify the samples representing *ca.* AD 1850, the chronology of the recent sediment record was estimated by linear interpolation, assuming constant sediment accumulation rates between the top of each core and radiocarbon dates at 80cm (350 cal. yrs. BP) for Upper Talley and 1.37m (1,030 cal. yrs. BP) for Llyn Pencarreg. This places 'bottom' samples at a depth of 34cm for Upper Talley and 20cm for Llyn Pencarreg. Following Bennion and Battarbee (2004), the direction and magnitude of floral dissimilarity between present day (top sample) and *ca.* 1850 (bottom sample) is estimated using the DCA Axis 1 scores (summarising 34% and 35% of the variance for Upper Talley and Llyn Pencarreg, respectively) and ecological status is assessed by examining the multiproxy data in the recent sediment record as a whole. Figures

11.1 (Upper Talley) and 11.2 (Llyn Pencarreg) show recent change at the study sites, with the top and bottom samples clearly marked.

11.2.1 Upper Talley

At Upper Talley DCA analysis indicates a deviation of 0.2 SD units from 0.3 SD in the bottom sample (34cm) to 0.5 SD in the top sample (2.5cm) (Figure 11.1), providing a measure that reflects the deterioration in lake conditions between *ca.* AD 1850 and present. As discussed in Section 10.3, although the basal sample is unlikely to represent a stable 'pristine' state, the changes in the diatom flora do at least indicate that the degree of difference between bottom and top samples provides a reliable summary of overall change in the intervening period. In this long-term study, the sample resolution is low, but changes in the diatom flora indicate unidirectional change over the last *ca.* 150 years as a response to recent eutrophication.

11.2.2 Llyn Pencarreg

At Llyn Pencarreg the DCA Axis 1 scores indicate an increase of 0.6 SD units from 0.4 SD in *ca.* AD 1850 sample (20cm) to 1.0 SD in the top sample (0.5cm) (Figure 11.2). As with Upper Talley, the bottom sample does not represent a pristine state. In contrast to Upper Talley, however, the diatom data exhibit complex shifts over the last *ca.* 150 years which are not unidirectional, indicating that the top-bottom approach does not adequately model the degree of response to impact over the last *ca.* 150 years. Variation in the DCA Axis 1 scores indicates a limnological shift from eutrophic to an improved, mesotrophic state in the mid-zone (with a decrease in

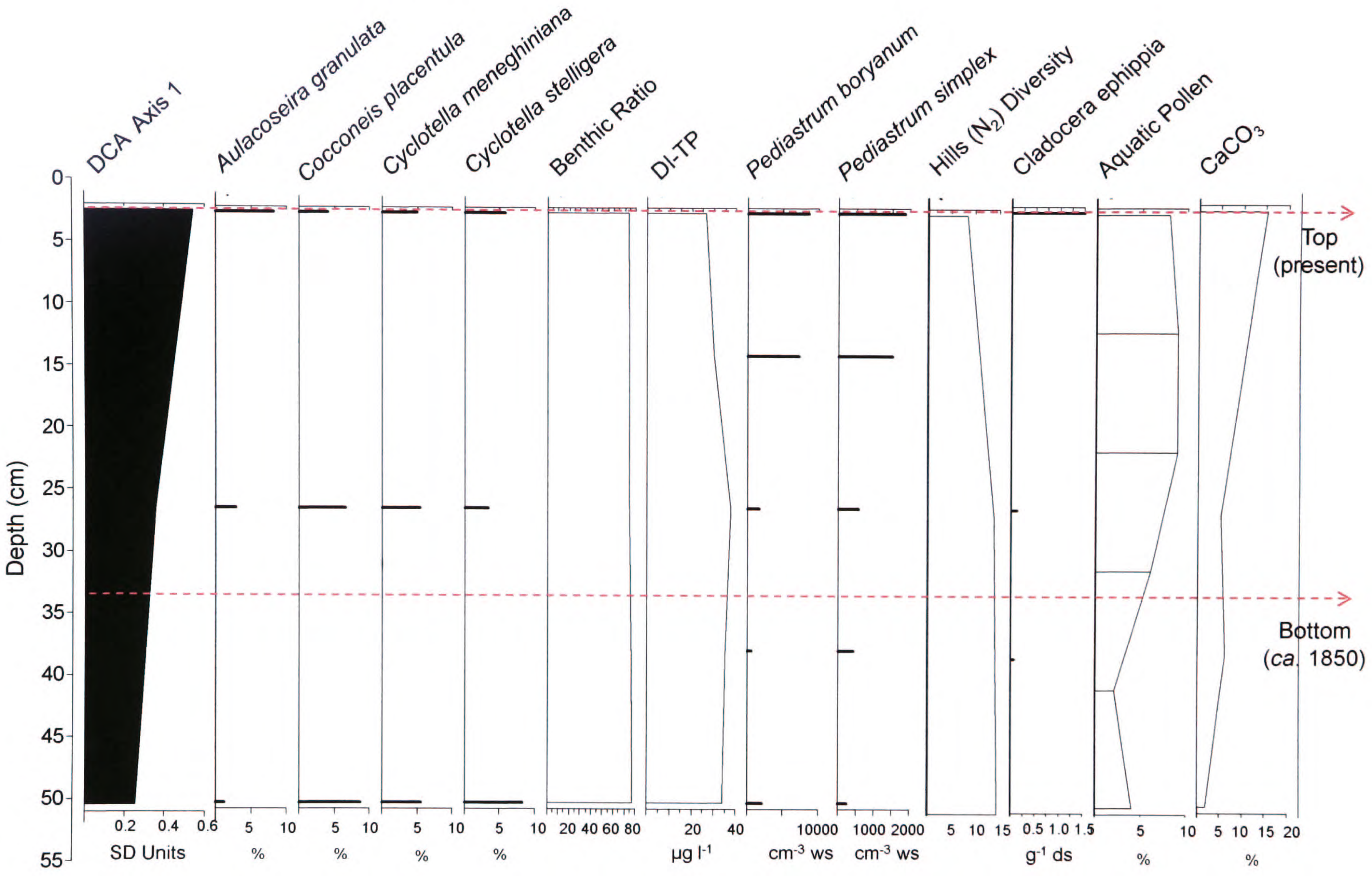


Figure 11.1 Diagram showing recent limnological change at Upper Talley , with the top and bottom samples clearly marked.

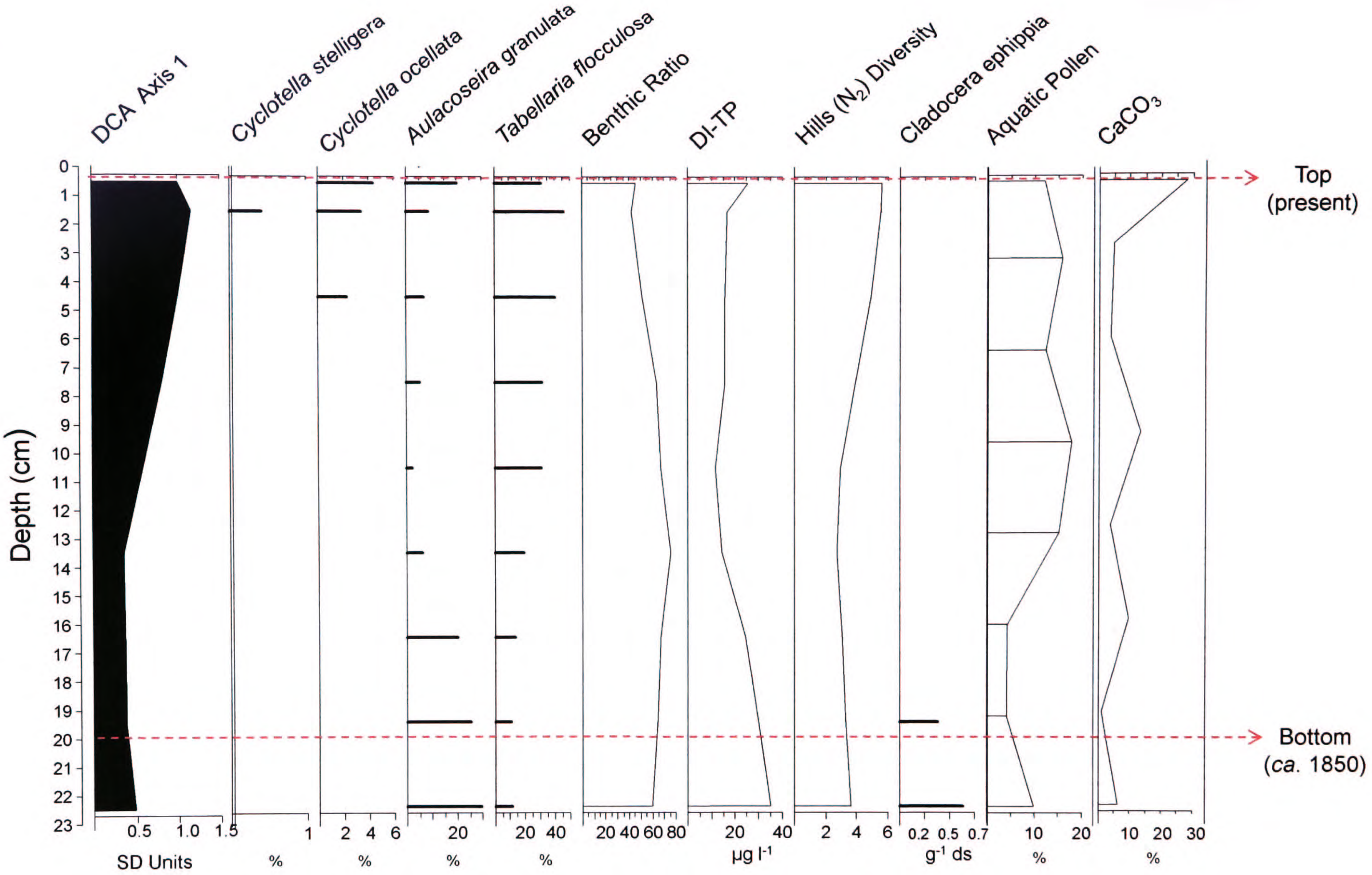


Figure 11.2 Diagram showing recent limnological change at Llyn Pencarreg, with the top and bottom samples clearly marked.

the relative abundance of *A. granulata,* accompanied by the loss of Cladoceran ephippia), followed by a return to eutrophic status over the last *ca.* 50 to 100 years. The *ca.* AD 1850 data provide no more than a 'snap shot' of more complex trends in nutrient enrichment that are missed by the top-bottom method.

11.3 Testing the Concept of a Long-term Stable State of Equilibrium prior to *ca.* AD 1850

Even on the basis of the top-bottom approach, it is clear that in neither site does the *ca.* AD 1850 sample represent a relatively undisturbed, reference state. Instead, the long-term Holocene data indicate that by *ca.* AD 1850, both lakes were turbid and eutrophic, and that the bottom sample represents part of a more complex, long-term trend of Holocene eutrophication and changing ecosystem dynamics.

11.3.1 Upper Talley

In terms of understanding eutrophication dynamics at Upper Talley, conditions in the *ca.* AD 1850 reference sample are the culmination of an enrichment trend that began *ca.* 400 years ago as part of a broader pattern of significant fluctuation throughout the Holocene. A shift in trophic status at *ca.* 0.4 k cal. yrs. BP was followed by a progressive increase in nutrient tolerant taxa (i.e. *P. boryanum*, *P. simplex*, *A. granulata, C. stelligera*, *C. meneghiniana*), and a reduction in aquatic pollen and spores, indicating that the increase in nutrient enrichment is more prolonged than the single 'bottom' sample suggests.

Although land-clearance at Upper Talley has increased significantly in the last *ca.* 150 years BP, the steady increase in available land (NAP; Figure 10.1), and sustained increase in catchment erosion (magnetic susceptibility; Figure 7.1b), suggest that intensified human impact within the lake catchment from *ca.* 0.4 k cal. yrs. ago is the major cause of these changes. As noted in previous sections, in general Upper Talley has been characterised by high instability throughout the Holocene, with major fluctuations over time in a variety of proxy indicators even prior to the *ca.* 0.4 k cal. yr. BP eutrophication trend.

11.3.2 Llyn Pencarreg

Over the long term, Llyn Pencarreg shows a greater degree of stability than Upper Talley, but the results indicate that conditions of poor ecosystem health in the *ca.* AD 1850 sample are also part of a longer-term trend of eutrophication. The cause of impact in the 'bottom' sample cannot be assessed in full due to the gap in the sediment sequence, although as noted, it may be partly a function of railway construction in addition to any agricultural impact. Before the gap, conditions at Llyn Pencarreg demonstrate that even at a site of relative 'stability', there had been significant human impact from as early as the medieval period, characterised by an increase in open land (NAP; Figure 10.2) and changes in the organic record (Figure 8.1b), which drove a long-term trend towards higher nutrient status.

In summary, neither lake exhibits a long-term pattern of stable equilibrium prior to the AD 1850 sample. Upper Talley demonstrates that the most recent increase in nutrient status began *ca.* 400 years ago, whilst in the past 150 years Llyn Pencarreg has fluctuated between eutrophic and mesotrophic states rather than displaying a straightforward increase in trophic status, and had previously shown a subtle but increasing trend towards higher nutrient status since the medieval period.

11.3.3 Testing the Baseline Reference State at Llangorse Lake, Carmarthenshire.

It is useful to assess the conclusions drawn in this study against those from another similar lake in the region which has been the subject of high resolution, long-term palaeolimnological analysis. Llangorse Lake (Bennion and Appleby, 1999), is situated in the Brecon Beacons National Park to the east of the study sites, and has similar climatic and geological parameters. A maximum depth of 9.00m also places Llangorse in the same typology class as the study sites. An interpolated date of AD 1850 at a depth of 1.10m (Bennion and Appleby, 1999) shows that the lake was hypereutrophic (*ca.* 170 $\mu g\ l^{-1}$). DCA analysis was not part of the study, though the change between 1850 and the present is unmistakable and in this case reflects a clear improvement in trophic status. However, the lake is still eutrophic, with a total phosphorus concentration of *ca.*100 $\mu g\ l^{-1}$, and is dominated by planktonic eutrophic taxa (e.g. *Aulacoseira* spp. and *Stephanodiscus* spp.), as opposed to the pre-Roman *Fragilaria* assemblage reported by Jones *et al.* (1974). Additionally, an increase in trophic status between the penultimate and top samples suggests that P loading is on the increase again, reinforcing the suggestion that the top-bottom approach, which ultimately sets the parameters for restoration targets, is of limited utility for assessing important changes in water chemistry and ecological status.

11.4 Can a Realistic Reference State be Defined?

The above discussion shows that in three similar lakes (Upper Talley, Llyn Pencarreg and Llangorse Lake), the bottom *ca.* AD 1850 sample does not represent conditions of minimal impact. Although DCA (and associated diatom data from single samples) provides a warning of ecosystem deterioration, the technique is not adequate to model the degree of limnological change in response to recent human impact. The Holocene palaeolimnological data allow changes in ecosystem health, water chemistry trajectories and patterns of ecosystem dynamics in response to nutrient enrichment to be examined from a longer perspective, to locate a realistic baseline state for restoration. Reconstruction of the patterns of ecosystem response to phases of eutrophication at Upper Talley and Llyn Pencarreg allows definition of the ecological status of a pre-enriched, baseline state that represents natural lake conditions, and (depending on how far back in time this may be) to assess whether it is feasible to use this as a restoration target.

Although examination of long-term nutrient dynamics at the study sites indicates phases of poor ecosystem health caused by anthropogenic impact during major archaeological phases (i.e. Bronze Age, Medieval Period) at both Upper Talley and Llyn Pencarreg, modern eutrophication has produced more marked shifts than at any other time, other than in the lakes' early development. Although not the case in Llangorse, which overall has improved in nutrient status since *ca.* AD 1850, this suggests that the *ca.* AD 1850 state may potentially be valid as an indicator of the *minimum* target for restoration. On the other hand, as in this study, there is evidence that the ecosystem may have been subject to multiple stressors (e.g. eutrophication, pH change). This is supported by the conclusions of regional top-bottom research; in Ireland, for example, Leira *et al.* (2006) found that due to the high sensitivity of the lakes, diatom assemblages were affected by a range of external drivers (e.g. pH, nutrient loading, ice cover), making regional classification more complex than anticipated.

At both lakes the effects of long-term local human occupation mean that a true natural baseline state representing minimal human impact was located at *ca.* 2.1 k cal. yrs. BP for Upper Talley and *ca.* 4.0 k cal. yrs. BP for Llyn Pencarreg when both lakes were mesotrophic and in good ecosystem health (Figure 10.2). Although the high proportion of benthic taxa may have caused an underestimation of DI-TP, their presence indicates the availability of benthic habitats, which is believed to encourage strong community composition and robust predator-prey relationships (Brinkhurst, 1974). High diatom diversity and the presence of *P. boryanum* reflect competition in the algal community, which can effectively summarise ecosystem health; photosynthesis is maintained at the benthic zone, whilst an associated increase in zooplankton diversity helps to develop strong trophic relationships between functional groups that is likely to have a positive effect on ecosystem health (Carpenter *et al.*, 1985; Allott and Monteith, 1999).

Good ecosystem health was also reflected by the lack of obligate eutrophic diatoms and in the stability of the aquatic macrophyte community, the benefits of which are described in Section 9.3.1. In the present study aquatic macrophytes have responded to catchment impact by initially increasing in abundance followed by a decrease, which causes the parallel loss of aquatic habitats and a decrease in ecosystem diversity (Krull, 1970), highlighting their usefulness as an indicator of ecosystem health. It is therefore assumed that the lack of change in relative abundance of aquatic pollen and spores at Upper Talley and Llyn Pencarreg during this period reflects a lack of environmental change.

Although Upper Talley and Llyn Pencarreg appear to regain equilibrium following medieval land-clearance, neither site demonstrates a complete return to the conditions described above, probably due to continuous human impact within the lake catchment. *C. stelligera* demonstrates changes in abundance that are indicative of the eutrophic sequence described in Section 9.2.1.1, whilst diatoms indicative of poor ecosystem health are consistently present (e.g. *C. placentula* and *C. meneghiniana*). Although there are changes in the abundance of aquatic macrophyte pollen and spores indicative of changes in ecosystem health, community composition was diverse and composed of species indicative of the mesotrophic state (e.g. *Nymphaea alba, Littorella uniflora, Ranunculaceae*), providing information that can be of great use to restoration plans.

At Llyn Pencarreg examination of the missing sediments would be necessary to evaluate in full the changes in ecosystem status following the Medieval Period, and the cause and magnitude of impact reflected in the *ca.* AD 1850 sample. However, high resolution sampling shows that the return to mesotrophic status, *ca.* 50 – 100 years ago, demonstrates characteristics of good ecosystem health; e.g. a decrease in *A. granulata*, increase in benthic taxa and a reduction in aquatic macrophytes.

11.5 Testing the Validity of Site Classification

The lakes of Upper Talley and Llyn Pencarreg are located within the same geographic region and would be classified within the same type group in the UKTAG (2008) system. Under the described typology framework, pristine lakes have been selected at sites in Ireland (Leira *et al.*, 2006) and Scotland (Bennion *et al.*, 2004) to analogue match with other lake types, with the intention of grading lake status into good, moderate, poor or bad quality in an attempt to reflect the degree of recent (past *ca.* 150 years) ecological deterioration. The validity of the classification system therefore has a bearing both on impact assessment, and on defining targets for restoration.

The following section shows that despite their similar physico-chemical characteristics, analysis of the biological proxy indicators at Upper Talley and Llyn Pencarreg describe two very different lakes. The two lakes would be classified in the same water depth category (>3.0 - 10.0 m). The most likely explanation of differences in ecosystem response between the two sites is that of differences in water depth, probably coupled with variation in morphometry. The results therefore indicate that the classification system is too broad to address the differences between the study lakes, that may not be initially obvious.

Throughout the Holocene diatom species composition at Upper Talley fluctuates significantly with shifts in benthic and planktonic taxa and the gradual introduction of new species. McCann (2000) refers to these frequent changes in ecosystem composition as 'dynamic stability' and explains that the system is constantly changing as it attempts to maintain a state of 'equilibrium' in response to subtle changes in water chemistry and minor environmental influences within the lake catchment. In contrast, ecosystem dynamics at Llyn Pencarreg reflected gradual changes in the diatom community related to long-term, lake infilling. Response to nutrient enrichment from changes in land-use became progressively stronger with each major phase of human impact possibly reflecting a long-term reduction of ecological integrity, which may be encouraged by intensified land-clearance.

During phases of nutrient enrichment the proxy indicators become difficult to compare due to differences in rates of response between the two sites. To explore between-site variation further, ecosystem status during the previously discussed mesotrophic state (Upper Talley; *ca.* 2.3 k cal. yrs. BP and Llyn Pencarreg *ca.* 4.1 k cal. yrs. BP) provides the opportunity to examine the lakes during a period of similar nutrient status. Although both lakes became mesotrophic following lake development, differences in ecological status reflect a contrast in levels of productivity and species diversity. At Upper Talley aquatic pollen abundance is between 25% and 30% with a rich and diverse flora (Figure 11.3a), dominated by Cyperaceae, and a range of submerged macrophytes typical of meso-eutrophic nutrient loads (e.g. *Ranunculus, Nympheae, Isoetes* spp.). In contrast macrophyte abundance at Llyn Pencarreg is considerably lower at 5% to 15% with generally lower diversity (aquatic pollen is limited to mesotrophic Cyperaceae and *Isoetes lacustris*). Diatom diversity (Figure 11.3b) and concentrations (Figure 11.3c) are also consistently higher at Upper Talley. Variations in diatom assemblage composition between the two lakes also reflect differences in the opportunity for colonisation and in the availability of habitat types (Cairns *et al.*, 1983).

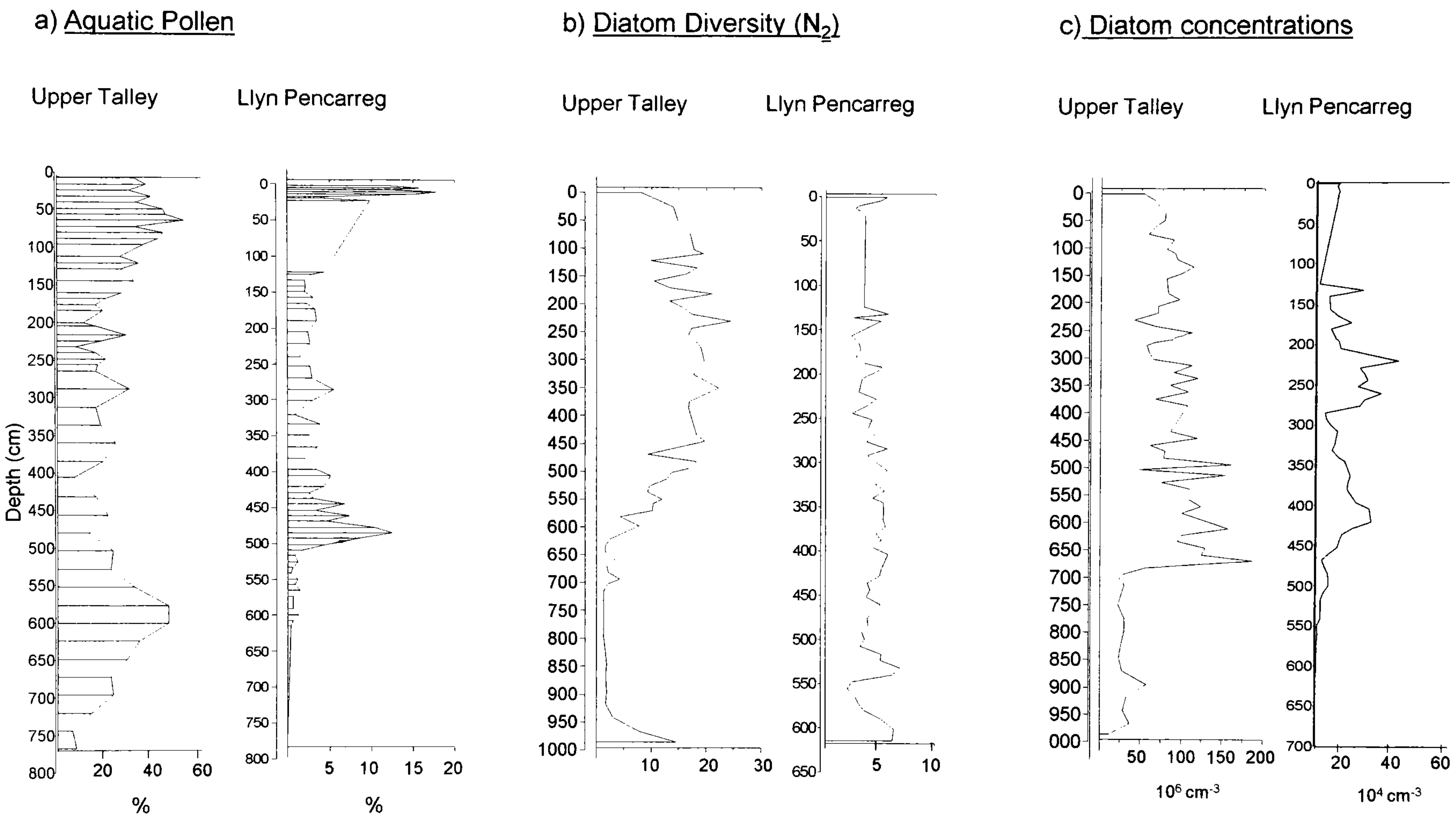


Figure 11.3 Diagram showing a comparison of baseline a) aquatic pollen abundance, b) diatom diversity (N_2) and c) diatom concentrations at Upper Talley and Llyn Pencarreg.

In this study, environmental parameters (e.g. climate, geology, water chemistry) are controlled to enable the comparison of two seemingly similar lakes. However, the above discussion shows that despite this, between-lake variations in baseline conditions reflect two lakes of contrasting nature. The control of environmental parameters also enables the examination of human impact at the study sites as a potential cause of these between-site differences. The timing of impact during the Bronze Age, medieval period and more recently (last 400 years) is analogous at both sites, whilst the shift in agricultural land-use during the early Bronze Age from a mixed farming economy to a predominantly arable regime is also comparable. However, the magnitude of land-clearance (Figure 10.1) is consistently higher at Llyn Pencarreg indicating that the constant fluctuations in ecosystem dynamics and higher productivity status at Upper Talley is not due to the influence of catchment impact.

Upper Talley has a maximum depth of 4.0m, whilst Llyn Pencarreg has a maximum of 10.0m and the palaeolimnological analysis indicates that Llyn Pencarreg has always been the deeper lake. At the same time Upper Talley has shallow banks that form a steady incline into the water, whereas apart from a narrow, wave-cut platform on the west bank and a steady incline at the east, the northern and southern littoral zones of Llyn Pencarreg are steep and rise high above the water's edge. Although the differences in plant abundance and diversity may be associated with the shallow status of Upper Talley and the closeness of the coring site to the littoral zone, it is likely that these differences are a function of differences in the morphometry of the lake basins; the steep sides of Llyn Pencarreg provide little space for aquatic plant growth, and although Upper Talley is presently devoid of littoral plants there is provision for a healthy littoral zone around the entire lake circumference. Additionally, the gentle slopes of the lakesides at Upper Talley provide a range of water depths for the establishment of a variety of submerged and emergent taxa (Lehmann, 1998), which may explain between-site differences in species richness.

The high diatom concentrations and diatom diversity at Upper Talley (Figures 11.3b and 11.3c) may be associated with differences in aquatic macrophyte growth and the availability of plant habitat, but it is also possible that the variations in baseline diversity and ecological behaviour are influenced by the difference in lake depth. This is because shallow lakes naturally support a wider range of functional groups with high rates of productivity due to their extensive habitat availability (Reynolds, 2003). Frequent changes in ecosystem status are often observed in shallow lakes due to their high sensitivity to environmental change (Schindler *et al.*, 1995; Scheffer, 2004). Not only do these changes strengthen ecosystem integrity by increasing the chance of species replacement during phases of impact (Section 1.9.1) (Reynolds, 1999; Kamenir *et al.,* 2004), but they also produce shifts in an attempt to maintain equilibrium and retain good ecosystem health (Harris, 1999). In contrast, communities in deeper lakes with fewer available habitats may become structurally simplified with reduced species composition (Persson *et al.,* 1992; Battarbee *et al.*, 2005). The slower response rates often reflected by deep lakes are therefore due to the presence of fewer functional groups and their poorer diversity record, which can also cause a gradual reduction in resilience pathways with each phase of human impact (Beklioğlu, 1997).

Under the EUWFD guidelines, UKTAG (2008) has developed an ecotype classification system that defines current lake status based on typology and water body delineation that helps to identify the measures required for ecological improvement. Within this framework Upper Talley is classed as a deep lake (>3m) but has produced information that typically describes a shallow lake environment; frequently changing ecosystem status due to a high sensitivity to environmental change and the ability to produce high diversity (e.g. plants and algae) and broad community composition (Lau and Lane, 2001; Moss *et al.*, 1997; Scheffer, 2004). On the other hand the relatively lower productivity status of Llyn Pencarreg and background of slow limnological change are typical of deeper lakes (Scheffer, 1998). The current study therefore provides clear evidence that despite being similarly classified under their physico-chemical characteristics, the broad category description for lake depth within the EUWFD, plus the lack of consideration for lake shape and distinct absence of biological classification, has allowed for two lakes of considerable ecological difference to fall within the same typology group.

The above analysis indicates that in order to provide realistic assessments of current lake status a more complex classification system is required that not only considers physico-chemical characteristics, but also ecosystem structure. There is already acknowledgement in the scientific community that the integration of current limnological knowledge is central to the evaluation of present lake conditions. In their determination of the ecological status of surface waters (ECOFRAME), Moss *et al.* (2003) show that addressing the demand for good ecological status requires liberal interpretation of the EUWFD guidelines with a strong emphasis on biological structure if we are to adequately address present ecological status.

Designed to address the high variability of shallow lakes, their classification system can easily be extended to all lake types to establish environmental objectives and meet the ecological requirements of the directive. In addition to the more obvious physico-chemical parameters (e.g. water chemistry, temperature, geology), the approach considers site specific plant diversity and abundance, the presence of invertebrate predators, zooplankton to phytoplankton ratio, the analysis of taxa sensitive to changes in water chemistry and fish community composition. In this sense ECOFRAME (Moss *et al.*, 2003) considers lakes as individual units that cannot be represented by a single set of ecological conditions, whilst emphasising that a deeper understanding of this concept is necessary to achieve good ecological status for the long-term, sustainable restoration of surface waters. The results of this study support these arguments in full.

11.6 Implications for Applied Management

Palaeolimnology is not being used to its full potential to address the requirements of the EUWFD. The drawbacks in using the top-bottom approach and associated classification system to define impact on lakes have major implications for the definition of realistic restoration plans. Although it has been demonstrated that the post- AD 1850 changes in Upper Talley and Llyn Pencarreg do indicate accelerated impact in the 20th century, when compared to earlier periods, and one might argue that the natural baseline state several thousand years ago cannot be used as a realistic restoration target for a modern lake, the conclusions drawn from this study as to differences in ecosystem response are directly relevant to applied management. The top-bottom approach does not contribute to the understanding of the complexities of ecosystem dynamics, which is necessary to devise successful restoration plans. In addition, as acknowledged by Leira *et al.* (2006) in their discussion of the limitations of modern analogue matching to determine deviation in impacted sites compared to putative pristine sites, the degree of impact which is already apparent by *ca.* AD 1850 (as in Upper Talley, Llyn Pencarreg and Llangorse) may invalidate the use of the pre-20th century state as a valid representation of the pristine state. The pristine, 19th century shallow, mesotrophic lake may not exist in the heavily-populated areas of Britain and Ireland.

If it is realistic to take the 'idealist' approach to restoration, the common occurrence of limnological impact in sediments representing *ca.* AD 1850 does, as noted, highlight its unsuitability as a restoration target. This was the case in both Upper Talley and Llyn Pencarreg, and restoring to the *ca.* 1850 state would without doubt not represent the achievement of good ecological status. As with other studies of long-term human impact (Bradshaw 2001; Ekdahl, 2004; Amsinck *et al.*, 2003; Bradshaw and Rasmussen, 2004), the results of this study clearly support the need for long-term palaeolimnological research to understand patterns of ecosystem response to external impact.

The analysis of multiple proxy limnological indicators is not carried out in most rapid regional assessment palaeolimnology studies, in spite of its widespread adoption in modern monitoring programmes. For restoration, even if not for impact assessment, the results of this study strongly support the contention that analysis of multiple proxy indicators is necessary for the definition of management plans. The present study has noted that many eutrophic diatom taxa have broad tolerance ranges, with apparently site-specific (e.g. differences in the response of *C. stelligera* during eutrophication in the two sites) water chemistry preferences, for example, which might give a false impression of ecosystem health, particularly if conditions are inferred from just one sample. The use of diatoms alone would not be adequate to produce a complete picture of ecosystem health as they only provide information for one level of the food chain and their response to environmental change is not always adequate enough to measure the full extent of limnological change.

Descriptions of baseline conditions and ecological response to nutrient loading at the study sites therefore support the idea that the evaluation of lake status should be approached on a site by site basis making use of the full palaeolimnological record to take account of the natural condition and function of each lake (White and Walker, 1997; Carpenter and Lathrop, 1999; Fozzard *et al.*, 1999, Moss *et al.*, 2003). From this study, the most valuable combination would be that of diatoms, cladocera and aquatic plant pollen (or macrofossils). The link between aquatic macrophytes and ecosystem health has been well demonstrated, highlighting the importance of regaining a healthy littoral zone as part of restoration plans. Reconstruction of changes in the macrophyte community can also identify plant taxa of the baseline reference state for re-establishment (Moss *et al.*, 1997).

The analysis of catchment impacts is also not carried out in regional palaeolimnological programmes. Although well demonstrated by previous research, it is worthy of note that the adoption of pollen and charcoal analysis, magnetic susceptibility and C/N analysis in this study has provided invaluable proxy data on human impact which is not available in regional impact assessment studies.

Significant changes in land-use at Upper Talley and Llyn Pencarreg correspond with shifts in ecosystem status, demonstrating the importance of palynology in assessing the connection between catchment impact and nutrient loading, and having an obvious role to play in devising restoration plans.

On a positive note, an important result of the study is that Upper Talley and Llyn Pencarreg is that the data appear to demonstrate a natural ability to revert to a state of equilibrium with the removal of major sources of human impact. Although recovery is not complete, with carefully planned restoration and catchment management aimed at encouraging limnological improvement it is possible that both lakes would regain a better state of ecosystem health. The evidence suggests that diatoms enter a phase of diapause and will reappear under their optimum conditions (Smetacek, 1985). In the absence of human impact a return to equilibrium is encouraged by the return of diatom taxa associated with good ecosystem health (e.g. Carpenter *et al,* 1992; Mittelbach *et al.,* 1995; Heiri and Lotter, 2004). The return of aquatic plants indicative of lower nutrient loads also helps to maintain water quality, trophic status and ecosystem diversity (Holmes *et al.*, 1998). In many cases the plant community has also been seen to re-establish itself naturally as part of a wider restoration programme that involves lowering nutrient inputs from the lake catchment (Moss *et al.,* 1997).

From the results of the present study, a compromise can be suggested for rapid, regional palaeolimnological impact assessment, to produce a clearer interpretation of environmental change within the constraints of limited time and finance. Spot sampling of a longer core using several carefully selected indicators (e.g. diatoms for eutrophication, chironomid exuviae for acidification (UKTAG, 2008) and Cladocera ephippia for ecosystem stress) would generally provide a reasonable assessment of recent shifts in ecosystem status, although a full Holocene approach would be unfeasible due to the field sampling time involved in collecting several metres of Livingstone core. Bennion and Battarbee (2007) suggest the use of a wider (14cm), longer (150cm) piston corer, developed at UCL to collect larger volumes of sediment for multiproxy analysis, whilst a Glew, messenger-operated, gravity corer (Glew, 1991), which is light and easy to transport, can also collect sediment sequences up to 100cm long. This would provide *ca.* 400 years of sediment accumulation at UT and *ca.* 600 years at Llyn Pencarreg, which would indeed be adequate to provide a realistic indication of ecosystem health, nutrient status and reference conditions. Were these techniques coupled with realistic attempts to examine individual lake ecology, their application would provide information pertaining to changes in ecosystem status during the eutrophication process and thus increase the likelihood of successful restoration.

CHAPTER 12

CONCLUSIONS

12.1 Précis

Due to the demand for improvements in the ecological and chemical status of surface waters within Europe, the European Water Framework Directive (EUWFD; Directive 2000/60/EC) intends to address the issues surrounding poor water quality by ordering the restoration of inland water bodies across Europe, where necessary, to *good ecological status* by 2015. UKTAG (2008), the body responsible for implementing the directive in the UK, must produce draft management plans and restoration objectives by December, 2008. Palaeolimnological techniques are being increasingly employed to complement large-scale biomonitoring programmes.

With a focus on diatom analysis, the 'top-bottom' approach prioritises water bodies for restoration depending on the degree of ecological deviation over the last *ca.* 150 years. After lake classification, the inferred ecological status at *ca.* 1850 may also be used as a reference state to define restoration plans for individual lakes of a similar type. The technique is based on a set of assumptions which can be tested by longer-term palaeolimnological research. The most important are (1) that the degree of compositional change between top and bottom samples is adequate to model ecological deviation over the last *ca.* 150 years, (2) that a stable state of equilibrium existed prior to a *ca.* 1850 undisturbed reference state, and (3) that lakes which are regionally classed as similar in terms of physico-chemical characteristics will have similar ecosystem characteristics.

To test these assumptions long cores were extracted from two moderately nutrient-enriched sites, Upper Talley and Llyn Pencarreg, in Carmarthenshire, South Wales. Both lakes would be classified as highly similar under the EUWFD typology framework. Multi-proxy palaeolimnological techniques were adopted to reconstruct ecosystem dynamics. The range of proxy indicators employed was selected primarily on the basis of their strength as indicators of mid- to long-term changes in ecosystem dynamics in response to nutrient enrichment (Diatoms, *Pediastrum* spp., Cladocera ephippia, $\delta^{13}C$). Phases of human impact within the lake catchment (pollen, charcoal, C/N, LOI, magnetic susceptibility, sediment lithology) were examined to identify possible causes of limnological impact and explore the degree to which observed limnological change has been driven by human activities rather than natural (ontogeny or climate-induced) change prior to the Industrial Revolution (*ca.* 1850).

On the whole, the proxy indicators performed well and the use of data taken from a single core from each site allowed for the comparative reconstruction of nutrient status and ecosystem dynamics in two similar lakes and an evaluation of the direct contribution of human impact to the eutrophication process. However, the results show that the observed differences between the two lakes were not due simply to contrasting land-use practices in the catchment. Examination of the long-term trophic histories of Upper Talley and Llyn Pencarreg indicates that despite their similar classification under the EUWFD typology framework, the consequences of eutrophication differ significantly from lake to lake due to differences in lake development, ecosystem composition, response mechanisms, and in basin morphometry.

Upper Talley produced a record characterised by relatively rapid, short-term fluctuations in community composition. This was in accordance with the widely reported increase in lake sensitivity and lack of stability observed in shallow, eutrophic lakes. Llyn Pencarreg demonstrated steady changes and possibly the gradual erosion of resilience mechanisms, which appears to be more indicative of deep lakes. Upper Talley and Llyn Pencarreg also show that the timing and extent of ecosystem change in response to human impact can vary with each phase of nutrient enrichment at the same site, which may be related to progressive increases in land-use within the lake catchments and changes in lake depth. Also demonstrated is the capacity of both lakes to return to a state of equilibrium when the source of impact is reduced or removed.

When the significant differences between Llyn Pencarreg and Upper Talley are set in context with other sites, it is clear that lake character is strongly influenced by individual history, which is unique at each site and central to the problem of defining the relatively undisturbed reference state. The results support the idea that each lake is not purely a product of a fixed set of environmental factors, and that individual components can produce discrete pathways of ecosystem change that can cause unpredictable changes in community composition (Scheffer, 2004).

The results were used to test the main assumptions of the palaeolimnological EUWFD approach. The results indicate that in some sites, where there is a relatively simple trajectory of increasing impact over the last *ca.* 150 years (Upper Talley), the 'top-bottom' approach is perfectly adequate to summarise the degree of recent deviation. In others (Llyn Pencarreg), ecosystem change over the last *ca.* 150 years may be more complex, and the technique is less effective. As has been acknowledged in applying the technique, the results confirm that at both sites (and in nearby Llangorse Lake) the assumption of a stable state of equilibrium prior to *ca.* 1850 is not valid, due to significant human impact over the longer term. While it may be unrealistic to aim to restore a lake to a pre-Medieval pristine state, this does have implications for using *ca.* 1850 as an undisturbed baseline for restoration targets in other lakes. The study demonstrates that ecological conditions in the bottom, *ca.* 1850 sample did not represent conditions of minimal impact.

Examination of the longer, multi-proxy record shows an extended history of limnological impact at Upper Talley from *ca.* 0.4 k cal. yrs. BP, and a more complex site history leading to a recent phase of recovery at Llyn Pencarreg, both of which are missed by the top-bottom approach. Finally, the assumption that lakes of similar physico-chemical characteristics, within the same locality, should exhibit similar patterns of ecosystem response is not valid in this case; the classification of water depth 3-10m is too broad, and lake morphometry has also led to important differences in ecosystem dynamics.

The long-term data from Upper Talley and Llyn Pencarreg has allowed a more realistic reference state to be defined based on conditions of minimal human impact. At both lakes this appears to be following Bronze Age impact, when diatoms and aquatic plants indicate a mesotrophic phase displaying all the characteristics of good ecosystem health. However, the interpretation of good ecosystem health is clearly broad regardless of regional classification, and the ability of Upper Talley and Llyn Pencarreg to revert to a state of equilibrium following phases of human impact suggests that with careful restoration based on achieving good ecological status, both lakes are likely to eventually return to a pre-impact state.

12.2 Future Recommendations

The present study shows that a detailed approach is essential for the development of restoration plans based on achieving good ecological status. A classification system which includes biological rather than physico-chemical typology groups is essential if good ecological status is to be achieved (Moss *et al.*, 2003), whilst multiple proxy indicators are necessary to examine and highlight the complex changes in ecosystem dynamics indicative of individual lakes. Quick and efficient techniques could still be applied, such as sampling at higher resolution (>4 samples in the top *ca.* 1.5m of a core) rather than a 'top' and 'bottom' approach. Proxy techniques other than the standard selection (e.g. diatoms), such as charophyte or cladocera ephippia analysis, could also be usefully employed (and, in some studies, are being) to address the lack of ecological information and produce reliable data to infer changes in ecosystem health. In this manner the aim of achieving good ecological status in a range of lake types is more likely to be achieved.

BIBLIOGRAPHY

Aalderink, R.H., Lijklema, L., Breukelman, J, Van Raaphorst, W. and Brinkman, A.G., 1985. Quantifications of wind induced re-suspension in a shallow lake. *Water Science and Technology* **17**, 903-914.

Abel, P.D., 1996. *Water Pollution Biology.* Taylor and Francis, England.

Alefs, J. and Müller, J., 1999. Differences in the eutrophication dynamics of Ammersee and Starnberger See (Southern Germany) reflected by the diatom succession of varved sediments. *Journal of Paleolimnology* **21**, 395-407.

Alhonen, P. and Ristiluoma, S., 1973. On the occurrence of subsoil *Pediastrum* algae in a core from Kirkkonummi, southern Finland. *Bulletins of the Geological Society of Finland, Helsinki* **45**, 73-77.

Alley, R.B. and MacAyeal, D.R., 1994. Ice-rafted dedris associated with binge/purge oscillations of the Laurentide Ice Sheet. *Palaeoceanography* **9 (4)**, 503-512.

Allott, T.E.H. and Monteith, D.T., 1999. Classification of lakes in Wales for conservation using integrated biological data. *CCW Science Report* **No. 314**.

Almquist-Jacobson, H., 1995. Lake-level fluctuations at Ljustjarnen, Central Sweden and their implications for the Holocene climate of Scandinavia. *Palaeogeography, Palaeoclimatology and Palaeoecology* **118 (3-4)**, 269-290.

Amsink, S.L., 2003. Vandrammedirektivet og danske søer. Del 2: Palaeoøkologiske undersøgelser. *Faglig rapport fra Danmarks Miljø-undersøgelser* **476**, 120. Cited in; Bradshaw, E. G. and Rasmussen, P., 2004. Using the geological record to assess the changing status of Danish lakes. *Geological survey of Denmark and Greenland Bulletin* **4**, 37-40.

Andersen, S.T., 1973. The different pollen productivity of trees and its significance for the interpretation of a pollen diagram from a forested region. In; Birks, H. J. B. and West, R. G. (eds.). *Quaternary Plant Ecology*. 108-114, Blackwell, Oxford.

Anderson, N.J., 1997. *Reconstructing Historical Phosphorus Concentrations in Rural Lakes Using Diatom Models.* Division of Environmental History and Climate, GEUS, Copenhagen, Denmark.

Anderson, N.J. and Odgaard, B.V., 1994: Recent palaeolimnology of three shallow Danish lakes. *Hydrobiologia* **256-257**, 411-422.

Anderson, N.J. and Rippey, B., 1988. Diagenesis of magnetic minerals in the recent sediments of a eutrophic lake. *Limnology and Oceanography* **33**, 1476-1492.

Anderson, N.J. and Rippey, B., 1994. Monitoring lake recovery form point-source eutrophication: the use of diatom-inferred epilimnetic total phosphorus and sediment chemistry. *Freshwater Biology* **32**, 625-639.

Anderson, N.J., Rippey, B. and Stevenson, A.C., 1990. Change of a diatom assemblage in a eutrophic lake following point source nutrient redirection: A palaeolimnological approach. *Freshwater Biology* **23**, 205-217.

Antoniades, D., Douglas, M.S.V. and Smol, J.P., 2005. Quantitative estimates of recent environmental changes in the Canadian High Arctic inferred from diatoms in lake and pond sediments. *Journal of Palaeolimnology* **33 (3)**, 349-360.

Archibold, O.W., 1995. *Ecology of World Vegetation.* Chapman and Hall, London.

Baier, J., Lücke, A., Negendank, J.F.W., Schleser, G.H. and Zolitschka, B., 2004. Diatom and geochemical evidence of mid- to late Holocene climatic changes at lake Holzmaar, West-Eifel (Germany). *Quaternary International* **113,** 81-96.

Balaya, D.J. and Moss, B., 2003. Spatial patterns and population dynamics of plant associated microcrustaceae (Cladocera) in an English shallow lake (Little Mere, Cheshire). *Aquatic Ecology* **37**, 417-435.

Barber, H.G., 1981. *A Guide to the Morphology of the Diatom Frustule.* Freshwater Biological Association, Scientific Publication 44.

Barber, K.E., Battarbee, R.W., Brooks, S.J., Eglinton, G., Hayworth, E.Y., Oldfield, F., Stevenson, A.C. Thompson, R., Appleby, P.G., Austin, W.E.N., Cameron, N.G., Ficken, K.J., Golding, P., Harkness, D.D., Holmes, J.A., Hutchinson, R., Lishman, J.P., Maddy, D., Pinder, L.C.V., Rose, N.L. and Stoneman, R.E., 1999. Proxy records of climate change in the UK over the last two millennia: documented change and sedimentary records from lakes and bogs. *Journal of the Geological Society* **156 (2)**, 369-380.

Barber, K.E., Maddy, D., Rose, N., Stevenson, A.C., Stoneman, R. and Thompson, R., 2000. Replicated proxy climate signals over the last 200 years from two distant UK peat bogs: new evidence for regional palaeoclimatic teleconnections. *Quaternary Science Reviews* **19 (6)**, 481-487.

Barber, K.E., Chambers, F.M. and Maddy, D., 2003. Holocene palaeoclimates from peat stratigraphy: macrofossil proxy climate rtecords from three oceanic raised bogs in England and Ireland. *Quaternary Science Reviews* **22**, 521-539.

Barclay, G. J., 1997. The Neolithic. In; Edwards, K.J. and Ralston, I.B.M. (eds). *Scotland: Environment and Archaeology, 8000 BC to AD 1000.* John Wiley, Chichester.

Barker, P., 1993. The Interpretation of the Evidence. In; *Techniques of Archaeological Excavation.* Routeledge, England.

Barton, N., 1999. The Late-Glacial or Late and Final Palaeolithic colonisation of Britain. In; Hunter, J. and Ralston, I. (eds). The Archaeology of Britain; an

Introduction from the Upper Palaeolithic to the Industrial Revolution. Routledge, London.

Battarbee, R.W., 1982. The use of electronically counted microspheres in absorlute diatom analysis. *Limnology and Oceanography* **27 (1)**, 184-188.

Battarbee, R.W., 1984. Diatom analysis and the acidification of lakes. *Philisophical Transaction of the Royal Society of London* **B 305**, 451-477.

Battarbee, R.W., 1986. Diatom Analysis. In: *Handbook of Palaeoecology and Palaeohydrology*, (ed. Berglund, B. E.). Wiley, Chichester.

Battarbee, R.W., 1991. Palaeolimnology and Climate Change. In: Freryel, B. (ed). *Evaluation of Proxy Data in Relation to the European Holocene.* Gustav Fischer Verlag, Stuttgart.

Battarbee, R.W., 1999. The importance of paleolimnology to lake restoration. *Hydrobiologia* **395/396** 149-159.

Battarbee, R.W., Anderson, N.J., Jeppesen, E. And Leavitt, P.R., 2005. Combining palaeolimnological and limnological approaches in assessing lake ecosystem response to nutrient reduction. *Freshwater Biology* **50 (19)**, 1772-1780.

Battarbee, R.W., Grytnes, J-A., Thompson, R., Appleby, P.G., Catalan, J., Korhola, Atte, Birks, H.J.B., Heegaard, E. And Lami, A., 2002. Comparing palaeolimnological and instrumental evidence of climate change for remote mountain lakes in the last 200 years. *Journal of Palaeolimnology* **28 (1)**, 161-179.

Battarbee, R.W., Juggins, S., Gasse, F., Anderson, N.J., Bennion, H. and Cameron, N.G., 2000. *European Diatom Database (EDDI). An information system for palaeoenvironmental reconstruction.* European Climate Science Conference, Vienna City Hall, Vienna, Austria, 19th-23rd October 1998.

Battarbee, R.W., Mason, J., Renberg, I. and Talling, J.F. (eds.), 1990. *Palaeolimnology and Lake Acidification.* The Royal Society of London, London.

Battarbee, R.W., Mckay, A.W., Jewson, D.H., Ryves, D.B. and Sturm, M, 2005. Differential dissolution of Lake Baikal diatoms: correction factors and implications for palaeoclimatic reconstruction. *Global and Planetary Change* **46**, 75-86.

Battarbee, R.W., Monteith, D.T., Juggins, S. Evans, C.D., Jenkins, A. and Simpson, G.L., 2005. Reconstructing pre-acidification pH for an acidified loch: A comparison of palaeolimnological and modelling approaches. *Environmental Pollution* **137 (1)**, 135-149.

Behre, K.E., (ed.) 1986. *Anthropogenic Indicators in Pollen Diagrams.* Balkema, Rotterdam.

Beklioğlu, M., 1997. A review on the control of eutrophication in deep and shallow lakes. *Turkish Journal of Zoology* **23**, 327-336.

Bell, M., 2007. Prehistoric Coastal Communities: The Mesolithic in Western Britain. *CBA Research Report* **149**.

Bengtsson, L. and Enell, M., 1986. Chemical Analysis. In: Berglund, B.E. (ed.). *Handbook of Holocene Palaeoecology and Palaeohydrology.* John Wiley and Sons, Ltd., 423-451.

Bennet, K.D., 1994. Confidence intervals for age estimates and deposition times in late-Quaternary sediment sequences. *The Holocene* **4**, 337-348.

Bennet, K.D., 1994b. 'Psimpoll' Version 2.23: a C program for analysing pollen data and plotting pollen diagrams. *Commission for the study of the Holocene Working group on data-handling methods, Newsletter* **11**, 4-6.

Bennet, K.D., Simonson, W.D. and Peglar S.M., 1990. Fire and man in the Post-glacial woodlands of Eastern England. *Journal of Archaeological Science* **17**, 635-642.

Bennike, O., Sarmaja-Korjonen, K. and Seppänen, A., 2004. Reinvestigation of the classic late-glacial Bølling Sø sequence, Denmark: chronology, macrofossils, Cladocera and chydorid ephippia. *Journal of Quaternary Science* **19**, 465-478.

Bennion, H., 1994. A diatom-phosphorus transfer function for shallow, eutrophic ponds in southeast England. *Hydrobiologia* **275/276**, 391-410.

Bennion, H., 1995. Quantitative reconstruction of the nutrient histories of 3 Anglesey lakes. *CCW Contract Science Report* **87**.

Bennion, H., 2001. Reconstructing nutrient histories in the Norfolk Broads: implications for the application of diatom-phosphorus transfer functions to shallow lake management. *Journal of Palaeolimnology* **26**, 181-204.

Bennion, H., Allott, T.E.H., Appleby, P.G., Hunt, M., Oliver, E. and Patrick, S.T., 1997. A study of recent environmental change within selected standing waters proposed as Special Areas of Conservation (SAC) in Wales - Llyn Idwal, Llyn Cwellyn and Llyn Safadden (Llangorse Lake) Phase II. *CCW Research Report* **187**.

Bennion, H., Allott, T.E.H. and Shilland, E., 1998. Investigation of environmental change in two mesotrophic lakes in Mid-Wales: Llyn Eiddwen and Llyn Fanod. Contract Number. FC-73-07-71A. *CCW Research Report* **46**.

Bennion, H. and Appleby, P., 1999. An assessment of recent environmental change in Llangorse Lake using palaeolimnology. *Aquatic Conservation: Marine and Freshwater Ecosystems* **9**, 361-375.

Bennion, H, Appleby, P.G. and Phillips, G.L., 2001. Reconstucting nutrient histories in the Norfolk Broads, UK: Implications for the role of diatom-total phosphorus transfer functions in shallow lake management. *Journal of Palaeolimnology* **26**, 181-204.

Bennion, H. and Battarbee, R.W., 2007. The European Union Water Framework Directive: opportunities for palaeolimnology. *Journal of Palaeolimnology* **38**, 285-295.

Bennion, H., Davidson, T., Simpson, G., Solovieva, N., Rose, N., Theophile, S., Yang, H., Anderson, N.J., Brooks, S. and Peglar, S., 2004. *Identification of reference lakes and evaluation of palaeoecological approaches to define reference conditions for UK (England, Scotland, Wales and Northern Ireland) ecotypes.* Final Report (project WFD08) to Scotland and Northern Ireland Forum for Environmental Research (SNIFFER).

Bennion, H., Duigan, C.A., Haworth, E.Y., Allott, T.E.H., Anderson, N.J., Juggins, S. and Monteith, D.T., 1996a. The Anglesey Lakes, Wales, UK- Changes in trophic status of three standing waters as inferred from diatom transfer functions and their implications for conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems* **6**, 81-92.

Bennion, H., Fluin, J. and Simpson, G.L., 2005. Assessing eutrophication and reference conditions for Scottish freshwater lochs using subfossil diatoms. *Journal of Applied Ecology* **41 (1)**, 124-138.

Bennion, H., Juggins, S. and Anderson N.J., 1996b. Predicting epilimnetic phosphorus concentrations using an improved diatom-based transfer function and its application to lake eutrophication management. *Environmental Science and technology* **30**, 2004-2007.

Bennion, H., Monteith, D. and Appleby, P., 2000. Temporal and geographical variation in lake trophic status in the English Lake District: evidence from sub-fossil diatoms and aquatic macrophytes. *Freshwater Biology* **45**, 394-412.

Bennion, H., Shilland, E. and Appleby, P. G., 2002. An assessment of recent environmental change in Llyn Tegid using the sediment record. In; Gritten, R. H., Duigan, C. A. and Millband, A. (eds.). *The ecology, conservation and environmental history of the largest natural lake in Wales.* Snowdonia National Park Authority.

Benson, D.G., Evans, J.G. and Williams, G.H., 1990. Excavations at Stackpole warren, Dyfed. *Proceedings of the Prehistorical Society* **56**, 179-245.

Berglund, B.E., 2003. Human impact and climate changes-synchronous events and a causal link? *Quaternary International* **105**, 7-12.

Bernasconi, S.M., Barbieri, A. and Simona, M., 1997. Carbon and nitrogen isotope variations in sedimenting organic matter in lake Laguno. *Limnology and Oceanography* **43**, 1755-1765.

Berridge, P.J., 1980. Waun-Fignen-Felin. *Archaeology in Wales* **20**, 10-20.

Bertrand, C., Fayolle, S., Franquet, E. and Cazaubon, A., 2003. Responses of the planktonic diatom Asrerionella Formosa Hassal to abiotic environmental factors in a reservoir complex (south-eastern France). *Hydrobiologia* **501**, 45-58.

Binford, M.W., 1986. Ecological correlates of net accumulation rates of Cladocera remains in lake sediments. *Hydrobiologia* **143 1**, 123-128.

Birks, H.J.B., 1998. Numerical tools in palaeolimnology: Progress, potentialities, and problems. *Journal of Paleolimnology* **20**, 307-332.

Birks, H.H., Battarbee, R.W. and Birks, H.J.B., 2000. The development of the aquatic ecosystem at Kråkenes Lake, western Norway, during the late-glacial and early-Holocene-a synthesis. *Journal of Paleolimnology* **23**, 91-114.

Birks, H.H. and Birks, H.J.B, 2000. Future uses of pollen analysis must include plant macrofossils. *Journal of Biogeography* **27**, 31-35.

Björck, S. and Håkansson, H., 1982. Radiocarbon dates from Late Weichselian lake sediments in South Sweden as a basis for chronostratagraphic subdivisions. *Boreas* **11**, 141-150.

Björck, S. and Wohlfarth, B., 2001. ^{14}C Chronostratigraphic Techniques in Palaeolimnology. In: Last, W.M. and Smol, J.P., (eds.). *Tracking Environmental Change Using Lake Sediments, Vol. 1: Basin Analysis, Coring and Chronological Techniques.* Kluwer Academic Press, Dordrecht, 205-245.

Blackmoor, N., 2002. A Fabric Type Series of Roman Pottery in Wales. *Unpublished Cardiff University Thesis.*

Blakemore, R.P., 1975. Magnetotactic bacteria. *Science* **190**, 277-379.

Blindow, I., Andersson, G., Hargeby, A. and Johansson, S., 1993. Long-term patterns of alternative stable states in two shallow eutrophic lakes. *Freshwater Biology* **30**, 159-167.

Blytt, A., 1876. *Essay on the immigration of the Norwegian flora during alternate rainy and dry periods.* Cammermeyer, Kristiana.

Bobbink, R., Boxman, D., Fremstad, E., Heil, G., Houdijk, A. and Roelofs J., 1993. Nitrogen eutrophication and critical load for nitrogen based upon changes in flora and fauna in (semi)-natural terrestrial ecosystems. In: *Critical loads for nitrogen.* Proceedings of a UN-ECE workshop at Lökeberg, Sweden. 6-10 April 1992. 111-159. Nordic Council of Ministers, Copenhagen, Denmark.

Boix, D., Gascon, S., Sala, J., Martinoy, M., Gifre, J., and Quintana, X.D., 2005. A new index of water quality assessment in Mediterranean wetlands based on crustacean and insect assemblages: the case of Catalunya (NE Iberian Peninsula). *Aquatic Conservation; Marine and Freshwater Ecosystems* **15 (6)**, 635-651.

Bondavalli, C., Bodini, A., Rossetti, G. and Allesina, S., 2006. Detecting stress at the whole-ecosystem level: The case of a mountain lake (Lake Santo, Italy). *Ecosystems* **9**, 768-787.

Bonsall, C., Macklin, M., Anderson, D.E. and Payton, R.W., 2002. Climate change and the adoption of agriculture in north-west Europe. *European Journal of Archaeology* **5 (1)**, 9-23.

Boon, P.J., 2000. The development of integrated methods for assessing river conservation value. *Hydrobiologia* **422**, 413-428.

Borgmark, A., 2005. Holocene climate variability and periodicies in south-central Sweden, as interpreted from peat humification analysis. *The Holocene* **15**, 387-395.

Bostock, J.L., 1980. The history of the vegetation of the Berwyn Mountains, North Wales, with emphasis on the development of blanket mire. Unpublished PhD thesis, University of Manchester.

Bottema, S., 1975. The interpretation of pollen spectra from prehistoric settlements (with special attention to Liguliflorae). *Palaeohistoria* **17**, 18-35.

Boutton, T.W., 1991. Stable carbon isotope ratios of natural materials: II. Atmospheric, terrestrial, marine and freshwater environments. In; Coleman, D. C. and Fry, B. (Eds.). *Carbon Isotope Techniques* 173-185. Academic Press, San Diego.

Bowen D.Q., 1978. *Quaternary Geology: A Stratigraphic Framework for Multidisciplinary Work.* Pergamon Press, Oxford.

Bowen D.Q., Phillips, F.M., McCabe, A.M., Knutz, P.C. and Sykes, G.A., 2002. New data for the Last Glacial Maximum in Great Britain and Ireland. Quaternary Science Reviews 21 (1-3), 89-101.

Boyle J.F., 2004. A comparison of two methods for estimating the organic matter content of sediments. *Journal of Paleolimnology* **31**, 125-127.

Bradbury, J.P., 1971. Palaeolimnology of Lake Texcoco, Mexico. Evidence from diatoms. *Limnology and Oceanography* **16**, 180-200.

Bradbury, J.P., 1986. A climatic-limnologic model of diatom succession for palaeolimnological interpretation of varved sediments at Elk Lake, Minnesota. *Journal of Palaeolimnology* **1**, 115-131.

Bradshaw, E., 2001. The response of lake nutrient regimes and diatoms to long-term land-use change in Denmark. *Unpublished PHd thesis,* The University of Copenhagen, Copenhagen, Denmark.

Bradshaw, E.G. and Rasmussen, P., 2004. Using the geological record to assess the changing status of Danish lakes. *Geological survey of Denmark and Greenland Bulletin* **4**, 37-40.

Brecon Beacons, 2006. At www.brecon-beacons.com/llangorse-lake.htm (cited on 15/05/06)

Brenner, M., Whitmore, T.J., Lasi, M.A., Cable, J.E. and Cable, P. H., 1996. A multi-proxy trophic state reconstruction for shallow Orange Lake, Florida, USA: possible influence of macrophytes on limnetic nutrient concentrations. *Journal of Paleolimnology* **21**, 215-233.

Brenner, M., Whitmore, T.J., Curtis, J.H., Hoddel, D.A. and Schelske, C.L., 1999. Stable isotope signatures of sedimented organic matter as indicators of historic lake trophic status. *Journal of Palaeolimnology* **22**, 205-221.

Breukers, C.P.M., van Dam, E.M. and de Jong, S.A., 1997. Lake Volkerak-Zoom: A lake shifting from the clear to the turbid state. *Hydrobiologia* **342/ 343**, 367-376.

Briffa, K. and Atkinson, T., 1997. Reconstructing Lateglacial and Early Holocene Climates. In; Hulme, M. and Barrow, W. (eds.). *Climates of the British Isles*, Routledge, London.

Brinkhurst, R.O., 1974. *The Benthos of Lakes*. The Blackburn Press, New Jersey, USA.

Brodersen, K.P., Melbourn, C., Whiteside, C. and Lindegaard, C., 1998. Reconstruction of trophic states in Danish Lakes using subfossil chydorid (Cladocera) assemblages. *Canadian Journal of Fisheries and Aquatic Science* **55**, 1093-1103.

Bronk Ramsey, C., 1995. Radiocarbon Calibration and Analysis of Stratigraphy: The OxCal Program. *Radiocarbon* **37**, 425-430.

Broström, A., Gaillard, M.J., Ihse, M. and Odgaard, B., 2004. Pollen landscape relationships of modern analogues of ancient cultural landscapes in southern Sweden – a first step towards quantification of vegetation openness is the past. *Vegetation History and Archaeobotany* **7 (4)**, 189-281.

Broström, A., Sugita, S., Gaillard, M.J. and Pilesjö, P., 2005. Estimating the spatial scale of pollen dispersal in the cultural landscape of southern Sweden. *The Holocene* **15 (2)**, 252-262.

Broström, B., Andersen, J.M., Fleisher, S., Jansson, M., 1998. Exchange of phosphorus across the sediment-water interface. *Hydrobiologia* **170 (1)**, 229-244.

Brown, A., 1977. An early Flandrian vegetation record from the floor of the Bristol Channel. *New* Phytologist **78**, 525-531.

Brown, A.G., 1999. Characterising prehistoric lowland environments using local pollen assemblages. *Quaternary Proceedings* **7**, 585-594.

Brown, T., 2002. Clearance and Clearings: Deforestation in Mesolithic/Neolithic Britain. *Oxford Journal of Archaeology* **16 (2)**, 133-146.

Brown, T.A., Nelson, D.E., Mathewes, R.W., Vogel, J.S. and Southon, J.R., 1989. Radiocarbon dating of pollen by Acclelerator Mass Spectrometry. *Quaternary Research* **32,** 205-212.

Brück, J., 1999. What's in a settlement? Domestic practice and residential mobility in Early Bronze Age Southern England. In; Brück, J. and Goodman, M (eds). *Making Places in the Prehistoric World: themes in settlement archaeology,* London: UCL Press.

Brugham, R.B. and Speziale, B.J., 1983. Human disturbance and the paleolimnological record of change in the zooplankton community of Lake harriet, Minnesota. *Ecology* **64**, 578-591.

Brylinsky, M. and Mann, K.H., 1973. An analysis of factors governing productivity in lakes and reservoirs. *Limnology and Oceanography* **18**, 1-14.

Burden, E.T., McAndrews, J.H. and Norris, G., 1986. Palynology of Indian and European forest clearance and farming in lake sediment cores from Awanda provincial park, Ontario. *Canadian Journal of Earth Science* **23**, 43-54.

Burn, C.R., 1997. Cryostratigraphy, palaeogeography and climate change during the early Holocene warm interval, western Arctic coast, Canada. *Canadian Journal of Earth Sciences* **34 (7)**, 912-925.

Butler, S., 1984. Preliminary investigations of the pollen record from the Talley Lakes. *The Carmarthenshire Antiquary* **20** 3-14.

Byrne, R., Michaelsen, J. and Soutar, A., 1977. Fossil charcoal as a measure of wildfire frequency in Southern California: A preliminary analysis. In; Mooney, H.A. and Conrad, C. E. (eds.*). Proceedings of the symposium on the environmental consequences of fire and fuel management in Mediterranean ecosystems,* 361-367. United States Department of Agriculture Forest Service General technical report WO-3, Washington.

Cairns, J., Plafkin, J.L., Kaesler, R.L. and Lowe, R.L., 1983. Early colonization patterns of diatom and protozoa in fourteen fresh-water lakes. *Journal of Eukaryotic Microbiology* **30 (1)**, 47-51.

Carmarthenshire, 2007. www.carmarthenshire.gov.uk (cited on 10/12/2007)

Carpenter, S.R., Caraco, N.F., Correll, D.L., Howarth, R.W., Sharpley, A.N. and Smith, V.H., 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications* **8 (3)**, 559-568.

Carpenter, S.R., Clifford, E.K., Wright, R., He, X., Soranno, P.A. and Hodgeson, J.R., 1992. Resilience and resistance of a lake phosphorus cycle before and after food web manipulation. *The American Naturalist* **140 (5)**, 781-798.

Carpenter, S. R., Elser, J.J. and Olsen, K.M., 1983. Effects of roots of *Myriophylum verticillatum* L. on sediment redox conditions. *Aquatic Botany* **17**, 243-250.

Carpenter, S. R., Kitchel, J.F. and Hodgson, J.R., 1985. Cascading trophic interactions and lake productivity. *Bioscience* **35**, 634-639.

Carpenter, S.R. and Lathrop, R.C., 1999. Lake restoration: capability and needs. *Hydrobiologia* **395/396**, 19-28.

Carpenter, S. R., Ludwig, D. and Brock, W. A., 1999. Management of eutrophication for lakes subject to potentially irreversible change. *Ecological Applications* **9 (3)**, 751-771.

Carvalho, L. and Moss, B., 1995. The current status of a sample of English sites of special scientific interest subject to eutrophication. *Aquatic Conservation; Marine and Freshwater Ecosystems* **5 (3)**, 191-204.

Caseldine, A., 1990. *Environmental Archaeology in Wales.* Cadw Welsh Historic Monuments, Archaeology Department, St. David's University College, Lampeter.

Catalan, J., Ventura, M., Brancelj, A., Granados, I., Thies, H., Nickus, U., Lotter, A.F., Barbieri, A., Stuchlik, E., Lien, L., Bitušik, P., Buchaca, T., Camarero, L., Goudsmit, G.H., Kopá, J., Strunecký, O. and Toro, M., 2002. Seasonal ecosystem variability in remote mountain lakes: implications for detecting climatic signals in sediment records. *Journal of Palaeolimnology* **28 (1)**, 25-48.

Chambers, F.M., 1982. Two radiocarbon dated pollen diagrams from high altitude blanket peats in South Wales. *Journal of Ecology* **70**, 445-459.

Chambers, F.M.,1983a. New applications of paleoecological techniques. Integrating evidence of arable activity in pollen. Peat and soil stratigraphies, Cefn Greanog, North Wales. In; Jones, M. (ed.). *Integrating the subsistence economy.* B.A.R. International Series **181**, 107-122.

Chambers, F.M.,1983b. Three radiocarbon pollen diagrams from upland peats north-west of Merthyr Tydfil, South Wales. *Journal of Ecology* **71**, 475-487.

Chambers, F.M., 1988. Palynology and archaeology in Wales; recent and current research. *Archaeology in Wales* **28**, 35-36.

Chambers, F.M., 1999. The Quaternary history of Llangorse Lake: implications for conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems* **9**, 343-359.

Chambers, F.M. and Price, S.M., 1985. Palaeoecology of Alnus: early post-glacial rise in a valley mire, north-west Wales. *New Phytologist* **101**, 333-344.

Chambers, F.M. and Price, S.M., 1988. The environmental setting of Erw-wen and Moel y Gerddi: prehistoric enclosures in upland Ardudwy, North Wales. *Proceedings of the Prehistorical Society* **54**, 93-100.

Champion, T., 1999. The Later Bronze Age. In; Hunter, J. and Ralston, I. (eds). The Archaeology of Britain; an Introduction from the Upper Palaeolithic to the Industrial Revolution. Routledge, London.

Chapman, J. L. and Reiss, M. J., 1997. *Ecology; Principles and Applications.* Cambridge University Press, Cambridge.

Charles, B.G., 1934. *Old Norse Relations with Wales.* University of Wales Press, Cardiff.

Charles, D.F., Whitehead, D.R., Engstrom, D.R., Fry, B.D., Hites, R.A., Norton, S.A., Owen, J.S., Roll, L.A., Schindler, S.C., Smol, J.P., Uutala, A J., White, J.R. and Wise, R.J., 1987. Palaeolimnological evidence for recent acidification of Big Moose Lake, Adirondack Mountains, N. Y., USA. *Biogeochemistry* **3**, 267-296.

Charlesworth, J.K., 1929. The South Wales end-morain. *Quarterly Journal of the Geological Society of London* **85**, 335-358.

Chavas, J.P. and Bromley, D.W., 2005. Modelling Population and Resource Scarcity in Fourteenth Century Britain. *Journal of Agricultural Economics* **56 (2)**, 217-237.

Cherapanova, M.V., Snyder, J.A. and Brigham-Grette, J., 2007. Diatom stratigraphy of the last 250 ka at Lake El'gygytgyn, northeast Siberia. *Palaeolimnology* **37**, 155-162.

Clark, J.G.D., 1938. Microlithic Industries from tufa deposits at Prestatyn, Flintshire anf Blashenwell, Dorset. *Proceedings of the Prehistoric Society* **4**, 330-334

Clark, J.S., 1990. Fire and climate change .during the last 750 years in northern Minnesota. *Ecological Monographs* **60**, 135–59.

Clark, J.S., 1988. Particle motion and the theory of charcoal analysis: source area, transport, deposition and sampling. *Quaternary Research* **30 (1)**, 67-80.

Clark, R.L., 1982. Point Count Estimation of Charcoal in Pollen Preparations and Thin Sections of Sediments. *Pollen et Spores* **24 (3-4)**, 523-535.

Clarke, G., Kernan, M., Marchetto, A., Sorvari, S. and Catalan, J., 2005. Using diatoms to assess geographical patterns of change in high altitude European lakes from pre-industrial times to present day. *Aquatic Science* **67**, 224-236.

Cleary, S.E., 1999. Roman Britain; Civil and Rural Society. In; Hunter, J. and Ralston, I. (eds). The Archaeology of Britain; an Introduction from the Upper Palaeolithic to the Industrial Revolution. Routledge, London.

Clegg, J., 1970. Excavations at Coyan Cave, near Laugharne. *Carmarthenshire Antiquity* **5-6**, 13-20.

Cloutman, E., 1983. Studies in the vegetational history of the Black Mountain Range, South Wales. *Unpublished PhD thesis*, University of Wales.

Colman, S.M., King. J.W., Jones, G.A., Reynolda, R.L. and Bothner, M.H., 2000. Holocene and recent sediment accumulation rates in Southern Lake Michigan. *Quaternary Science Review* **19 (16)**, 1563-1580.

Conolly, A.P. and Dickson, 1969. A note on a late-Weilchselian Splanchinum capsule from Scotland. *New Phytologist* **68**, 197-199.

Conway, J.S. and Younis, M.G.A., 1984. The mineralogy of soils from Moel Goedeg 1. *Archaeologia Cambrensis* **133**, 43-44.

Coope, G.R. and Brophy J.A., 2008. Late Glacial environmental changes indicated by a coleopteran succession from North Wales. *Boreas* **1 (2)**, 97-142.

Cooper, S. R., 1999. Esturine reconstruction using diatoms. In; Stoermer, E. F. and Smol, J. P. (eds.). *The Diatoms: Applications for the Environmental and Earth Sciences*, 352-373, Cambridge University Press, Cambridge.

Coops, H., 2002. Ecology of Charophytes: An Introduction. *Aquatic Botany* **72 (3-4)**, 205-208.

Cottingham, K.L., 1996. *Phytoplankton responses to whole-lake manipulations of nutrients and food webs*. University of Wisconsin, Madison, USA.

Coulthard, J.C. and Macklin, M.G., 2001. How sensitive are river systems to climate and land-use changes? A model based evaluation. *Journal of Quaternary Science* **16 (4)**, 347-351.

Coveney, M.F., Lowe, E.F., Battoe, L.E., Marzolf, E.R. and Conrow, R., 2005. Response of eutrophic, shallow subtropical lake to reduced nutrient loading. *Freshwater Biology* **50**, 1718-1730.

Crabtree, K., 1965. *Late Quarternary deposits near Capel Curig*, North Wales. PhD thesis, University of Bristol.

Crabtree, K., 1969. Post-glacial diatom zonation of Quaternary deposits in North Wales. *Mitt. Internat. Verein. Limnol.* **17**, 165-171.

Crabtree, K., 1972. Late-glacial deposits near Capel Curig, Caernarvonshire. *New Phytologist* 71, 1233-1243.

Crampton, C.B., 1973. An Interpretation of the pollen distribution through the Hardings' Down Hillfort. *Archaeologia Cambrensis* **122**, 55-68.

Creighton, O.H., 2005. Castles *and Landscapes: Power, Communication and Fortification in Medieval England*. Equinox Publishing Ltd., London.

Crisman, T.L. 1980. Chydorid cladoceran assemblages from subtropical Florida. In *Evolution and Ecology of Zooplankton Communities*. Kerfoot, W.C. (ed.). American Society of Limnology and Oceanography, Special Symposium 3, 657–668.

Crisman, T.L. and Whitehead, 1978. Palaeolimnological studies on small New England (USA) ponds. II Cladoceran community responses to trophic oscillations. *Polski Archivum Hydrobiologie* **25** 75-86.

Cunliffe, B., 2005. *Iron Age Communities in Britain.* Routledge, London.

Dalton, C., Birks, H.J.B., Brooks, S.J., Cameron, N.J., Evershed, R.P., Peglar, S.M., Scott, J.A. and Thompson, R., 2005. A multi-proxy study of lake development in response to catchment changes during the Holocene at Lochnagar, north-east Scotland. *Palaeogeography, Palaeoclimatology, Palaeoecology* **221 (3-4)**, 175-201.

Dark, P., 1976. Palaeoecological evidence for landscape continuity and change in Britain ca. AD 400-800. In; Dark, K.R. (ed). *External Contacts and the Economy of Late Roman Britain and Post-Roman Britain.* Boydell Press.

Dark, P., 2006. Climate deterioration and land-use change in the first millennium BC: perspectives from the British palaeolimnological record. *Journal of Archaeological Science* **33**, 1381-1395.

Das, B., Nordin, R. and Mazumder, A., 2008. Relationship between phytoplankton palaeoproduction and diversity in contrasting trophic states. Aquatic Ecosystem Health and Management 11 (1), 78-90.

da Silva, C.A., Train, S. and Rodrigues, L.C., 2005. Phytoplankton assemblages in a Brazilian subtropical cascading reservoir system. *Hydrobiologia* **537**, 99-109.

David, C., Dearing, J. and Roberts, N., 1998. Land-use history and sediment flux in a lowland lake catchment: Groby Pool, Leicestershire, UK. *The Holocene* **8 (4)**, 383-394.

Davidson, T.A., Sayer, C.D., David, C., Rose, N. and Wade, M. 2005. A 250 year comparison of historical, macrofossil and pollen records of aquatic plants in a shallow lake. *Freshwater Biology* **50**, 1671-1686

Davies, M.B., Brubaker, L.B. and Beiswenger, J.M., 1971. Pollen grains in lake sediments: Pollen percentages in surface sediments in southern Michigan. *Quaternary Research* **1,** 450-467.

Davies, W., 1982. *Wales in the Early Middle Ages.* Leicester University Press, Leicester.

Davis, R.B., 1967. Pollen studies of near surface sediments in Maine lakes. In; Cushing, E. J. and Wright, H. E. (eds.). *Quaternary Palaeoecology.* Yale University Press, New Haven.

Davis, M.B., 1981. Outbreaks of forest pathogens in Quaternary history. *Proceedings of VI International Conference on Palynology, Lucknow, India* **3**, 216-227.

Davis, O.K., 1999. Pollen analysis of Tulari Lake, California: Great basin-like vegetation in central California during the full Glacial and Early Holocene. *Review and Paleobotony and Palynology* **107 (3-4)**, 249-257.

Day, S.P., 1993. Preliminary results of high resolution palaeoecological analysis at Star Carr, Yorkshire. *Cambridge Archaeological Journal* 3, 129-140.

Dean, W.E., 1974. Detection of carbonate and organic matter in calcareous sedimentary rocks by loss on ignition. *Chemical Geology* **44**, 151-185.

Dean, W.E., 1999. The carbon cycle and biogeochemical dynamics in lake sediments. *Journal of Palaeolimnology* **21,** 397-393.

Dearing, J.A., 1991. Erosion and Landuse. In; Berglund, B. E. (Ed.). The cultural landscape during 6000 years in southern Sweden. *Ecological Bulletins* **41**, 283-292.

Dearing, J.A., 1999. Holocene environmental change from magnetic proxies in lake sediments. In Maher, B.A. and Thompson, R. (eds.). *Quaternary Climates, Environments and Magnetism.* Cambridge University Press. 231-278.

Dearing, J.A. and Flowers, R.J., 1982. The magnetic susceptibility of sedimenting material trapped in Lough Neagh, Northern Ireland, and its erosional significance. *Limnology and Oceanography* **27**, 969-975.

DEFRA, 2004. At www.defra.gov.uk/environment/water (cited on 16/07/04).

DEFRA, 2005. At www.defra.gov.uk/wildlife-countryside (cited on 12/04/06).

De Geer, 1912. A geochronology of the last 12000 years. *11th International Geological Congress* **1**, Stockholm, Sweden, 241-253.

Delcourt, A.D. and Delcourt, R.D., 1991. *Quaternary Ecology: A Palaeoecological Perspective.* Chapman and Hall, London.

deNicola, D. M., 2000. A review of diatoms found in highly acidic environments. *Hydrobiologia* **422 (1-3)**, 111-122.

Dent, C.L., Cumming, G.S. and Carpenter, S.R., 2002. Multiple states in river and lake ecosystems. *Philosophical Transactions of the Royal Society London B* **357**, 635-645.

Didham, R.K. and Norton, D.A., 2006. When are alternate stable states likely to occur? *Oikos* **113 (2)**, 357.

Diekmann, M., Bramick, U., Lemcke, R., Mehner, T., 2005. Habitat-specific fishing revealed distinct indicator species in German lowland lake fish communities. *Journal of Applied Ecology* **42 (5)**, 901-909.

Diggerfeldt, G., 1988. Reconstruction and regional correlation of Holocene lake-level fluctuations in Lake Bysjön, south Sweden. *Boreas* **17**, 165-182.

Dillon, P.J. and Rigler, F.H., 1974. The phosphorus-chlorophyll relationship in lakes. *Limnology and Oceanography* **19**, 767-773.

Dokulil, M.T. and Teubner, K., 2005. Do phytoplankton communities correctly track trophic changes? An assessment using directly measured and palaeolimnological data. *Freshwater Biology* **50 (10)**, 1594-1610.

Dokulil, M.T. and Skolout, C., 1986. Succession of phytoplankton in a deep stratifying lake: Mondsee, Austria. *Hydrobiologia* **138**, 9-24.

Donabaum, K., Schagerl, M. and Dokulil, M.T., 1999. Integrated management to restore macrophytes domination. *Hydrobiologia* **395/396**, 87-97.

Dorsetcoast, 2006. At www.dorsetcoast.com/media/waterlegislationbriefingpaper (cited on 26/09/06)

Dumayne, L. and Barber, K.E., 1994. The impact of Romans on the environment of Northern England; pollen data from three sites close to Hadrians Wall. *The Holocene* **4** 165-173.

Duthie, H.C., Yang, J-R., Edwards, T.W.D., Wolfe, B.B. and Warner, B.G., 1996. Hamilton Harbour, Ontario: 8300 years of environmental change inferred from microfossil and isotopic analysis. *Journal of Palaeolimnology* **15**, 79-97.

Eckmeier, E., 2007. Conversion of biomass to charcoal and the carbon mass balance from a slash-and-burn experiment in a temperate deciduous forest. *The Holocene* **17**, 539-542.

Edmonds, M., 1993. Interpreting causewayed enclosures in the past and present. In; Tilley, C. (ed). *Interpretative Archaeology*. Oxford Press.

Edmonds, M., 1995. *Stone Tools and Society*. London, Batsford.

Edmondson, W.T., 1991. *The Uses of Ecology; Lake Washington and Beyond.* University of Washington Press, Seattle.

Edward, S. and Deevey, Jr., 1984. Stress, strain and stability of lacrustine ecosystems. In: Hayworth, E.Y. and Lund, J.W.G., (eds.). *Lake Sediments and Environmental History*. University of Minnisota Press, Minneapolis.

Edwards, K.J. and Whittington, G., 2001. Lake sediments, erosion and landscape change during the Holocene in Britain and Ireland. *Catena* **42 (2-4)**, 143-173.

Eggilshaw, H.J. and Shackley, P.E., 1971. Suspended organic matter in fast-flowing streams in Scotland. I: Downstream variations in microscopic particles. *Freshwater Biology* **1 (3)**, 273-285.

Eilers, J.M., Kann, J., Cornett, J., Moser, K. and St. Amand, A., 2004. Palaeolimnological evidence of change in a shallow, hypereutrophic lake: Upper Klamath Lake, Oregon, USA. *Hydrobiologia* **520 (1-3)**, 7-18.

Ekdahl, E.J., Teranes, J.L., Guilderson, T.P., Turton, C.L., McAndrews, J.H., Wittcop, C.A. and Stoermer, E.F., 2004. Prehistorical record of cultural eutrophication from Crawford Lake, Canada. *Geological Society of America* **32 (9)**, 745-748.

El-Bestawy, E., 2000. X-ray Microanalytical study on *Cyclotella meneghiniana* (Bacillariophyceae) as a bioindicator for metal pollution in marine and freshwater environments. *Pakistan Journal of Biological Sciences* **3 (9)**, 1500-1505.

Elias, S.A. and Toolin, L.J., 1990. Accelerator dating of a mixed assemblage of Late Pleistocene insect fossils from the Lamb Spring Site, Colerado. *Quaternary Res*earch **36**, 307-321.

Ellis-Gruffydd, I.D., 1977. Late Devensian Glaciation in the Upper Usk Basin. *Cambria* **4**, 46-55.

Ellis. C.J. and Tallis, J.H., 2001. Climatic control of peat erosion in a North Wales blanket mire. *New Phytologist* **152**, 313-324.

Elmqvist, T., Folke, C., Nyström, M., Peterson, G., Bengtsson, J., Walker, B. and Norberg, J., 2003. Response diversity, ecosystem change and resilience. *Frontiers in Ecology and the Environment* **1 (9)**, 488-494.

Elner, J.K. and Happey-Wood, C. M., 1978. Diatom and crysophyte cyst profiles in sediment cores from two linked but contrasting Welsh Lakes. *Journal of British Phycology* **13**, 341-360.

Elner, J.K. and Happey-Wood, C. M., 1980. The history of two linked but contrasting lakes in North Wales from a study of pollen, diatoms and chemistry in sediment cores. *Journal of Ecology* **68**, 95-121.

Elton, C.S., 1958. *Ecology of invasions by animals and plants.* Chapman and Hall, London.

Engel, S., 1988. The role and interactions of submersed macrophytes in a shallow Wisconsin Lake USA. *Journal of Freshwater Ecology* **4**, 329-342.

Engstrom, D.R. and Fritz, S.C., 2006. Coupling between primary terrestrial succession and the trophic development of lakes at Glacier Bay, Alaska. *Journal of Palaeolimnology* **35**, 873-880

Engstrom, D.R, Fritz, S.C., Almendinger, J.E. and Juggins, S., 2000. Chemical and biological trends during lake evolution in recently deglaciated terrain. *Nature* **408**, 161-166.

Engstrom, D. R. and Wright, H. E. Jr., 1984. Chemical stratigraphy of lake sediments as a record of environmental change. In; Haworth, E. Y. and Lund, W. G., (Eds.). *Lake Sediments and Environmental History.* University of Minnesota Press, Minneapolis.

Etienne, J.L., Jansson, K.N., Glasser, N.F., Hambrey, M.J., Davies J.R., Waters, R.A., Maltman, A.J. and Wilby, P.R., 2006. Palaeoenvironmental interpretation of an ice-contact glacial lake succession: an example from the late Devensian of southwest Wales, UK. *Quaternary Science Reviews* **25**, 739-762.

European Union, 2000. Directive 2000/60/EC of the European Parliament and the Council of 23 October 2000 on establishing a framework for community action in the field of water policy. *The Official Journal of the European Community* **L327**, 1-327.

Evans, G.H. and Walker, R., 1977. The Lake Glacial history of the diatom flora of Llyn Clyd and Llyn Glas, two small oligitrophic high mountain tarns in Snowdonia (Wales). *New Phytologist* **78**, 221-236.

Eyles N. and McCabe, A.M., 1989. The Late-Devensian Irish Sea Basin: The sedimentary record of a collapsed ice sheet margin. *Quaternary Science Reviews* **8 (4)**, 307-351.

Faegri, K. and Iversen, J., 1989. *Textbook of Pollen Analysis.* John Wiley and Sons, Chichester.

Fellerhoff, C., Voss, M. and Wantzen, K.M., 2003. Stable carbon and nitrogen isotope signatures of decomposing tropical macrophytes. *Aquatic Ecology* **37**, 361-375, 2003.

Flower, R.J., 1993. Diatom preservation- Experiments and observations on dissolution and breakage in modern and fossil material. *Hydrobiologia* **269**, 473-484.

Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L. and Holling, C.S., 2004. Regime shifts, resilience, and biodiversity in ecosystem management. *Annual Review of the Ecology and Evolution of Systems* **35**, 557-581.

Forrester, J.W., 1976. *Principles of Systems: 2nd Edition.* MIT Press, Cambridge.

Forster, T. 1993. *Carmarthenshire Birds.* Carmarthenshire Ornithological Records Committee, Carmarthenshire,Wales.

Fossit, J.A., 2000. *A Guide to Habitats in Ireland.* The Heritage Council, Ireland.

Fox, C.F., 1946. Finds of the Early Iron Age from Llyn Cerrig Bach, Anglesey. National Museum of Wales, Cardiff.

Foy, R.H., Lennox, S.D. and Gibson, C.E., 2003. Changing perspectives on the importance of urban phosphorus inputs as the cause of nutrient enrichment in Lough Neagh. *The Science of the Total Environment* **310 (1-3)**, 87-99.

Fozzard, I., Doughty, R., Ferrier, R.C., Leatherland, T., and Owen, R, 1999. A quality classification for management of Scottish standing waters. *Hydrobiologia* **395/396**, 443-453.

France , R.L., 1995. Differentiation between littoral and pelagic food webs in lakes using stable isotopes. *Limnology and Oceanography* **40 (7)**, 1310-1313.

French, C.N. and Moore, P.D., 1986. Deforestation, cannabis cultivation and schwingmoor formation at Cors Llyn (Llyn Mire), central Wales. *New Phytology* **102**, 469-482.

Frey, D.G. 1960. The ecological significance of Cladocera remains in lake sediments. *Ecology* **41**, 684–699.

Frey, D.G., 1982. Contrasting strategies of gamogenesis in northern and southern populations of Cladocera. *Ecology* **63**, 223-241.

Fritz, S.C., 1989. Lake development and limnological response to prehistoric and historic land-use in Diss, Norfolk, UK. *Journal of Ecology* **77**, 182-202.

Fritz, S.C., Kingston, J.C. and Engstrom, D.R., 1993. Quantitative trophic reconstruction from sedimentary diatom assemblages: a cautionary tale. *Freshwater Biology* **30**, 1-23.

Fryer, G., 1972. Observations on the ephippia of certain macrothricid cladocera. *Zoological Journal of the Linneaus Society* **51**, 79-96.

Fryer, G. And Frey, D.G., 1981. Two egged ephippia in the chydorid Cladocera. *Freshwater Biology* **11 (4)**, 391-394.

Gaillard, M.J., 1984. *A palaeohydrological study of Krageholmssjön (Scania, south Sweden). Regional vegetation history and water-level changes.* Diatom Analysis, Lundqua Report 25, Department of Quaternary Geography, Lund University, Lund.

Gaillard, M.J., Dearing, J.A., El-Daoushy, F., Enell, M. and Håkansson, H., 1991. A late Holocene record of land-use history, soil erosion, lake trophy and lake-level fluctuations at Bjäresjösjön (South Sweden). *Journal of Palaeolimnology* **6**, 51-81.

Gaillard, M.J. and Diggerfeldt, G., 1991. Palaeohydrological studies and their contribution to palaeoecological and palaeoclimatic reconstructions. *Ecological Bulletins* **41**, 275-282.

Gaillard, M.J., Birks, H.J.B., Emmanuelsson, U. and Berglund, B.E., 1992. Modern pollen/land-use relationships as an aid in the reconstruction of past land-uses and cultural landscapes: an example from south Sweden. *Vegetation History and Archaeobotony* **1**, 3-17.

Gandouin, E., Franquet, E. and Van Vliet-Lanoë, B., 2005. Chironomids (Diptera) in river floodplains: their status and potential use for palaeoenvironmental reconstruction purposes. *Arch. Hydrobiol.* **162**, 511-534.

Gandouin, E., Maasri, A., Van Vliet-Lanoë, B. And Franquet, E., 2006. Chironomid (Insecta: Diptera) assemblages from a gradient of lotic and lentic waterbodies in river floodplains of France: a methodological tool for palaeoenvironmental applications. *Journal of Palaeolimnology* **35**, 149-166.

Gassner, H., Wanzenbock, J., Zick, D., Tischler, G. and Pamminger-Lahnsteiner, B., 2005. Development of a fish based lake typology for natural Austrian lakes > 50 ha based on the reconstructed historical fish communities. *International Review of Hydrobiology* **90 (4)**, 422-423.

Gaudin, L., Marguerie, D. and Lanos, P., 2006. Correlations between spatial distributions of pollen data, archaeological records and physical parameters from north-western France: a GIS and numerical analysis approach. *Vegetation History and Archaeobotany* **17 (5)**, 585-595.

Germain, H., 1981. *Floré des Diatomeés.* Société Nouvelle des Éditions Boubée, Paris.

Geyh, M.A., Krumbein, W.E. and Krudass, H.R., 1974. Unreliable ^{14}C dating of long stored deep-sea sediments due to bacterial activity. *Marine Geol*ogy **17**, M45-M50.
Girvan, J. and Foy, R.H., 2006. Trophic stability in an Irish mesotrophic lake: Lough Melvin. *Aquatic Conservation* **6**, 623-636.

Glew, J., 1991. Minature gravity corer for recovering short sediment cores. *Journal of Paleolimology* **5**, 285-287.

Godwin, H., 1940. Pollen analysis and forest history of England and Wales. *Linnean Society of London*, **14 March**, 370-399.

Goldman, C.R., Marzolf, E.R. and Elser, J.J., 1990. Phosphorus and nitrogen limitation of phytoplankton growth in the freshwaters of North America: A review and critique of experimental enrichment. *Canadian Journal of Fisheries and Sciences* **47 (7),** 1468-1477.

Goulden, C.E., 1970. The fossil fauna and fauna (other that siliceous fossils, pollen and chironomid head capsules). In; Hutchinson, I. (ed.). An account of the history and development of Lago di Monterosi, Latinum, Italy. *Transactions of the American Philosophical Society* **60/4**, 102-111.

Graham, L.E. 1993. *Origin of Land Plants.* John Wiley and Sons, Inc. New York.

Green, S., 1986. The Palaeolithic Settlement of Wales Research Project: A Review of Progress 1978-1985. In; Collcutt, S.N. (ed). The Palaeolithic of Britain and Its Nearest Neighbours: Recent Trends. John R. Collis, Sheffield.

Greig, J.R.A., 1987. Pollen and Plant Macrofossils. In; Smith, C.A. and Lynch, F.M., *Cambrian Archeological Monographs* **3**, 39-44.

Griffin, J.J. and Goldberg, E.D., 1979. Morphologies and origin of elementary carbon in the environment. *Science* **206**, 563-565.

Griffiths, B.M., 1939. Early references to water blooms in British lakes. *Proceedings of the Linneaean Society* (London) **151 (1)**, 14-19.

Grimm, E., 1987.Coniss: a FORTRAN 77 program for stratigraphically constarined cluster analysis by the method of incremental sum of squares. *Computers and Geosciences* **13**, 13-35.

Grimm, E., 1991. *Tilia and Tiliagraph.* Springfield, Illinois State Museum, Illinois.

Grimm, E., 2004. *TG View.* Springfield, Illinois State Museum, Illinois.

Grölund, T. and Kauppila, T., 2002. Holocene history of Lake Soldatskije (Kola Peninsula, Russia) inferred from sedimentary diatom assemblages. *Boreas* **31**, 273-284.

Gu, B., Schelske, C.L. and Brenner, M., 1996. Relationship between sediment and plankton isotope ratios ($\delta^{13}C$ and $\delta^{15}N$) and primary productivity in Florida Lakes. *Canadian Journal of Fish and Aquatic Science* **53,** 875-883.

Guildbert, G., 1975. Moel-y-Gaer, 1973: an area excavation on the defences. *Antiquity* **49 (194)**, 109-117.

Guisan, A. and Theurillat, J.P., 2000. Assessing alpine plant vulnerability to climate change: a modelling perspective. *Integrated Assessment* **1 (4)**, 307-320.

Guiter, F., Andrieu-Ponel, V., Diggerfeldt, G., Reille, M., de Beaulieu, J-L. and Ponel, P., 2005. Vegetation history and lake-level changes from the Ypunger Dryas to the present in Eastern Pyrenees (France): pollen, plant macrofossils and lithostratigraphy from Lake Racou (2000 m a.s.l.). *Vegetation History and Archaeobotony* **14 (2)**, 99-118.

Hakala, A., Sarmarja-Korjonen, K. and Miettinen, A., 2004. The origin and evolution of lake Lake Vähä-Pitcusta, SW Finland – a multi-proxy study of a meromictic lake. *Hydrobiologia* **527**, 85-97.

Håkansson, H. and Regnéll, J., 1993. Diatom succession related to land-use during the last 6000 years: a study of small eutrophic lakes in southern Sweden. *Journal of Palaeolimnology* **8**, 49-69.

Hall, R.I., Leavitt, P.R., Smol, J.P. and Zirnhelts, N., 1997. Comparison of diatoms, fossil pigments and historical records of measures of lake eutrophication. *Freshwater Biology* **38**, 401-417.

Handa, S and Moore, P., 1976. Studies of the vegetational history of Mid-Wales. IV Pollen analysis of some pingo basins. *New Phytologist* **77**, 205-225.

Hann, B.J., 1990. Cladocera. In: Warner, B.G., (ed.). *Methods in Quaternary Ecology*. Geoscience Canada **5**, 81-91.
Hannon, G.E. and Gaillard, M.J., 1997. The plant macrofossil record of past lake-level changes. *Journal of Palaeolimnology* **18 (1)**, 15-28.

Hanson, W.S., 1999. Roman Britain; the Military Dimension. In; Hunter, J. and Ralston, I. (eds). The Archaeology of Britain; an Introduction from the Upper Palaeolithic to the Industrial Revolution. Routledge, London.

Hargeby, A., Andersson, G., Blindow, I. And Johansson, S, 1994. Trophic web structure in a shallow eutrophic lake during a dominance shift from phytoplankton to submerged macrophytes. *Hydrobiologia* **280**, 83-90.

Harmata, K., 1995. Traces of Human impact reflected in the pollen diagram from Tarnowiec Mire near Jaslo, SE Poland. *Vegetational History and Archaeobotony* **4 (4)**, 235-243.

Harris, G.P., 1994. Patterns, process and prediction in aquatic ecology. A limnological view of some general ecological problems. *Freshwater Biology* **32**, 143-160.

Harris, M.A., Cumming, B.F. and Smol, J.P., 2006. Assessment of recent environmental changes in New Brunswick (Canada) lakes based on paleolimnological shifts in diatom species assemblages. *Canadian Journal of Botany* **84 (13)**, 151-163.

Haselgrove, C., The Iron Age. In; Hunter, J. and Ralston, I. (eds). The Archaeology of Britain; an Introduction from the Upper Palaeolithic to the Industrial Revolution. Routledge, London.

Haslam, S.M., Sinker, C.A. and Wolsley, P.A., 1995. British Water Plants. *Field Studies* **4**, 243-351.

Haworth, E.Y., Pinder, L.C.V., Lishman, J.P. and Duigan, C., 1998. The Anglesey lakes, Wales, UK – A palaeolimnological study of the eutrophication and conservation status. *Aquatic Conservation : Marine and Freshwater Ecosystems* **2 (2)**, 61-80.

Hecky, R. E. and Kilham, P., 1973. Diatoms in Alkaline, Saline Lakes: Ecology and Geochemical Implications. *Limnology and Oceanography* **18 (1)**, 53-71.

Hedenström, A. and Risberg, J., 1999. Early Holocene shore-displacement in southern central Sweden as recorded in elevated isolated basins. *Boreas* **28**, 490-504.

Hedges, R.E.M., Housley, R.A., Law, I.A. and Bronk, C.R., 1989. Radiocarbon dates from the Oxford AMS system: Archaeometry datelist 9. *Archaeometry* **31**, 207-234.

Heinselman, M.L., 1973. Fire in the Virgin Forests of the Boundary Waters Canoe Area, Minnesota. *Quaternary Research* **3**, 329-382.

Heiri, O. and Lotter, A.F., 2003. 9000 years of chironomid assemblage dynamics in an Alpine lake: long term trends, sensitivity to disturbance and resilience of the fauna. *Journal of Palaeolimnology* **30 (3)**, 273-289.

Heiri, O., Lotter, A.F. and Lemcke, G., 2001. Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. *Journal of Palaeolimnology* **25**, 101-110.

Henning, S., 2002. Indicators of the long-term eutrophication of a Danish Lake (Karlsø), and water pollution management. *The Journal of Transdisciplinary Environmental Studies* **1 (1)**, 1-10.

Hibbert, F.A. and Switsur, V.W., 1976. Radiocarbon dating of Flandrian zones in Wales and Northern England. *New Phytologist* **77**, 793-807.

Hickman, M. and Reasoner, M.A., (1998). Late Quaternary diatom response to vegetation and climate change in a subalpine lake in Banff national Park, Alberta. *Journal of Palaeolimnology* **20**, 253-265.

Hickman, M. and Schweger, C.E., 1991. A palaeoenvironmental; study of Fairfax Lake, a small lake situated in the Rocky Mountain Foothills of west central Alberta. *Journal of Palaeolimnology* **6**, 1-15.

Hillman, G.C., 1978. Remains of crops and other plants from Carmarthen (Church Street). In; Boon, G.C., (ed). *Cambrian Archeological Association* **1**, 107-112.

Hillman, G.C., 1985. Plant Remains. In; White, R.B. (ed). *Archaeologia Cambrensis* **134**, 101-103.

Hills, C., 1999. Early Historic Britain. In; Hunter, J. and Ralston, I. (eds). The Archaeology of Britain; an Introduction from the Upper Palaeolithic to the Industrial Revolution. Routledge, London.

Hilton, J. and Lishman, J.P., 1985. The effects of redox changes in the magnetic susceptibility of sediments from a seasonally anoxic lake. *Limnology and Oceanography* **30**, 907-909.

Hingley, R., 1992. Society in Scotland from 700 BC – AD 200. *Proceedings of the Society of Antiquaries of Scotland* **122**, 7-53.

Hinzman, L.D., Bettez, N.D., Bolton, W.R., Chaplin, F.S., Dyurgerov, M.B., Fastie, C.L., Griffith, B., Hollister, R.D., Hope, A., Huntington, H.P., Jensen, A.M., Jia, G.J., Jorgenson, T., Kane, D.L., *et al.,* 2005. Evidence and implications of recent climate change in Northern Alaska and other Arctic Regions, *Climate Change* **72 (3)**, 251-298.

Hodell, D.A. and Schelske, C.L., 1998. Production, sedimentation and isotopic composition of organic matter in Lake Ontario. *Limnology and Oceanogaphy* **43**, 200-214.

Hoerling, M., Hurrell, J.W. and Xu, T., 2001. Tropical origins for recent North Atlantic climate change. *Science* **292**, 90-92.

Hogg, A.H.A. and King, D.J.C., 1963. Early Castles in Wales. *Archaeologia Cambrensis* **112**.

Hollander, D.J., MacKenzie, J.A. and ten Haven, H.L., 1992. A 200 year sedimentary record of progressive eutrophication in Lake Greifen (Switzerland): Implications for the origin of organic rich sediments. *Geology* **20**, 825-828.

Holling, C.S., 1973. Resilience and stability of ecological systems. *Annual Review of Ecological Systems* **4**, 1-23.

Holmes, N.T.H., Boon, P.J. and Rowell, T.A., 1998. A revised classification system for British rivers based on their aquatic plant community. *Aquatic Conservation: Marine and Freshwater Ecosystems* **8 (4)**, 555-578.

Hope, G.S. and Peterson, J.A., 1976. Palaeo-environments. In; Hope, G.S., Peterson, J.A., Radok, U. and Allison, I., 1976. *The Equitorial Glaciers of New Guinea,* 173-205. A.A. Balkema, Rotterdam.

Hörnberg, G., Bohlin, E., Hellberg, E., Bergman, I., Zackrisson, O., Olofsson, A., Wallin, J-E. and Påsse, T., 2006. Effects of Mrsolithic hunter-gatherers on local vegetation in a non-uniform glacio-isostatic land uplift area, northern Sweden. *Vegetation History and Botany* **15 (1)**, 13-26.

Howard-Williams, C., 1981. Studies on the ability of a *Potamogeton pectinalis* community to remove dissolved nitrogen and phosphorus compounds from lake water. *Journal of Applied Ecology* **18**, 619-637.

Huber, J.K., 1996. A postglacial pollen and nonsiliceous algae record from Gegoka Lake, Lake County, Minnesota. *Journal of Palaeolimnology* **16 (1)**, 23-35.

Hughes, H.H., 1922. Prehistoric remains on Penmaenmawr. *Archaeological Cambrensis* **77**, 346-360.

Hughes, P.D.M. and Barber, K.E., 2003. Mire development across the fen-bog transition on Teifi floodplaon at Tregaron Bog, Ceredigion, Wales, and a comparison with 13 other raised bogs. *Journal of Ecology* **91**, 253-264.

Hughes, P.D.M., Mauquoy, D., Barber, K.E. and Langdon, P.G., 2000. Mire-development pathways and palaeoclimatic records from a full Holocene peat archive at Walton Moss, Cumbria, England. *The Holocene* **10 (4)**, 465-479.

Huntley, B. And Birks, H.J.B., 1983. *An Atlas of Past and Present Pollen Maps for Europe: 0-13,000 Years Ago.* Cambridge University Press, Cambridge.

Hutchinson, G.E, 1967. *A Treatise on Limnology. II. Introduction to Lake Biology and the Limnoplankton.* John Wiley and Sons Inc., New York.

Hutchinson, G.E. and Goulden, C.E., 1966. The history of Laguna de Petenxil: The plant macrfossils. *Mem. Connect. Acad. Arts Sci.* **17**, 67-73.

Huttunen, P. and Meriläinen, J., 1983. Interpretation of lake quality from contemporary diatom assemblages. *Hydrobiologia* **103**, 91-97.

Hynynen, J., Palomaki, A., Merilainen, J.J., Witick, A. and Mantykoski, K., 2004. Pollution history and recovery of a boreal lake exposed to a heavy bleached pulping effluent load. *Journal of Paleolimnology* **32 (4)**, 351-374.

Ince, J., 1988. Two post-glacial pollen profiles from the uplands of Snodonia, Gwynedd, N. Wales. *New Phytologist* **95**, 159-172.

Ingrem, C., 2000. Pre-historic Intertidal Archaeology in the Welsh Severn Estury. In; Bell, M., Caseldine, A.M. and Neumann, H. (eds.). *Pre-historic Intertidal Archaeology in the Welsh Severn Estury*, CBA Research Report 120, Council for British Archaeologists, 53-54.

Issar, A., 2003. *Climate Changes During the Holocene and Their Impact on Hydrological Systems*. Cambridge University Press.

Iversen, J., 1949. The influence of prehistoric man on vegetation. *Danmarks Geologiske Undersögelse* **3 (6)**, 5-25.

Iversen, J., 1954. The Late-Glacial flora of Denmark and its relation to climate and soil. *Dansk. Geol. Unders.* II. Series No. **80**, 87-119.

Izaguirre, I., Mataloni, G., Vinocur, A. and Tell, G., 1993. Temporal and spatial variations of phytoplankton from Boeckella Lake (Hope Bay, Antarctic Peninsula). *Antarctic Science* **5 (2)**, 137-141.

Jacobi, R.M., 1976. Britain inside and outside Mesolithic Europe. *Proceedings of the Prehistoric Society* **42**, 67-84.

Jacobi, R.M., 1978. Population and Landscape in Mesolithic lowland Britain. In; Limbrey, S. and Evans, J.G. (eds.). *The Effect of Man on the Landscape: The Lowland Zone.* CBA Research Report 21, 75-85.

Jacobi, R.M., 1979. Early Flandrian Hunters in the south-west. *Devon Archaeological Society* **37**, 48-93.

Jacoby, J.G. and D'Arrigo, R., 1989. Reconstructed Northern hemisphere annual temperature since 1671 based on high-latitude tree-ring data. *Climatic Change* **14 (1)**, 39-59.

James, W. F. and Barko, J. W., 1990. Macrophyte influences on the zonation of sediment accretion and composition in a north temperate reservoir. *Archiv für Hydrobiologie* **120**, 129-142.

Janokovská, V. and Komárek, J., 2000. Indicative Value of *Pediastrum* and Other Green Algae in Palaeoecology. *Folia Geobotanica* **35**, 59-82.

Janokovská, V. and Panova, N.K., 1998. Late Glacial and Holocene history of vegetation in Polar Ural region, Russia. Abstract on "*The 5th European Palaeobotanical-Palynological Conference*", Poland, Kraków, June 26-30, 1998.

Janse, J. H., 1997. A model of nutrient dynamics in shallow lakes in relation to multiple stable states. *Hydrobiologia* **342/343**, 1-8.

Janssen, G.R., 1968. Myrtle Lake: A Late and Postglacial pollen diagram from northern Minnesota. *Canadian Journal of Botany* **46**, 1394-1408.

Jansson, K.N. and Glasser, N.F., 2005. Palaeoglaciology of the Welsh sector of the British-Irish ice sheet. *Journal of the Geological Society* **162 (1)**, 25-37.

Jenny H (1980) *The Soil Resource Handbook.* Springer-Verlag, New York.

Jeppesen, E., 2001. Functional ecology and palaeolimnology: using cladoceran remains to reconstruct anthropogenic impact. *Trends in Ecology and Evolution* **16 (4)**, 191-198.

Jeppesen, E., Jensen, J.P., Søondergaard, M. and Lauridsen, T.L., 2005. Response of fish and plankton to nutrient loading reduction in eight shallow Danish lakes with special emphasis on seasonal dynamics. *Freshwater Biology* **50,** 1616-1627.

Johnson, C.A. and Naiman, R.J., 1990. Aquatic patch creation in relation to beaver population trends. *Ecology* **71**, 1617-1621.

Johnson, W.K., 1998. *The Palaeoecology of the Late Glacial and Holocene Sediments of the Talley Lakes, Carmarthenshire.* Unpublished PhD. Thesis, University of Wales, Aberystwyth.

Jones, R. and Benson-Evans, K., 1974. Nutrient and phytoplankton studies of Llangorse Lake, a eutrophic lake in the Brecon Beacons National Park. *Field Studies* **4**, 61-65.

Jones, R., Benson-Evans, K., Chambers, F.M., Abell, B. and Tai, Y.C., 1978. Biological and chemical studies of sediments from Llangorse Lake, Wales. *Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandle Limnologie* **20**, 642-648.

Jones, R., Benson-Evans, K. and Chambers, F.M., 1985. Human influence upon sedimentation in Llangorse Lake, Wales. *Earth Surface Processes and Landforms* **10**, 1377-1382.

Jones, R., Chambers, F.M. and Benson-Evans, K., 1991. Heavy metals (Cu and Zn) in recent sediments of Llangorse Lake, Wales: non-ferrous smelting, Napoleon and the price of wheat - a paleoecological study. *Hydrobiologia* **214**, 149-154.

Jones, G. and Miles, A., 1984. Plant Remains from Arddleen. In Britnell, W.J. and Musson, C.R. (eds). *Archaeologia Cambrensis* **133**, 96-97.

Jongman, R.H.G., ter Braak, C.J.F. and van Tongren, O.F.R., 1995. Data Analysis in Community Landscape and Ecology. Cambridge University Press.

Jordan, P., Menary, W., Daley, K., Kiely, G., Morgan, G., Byrne, P. and Moles, R., 2005. Patterns and processes of phosphorus transfer from Irish grassland soils to rivers-integration of laboratory and catchment studies. *Journal of Hydrology* **304**, 20-34.

Kaland, P.E., Krzywinski, K. and Stabell V., 1984. Radiocarbon dating of transitions between marine and lacrustine sediments and their relation to the development of lakes. *Boreas* **13**, 243-258.

Kamenik, C., Schmidt, R., Kum, G. and Psenner, R., 2001. The Influence of catchment Characteristics on the Water Chemistry of Mountain Lakes. *Arctic, Antarctic and Alpine Research* **33**, 404-409.

Kamenir, Y., Dubinsky, Z., Zohary, T., 2004. Phytoplankton size structure stability in a meso-eutrophic subtropical lake. *Hydrobiologia* **520**, 89-104.

Karst, T.M. and Smol, J.P., 2000. Palaeolimnological evidence of limnic nutrient concentration equilibrium in a shallow macrophyte-dominated lake. *Aquatic Science* **62**, 20-38.

Kaushal, S. and Binford, M.W., 1999. Relationship between C: N ratios of lake sediments, organic matter sources, and historical deforestation in Lake Pleasant, Massachusetts, USA. *Journal of Palaeolimnology* **22**, 439-442.

Kawecka, B. and Olech, M., 1993. Diatom Communities in the vanishing Ornithologist Creek, King George Island, South Shetland, Antarctica. *Hydrobiologia* **269**, 327-333.
Kerfoot, W.C., 1981. Long-term replacement cycles in cladoceran communities. *Ecology* **55**, 51-61.

Keeley, H.C.M., 1986. The Soils. *Cambrian Archaeological Monographs* **3**, 35-38.

Kelly, M., Juggins, S., Guthrie, R., Pritchard, S., Jamieson, J., Rippey, B., Hirst, H. and Pritchard, S., 2008. Assessment of ecological status in UK rivers using diatoms. *Freshwater Biology* **53**, 403-422.

Kenward, H.K., 1975. The biological and archaeological implications of the beetle *Aglenus brunneus* (Gyllenhal) in ancient faunas. *Journal of Archaeological Science,* **2**, 63-69.

Kerfoot, W.C., 1981. Long-term replacement cycles in cladoceran communities: a history of predation. *Ecology* **62**, 216-233.

Kilham, S.S., 1986. Dynamics of Lake Michigan natural phytoplankton communities in continuous cultures along a Si: P loading gradient. *Canadian Journal of Fisheries and Aquatic Sciences* **43**, 351-360.

Kilham, S.S., Theriot, C. and Fritz, S. C., 1996. Linking planktonic diatoms and climate change in the large lakes of the Yellowstone ecosystem using resource theory. *Limnology and Oceanography* **41 (5)**, 1052-1062.

Kilinç, S. and Sivaci, R., 2001. A study on the past and present diatom flora of two alkaline lakes. *Turkish Journal of Botany* **25**, 373-378.

Kitner, M. and Poulíčková A., 2003. Littoral distoms as indicators for the eutrophication of shallow lakes. *Hydrobiologia* **506-509**, 519-524.

Knudson, B.M., 1957. Ecology of the epiphytic diatom Tabelleria flocculosa (Roth) Kutz. Var. flocculosa in three English lakes. *Journal of Ecology* **45 (1)**, 93-112.

Köhler, J., Hilt, S., Adiran, R., Nicklisch, A. Kozerski, H.P. and Walz, N., 2005, *Freshwater Biology* **50**, 1639-1650.

Komárek, J. and Jankovská, V., 2001. Review of the green algal genus *pediastrum*: implications for pollen analytical research. *Bibliotheca Phycologia* **108**, J. Cramer, Stuttgart.

Korhola, A. and Blom, T., 1996. Marked early 20th century pollution and the subsequent recovery of Töölö Bay, central Helsinki, as indicated by subfossil diatom assemblage changes. *Hydrobiologia* ***341***, 169-179.

Korhola, A., Weckström, J., Holmström, L. and Erästo, P., 2000. A Quantitative Holocene climate record from diatoms in Northern Fenoscandia. *Quaternary Research* **54 (2)**, 284-294.

Köster, D. and Pienitz, R., 2006. Seasonal diatom variability and palaeoinferences – a case study. *Journal of Paleolimnology* **35**, 395-416.

Krammer, K. and Lang-Bertalot, H., 1986. *Süßwasserflora von Mitteleuropa. Bacillariophycae. 1 Tiel: Naviculaceae. Vol. 2/1.* Gustav Fischer Verlag, Stuttgart.

Krammer, K. and Lang-Bertalot, H., 1988. *Süßwasserflora von Mitteleuropa. Bacillariophycae. 2 Tiel: Epithemiaceae, Bacillariaceae, Surirellaceae.* Vol. 2/2. Gustav Fischer Verlag, Stuttgart.

Krammer, K. and Lang-Bertalot, H., 1991a. *Süßwasserflora von Mitteleuropa. Bacillariophycae. 3 Tiel: Centarles, Fragilariaceae, Eunotiaceae. Vol. 2/3.* Gustav Fischer Verlag, Stuttgart.

Krammer, K. and Lang-Bertalot, H., 1991b. *Süßwasserflora von Mitteleuropa. Bacillariophycae. 4 Tiel: Achnanthaceae. Vol 2/4.* Gustav Fischer Verlag, Stuttgart.

Kravtsova, L.S., Weinberg, I.V., Khanaevl, I.V., Sherbakov, D.Yu., Semovsky, S.V., Adov, F.V., Rozhkova, N.A., Pomazkina, G.V., Shevelyova, N.G. and Kaygorodova, I.A., 2006. The forming of a fouling community on artificial substrate in Lake Baikal. *Hydrobiologia* **568**, 51-55.

Krull, J.N., 1970. Aquatic plant macroinvertebrate associations and waterfowl. *Journal of Wildlife Management* **34 (4)**, 707-718.

Kufel, L. and Kufel, I., 2002. Chara beds acting as nutrient sinks in shallow lakes – a review. *Aquatic Botany* **72 (3-4)**, 249-260.

Lacaille, A.D. and Grimes, W.F., 1955. The pre-history of Caldy. *Archaologia Cambrensis* **104**, 85-165.

Lamb, A.L., Leng, M.J., Mohammed, U.M. and Lamb, A.L., 2004. Holocene climate and vegertaion change in the Main Etheopian Rift Valley, inferred from the composition (C/N and $\delta^{13}C$) of lacustrine organic matter. *Quaternary Science Reviews* **23**, 7-8.

Lamb, H.H., 1985. Climate and Landscape in the British Isles. In; Woodell, S.R.J. (ed.) *The English Landscape, Past, Present and Future.* Oxford University Press, Oxford, 148-233.

Lamb, H.H., 1997. *Climate History and the Modern World.* Routeledge, London.

Lambert, C.A., 1963. Report on peat samples from the submerged forest at Freshwater West. *Procedings of the Prehistorical Society* **29**, 128-130.

Lammens, E.H.R.R., 1989. Causes and consequences of the success of bream in Dutch eutrophic lakes. *Hydrobiological Buletin* **23**, 11-18.

Lammens, E.H.R.R., Klein Breteler, J.G.P., Mooij, W.M. and Claassen, T., 1997. The direct and indirect effects of a seine and gill-net fishery on fish stock and water quality in the Frisian Lakes, The Netherlands. *Canadian Journal of Fisheries and Aquatic Science*, submitted. Cited in; Scheffer, M., 2004. *Ecology of Shallow Lakes.* Population and Community Biology Series, Kluwer Academic Publishers, London.

Lampeter, 2005. At www.lampeter.org/english/history/railway (cited on; 19/12/05.

Langdon Brooks, J., 1959. Cladocera. In; Edmondson, W.T., (ed.) *Freshwater Biology.* Wiley, London.

Lau, S.S.S. and Lane, S.N., 2001. Continuity and change in environmental systems: the case of shallow lake ecosystems. *Progress in Physical Geography* **25 (2)**, 178-202.

Lauderback, L.A. and Rhode, D.E., 2008. 15,000 years of vegetation change in the Bonneville basin, the Blue Lake pollen record. *Quaternary Science Reviews* **27 (17-18)**.

Lawton, J.H., 1994. What do species do in ecosystems? *Oikos* **71**, 367-374.

Lazzarotto, P., Prasuhn, V., Butscher, E., Crespi, C., Flühler, H., Stamm, C., 2005. Phosphorus export dynamics from two Swiss grassland catchments, *Journal of Hydrology* **304**, 139-150.

Lehmann, A., 1998. GIS modelling of submerged macrophyte distribution using Generalized Additive Models. *Plant Ecology* **139 (1)**, 113-124.

Leira, M., Jordan, P., Taylor, D., Dalton, C., Bennion, H., Rose, N. And Irvine, K., 2006. Assessing the ecological status of candidate reference lakes in Ireland using palaeolimnology. *Journal of Applied Ecology* **43**, 816-827.

Leland, J.J., 2003. Macrophyte-dominated and turbid states of shallow lakes: Evidence from Alberta Lakes. *Ecosystems* **6**, 213-223.

Levin, S.A., 1992. The problem of pattern and scale in ecology. *Ecology* **73**, 1943-1967.

Lewis, S., 1833. *A Topographical Dictionary of Wales*, Vol II (First Edition).

Lewis, P.R. and Jones, G.D.B., 1969. The Dolaucothi Gold Mines I: the surface evidence. *The Antiquaries Journal* **XLIX**, 244-272.

Lewis, C.A. and Thomas, G.S.P., 2005. The Upper Wye and Usk regions. In; Lewis, C.A. (ed.), *The Glaciations of Wales and Adjoining Regions.* Longman Press, London.

Likens, G. E., 1985. *An Ecosystem Approach to Aquatic Ecology; Mirror Lake and its Environment.* Springer-Verlag, New York.

Likens, G.E., Borman, F.H., Pierce, F.H., Eaton, J.S. and Johnson, N.M., 1977. *Biogeochemistry of a forested ecosystem.* Springer-Verlag, New York.

Lillie, M., Page, N., Kirby, J. and Griffiths, H., 2003. Neolithic-Bronze Age landscape development at Machynys, Llanelli. *Archaeology in Wales* **43**, 45-57.

Livingstone, D.A., 1955. A lightweight piston sampler for lake deposits. *Ecology* **36**, 137-139.

Lloyd, D. (1998) The Medicinal Leech (Hirudo medicinalis) in Wales. *CCW Contract Science Report* **311**.

Lobo, E.A., Katoh, K. and Aruga, Y., 1995. Response of epilithic diatoms to water pollution in rivers in the Tokyo Metropolitan area. *Freshwater Biology* **34**, 191-204.

Lotter, A.F., 1998. The recent eutrophication of Baldeggersee, (Switzerland) as assessed by fossil diatom assemblages. *The Holocene* **8 (4)**, 395-405.

Lotter, A.F., 2001. The palaeolimnology of Soppensee (Central Switzerland), as evidenced by diatom, pollen and fossil pigment analysis. *Journal of Palaeolimnology* **25 (1)**, 65-79.

Lowe, J.J. and Walker M.J.C., 1999. Reconstructing Quaternary Environments: 2nd Edition. Longman Press.

Lynch, F. and Musson, C., 2001. A prehistoric and early medieval complex at Llandegi

MacArthur, R.H., 1955. Fluctuations of animal populations and a measure of community stability. *Ecology* **36**, 533-536.

Macklin, M.G. and Lewin, J., 2003. River sediments, great floods and centennial-scale Holocene climate change. *Journal of Quaternary Science* **18 (2)**, 101-105.

Manning, S.W., Ramsey, C.B., Kutschera,W., Higham, T., Kromer, B., Steier, P.,Wild, E.M., 2006. Chronology for the Aegean Late Bronze Age 1700-1400 B.C. *Science* **312**, 565 – 569.

Marchetto, A., Lami, A., Musazzi, S., Massaferro, J. Langone, L. and Guilizzone, P., 2004. Lake Maggiore (N. Italy) trophic history: fossil diatom, plant pigments, chironomids and comparison with long-term limnological data. *Quaternary International* **113**, 97-110.

MDKI (MacBeth Division of Kollmorgen Instruments Corportion), 1994. Munsell Soil Colour Charts, 1994 edition. Baltimore, MD.

Maher, B.A. and Thompson, R., 1995. Palaeorainfall reconstructions from pedogenic magnetic susceptibility variations in the Chinese loess and palaeosols. *Quaternary Research* **44**, 383-391.

Manning, W.H., 2001. *A Pocket Guide to Roman Wales*. University of Wales Press and the Western Mail.

Mannion, A.M. and Bowlby, S.R., 1995. *Environmental Issues in the 90's*. John Wiley and Sons, Chichester, England.

Marchetto, A. and Bettinetti, F., 1995. Reconstruction of the phosphorus history of two deep sub-Alpine lakes from sedimentary diatoms, compared with long-term chemical measures. *Mem. Ist. Ital. Idrobiol.* **53**, 27-38.

Mason, C.F., 2002. *Biology of Freshwater Pollution*. Longman Scientific, London.

May, R.M., 1973. *Stability and complexity in model ecosystems.* Princeton University Press.

McArthur, R. H., 1955. Fluctuations of animal populations and a measure of community stability. *Ecology* **36**, 533-536.

McCann, K. S., 2000. The diversity-stability debate. *Nature* **405**, 228-233.

McCarthy, V., Donohue, I. and Irvine, K., 2006. Field evidence for stoichiometric relationships between zooplankton and P and N availability in a shallow calcareous lake. *Freshwater Biology* **51**, 1589-1604.

McBurney, C.B.M., 1959. Report on the first season's field work on British Upper Palaeolithic cave deposits. *Proceeding of the Prehistory Society* **25**, 260-269.

McGowan, S., Leavitt, P.R., Hall, R.I., Anderson, N.J., Jeppersen, E. and Odgaard, B.V., 2005. Controls of algal abundance and community composition during ecosystem state change. *Ecology* **86 (8)**, 2200-2211.

Meijer, M.-L., Raat, A.J.P. and Doef, R.W., 1989. Restoration by biomanipulation of Lake Bleiswijkse Zoom the Netherlands first results. *Hydrobiological Bulletin* **23**, 49-58.

Melzer, A., 1999. Aquatic macrophytes as a tool for lake management. *Hydrobiologia* **395–396**, 181-190.

Mercuri, A.M., Accorsi, C.A. and Mazzanti, M.B., 2002. The long history of Cannabis and its cultivation by the Romans in central Italy, shown by pollen records from Lago Albano and Lago di Nemi. *Vegetation History and Archaeobotony* **11**, 263-276

Meyers, P.A. and Ishiwatari, R., 1993. Lacrustine organic chemistry-an overview of indicators of organic matter sources and diagenesis in lake sediments. *Organic Geochemistry* **20**, 867-900.

Meyers, P.A. and Lallier-Verges, E., 1999. Lacustrine sedimentary organic matter records of Late Quaternary palaeoclimates. *Journal of Palaeolimnology* **21**, 345-372.

Meyers, P.A., Leenheer, M.J. and Bourboniere, R.A., 1995. Diagenesis of vascular plant organic matter components during burial in lake sediments. *Aquatic Geochemistry* **1**, 35-52.

Meyers, P.A. and Teranes, J.L., 2001. Sediment Organic Matter. In: Last, W.M. and Smol, J.P. (eds.). *Tracking Environmental Change Using Lake Sediments. Vol. 2: Physical and Geochemical Methods,* 239-269. Kluwer Academic Publishers Dordrecht, Netherlands.

Mighall, T.M. and Chambers, F.M., 1995. Holocene vegetation history and human impact at Bryn y Castell, Snowdonia, north Wales. *New Phytologist* **130**, 299-321.

Mighall, T.M., Abrahams, P.W., Grattan, J.P., Hayes, D, Timberlake, S. and Forsyth, S., 2002. Geochemical evidence for atmospheric pollution derived from prehistoric copper mining at Copa Hill, Cwmystwyth, mid-Wales, UK. The Science of the Total Environment **1-2**, 69-80.

Mighall, T.M, Grattan, J.P., Timberlake, S., Lees, J.A. and Forsyth, S., 2002a. An atmospheric pollution history for lead-zinc mining from the Ystwyth Valley, Dyfed, mid-Wales, UK as recorded by an upland blanket peat. *Geological Society of London* **2 (2)**, 175-184.

Mighall, T.M., Timpany, S., Blackford, J.J., Innes, J.B., O'Brien, C.E., O'Brien, W. and Harrison, S., 2006. Vegetation change during the Mesolithic and Neolithic on the Mizen Peninsula, Co. Cork, south-west Ireland. *Vegetation History and Archaeobotany* **17 (6)**, 617-628.

Millspaugh, S.H. and Whitlock, C., 1995. A 750-year history based on lake sediment records in central Yellowstone National Park, USA. *The Holocene* **5 (3)**, 283-292.

Mittelbach, G.G., Turner, A.M., Hall, D.J., Rettig, J.E. and Osberg, C.W., 1995. Perturbation and Resilience: A long-term, whole lake study of predator extinction and reintroduction. *Ecology* **76 (8)**, 2347-2360.

Molnar, P., 2004. Late Cenozoic increase in accumulation rates of terrestrial sediment: How might climate change have affected erosion rates? *Annual Review of Earth and Planetary Sciences* **32**, 67-89.

Moore, P.D., 1968. Human influence upon vegetational history in north Cardiganshire. *Nature* **217**, 1006-1007.

Moore, P.D., 1978. Forest Fires. *Nature* **272**, 754.

Moore, P.D., 1990. Studies in the vegetational history of mid-Wales. *New Phytologist* **71 (5)**, 947-959.

Moore, P.D. and Chater, E.H., 1969. The changing vegetation of west-central Wales in the light of human history. *Journal of Ecology* **57**, 361-379.

Moore, P.D., Webb, J.A. and Collinson, M.E., 1991. *Pollen Analysis: 2nd Edition.* Blackwell Science, Oxford.

Moos, M.T., Laird, K.R. and Cumming, B., 2005. Diatom assemblages and water depth in Lake 239 (Experimental Lakes Area, Ontario): implications for paleoclimatic studies. *Journal of Paleolimnology* **34 (2)**, 217-227.

Morris, K., Bailey, P.C., Boon, P.I. and Hughes, L., 2003. *Marine and Freshwater Research* **54 (3)**, 201-215.

Moss, B., 1988. *Ecology of Fresh Waters: Man and Medium, Past to Future. 3rd Edition.* Blackwell Science, Oxford.

Moss, B., 1996. A land awash with nutrients-the problem of eutrophication. *Chemistry and Industry,* 3 June 1996.

Moss, B., Barker, T., Stephen, D., Williams, A.E., Balaya, D.J., Beklioglu, M and Carvalho, L, 2005. Consequences of reduced nutrient loading on a lake system in a lowland catchment: deviations from the norm? *Freshwater Biology* **50**, 1687-1705.

Moss, B., Madgwick, J. and Phillips, G., 1997. *A Guide to the Restoration of Nutrient Enriched Shallow Lakes.* The Broads Authority, Norwich, Norfolk.

Moss, B., McGowan, S. and M., Carvalho, 1994. Determination of phytoplankton crops by top-down and bottom-up mechanisms in a group of English lakes, the West Midland meres. *Limnology and Oceanography* **39**, 1020-1029.

Moss, B., Stansfield, J., Irvine, K., Perrow, M., and Phillips, G., 1996. Progressive restoration of a shallow lake: a 12-year experiment in isolation, sediment removal and biomanipulation. *Journal of Applied Ecology* **33**, 71-86.

Moss, B., Stephen, D., Alvarez, C., Becares, E., Bund, W. van de, Collings, S. E., Donk, E. van, Eyto, E. de, Feldmann, T., Fernández-Aláez, C., Fernández-Aláez, M., Franken, R. J. M., García-Criado, F., Gross, E. M., Gyllström, M., Hansson, L. A., Irvine, K., Järvalt, A., Jensen, J. P., Jeppesen, E., Kairesalo, T., Kornijów, R., Krause, T., Künnap, H., Laas, A., Lill, E. (*et al.*), 2003. The determination of ecological status in shallow lakes - a tested system (ECOFRAME) for implementation of the European Water Framework Directive. *Aquatic Conservation: Marine and Freshwater Ecosystems* **13 (6)**, 507-549.

Mytum, H.C., 1988. On-site and off-site evidence for changes in subsistence economy: Iron Age and Romano-British West Wales. In; Bintliff, J. Davidson, D. and Grant, E. (eds). *Conceptual Issues in Environmental Archaeology*, Edinburgh University Press, 72-81.

Naeem, S. and Li, S., 1997. Biodiversity enhances ecosystem reliability. *Nature* **390**, 507-509.

Naeem, S., 1998. Species redundancy and ecosystem reliability. *Conservation Biology* **12**, 39-45.

Neal, C., Robson, A.J., Neal, M. and Reynolds, B., 2005. Dissolved organic carbon for upland acidic and acid sensitive catchments in mid-Wales. *Journal of Hydrology* **304** 203-220.

Negro, A.I., Hoyos, C.D. and Vega, J.C., 2000. Phytoplankton structure and dynamics in Lake Ssanabria and Valparaiso reservoir (NW Spain). *Hydrobiologia* **424**, 25-37.

Nielsen, H. and Sørensen, I., 1992. Taxonomy and stratigraphy of Late-Glacial Pediastrum taxa from Lysmosen, Denmark – a preliminary study. *Review of Palaeobotany and Palynology* **74**, 55-75.

Nowaczyk, N.R., 2001. Logging of magnetic susceptibility. In; Last, W.M. and Smol, J.P. (eds.). *Tracking Environmental Change Using Lake Sediments: Volume 2.* Kluwer Academic Publishers, London. 155-170.

Nyman, M.T. and Korhola, A.A., 2005. Chironomid-based classification of lakes in western Finnish Lapland. Boreal Environment Research **10 (4),** 239-254.

Odum, E.P., 1953. Fundamentals of Ecology. Saunders, Philadelphia.

Oertli, B., Joye, D.A., Castella, E., Juge, R., Lehmann, A. and Lachavanne, J.B., 2005. PLOCH: a standardized method for sampling and assessing the biodiversity in ponds. *Aquatic Conservation-Marine and Freshwater Ecosystems* **15 (6),** 665-679.

O' Farrel, I.O., 1994. Comparative analysis of the phytoplankton of fifteen lowland fluvial systems of the River Plate Basin (Argentina). *Hydrobiologia*, **289 (1-3)**, 109-117.

Olsson, I., 1968. radiocarbon analysis of lake sediment samples from Björnöya. *Geografiska Annaler* **50A**, 246-247.

O'Sulivan, P.E., Oldfield, F. and Battarbee, R.W., 1973. Preliminary studies of Lough Neigh sediments I. Stratigraphy, chronology and pollen analysis. In; Birks, H.J.B. and West, R.G. (eds.). _Quaternary Plant Ecology_. 267-278.

Owen, E., 1893a. A Contribution to the History of the Premonstratensian Abbey of Talley, Part 1. *Archaeologia Cambrensis* Vol. **X,** 5th Series.

Owen, E., 1893b. A Contribution to the History of the Premonstratensian Abbey of Talley, Part 2. *Archaeologia Cambrensis* Vol. **XI**, 5th Series.

Owen, R.B. and Crossley, R., 1992. Spatial and temporal distribution of diatoms in sediments of Lake Malawi, Central Africa, and ecological implications. *Journal of Palaeolimnology* **7 (1)**, 55-71.

Ozimek, T., Gulati, R.D. and van Donk, E., 1990. Can macrophytes be useful in the biomanipulation of lakes? The Lake Zwemlust example. *Hydrobiologia* **200/201**, 399-407.

Pacey, C.A., 1993. A study of the distribution of potential macrofossils in a lake bed and in the littoral muds in relation to the vegetational composition of the lake and its surroundings. _Unpublished Dissertation._ University of Aberystwyth, Wales.

Palmer, A.J.M. and Abbott, W.H., 1986. Diatoms as indicators of sea-level change. In; van de Plassche, O. (ed.). *Sea-level Research; a manual for the collection and evaluation of data*, 457-487. Geo Books, Norwich.

Parker Pearson, M., 1999. The Early Bronze Age. In; Hunter, J. and Ralston, I. (eds). The Archaeology of Britain; an Introduction from the Upper Palaeolithic to the Industrial Revolution. Routledge, London.

Parker, B.R. and Schindler, D.W., 2006. Cascading trophic interactions in an oligotrophic species-poor Alpine lake. *Ecosystems* **9**, 157-166

Parkinson, J.A., 1976. Animal Bones. *Archaeologia Cambrensis* **125**, 38-39.

Pearson. A.F., 1999. Building *Anderita* Late Roman Coastal Defences and the Construction of the Saxon Shore Fort at Pevensey. *Oxford Journal of Archaeology* **18 (1)**, 95-117.

Peglar, S.M. and Birks, H.J.B., 1993. The mid-Holocene Ulmus fall at Diss Mere, south-east England – disease or human impact? *Vegetation History and Archaeobotany* **2 (2)**, 61-68.

Peglar, S.M., Fritz, S.C. and Birks, H.J.B., 1989. Vegetational land-use history at Diss, Norfolk. *Journal of Ecology* **77**, 203-222.

Penczak, T., Galicka, M., Molinski, M., Kusto, E., Zalewski, M., 1982. The enrichment of a mesotrophic lake by, carbon, phosphorus and nitrogen from the cage aquaculture of rainbow trout, *Salmo gairdneri*. *Journal of Applied Ecology* **19 (2)**, 271-393.

Pennack, R.W., 1989. Cladocera (Water Fleas). In; *Freshwater Invertebrates of the US*, 350-387. John Wiley and Sons Inc., New York.

Pennington W., 1999. Lake Sediments: The Bottom Deposits of The North Basin of Windermere, with Special reference to Diatom Succession. *New Phytologist* **42**, 1-26.

Persson, A., Hansson, L.A., Brönmark, C., Lundberg, P., Petterson, L.B., Greenberg, L., Nilsson, P.A., Nyström, P., Romare, P. and Tranvik, L., 2001. Effects of enrichment on simple aquatic food webs. *American Naturalist* **157**, 654-699.

Persson, L., Diehl, S., Johansson, L., Andersson, G. and Hamrin, S.F., 1992. Trophic interavctions in temperate lake ecosystems: A test of food chain theory. *The American Naturalist* **140 (1)**, 59-84.

Perrow, M.R., Jowitt, A.J.D., Stansfield, J.H. and Phillips, G.L., 1999. The practical importance of the interactions between fish, zooplankton and macrophytes in shallow lake restoration. *Hydrobiologia* **395-396**, 199-210.

Peterson, G., Allen, C.R. and Holling, C.S., 1998. Ecosystem resilience, biodiversity and scale. *Ecosystems* **1**, 6-18.

Philbert, A. and Prairie, Y.T., 2002. Is the introduction of benthic species necessary for open water chemical reconstruction in diatom-based transfer functions? *Canadian Journal of Fisheries and Aquatic Science* **59**, 398-951.

Phillips, G.L., Eminson, D. and Moss, B., 1978. A mechenism to account for macrophyte decline in progressively eutrophicated fresh waters. *Aquatic Botany* **4**, 103-126.

Phillips, G.L. and Moss, B., 1994. Is biomanipulation a useful technique in lake management? *National Rivers Authority R & D,* Note **276.** Peterborough.

Phillips, G. L., Perrow, M. R. and Stansfield, J., 1996. Manipulating the fish-zooplankton interaction in shallow lakes: a tool for restoration. In; Greenstreet, D. P. R. and Tasker, M. L. (eds.). *Aquatic Predators and Their Prey,* Blackwell Science, Oxford, 174-183.

Phillips, G., Bramwell, A, Pitt, J., Stansfield, J. and Perrow, M., 1999. Practical application of 25 years' research into the management of shallow lakes. *Hydrobiologia* **395/396**, 61-76.

Pickering, A. D., 2001. Windermere: Restoring the health of England's largest lake. *Freshwater Biological Association* **11**.

Pimm, S.L. and Lawton, J.H., 1978. On feeding on more than one trophic level. *Nature* **275**, 542-544.

Pitkänen, A., 1999. A 1300-year forest-fire history at a site in eastern Finland based on charcoal and pollen records in laminated lake sediment. *The Holocene* **9 (3)**, 311-320.

Plater, A., Boyle, J., Mayers, C., Turner, S. and Stroud, R., 2006. Climate and human impact on lowland lake sedimentation in Central Coastal California: the record from *ca.* 650 AD to present. *Regional Environmental Change* **6 (1-2)**, 71-85.

Platt, B., Cumming, B. and Laird, K., 2004. A 1500-year record of climatic and environmental change in Elk Lake, Minnesota III: measures of past primary productivity. *Journal of Paleolimnology* **27 (3)**, 321-30.

Prartono, T. and Wolff, G.A., 1998. Organic geochemistry of lacrustine sediments: a record of the changing trophic status of Rostherne Mere, UK. *Organic Geochemistry* **28**, 729-747.

Prather, C. and Hickman, M., 2000. History of a presently acidic lake in northeastern Alberta, Canada as determined through analysis of the diatom record. *Journal of Paleaolimnology* **24**, 183-198.

Prescott, G.W., 1962. *Algae of the Great Western Lakes Area.* W.M.C. Brown Publications, Dubuque, Iowa.

Price, D.L., 1894. Talley Abbey, Carmarthenshire. *Archaeologia Cambrensis* Vol. **X**, 4th Series.

Price, M.D.R., 1981. Palynological studies on the Black Mountains, South wales. Unpublished PhD thesis, University of London.

Price, M.D.R. and Moore, P.D., 1984. Pollen dispersion in the hills of Wales. A pollen shed hypothesis. *Pollen et Spores* **26**, 127-136

Pringle, J. and George, T.N., 1948. *British Regional Geology, South Wales: 2nd Edition.* Her Majesty's Stationary Office, London.

Rasmussen, P. and Anderson, N.J., 2005. Natural and anthropogenic forcing of aquatic macrophyte development in a shallow Danish lake during the last 7000 years. *Journal of Biogeography* **32**, 1993-2005.

Rautio, M., Sorvari, S. and Korhola, A., 2000. Diatom and crustacean zooplankton communities, their seasonal variability and representation in the sediments of subarctic Lake Saanajärvi. *Journal of Limnology* **59** (suppl. 1; Palaeolimnology and ecosystem dynamics at remote Alpine lakes, 81-96.

Rawlence, D.J., 1988. The post-glacial diatom history of Splan lake, New Brunswick. *Earth and Environmental Science* **1** **(1)**, 51-60.

Rawson, D.S., 1955. Morphometry as a dominant factor in the productivity of large lakes. *Proceedings of the International Association of Geography* **12**, 164-175.

Reavie, E.D., Smol, J.P., Carignan, R. and Lorrain, S., 1998. Diatom palaeolimnology of two fluvial lakes in the St. Lawrence River: A reconstruction of environmental change during the last century. *Journal of Phycology* **34** **(3)**, 446-456.

Reed, J.M., Stevenson, A.C. and Juggins, S., 2001. A multi-proxy record of Holocene climate change in southwest Spain: the Laguna de Medina, Cádiz. *The Holocene* **11**, 705-717.

Rees, A.W.G., Hinton, G.C.F., Johnson, F.G. and O'Sullivan, P.E., 1991. The sediment column as a record of trophic status: examples from Bosherston Lakes, SW Wales. *Hydrobiologia*, **214** **(1)**, 171-180.

Regnéll, J., 1992. Preparing pollen concentrates for AMS dating: A methodological study from a hardwater lake in southern Sweden. *Boreas* **21**, 373-377.

Rehakova, Z., 1983. Diatom succession in the post-glacial sediments of the Komorany Lake, Northwest Bohemia, Czechoslovakia. *Hydrobiologia* **103**, 241-245.

Rellstab, C., Maurer, V., Zeh, M., Bürgi, H.R. and Spaak, P., 2007. Temporary collapse of the Daphnia population in turbid and ultra-oligotrophic Lake Brienz. *Aquatic Science* **69**, 257-270

Renberg, I., Korsman, T. and Anderson, N.J., 1993. A temporal perspective of lake acidification in Sweden. *Ambio* **22,** 264-271.

Reynolds, C.S., 1971. The ecology of the pl;anktonic blue-green algae in the north Shropshire meres, England. *Field Studies* **3,** 409-432.

Reynolds, C.S., 1973. The seasonal periodicity of planktonic diatoms in a shalloe eutrophic lake. *Freshwater Biology* **3 (1),** 89-110.

Reynolds, C.S., 1998. What factors influence the species composition of phytoplankton from lakes of different trophic status? *Hydrobiologia* **369/370,** 11-26.

Reynolds, C.S., 2000. Phytoplankton designer – or how to predict compositional responses to changes to trophic-state change. *Hydrobiologia* **424,** 123-132.

Reynolds, C.S., Descy, J-P. and Padisák, J., 1994. Are phytoplankton dynamics in rivers so different from those in shallow lakes? *Hydrobiologia* **289,** 1-7.

Reynolds, C.S., Jaworski, G.H.M., Roscoe, J.V., Hewitt, D.P. and George, D.G., 1998. Response of the phytoplamkton to a deliberate attempt to raise the trophic status of an acid, oligotrophic mountain lake. *Hydrobiologia* **369/370,** 127-131.

Richards, J.D., 1999. In; Hunter, J. and Ralston, I. (eds). The Archaeology of Britain; an Introduction from the Upper Palaeolithic to the Industrial Revolution. Routledge, London.

Richer, M., (ed.), 1974. *Giraldus Cambrensis: Speculum Duorum (or A Mirror of Two Men).* Board of Celtic Studies, University of Wales History and Law Series. No. 27. Cardiff University of Wales Press.

Rippey, B., Anderson, N.J. and Foy, R.H., 1997. Accuracy of diatom-inferred total-phosphorus concentrations and the accelerated eutrophication of a lake due to reduced flushing and increased internal loading. *Canadian Journal of Fisheries and Aquatic Sciences* **54,** 2637-2646.

Ripley, B., 1983. The physical limnology of Lough Augher (Northern Ireland). *Freshwater Biology* **13 (4)**, 353-362.

Roberts, N., 1998. *The Holocene: An Environmental History*. Blackwell publishing, London.

Robinson, D.M. and Platt, C., 1998. *Strata Florida Abbey and Talley Abbey*. CADW, Welsh Historic Monuments, Cardiff.

Romo, S., 1998. Comparative study of phytoplankton in an ologotrophic soft water lake under different pH-phosphate ranges. *Hydrobiologia* **369/370**: 133-127.

Romo, S. and Miracle, M.R., 2004. Population dynamics and ecology of subdominant phytoplankton species in a shallow hypertrophic lake (Albufera of Valencia, Spain).

Rose, N.L, Appleby, P.G., Boyle, J.F., Mckay, A.W. and Flower, R.J., 1998. The spatial and temporal distribution of fossil derived pollutants in the sediment record of Lake Baikal, eastern Siberia. *Journal of Palaeolimnology* **20**, 151-162.

Rose, N.L., Harlock, S. and Appleby, P.G., 1999. Within-basin profile variability and cross correlation of sediment cores using the spheroidal carbonaceous particle record. *Journal of Palaeolimnology* **21**, 85-96.

Rose, N.L., 2001. Fly Ash Particles. In; Last, W.M. and Smol, J.P. (eds.). *Tracking Environmental Change Using Lake Sediments: Volume 2*. Kluwer Academic Publishers, London. 319-349.

Rosentzweig, M.L., 1971. Paradox of enrichment: Destabilisation of exploitation ecosystems in ecological time. *Science* **171**, 385-387.

Round, F.E., 1964. Studies on bottom-living algae in some lakes of the English Lakes district: IV. The seasonal cycles of the Bacillariophyceae. *The Journal of Ecology* **48 (3)**, 529-547.

Round, F.E., 1964. The diatom sequence in lake deposits: some problems of interpretation. *Verh. Internat. Verein. Limnol.* **15,** 1012-1020.

Roy, S. and Chattopadhyay, J., 2007. The stability of ecosystems: A brief overview of the paradox of enrichment. *Journal of Biosciences* **32 (2)**, 421-428.

Rühland., K.M., Smol, J.P. and Pienitz , R., 2003. Ecology and spatial distributions of surface-sediment diatoms from 77 lakes in the subarctic Canadian treeline region. *Canadian Journal of Ecology* **81 (1)**, 57-73.

St. Jaques, J,-M, Douglas, M.S.V., Price, N., Drakulic, N. and Gubala, C. P., 2005. The effect of fish introductions on the diatom and cladocera communities of Lake Opeongo, Ontario, Canada. *Hydrobiologia* **549**, 99-113.

Salmaso, N., 1996. Seasonal variation in the composition and rate of change of the phytoplankton community in a deep subalpine lake (Lake Garda, northern Italy). An application of nonmetric multidimensional scaling and cluster analysis. *Hydrobiologia* **337 (1-3)**, 49-68.

Salomoni, S.E., Rocha, O., Callegaro, V.L. and Lobo, E.A., 2006. Epilithic diatoms as indicators of water quality in the Gravatai river, Rio Grande do Sul, Brazil. *Hydrobiologia* **559**, 233-246.

Sampei, Y. and Matsumoto, E., 2001. C/N ratios in a sediment core from Nakaumi Lagoon, southwest Japan - usefulness as an organic source indicator. *Geochemical Journal* **35,** 189-205.

Sapota, T., Häkanson, L., Aldahan, A. and Possnert, G., 2006. Sediment flux to Lake Baikal (Siberia, Russia): Modelling approach. *Geomorphology* **80 (1-2)**, 105-113.

Sarazin, G., Michard, G., Al Gharib, I. and Bernat, M., 1992. Sedimentation rate and early diagenesis of particulate organic nitrogen and carbon in Ayat Lake (Puy de Dôme, France). *Chemical Geology* **98,** 307-316.

Sarmaja-Korjonen, K., 1999. Headshields of ephippial *Chydorus piger* Sars (Cladocera, Chydoridae) females from northern Finnish Lapland: A long period of gamogenesis? *Hydrobiologia* **390,** 11-18.

Sarmaja-Korjonen, K., 2004. Chydorid ephippia as indicators of past environmental changes – a new method. *Hydrobiologia* **526** , 129-136.

Sarmaja-Korjonen, K., Szeroczynska, K. and Gasiorowski, M., 2003. Subfossil chydorid taxa and assemblages from lake sediments in Poland and Finland with special reference to climate. Studia *Quaternaria* **20**. Warszawa: Polish Academy of Sciences, 25-34.

Savory, H.N., 1961. Mesolithic sites on Craig-y-Llyn, Glamorgan. *B.B.C.S* **19**, 163-165.

Savory, H.N., 1980. The Neolithic in Wales. In; Taylor, J.A. (ed). B.A.R. Brit Series 76, 207-231.

Sayer, C., 2001. Problems with the application of diatom-phosphorus transfer functions: examples from a shallow English lake. *Freshwater Biology* **46**, 743-757.

Sayer, C.D., Roberts, N., Sadler, J., David, C. and Wade, M., 1999. Biodiversity changes in a shallow lake ecosystem: a multi-proxy palaeolimnological analysis. *Journal of Biogeography* **26**, 97-114.

Sayer, C.D., Burgess, A., Kari, K., Peglar, S., Davidson, T.A., Yang, H. and Rose, N., Accepted 2007. Long-term dynamics of submerged macrophytes and algae in a small and shallow, eutrophic lake: implications for the stability of the macrophyte-dominated state. *Freshwater Biology*.

Scheffer, M., 1990. Multiplicity of stable states of freshwater systems. *Hydrobiology* **200/2001**, 475-486.

Scheffer, M., 2004. *Ecology of Shallow Lakes*. Population and Community Biology Series, Kluwer Academic Publishers, London.

Scheffer, M., Carpenter, S., 2003. Catastrophic regime shifts in ecosystems: linking theory to observation. *Trends in Ecology and Evolution* **18 (12),** 648-656.

Scheffer, M., Carpenter, S., Foley, J.A., Folke, C. and Walker, B., 2001. Catastrophic shifts in ecosystems. *Nature* **413**, 591-596.

Scheffer, M., Hosper, S.H., Meijer, M.–L., Moss, B. and Jeppesen, E., 1993. Alternate equilibria in shallow lakes. *Trends in Ecology and Evolution* **8**, 275-279.

Scheffer, M., van den Berg, M., Breukelaar, A., Breukers, C., Coops, H., Doef, R. and Meijer, M.–L., 1994. Vegetated areas with clear water in turbid lakes. *Aquatic Botany* **49**, 193-196.

Schelske, C.L.,Conley, D.J., Stoermer, E.F., Newberry, T.L. and Campbell, C.D., 1986. Biogenic silica and phosphorus accumulation in sediments as indices of eutrophication in the Laurentian Great Lakes. *Hydrobiologia* **143**, 79 – 86.

Schindler, D.W., 1974. Eutrophication and recovery in experimental lakes: Implications for lake management. *Science* **184,** 897-899.

Schindler, D.W., 1990. Experimental perturbations of whole lake as test of hypotheses concerning ecosystem structure and function. *Oikos* **57**, 25-41.

Schindler, D.W., 1998. Whole lake experiments: replication versus realism; the need for ecosystem-scale experiments. Ecosystems 1 (4), 323-334.

Schindler, D.W., Mills, K.H. Malley, D.F. and Findlay, D.L., Shearer, J.A., Turner, M.A., Linsey, G.A. and Cruikshank, D.R., 1985. Long-term ecosystem stress: the effects of years of experimental acidification on a small lake. *Science* **228**, 1395-1401.

Schiemer, F. and Prosser, M., 1976. Distribution and biomass of submerged macrophytes in Neusiedlersee. *Aquatic Botany* **2**, 289-307.

Schlosser, I.J. and Karr, J.R., 2000. Riparian vegetation and channel morphology impact on spatial patterns of water quality in agricultural watersheds. *Environmental Management* **5 (3)**, 233-245.

Schmidt, M.W.I., Rumpel, C. and Kögel-Knabner, I., 1999. Particle size fractionation of soil containing coal and combusted particles. *European Journal of Soil Science* **50 (3)**, 515–522.

Schofield, J., 1999. Landscapes of the Middle Ages. In; Hunter, J. and Ralston, I. (eds). The Archaeology of Britain; an Introduction from the Upper Palaeolithic to the Industrial Revolution. Routledge, London.

Scotland, 2006. At www.scot.gov.uk/Topics/Environment/Water/17670/9395 (cited on 26/09/06).

Selby, K., 2004. Lateglacial and Holocene vegetation change on the Isle of Sky. New data from three coastal locations. *Vegetation History and Archaeobotony* **13 (4)**, 233-247.

Sernander, R., 1908. On the evidences of Post-glacial changes of climate furnished by the peat mosses of northern Europe. *Geol. Fören. Stockholm Fören.* 30. 465-473.
Seymour, W.P., 1985. *The environmental history of the Preseli region of south-west Wales over the past 12000 years.* Unpublishes pHD thesis, University of Wales.

Sharpley, A. N., Hedley, M. J., Sibbesen, E., Hillbricht-Ilowska, A., House, W. A. and Ryszkowski, L, 1995. Phosphorus transfers from terrestrial to aquatic ecosystems. In; Tiessen, H. (Ed.). *Phosphorus in the Global Environment.* J. Wiley, Chichester.

Shen, C., Lui, K., Morrill, C., Overpeck, J.T., Peng, J. and Tang. L., 2008. Ecotone shift and major droughts during the mid-late Holocene in the Central Tibetan Plateau. *Ecology* **89 (4)**, 1079-1088.

Shovonlal, R. and Chattopadhyay, J., 2007. The stability of ecosystems: a brief overview of the paradox of enrichment. *Journal of Bioscience* **32 (2)**, 421-428.

Schrage, L.J. and Downing, J.A., 2004. Pathways of increased water clarity after fish removal from Ventura Marsh; a shallow, eutrophic wetland. *Hydrobiologia* **511 (1-3)**. 215-231.

Shepherd, R., 1993. *Ancient Mining.* Institution of Mining and Metalurgy, Elsever Applied Science, Chapman and Hall.

Simmonds, I.G. and Innes, J.B., 1996. Prehistoric charcoal in peat profiles at North Gill, North Yorkshire Moors, England. *Journal of Archaeological Science* **23**, 193-197.

Simpson, G.L., 2000. Palaeolimnological analogues in defining target assemblages for the recovery of acidified surface waters: a desk study. *ECRC Research Report* **74**.

Simpson, G.L., 2001. Biological targets for recovery from lake acidification: Developing the analogue matching procedure. In Curtis, C. and Simpson, G. L. (eds). In; Summary of research under DETR contract "*Acidification of fresh waters: the role of nitrogen and the prospects for recovery: Work Package 2: Recovery*". *ECRC Research Report* **79**.

Sims, P. A., 1996. *An Atlas of British Diatoms.* Biopress Ltd., Bristol.

Slusarczyk, M., 2001. Food threshold for diapause in Daphnia under threat of fish predation. *Ecology* **82**, 1089-1096.

Smetacek, V.S., 1985. Role of sinking in diatom life-history cycles: ecological, evolutionary and geological significance. *Marine Biology* **84 (3)**, 239-251.

Smith, A.G. and Cloutman, E.W., 1988. Reconstruction of Holocene vegetation history in three dimensions at Waun-Fignen-Felen, an upland site in south Wales. *Philosophical Transactions of the Royal Society of London B* **322**, 159-219.

Smith, G.M., 1950. *The Freshwater Algae of the United States. 2nd Edition.* McGraw-Hill Book Company, New York.

Smith, D., Osborne, P. and Barret, J., 2000. Chapter 14: Beetles as Indicators of Past Environments and Environmental History at Goldcliff. In; Bell., M., Caseldine, A. and Neumann H. (eds.). *Prehistoric Intertidal Archaeology in the Welsh Severn Estury*. CBA Research Report **120**, 245-259.

Smith, R.W., 1984. The ecology of Neolithic farming systems exemplified by the Avebury region of Wiltshire. *Proceedings of the Prehistoric Society* **50**, 99-120.

Smith, R.T. and Taylor, J.A., 1969. The Post-glacial development of vegetation and soils in northern Cardiganshire. *The Institution of British Archaeology* **48**, 75-95.

Smol, J.P., 1990. Freshwater Algae. In; Warner, B.G., (ed.). *Methods in Quaternary Ecology*. *Geoscience Canada*, Series **5**.

Smol, J.P., 1992. Paleolimnology; An important tool for effective ecosystem management. *Journal of Aquatic Ecosystem Health* **1**, 49-58.

Smol, J.P., 2002. *Pollution of Lakes and Rivers: A Palaeoenvironmental Perspective.* Hodder and Arnold, London.

Snowball, I.F. and Thompson, 1988. The occurrence of greigite in sediments from Loch Lomond. *Journal of Quaternary Science* **3**, 121-125.

Smol, J.P., 2002. *Pollution of Lakes and Rivers: A Palaeoenvironmental Perspective.* Arnold, London.

Søndergaard, M., Kristensen, P. and Jeppersen, E., 1992. Phosphorus release from resuspended sediment in the shallow wind-exposed lake Arresø. *Hydrobiologia* **228** **(1)**, 91-99.

Søndergaard, M., Jensen, J.P. and Jeppesen, E., 2003. Role of sediment and internal loading of phosphorus in shallow lakes. *Hydrobiologia* **506-509,** 135-145.

Spurgeon, C.J., 1987. Mottes and castle ringworks in Wales. In; Kenyon, J.R. and Aventg, R. (eds). *Castles in Wales and the Marches*, Cardiff University Press, 23-50.

Stager, J.C., Reinthal, P.N. and Livingstone, D.A., 1986. A 25,000-year history for Lake Victoria, East Africa, and some comments on its significance for the evolution of cichlid fishes. *Freshwater Biology* **16 (1)**, 15-19.

Stenger-Kovács, C, Buczkó, K., Hajnal, É. And Padisák, J., 2007. Epiphytic, littoral diatoms as bioindicatirs of shallow lake trophic status: Trophic Diatom Index for Lakes (TDIL) developed in Hungary. *Hydrobiologia* **589**, 141-154.

Stevenson, J., Dodson, J.R. and Prosser, I.P. 2001. A late Quaternary record of environmental change and human impact from New Caledonia. *Palaeogeography and Palaeoclimate* **168 (1-2)**, 97-123.

Stoermer, E.F. and Smol J.P., 1999. *The Diatoms: Applications for the Environmental and Earth Sciences.* Cambridge University Press, Cambridge.

Sugita, S., 1993. A model of pollen source area for an entire lake surface. *Quaternary Research* **39 (2)**. 239-244.

Sugita, S., Gaillard, M.J. and Broström, A., 1999. Landscape openness and pollen records; a simulation approach. *Holocene* **9 (4)**, 409-421.

Summerfield, M.A., 1999. *Global Geomorphology; An introduction to the study of landforms.* Longman, Essex, England.

Sutherland, R.A., 1998. Loss-on-ignition estimates of organic matter and relationships to organic carbon in fluvial bed sediments. *Hydrobiologia* **389**, 153-167.

Tansley, A.G., 1939. *The British Islands and Their Vegetation.* Cambridge University Press, Cambridge.

Taylor, J.A., 1973. Chronometers and chronicles: a study of palaeoenvironments in west central Wales. *Progress in Geography* **5**, 247-334.

Taylor, B.R., Parkinson, D. and Parsons, W.F.J., 1989. Nitrogen and lignin content as predictors of litter decay rates: A microcosm test. *Ecology* **70 (1)**, 97-104.

Taylor, D., Dalton, C., Leira, M., Jordan., P., Chen., G.,l León-Vintro, L., Irvine, K. Bennion, H. and Nolan, T., 2006. Recent histories of six productive lakes in the Irish Ecoregion based on multiproxy palaeolimnological evidence. *Hydrobiologia* **571**, 237-259.

Telesh, I.V., Alimov, A.F., Golubkov, S.M., Nikulina, V.N. and Panov, V.E., 1999. Response of aquatic communities to anthropogenic stress: a comparitive study of Neva Bay and the Eastern Gulf of Finland. *Hydrobiologia* **393**, 95-105.

Telford, R.J., Heegaard, E. and Birks, H.J.B., 2004. All age-depth models are wrong: but how badly? *Quaternary Science Reviews* **23**, 1-5.

Telford, R.J. and Lamb, H.F., 1999. Groundwater-mediated response to Holocene climatic change recorded by the diatom stratigraphy of an Ethiopian crater lake. *Quaternary Research* **52**, 63-75.

ter Braak, C.J.F. and van Dam, H., 1989. Inferring pH from diatoms: a comparison of old and new calibration methods. *Hydrobiologia* **178**, 209-223.

Thain, M. and Hickman, M., 1999. *Dictionary of Biology.* Penguin Books, London.

Thomas, K.W., 1965. The stratigraphy and pollen analysis of a raised peat bog at Llanllwch, near Carmarthen. *New Phytologist* **64**, 101-117.

Thomas, C., 1975. Peasant Agriculture in medieval Gwynedd; an interpretation of documentary evidence. *Folk Life* **13**, 24-37.

Thomas, J., 1999. *Understanding the Neolithic* (London: Routledge)

Thompson, R., Battarbee, R.W., O'Sullivamn, P.E. and Oldfield, F., 1975. Magnetic Susceptibility of Lake Sediments. *Limnology and Oceanography* **20**, 73-80.

Thoms, M.C., Ogden, R.W. and Reid, M.A., 1999. Establishing the condition of lowland floodplain rivers: A palaeoecological approach. *Freshwater Biology* **41**, 407-423.

Tilman, D., 1996. Biodiversity: population versus ecosystem stability. *Ecology* **77**, 350-363.

Timms, R. M. and Moss, B., 1984. Prevention of growth of potentially dense phytoplankton populations by zooplankton grazing in the presence of zooplanktiverous fish in a shallow wetland ecosystem. *Limnology and Oceanography* **29**, 472-486.

Tolonen, M., 1978. Palaeoecology of annually laminated sediments in Lake Ahvenainen, S. Finland. I. Pollen and charcoal analysis and their relation to human impact. *Ann. Bot. Fenn.* **15**, 177-208.

Tolonen, M., 1986. Charred Particle Analysis. In; In: Berglund, B. (ed). *Handbook of Holocene Paleoecology and Palaeohydrology,* 485-495. John Wiley and sons, Ltd., Chichester.

Tomlinson, P., 1987. Pollen. *Journal of Medieval Archaeology* **31**, 41.

Troels-Smith, 1955. Characterization of unconsolidated sediments. *Danmarks Geologiske Undersøgelse,* series IV, 3, 38-73. In: Jones, A.P., Tucker, M.E. and Hart, J.K. (eds.), 1999. The Description and Analysis of Quaternary Stratagraphic Field

Sections. *Quaternary Research Association, Technical Guide No. 7*, Quaternary Research Association, London.

Tsudaka, M. and Deevey, E.S., 1967. Pollen analysis from four lakes in the Southern maya area of Guatemala and El Salvador. In; Cushing, E.J. and Wright, H.E. (eds.). *Quaternary Palaeoecology.* Yale University Press, New Haven.

Tsukada, M., Sugita, S. and Tsukada, Y., 1986. Oldest primative agriculture and vegetational environment in Japan. *Nature* **322**, 632-634.

Turner, J., 1964. The anthropogenic factor in vegetational history. I. Tregaron and Whixall Mosses. *New Phytologist* **63 (1)**, 73-90.

Turner, J., 1965. A contribution to the history of forest clearance. *Proceedings of the Royal Society B* **161**, 343-354.

Turner, R.S., Cook, R.B., Van Miegroet, H., Johnson, D.W., Elwood, J.W., Bricker, O.P., Lindberg, S.E. and Hornberger, G.M., 1989.Watershed and lake processes affecting chronic surface water acid-base chemistry. *State-of-the-Science Technical Report* **10**. National Acid Precipitation Assessment Program, Washington DC.

UK Biodiversity, 2001. At www.ukbap.org.uk. (cited on 22/04/06)

UKTAG, 2008. At www.wfduk.org. (cited on 06/01/08)

Urrutia, R., Sabbe, K., Cruces, F., Pozo, K. and Becerra, J., 2000. Paleolimnological studies of Laguna Chica of San Pedro (VIII Region): Diatoms, hydrocarbons and fatty acid records. *Review of Chilian Natural History* **73**, 717-728.

van Dam, H., Mertens, A. and Sinkeldam, J., 1994. A coded chexklist and ecological indicator values of freshwater diatoms of the Netherlands. Netherlands Journal *Aquatic Ecology* **28 (1)**, 117-133.

van den Berg, M.S., Scheffer, M., van Ness, E. and Coops, H., 1999. Dynamics and stability of *Chara* sp. and *Potamogeton pectinatus* in a shallow lake changing in eutrophic level. *Hydrobiologia* **408-409**, 335-342.

van Donk, E., and Kilham, S.S., 1990. Temperature effects on silicon and phosphorus-limited growth and competitive interactions among three diatoms. *Journal of Phycology* **26**, 40–50.

van Eerden, M.R. and Voslamber, B., 1995. Mass fishing by Cormorants, Phalacrocorax carbo sinensis, at Lake Ijsselmeer, the Netherlands: a recent and successful adaptation to a turbid environment. *Ardea* **83**, 199-212.

van Geel, B., Klink, A.G., Pals, J.P. and Wigers, J., 1986. An Upper Eemian lake deposit from Twente, eastern Netherlands. *Review of Palaeobotony and Palynology* **47**, 31-61.

van Ness, E.H., Scheffer, M., van den Berg, M. and Coops, M., 2002. Aquatic macrophytes: restore, eradicate or is there a compromise? *Aquatic Botony* **72**, 387-403.

van Zeist, E. and Bottema, S., 1977. Palynological investigations in Western Iran. *Palaeohistoria* **19**, 19-95.

von Post, L., 1916. Forest tree pollen in south Swedish peat bog deposits. *Pollen Spores* **9**, 375-401.

Vollenweider, R.A., 1968. *Scientific Fundamentals of Lake and Stream Eutrophication, with Particular Reference to Phosphorus and Nitrogen as Eutrophication Factors*. Technical Report DAS/DSI/68.27. OECD, Paris.

Vollenweider, K. and Kerekes J., 1982. *Eutrophication of waters, monitoring assessment and control.* OECD, Paris

Waddington, J.C.B., 1969. A stratigraphic record of the pollen influx to a lake in the Big Woods of Minnesota. *Geological Society of America, Special Paper* **123**, 236-282.

Walker, I.R., Smol, J.P., Engstrom, D.R. and Birks, H.J.B., 1991. An assessment of Chironomidae as quantitative indicators of past climatic change. *Canadian Journal of Fisheries and Aquatic Sciences*, **48**, 975-987.

Walker, M.J.C., 1981. The late-glacial and early Flandrian deposits at Traeth Mawr, Brecon Beacons, South Wales. *New Phytologist* **90**, 177-194.

Walker, M.J.C., 1985. Pollen Analysis. *Carmarthenshire Antiquity* **21**, 108-109

Walker, M.J.C., 1993. Holocene (Flandrian) vegetation change and human activity in the Carneddau area of upland mid-Wales. In; *Climatic Change and Human Impact on the Landscape* (edited by F. M. Chambers), Chapman and Hall, 163-183.

Walker, M.J.C., Griffiths, H.I., Ringwood, V. and Evans. J.G., 1993. An early-Holocene pollen, mollusc and ostracod sequence from lake marl at Llangorse Lake, South Wales, UK. *The Holocene* **3 (2)**, 138-149.

Walker, M.J.C. and Harkness, N.D.D., 1990. Radiocarbon Dating the Devensian Late-glacial in Britain: New evidence from Llanilid, south Wales. *Journal of Quaternary Science* **5 (2)**, 135-144.

Walker, M.F. and Taylor, J.A., 1976. Post-neolithic vegetation changes in the western Rhinogau, Gwynedd, North West Wales. *Transactions of the Institute of British Geography* **1**, 323-345.

Warner, B.G., 1990. Plant Macrofossils. In; Warner, B.G., (ed.). *Methods in Quaternary Ecology*. Geoscience Canada, **Series 5**.

Watkins, R. 1990. The postglacial history of lowland Gwynedd - Llyn Cororion. In; Addison, K., Edge, M.J. And Watkins, R. (eds). *North Wales Field Guide*. Quaternary Research Association, 131-136.

Weale, M.E., Weiss, D.A., Jager, R.F., Bradman, N. and Thomas, M.G., 2002. Y Chromosome Evidence for Anglo-Saxon Mass Migation. *Molecular Biology and Evolution* **19**, 1008-1021.

Webley, D.P., 1974. Pollen analysis from the site of Rhos-y- Clegryn. In; Lewis, J.M. *Archeaologia Cambrensis* **123**, 34-35.

Weismann, A., 1877. Beitrage zur Naturgeschichte der Daphnoiden. II. Die Eibildung bei den Daphnoiden. Cited in; Frey, D.G., 1982. Contrasting strategies of gamogenesis in northern and southern populations of Cladocera. *Ecology* **63**, 223-241.

Welch, E.B. and Cooke, G.D., 2005. Internal phosphorus loading in shallow lakes: Importance and control. *Lake and Reservoir Management* **21 (2)**, 209-217.

Wetzel, R.G., 2001. *Limnology: Third Edition.* Saunders College Publishing, London.

White, P.S. and Walker, J.L., 1997. Approximating natures variation: selecting and using reference information in restoration ecology. *Restoration Ecology* **5 (4)**, 338-349.

Whitehead, D.R., Charles, D.F., Jackson, S.T., Reed, S.E. and Sheenan, M.C., 1986. Late Glacial and Holocene acidity changes in the Adirondack (NY) lakes. In; Smol, J.P., Battarbee, R.W., Davis, R.B. and Meriläinen (eds.). *Diatoms and Lake Acidity. Dr. W. Junk Press, Dordrecht.*

Whiteside, M.C., 1970. Danish chydorid Cladocera: modern ecology and core studies. *Ecological Monographs* **40**, 79-118.

Whiteside, M.C., 1983. The mythical concept of eutrophication. *Hydrobiologia*, **103** 107-111.

Whitmore, T. J., 2004. Sedimentary diatom concentrations and accumulation rates as predictors of lake trophic status. *Hydrobiologia* **214 (1)**, 163-169.

Whittaker, R.H., 1975. *Communities and Ecosystems: 2^{nd} Edition.* Colier Macmillan Publishers, London.

Whittle, A., 1999. The Neolithic Period, *c.* 4000-2500/2200 BC. Changing the World. In; Hunter, J. and Ralston, I. (eds). *The Archaeology of Britain: An Introduction from the Upper Palaeolithic to the Industrial Revolution.* Routledge, London.

Wik, M., Renberg, I. and Darley, J., 1986. Sedimentary records of carbonaceous particles from fossil fuel combustion. *Hydrobiologia* **143 (1)**, 387-394.

Wilcox, G.H., 1974. A History of deforeststion as indicated by charcoal analysis at four sites in Eastern Anatolia. *Anatolian Studies* **24**, 117-133.

Williams, H., 2003. Archaeologies of Remembrance; Death and Memory in Past Societies. Springer, New York.

Wilson, D.G., 1981. Flora and fauna of the Well . 3; The Plant Remains. In; Jarret, M.E. and Wrathmell, S. (ed). *An Iron Age and Roman Farmstead in South Glamorgan.* University of wales press.

Witkowski, A., 1994. Recent and fossil diatom flora of the Gulf of Gdansk, Southern Baltic Sea. Origin, composition and changes of diatom assemblages during the Holocene. *Bibliotheca Diatomol* **28**, 1-230.

Wohlfarth, B., Possnert, G., Skog, G. and Holmquist, B., 1998. Pitfalls in the radiocarbon dating of terrestrial macrofossils. *Journal of Quaternary Science* **13**, 137-145.

Wolf, B.B., Edwards, T.W.D., Elgood, R.J. and Beuning, K.R.M., 2001. Carbon and oxygen isotope analysis of lake sediment cellulose: methods and applications. In:

Last, W.M. and Smol, J.P. (eds.). *Tracking Environmental Change Using Lake Sediments. Vol. 2: Physical and Geochemical Methods,* 373-400. Kluwer Academic Publishers Dordrecht, Netherlands.

Wolin, J.A., 1996. Late Holocene lake level and lake development signals in Lower Herring Lake, Michigan. *Journal of Palaeolimnology* **15**, 19-45.

Wolin, J.A. and Duthie, H.C., 1999. Diatoms as indicators of water level changes in freshwater lakes. In; Stoermer, E.F. and Smol, J.P. (eds.). *The Diatoms: Application for the Environmental and Earth Sciences*, 183-202. University Press, Cambridge.

Woods, J.A., 1989. Peatland acidity budgets and the effect of acid deposition. Ecological applications research division, sustainable development, Environment Canada, *Discussion Paper* **5**.

Wright, H.E., 1980. Cores of soft lake sediments. *Boreas* **9**,107-119

Wright, J.F., Sutcliffe, D.W. and Furse, M.T., 2000. *Assessing the Biological Quality of Fresh Waters.* Freshwater Biological Association, Ambleside.

Wymer J., 1981. The Palaeolithic. In; Simmonds, I.G. and Tooley, M.J. (eds.). *The Environment in British Pre-history.* Duckworth, London, 49-81.

Wymer J., 1999. *The Lower Palaeolithic Occupation of Britain: Volume 1.* Wessex Archaeology and English Heritage, Trust for Wessex Archaeology Ltd., Salisbury, p. 181.

Yang, X.D., Dong, X.H., Gao, G., Pan, H.X. and Wu, J.L., 2005. Relationship between surface sediment diatoms and summer water quality in shallow lakes of the middle and lower reaches of the Yangtze River. *Journal of Intergrative Plant Biology* **47 (2)**, 153-164.

Yangdong, P. and Brugham, R., 1997. Human disturbance and trophic status changes in Crystal Lake, McHenry County, Illinois, USA. *Journal of Palaeolimnology* **17**, 369-376.

Yodzis, P. and Innes, S., 1992. Body-size and consumer-resource dynamics. *American Nature* **139,** 1151–1175

Zhao, Y., Sayer, C.D., Birks, H.H., Hughes, M. and Peglar, S.M., 2006. Spatial representation of aquatic vegetation by macrofossils and pollen in a small, shallow lake. *Journal of Paleolimnology* **35**, 335-350.

Zbinden, H., Andree, M., Oeshger, H., Ammann, B., Lotter, A., Bonani, G. and Wolfi, W., 1989. Atmospheris radiocarbon at the end of the last glacial: An estimate based on AMS radiocarbon dates on terrestrial macrofossils from lake sediments. *Radiocarbon* **31**, 795-804.

Zohary, T., 2004. Changes to the phytoplankton assemblage of Lake Kinneret after decades of a predictable, repetitive pattern. *Freshwater Biology* **49** (10), 1355-1371.

Zong, Y. and Tooley, M.J., 1096. Holocene sea-level changes and crustal movements in Morcambe Bay, northwest England. *Journal of Quaternary Science* **11 (1),** 43-58.