Report

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The limnology of Esthwaite Water: historical change and its causes, current state and prospects for the future

A report to Natural England

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Executive Summary

1. Esthwaite Water is one of the most nutrient-enriched lakes in the English Lake District and one of the best-studied lakes in the world. The purpose of this report is to produce a comprehensive overview describing the current ecological status of Esthwaite Water, how it has changed in response to external and internal pressures and to forecast how it may function in the future.

2. The catchment of Esthwaite Water lies on Silurian slates, grits and Bannisdale slates overlain by glacial till. It drains into the South Basin of Windermere. It is a SSSI, a Ramsar site and the fen at the north end is a National Nature Reserve. It is 0.96 km^2 in area, has a mean depth of 6.9 m and an average retention time of 91 days.

3. Palaeolimnological records extending back to the end of the last glaciation, when the lake was formed, show how the catchment vegetation has altered in response to climate change and man's activities. There is clear evidence for nutrient enrichment that began around 1850 and accelerated after 1970.

4. The main land cover in the catchment today is pasture (55%), coniferous forest (20%), broad leaved forest (13%) and natural grassland (11%).

5. The average outflow from the lake is 0.85 $m^3\ s^{-1}$ leading to an average discharge of 26.9 $Mm^3\ y^{-1}$, of which about 55% derives from Black Beck.

6. The load of phosphorus, the main limiting nutrient, has varied over time in magnitude and source. The main sources of phosphorus to Esthwaite Water today are the catchment (fertilizers and septic tanks), rain, wastewater treatment works, the fish farm and internal load from the lake sediment. Before the WwTW and fish farm were in operation, the catchment was estimated to deliver 96% of the total load of 1378 kg TP y^{-1}. The total load had increased to 1541 kg TP y^{-1} in 1985-6 and the WwTW was the dominant source (37%). In 1992-3, the total load of 1677 kg TP y^{-1} was mainly contributed by the fish farm (48%), the catchment (29%) and the WwTW operations had declined to 20% as a result of tertiary P-removal at the Hawkshead works. In 2010 following the closure of the fish farm and redirection of the discharge from near Sawrey,

the load was estimated to have fallen to 747 kg TP y^{-1} with the main contribution from the catchment (40%) and the WwTW (34%). Internal load is a small component of total load on an annual basis, but can be important in summer when inflow from the land is low.

7. The lake sediment is over 5 m deep and the present day sedimentation rate, much greater than in previous years, is about 0.06 g cm^{-2} y^{-1}, equivalent to about 0.6 mm y^{-1} although there is a high degree of spatial variability in sedimentation rate in different regions caused by variation in physical mixing processes.

8. The physical conditions on the lake are known from the CEH automatic water quality monitoring station on the lake and the longer-term weekly or fortnightly measurements that extend back to the 1940s. Many physical conditions are directly influenced by climate change. Surface water temperature has increased by about 0.27 ^{o}C per decade since 1957. The stratification strength (using an index of top minus bottom temperature) has increase by about 0.25 ^{o}C per decade over the same period. Stratification has also been starting earlier by about 7 days per decade and breaking down later by about 4 days per decade.

9. Average wind-induced water currents are about 5 cm s^{-1} at the lake surface. Wave-mixed depths are about 0.5 m under average wind conditions and 2 m with strong winds, so a relatively small amount of the lake bed is directly affected by wind-induced mixing.

10. Long-term records of water transparency, measured with a Secchi disc, show a strong seasonal pattern with clearest water in May, after the spring diatom bloom, and the most turbid water in the summer. There have been significant reductions in Secchi depth in most months between May and October. There has been a marked decline in annual average Secchi depth between 1972 and the present day with a downward step-change around 1987. Based on measurements with light-meters, the current average depth at which light is attenuated to 1% (rough indication of the photic zone) is 5 m.

11. The composition of major anions and cations over the last 50 years is presented. The most noticeable change is the increase in alkalinity (acid neutralising capacity) linked to an increase in concentration of sulphate as a result of declining sulphate-deposition.

Esthwaite Water has a current average alkalinity of 431 mequiv m^{-3}, placing it within the 'medium alkalinity' category of the Water Framework Directive.

12. Uptake of inorganic carbon in photosynthesis and production of carbon dioxide by respiratory process in the lake and catchment lead to rapidly-changing pH, especially during summer stratification. Minimum surface pH values are slightly below pH 7 and maximum values are over pH 10. Although at high pH the surface water is strongly depleted in carbon dioxide and taking up this gas from the atmosphere, over a year the lake is a source of CO_2; in large part because it receives CO_2-rich water from inflowing streams.

13. The long-term record of soluble reactive phosphorus (SRP) extending back to 1945 demonstrates the dramatic increase in winter concentrations. Summer concentrations are low then and now because of high biological demand for this limiting resource. The total phosphorus (TP) concentration has also increased but show indications of a reduction in the last decade.

14. Nitrate-nitrogen has also increased, but less dramatically than for phosphorus. The late summer depletion may impose a transient nitrogen-depletion on the phytoplankton, although this may be overcome by growth of nitrogen-fixing cyanobacteria.

15. Dissolved organic carbon is not measured routinely. Measurements indicate and annual average of about 3 g m^{-3}; higher than many of the lakes in the English Lake District, but low on a national scale.

16. There are strong depth-variation in water chemistry during summer stratification caused by a separation of photosynthesis in the upper well-lit epilimnion and respiration in the 'dark' lower hypolimnion. Oxygen is close to zero in much of the hypolimnion in summer and this has been the case for the last 45 years. Nutrient concentrations are higher at depth because of regeneration by decomposition, and release from the sediment surface because of redox-changes linked to anoxia.

17. Phytoplankton are the major primary producers in Esthwaite Water today. Chlorophyll *a* is the main photosynthetic pigment and its concentration is a convenient measure of phytoplankton density. Measurements extending back to 1964 show increased

concentration from May to July and in October. Over 500 taxa of phytoplankton have been recorded in Esthwaite Water over the last 66 years. Of these 190 have only been recorded once, while the diatom *Asterionella formosa* has been recorded each year, and four more species have been recorded in 64 years or more. In general, the number of phytoplankton taxa recorded each year has increased. Average seasonal patterns show light- and temperature-limited *A. formosa* and *Aulacoseira subarctica* in spring, and colonial green algae such as *Coenochloris fottii* in midsummer and dinoflagellates and cyanobacteria in late summer at times of nutrient depletion and strong stratifciation.

18. Early studies on submerged macrophytes were carried out by W.H. Pearsall between 1914 and 1916 when 22 species were recorded. Two rare species, *Hydrilla verticillata* and *Najas flexilis*, appear to be lost from the lake and the species number has declined to about eleven. Depth limits also appear to have declined with worsening light climate from about 3.6 m to 2.3 m in 2011.

19. Protozoa are single-celled organisms that consume phytoplankton, bacteria and detritus and live in the water column and sediment. Esthwaite Water supports a diverse community of protozoa: over 120 species have been identified. They have large seasonal patterns of change, typically with peaks in spring-early summer.

20. Esthwaite Water supports a diverse community of zooplankton which live in the open water or in association with sediments or vegetation. They occupy an important intermediate position in the food web as they consume phytoplankton and are consumed by fish. Twenty-four species of rotifers, 25 species of cladocerans, 11 species of cyclopoid copepods, one species of calanoid copepod and one species of harpacticoid copepod have been identified in Esthwaite Water. The numerically dominant cladocerans typically produce a large cohort in late spring and a much smaller cohort in late summer. The dominant species are *Daphnia hyalina/galeata*, *Bosmina longirostris* and *Ceriodaphnia quadrangula*.

21. The different littoral and profundal habitats in Esthwaite Water support different types of benthic invertebrates. These include 81 insects, 3 crustaceans, 16 molluscs, 11 leeches,

6 flatworms, 10 oligochaete worms and some less-well identified species from other groups.

22. The vertebrate fauna of Esthwaite Water has not been particularly well-studied. Brown trout (*Salmo trutta*), perch (*Perca fluviatilis*) and pike (*Esox lucius*) are the main species with Altantic salmon (*Salar salar*) passing though on migration and the cyprinids roach (Rutilis rutilis) and rudd (*Scardinius erythrophthalmus*) and their hybrids also present. The fish farm stocks the lake with rainbow trout (*Oncorhynchus mykiss*) but this will be phased out in the next few years.

23. No amphibians or reptiles are recorded for Esthwaite Water and its environs, although they are likely to be present. The lake's waterfowl are of national and international importance and include a large number of species. Recently an osprey (*Pandion haliaetus*) has been observed hunting at Esthwaite Water. The otter (*Lutra lutra*), American mink (*Mustela vison*) and coypu (*Myocaster copypus*) are among the 'aquatic' mammals recorded.

24. Esthwaite Water, like many lakes, is sensitive to climate change. Regional weather patterns such as the North Atlantic Oscillation, the position of the Gulf Stream and Rossby breaking waves all affect the ecological functioning of the lake. Phenological changes have been linked to warming water temperature, but also to nutrient enrichment. Forecasts of future conditions using models suggest a greater preponderance of the potentially toxic cyanobacteria *Anabaena* and *Aphanizomenon* in response to warmer summers with lower rainfall and hence lower discharge.

25. Esthwaite Water has clearly responded to Man's activity and past management actions within the catchment. More recently, the fish farm ceased in autumn 2009 and the Hawkshead WwTW upgrades should be completed in 2012. Although there are signs of recent improvement in water quality, these actions, as a minimum, will be needed to shift the lake from its current 'Moderate' to 'Good' Ecological Status under the Water Framework Directive.

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1. Objective of the review

Esthwaite Water is one of the most nutrient enriched lakes in the English Lake District, but recently two changes have been implemented to reduce nutrient loading to the lake: the removal of the fish hatchery on the lake and upgrading of the wastewater handling and treatment facilities at the Hawkshead Wastewater Treatment Work (WwTW). At the same time, it is experiencing changes linked to variable weather patterns and climate change that have the potential to have widespread effects on the way the lake functions. It is also one of the best-studied lakes in the world with detailed scientific studies that extend back for around 100 years and is part of the Cumbrian lakes long-term monitoring programme undertaken by the Centre for Ecology and Hydrology. It has been the subject of numerous papers and report. The earlier ones have been summarised by Talling & Heaney (1983) so this review seeks to update this with the new data, publications and responses to recent pressures in the subsequent nearly 30 years.

The aim of this report is not to cite every paper or report produced on Esthwaite Water but to produce a comprehensive overview describing the current ecological status of Esthwaite Water, how it has changed in response to external and internal pressures and to forecast how it may function in the future.

2. Background to Esthwaite Water

2.1 Basic features

Esthwaite Water (54° 22' N, 2° 56' W) is situated in a side-valley that drains into the South Basin of Windermere in the English Lake District, Cumbria UK. It is about 2.8 km long and 0.547 m at its widest (Pearsall 1917). It was formed at the end of the last glaciation, around 12,000 year ago, and is held-up by morainic material. Priest Pot to the north (Finlay & Maberly 2000) and Out Dubs Tarn to the south were once part of the lake. Esthwaite Water is surrounded by an agricultural landscape of improved grassland with small copses and some deciduous forests (Fig. 2.1). Studies by Pearsall (1921a), Jones (1972) and Gorham *et al.* (1974) ranked Esthwaite Water as the most productive of the 20 major lakes in the English Lake District. It has extremely rare long-term data extending back consistently to 1945 but has relatively little information on fish populations. It lies within the Lake District National Park, is a Site of Special Scientific Interest (SSSI) for Natural England and an international Ramsar Convention site. The fen at the north end of the lake is a National Nature Reserve (Piggott & Wilson 1978). Talling (1999) provided a general overview of the lake.

Figure 2.1 Aerial view of Esthwaite Water and its immediate surroundings.

Esthwaite Water is a relatively small lake compared to many in the English Lake District (Table 2.1): it has an area of 0.96 km^2 (10 have a larger area) and a volume of 6.7 Mm^3 (11 have a larger volume). Its catchment is 17 km^2 and is more low-lying (average 148 m) than most of the other major English Lake District catchments. The average discharge recorded between 1976 and 2009 of 26.9 $Mm^3\ y^{-1}$ (equivalent to 0.85 $m^3\ s^{-1}$) gives an average retention time of 91 days.

Table 2.1 *Key characteristics of Esthwaite Water and its catchment.*

Catchment	**Value**	**Lake**	**Value**
Catchment area (km^2)	17	Lake area (km^2)	0.96**
Maximum catchment altitude (m)	306	Altitude (m)	65
Average catchment altitude (m)	148	Maximum depth (m)	16.0**
Average slope ($m\ km^{-1}$)	118	Mean depth (m)	6.9**
Average rainfall (1961-1990; $m\ y^{-1}$)	1.911	Volume (Mm^3)	6.7
Average discharge* (1976-2009; $Mm^3\ y^{-1}$)	26.9	Average retention time (d)	91

** See Section 4 on hydrology. ** New bathymetric survey Mackay et al. (2011).*

The lake is divided into three main basins. The northern basin is the largest in area and also the deepest with a maximum depth of 16 m (Fig. 2.2). This deep point is where the long-term monitoring samples are collected. A sill at about 5 m separates this from a smaller central basin with a maximum depth of around 12 m and another sill at about 6 m separates this central basin from a small southern basin that has a maximum depth of around 10 m. The main inflows enter the lake to the north and the ouflow is to the south from the southern basin. The mean depth of 6.9 m places Esthwaite Water in the 'shallow' depth-category within the Water Framework Directive (2000/60/EC).

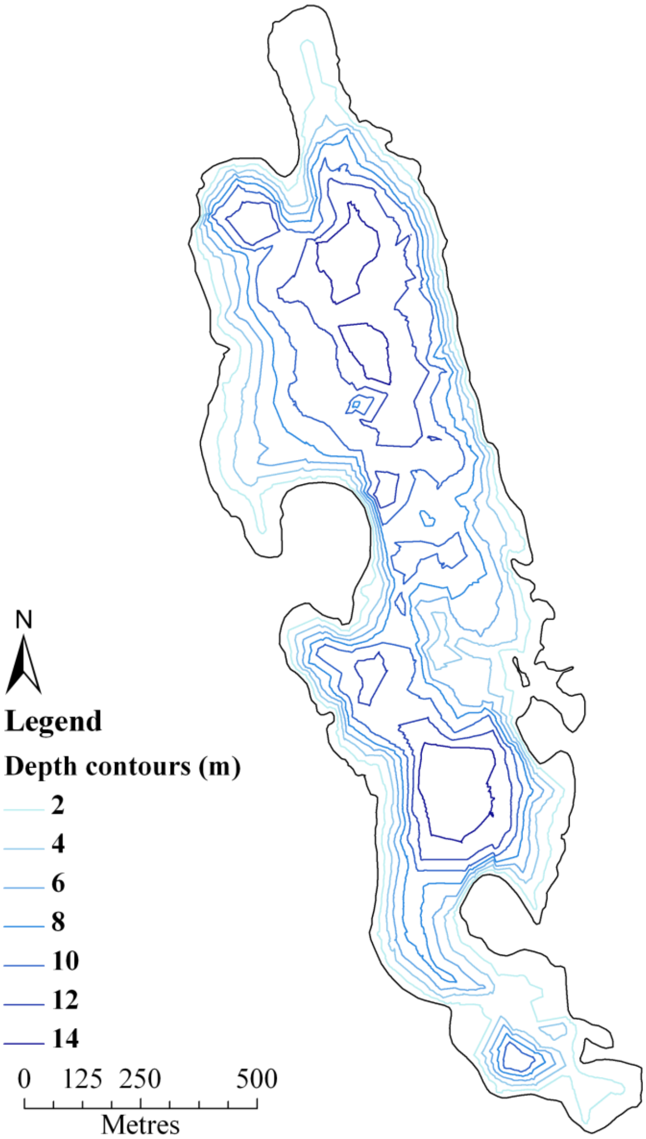

Figure 2.2 Bathymetry of Esthwaite Water, from Mackay et al. *(2011).*

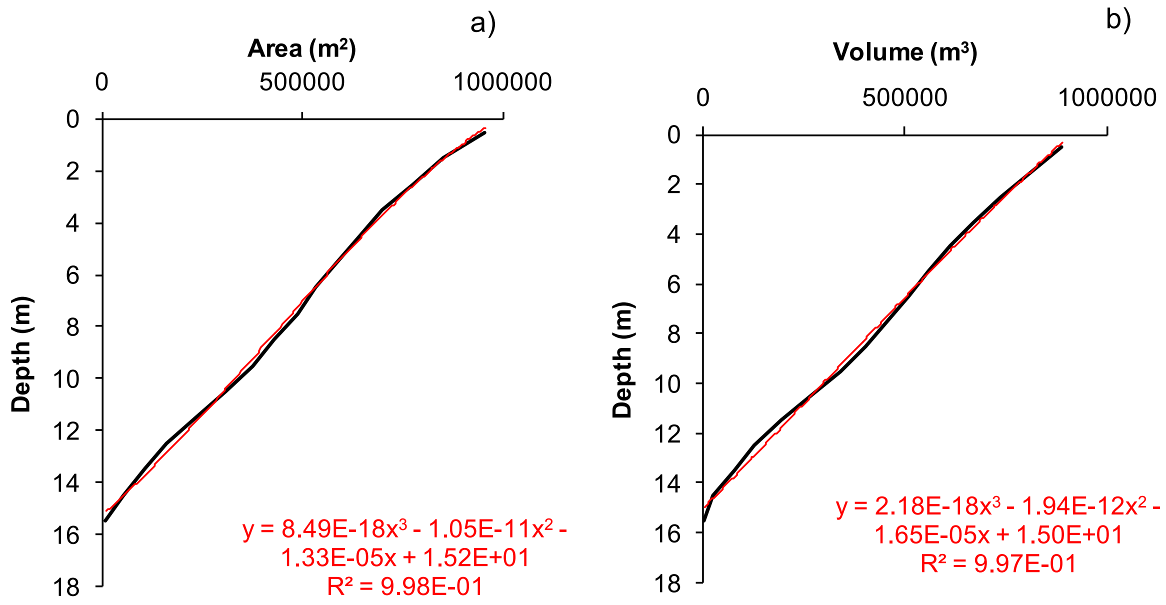

Figure 2.3. Hypsographic curves for Esthwaite Water showing a) area and b) volume calculated from data in Mackay et al. *2011. Results are based on 1 m depth intervals and areas and volumes are plotted against the mid-depth for each 'slice'. The trendline shown in red is a third-order polynomial equation fitted to the data (equation shown).*

2.2 Weather

The English Lake District is situated close to the western seaboard of north-west England and receives predominantly westerly or southwesterly winds from the North Atlantic, giving the region a relatively equitable maritime climate. The rainfall is generally high because as the Atlantic air rises over the mountains and cools, water condenses and falls as rain or snow. Meteorological data collected at Ambleside, about 8 km away, illustrates the average seasonal weather pattern (Fig. 2.4). None of the weather variables show any long-term trends at the annual scale between 1965 and 2010, apart from air temperature which has increased on average by 0.04 ^{o}C y^{-1} (Fig. 2.5).

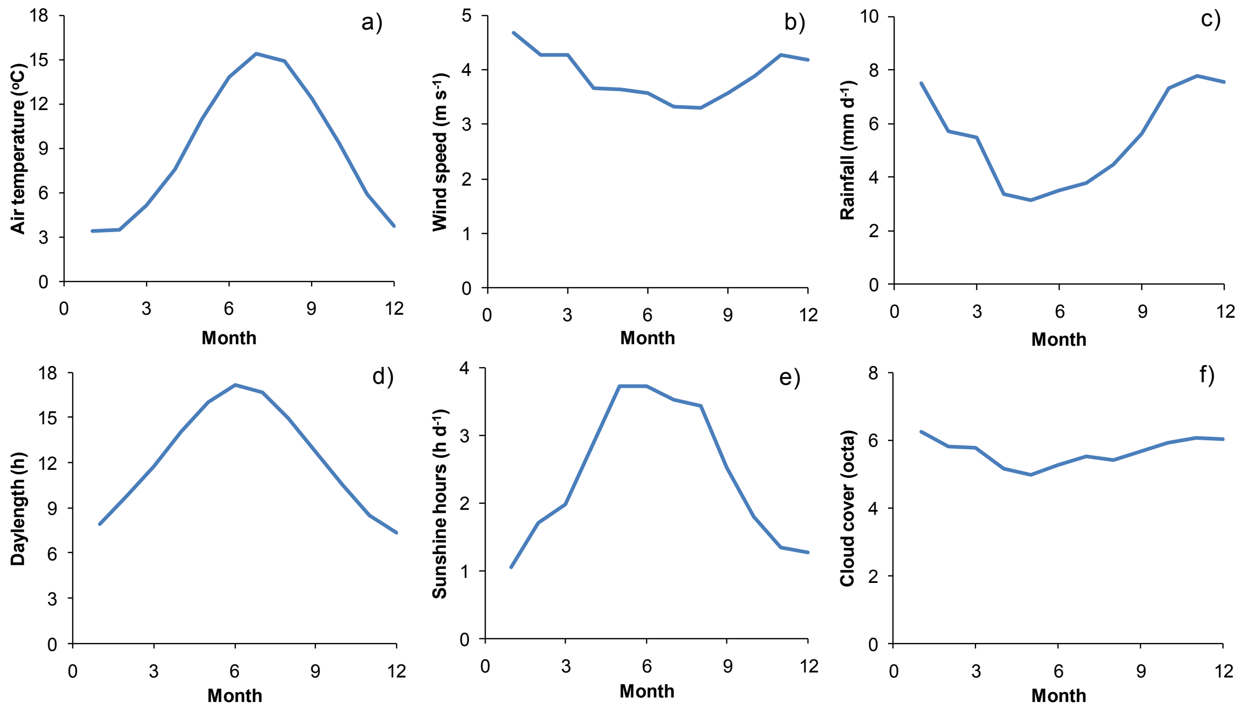

Figure 2.4. Average monthly weather- patterns at Ambleside (8 km NW of Esthwaite Water) between 1965 and 2010 showing: a) air temperature; b) wind speed, c) rainfall; d) calculated daylength; e) sunshine hours and f) cloud cover.

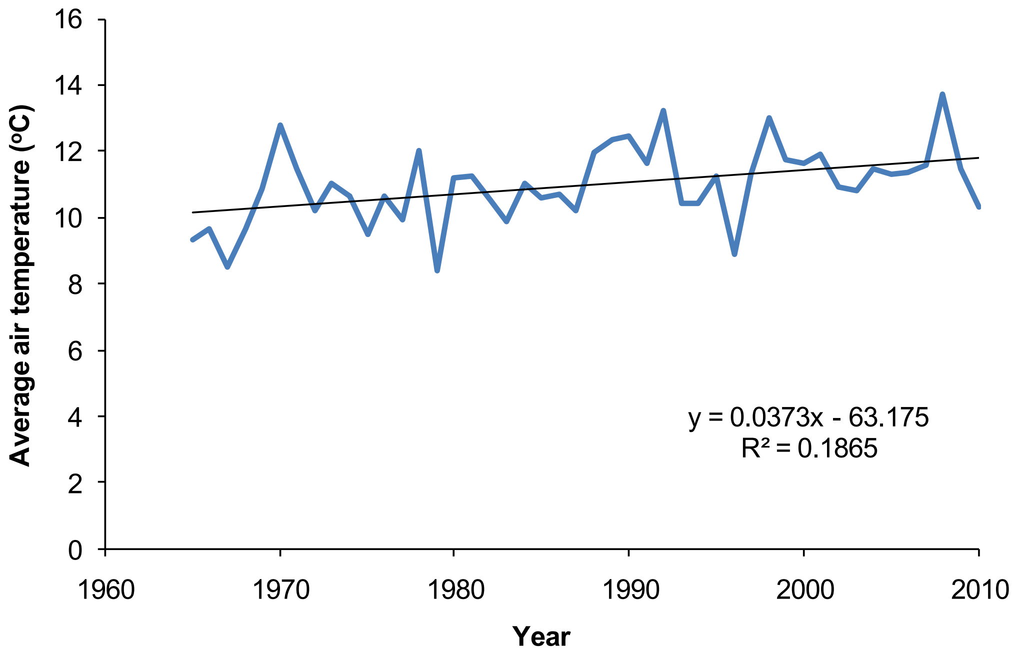

Figure 2.5. Long-term average annual air temperature at Ambleside (8 km NW of Esthwaite Water).

2.3 Pre-historical & historical changes

Palaeolimnological studies based on material preserved in the lake sediment (see Section 6.1) have allowed the past conditions in the catchment and the lake to be inferred. At the end of the last glaciation the retreat of the glaciers into the central mountains exposed bare ground that became colonised by sedge tundra and arctic-alpine shrubs such as *Betula nana* and *Salix herbacea.* As the climate warmed, copses of *Betula pubescens* and *B. verrucosa* developed but did not form a closed canopy (Franks & Pennington 1961). In subsequent cold periods the *Betula* copses reduced or disappeared and grass and sedge with *Betula nana* and *Salix herbacea* returned. Within the lake there are early records of the aquatic moss *Fontinalis* and later *Myriophyllum alterniflorum*, *M. spicatum*, *Littorella uniflora and Nuphar* sp. and diatoms such as *Melosira arenaria* (possibly a synonym for *Ellerbeckia arenaria* (Moore ex Ralfs)) (Round 1961). During the warmer post-glacial period, a *Pinus-Betula* woodland developed with increasing amounts of *Corylus.* Later, *Quercus, Ulmus* and *Tilia* appeared and *Alnus* became increasingly dominant. As *Betula* and *Pinus* declined a mixed *Quercus* forest developed around 8500 years ago (Franks & Pennington 1961; Pearsall & Pennington, 1947).

At the time of the colonisation of the lake district by megalithic people about 4000 years ago (Pearsall & Pennington, 1947) the oak forest would have been the dominant vegetation intermixed with *Festuca-Agrostis* grasslands and boggy valley bottoms with *Alnus* and *Salix*. Later Neolithic people appear to have mainly occupied the upper areas above the tree-line or in upland woods with little understory vegetation. The Roman invasion appeared not to have influenced the vegetation greatly and from about 3500 to 1100 years ago the main influence of man on the vegetation was the reduction of upland pine-heaths. The colonisation by Norse populations (around 900 to 1000 AD) led to the valley *Alnus*-boggy land to start to be cleared to provide land to graze sheep, a process that later expanded under the influence of the Cistercian abbeys. An additional reason for the destruction of the woodland was to provide fuel, especially to produce charcoal for smelting iron ore. By 1600 – 1700 AD deforestation was perhaps at its greatest and the large forges were forced to move to western Scotland where timber for charcoal was plentiful. The period following, up to the start of the Second World War, saw some reforestation as a result of the 'Romantic' influence and the start of commercial forestry. During the 1800s populations began to expand helped perhaps by the building of the railway line which reached Windermere in 1847 and increased access and tourism in the area. The period from 1850 onwards sees a steady expansion and intensification of agriculture (Dong *et al.* 2011a).

Although the focus in the above has been on land-use changes, there are additional, linked, changes in climate. For example, the period from 880 to 1350 AD is referred to as the 'Medieval Warm period' that was followed, 1350 – 1800 AD, by the era known as the 'Little Ice Age' (Dong *et al.* 2011a). Since around 1990, there is clear evidence for warming air and lake water temperatures in Cumbria, and elsewhere in Europe (Hari *et al.* 2006). There have also been recorded changes in precipitation in Cumbria over the last 200 years (Barker *et al.* 2004) with lower than average annual means between 1850 and 1900 and slightly higher than average precipitation between 1920 and 1960.

Recently, Esthwaite Water has been particularly heavily influenced by human activity. The potentially major recent impacts include:

- 1973 (November) input of sewage from the Wastewater Treatment Work (WwTW) serving Hawkshead
- 1981 (March) lowering the lake level by about 0.5 m
- 1981 introduction of a fish farm on the lake
- 1986 implementation of tertiary phosphorus removal at Hawkshead WwTW
- 1993 output from Near Sawrey WwTW redirected into Cunsey Beck below Esthwaite
- 2009 closing of the fish farm
- 2011-12 planned upgrade to Hawkshead WwTW

3. The catchment & its land cover

3.1 Geology & soils

Esthwaite Water has a catchment area of about 17 km^2 (Table 2.1). The underlying geology of the catchment comprises Silurian slates, grits and Bannisdale slates (Fig. 3.1) with the lower slopes covered in glacial tills of varying thicknesses. A thin band of Coniston limestone runs through the north of the catchment. The lake itself is underlain by Silurian slates (Fryer, 1991).

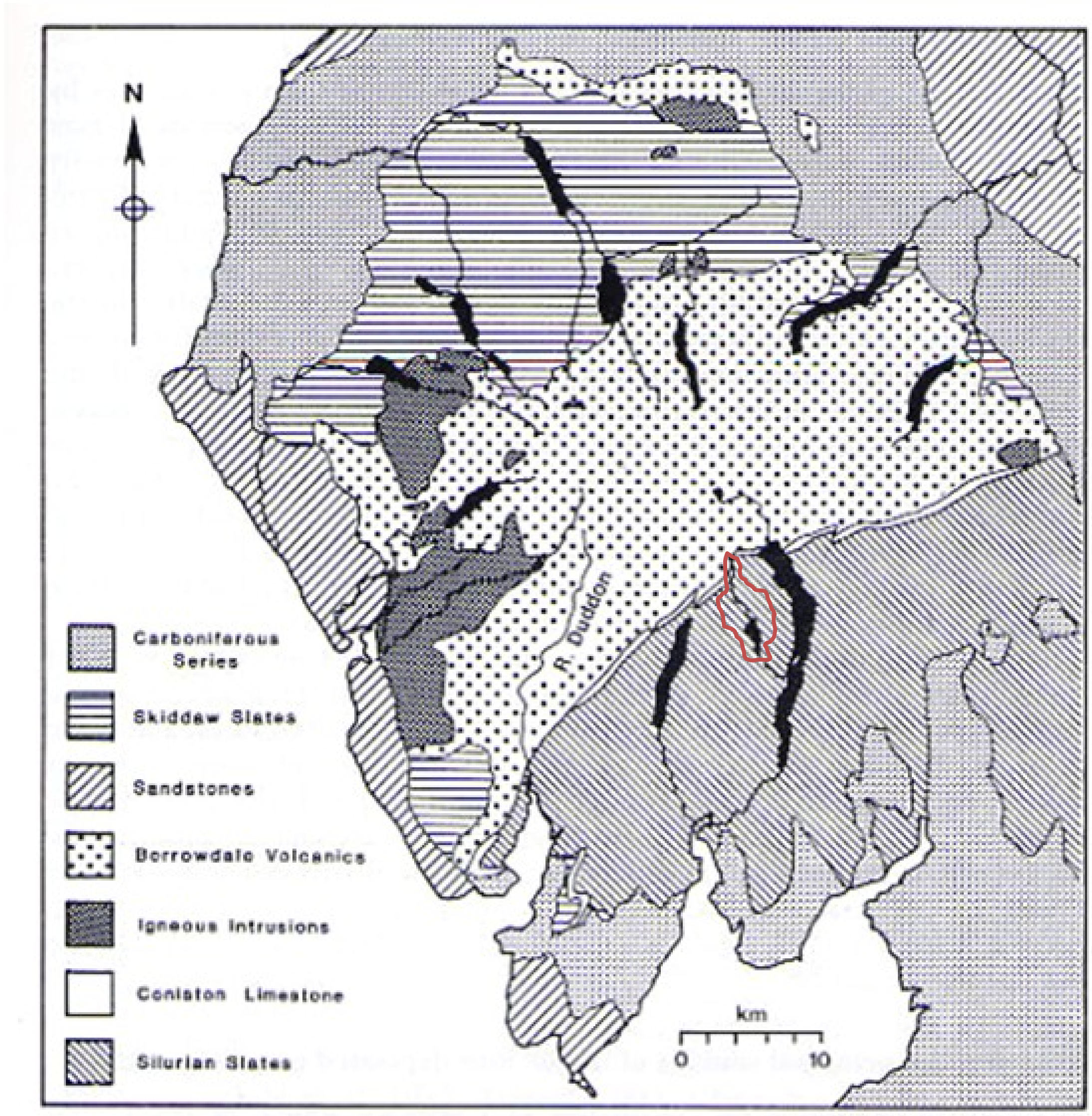

Figure 3.1. The underlying geology of the English Lake District (based on Sutcliffe, 1998). The approximate catchment area of Esthwaite Water is outlined in red.

3.2 Land cover

In 1988, the Lake District National Park Authority (LDNPA, Kendal) provided land cover data for the Esthwaite Water catchment in 38 different categories. These were aggregated into the 12 composite land cover groups shown in Figure 3.2 (May *et al.* 1995).

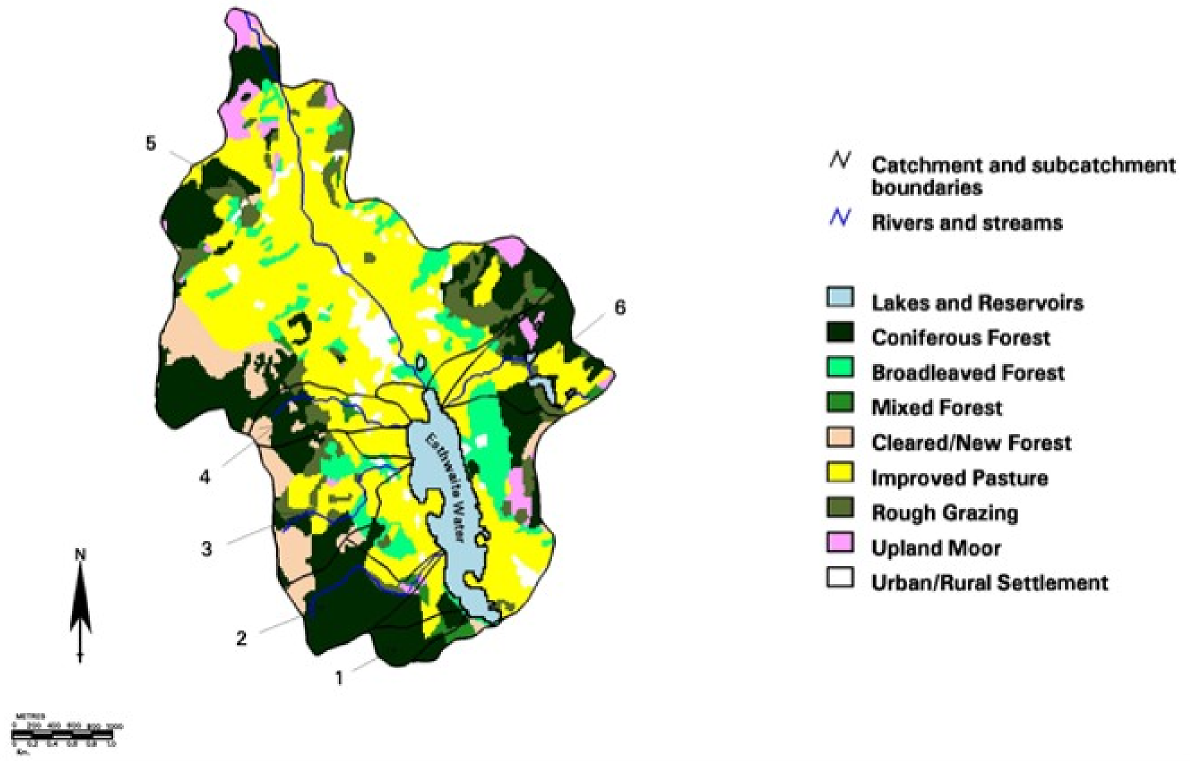

Figure 3.2. Land cover groups in the catchment of Esthwaite Water in 1988 based on data provided by the Lake District National Park Authority.

An alternative, further simplified, land cover categorisation of the land cover in the catchment of Esthwaite Water is that based on the European Corine Land Cover map (http://www.ceh.ac.uk/sci_programmes/BioGeoChem/CORINELandCoverMap.html). The areal coverage of each category is shown in Table 3.1. These data show that pasture contributes just over half of the total land area within the catchment and the two types of woodland about another third.

Table 3.1. Areal coverage of Corine land cover types within the Esthwaite Water catchment (excluding the lake).

Corine Gridcode	Land cover type	Area (ha)	% area
231	Pasture	867	55
311	Broad leaved forest	210	13
312	Coniferous forest	315	20
321	Natural grassland	170	11

4. Hydrology

The main national gauging station downstream of Esthwaite Water is at Eel House Bridge on Cunsey Beck (NGR SD369941), about 1.2 km below the main ouflow from the lake and below the small (1.1 ha) pond, Out Dubs Tarn. The gauging station drains a slightly larger catchment than Esthwaite Water of about 18.6 km^2 but in the period 1961 to 1990 had a slightly lower average rainfall of 1.899 mm y^{-1}. Consequently, the estimated outflow from Esthwaite Water is 0.92 of that at Eel House Bridge and this value has been used to convert all discharge data to appropriate values for Esthwaite Water. The National River Flow Archive (NRFA) at the Centre for Ecology & Hydrology (http://www.ceh.ac.uk/data/nrfa/index.html) archives these flow data and provides summary statistics.

Table 4.1. Summary time series data for site 73006 - Cunsey Beck at Eel House Bridge and equivalent values for the outflow from Esthwaite Water at Ees Bridge. All flow data as $m^3\ s^{-1}$.

Statistic	**Value at Eel House Bridge**	**Value at Ees Bridge**
Period of Record:	1976 - 2009	-
Percent Complete:	90 %	-
Base Flow Index:	0.41	-
Mean Flow:	0.928	0.85
95% Exceedance (Q95):	0.041	0.038
70% Exceedance (Q70):	0.263	0.242
50% Exceedance (Q50):	0.5	0.46
10% Exceedance (Q10):	2.269	2.088

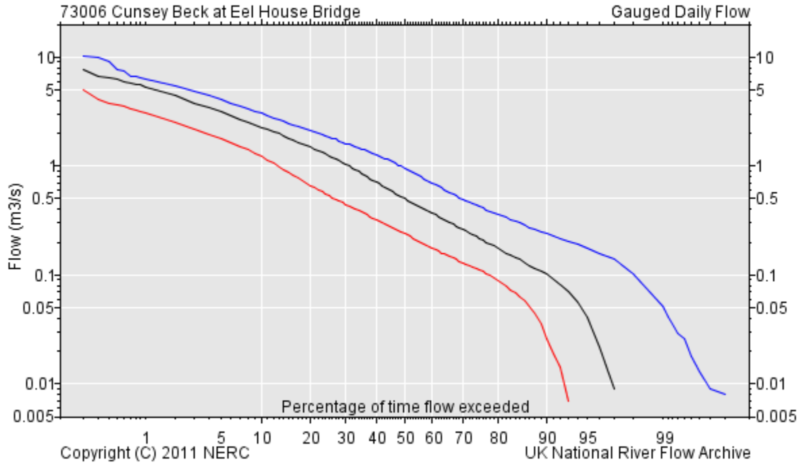

Figure 4.1. Flow-duration curve from the UK NRFA for station 73006 at Eel House Bridge showing annual (black line), December to March (blue line) and June to September (red line) curves.

The long-term hydrology is summarised in Fig. 4.2. The format used here will be used throughout the report for all the long-term records. The upper panel (a) shows the actual long-term record. Panel (b) shows changes in the annual mean value with error bars representing the annual standard deviation. Panel (c) shows the monthly seasonality with error bars representing the monthly standard deviation and also shows the correlation coefficient for long-term change in a particular month. This shows whether there has been a long-term increase or decrease in a value in a particular month and coloured circles on the correlation-plot show the statistical significance of the correlation.

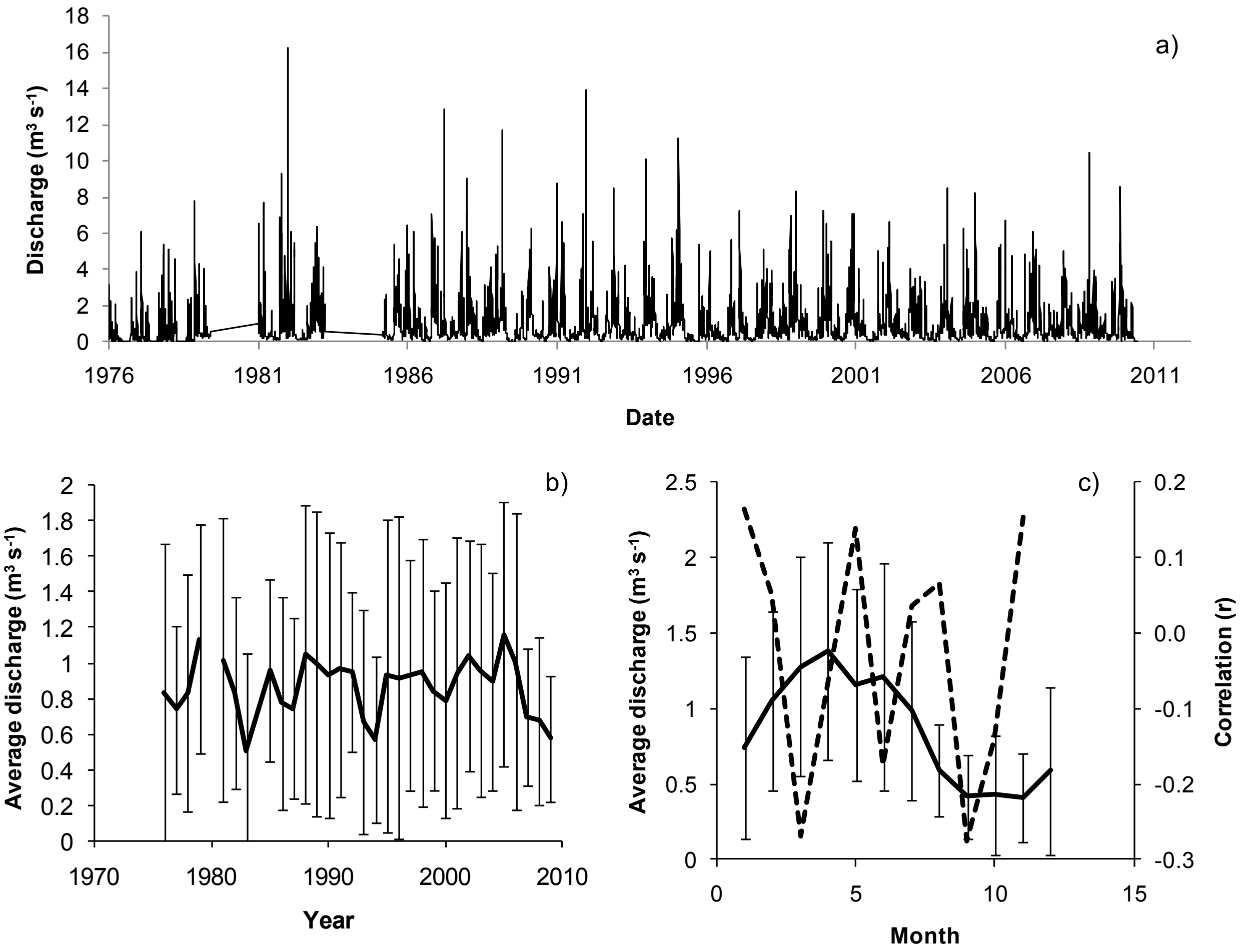

Figure 4.2. Discharge leaving Esthwaite Water, based on data from Eel House Bridge, corrected to outflow at Ees Bridge, see text. a) long-term record from 1976 to 2010; b) annual mean (error bars show standard deviation); c) monthly mean (error bars show standard deviation) plus long-term correlation of monthly change (dashed line), none of the correlations are statistically significant. Data kindly provided by the Environment Agency.

The discharge from Esthwaite Water has no consistent long-term change at an annual or monthly level, although there is a clear inter-annual variation, but the large error bars show the large range in discharge each year. There is a strong seasonal pattern with greatest discharge in the spring between March and June and lowest discharge in the autumn between September and November.

On the basis of catchment area, over half the catchment, and hence likely half the discharge, will derive from Black Beck that enters Esthwaite Water at the northernmost end of the lake (Table 4.2, Fig. 4.3). Elder Gill and Smooth Beck contribute around another 10 and 8 % respectively.

Table 4.2. Proportion of the total catchment drained by different streams.

Stream	**Proportion of Catchment area**
Black Beck	0.546
Elder Gill	0.096
Smooth Beck	0.083
Esthwaite Hall Beck	0.062
Howe Beck	0.061
Esthwaite Intake	0.028
Other	0.124

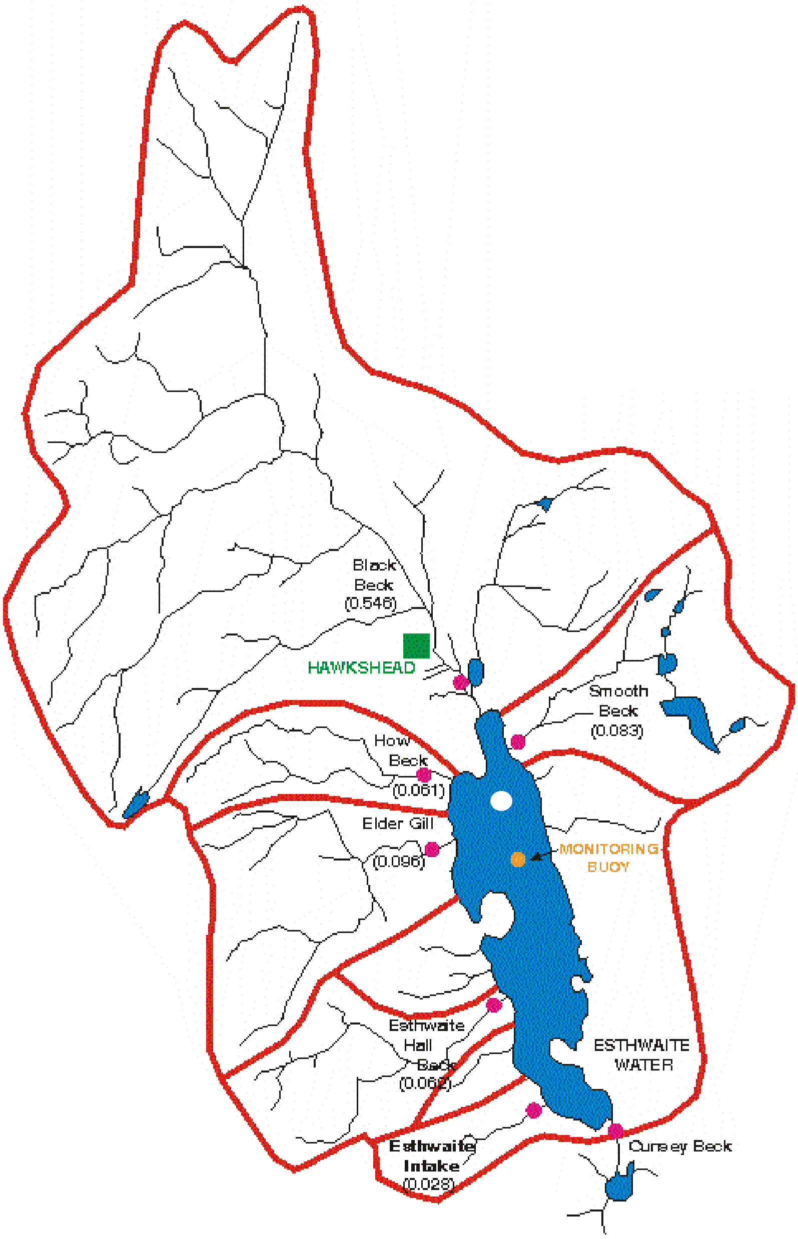

Figure 4.3. Map of Esthwaite Water showing the main inflowing streams and the proportional area of the different sub-catchments in parentheses. The location of the Hawkshead WwTW is shown in green, the long-term monitoring buoy is shown in white, the automatic monitoring buoy (see Section 7.2) on the lake is shown in orange and the sampling for phosphorus-loads from streams is shown in pink.

5. External & internal phosphorus loads to Esthwaite Water

5.1 Introduction

It is intrinsically very difficult to estimate nutrient loads to a lake because of the high degree of temporal variation in load and because a large number of different sources need to be taken into account. Nevertheless, a large number of studies on nutrient load to Esthwaite Water have been undertaken and the extensive in-lake information provides a way of testing the accuracy of the load estimates. The main external sources of nutrients to Esthwaite Water are: the load leaving the land, which includes losses from septic tanks and farming activities, the load from centralised Wastewater Treatment Works (WwTW), the load from the fish farm (external in the sense that the food is derived from outside the lake) and direct rainfall also carries a phosphorus load. In addition, because of the long history of nutrient enrichment of the lake, there is the possibility that nutrients in the sediment can be released back into the overlying water under some conditions, constituting an internal load. This review has focussed on phosphorus as it is the main element limiting productivity in the lake.

5.2 Wastewater Treatment Works (WwTW)

The new WwTW that had been constructed at Hawkshead in 1973 was estimated to be discharging about 409 kg y^{-1} of total phosphorus (TP) into the main inflow to Esthwaite Water (i.e. Black Beck) by Agar *et al.* (1988). However, May *et al.* (1997b) suggested that, while making a significant contribution to the total nutrient load into the lake, its net effect in terms of altering the overall TP load to the lake was probably relatively small at first. This was because it replaced many on-site sewage treatment facilities, such as septic tanks, that were in the vicinity of Hawkshead and which had previously discharged nutrient laden waste into Black Beck (Talling & Heaney, 1983). There was, however, a long-term advantage associated with this development in that it converted a diffuse source of TP (septic tanks) into a point source of TP (sewage effluent pipe); discharge from the latter could be controlled relatively easily at a later date. In 1986, North West Water (NWW) addressed this issue by introducing TP stripping by chemical precipitation into the treatment process at the Hawkshead works. Initially, this was carried out between April and October, only, but from 1989 onwards TP stripping was carried out all year round. This reduced the TP-load to the lake from this source by about 60%.

United Utilities are currently implementing further improvements to their wastewater treatment and handling processes. Currently, about 90 kg TP y^{-1} is estimated to reach the lake from overflow of untreated wastewater during times of high rainfall. The capacity of pumping stations and storage of wastewater for subsequent treatment is being increased to eliminate this source of phosphorus in the future. In addition, the tertiary treatment is being upgraded so that the consent annual average concentration of TP is reduced from 1.5 to 1.0 g m^{-3}. In practice, the actual averages are already below than the consent limit so it is not easy to estimate the reduction in TP-load from the WwTW in the future: but this has been conservatively estimated as 50 kg TP y^{-1} (Steve Rimmer, pers. comm.).

In addition to the above, Talling & Heaney (1983) also noted that, in the 1970s, the sewage effluent from the villages of Near Sawrey was also piped into the lake. Agar *et al.* (1988) estimated the TP-load from this source to be about 70 kg y^{-1} (April 1985 to March 1986; about 8% of their estimated TP-load from the Hawkshead WwTW).

5.3 The Fish Farm

A fish farm was established in the lake in 1981. This cultivated rainbow trout in floating cages positioned near the outflow. In the years that followed, fish production increased and this is thought to have been the main reason for the increasing autumn-winter soluble reactive phosphorus (SRP) levels that were recorded within the lake from the mid-1980s onwards (Section 8.5). Although exact figures were unknown, the total annual biomass of fish produced by this fish farm was estimated to be to be about 100 tonnes y^{-1} by Hall *et al.,* 1993. On the basis of this figure, and of work carried out on an experimental fish cage, Hall *et al.* (1993) estimated the likely TP-load to the lake from this source to be about 812 kg TP y^{-1}. The authors also noted that, as it was not possible to treat the effluent from floating fish cages to remove TP, any reduction in TP-load from this source would depend on good, environmentally friendly management and the use of low phosphorus fish food. From 1998 to 2008, smaller amounts of fish-food have been used at the fish farm in the switch to organic rearing of fish, although the food probably had a slightly higher P-content (N. Woodhouse, pers comm.). In autumn 2009, the fish cages were removed from Esthwaite Water so the direct input of food from this source will subsequently have been dramatically reduced. Based on the reductions in fish-food application, TP-load from the fish farm is roughly estimated as 338 kg y^{-1} in 2008, 135 kg y^{-1} in 2010 and a rough forecast of 100 kg y^{-1} in 2013. At the start of the fish farm operation less fish were being reared and the TP-load is estimated as 433 kg y^{-1}.

5.4 The catchment

Although the nutrient loads to the lake from the WwTW and fish farm were fairly well documented, May *et al.* (1997b) found that nutrient losses from land-based, diffuse sources were less well understood. Hall *et al.* (1993) had monitored the TP input to the lake from its six main feeder streams, but this had only given composite runoff figures for the TP-load from each sub-catchment; no information about catchment sources had been collected. May *et al.* (1997b) used a GIS-based export coefficient approach to investigate the potential catchment sources of these inputs by combining the land cover data for 1988 supplied by the LDNPA (Fig. 3.2) with nutrient export coefficients (Table 5.1), to estimate the TP-load.

5.4.1 Long-term changes in concentrations of phosphorus from Black Beck

The Centre for Ecology & Hydrology (and earlier the FBA) has been monitoring the nutrient concentrations at the main inflow to Esthwaite Water, Black Beck, for a number of years. This monitoring point is at Pool Bridge and so is downstream of Hawkshead village but upstream of the Hawkshead WwTW and so represents the major catchment input to the lake. The results in

Figure 5.1 show that there has been a marked decline in the concentration of both available SRP and TP over a number of years. Comparing the mean concentrations from 1986 to 1990 with those from 2006 to 2010, the mean concentration of SRP has fallen 2.2-fold from 22 to 10 mg m^{-3} and the concentration of TP has fallen 1.8-fold from 42 to 23 mg m^{-3}. The precise cause of this reduction is unknown but will relate to changes in application of fertilizer to the fields in the catchment and improved management of human waste from septic tanks. Since Black Beck contributes about 55% of the total hydraulic load to the lake and 86% of the estimated TP-load (Table 5.2), this reduction is likely to have had a significant effect on the phosphorus budget to the lake.

It is notable that the concentrations of SRP for June 1968 to June 1969 presented in Table 3 of Talling & Heaney (1983) are much higher than current values. The average was 81 mg m^{-3} and was strongly seasonal with summer concentrations reaching 540 mg m^{-3} in August 1968. These high concentrations presumably derive from high visitor numbers in summer and discharge from septic tanks, much of which is now processed at the WwTWs.

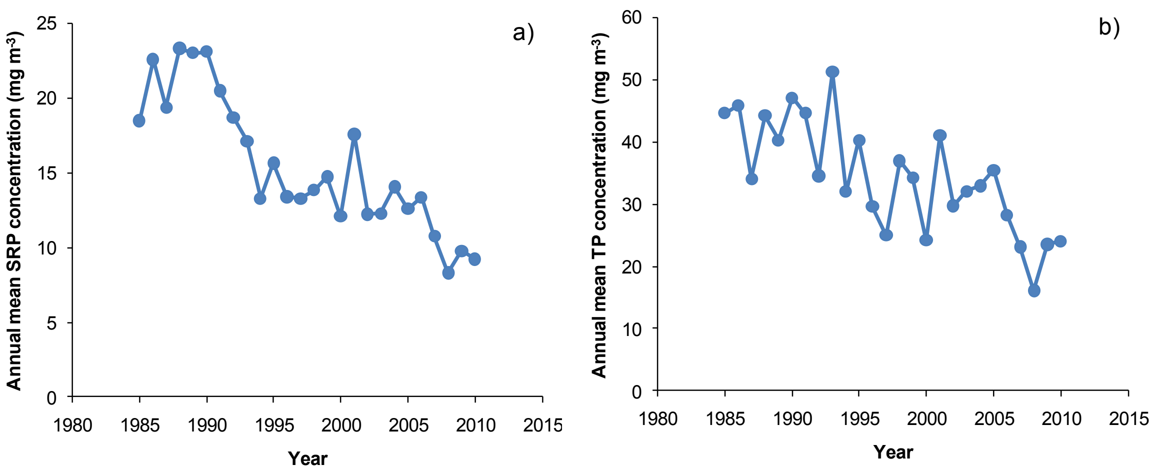

Figure 5.1. Long-term changes in annual mean concentration (mg m^{-3}) of a) SRP and b) total P in Black Beck at Pool Bridge.

5.4.2 Measured phosphorus load to the lake

The TP-load to Esthwaite Water from its inflows was estimated from measurements of concentration and discharge from June 1992 to May 1993, at a series of field sites collected at 4-weekly intervals from locations close to the mouths of the following feeder streams (Fig. 4.3):

1) Esthwaite Intake (Ridding Wood Beck)
2) Esthwaite Hall Beck
3) Elder Ghyll
4) Howe Beck
5) Black Beck (above the WwTW effluent discharge)
6) Smooth Beck

The results of this survey are summarised in Table 5.1. They showed that most of the inflows were small, each contributing less than 20 kg TP y^{-1} to the lake. The exception was Black Beck, the main inflow, which accounted for an annual TP-load from diffuse (land) sources of more than 400 kg y^{-1}. Hall *et al.* (1993) concluded that the total annual TP-load to the lake from diffuse sources within the catchment in 1992/1993 was about 480 kg y^{-1}. However, it should be noted that this value excluded any TP runoff from land close to the shore that drained directly into the lake and any TP input from rain falling directly onto the surface of the lake.

Table 5.1. Estimated annual TP-load to Esthwaite Water from its 6 main inflows and 2 main point sources, 1992 to 1993 (after Hall et al.*, 1993).*

Source	**TP-load (kg TP y^{-1})**	**TP-load (%)**
(1) Esthwaite intake (Ridding Wood Beck)	10.9	0.8
(2) Esthwaite Hall Beck	6.0	0.4
(3) Elder Ghyll	18.1	1.2
(4) Howe Beck	14.5	1.0
(5) Black Beck (above the WwTW effluent discharge)	415.8	28.4
(6) Smooth Beck	16.9	1.2
Total	**482.2**	**33.0**

5.4.3 Export-coefficient estimate of phosphorus load

May *et al.* (1997b) used a GIS-based analysis to attribute the measured TP-load to likely sources within the catchment. The 38 land cover classes within the original dataset, which was compiled in 1988, were reclassified into 12 composite land cover groups, as described by May *et al.* (1995). The reclassified data were then summarised for each land cover group in terms of their areal coverage within each sub-catchment. The TP-load to the lake from each sub-catchment was estimated as the product of the areal extent of each land cover group and its associated TP export coefficient (Table 5.2). The TP losses from land cover within the catchment as a whole were calculated by summing the individual estimates for each of the sub-catchments and for the un-gauged catchment. The TP-load from rain falling directly onto the lake surface was estimated from the mean annual rainfall at Ambleside and the estimated TP concentration of freshly fallen rain. This value was added to the estimated total TP-load for the lake.

Table 5.2 Land cover categories and related export coefficients (after May et al.*, 1997a).*

Land cover category	**TP Export coefficient (kg ha^{-1} yr^{-1})**	**Reference**
Urban/rural settlement (runoff, only)	0.83	*Bailey-Watts, Sargent , Kirika & Smith (1987)*
Upland moor	0.1	*Harper & Stewart (1987)*
Improved pasture	0.38	*May* et al. *(1997a)*
Coniferous forest	0.15	*May* et al. *(1997a)*
Cleared/new forest	0.2	*May* et al. *(1997a)*
Broadleaved forest	0.15	*Dillon & Kirchner (1975)*
Mixed forest	0.15	*Hancock (1982); Dillon & Kirchner (1975)*
Bogs & peat	1.0	*Casey* et al. *(1981)*
Inland bare rock	0.1	*May* et al. *(1995)*
Rough grazing	0.07	*Cooke & Williams (1973)*
Arable	0.25	*Cooke & Williams (1973)*
Other	0.1	*May* et al. *(1995)*

The TP losses from different land cover sources within the catchment, as estimated by May *et al.* (1997b), are summarized in Table 5.3. The TP-load from the 'ungauged' area of the catchment was estimated to be about 83 kg y^{-1} and a further 40 kg y^{-1} were found to come from rain falling directly onto the surface of the lake. These TP inputs were not included in the study of Hall *et al.* (1993), so the overall TP-loading estimate of 482 kg TP y^{-1} from the catchment that they gave may have underestimated the external TP-load to the lake by about 120 kg TP y^{-1}. Of the different land use categories, improved pasture and forestry were the greatest contributors to TP-load. Overall, the direct measurements from stream flow and the estimate from land cover gave similar, but not identical, estimates of TP-load to Esthwaite Water

Table 5.3. Estimated annual TP-load to Esthwaite Water based on land cover (after May et al.*, 1997b).*

	Area		**TP-load to lake**
Source	**(ha.)**	**(%)**	**(kg y^{-1})**
Urban/rural settlement (runoff only)	54.2	3.5%	45.0
Upland moor	52.2	3.3%	5.2
Improved pasture	704.2	45.0%	267.6
Forestry	638.8	40.8%	100.9
Rough grazing	108.3	6.9%	7.6
Other	0.1	0.0%	0.0
Subtotal	**1557.8**	**99.5%**	**426.3**
Rainfall			40.5
Total			**466.8**

5.5 Internal load

The phosphorus present in lake sediments can be released back into the lake water by physical and chemical processes. In shallow sediments, physical disturbance by wind and wave can mix the interstitial water, relatively rich in phosphate, into the overlying water. In Esthwaite Water, for example, interstitial concentrations of SRP can reach 400 to 600 mg m^{-3} (Drake & Heaney, 1987). Chemical conditions at the sediment surface, particularly pH and redox potential, can also regulate nutrient release via their effect on iron chemistry (Bostrom *et al.*, 1988). Ferric iron (Fe^{3+}) is present in oxic conditions and this binds phosphate effectively. If the oxygen concentration at the sediment surface falls, the redox potential also falls and ferric ions change to ferrous (Fe^{2+}) ions that no-longer bind phosphate. This phosphate is then free to diffuse out of the sediment into the overlying water. This principle was originally demonstrated by Mortimer (1941, 1942) while working on Esthwaite Water. An additional chemically-mediated mechanism leading to the release of phosphorus from sediments is the effect of elevated pH. Drake & Heaney (1987), working on Esthwaite Water, showed that rates of phosphorus release increased rapidly with increasing pH. Since high pH can be present in the surface of Esthwaite Water in the summer as a result of inorganic carbon depletion (see Sections 8.3 and 8.4), this could be an important additional mechanism of internal P-loading.

Quantifying the internal load of phosphorus to a lake is even more difficult than quantifying the external load. Unpublished work carried out by Helen Miller as part of her PhD thesis (Miller 2008) attempted to estimate the internal load from the hypolimnion to the epilmnion in Esthwaite Water. Detailed measurements of depth-profiles of SRP were combined with depth-profiles of stability and estimates of eddy-diffusion (the process leading to the transport of an ion from high to low concentration). The calculations suggested that in August and September when SRP concentrations at depth are high, the internal load can contribute a large proportion of the total SRP supply (Fig. 5.2). On an annual basis, this is equivalent to about 16 kg. Although relatively small, it could be important ecologically, as in summer phosphorus is in short supply and in particular potentially-harmful cyanobacterial blooms are most likely to occur then.

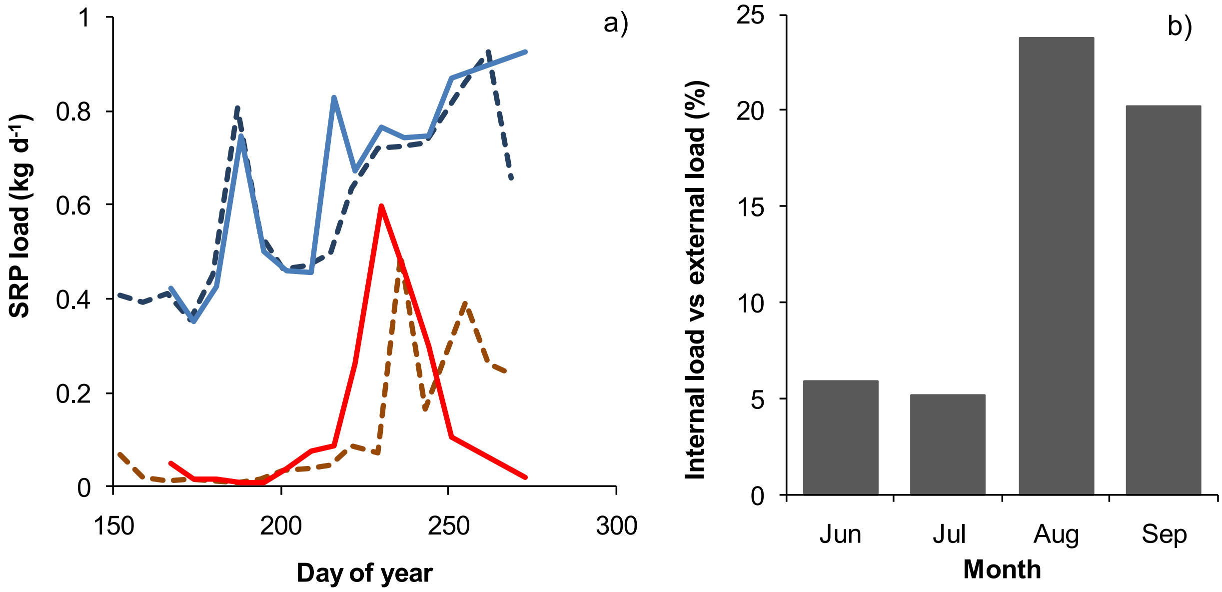

Figure 5.2. Estimated internal load of phosphorus from the hypolimnion to the epilimnion of Esthwaite Water, a) daily SRP-load from external sources (blue lines) and from the hypolimnion (red lines) in 2004 (solid lines) and 2005 (dashed lines) and b) as a monthly percent of total load as an average of the two years. Miller (2008).

5.6 Source apportionment

The different contributing sources of TP to Esthwaite Water are brought together here to estimate the total load, and their percent contribution to the total, based on the numbers discussed in Sections 5.2 to 5.5 which are largely based on measurements in 1992-93. The total estimated load for that period is 1677 kg TP y^{-1} (Table 5.4) of which, roughly, 29% derives from the catchment, 20% derives from the WwTW and their operation in aggregate and 48% derives from the fish farm.

Table 5.4. Source apportionment of the estimate annual TP-load to Esthwaite Water 1992-3.

Source	Annual TP-load (kg y^{-1})	Percent total load	Notes
Catchment	482	29	Table 5.1
Direct rain	41	2	Table 5.3
Hawkshead WwTW	166	10	Section 5.2
Hawkshead intermittent discharge	90	6	Section 5.2
Near Sawrey WwTW	70	4	Section 5.2
Fish farm	812	48	Section 5.3
Internal load	16	1	Section 5.5
TOTAL	**1677**	**100**	

5.7 Changes in phosphorus load and effect on lake TP concentration

This section attempts to reconstruct the phosphorus load to Esthwaite Water for six time periods:

i) 1968-1969, before the WwTW and fish farm were in operation;

ii) 1985-1986, based largely on the report of Agar *et al*. (1988) and immediately before the implementation of the tertiary treatment at the Hawkshead WwTW;

iii) 1992-1993, after tertiary treatment and a time period for which load data are available (Table 5.4);

iv) 2008, a year when the intensity of the fish farm was reduced;

v) 2010 after the closure of the fish farm;

vi) forecast for the future after the upgrade to the WwTW (2013).

It is apparent from the sections above that estimating nutrient loads is extremely approximate so all the values used below are best approximations and not absolute numbers.

Table 5.5. Approximate loads of phosphorus (kg y^{-1}) to Esthwaite Water from various sources in different time periods. The column and row identifiers provide a link to the notes below the table.

	Column identifier	**A**	**B**	**C**	**D**	**E**	**F**
Row identifier	**Source**	**1968-9**	**1985-6**	**1992-3**	**2008**	**2010**	**Forecast 2013**
1	Catchment	1321	482	482	257	299	278
2	Direct rain	41	41	41	41	41	41
3	Hawkshead WwTW	0	409	166	166	166	115
4	Hawkshead intermittent discharge	0	90	90	90	90	0
5	Near Sawrey WwTW	0	70	70	0	0	0
6	Fish farm	0	433	812	338	135	100
7	Internal load	16	16	16	16	16	16
	TOTAL	**1378**	**1541**	**1677**	**908**	**747**	**550**

Notes

A1, derived from weekly measurements on Black Beck in Talling & Heaney (1982)
B1, based on C1 and annual mean TP concentration in Black Beck (Fig. 5.1b)
C1, Table 5.2
D1, based on C1 and annual mean TP concentration in Black Beck (Fig. 5.1b)
E1, based on C1 and annual mean TP concentration in Black Beck (Fig. 5.1b)
F1, Average of D1 and E1
A2-F2, Table 5.3
A3, Before Hawkshead WwTW commissioned
B3, Agar et al. *(1988)*
C3-E3,Table 5.4
F3, Steve Rimmer, pers. comm. & Section 5.2
A4, Before Hawkshead WwTW commissioned
B4-E4, from Steve Rimmer pers. comm.& Section 5.2
F4, After upgrades to pumping station & tank capacity (Steve Rimmer, pers. comm.) & Section 5.2
A5, Before Near Sawrey WwTW commissioned
B5-C5, Agar et al. *(1988), Steve Rimmer pers comm. & Section 5.2*
D5-F5, After Near Sawrey WwTW re-routed below the Cunsey Beck outflow
A6, Before fish farm in operation
B6, Based on C6 and reduced fish food application, Section 5.3
C6, Table 5.4
D6, Based on C6 and reduced fish farm application, Section 5.3
E6, Based on C6 and reduced fish farm application, Section 5.3
F6, Based on C6 and reduced fish farm application, Section 5.3.

The results are shown in Table 5.5 along with the assumptions made to produce them. The results are also presented graphically as total load (Fig. 5.3a) and percent load (Fig. 5.3b). The results show that before the operation of the WwTWs and the fish farm, the catchment supplied most of the phosphorus to the lake, some of which will have derived from inputs from septic tanks. With the operation of the WwTWs, in the mid 1980s, the sum of loads from the catchment plus WwTWs is about 1051 kg TP y^{-1}, which on the face of it implies that about 270 kg TP y^{-1} is removed at the WwTWs. The operation of the fish farm imposed a large additional load of TP to the lake, in both absolute and relative terms (Fig. 5.3), reaching a peak in the early 1990s before declining. Estimated loads in 2010 derive mainly from the catchment (40%), the WwTWs (34%) and the fish farm (18%).

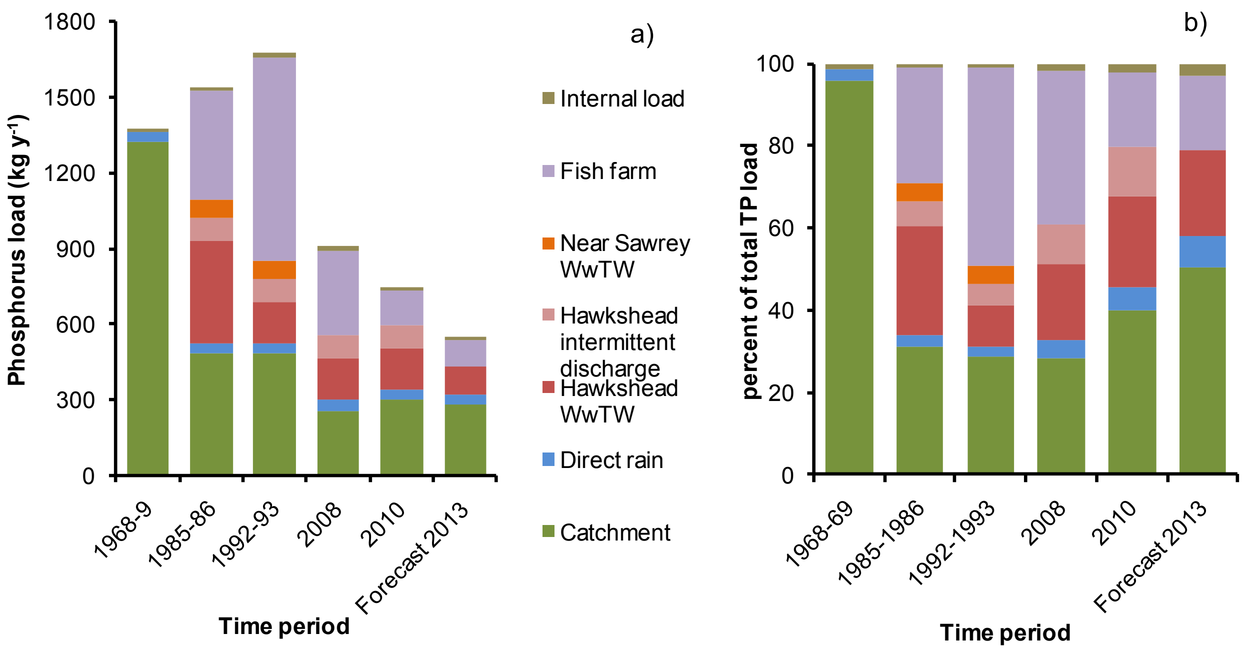

Figure 5.3. Proportion of the total annual load of TP to Esthwaite Water from different sources at different periods of time. Derived from values in Table 5.5.

It is possible to estimate the in-lake concentration of TP from the nutrient load, hydrology and bathymetry of a lake. A number of different formulations exist, all are approximations. One that has been shown to work well for Windermere (Maberly 2009) is that of Kirchner & Dillon (1975):

$$P = (L_p / q_s) * (1 - R)$$ Equn 1

where: P is in-lake concentration of TP (g m^{-3}), L_p = annual TP-load (g m^{-2} y^{-1}), q_s = water discharge height (m y^{-1}) and R is a dimensionless retention rate calibrated using the following equation:

$$R = 0.426e^{-0.271*Zm/qs} + 0.574e^{-0.00949*Zm/qs}$$ Equn 2

where Z_m is mean depth (m). Z_m is derived from Table 2.1, as is q_s and this was not altered to take account of inter-annual variation in hydrology.

There is a reasonable agreement between estimated and measured annual average concentration of TP (Fig. 5.4). Based on the measured concentrations, in 1968 and 1985 the loads may be a slight overestimate, while in 2010 they appear to be an underestimate of the real value.

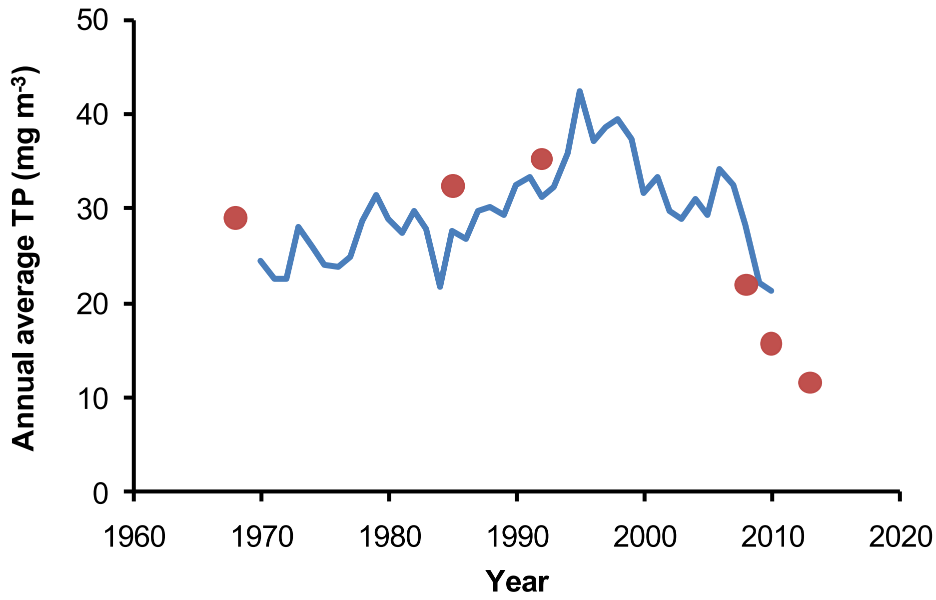

Figure 5.4. Comparison between annual mean concentrations of TP measured in the long-term monitoring (blue line) and estimated from the annual loads using the formula of Kirchner & Dillon (1975).

6. Sediment

6.1 Variation with depth

When Esthwaite Water was forming by the retreat of glaciers and ice sheets around 12,000 years ago (Section 2.3), the base of the lake will have comprised bedrock covered by recent glacial till that was accumulating from in-wash of easily-transportable boulders, gravel, sand and clay from the surrounding moraines and catchment. The rapid inflow of material from the catchment could have produced a lake with a relatively high nutrient concentration and productivity. Since those post-glacial times, the sediment in the lake has been gradually accumulating and the inorganic fractions supplemented by organic carbon produced within the catchment and the lake itself. In the deepest parts of the lake the sediment is over 5 m deep and, at about 5 m, comprises greyish silts and clays that were laid down in the late glacial period (Round 1961). These are overlain by brown lake muds and clays up to a depth of 3 m which derive from the Boreal and Atlantic periods and the top 3 m comprise brown lake mud from the post-Atlantic period (Round 1961). More recent studies of sediment accumulation (Bennion *et al.* (2000) and especially the detailed study of Dong *et al.* (2011a) who analysed an 86 cm core where the base represent about 780 AD give greater detail. There have been large fluctuations in chemical composition and rate of sedimentation over the last 1200 years. Sedimentation rate was about 0.02 g cm^{-2} y^{-1} in 800 AD, increasing to about 0.04 g cm^{-2} y^{-1} in the 1800s and it has been about 0.06 g cm^{-2} y^{-1} in the last thirty years. These and other changes can be attributed to changes in land-use in the catchment from earlier forest clearing and commencement of agriculture around 800 AD, more extensive clearance after 1700 AD and intensification of agriculture after around 1880 AD. The current sedimentation rate is in the order of 0.057 cm y^{-1} based on a density of 1.05 g cm^{-3} (Hilton & Gibbs 1984).

Sediments also preserve fossilised algal, plant and animal remains. These can be identified and enumerated and used to infer past conditions within the lake and its catchment. There is an extensive literature on the palaeolimnology and Round (1961), Bennion *et al.* (2000) and Dong *et al.* (2011a) give examples of the powerful information that can be learnt from this approach.

Bennion *et al.* (2000) analysed a core taken in 1995 from the deepest point of Esthwaite Water that dated back to about 1740 and calculated sediment accumulation rates to be 0.0379 ±0.002 g cm^{-2} y^{-1} up until around the mid 1850s but contemporary rates were two to three times higher indicating a large increase in input of material to the lake in recent times or an increased input by production in the lake, or both. A more recent palaeolimnological study on a core collected in 2006 dated back to 780 AD also estimated concentrations of TP from the diatom composition (Dong *et al.* 2011a). There was little change in inferred TP concentration up until 1850, with an average concentration of 10 mg m^{-3} (Fig. 6.1). This was followed by a slow but steady increase in TP concentration up to around 1970 when it had reached about 19 mg m^{-3}. This was followed by a dramatic increase in concentration to around 60 mg TP m^{-3} at the top of the core in 2006. While the inferred TP concentrations overestimate the actual TP concentrations compared to actual measurements in the lake, the timing and rates of change are likely to be very reliable and the dramatic increase in concentration around 1970 matches the installation of the WwTW works serving Hawkshead in 1973. These data show that Esthwaite Water has been influenced by Man's activities for several centuries, but the largest and most rapid change has occurred within the last 40 years.

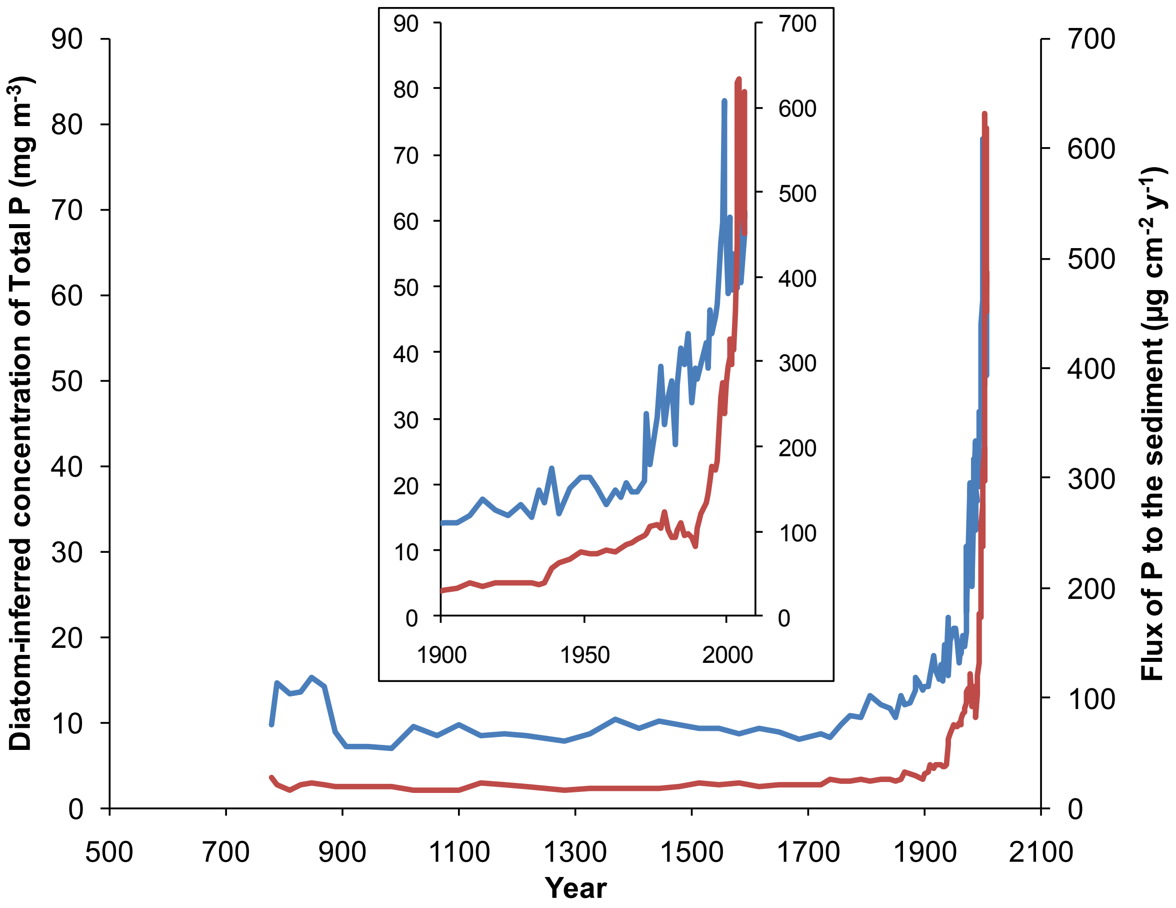

Figure 6.1. Long-term changes in the Total phosphorus concentration inferred from diatom samples (blue line) and flux of phosphorus to the sediment (red line) to Esthwaite Water from AD780 to the present day. The inset shows values from 1900 onwards. Dong et al. *2011a & Dong, Maberly, Sayer, Bennion & Battarbee (unpublished).*

6.2 Horizontal variation

Although there has been a tendency to study vertical patterns within a core from a single location within a lake, there is also a large amount of horizontal heterogeneity in chemical composition (Hilton & Gibbs, 1984). Based on an extensive survey of 116 samples, Hilton & Gibbs (1984) found that the average (standard deviation) density was 1.05 g cm^{-3} (0.03), the percent dry solids was 9.2% (2.8%) and as a percent of dry solids, the organic matter content was 33.3% (5.5%) the carbon content was 15.9% (3.2%) and the nitrogen content was 1.29% (0.43%). The P_2O_5 content was 1.15% (0.61%) of the ash. Different elements were distributed differently in the lake. For example, dry weight as percent of wet volume tended to be greater in the northern and western part of Esthwaite Water whereas organic carbon and its correlates such as N and P tended to be greatest in deeper water.

Table 6.1. Characteristics of the surface sediments in Esthwaite Water (Mackay et al. *2011).*

Sediment characteristic	**Mean**	**Standard deviation**	**Coefficient of variation (%)**
Water content (%)	86.5	4.0	4.6
Clay (<2 µm)	1.8	0.4	19.4
Silt (2 – 63 µm)	82.9	10.4	12.6
Sand (63 – 1000 µm)	15.3	10.6	69.5
Total phosphorus (g kg^{-1})	3.14	1.48	47.3

A recent study provided average sediment characteristics for Esthwaite Water based on samples from 29 sites (Mackay *et al.* 2011). This also showed that while water content was very constant among sites, sand and TP content were very variable (Table 6.1). In the case of phosphorus, the content increased markedly with water depth (R^2 = 0.82): values in shallow water were about 1 g kg^{-1} increasing to about 5 g kg^{-1} at 14 to 15 m.

The spatial patterns of sediment characteristics result from physical mixing processes (Section 7.5). Wind-induced, water-current driven resuspension and transport of small particles, which often have a high phosphorus-content, is the main cause of the spatial patterns in sediment characteristics observed in Esthwaite Water (Mackay *et al.* 2011).

6.3 Rôle of sediments in the phosphorus budget of the lake

The sediment is a site where material can be stored or released back to the water. The material can be brought in from the catchment (allochthonous) or produced within the lake (autochthonous). A recent, spatially-resolved, study by Mackay *et al.* (2011) estimated a contemporary phosphorus burial rate of 1000 kg y^{-1}. The samples were taken in 2009 from the top 2 cm which represents roughly 35 years based on an accumulation rate of 0.057 cm y^{-1} (Section 6.1). With an average discharge of 26.9 Mm^3 y^{-1} (Table 2.1) and an average TP concentration from 1974-2009 of 30.5 mg m^{-3}, the average loss of phosphorus from the lake is about 820 kg y^{-1}, implying that about 55% of the phosphorus load is retained within the sediment.

Early pioneers within the Freshwater Biological Association recognised the importance of the sediment as a record of past change and as a source of material to the water. The seminal work of Mortimer (Mortimer 1941-2) was based largely on Esthwaite Water. He showed that when the sediment surface becomes anoxic, the redox-potential becomes negative and iron changes valence from ferric (3+) to ferrous (2+). The significance of this is that Fe^{3+} is able to bind phosphate ions whereas Fe^{2+} is not so that, under reducing conditions, phosphate in the interstitial water is free to diffuse across the sediment surface into the hypolimnion. This can then recycle phosphorus within the lake constituting an internal load (Sections 5.5, 8.5). The internal load of phosphorus to the epilimnion via eddy-diffusion from the hypolimnion is a small proportion of that deposited to the sediment but could be ecologically important (Section 5.8).

7. Physical characteristics: meteorology, temperature, water movement & light attenuation

7.1 Introduction

Physical processes can have a large effect on lake function and are the link between many of the responses of a lake to climate change. This section draws on much new and unpublished information and so the way the data were collected is briefly described.

7.2 Data sources and calculations

Meteorological and in-lake temperature measurements have principally been taken from two sources: an automated monitoring station deployed on the lake since 2004 and a long-term monitoring programme started just prior to 1950. The long-term monitoring records included temperature profiles taken weekly or fortnightly at the deep point within the north basin of the lake. Although the earlier profiles did not regularly penetrate deep into the water column, since summer 1956 vertical profiling became the norm. Analysis here has therefore been restricted to the period from 1957 to 2010, inclusive, between 0 m ('surface temperature') and 14.5 m ('bottom temperature'). These temperature profiles have been linearly interpolated between depths when necessary to fit a 1 m vertical grid, and then linearly interpolated between sampling times to form a daily grid. This 1 m daily grid has then been used for the analysis below.

The automated monitoring station in the north basin of Esthwaite Water consisted of a set of 12 in-lake temperature sensors (Thermospeed, Bolton, UK) hanging each metre between depths of 0.5 m and 11.5 m, and a suite of meteorological instruments on a mast approximately 2.5 m above the water surface. These instruments included a pyranometer (CM3, Kipp & Zonen, Delft, Holland) to measure short-wave radiation, an anemometer (SKH2012, Skye Instruments, Llandrindod Wells, Wales) measuring wind speed at 2.9 m, an air temperature sensor (A100L2-WR, Vector Instruments, Rhyl, North Wales) and a relative humidity sensor (Hobo Pro RH/Temp, Onset Computer Corporation, Pocasset, USA) both initially sited at 1.5 m, but raised to 2.1 m in April 2007. The short-wave radiation, air temperature and wind speed were measured every minute, the in-lake temperatures every two minutes and the relative humidity every four minutes. All data were then automatically averaged each hour and recorded, and ultimately then daily averaged. As there are inevitably gaps in this type of automated data, extra meteorological data have been obtained from an array of additional sources; mainly a meteorological station located on the shore of the north basin of the lake, but also from meteorological stations based at neighbouring Blelham Tarn, at Windermere and at The Ferry House. The shore station was also used as the source for wind direction data because the lake buoy is not a rigidly-fixed platform. Additional relative humidity data were taken from British Atmospheric Data Centre meteorological stations at Shap, Walney Island and Keswick, particularly for 2004 when there was no relative humidity instrument on the lake or the shore station. Data obtained from these other sources were regressed against the Esthwaite Water monitoring buoy data during periods when both data sets were available, and these regressions then used to correct the additional data. In total 97.5 % and 2.3 % of air temperature data were taken from the Esthwaite lake buoy and the Esthwaite shore station respectively, as were 92.9 and 6.4 % of wind speed data, 73.6 and 23.4 % of short-wave radiation data and 77.7 and 2.7 % of relative humidity data. In-lake temperatures were all but continuous from start 2004 to end 2009 and linearly interpolated through time when missing, but no data were recorded in 2010, so monitoring buoy data have been analysed from January 1st 2004 to December 31st 2009. Cloud cover data were taken by Mr Bernard Tebay at Ambleside, twice daily during the period.

The buoy data were used to derive some relevant secondary parameters, most notably the principal fluxes of heat to the lake. These fluxes, in combination with mixing from the wind, drive the temperature structure of the lake and are the mechanism for which any change in climate will initially impact on a lake. There are a number of these fluxes, each dependent on a different combination of meteorological and in-lake parameters. Of these the most obvious is the solar radiation (sunlight) usually referred to as short-wave radiation as it occupies a characteristic band in the electromagnetic spectrum. This has been directly measured by an instrument on the buoy. A proportion of this radiation, dependent on the time of year, is reflected from the surface of the lake. This reflected short-wave radiation has been calculated from Cogley (1979). Both the atmosphere and the lake emit radiation, but in the longer part of the electromagnetic spectrum. The upward long-wave radiation from the lake depends on the surface temperature of the lake, whilst the downwelling long-wave radiation from the sky may be estimated from atmospheric temperature, atmospheric relative humidity and cloud cover. Both types of long-wave radiation have been calculated from Josey *et al.* (2003). There are also two so-called 'turbulent' fluxes of heat which are driven by the turbulent motion of fluid at the water-air interface. One, the sensible heat flux, that is heat that can be sensed, may be calculated from the speed of the wind, and the temperature gradient between the air and the surface water. The other, the latent heat flux depends on the evaporation of water and may be estimated from the wind speed, the temperatures of the lake surface and the air and the relative humidity of the air. Both these turbulent fluxes may be calculated from standard formulae (e.g. Gill, 1982), but do require a complex iterative procedure for estimating the effects of atmospheric stratification on the process. Here the procedure outlined in MacIntyre *et al.* (2002) has largely been followed, although parameterisations from the review by Högström (1996) have been used, as well as a neutral 10 m transfer coefficient of 1.0×10^{-3}. Here downward heat fluxes are denoted as negative and upward fluxes are denoted as positive. Therefore, negative heat fluxes would warm the lake and positive heat fluxes would cool the lake.

The mixed layer of the lake has been defined as the shallowest depth which has a temperature more than 1 ^{o}C less than the surface temperature. Similarly, the lake has been defined to be stratified so long as the surface temperature is more than 1 ^{o}C higher than the bottom temperature. The strength of the stratification has been estimated from the difference between the top and bottom temperatures from the long-term monitoring data. As the buoy does not record temperatures at the very bottom of the lake a separate measure of the temperature gradient (i.e temperature difference divided by distance) between the shallowest (0.5 m) and deepest (11.5 m) temperature sensors has been calculated from the high resolution data.

Wave-theory (see Smith and Sinclair, 1972) has been used to estimate the depth of the wave-mixed layer, indicative of the depth to which wave mixing may impact the lake bed. The wave-mixed layer depends on both wind speed and ‘effective fetch’, estimated as a weighted averaged of fetches from a small directional segment. Effective fetch was calculated in ESRI ArcMap using the USGS Wind Fetch Model (Finlayson, 2005; Rohweder *et al.*, 2008) and wave-mixed depth then calculated following Spears & Jones (2010) as detailed by Mackay *et al.* (2011).

Average current speeds in the lake may be estimated from the wind speed, following Smith (1979), using the assumptions that the surface current will be a proportion of the wind speed, there will be an exponential decay in this current through the water column, and to conserve mass, a return current equally spread throughout the depth.

7.3 Temperature & stratification

The annual surface water temperature of the lake has risen since 1957 in a statistically significant manner at an average rate of 0.27 ^{o}C per decade (Fig. 7.1). In contrast the temperature at 14.5 m in the lake has remained unchanged on average, with a result that the volume-averaged temperature of the lake has risen in a statistically significant manner, but at a slower rate, 0.17 ^{o}C per decade, than the surface temperature. Despite this secular change in temperatures it is worth noting that the interannual variability in both the annually averaged surface and bottom water temperature was somewhat larger than the yearly trend (Fig. 7.1).

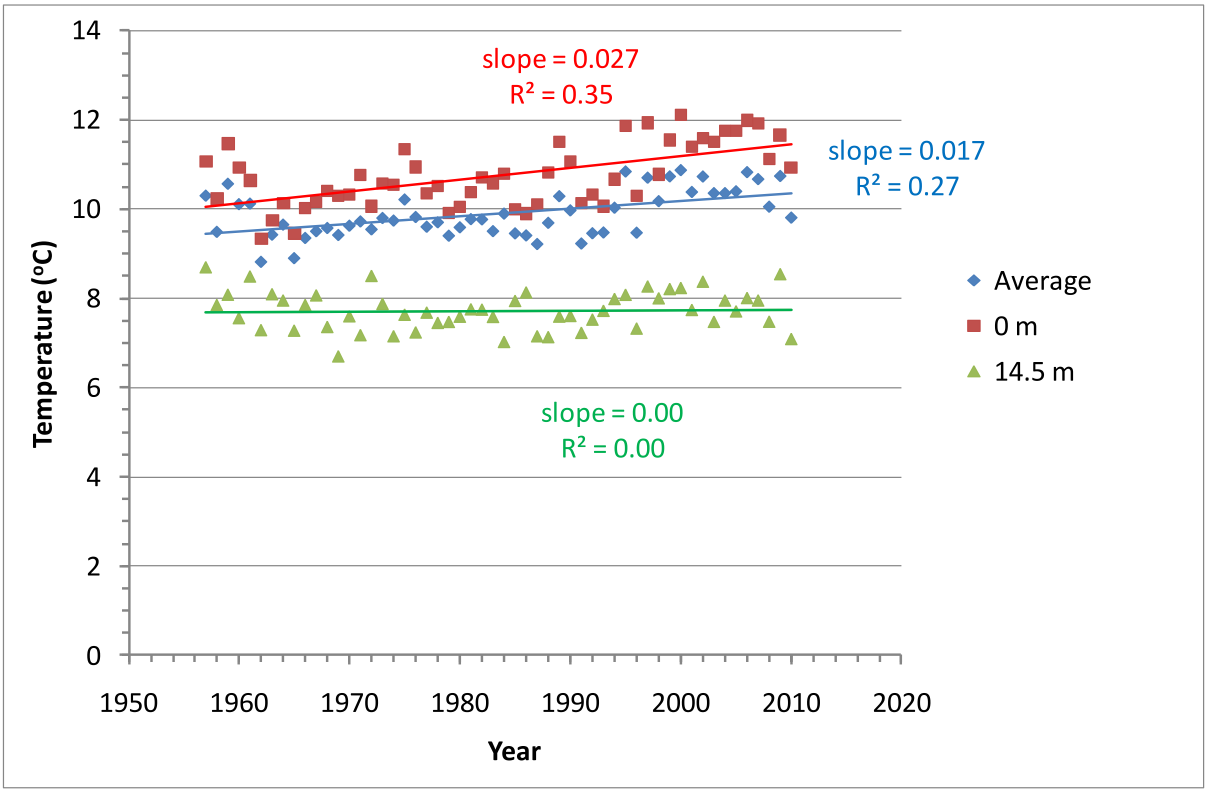

Figure 7.1. Annually averaged surface (red squares), average (blue diamonds) and bottom (green triangles) temperatures for Esthwaite Water from 1957 to 2010.

As the surface temperatures have been slowly but steadily rising, whilst the bottom temperatures have remained largely unchanged there has been an associated statistically significant increase in the strength of stratification in the lake (Fig. 7.2). The increase in annually averaged top-minus-bottom temperature has been 0.25 °C per decade. Similarly, the annually averaged mixed layer depth has also been shallowing at a statistically significant rate of 0.29 m per decade. Again interannual variability was larger than the yearly trend (Fig. 7.2). It should be noted that these results are annual averages and therefore include several months of the year when the lake is isothermal with a mixed depth of 14.5 m and a top-minus-bottom temperature difference of 0 °C.

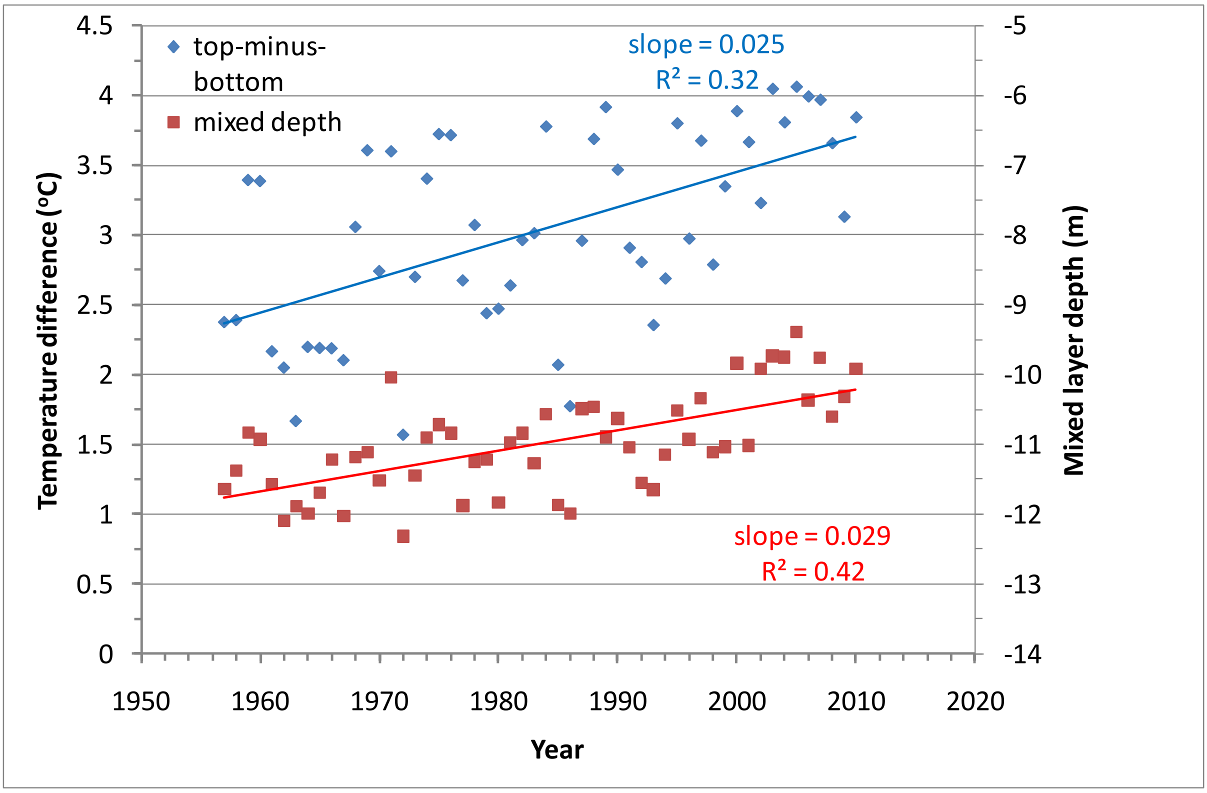

Figure 7.2. Annually averaged temperature difference between the top and bottom of Esthwaite Water (blue diamonds and left hand axis) and annually averaged mixed layer depth of Esthwaite Water (red squares and right hand axis) from 1957 to 2010.

The interpolated daily grid of temperature data may be used for estimating when stratification starts and ends each year and the corresponding length of summer stratification (Fig. 7.3). Whilst there was always a period of continuous stratification in the summer, this was occasionally preceded or followed by shorter periods of temporary stratification following particularly clement weather in the spring or autumn. As these occasional periods of stratification may occur at almost any time of year we have focussed solely on the period of continuous summer stratification. Since 1957, this summer stratification has been starting earlier, at a statistically significant average rate of nearly seven days per decade and finishing later at a statistically significant average rate of over four days per decade. In total summer stratification now lasts about 50 days longer than it did in the late 1950s, typically starting at the beginning of April and overturning towards the end of October (Fig. 7.3).

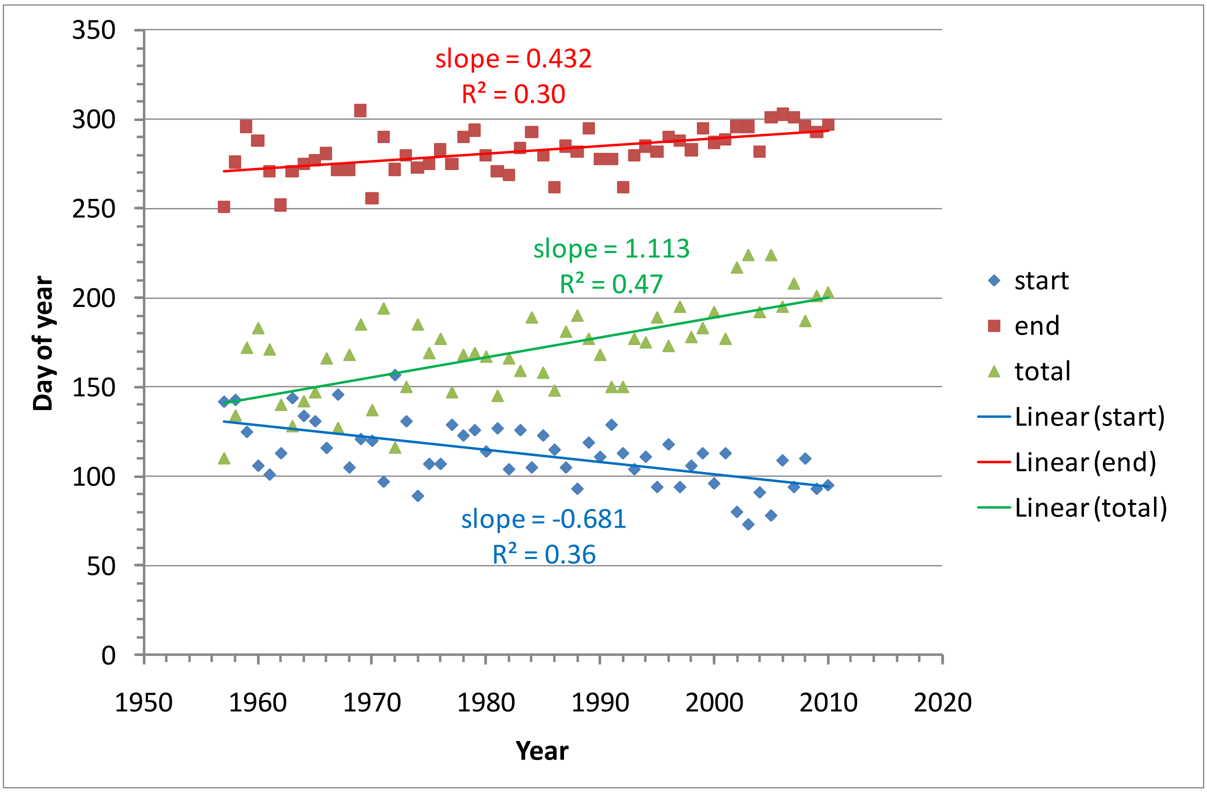

Figure 7.3. Day of start of continuous stratification (blue diamonds), end of continuous stratification (red squares) and total number of continuously stratified days (green triangles) for Esthwaite Water from 1957 to 2010.

Average daily surface and bottom temperatures from the 1960s' decade and the 2000s' decade (Fig. 7.4) are noticeably different. Surface temperatures in the 2000s were greater than those in the 1960s throughout the year, with differences approaching 3 °C at times. In contrast, though bottom temperatures in the 2000s were substantially warmer in the winter during isothermal conditions than those in the 1960s, they were usually a little cooler in the summer (Fig. 7.4).

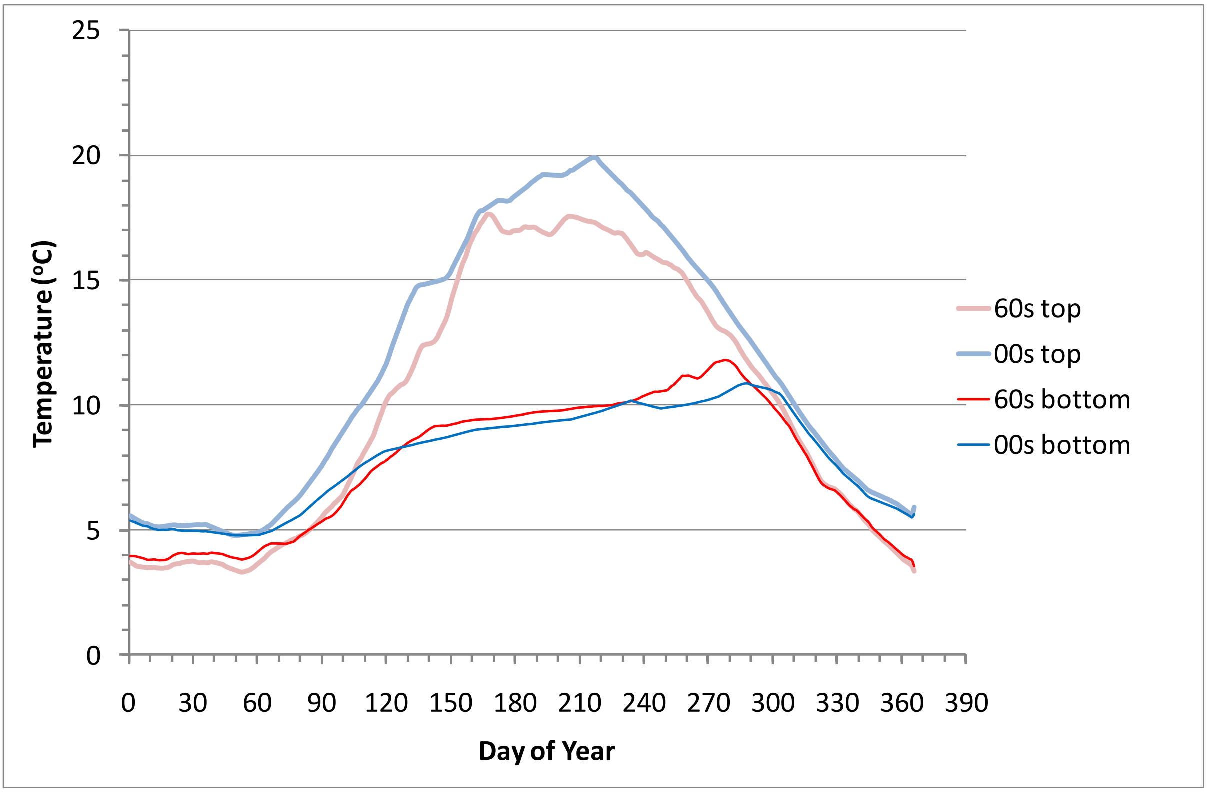

Figure 7.4. Average daily surface (thick lines) temperature and bottom (thin lines) temperature for the 1960s (red lines) and 2000s (blue lines) for Esthwaite Water.

A much more detailed picture of typical modern conditions emerges from the high resolution data from the automatic monitoring buoy. Data from 2006 are used here to show a typical annual pattern in the recent thermal structure of the lake. Daily averaged temperatures from the 12 different depths showed isothermal conditions throughout the winter and early spring with occasional signs of reverse stratification when the surface temperature fell below 4 °C (Fig. 7.5). In about mid-April the greater heating of the surface waters than the bottom waters enabled stratification to develop. Although this separation in temperature between the shallow and deep water tended to increase through the summer and then decrease towards autumn it was extremely variable with noticeable periods of substantial reductions in stratification (Fig. 7.5). Final overturn took place towards the end of October and the lake remained isothermal for the remainder of the year.

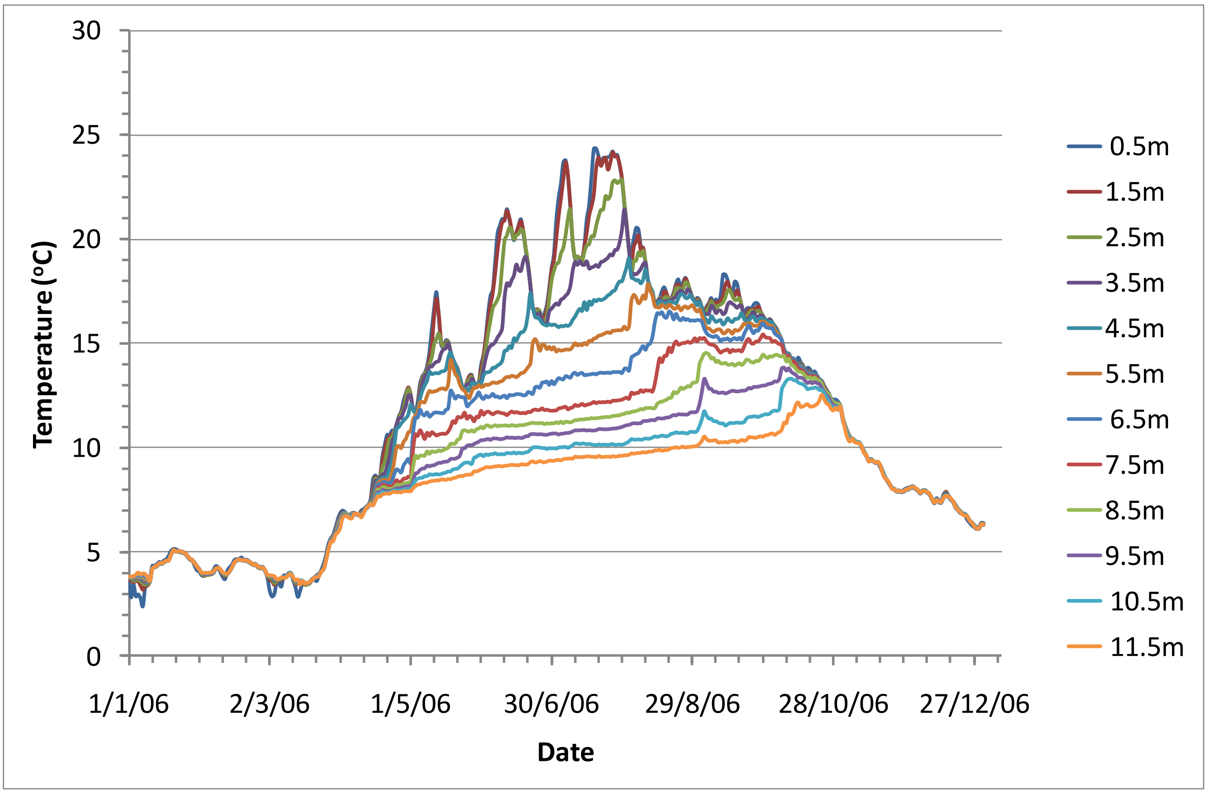

Figure 7.5. Daily average temperatures at 12 depths at metre intervals from 0.5 to 11.5 m for Esthwaite Water in 2006.

The pattern seen in the full thermal structure (Fig. 7.5) may be usefully parameterised by looking at the mixed layer depth and the temperature gradient between 0.5 and 11.5 m (Fig. 7.6). The mixed layer depth can be seen to have shoaled and then deepened dramatically at the start and end of the stratified period. It was typically about 4 m deep through the summer, but at times was as shallow as 1.5 m or as deep as 7.5 m, with a large degree of variability throughout the stratified period. The typical temperature gradient between 0.5 and 11.5 m was between 0.5 and 1.0 ^{o}C m^{-1}, although at times reached over 1.3 ^{o}C m^{-1} (Fig. 7.6).

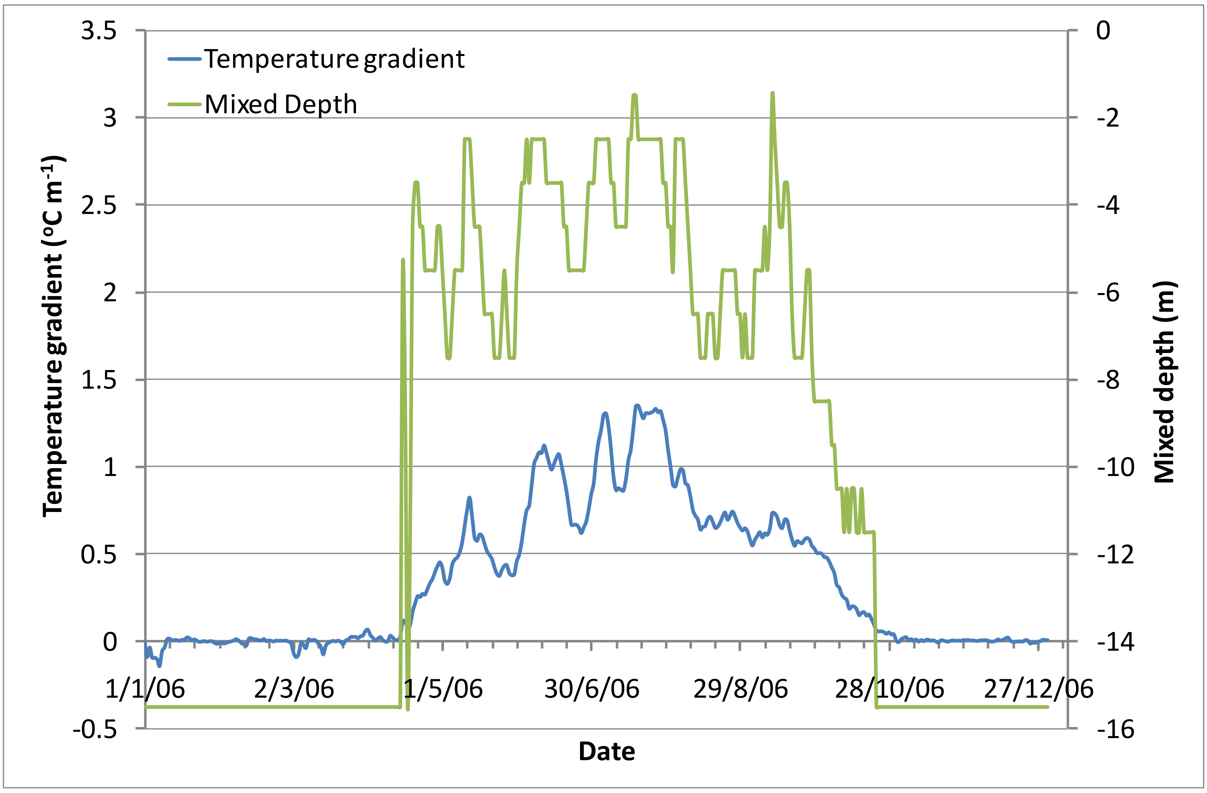

Figure 7.6. Daily average temperature gradient (°C m^{-1}) between 0.5 and 11.5 m (left hand axis, blue line) and daily averaged mixed layer depth (right hand axis, green line) for Esthwaite Water in 2006.

7.4 Average meteorological conditions from 2004 to 2009

Daily averaged meteorological parameters measured on the buoy were all highly variable (Fig. 7.7). The average daily wind speed from 2004 to 2009 was about 2.3 m s^{-1}. There was little systematic seasonal pattern with summer wind speeds only being a little less than winter ones. The standard deviation in daily averaged wind speed over the six years was typically comparable to the actual wind speed (Fig. 7.7a). In contrast there was a clear and expected seasonal cycle in air temperature, varying from, on average, less than 5 °C in the winter to more than 15 °C in the summer, but with much day-to-day variation (Fig. 7.7b). The relative humidity was continuously high, averaging approximately 85 %, and virtually never dropping below 60 %, with late spring humidities typically being a little less than winter ones (Fig. 7.7c).

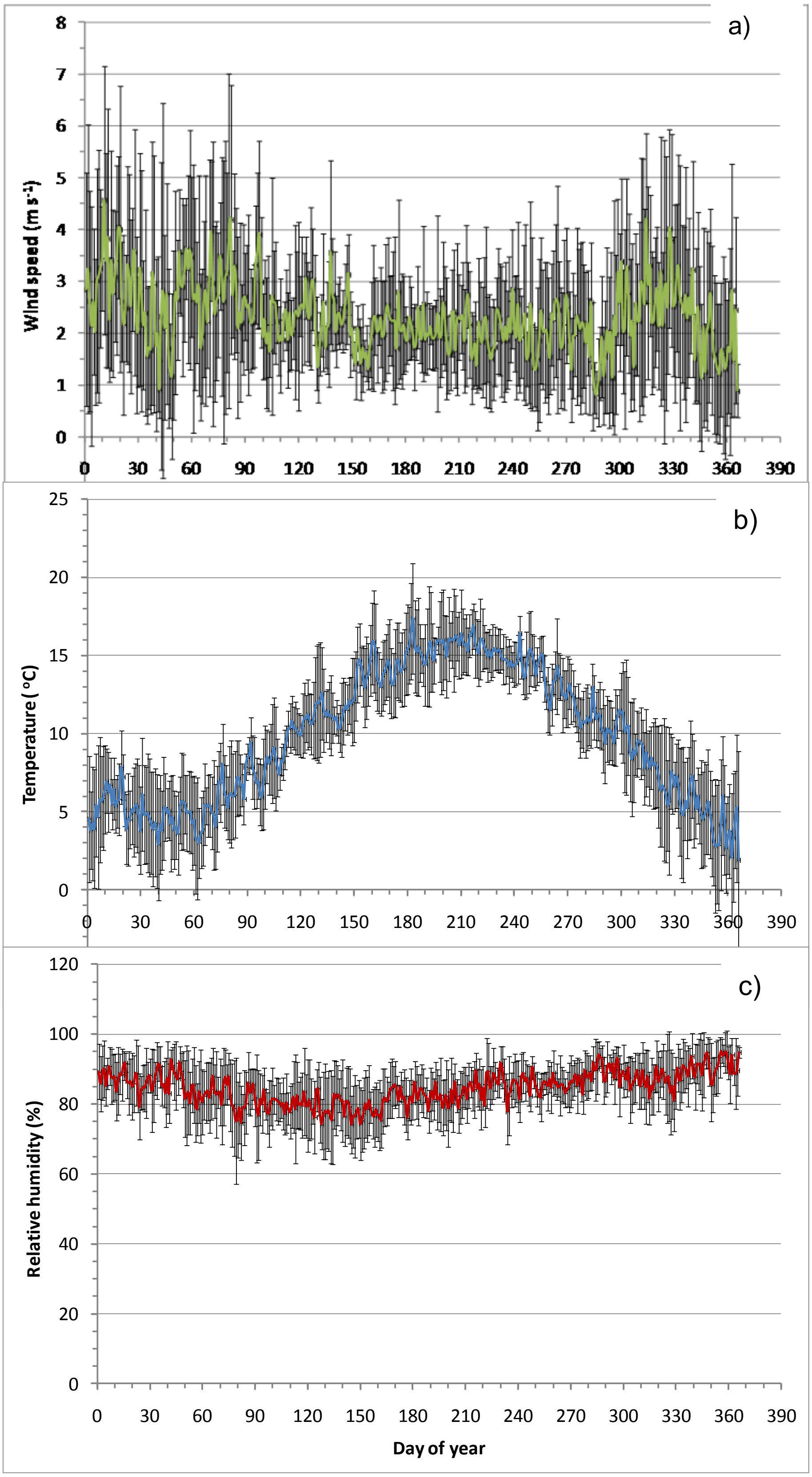

Figure 7.7. Daily averages from 2004 to 2009 and standard deviation over the six years of a) wind speed (m s^{-1}), b) temperature (^{o}C), and c) relative humidity (%).

By examining the hourly data over the six year period the diel cycle in meteorological parameters may be diagnosed (Fig. 7.8). It is clear that wind speed, air temperature and relative humidity all changed systematically through the day. The most interesting change was possibly that in the wind speed which was characterised by late afternoon wind speeds being more than 50 % greater than night-time wind speeds. Temperatures on average rose by about 3 ^{o}C from the early morning to the mid-afternoon, while relative humidities correspondingly dropped by about 15 %.

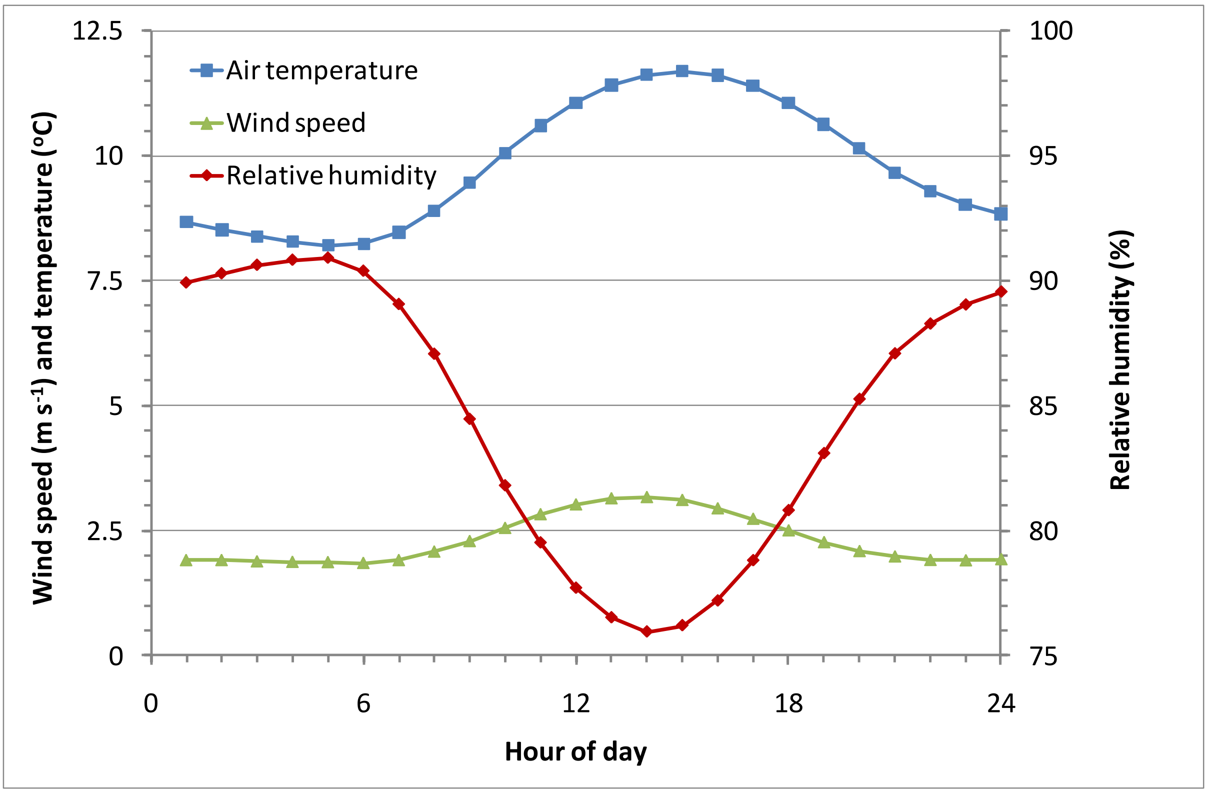

Figure 7.8. Hourly averages from 2004 to 2009 of wind speed (left hand axis, green triangles), air temperature (left hand axis, blue squares), and relative humidity (right hand axis, red diamonds).

7.5 Water currents in Esthwaite Water

Indicative average wind-induced currents in the lake would be a little more than 5 cm s^{-1} in the surface of the lake, but less than 1 cm s^{-1} in the deeper waters (Fig. 7.9). The negative values at depth in Figure 7.9 indicate water flowing in the opposite direction from that at the surface.

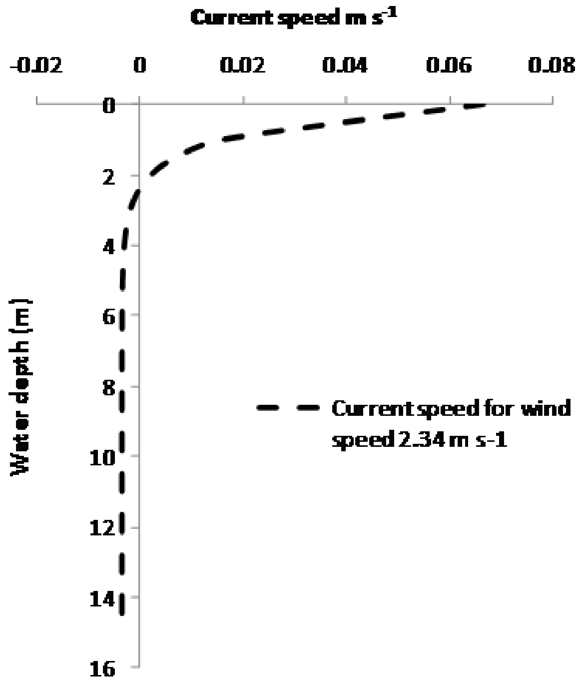

Figure 7.9. Indicative wind-induced current speeds calculated from average wind speeds for Esthwaite Water.

Wind direction is particularly relevant to the lake due to its elongated morphology, and showed an association with wind speed (Fig. 7.10). The prevailing winds came from a southerly to south-westerly direction. A much smaller proportion came from the north west although these were typically a little stronger. The lake appeared to be sheltered from easterly winds (Fig. 7.10).

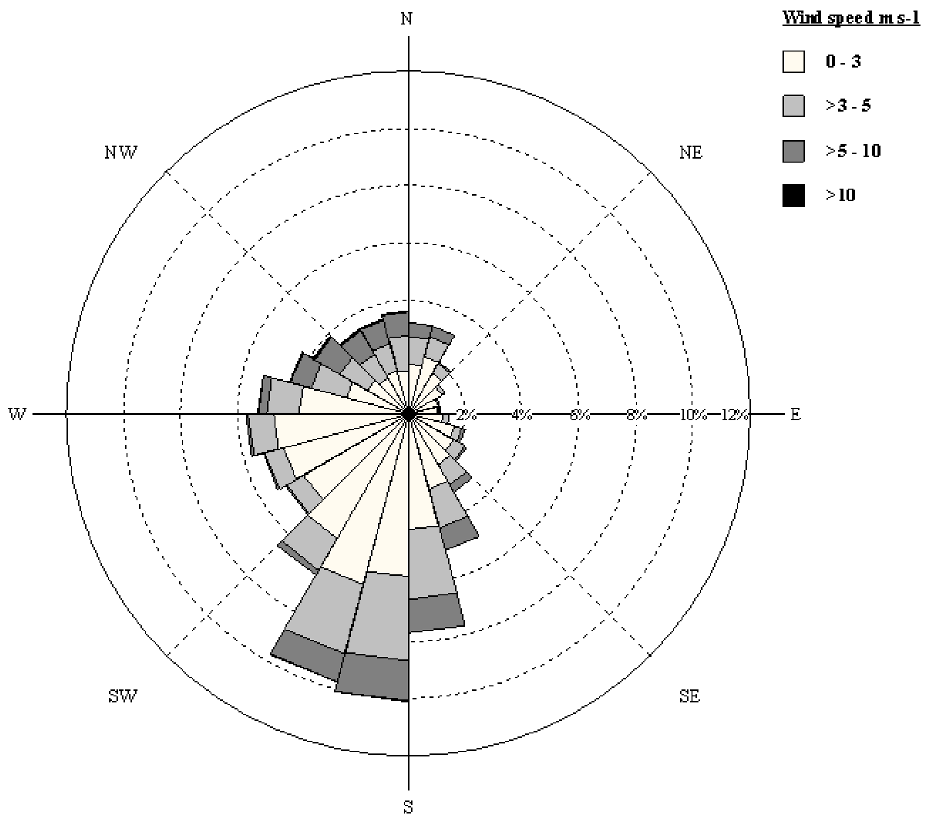

Figure 7.10. Hourly wind rose for Esthwaite Water from 2005 to 2009.

The combination of wind speed and wind direction data allowed wave-mixed depths to be calculated, both from the average wind speeds for 2005 to 2009 (Fig. 7.11a) and the maximum wind speeds over this period (Fig. 7.11b). As the prevailing winds were southwesterlies, wave-mixed depths were usually greater in the north east part of the lake, but were only about half a metre depth or less during average winds, and less than 2 m depth even in the strongest winds. It is of particular interest to note that little of the lake bed would be impacted by the wave-mixed depths generated by either the average or maximum wind speeds (Fig. 7.12).

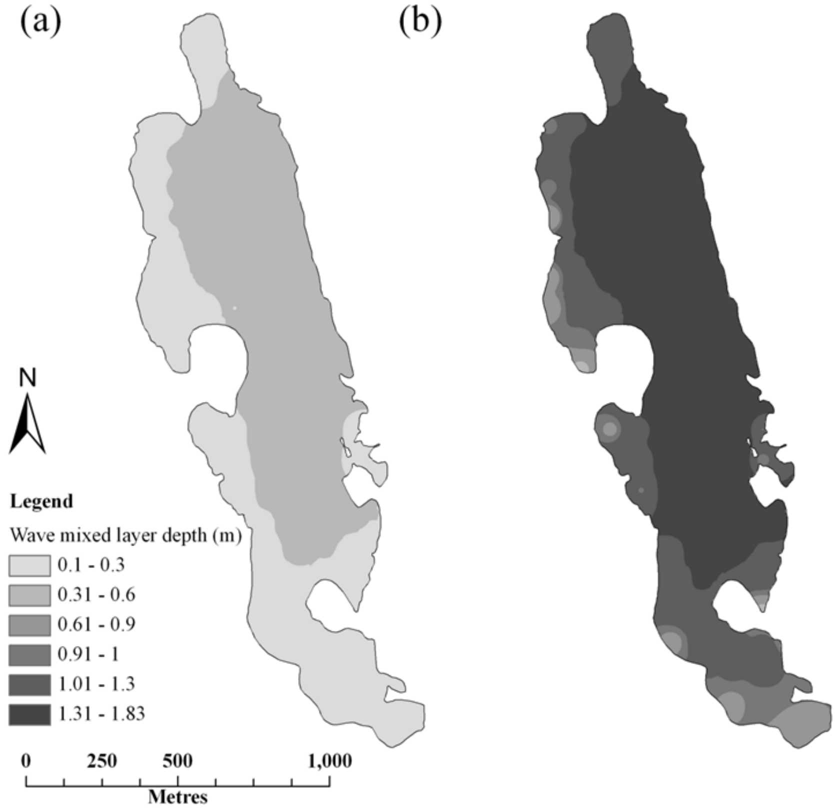

Figure 7.11. Wave-mixing layer for Esthwaite Water from 2005 to 2009 for a) average wind speeds, and b) maximum wind speeds.

Figure 7.12. Area of lake bed impacted by average and maximum wave-mixing

7.6 Heat budget

There was a recognisable seasonal cycle in the average daily heat flux over the six year period, but there was a huge daily and interannual variation in this total heat flux (Fig. 7.13a). This great variation was a result of the total heat flux being the sum of many component parts, representing both heat fluxes out of the lake (typically upward long-wave radiation, latent and sensible heating) and heat fluxes entering the lake (typically short-wave and atmospheric long-wave radiation). It is worth noting that the radiative fluxes were each much larger individually than the total heat flux (Fig. 7.13b). Although usually smaller than the radiative fluxes, the two turbulent fluxes are important as they can both be either warming or cooling fluxes depending on the air-water temperature difference. As such they act as a feedback mechanism, cooling the lake if it becomes warmer than the overlying air and warming the lake if it becomes cooler than the air. By monthly averaging and combining the fluxes into net short-wave, net long-wave, and turbulent parts, the seasonal cycle becomes more evident. Heat can be seen to be leaving the lake at the start of the year, but starts entering the lake around March, reaching its maximum in April-May and then starts leaving the lake again in September (Fig. 7.14). As the total heat flux is the sum of its constituent parts the seasonal cycle was not necessarily the same as any of the individual heat fluxes. In particular the seasonal cycle of total heat flux was typically different from that of the short-wave radiation.

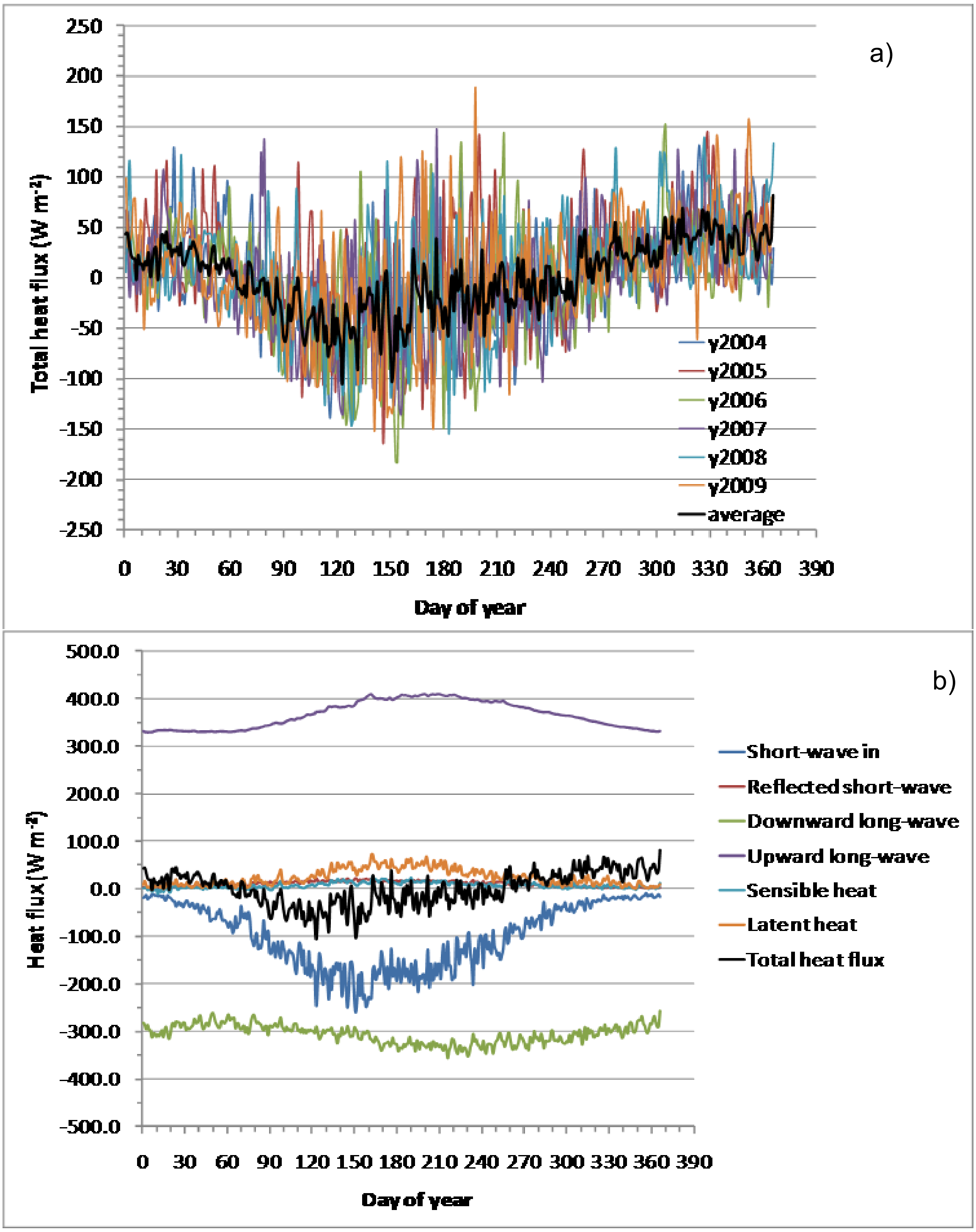

Figure 7.13. a) Total daily heat flux (W m^{-2}) for Esthwaite Water for each year 2004 to 2009 and for the daily average (black line) over the six years, and b) daily average individual heat fluxes (W m^{-2}) for 2004 to 2009.

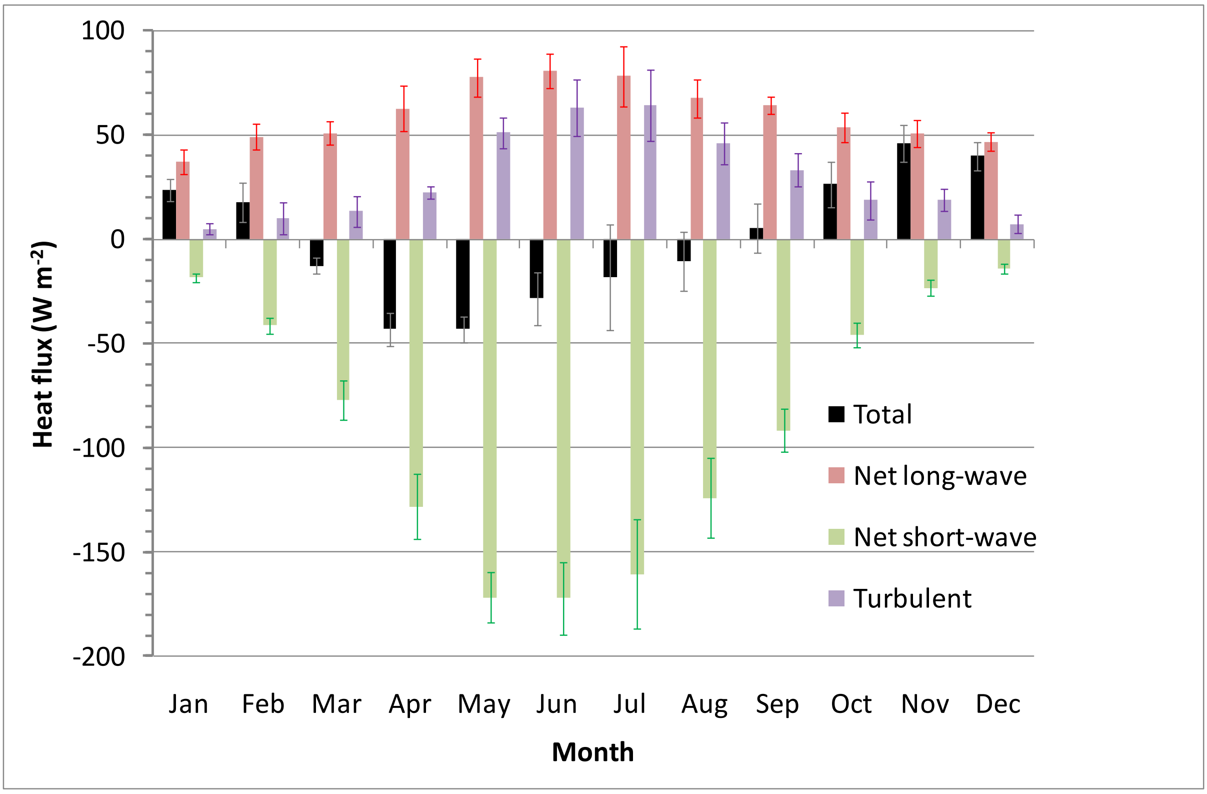

Figure 7.14. Monthly average heat fluxes from 2004 to 2009 and yearly standard deviations.

7.7 Light attenuation

The simplest and most widespread measure of light attenuation in lakes is that measured by the Secchi disc. This is a white 30 cm diameter disc that is lowered down the water column. The Secchi depth is depth between the depths where it disappears and appears. Regular Secchi depths are available from Esthwaite Water since the early 1970s (Fig. 7.15a) and the average Secchi depth over the period of record is 2.64 m. There has been a significant decrease (i.e. deterioration in light climate) over the 38 years analysed (long-term correlation -0.62, $P<0.001$) with a step-change reduction around 1987 (Fig. 7.16b). The monthly pattern of change is for relatively clear water in May (mean 3.8 m) representing a 'clear water phase' between the spring diatom bloom and summer bloom of cyanobacteria and other species (Fig. 7.16c). On average, August and September had the poorest water clarity with mean Secchi depths of around 1.6 m. The months with the greatest decline in Secchi depth are May, June, July and October.

Secchi depth gives a convenient measure of water clarity but is not directly quantitatively linked to light attenuation or spectral attenuation (i.e. the differential loss of different wavelengths of light). Extensive measurements of light attenuation in different spectral bands were made by Talling (e.g. Talling 1971). More recently, the attenuation coefficient for photosynthetically active radiation (PAR, 400 – 700 nm) has been measured in parallel with Secchi depth. Fig. 7.16 shows the relationship between these two variables and highlights that while they are related, they are not measuring exactly the same thing. The range of attenuation values measured in Esthwaite Water equate to a depth at which light reaches 1% of the subsurface (the bottom of the photic zone) of between 2.9 and 9.9 m with a mean of about 5 m. This is slightly shallower than a mean computed from measurements made by Pearsall in August 1915 with an attenuation of 0.71 m^{-1} that is equivalent to a 1% depth of 6.5 m (Pearsall 1917), i.e. deeper than the current mean. It is hard to infer whether or not there has been a real increase in light attenuation in the intervening years since the value in 1915 is within the recently-measured range and the methods of measurement differ.

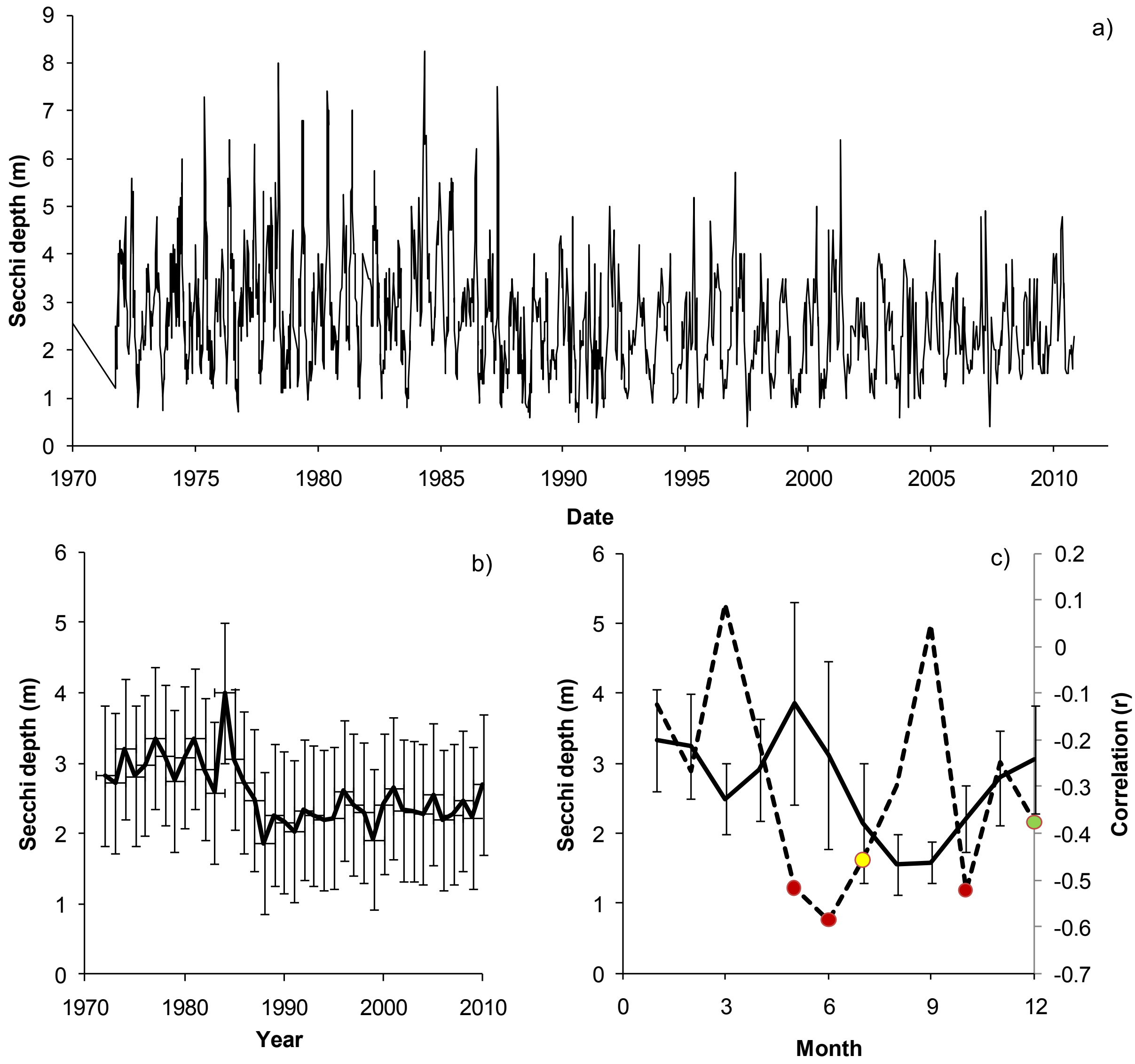

Figure 7.15. Secchi depth measurements in Esthwaite Water. a) long-term Secchi depth record from 1972 to 2010; b) annual mean (error bars show standard deviation); c) monthly mean (error bars show standard deviation) plus long-term correlation of monthly change (dashed line), green circle P<0.05, yellow circle P<0.01, red circle P<0.001.

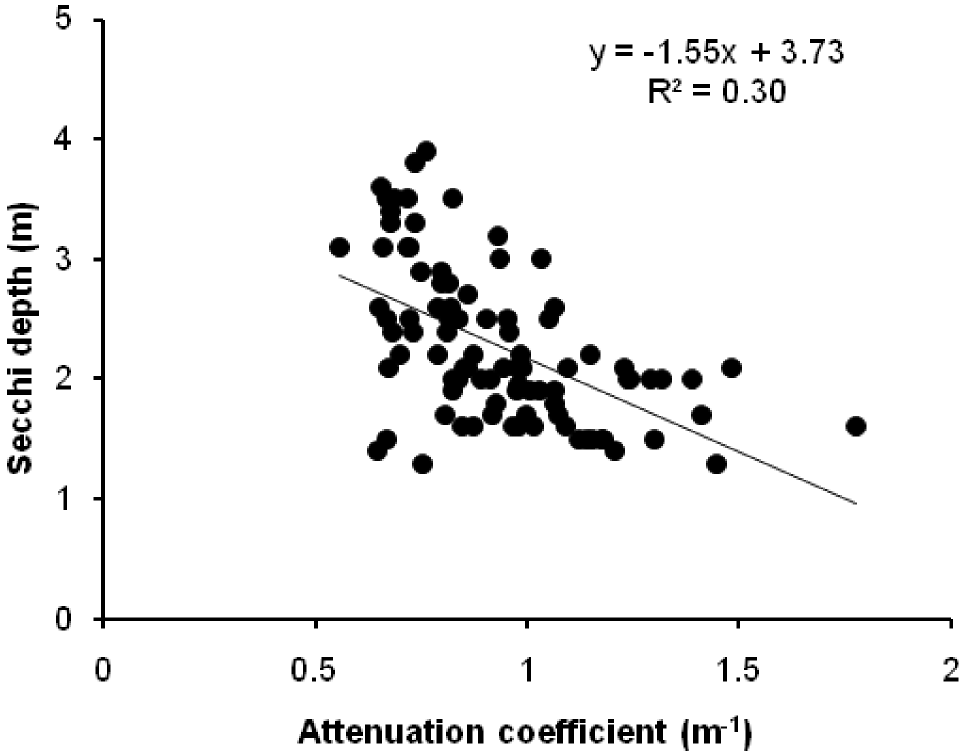

Figure 7.16. Comparison between Secchi depth and attenuation coefficient for photosynthetically active radiation (PAR, 400 – 700 nm) in Esthwaite Water between 2006 and 2009. The equation is given for the linear regression between the two measures.

Measurements of spectral attenuation show least rapid attenuation between 500 and 600 nm and greater attenuation in the blue region from 400 to 500 nm. The attenuation is much greater in the UV-A region from 320 to 400 nm and even more in the UV-B for wavelengths up to 320 nm (Fig. 7.17). The rapid attenuation in the blue region is consistent with the relatively high concentration of dissolved organic carbon (DOC) measured in Esthwaite Water (Tipping *et al.* 1988), although concentrations are low on a national scale and much of the DOC may not be coloured (Section 8.8).

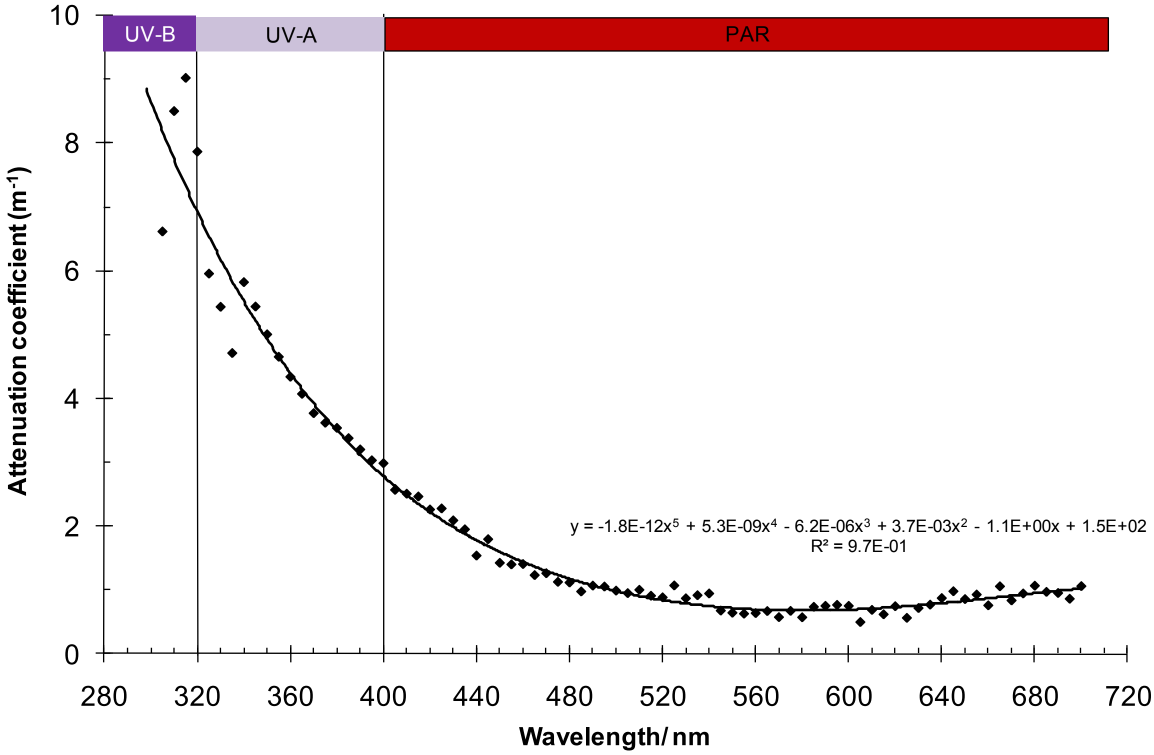

Figure 7.17. Spectral attenuation in Esthwaite Water on 21/06/1995. Redrawn from Olesen & Maberly (2000).

8. Water chemistry

8.1 Introduction

Esthwaite Water is one of the most ionically-rich lakes in the English Lake District, but on a national or global scale it is relatively dilute because it drains land which is slow to weather and yields relatively few ions. There is little base material in the catchment, apart from a thin band of Coniston limestone in the north of the catchment. The chemistry of Esthwaite Water is one of the best studied in the world. The earlier work is summarised in Talling & Heaney (1983), Heaney *et al.* (1986) and Talling (1999). Most of the water chemistry data analysed here has been obtained from an integrated 0 to 5 m water sample collected over the deepest point, with the exception of the long-term alkalinity and pH data which derive from a sub-surface sample.

8.2 Major cations & anions

Sodium and calcium are the major cations and bicarbonate, chloride and sulphate are the major anions in Esthwaite Water (Table 8.1). Over the last 35 years there has been a decrease in conductivity ($P<0.05$), and a very large reduction in the concentration of sulphate ($P<0.001$). Conductivity is strongly related to the total ionic concentration (Fig. 8.1) and using this relationship to subtract the effect of sulphate concentration on conductivity, the reduction in conductivity is largely caused by the reduction in sulphate concentration. This reduction has been caused by the large reduction in atmospheric sulphur deposition as a result of legislation to reduce acid-deposition: emissions of sulphur dioxide declined by 92% between 1970 and 2010 and deposition declined by 80% between 1986 and 2006 (ROTAP 2011). There has been a slight increase in alkalinity. Although the change is not quite statistically significant, about 70% may be explained by the reduction in sulphate deposition. Further information on the ion chemistry of Esthwaite Water can be found in Gorham *et al.* (1974), Carrick & Sutcliffe (1982), Sutcliffe *et al.* (1982), Sutcliffe (1998).

Table 8.1. The major ions in Esthwaite Water. Data derived from four measurements a year during in 'Lake Tours' apart from the data from 194-5 and 1974-8 which are derived from Carrick & Sutcliffe (1982). Concentrations of anions and cations as mequivalent m^{-3}.

	Cations				**Anions**				**Conduct-ivity (µS cm^{-1})**	**Ionic strength (mol m^{-3})**
Year	**Na^+**	**K^+**	**Ca^{2+}**	**Mg^{2+}**	**Cl^-**	**SO_4^{2-}**	**HCO_3^-**	**NO_3^-**		
1954-5	202	23	412	116	214	197	306	13		1.10
1974-8	249	25	526	123	282	231	386	31		1.37
1984	303	27	544	122	417	214	320	61	116.8	1.44
1991	325	34	554	129	459	180	405	42	124.9	1.49
1995	279	27	514	109	314	141	440	38	105.3	1.31
2000	254	23	478	110	317	120	425	28	102.8	1.23
2005	302	26	547	122	375	111	463	33	-	1.38
2010	267	22	467	95	294	98	431	25	92.6	1.18

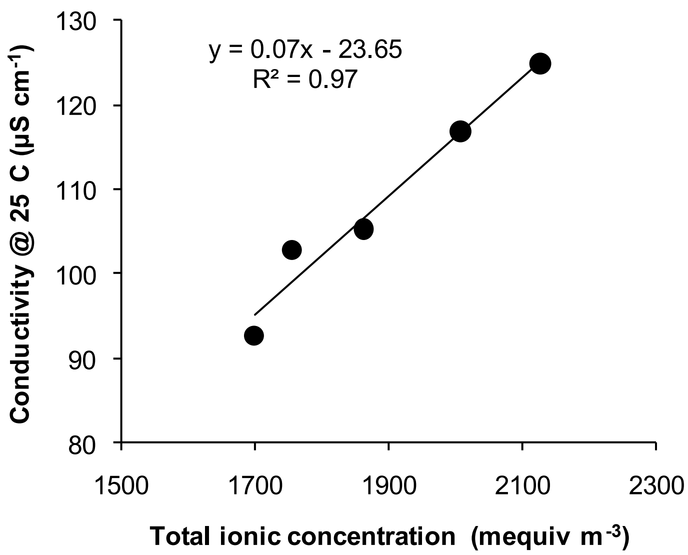

Figure 8.1. Relationship between conductivity and total ionic concentration in Esthwaite Water between 1974 and 2010. Data derived from Table 7.1.

8.3 Alkalinity & pH

Alkalinity, or more correctly acid-neutralising capacity, represents the sum of the anions of weak acids and controls the ability of a water to buffer change in pH (Stumm & Morgan 1996). In the

English Lakes, bicarbonate is the main form of alkalinity and of the major lakes, Esthwaite Water and Blelham Tarn have the highest alkalinity, although alkalinity is low from a UK and an international perspective since there is little base-generating material within the catchment. As was identified in the previous section, alkalinity (expressed as the concentration of bicarbonate in Table 8.1) has increased in Esthwaite Water. Based on weekly or fortnightly measurements, annual mean has increased statistically significantly (Fig. 8.2a; $P<0.001$). There is an indication in the data that alkalinity has stabilised since the start of the 2000s. There is a very regular seasonal pattern of alkalinity change, alkalinity is greatest at the end of the growing season in September and alkalinity and lowest in February and March, probably largely as a result of inflowing concentrations influenced by dilution by varying amounts of rainwater. The average alkalinity between 1984 and 2010 of 0.42 equiv m^{-3} places Esthwaite Water in the 'medium alkalinity' category within the Water Framework Directive (UKTAG 2008).

In productive lakes, pH can be highly variable. This results from variation in the supply and demand for inorganic carbon. Large crops of phytoplankton can remove inorganic carbon faster than it can be resupplied from reserves at depth, inflow from influent streams or exchange with the atmosphere across the air-water interface. Net carbon uptake in photosynthesis thus causes the concentration of total inorganic carbon (CT) to decrease causing pH to increase and the equilbrium between the different forms of inorganic carbon to shift away from CO_2 and towards HCO_3^- and CO_3^{2-} (Fig. 8.3). Conversely when respiration predominates (for example at night) or when input of CO_2 from depth or by transport across the air-water interface exceeds photosynthetic carbon uptake, the concentration of CT rises, pH falls and the equilibria between the different forms of inorganic carbon shift towards CO_2 and away from HCO_3^- and CO_3^{2-}.

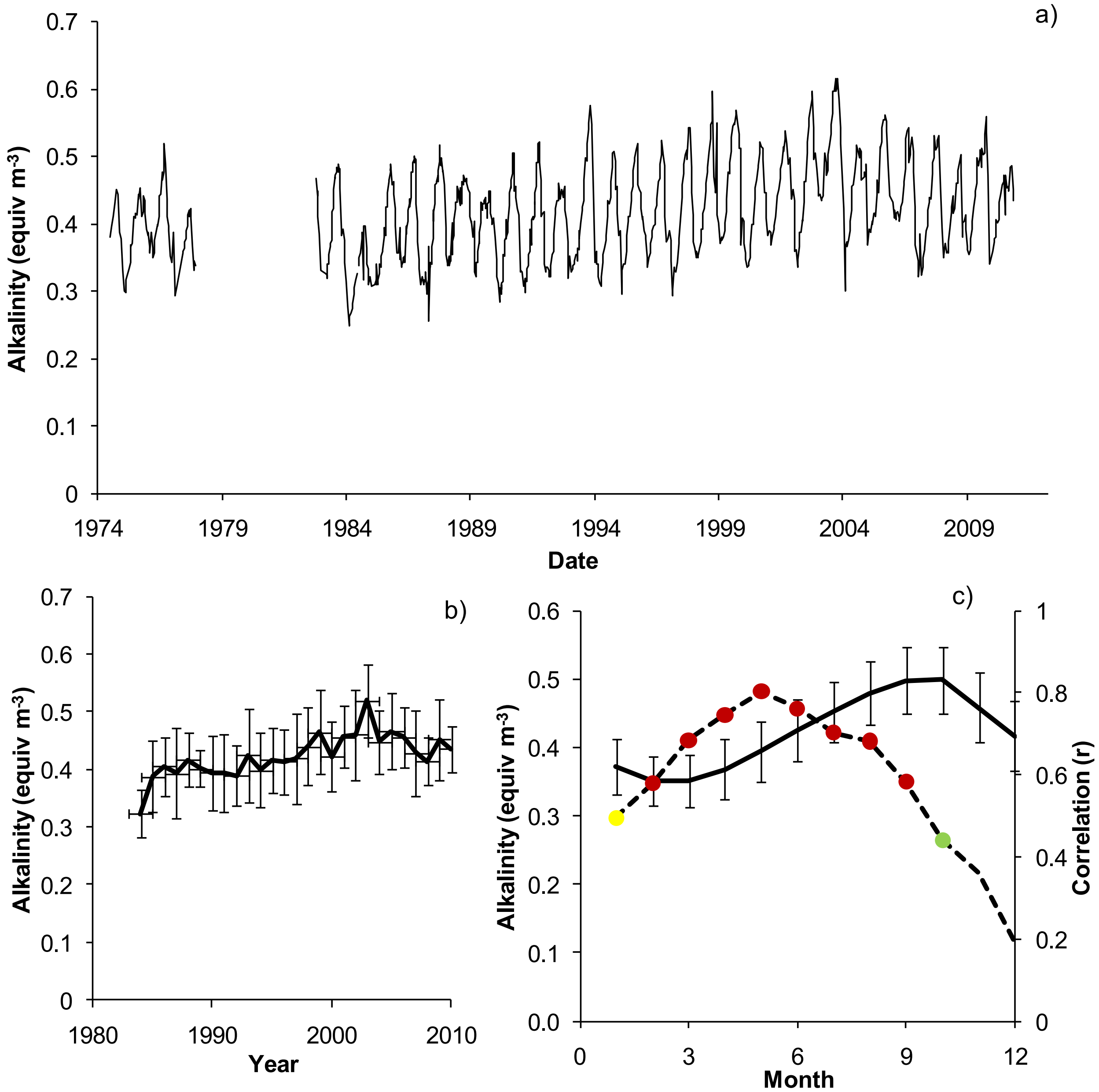

Figure 8.2. Alkalinity in Esthwaite Water. a) long-term alkalinity record from 1974 to 2010; b) annual mean (error bars show standard deviation); c) monthly mean (error bars show standard deviation) plus long-term correlation of monthly change (dashed line), green circle P<0.05, yellow circle P<0.01, red circle P<0.001.

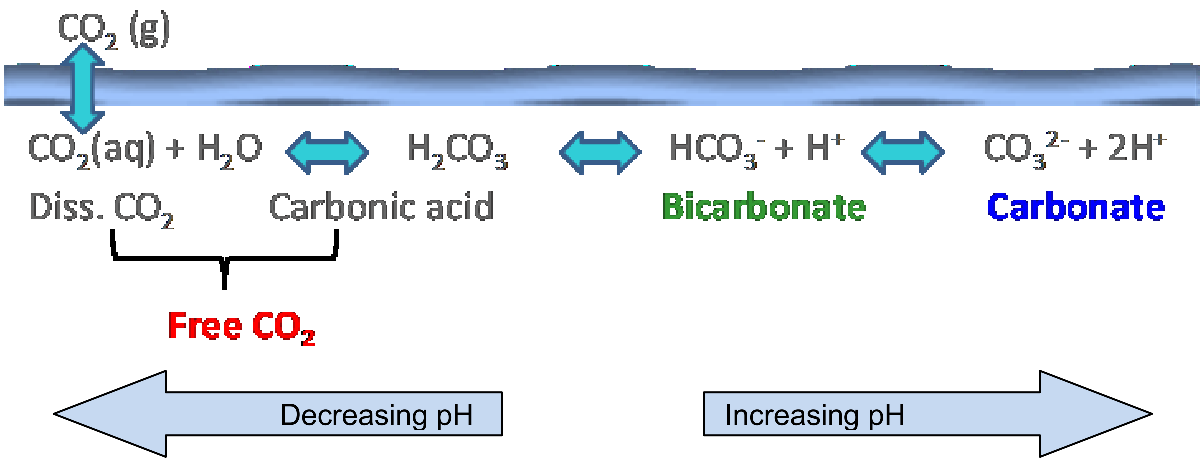

Figure 8.3. The carbonate system in freshwaters. Transfer across the air-water interface and equilibria between the different forms of inorganic carbon are shown by double-headed arrows.

The large crops of phytoplankton during summer stratification cause substantial drawdown of inorganic carbon and hence causes pH to increase (Fig. 8.4; Talling 1976; Maberly 1996). At the end of summer stratification there is a rapid reduction in pH as conditions become less favourable for phytoplankton photosynthesis and the CO_2 produced at depth by microbial respiration becomes entrained into the surface water. There is an indication that demand for inorganic carbon was lower in the summers of 2008, 2009 and to a lesser extent 2010, as summer pH values were relatively low. However, the long-term average pH has not changed significantly annually or in any of the study months (Fig. 8.4). This might be partly because episodes of high pH are often short-lived and tend to occur during calm, warm weather.

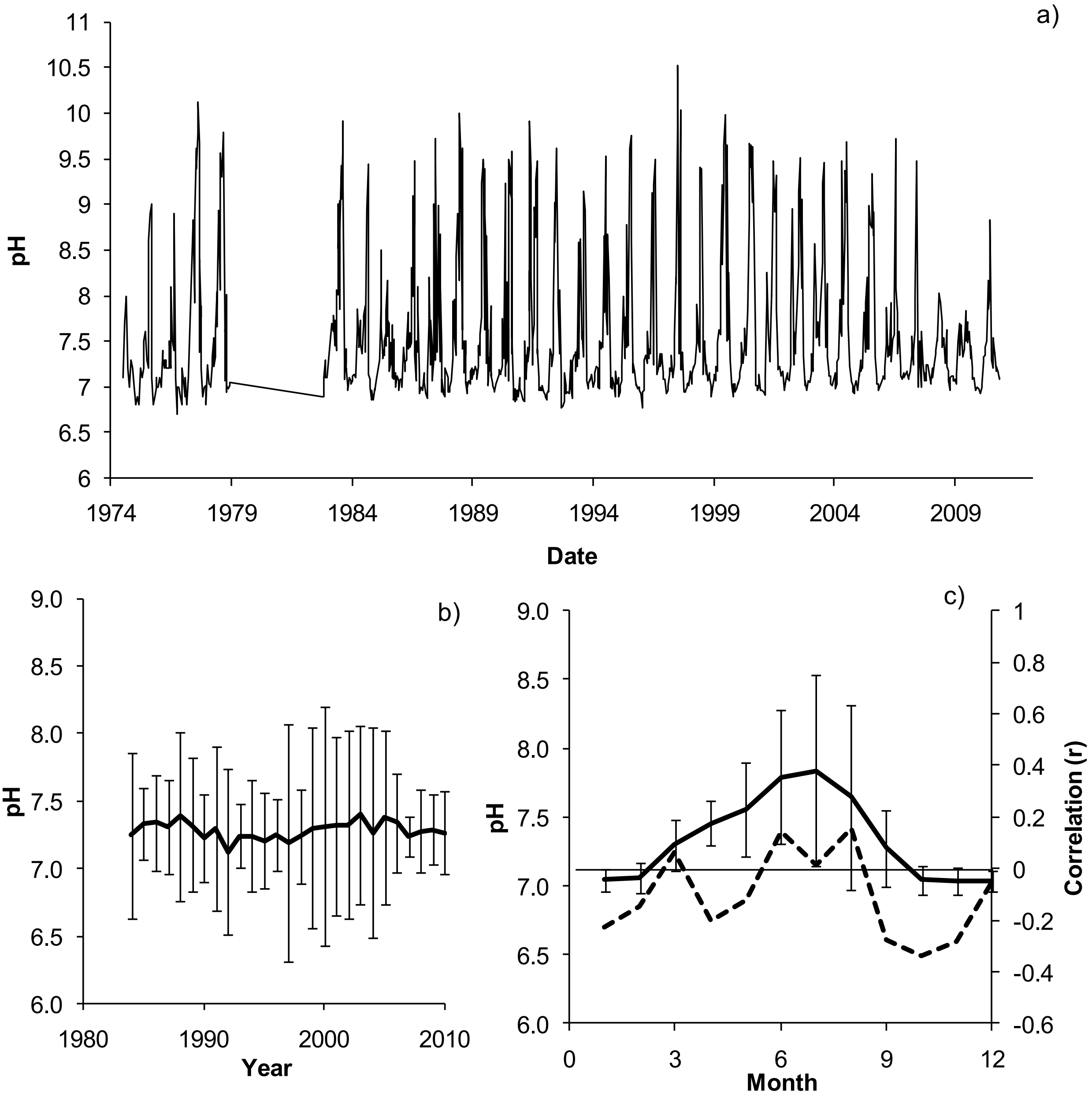

Figure 8.4. pH in Esthwaite Water. a) long-term pH record from 1974 to 2010; b) annual mean (error bars show standard deviation); c) monthly mean (error bars show standard deviation) plus long-term correlation of monthly change (dashed line), none of the correlations are statistically significant. Averages of pH were calculated geometrically because pH is a logarithmic number.

Because pH is so dynamic, a better picture of the variability is gained by high frequency measurements. Since the end of 1992, pH has been recorded every 15 minutes at the surface of Esthwaite Water (Maberly 2006). An example of this high resolution record, degraded to daily means, is shown in Fig. 8.5a. The true scale of the variability can be seen in Figure 8.5b.

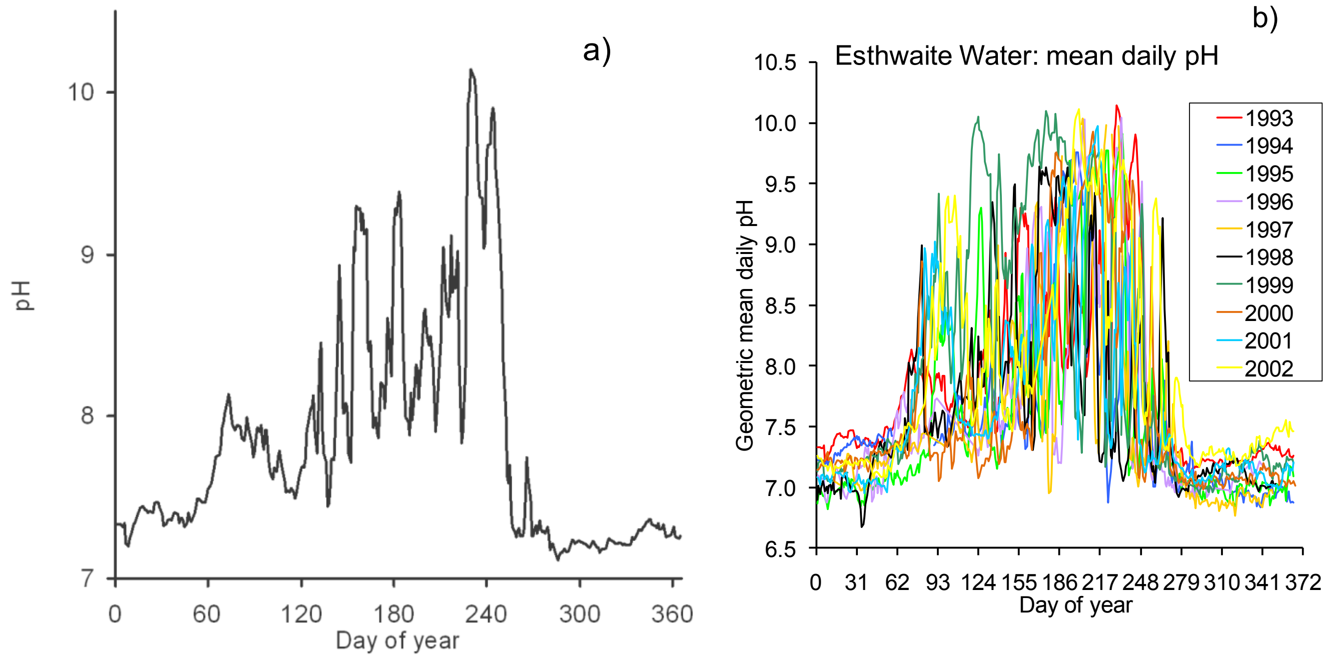

Figure 8.5. Daily change in pH in the surface of Esthwaite Water in; a) 1993 (Maberly 1996). and (b) from 1993 to 2002.

Diel changes in pH (not shown in the above figures above based on daily means) can be extensive. Most frequently they are 0.1 to 0.2 pH units per day, driven by changes in the balance between photosynthesis (pH increase) and respiration (pH decrease) but much larger changes of around 1 pH unit can occur during particularly active photosynthesis by dense populations of phytoplankton (Fig. 8.6a) and occasionally a large pH change can occur of over 2 pH units that is usually caused by a large drop in pH caused by an extremely large mixing event (Fig. 8.6b).

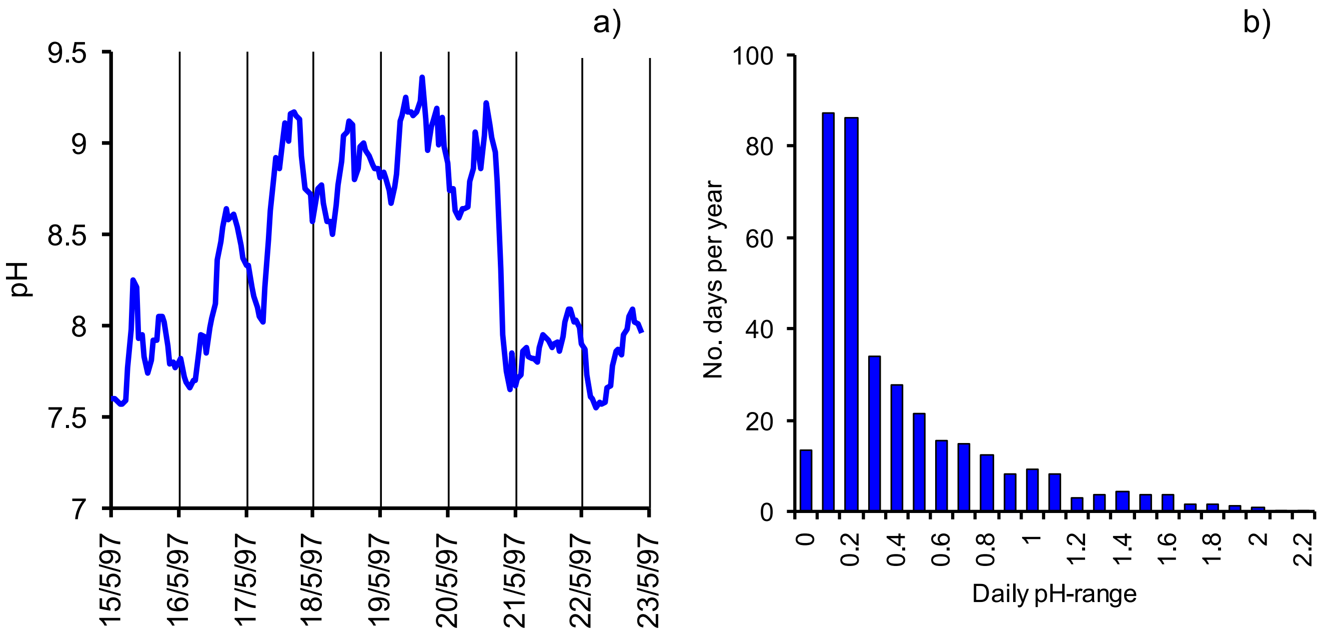

Figure 8.6. Example of pH change in Esthwaite Water over 8 days (a) and frequency of diel variation in pH (b).

8.4 CO_2 & inorganic carbon

The alkalinity is a major descriptor of the amount of inorganic carbon in a freshwater. However, the concentration of inorganic carbon, and in particular the form of inorganic carbon that constitutes the total is strongly dependent on pH (Figs 8.3, 8.7). Inorganic carbon comprises dissolved CO_2 and carbonic acid (together taken as free CO_2- or just CO_2 for short), bicarbonate and carbonate.

The 15-minute records of surface pH and temperature, plus weekly or fortnightly measurements of alkalinity can be used to calculate the concentration of CO_2. The example record in Fig. 8.8 shows that in the summer the concentration of CO_2 falls close to zero and the lake is then taking CO_2 up from the atmosphere. For much of the year however Esthwaite Water is losing CO_2 to the atmosphere, particularly immediately after the breakdown of stratification when CO_2 trapped in the hypolimnion is mixed into the whole water column. Esthwaite Water has been a source of CO_2 to the atmosphere in every year when measurements have been made (Fig. 8.9).

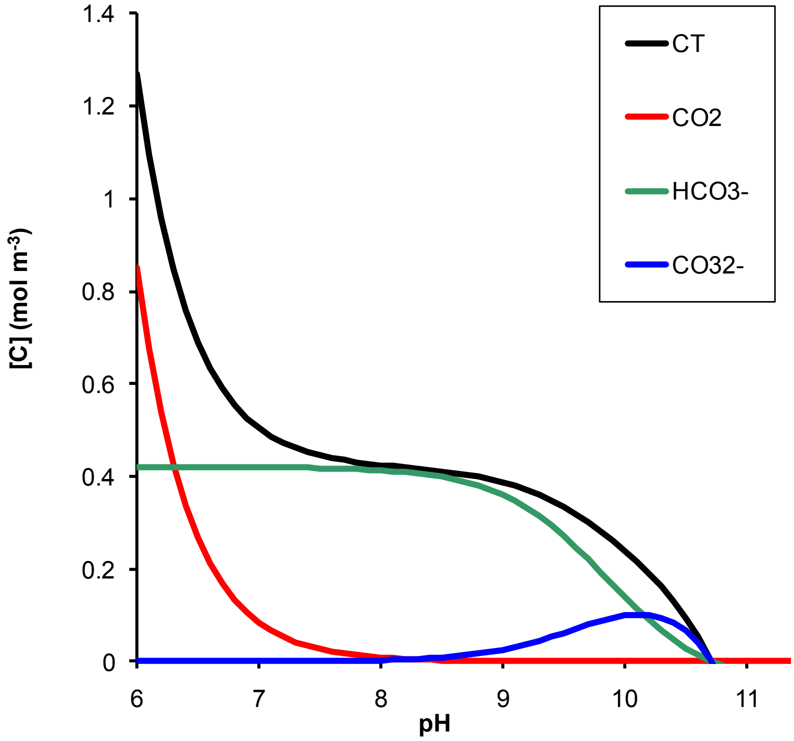

Figure 8.7. Carbon speciation in Esthwaite Water at 20°C, and alkalinity of 0.42 equiv m^{-3}. CT = total inorganic carbon, the sum of the three constituent forms of inorganic carbon.

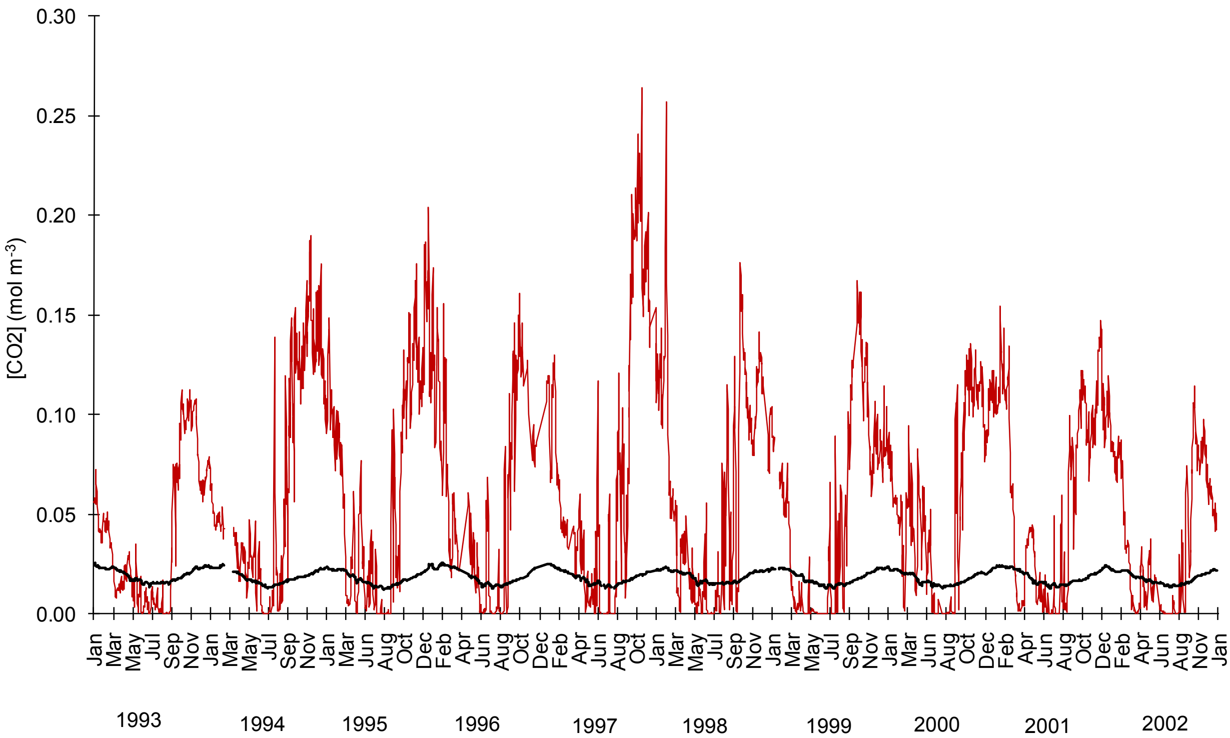

Figure 8.8. Calculated change in the concentration of CO_2 in the surface water of Esthwaite Water (red-line) based on automatic 15-minute readings (Maberly, 1996). The temperature-dependent air-equlbrium concentration is shown as a black line.

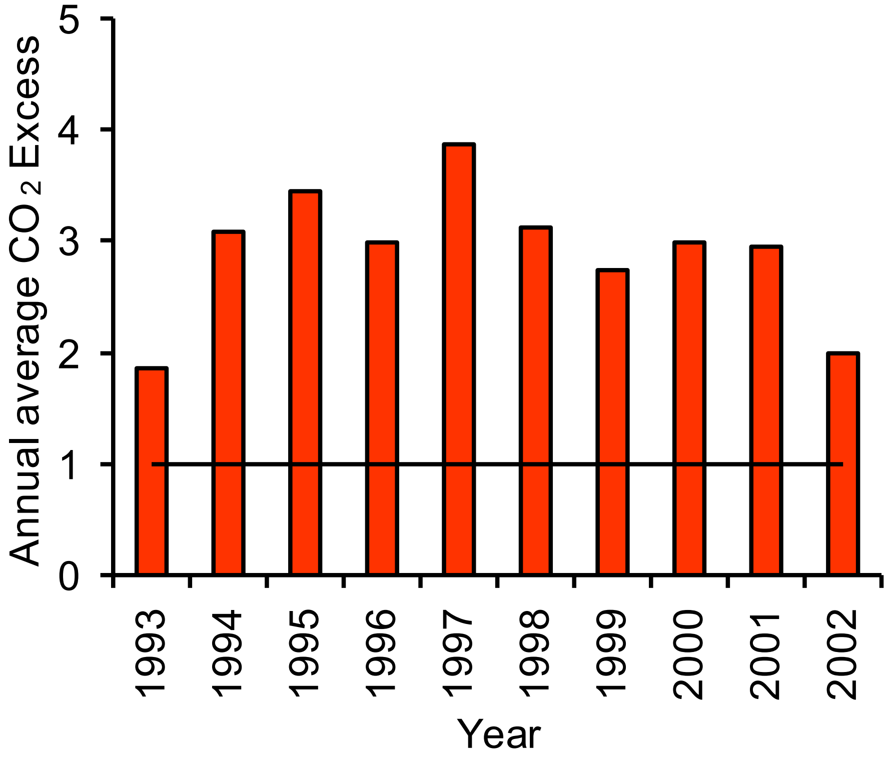

Figure 8.9. Annual estimates of CO_2-excess in Esthwaite Water over ten-years. The horizontal line signifies a balance between carbon-uptake and carbon-loss.

For most of the year bicarbonate is the dominant form of inorganic carbon in the surface of Esthwaite Water. However, during episodes of high summer pH the equilibria shift so that carbonate becomes increasingly important (Fig. 8.10). Although the carbonate solubility coefficient can be exceeded at times, there is no indication for episodes of calcite precipitation (Maberly 1996).

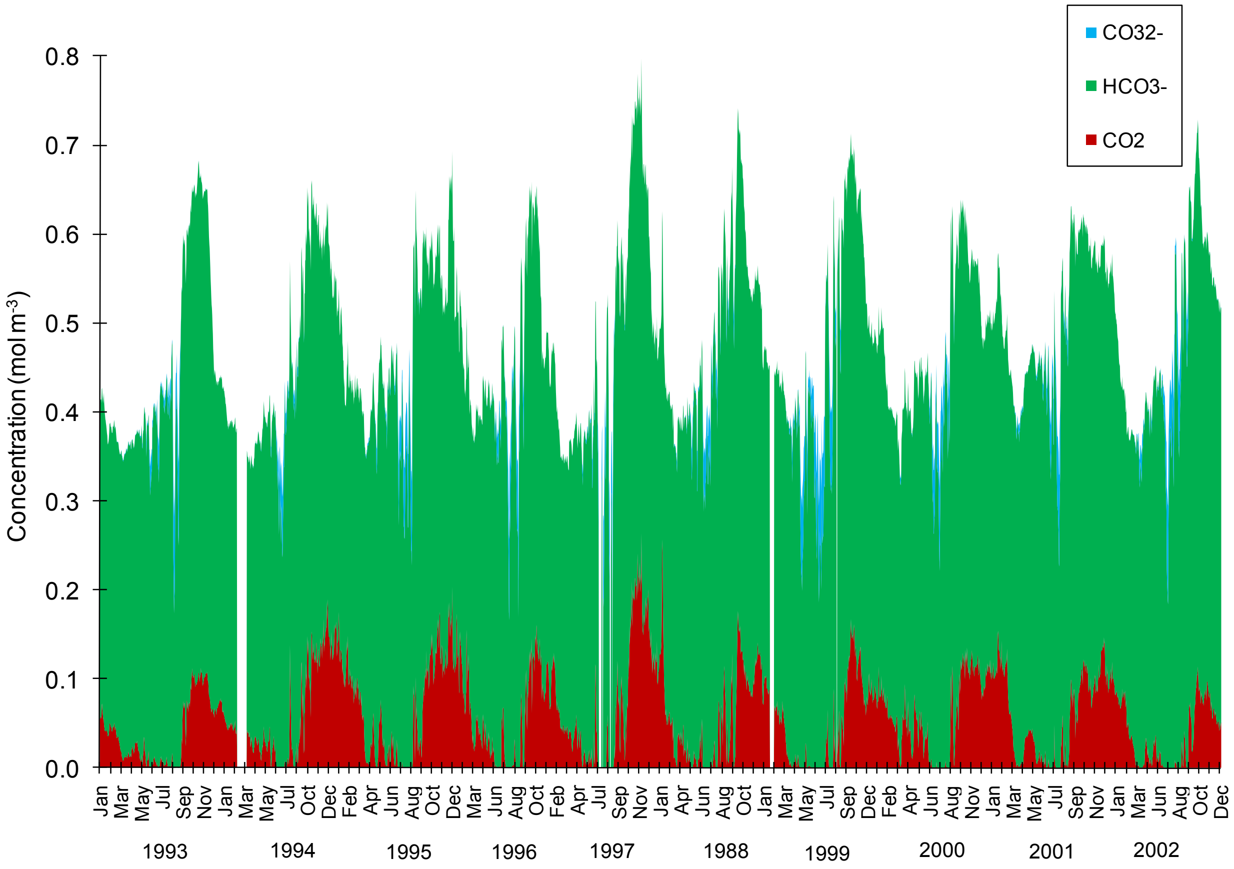

Figure 8.10. Calculated carbon speciation in the surface of Esthwaite Water based on automatic 15-minute readings (Maberly, 1996).

8.5 Phosphorus

In many larger lowland lakes, the element phosphorus (P) is in short supply and can limit phytoplankton productivity. Two forms of phosphorus are commonly distinguished. The first is clearest: total phosphorus is the total amount of this element in a given volume of water. This will include dissolved inorganic P, dissolved organic P, particulate P such as that associated with inorganic particles and the biota themselves, although this excludes larger life forms such as fish. Not all of these fractions (such as those bound to clay particles for example) are available as a resource to phytoplankton or other microbes. The second fraction is variously called Soluble Reactive Phosphorus (SRP), Dissolved Reactive Phosphorus (DRP), or Molybdate Reactive Phosphorus (MRP) and is defined operationally as the fraction that passes through a filter (pore size around 1 µm) and reacts with the reagent, molybdate used to measure phosphate. It is generally regarded as the fraction that is available to phytoplankton although in reality it is not necessarily just the concentration but also the rate of turnover in the 'microbial-loop' that will determine availability.

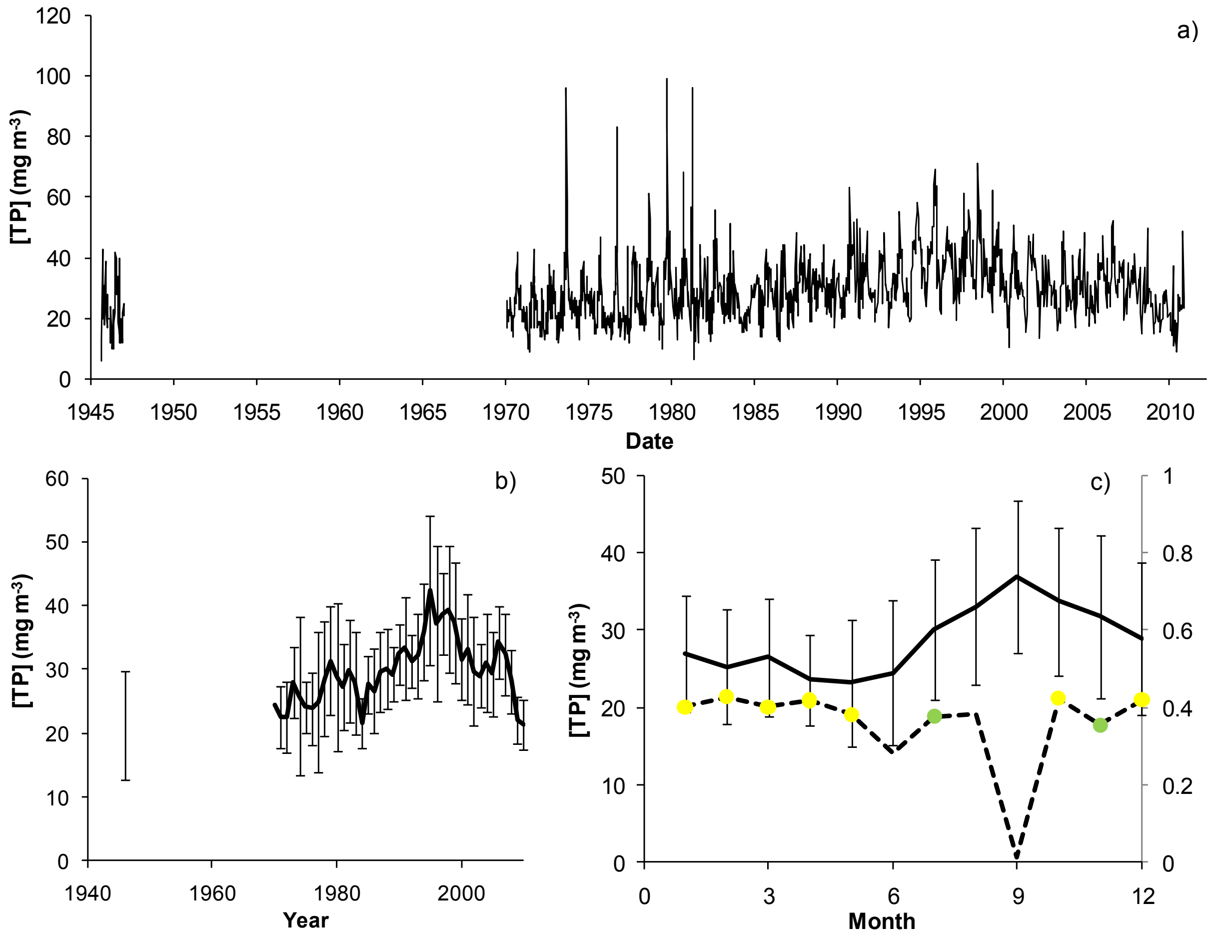

Figure 8.11. Concentration of total phosphorus (TP) in Esthwaite Water. a) long-term record from 1946 to 2010; b) annual mean (error bars show standard deviation); c) monthly mean (error bars show standard deviation) plus long-term correlation of monthly change (dashed line), green circle P<0.05, yellow circle P<0.01, red circle P<0.001.

Total phosphorus has been measured between 1945 and 1947 and in the only complete year of measurement, 1946, the concentration was about 21 mg m^{-3} (Fig. 8.11). TP measurements recommenced in 1970 by which time the annual mean had risen slightly to 24 mg m^{-3}. This was followed by an irregular rise to a peak of 42 mg m^{-3} in 1995. Since then there has been an irregular decline and a remarkably steep decline since 2006 to the mean value in 2010 of 21 mg m^{-3}, very close to the first measurement in 1946. The typical seasonal pattern is rather muted, with the lowest concentrations occurring in May, possibly as a result of loss of TP to the sediment following decline and fall of the spring diatom populations. There has been a statistically significant increase in almost all months, apart from in September, the month of peak concentration (Fig. 8.11c). The lack of a long-term trend and the peak concentration itself probably result from erosion and breakdown of summer stratification releasing phosphorus, produced at depth from decomposition and sedimentary release, into the epilimnion.

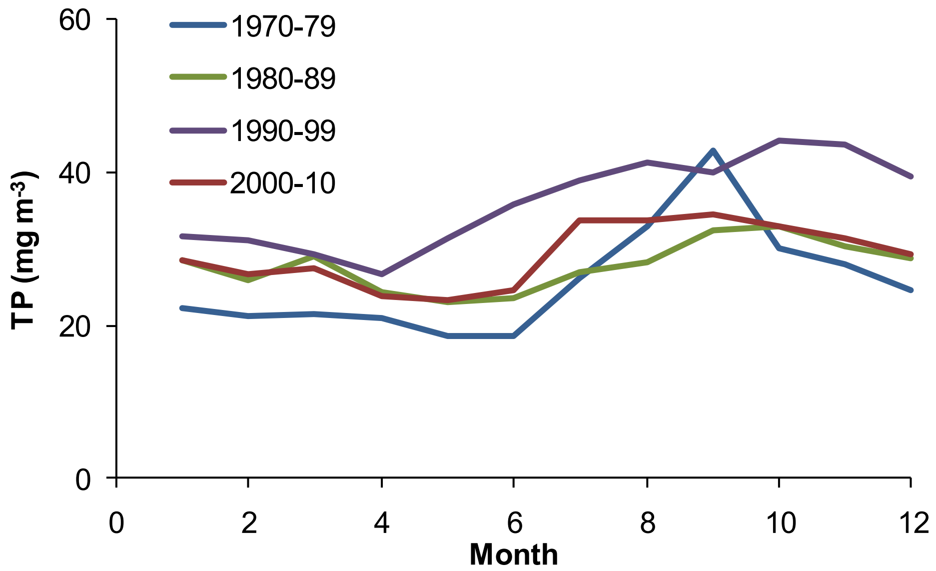

Figure 8.12. Patterns of seasonal changing total phosphorus concentration in Esthwaite Water in different decades.

There have not been major changes in seasonality of TP concentration over the decades that data are available, but more a general shifting in overall concentration (Fig. 8.12).

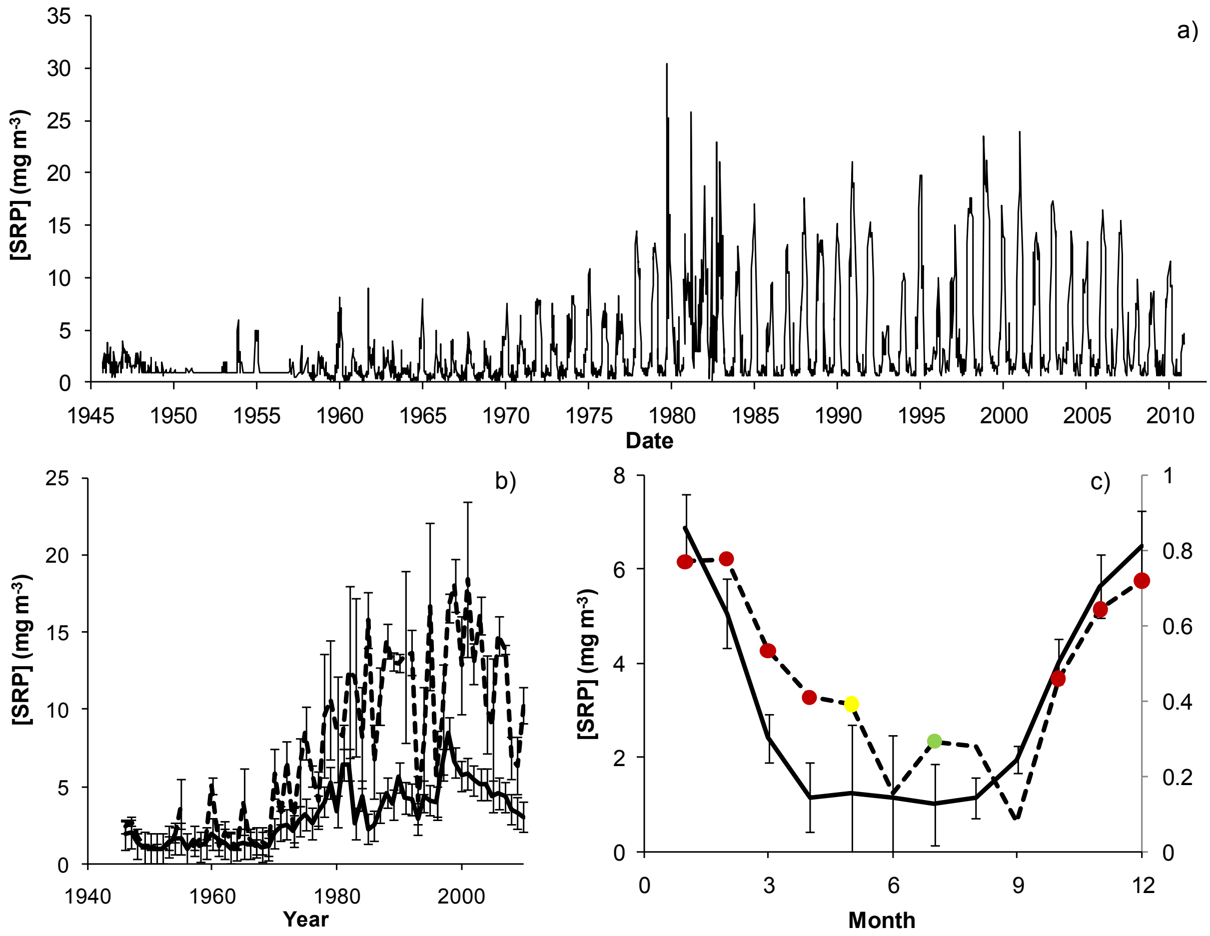

Figure 8.13. Concentration of soluble reactive phosphorus (SRP) in Esthwaite Water. a) long-term record from 1946 to 2010; b) annual mean and winter (DJF) mean (dashed line) (error bars show standard deviation); c) monthly mean (error bars show standard deviation) plus long-term correlation of monthly change for annual data (dashed line), green circle P<0.05, yellow circle P<0.01, red circle P<0.001.

The concentration of SRP has been measured since autumn 1945 (Fig. 8.13a). There is a strong seasonal pattern with highest concentrations in winter and lowest concentrations during the growing season (Fig. 8.13c). The annual mean concentration in 1946 (the first year with a complete record) was 1.9 mg m^{-3} and even winter concentrations were only 2.4 mg m^{-3}. These concentrations remained relatively stable until 1974 when the annual mean was 2.8 mg m^{-3} and the winter mean had jumped from 2.6 mg m^{-3} in 1973 to 6.0 mg m^{-3} in 1974. This step-change coincides precisely with the commissioning of the WwTW at Hawkshead. There followed a large increase up to an annual mean of 8.8 mg m^{-3} in 1998 and a winter mean of 17.8 mg m^{-3} in 2001. Both measures subsequently declined and in 2010 the annual and winter means were 3.1 and 10.3 mg m^{-3} respectively. Over all the available data, SRP comprises about 10% of TP, but this percentage is lowest during the summer growing season when demand is high and bioavailable phosphorus is in short-supply and largely bound up in the biota. The tertiary treatment introduced at the WwTW at Hawkshead did not have a discernible effect on winter or annual mean SRP, indicating either that the treatment was not very effective or that other factors, such as the expansion of the fish farm on the lake, masked any beneficial effect of the removal of this source of P.

8.6 Nitrogen

Nitrogen is an important requirement for growth of phytoplankton and aquatic macrophytes. In balanced growth it is required in 16-times greater amount than phosphorus on a molar basis (about 7.2-times on a mass-basis). Nitrogen can be an important limiting or co-limiting nutrient in upland lakes, including Cumbrian tarns (Maberly *et al.* 2002) and lowland shallow lakes (James *et al.* 2003). Although it is possible that nitrogen was more important in the past in controlling lake productivity before extensive man-made nitrogen deposition (Bergström & Jansson 2006), in many temperate low-elevation lakes it is often less important than phosphorus in controlling productivity and leading to eutrophication.

The long-term record shows that, like phosphorus, there has been an increase in nitrate concentration over the period of record (Fig. 8.14). This appeared to start in the 1960s, i.e. pre-dating the operation of the Hawkshead WwTW, and so is possibly linked to application of nitrogenous fertilizers to the improved pasture in the catchment, or possibly to atmospheric deposition. There are marked interannual variations in annual mean and annual maximum concentrations of nitrate that are linked to regional weather patterns such as the North Atlantic Oscillation (see Section 14.2 for more details). Since the 1980s the concentration of nitrate has shown no tendency to increase further (Fig. 8.14b). Like phosphorus, nitrate shows a strong seasonal pattern of change. Concentrations fall steadily from around March and reach a minimum in August and September (Fig. 8.14c). As a rule of thumb, when the concentration of nitrate-nitrogen falls below around 80 or 100 mg m^{-3} and in the absence of substantial ammonium as an alternative nitrogen source, there is a possibility of nitrogen-limitation (Maberly & Carvalho, 2010; Elliott *et al.* 2010). Inspection of Figure 8.14a shows that this concentration is reached in many, although not all, summers and so there may be a transient period of nitrogen-limitation at the end of the summer in Esthwaite Water. The possible consequences of this for the phytoplankton are discussed in Section 9.

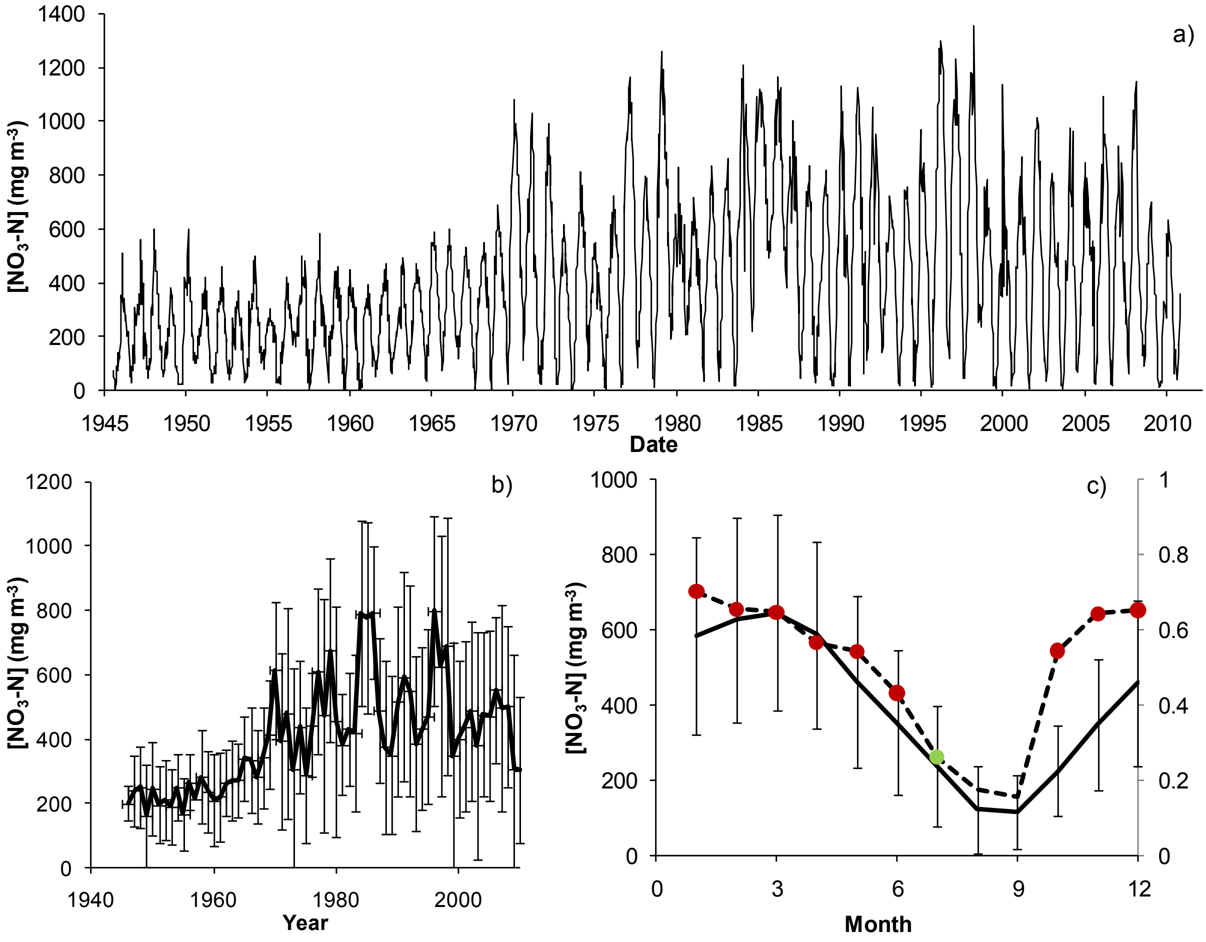

Figure 8.14. Concentration of nitrate-nitrogen in Esthwaite Water. a) long-term record from 1946 to 2010; b) annual mean (error bars show standard deviation); c) monthly mean (error bars show standard deviation) plus long-term correlation of monthly change for annual data (dashed line), green circle P<0.05, yellow circle P<0.01, red circle P<0.001.

Ammonium is generally present at lower concentration than nitrate, although it is a preferred nitrogen source for many phytoplankton. Annual mean concentrations are typically between 20 and 80 mg m^{-3} and annual maxima generally less than 300 mg m^{-3} (Fig. 8.15). Since around 1997, annual mean concentrations have fallen and monthly means show significant long-term decline in winter but it is not clear why this may have occurred. Ammonium concentrations are low throughout most of the year (Fig. 8.15c) but peak in October and November following the breakdown of stratification when ammonium at depth is entrained into the surface water.

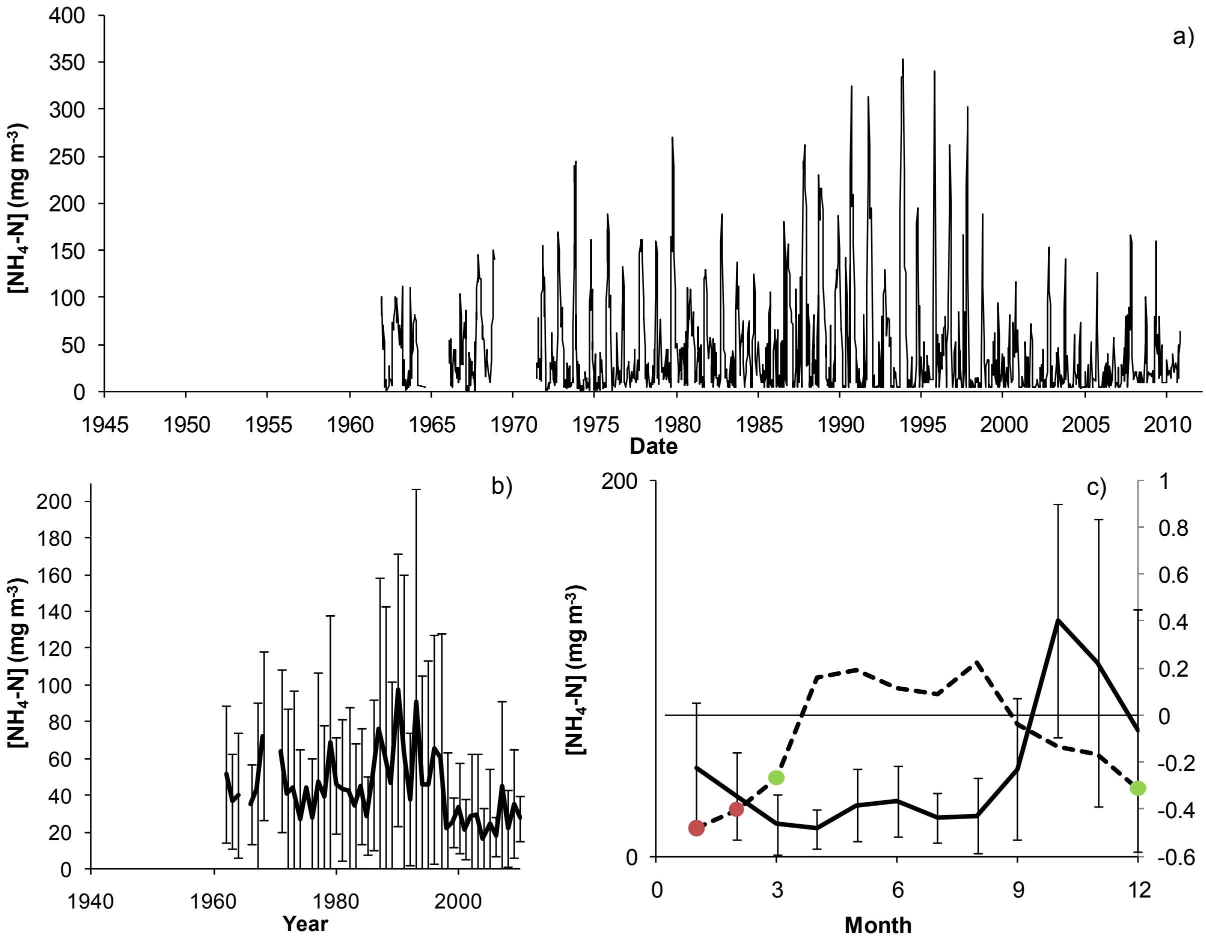

Figure 8.15. Concentration of ammonium-nitrogen in Esthwaite Water. a) long-term record from 1962 to 2010; b) annual mean (error bars show standard deviation); c) monthly mean (error bars show standard deviation) plus long-term correlation of monthly change for annual data (dashed line), green circle P<0.05, yellow circle P<0.01, red circle P<0.001.

8.7 Silica

Silica is required in large amounts by an important group of phytoplankton, the diatoms, and in smaller amounts by some other species such as the chrysophytes. Typically, winter concentrations are between 2500 and 3500 mg m^{-3} (as SiO_2) and in summer silica falls to very low concentrations, although in the 1940s and early 1950s summer minima were higher than later in the record (Fig. 8.16a). This difference is consistent with a lower demand for silica at the start of the record because of a lower productivity in the lake. The strong seasonal pattern with minima in May, June and July reflects the uptake and growth by spring phytoplankton that is dominated by diatoms. These low concentrations may be one of several reasons why diatoms are not present in great abundance in early summer. The subsequent net replenishment in late summer and autumn results from lower demand for silica because diatoms are less abundant and inflow of silica from the catchment.

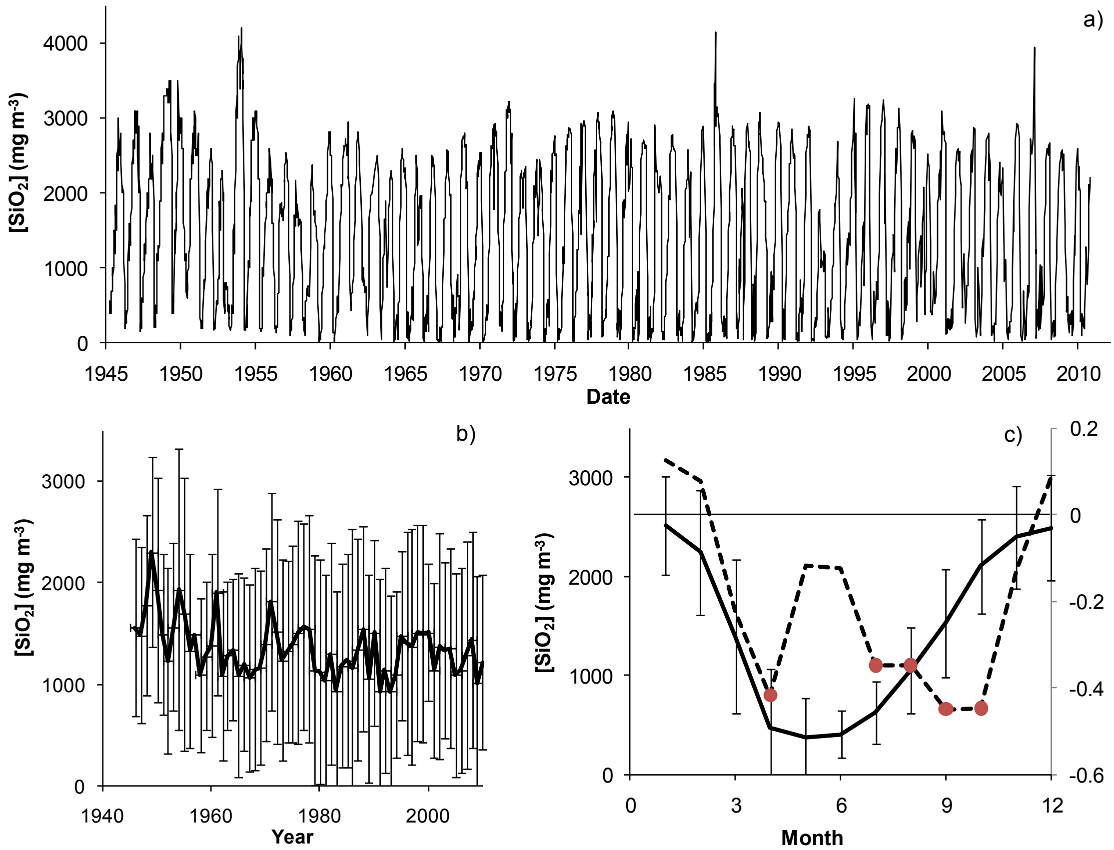

Figure 8.16. Concentration of silica in Esthwaite Water. a) long-term record from 1945 to 2010; b) annual mean (error bars show standard deviation); c) monthly mean (error bars show standard deviation) plus long-term correlation of monthly change for annual data (dashed line), green circle P<0.05, yellow circle P<0.01, red circle P<0.001.

8.8 Dissolved Organic Carbon (DOC)

In many northern lakes, concentration of dissolved organic carbon can be very high. In these lakes the DOC is usually derived from breakdown of terrestrial plant material and is often highly coloured. It can have a significant effect on the underwater light climate, sometimes reducing lake productivity (Karlsson *et al.* 2009) but also contributing fixed carbon that can be used by the lake microbes. DOC can also be produced within a lake and then it tends to be less highly coloured. Concentrations of DOC are tending to increase in many lakes as catchments recover from acidification (Monteith *et al.* 2007).

DOC is not measured as part of the long-term monitoring programme and most lakes in the English Lake District are not coloured and have low concentrations of DOC with measured values between 0 and 6.2 g m^{-3} for thirteen lakes (Tipping *et al.* 1988). Esthwaite Water had DOC concentrations varying from 1.4 to 6.2 g m^{-3} over the study period of May to November 1986, with an average value of 2.6 g m^{-3}. Fourteen inflowing streams had mean DOC concentrations of between 0.6 and 3.4 g m^{-3} over the same time period, suggesting that at least some of the DOC in Esthwaite Water is produced within the lake and this is further supported by higher concentrations in summer than winter (Tipping *et al.* 1988). Furthermore, the absorption of the DOC in summer at 340 nm is relatively low, suggesting that it is not coloured DOC derived from the catchment but DOC produced within the lake with low colour.

8.9 Depth-variation in water chemistry

The water chemistry reported above is largely derived from an integrated water-sample from the top 5 m, reflecting surface conditions. However, in stratified productive lakes, the combination of a strong separation into an upper epilimnion and lower hypolimnion during much of the summer (Section 6.3), with rapid attenuation of light with depth (Section 7.7) leads to zones of production and consequently nutrient and CO_2 depletion and oxygen production in the epilimnion and zones of decomposition and consequently nutrient generation and oxygen consumption in the hypolimnion. These depth variations have been studied extensively in Esthwaite Water, especially by J.F. Talling and S.I. Heaney who summarised much of their work in Heaney *et al.* (1986). One of the major variables that can change with depth in stratified lakes is the concentration of dissolved oxygen. In unproductive lakes, such as in Wastwater in the west of the English Lake District, oxygen concentrations vary little with depth because biological activity in low. In productive lakes, however, the surface is typically saturated or super-saturated with oxygen as high light levels allow photosynthetic oxygen production. Light decreases with depth (Section 7.7) and so the rate of oxygen production declines until respiration exceeds photosynthesis at low light levels. During stratification, water within the hypolimnion is effectively separated from oxygen replenishment from the atmosphere and from photosynthetic production, and decomposition processes cause the oxygen concentration to decline. There is a strong relationship between the minimum oxygen concentration at depth and the productivity of the lake (Fig. 8.17).

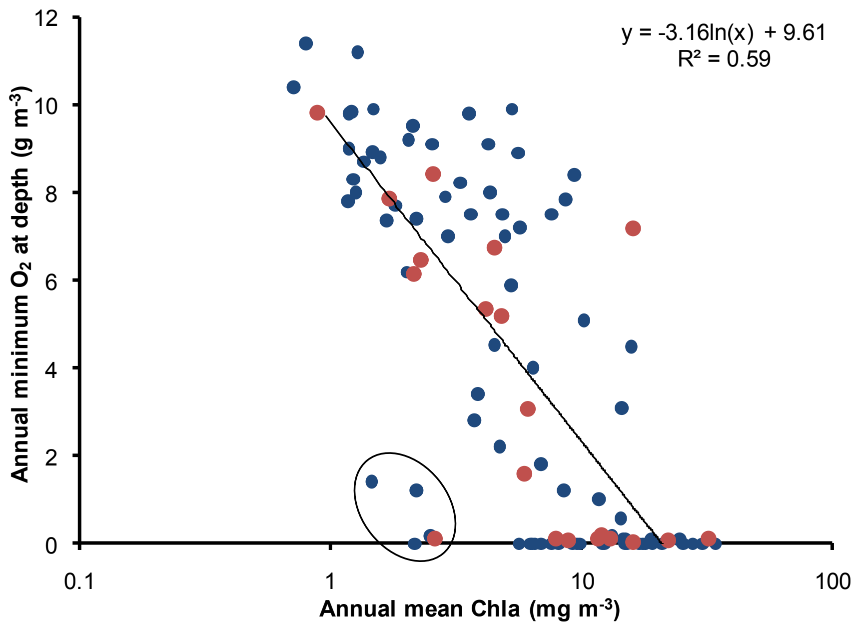

Figure 8.17. Depth variation in annual minimum concentration of oxygen at depth versus annual mean concentration of chlorophyll a (log scale). Data from the 20 lakes in the Lakes Tour (Maberly et al*. 2011). The red symbols represent data from 2010, the blue symbols data from 1984, 1991, 1995 2000 and 2005. The five circled symbols derive from Brothers Water.*

In Esthwaite Water, in the summer, oxygen concentration decline with depth and are typically less than 50% of air-saturation below about 6 m (Fig. 8.18). Oxygen concentrations have declined with depth even in the earliest records from the last 1960s (Fig. 8.18) and from earlier studies in 1939 and 1940 (Mortimer 1941-2).

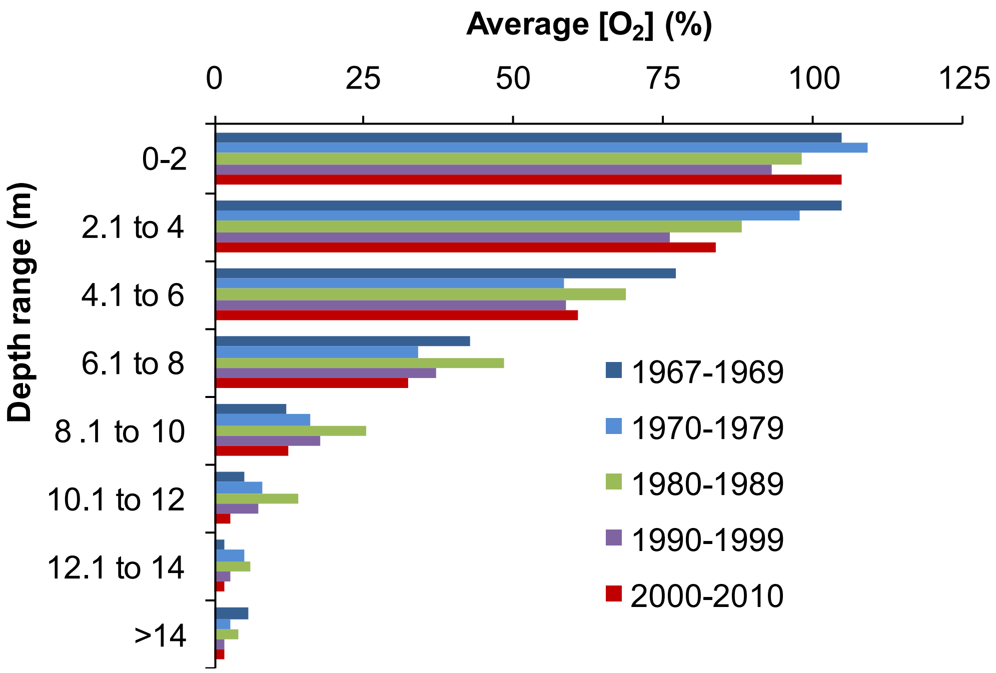

Figure 8.18. Changes in average oxygen concentration (as percent of air-saturation) with depth in Esthwaite Watrer for five different decades.

There is a large amount of year-to-year variation in the long-term annual minimum concentration of oxygen (Fig. 8.19), but there has been a tendency for this concentration to decline since 1982 (correlation coefficient = -0.54, P<0.01). This is supported by the monthly analysis showing that there has been a significant decline in oxygen concentrations, for much of the summer (Fig. 8.19).

The depth-profiles of temperature, light, and oxygen are associated with other depth-profiles such as increasing concentrations of CO_2 and nutrients, especially phosphate and ammonium: associated with nutrient regeneration at depth and redox-linked sediment release (Sections, 5.5, 6.3). Other water chemistry variables that increase at depth include conductivity, alkalinity, concentrations of iron and manganese (Heaney *et al.* 1986, Talling 1999). These depth-changes also affect the distribution of phytoplankton, zooplankton (Heaney *et al.* 1986) and presumably by analogy with Windermere (Jones *et al.* 2008), fish.

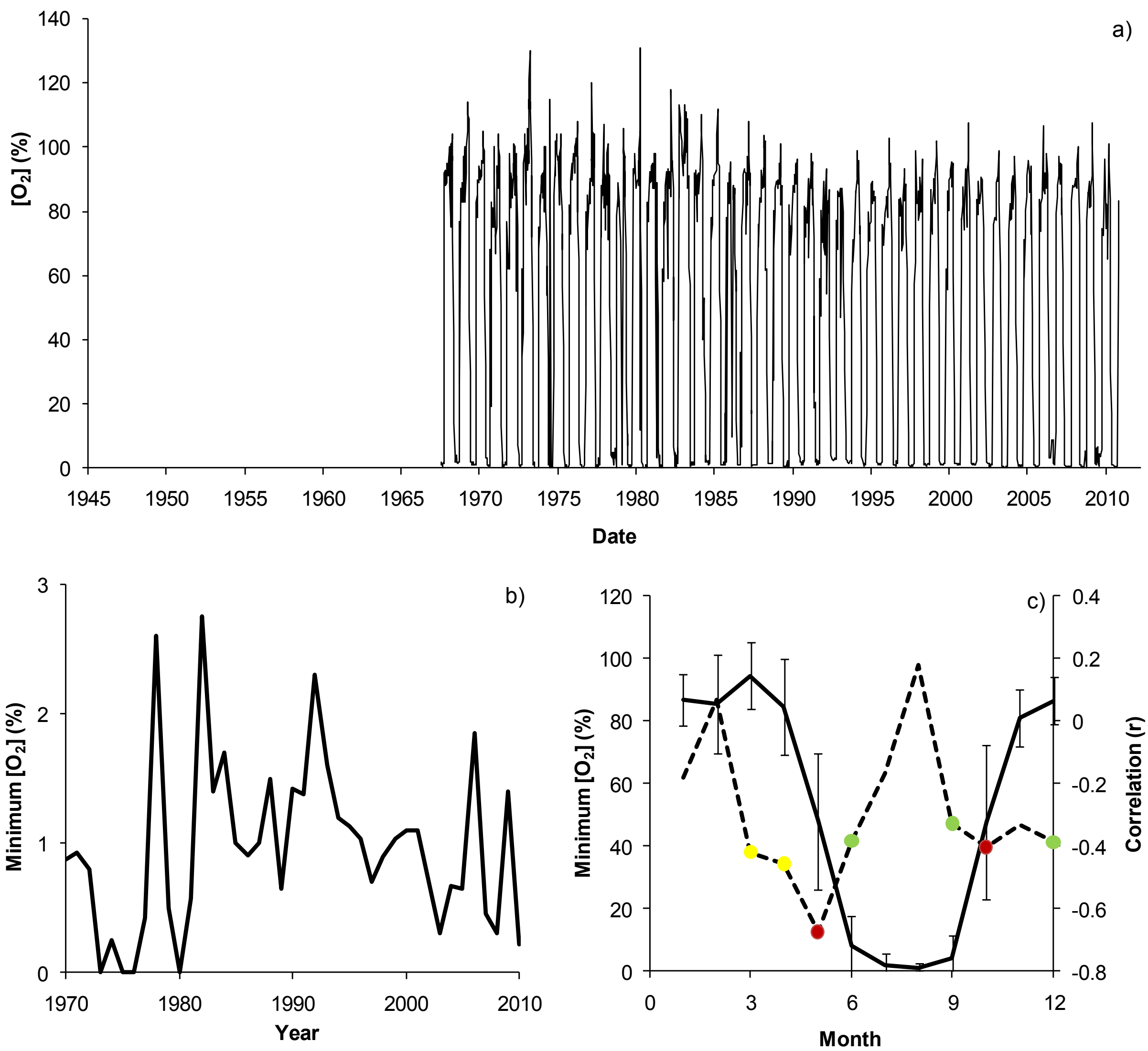

Figure 8.19. Minimum concentration of oxygen in Esthwaite Water: a) long-term record from 1968 to 2010; b) annual minimum; c) monthly mean of the minimum (error bars show standard deviation) plus long-term correlation of monthly change for annual data (dashed line), green circle P<0.05, yellow circle P<0.01, red circle P<0.001.

Figure 8.20 shows and example depth- profile taken in July 2005. In the upper 4 m which represent the epilimnion, concentrations of nutrients are low and probably limiting. At depth, concentrations of ammonium, TP and SRP all increase, in the case of the phosphorus forms, the increase only occurs below 10 m while ammonium shows elevated concentrations higher in the water column, below about 6 m. Nitrate concentrations are more complicated with mid-depth maxima at about 8 m. Lower concentrations at depth probably results from the conversion of nitrate to ammonium in dissimilatory nitrate reduction and denitrification of nitrate to nitrogen gas.

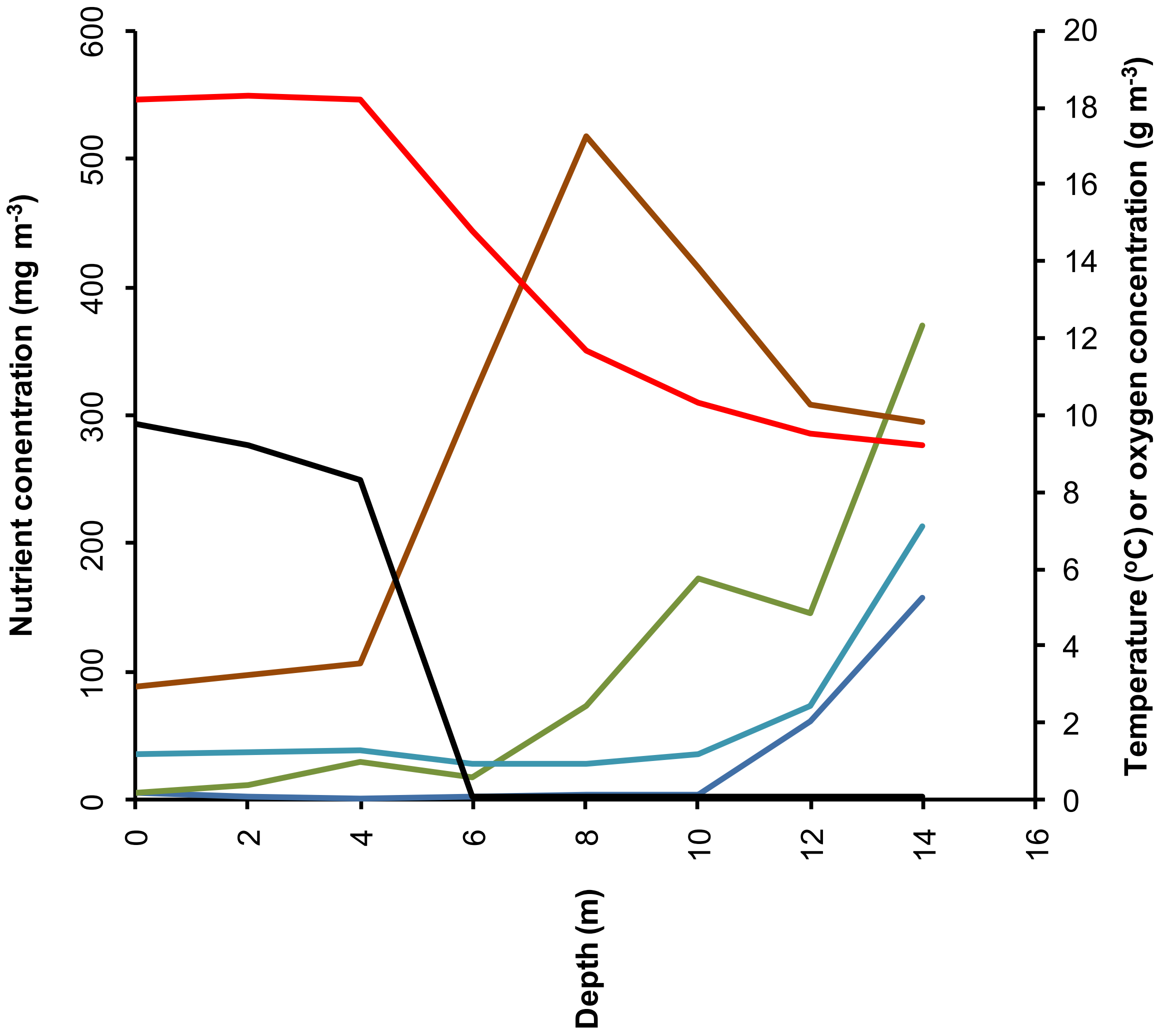

Figure 8.20. Example of depth-profiles of Esthwaite Water on 26/07/2005 showing temperature (red line), oxygen concentration (black line), nitrate-N concentration (brown line), ammonium-N concentration (green line), SRP concentration (dark blue line) and TP concentration (light blue line).

9. Phytoplankton

9.1 Introduction

The phytoplankton are a phylogenetically disparate group of eukaryotic algae plus prokaryotic cyanobacteria. Along with macrophytes, they form the base of the food-chain in most lakes (a few rely partly on production by photosynthetic bacteria). In Esthwaite Water today, macrophytes are relatively scarce and the phytoplankton are the main primary producers. They are also one of the most visible symptoms of nutrient enrichment as some, particularly certain cyanobacteria can form surface scums and be potentially toxic. All cause a reduction in light attenuation and when they decompose and settle into the hypolimnion can provide the organic material for microbial decomposition that results in oxygen depletion at depth.

9.2 Chlorophyll *a*

All types of photosynthetic algae produce chlorophyll *a* as the main light harvesting and processing pigment in addition to the group-specific accessory pigments such as chlorophyll *b*, *c*, fucoxanthin and phycocyanin. It is therefore a very useful 'chemical' measure of phytoplankton abundance although it does not necessarily relate directly to other measures of phytoplankton abundance such as biovolume or biomass.

Chlorophyll *a* has been measured in Esthwaite Water since 1964 (Fig. 9.1) as a result of the pioneering work of Jack F Talling of the FBA. In the period from 1964 to 1969, the mean concentration was 16 mg m^{-3} (Fig. 9.1b). Mean concentrations in subsequent decades were higher at 22, 20, 25 and 20 mg m^{-3} for the 70s, 80s, 90s and 2000s respectively. In the 1970s in particular some very high concentrations were recorded, with a maximum spot reading of 344 mg m^{-3} in September 1973 and a monthly maximum of 199 mg m^{-3} in the previous month. It is likely that these very high concentrations of chlorophyll *a* are at least partly the result of local concentrations of algae, mainly the dinoflagellate *Ceratium* sp and the cyanobacterium

Microcystis. sp.. The seasonal pattern over the period of record is for a spring bloom peaking in April followed by a small decline to May and a subsequent large increase to late summer annual maxima in August and September of between 40 and 50 mg m^{-3}. Over the period of record there have been statistically significant increases in concentration of chlorophyll *a* in May, June, July and October and a significant reduction in September (Fig. 9.1), probably as a result of changed phytoplankton species composition over the years.

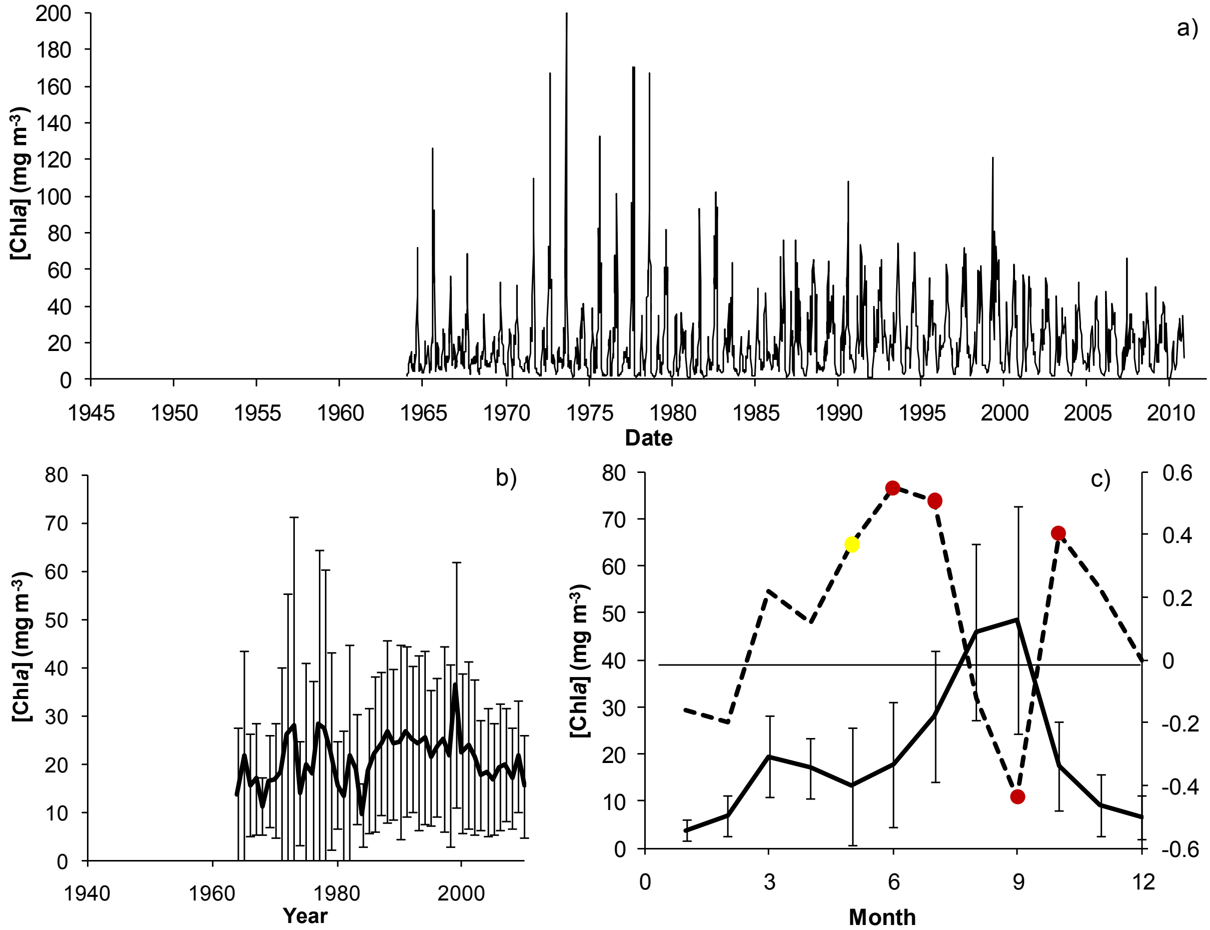

Figure 9.1. Concentration of phytoplankton chlorophyll a *in Esthwaite Water. a) long-term record from 1964 to 2010; b) annual mean (error bars show standard deviation); c) monthly mean (error bars show standard deviation) plus long-term correlation of monthly change for annual data (dashed line), green circle P<0.05, yellow circle P<0.01, red circle P<0.001.*

A more detailed analysis of the different patterns of seasonal change (Fig. 9.2) shows that the spring bloom has been largely unaffected by nutrient enrichment although there has been a tendency for the spring bloom to be occurring earlier (see also Section 14.3). In contrast to the 1960s, summer and autumn populations of phytoplankton were larger in the 1970s. Some of the changes appear to slightly pre-date the commissioning of the Hawkshead WwTW in November 1983 (Fig. 9.1), but it is possible that nutrient inputs were increased in the months leading to its full operation. A feature of the 1970s to 1990s is greater mid-summer populations. Fig. 9.2 shows that there has been a slight reduction in summer and early autumn phytoplankton chlorophyll *a* in the 2000s compared to the 1990s.

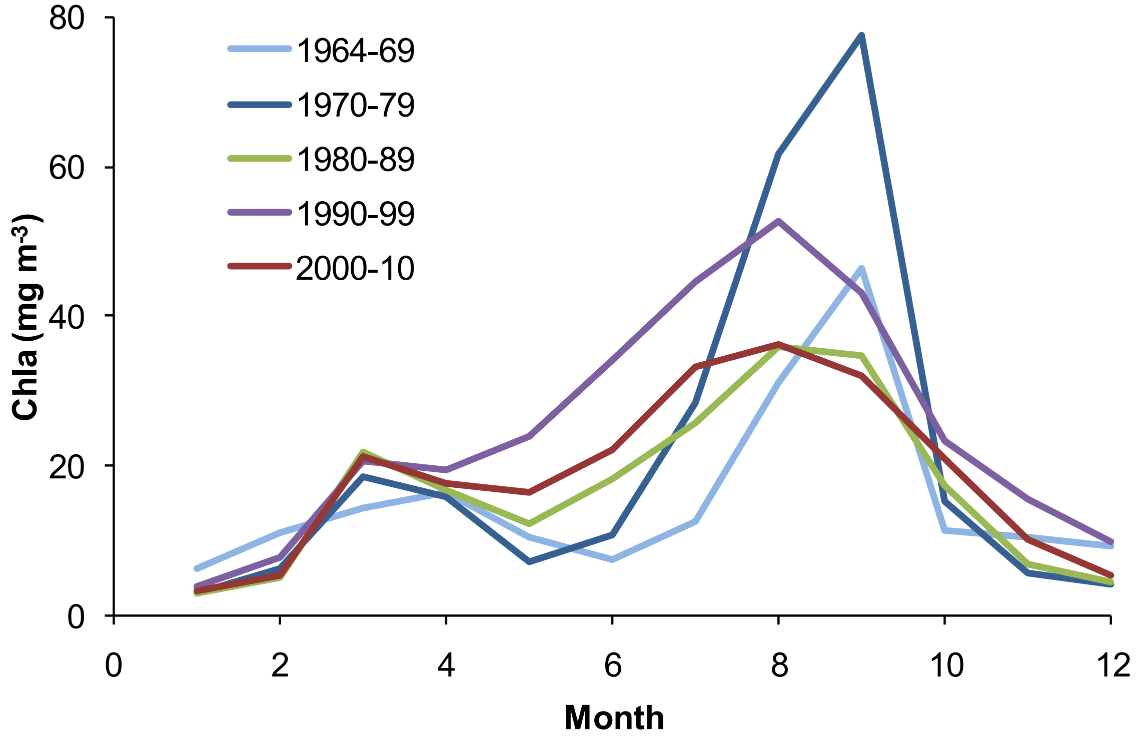

Figure 9.2. Patterns of seasonal changing chlorophyll a in Esthwaite Water in different decades.

9.3 Ratio of chlorophyll *a* to total phosphorus (Chla:TP)

A useful measure of the efficiency of conversion of phosphorus into phytoplankton is the ratio of phytoplankton chlorophyll *a* to total phosphorus. At the annual scale, the record shows interannual variation but no clear trend, apart from possibly a slightly greater ratio at the start of the 1970s. Seasonally, August and September are the months that support the greatest amount of chlorophyll *a* per unit of total phosphorus and there has been a significant reduction in this ratio in these two months but inspection of the raw data shows that this is largely the result of high ratios in the 1970s (not shown).

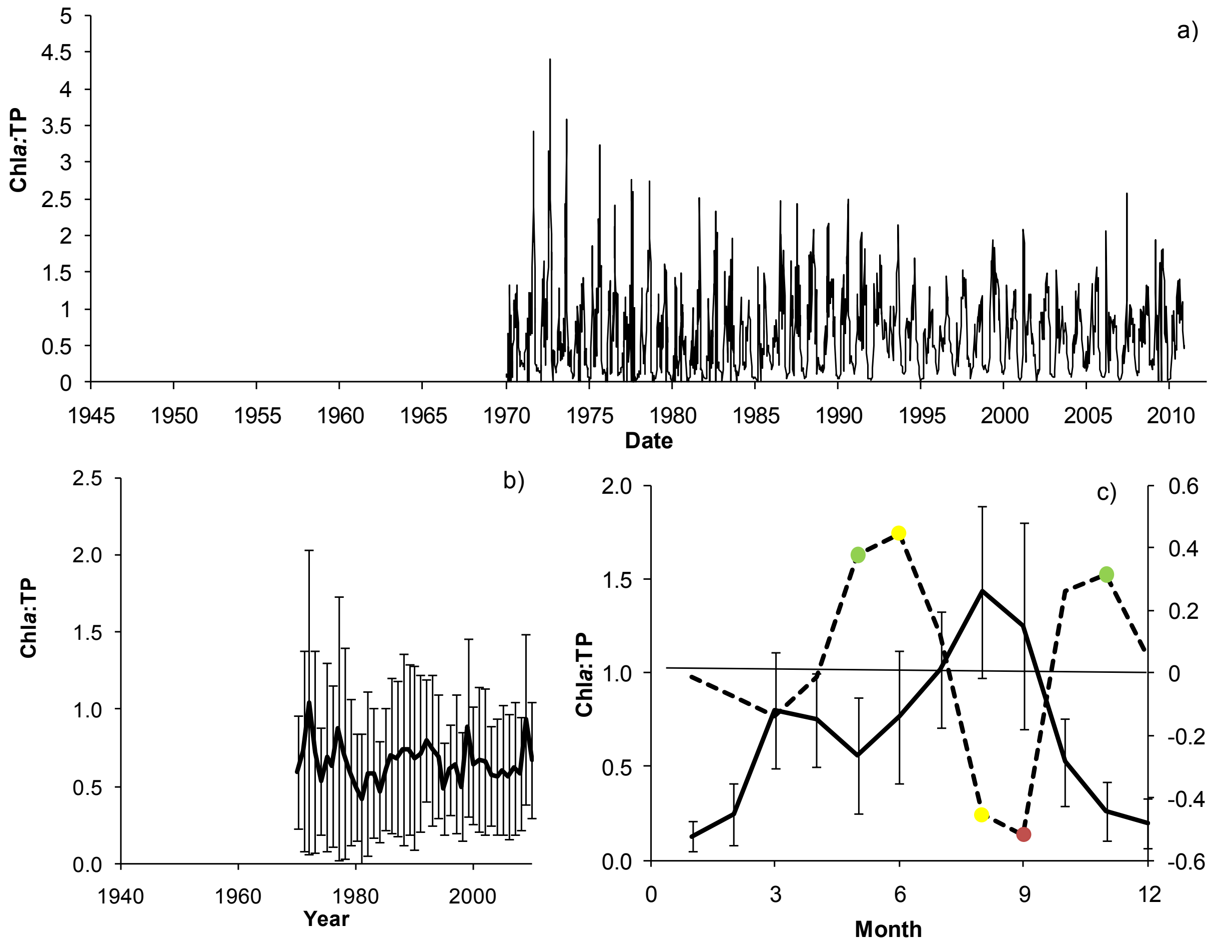

Figure 9.3. The ratio of concentration of phytoplankton chlorophyll a to total phosphorus *in Esthwaite Water. a) long-term record from 1970 to 2010; b) annual mean (error bars show standard deviation); c) monthly mean (error bars show standard deviation) plus long-term correlation of monthly change for annual data (dashed line), green circle P<0.05, yellow circle P<0.01, red circle P<0.001.*

9.4 Species composition

Counts and phytoplankton identification have been recorded in Esthwaite Water over 66 years since 1945. Over 500 taxa have been recorded, of which, 190 have only been recorded once. Table 9.1 lists the most frequently occurring taxa: those that have been recorded in more than half the 66 years of record. The most persistent species in the record are the three diatoms

Asterionella formosa, Aulacoseira subartica (= *Melosira subartica*) and *Fragilaria crotonensis*, the colonial green algae, *Coenochloris fottii* and *Dictyosphaerium pulchellum*, the two dinoflagellates *Ceratium hirundinella* and *Peridinium* sp. and the cyanobacterium *Planktothrix mougeotii* (=*Oscillatoria agardhii* f. isothrix).

Table 9.1. Phytoplankton recorded in more than half the 66 years of record from Esthwaite Water.

Phytoplankter	**Number of years**
Asterionella formosa HASSALL 1855	66
Aulacoseira subarctica (O MÜLLER) E Y HAWORTH 1988	65
Coenochloris fottii (HINDÁK) TSARENKO 1990	65
Fragilaria crotonensis KITTON 1869	65
Ceratium hirundinella (OF MÜLLER) DUJARDIN 1846	64
Dictyosphaerium pulchellum WOOD 1874	63
Peridinium EHRENBERG 1832 sp.	62
Planktothrix mougeotii (BORY EX GOMONT) ANAGNOSTIDIS ET KOMÁREK 1988	62
Staurastrum chaetoceras (W & GS WEST) GM SMITH 1924	59
Tabellaria flocculosa var. *asterionelloides* (GRUNOW) KNUDSON 1952	55
Scenedesmus MEYEN 1829 sp.	54
Botryococcus braunii KÜTZING 1849	53
Gymnodinium STEIN 1878 sp. (large)	51
Staurastrum cingulum (W & G.S.WEST) G.M.SMITH 1922	51
Dinobryon divergens IMHOF 1887	50
Gymnodinium STEIN 1878 sp. (small)	50
Mallomonas caudata IVANOV 1899	49
Anabaena solitaria KLEBAHN 1895	46
Cosmarium abbreviatum RACIBORSKI 1885	45
Aphanothece clathrata WEST & G.S.WEST 1906	44
Microcystis aeruginosa (KÜTZING) KÜTZING 1846	44
Peridinium EHRENBERG 1832 sp. (medium)	44
Anabaena flos-aquae BRÉBISSON EX BORNET & FLAHAULT 1888 SENSU KOMÁREK & ETTL 1958	43
Gymnodinium helveticum forma *achroum* SKUJA 1948	42
Trachelomonas varians DEFLANDRE 1924	42
Anabaena circinalis &/or flos-aquae	40
Monoraphidium contortum (THURET) KOMÁRKOVÁ-LEGNEROVÁ 1969	40
Cryptomonas EHRENBERG 1838 spp. 'large' (C. curvirostrata)	39
Gymnodinium STEIN 1878 sp. (medium) = *Gymnodinium helveticum*	39
Cryptomonas EHRENBERG 1838 spp. 'medium' (C. ovata)	37
Peridinium lomnickii WOLOZYNSKA 1916	37
Cryptomonas EHRENBERG 1838 spp.	35
Anabaena circinalis RABENHORST EX BORNET & FLAHAULT 1888 SENSU KOMÁREK & ETTL 1958	34
Chlorella BEIJERINCK 1890 sp.	34
Peridinium cinctum (OF MÜLLER) EHRENBERG 1838	34

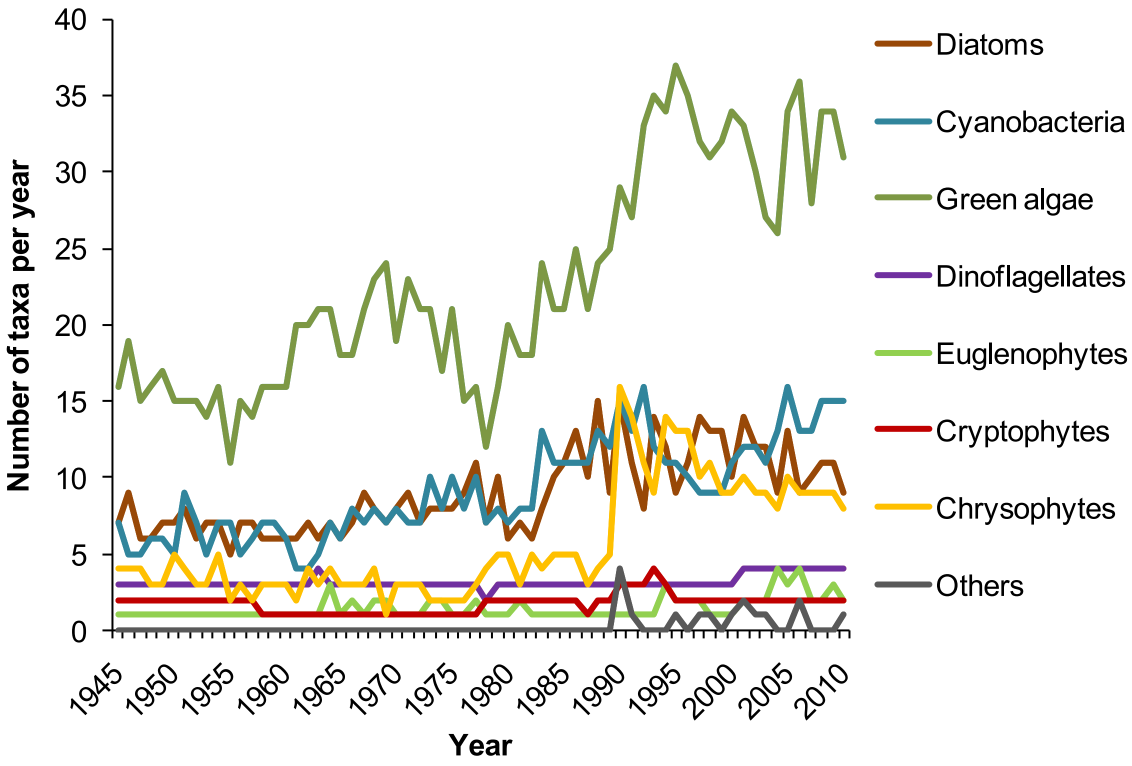

Figure. 9.4. Number of phytoplankton taxa in different phylogenetic groups in Esthwaite Water from 1945 to 2010.

9.5 Seasonal changes in phytoplankton

In addition to seasonal changes in chlorophyll *a* (Figs 9.1 and 9.2) the species that make up the phytoplankton community also change. Figure 9.5 shows average monthly patterns for the eight most consistently recorded taxa. The two diatoms *Asterionella formosa* and *Aulacoseira subarctica* produce maxima in spring. Their growth is controlled by physical factors such as water temperature and light availability which is the reason the size of the spring bloom has been relatively unaffected by changes in nutrient availability. Diatoms have dense silica cell walls and their rapid decline in late spring corresponds to when silica runs out (Section 8.7) and the lake stratifies (Section 7.3) and water currents are not sufficient from keeping them suspended in the water column. The colonial green alga *Coenochloris fottii* produces a mid-summer population peak and another colonial green algal produces a broader peak slightly later in the year on average. The two dinoflagellates *Ceratium* spp. and *Peridinium* spp. tend to produce late-summer maxima at times of usually severe nutrient limitation, but their motility gives them the opportunity to exploit nutrients diffusing out from below the thermocline.

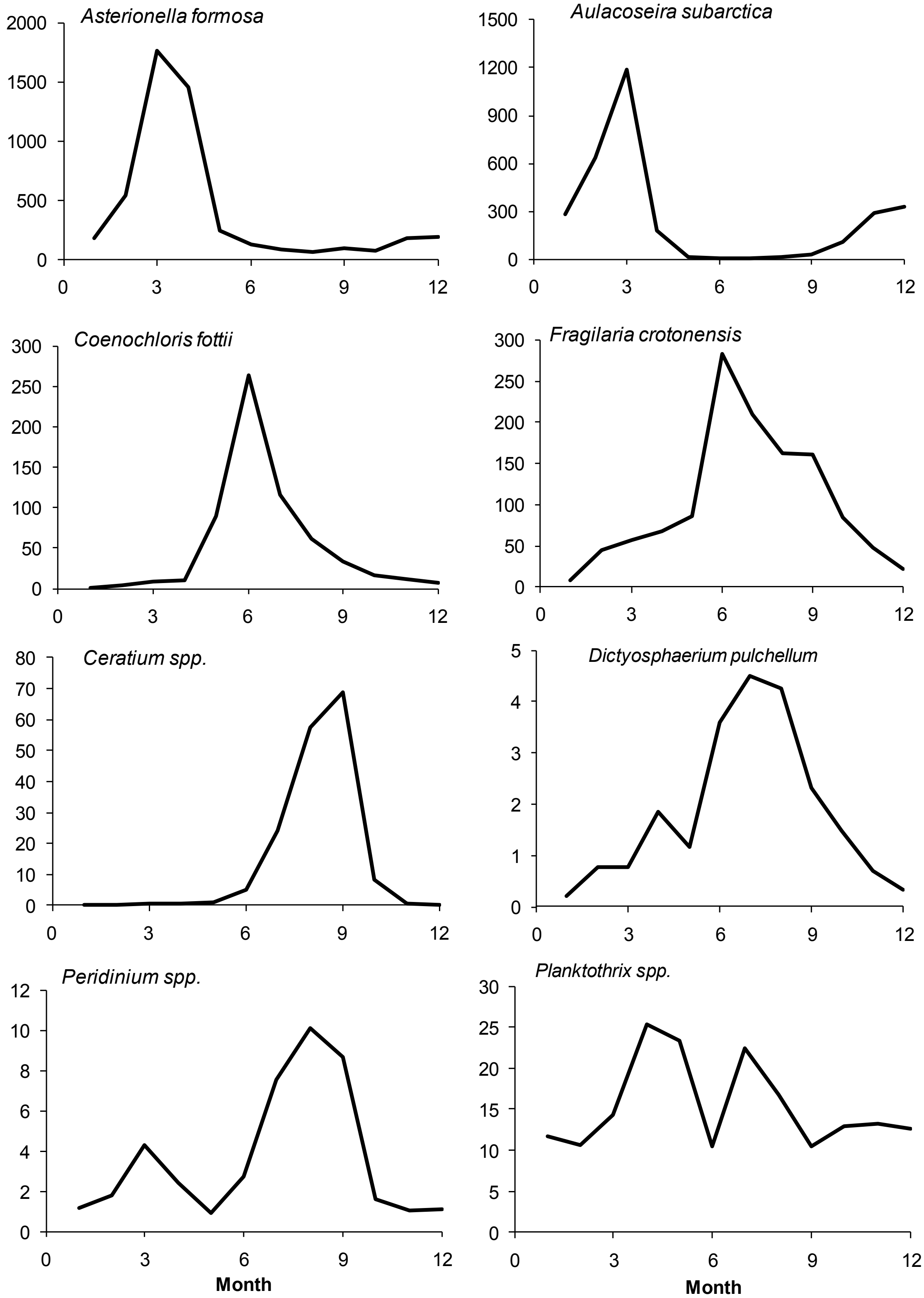

Figure 9.5. Long-term average seasonal patterns in the eight most consistently recorded taxa (Table 9.1) based on data from 1945- 2010. Values show monthly average cell density (cell cm-3) apart from Dictyosphaerium pulchellum *(colony cm^{-3}) and* Planktothix *spp. (filament cm^{-3}).*

10. Macrophytes

10.1 Introduction

Macrophytes are an ecological group that comprise flowering plants, mosses and fern-allies and large-celled green algae (charophytes). They are mainly rooted in the sediment, although some are free-floating in the water column for at least part of their life-cycle. They grow along a gradient, depending on species and water-level from completely submerged, species with floating leaves, emergent species in water and fringing species whose roots are permanently in waterlogged soil but that otherwise act as terrestrial plants. This latter group will not be included in the review. In shallow lakes, macrophytes are often the main primary producers in the lake although disturbance, often nutrient enrichment, can shift the lake from a macrophyte-dominated clear water lake to an alternative 'stable-state' with turbid water and dominated by phytoplankton (Scheffer 1997). In deeper and more exposed lakes, the macrophyte-dominated zone can be restricted by unsuitable substrate and high wave exposure in shallow water and low light levels at depth, so that macrophytes contribute little to the primary productivity of the lake. Even when this is so, macrophyte beds can be ecologically important in intercepting nutrient runoff from the land, facilitating exchange of nutrients between the sediment and water, stabilising sediment, altering the physico-chemical conditions within a macrophyte stand, acting as conduits for loss of gases to the atmosphere, especially the greenhouse gas methane, and providing structured habitat that can be used by zooplankton, macroinvertebrates and fish.

10.2 The macrophyte flora of Esthwaite Water

One of the earliest scientific ecological studies on macrophytes was carried out between 1914 and 1916 by W.H. Pearsall on Esthwaite Water (Pearsall 1917). He recorded 22 species of submerged or floating-leaved macrophytes in addition to several emergent species. He produced detailed maps of aquatic plant communities around Strickland Ees, in the northernmost bay and around the Ees south of Elter Holme. Subsequent surveys were made by Stokoe (1983) between 1974 and 1980. Wade carried out a survey in 1982 specifically to look for *Najas flexilis* (Wade 1994) and the most recent published survey is that carried out by Darwell in 1999 (Darwell 2000). A list of species recorded in these different surveys is shown in Table 10.1.

Pearsall recorded 22 species including five species that have not subsequently been recorded: *Hydrilla verticillata*, *Najas flexilis, Potamogeton alpinus, P. perfoliatus* and *Ranunculus truncatus (probably R. aquatilis).* Of these, *H. verticillata* (sometimes known as Esthwaite Waterweed) is one of the most celebrated. It was found in Esthwaite Water by Pearsall in 1914 and observed over several years (Pearsall 1921b, 1936) and last seen in 1941. *H. verticillata* is native to north east Europe and is has been suggested that it was carried to Esthwaite by birds. *Potamogeton alpinus* and *P. perfoliatus* are both plants of mid-depth water, *P. perfoliatus* slightly deeper than *P. alpinus*, both are fairly widespread in the UK and it is not clear why these species should have been lost. *P. perfoliatus* is fairly abundant in nearby Windermere. *Najas flexilis* grows in deep clear water in mesotrophic lakes (Preston & Croft 1997). It was recorded as abundant by Pearsall in 1914 growing in stands with linear-leaved Potamogetons *Nitella flexilis* and *Elodea canadensis* at depths of around 1.5 to 2.6 m. It is a rare species in the UK and Esthwaite Water was its only English location. It is regarded as being vulnerable to eutrophication, but the precise sensitivity of this species is not known. Despite an extensive search by an experienced surveyor (Darwell, 2000) it now appears to be extinct in Esthwaite Water. Pearsall (1917) notes that this species produces abundant seeds so it is possible that viable seeds may still exist in the sediment.

Table 10.1. Species of submerged and floating-leaved macrophytes recorded in Esthwaite Water in 1914 (Pearsall 1917), 1974-1980 (Stokoe 1983) and 1999 (Darwell 2000).

Species	1914	1980	1999	Notes
Callitriche hamulata		✓		
Callitriche hermaphroditica	✓	✓		*C. autumnalis* in Pearsall
Callitriche spp.			✓	
Chara delicatula		✓		
Elodea canadensis	✓	✓		
Elodea spp.			✓	
Fontinalis antipyretica	✓	✓	✓	
Hydrilla verticillata var. *pomeranica*	✓			
Isoetes lacustris	✓	✓	✓	
Lemna trisulca		✓		
Littorrella uniflora	✓	✓	✓	
Lobelia dortmanna	✓	✓		
Myriophyllum alterniflorum	✓	✓		
Najas flexilis	✓			
Nuphar lutea	✓	✓	✓	
Nitella flexilis/ sp.	✓	✓	✓	
Nymphaea alba	✓	✓		*Castalia alba* and *C. minor* in Pearsall
Potamogeton alpinus	✓			
Potamogeton berchtoldii	✓	✓	✓	Pearsall recorded as *P. pusillus* var. *tenuissimus*
Potamogeton crispus	✓	✓	✓	Pearsall also recorded *P. crispus* var. *serratus*
Potamogeton gramineus	✓	✓		
Potamogeton natans		✓		
Potamogeton obtusifolius	✓	✓	✓	Includes Pearsall's *P. sturrockii*
Potamogeton perfoliatus	✓			
Potamogeton pusillus	✓	✓		
Ranunculus peltatus	✓	✓		
Ranunculus aquatilis	✓			Pearsall recorded as *R. truncatus*
Sparganium erectum		✓		
Sparganium natans	✓	✓	✓	Recorded as *S. minimum* before Darwell
Total number of species	**22**	**22**	**11**	

There appears to have been a dramatic loss of species in Esthwaite Water in recent years. Pearsall's survey around 1914 and surveys by Ralph Stokoe in the late 1970s and 1980 both recorded 22 species of submerged and floating-leaved macrophytes (although not the same species). A re-survey in 1999 only recorded eleven species, so the species richness appears to have halved. A survey carried out by S.C. Maberly and I.J. Winfield on 27 July 2011 (report in preparation) found a similar low number of species and the non-native species *Elodea nuttallii* was by far the most dominant and widespread species.

Species loss is a typical response to nutrient enrichment and increasing abundance of phytoplankton via an effect of the phytoplankton on light attenuation and carbon-dioxide depletion. One species that is particularly sensitive to the effects of nutrient enrichment is the isoetid *Lobelia dortmanna*. This was last recorded in the 1974 to 1980 surveys by Stokoe (1983) although it was subsequently seen flowering, and in good condition, in 1983, but in a small bay, Robin's Wyke, that is partly separated from the main lake (Maberly S.C. *pers obs*). It was absent in a subsequent visit in the early 2000s. Small low-growing species may be able to survive carbon-depletion by growing close to the sediment surface where concentrations of CO_2 are higher than in the overlying water. This was shown to allow the survival of the aquatic moss *Fontinalis antipyretica* in the shallow water of the northern bay (Maberly 1985a,b). Depth limits for macrophytes noted by Pearsall appeared to be about 3.6 m and this was recorded for *Nitella flexilis*. Maberly (1993) recorded the same depth as the limit for *Potamogeton obtusifolius* in 1982. However, the macrophyte survey in July 2011 found a depth limit of about 2.3 m, substantially less than in earlier studies. The balance of probabilities is that light attenuation will be poorer now than in previous years, accounting for the reduced depth-limits, and this is borne out by the records of Secchi depth that show an annual reduction (Fig. 7.15b), especially since about 1986, and significant long-term reductions in May, June and July (Fig. 7.16c), which are critical months for macrophyte growth.

11. Protozoa & zooplankton

11.1 Protozoa

The protozoa are a community of microscopic, single-celled, organisms that inhabit both open waters and sediments in aquatic environments. These organisms consume phytoplankton, detritus, bacteria and each other and may indeed be significant grazers in aquatic systems. They are also consumed by the multi-cellular members of the zooplankton; the rotifers and crustaceans. In Esthwaite Water, studies of protozoan communities have focussed largely on spatial and temporal variations in abundance with a particular emphasis on the effects of seasonal changes in water column structure and oxygen concentrations.

11.2 Protozoan community composition

Esthwaite Water supports a diverse community of benthic protozoa, including representatives of the amoebae, flagellates and ciliates. In an intensive survey of sediments at different depths in the lake Webb (1961) found more than 120 distinct species, the majority of which (approx. 90 species) were ciliates. It is perhaps as a result of this diversity that much of the subsequent research on the protozoan community of Esthwaite Water has focussed upon this group. The species present in the lake differ in their ecological requirements, particularly with respect to their tolerance of oxygen depletion. Indeed, Webb (1961) distinguished three categories: *i)* species found under oxygenated conditions, *ii)* species that aggregate under conditions of oxygen depletion (though not complete anoxia) and *iii)* species that can be found in completely anoxic conditions.

The protozoan, particularly ciliate, community of Esthwaite Water has been used as a model system to test hypotheses relating to the mechanisms behind community assembly, and therefore diversity. The genus *Loxodes* has been a particular focus for studies of interspecific competition. In Esthwaite Water closely related *L. magnus* and *L. striatus* coexist, with no evidence of intense competition for resources (Goulder 1974a, 1980). This coexistence could be mediated by among-species differences in food resources; the larger *L. magnus* has a larger mouth and can therefore consume larger prey (Finlay & Berninger 1984).

11.3 Protozoan seasonal dynamics

Populations of benthic protozoa show marked temporal variation, with seasonal peaks in abundance that differ greatly in timing and magnitude from one year to the next (Goulder 1974b, 1980). For many of the common species of large ciliate, peak abundances in sediments are detected during periods when oxygen concentrations are high in the bottom waters of the lake. These changes in abundance do not necessarily imply mass mortality of all common benthic species under conditions of anoxia, as some species migrate from the sediment surface into the overlying water column when bottom waters become deoxygenated (Finlay 1981, Laybourn-Parry *et al.* 1990b). Due to these migrations, these organisms simply cannot be captured in great numbers in sediments during the anoxic period. An intensive survey of the wider protozoan community in the sediments suggested that abundances of flagellates and amoebae vary rather less than those of large ciliates during deoxygenation, and the subsequent return of high oxygen concentrations when the lake mixes in the autumn (Bark 1981).

Due to the migrations of benthic ciliates, it is possible to distinguish two communities in the water column of Esthwaite Water; a truly planktonic assemblage and a collection of benthic migrants (Laybourn-Parry *et al.* 1990b). Planktonic ciliate populations show equally pronounced seasonal variations in abundance (Bark 1981, Laybourn-Parry *et al.* 1990a), presumably as a result of temporal changes in the physical and chemical attributes of the water column. Total planktonic ciliate numbers typically peak in the spring-early summer after a period of very rapid population growth, with constituent species differing in the precise timing of their maximal abundances (Laybourn-Parry *et al.* 1990a). Some taxa are free-swimming whilst others, such as *Vorticella*, live attached to phytoplankton cells. When in the water column, protozoans can be important grazers and have the potential to bring about collapses in populations of phytoplankton. For example Canter & Lund (1968) observed that *Pseudospora* can infest colonies of some phytoplankton species and ingest the constituent cells. Observations from Esthwaite Water indicated that populations of prey species frequently collapsed after these protozoans were first detected feeding upon them.

11.4 Protozoan spatial distribution

The highest abundances of benthic ciliates are found in flocculent material at the sediment-water interface (Webb 1961, Goulder 1971, 1980) while flagellates and amoebae can be found deeper, at a depth of 8 cm within the sediment (Bark 1981). It has been suggested that surface sediments are a favourable habitat due to the presence of interstitial spaces that can accommodate the ciliate fauna, and the likely higher food abundances at the sediment surface (Goulder 1971, 1980). There is a high degree of spatial heterogeneity in abundance across the sediment surface. Spatial variations in sediment density and food quality/quantity are believed to account for observations of higher abundances of ciliates at the surface of deep sediments (Goulder 1974b, 1980). As this flocculent material is readily disturbed by water currents, aggregations of protozoa can be broken up under windy conditions or after storm events (Webb 1961).

Water column ciliate communities show marked spatial variations in association with patterns of thermal stratification and correlated vertical variations in oxygen concentrations. During anoxia-driven seasonal migrations from the sediment surface to the overlying water, many large ciliates (>150 µm) aggregate around the vertical oxygen gradient (Bark 1981, Finlay 1981). However, these species are still found throughout the anoxic zone, perhaps indicating that they migrate periodically between the sediment and oxygenated waters, spending much of this time in anoxic waters. Whilst some taxa (*Brachonella, Caenomorpha, Metopus*) appear to be restricted to anoxic bottom waters, others (*Vorticella, Epistylis*) are equally restricted to oxygenated surface waters (Bark 1981). There is also evidence that, when in the water column, some ciliate species aggregate around vertical layers of high bacterial abundance (Guhl *et al.* 1996) or are associated with high abundances of flagellates (Laybourn-Parry *et al.* 1990a), both important food resources.

11.5 Zooplankton

In an earlier section of this review, existing knowledge on the protozoan fauna of Esthwaite Water was summarised. The lake also supports a diverse community of multi-cellular zooplankton, which were studied intensively in the 1950s-1960s. These organisms live either suspended in the open water of the lake, or in association with sediments and vegetation. They occupy an important intermediate position in the aquatic food web; they consume phytoplankton (and each other), and are consumed by fish and larger invertebrates. Therefore, the zooplankton are an important conduit for the transfer of energy and materials through the lake food web. Research has focussed upon spatial and temporal patterns in the resident populations and has yielded much useful baseline information to which the results of contemporary and future studies could be compared.

11.6 Zooplankton species composition

Esthwaite Water supports populations of cladocerans, copepods and rotifers. The available literature provides a more exhaustive listing of the taxonomic composition of the planktonic crustacea (cladocera and copepods), than of the rotifers. Much of the existing information on the species composition of the planktonic crustacea of Esthwaite Water originates from samples collected in the 1920s (Gurney 1923), 1950s and 1960s (Goulden 1964, Smyly 1968a, Smyly 1972). Comparisons of community structure over the time period encompassed by these early surveys suggested that relatively little compositional change had occurred. A species list compiled from these surveys, and from a few other sources, is presented in Table 11.1. Only one species of calanoid copepod has been recorded in Esthwaite Water, and only two species of pelagic cyclopoid copepods (*Cyclops strenuus*, *Mesocyclops leuckarti*). The coexistence of the two cyclopoids in Esthwaite Water appears to be unusual for the Lake District lakes and may suggest that these species occupy different spatial and/or temporal niches as a way of avoiding competition or predator-prey interactions (Smyly 1978). Of the extensive list of cladocera recorded at the site, only a few were reported as numerically abundant (*Daphnia hyalina/galeata*, *Bosmina longirostris* and *Ceriodaphnia quadrangula*). Molecular genetic studies of extant populations of *Daphnia* in the lake and of resting eggs in sediment samples suggest that, while in the early surveys both *D. hyalina* and *D. galeata* coexisted in the lake, contemporary populations are dominated by *D. galeata* (Reid *et al.* 2000). Based upon a 1920s survey of many of the larger Lake District lakes (Gurney 1923), Esthwaite Water would appear to have had a comparatively diverse zooplankton community including a number of species that were considered to be "warm water" taxa (*Diaphanosoma brachyurum*, *Ceriodaphnia* spp., *Mesocyclops leuckarti*). Collections of crustacean fauna in the open waters of the lake typically also contain small numbers of, strictly, non-planktonic species that are associated more closely with littoral or benthic habitats (*Ceriodaphnia megalops*, *Polyphemus pediculus*, *Sida crystallina*).

Table 11.1. Zooplankton species recorded in Esthwaite Water based upon Edmondson (1965), Galliford (1948), Galliford (1950), Gurney (1923), Goulden (1964), Harding & Smith (1974), Reid et al*. (2000), Ruttner-Kolisko (1989), Scourfield & Harding (1966), Smyly (1968a), Smyly (1972) and Tinson & Laybourn-Parry (1985). Species marked with asterisks are those believed to be more closely associated with littoral and benthic habitats.*

Rotifera	*Kellicottia longispina*
Asplanchna priodonta	*Notholca foliacea*
Chromogaster ovalis	*Notholca longispina*
Chromogaster testudo	*Notholca striata*
Collotheca mutabilis	*Polyarthra trigla*
Collotheca libera	*Polyarthra vulgaris*

Collotheca pelagica	*Synchaeta kitina*
Conochilus unicornis	*Synchaeta oblonga*
Conochiloides dossuarius	*Synchaeta pectinata*
Filinia terminalis	*Trichocerca capucina*
Gastropus stylifer	*Trichocerca porcellus*
Keratella cochlearis	*Trichocerca similis*
Keratella quadrata	
Cladocera	*Ceriodaphnia quadrangula*
*Alona affinis**	*Chydorus piger**
*Alona guttata**	*Chydorus sphaericus*
*Alona rectangula**	*Daphnia hyalina*
*Alonella excisa**	*Daphnia galeata*
*Alonella exigua**	*Diaphanosoma brachyurum*
*Alonella nana**	*Eurycercus lamellatus**
*Alonopsis elongata**	*Graptoleberis testudinaria**
Bosmina coregoni	*Leptodora kindtii*
Bosmina longirostris	*Peracantha truncata**
Bythotrephes longimanus	*Polyphemus pediculus**
*Ceriodaphnia megalops**	*Scapholeberis mucronata**
Ceriodaphnia pulchella	*Sida crystallina**
Cyclopoida	*Eucyclops agilis**
*Acanthocyclops viridis**	*Eucyclops macruroides**
*Cyclops dybowskii**	*Eucyclops macrurus**
*Cyclops fuscus**	*Macrocyclops albidus**
Cyclops strenuus	*Mesocyclops leuckarti*
*Diacyclops bicuspidatus**	*Paracyclops fimbriatus**
Calanoida	**Harpacticoida**
Eudiaptomus gracilis	*Canthocamptus staphylinus**

11.7 Zooplankton temporal dynamics

Most information on the temporal dynamics of the zooplankton of Esthwaite Water derives from a period of monitoring in the 1950s, 1960s and early 1970s. As in other temperate lakes, there are pronounced seasonal variations in the crustacean zooplankton community. Furthermore, the dominant species differ markedly in their seasonal life history strategies. The numerically dominant cladocerans *Daphnia hyalina* and *Bosmina longirostris* and the rotifer *Filinia terminalis* typically show a single population maximum in spring (Smyly 1968a, George *et al.* 1990, Ruttner-Kolisko 1989, Talling 2003). For *Daphni*a this follows a period of overwintering as active individuals in the water column (Smyly 1979). In the case of *Daphnia*, additional summer population maxima occur in some years (Smyly 1979). These patterns are believed to be driven by bouts of rapid reproduction in response to seasonal "windows of opportunity" typified by pulses in the production of readily ingestible phytoplankton taxa during periods of high water temperature (George *et al.* 1990). At these times, the grazing pressure exerted by *Daphnia* populations may be considerable and is believed to contribute to seasonal declines in phytoplankton biomass (Talling 2003). These conditions of warm water and high abundances of smaller phytoplankton also promote population growth for the lakes rotifers (Edmondson 1965). In contrast to *Daphina* and *Bosmina*, the cladocerans *Diaphanosoma brachyurum* and *Ceriodaphnia quadrangula* produce defined summer-late summer maxima (Smyly 1968a, 1974, Heaney *et al.* 1986).

Life-cycles of the resident copepods are longer and more complex. Detailed studies of the omnivorous cyclopoid *Mesocyclops leuckarti* (Fryer & Smyly 1954, Smyly 1961, 1968a) have shown that the Esthwaite Water population typically produces two generations per year. In the spring, adults produce offspring that form the basis of a later summer adult population. The progeny of these summer adults enter a period of arrested development, lying dormant at the sediment-water interface over winter, before resuming development the following spring. *Cyclops strenuus*, however, typically produces one generation each year with the main period of reproduction in the spring, giving rise to progeny that overwinter as juveniles in a dormant state, before emerging to breed the following spring (Smyly 1973). The calanoid copepod *Eudiaptomus gracilis* does not typically show such rigid seasonality and adults may be abundant throughout the year. For this species breeding occurs throughout much of the year, though peak numbers of eggs are often produced in the spring (Smyly 1968b). A variable number of generations can be produced from one year to the next (Smyly 1968a).

At present our knowledge of interannual variations in the zooplankton community, and of long-term trends, is limited to a period up until the early 1990s. It would appear that there were few qualitative changes in the community structure (i.e. species presence/absence) of the crustacean zooplankton over this period (Gurney 1923, Smyly 1968a, George *et al.* 1990). However, interannual changes in the abundance of dominant taxa have been recorded during this time. A combination of paleaolimnological evidence and water column sampling showed an increasing abundance of *Daphnia* during the 1950s and 1960s, as the lake was becoming more enriched with limiting nutrients (Goulden 1964, Smyly 1972). At the same time, fewer ephippia were produced by these organisms, suggesting a switch in reproductive mode from sexual to asexual. It was believed that this was a response to nutrient enrichment but later studies did not show evidence for a trend of increasing abundance in winter and spring (George & Hewitt 1999, George *et al.* 2000). As yet, contemporary zooplankton samples have not been analysed to reveal whether a long-term trend in annual mean abundance is indeed apparent over a longer time scale.

As well as changing lake trophic status, weather patterns are thought to have a strong influence upon zooplankton dynamics. Weather-driven interannual variations in water temperature, water column structure and mixing are thought to drive year-to-year fluctuations in the abundance of *Daphnia* and *Eudiaptomus* by affecting the growth of phytoplankton food resources (George & Hewitt 1999, George *et al.* 2000). Predator-prey interactions may also have a role in shaping temporal dynamics. Smyly (1972) suggested that interannual variations in *Cyclops strenuus* abundance may be affected by the predation pressure exerted by the phantom midge larvae *Chaoborus*, which feeds on the larval stages of this species.

11.8 Zooplankton spatial distribution

The zooplankton of Esthwaite Water are rather patchily distributed throughout the open water zone. To date, research has focussed upon depth-related variations in zooplankton abundance, driven by seasonal changes in stratification and deep anoxia. Whilst zooplankton might be distributed throughout much of the water column during conditions of overturn in winter and early spring, most species appear to avoid the anoxic deep waters during the summer months, inhabiting the warmer food rich surface waters (Heaney *et al.* 1986, Thackeray *et al.* 2005). Species differ in their depth selection behaviour within these surface waters, with both *Eudiaptomus gracilis* and *Cyclops strenuus* occupying deeper strata than *Daphnia hyalina* (Thackeray *et al.* 2005). During calm conditions small-scale “swarms” of *Daphnia hyalina* have been observed in the surface waters, perhaps generated by behavioural responses of the animals to each other and the surrounding environment (George 1981).

Some species do inhabit deeper deoxygenated waters, though they do avoid complete anoxia. The small cladoceran *Ceriodaphnia quadrangula* is typically most abundant in the region of the thermocline and associated oxygen gradient, and has been observed in the oxygen depleted waters below (Heaney *et al.* 1986, Smyly 1974). Benthic copepods such as *Acanthocyclops viridis*, *Macrocyclops albidus*, *Eucyclops agilis*, *Paracyclops fimbriatus* and *Diacyclops bicuspidatu*s may also tolerate low oxygen concentrations until complete anoxia drives them to migrate into surface waters or laterally towards shallow, more oxygenated waters in the littoral zone (Laybourn-Parry & Strachan 1980, Tinson & Laybourn-Parry 1985). The rotifer *Filinia terminalis* has also been observed at very high population densities at the sediment-water interface during the early stages of seasonal oxygen depletion, before the onset of complete anoxia, perhaps as a result of mass emergence from benthic resting eggs (Ruttner-Kolisko 1980, 1989).

11.9 Long-term changes in zooplankton density

During the collection of phytoplankton for analysis of concentration of chlorophyll *a*, some zooplankton are trapped on the filter and their numbers are counted. The water for the chlorophyll analysis has been collected from 0 to 5 m but in the summer relatively small volumes of water are filtered so the estimates of zooplankton abundance are imprecise. Nevertheless, this provides a consistent method to assessing changes in zooplankton density. The data (Fig. 11.1) show relatively small changes in zooplankton density between 1967 and 2010. Peak abundance annual average density was recorded in 1987 followed by a decline to a low average density in 1998 (Fig. 11.1b). Densities between 2005 and 2010 have generally been low. The average seasonal pattern is a single large cohort centred around May (Fig. 11.1c). There have been statistically significant long-term reductions in density in June, July, October and November.

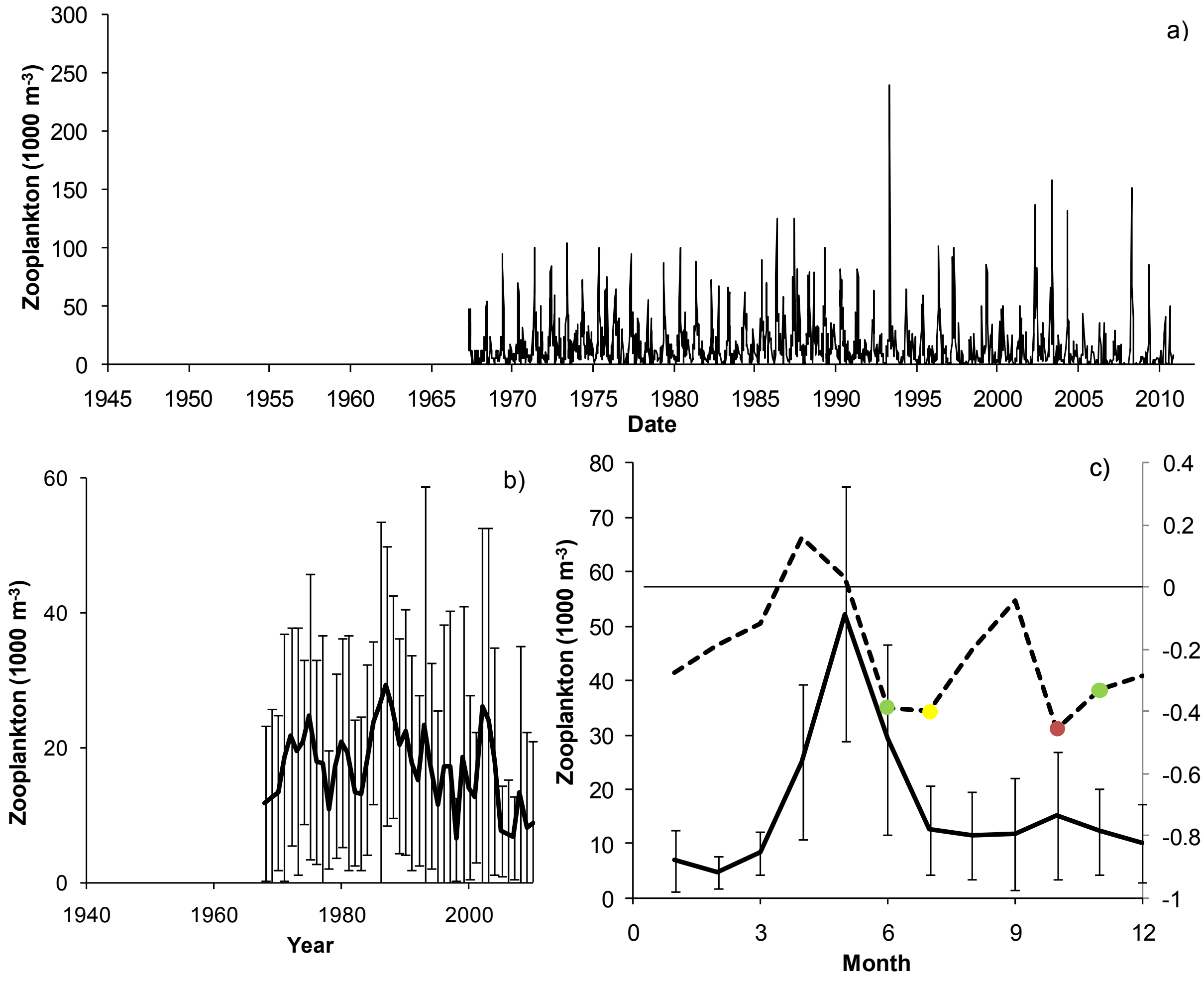

Figure 11.1. *Zooplankton density in Esthwaite Water. a) long-term record from 1967 to 2010; b) annual mean (error bars show standard deviation); c) monthly mean (error bars show standard deviation) plus long-term correlation of monthly change for annual data (dashed line), green circle P<0.05, yellow circle P<0.01, red circle P<0.001.*

12. Benthic invertebrates

12.1 Introduction

The benthic invertebrate community of Esthwaite Water has been studied over a number of decades. Much of the published work on these communities reports the results of faunistic surveys that aimed to provide information on the species that can be found in the lake, in a variety of benthic environments. As there are few studies detailing species interactions and temporal dynamics within the lake, this review has been structured in a way that best represents the existing literature; outlining findings on community composition in different major habitat types. Published species records are listed in Table 12.1. It should however be noted that the present review focusses only on published records and that there may, of course, be unpublished records that would serve to augment the species list that is presented.

12.2 Community composition on rocky shores & in vegetation

Within Esthwaite Water are a number of distinct habitat types, each of which is home to a variety of macroinvertebrate species. In shallow waters, invertebrate collections have been made within aquatic plant beds and on bare stony substrates, with the latter having been more intensively sampled. Beds of emergent *Phragmites* and *Carex* rooted in sediments rich in organic matter provide a habitat for corixids (Macan 1938, Frost & Macan 1948, Macan 1956). It is believed that, within this community, both *Sigara falleni* and *S. semistriata* are species that are indicative of the highly productive nature of the lake (Macan 1970). Submerged plant beds also accommodate various oligochaete worms (e.g. *Dero limosa*, *Nais pseudoobtusa*, *Stylaria lacustris*, *Cernosvitov* 1945), mayflies (e.g. *Siphlonurus lacustris*, *Caenis horaria*, Kimmins 1943, Macan 1970), the limpet *Acroloxus lacustris* (Geldiay 1956) and the leech *Helobdella stagnalis* (Mann 1955).

Stony substrata accommodate a diversity of fauna, including insects, crustacea, molluscs, oligochaete worms and flatworms (Cernosvitov 1945, Macan 1980). By comparing macroinvertebrate samples collected on these substrates in Esthwaite Water with samples from a number of other, less productive, Lake District lakes, Macan (1970, 1980) showed that the former supports higher numbers of organisms but a qualitatively different community that is poorer in insect species [e.g. Ephemeroptera (mayflies) and Plecoptera (stoneflies)] but richer in Crustacea and Platyhelminthes (flatworms). The stony substratum of the lake is also inhabited by the freshwater limpet *Ancylus fluviatilis* (Geldiay 1956) and leeches such as *Erpobdella octoculata* (Mann 1955). This habitat type may be preferable due to the presence of food organisms; epilithic algae in the case of *Ancylus* and chironomids/trichopterans in the case of *Erpobdella*. Stony substrates just above the water's edge are an important habitat for the leech *Haemopis sanguisuga* which feeds upon terrestrial organisms such as earthworms (Mann 1955), and is capable of swallowing smaller invertebrate prey whole (Elliott & Mann 1979). The other species of leech found in Esthwaite Water are known to demonstrate a variety of feeding habits (Elliott & Mann 1979); some are parasitic on fish (e.g. *Piscicola geometra*), water birds (e.g. *Theromyzon tessulatum*), molluscs (e.g. *Glossiphonia heteroclita*) and even other leeches (e.g. *Glossiphonia complanata*) whilst others are carnivores that prey on a range of invertebrates (e.g. *Erpobdella octoculata*).

A currently abundant member of the Crustacea, *Asellus aquaticus*, is believed to have invaded Esthwaite Water after the mid 1940s (Moon 1968). Specimens of *Asellus* were first collected in the *Phragmites* beds at the north end of the lake in 1944 and, over a period of approximately 18 years, this species dispersed around the shoreline of the whole lake. While it was speculated that anthropogenic modification of the lake ecosystem, or human activities in the catchment, were at the root of this invasion the precise mechanisms behind the colonisation remain largely unknown.

Table 12.1. Macroinvertebrate species recorded in Esthwaite Water according to Cernosvitov (1945), Brinkhurst (1964), Elliott (1995), Fryer (1991), Geldiay (1956), Grey et al. *(2004a), Kimmins (1943, 1972), Macan (1938, 1950, 1955, 1956, 1970, 1977, 1980), Mann (1955), Reynoldson (1978, 1990), Smyly (1972) and records collected between 1999-2005 and held by the Environmental Change Network.*

Alderflies	
Sialis lutaria	
Beetles	*Haliplus ruficollis*
Agabus spp.	*Hydraena gracilis*
Elmis aenea	*Hydraena riparia*
Esolus parallelepipedus	Hydrophilidae
Ilybius spp.	*Nebrioporus depressus*
Haliplus lineatocollis	*Noterus clavicornis*
Haliplus lineolatus	*Oulimnius tuberculatus*
Caddis flies	*Limnephilus extricatus*
Agapetus fuscipes	*Limnephilus lunatus*
Agraylea multipunctata	*Limnephilus marmoratus*
Agrypnia spp.	*Molanna angustata*
Anabolia nervosa	*Mystacides azurea*
Chaetopteryx villosa	*Polycentropus flavomaculatus*
Cyrnus trimaculatus	*Tinodes waeneri*
Halesus spp.	
Damselflies	
Enallagma cyathigerum	
Flies	*Microtendipes chloris*
Ablabesmyia monilus	*Microtendipes pedellus*
Ceratopogonidae	*Parachironomus baciliger*
Chaoborus spp.	*Pericoma* spp.
Chironomus anthracinus	*Polypedilum* spp.
Chironomus plumosus	*Procladius* spp.
Cladotanytarsus atridorsum	*Tanytarsus glabrescens*
Cladotanytarsus mancus	*Tanytarsus holochlorus*
Corynoneura lacustris	*Tanytarsus lugens*
Dixella spp.	*Tanytarsus samboni*
Endochironomus albipennis	Tipulidae
Mayflies	*Electrogena lateralis*

Baetis rhodani	*Ephemera danica*
Caenis horaria	*Leptophlebia marginata*
Caenis luctuosa	*Leptophlebia vespertina*
Centroptilum luteolum	*Paraleptophlebia submarginata*
Cloeon dipterum	*Serratella ignita*
Cloeon simile	*Siphlonurus lacustris*
Ecdyonurus dispar	
Stoneflies	*Leuctra hippopus*
Chloroperla torrentium	*Nemoura avicularis*
Water bugs	*Sigara distincta*
Callicorixa praeusta	*Sigara dorsalis*
Corixa dentipes	*Sigara falleni*
Corixa punctata	*Sigara fossarum*
Cymatia bonsdorffi	*Sigara lateralis*
Hesperocorixa linnei	*Sigara semistriata*
Hesperocorixa sahlbergi	*Sigara striata*
Notonecta glauca	*Sigara scotti*
Crustaceans	*Crangonyx pseudogracilis*
Asellus aquaticus	*Gammarus pulex*
Molluscs	*Planorbis contortus*
Acroloxus lacustris	*Potamopyrgus antipodarum*
Ancylus fluviatilis	*Radix auricularia*
Bathyomphalus contortus	*Radix balthica*
Hippeutis complanatus	Sphaeriidae
Lymnaea palustris	*Stagnicola palustris*
Lymnaea peregra	*Valvata piscinalis*
Physa fontinalis	*Valvata cristata*
Planorbis albus	
Leeches	*Glossiphonia heteroclita*
Batracobdella paludosa	*Haemopis sanguisuga*
Dina lineata	*Helobdella stagnalis*
Erpobdella octoculata	*Hemiclepsis marginata*
Erpobdella testacea	*Piscicola geometra*
Glossiphonia complanata	*Theromyzon tessulatum*
Flatworms	*Dugesia polychroa*.
Bdellocephala punctata	*Polycelis nigra*
Dendrocoelum lacteum	*Polycelis tenuis*
Dugesia lugubris	

Oligochaete worms	*Nais variabilis*
Aulodrius pluriseta	*Potamothrix hammoniensis*
Dero limosa	*Stylaria lacustris*
Limnodrilus hoffmeisteri	*Stylodrilus heringianus*
Lumbricidae	*Tubifex tubifex*
Nais pseudoobtusa	
Other groups	Nematoda
Hydridae	Ostracoda
Hydracarina	

12.3 Community composition in deep sediments

Soft sediments below the lake are home to dense communities of oligochaete worms, dominated by a different assemblage of species to those found in submerged plant beds in shallow waters. *Limnodrilus hoffmeisteri, Potamothrix hammoniensis* and *Tubifex tubifex* are all common species in deep sediments, the latter being widespread across the Lake District (Brinkhurst 1964, Reynoldson 1990). It is likely that growth and reproduction of the oligochaetes is restricted to the spring and autumn in Esthwaite Water, when deep waters are sufficiently warm but not deoxygenated (Reynoldson 1987). Abundances of the worms show significant spatial variation across the sediment surface. The abundant chironomid larvae in the deepest sediments (Mundie 1956, Grey *et al.* 2004a) also show marked spatial heterogeneity, and were particularly numerous below the cages of the commercial fish farm (Grey *at al.* 2004b). There is also an inshore-offshore gradient in the abundance and diversity of chironomid larvae with maximum densities and more species being found in the structurally diverse substrates of the shallower littoral zone, than in the more homogenous deep profundal zone (Fryer 1991). Both oligochaetes and chironomid larvae are likely to feed on bacteria in sediments and organic material that has settled from overlying waters.

Studies of invertebrates living in the deep sediments of Esthwaite Water have yielded significant insights into patterns in the transfer of energy and matter through the lake food web. A particular focus has been on elucidating less well known sources of nutrition to macroinvertebrates;

methane-consuming bacterial communities and materials derived from food pellets given to rainbow trout at the fish farm that was present in the lake until recently. While benthic chironomid larvae may depend upon water column production for much of the year, feeding upon sinking phytoplankton and detritus, they show a seasonal shift in their use of food resources and ingest methane-consuming bacteria in surface sediments during periods of anoxia and subsequent overturn (Grey *et al.* 2004a, Grey *et al.* 2004c, Deines *et al.* 2009). However, below the fish farm, chironomids appeared to derive a significant proportion of their food resources from sedimenting food pellets given to the rainbow trout confined in the cages (Grey *et al.* 2004b). The nutrition provided by these pellets was not restricted to the chironomids and also appeared to “fuel” many additional food web components, including zooplankton and roach.

13. Vertebrates

13.1 Introduction

In contrast to the preceding subjects of this review, the vertebrates of Esthwaite Water and its immediate environment have been subjected to little scientific study. Consequently, in addition to the limited relevant primary scientific literature that does exist, the following sections have also drawn on a number of secondary and semi-popular articles, and on unpublished data holdings. Key amongst these are the publications of Horne & Horne (1985), Frost (1989), Talling (1999) and Armsby (2010), together with unpublished data held by the Cumbria Biodiversity Centre (CBDC), Cumbria Wildlife Trust (CWT) and the Environment Agency (EA).

13.2 Fish

In contrast to the long-standing and extensive studies of the fish populations of nearby Windermere (reviewed, for example, by Le Cren (2001)), those of Esthwaite Water and its immediate environment have received only very limited and intermittent scientific attention. One of the most authoritative, but still brief, descriptions of the fish community of this lake is that provided by Frost (1989), who noted that brown trout (*Salmo trutta*), perch (*Perca fluviatilis*) and pike (*Esox lucius*) were the main species present, with Atlantic salmon (*Salmo salar*) passing through on migration and with the cyprinids roach (*Rutilus rutilus*) and rudd (*Scardinius erythrophthalmus*) and their hybrid also present. In addition, the same author observed that roach and rudd also occurred in Priest (or Priest's) Pot, a small pond at the north end of the lake. A morphological study of roach, rudd and their hybrid in Esthwaite Water was provided by Wheeler (1976).

Taking into account the, with the exception of minnow (*Phoxinus phoxinus*), very restricted distribution of cyprinids in the Lake District as a whole where the then still limited populations had occurred 'within the past ninety years', Frost (1989) concluded that the apparently recent appearance of such populations was the result of live-baiting activities by anglers. Somewhat in contrast, Le Cren *et al.* (1972) suggested that this local occurrence of cyprinids may be related to the fact that the fisheries of Esthwaite Water were historically owned by monks, who have a long history of cyprinid cultivation and stocking. Le Cren *et al.* (1972) commented specifically on the locally unusual occurrence of roach and rudd in Esthwaite Water and Priest Pot and, given the potential for downstream emigration from the former by the connecting Cunsey Beck, expressed some surprise that roach, at least, was not a more prominent feature of the Windermere fish community at the time of writing. Smyly (1957) recorded the presence of bullhead (*Cottus gobio*) in the Esthwaite Water catchment, although this species is unlikely to inhabit the lake itself. Finally in terms of a historical perspective on the Esthwaite Water fish community, Pennington & Frost (1961) found salmonid vertebrae and scales within a sediment core taken from the lake. Although the species from which the materials originated could not be identified with certainty, it did appear to be either brown trout or Arctic charr (*Salvelinus alpinus*). As there is no historical evidence for the occurrence of Arctic charr in the lake, nor for that matter of the coregonids vendace (*Coregonus albula*) and schelly (*Coregonus lavaretus*), it seems most likely that the materials originated from brown trout.

Two further species were noted in the Esthwaite Water fish community a review by Talling (1999), i.e. stone loach (*Barbatula barbatula*), which extends some way into the lake from an inflow where its biology has been studied (Smyly, 1955), and eel (*Anguilla anguilla*). With respect to the latter species, Talling (1999) made particular reference to 1940s studies of eel local growth (Frost, 1945), diet (Frost, 1946) and diel migratory behaviour (Lowe, 1952). The migration of maturing eel from Esthwaite Water along Cunsey Beck to Windermere was also briefly considered by Bagenal (1970).

The fish community of Priest Pot has itself also received some attention, with Frost & Le Cren (1957) providing brief comment on the local occurrence of rudd and Bagenal (1974) and Hewitt (1979) using this small pond as a location to develop a method for the quantitative sampling of larval roach, rudd and their hybrids. Venugopal & Winfield (1993) investigated the spatial distributions of juvenile roach, rudd, roach-rudd hybrids and perch with respect to macrophyte beds around the pond's perimeter, and also noted the presence of pike and the probable local occurrence of eel.

Talling (1999) also described the establishment in 1981 of a cage-based fish farm for rainbow trout (*Oncorhynchus mykiss*) near the outflow of Esthwaite Water. In addition to producing fish for the market, a component of annual production was used to stock the lake in a successful put-and-take fishery which subsequently developed a national reputation amongst the trout angling community. The lake is also frequently visited, particularly during the winter months, by anglers seeking specimen pike which have probably benefitted from the local stocking activities. However, for many years concerns have been expressed over nutrient loading to the lake from the fish farm. In addition to general eutrophication effects considered elsewhere in this review, Grey *et al.* (2004b) used stable isotope analyses to study the fate of waste pelleted food and tentatively concluded that roach were probably partly short-circuiting the lake's food chain by directly consuming significant amounts of particulate pellet material, such that over 80% of their body carbon may be ultimately derived from pellet material. In addition to these wider environmental concerns, in the early 2000s the fish farm's stocking activities became incompatible with the then newly-developed Environment Agency National Trout & Grayling Fisheries Strategy (Environment Agency, 2003). Following extensive discussions, in early 2009 the owner of Hawkshead Trout Farm agreed to remove the cages as part of a voluntary agreement with Natural England. The agreement reached saw the removal of the cages in late 2009, with Esthwaite Water planned to become an all brown trout fishery by 2013. The shift from stocking with rainbow trout to stocking with brown trout will be a gradual transition and the 2012 season will see the last stockings of rainbow trout. The stocked brown trout will be high quality triploids. In 2015, a fish screen at the outflow of Esthwaite Water originally installed to retain stocked rainbow trout will be removed to allow the free passage of native salmonids. In addition, Natural England and the Lake District National Park Authority have agreed to work with the fishery to ensure it remains a successful rural business.

Finally, the most recent information on the fish of Esthwaite Water and its immediate environment is that provided by unpublished electrofishing surveys of its streams made between 2005 and 2010 by EA and an unpublished gill-net survey of the lake itself conducted jointly on 23 September 2009 by EA and the Centre for Ecology & Hydrology (CEH) (EA and EA/CEH, unpublished data). In 2005, electrofishing at a site (Hawkshead) in Black Beck recorded brown trout fry at a population density of 16.96 individuals 100 m^{-2} and brown trout parr at a population density of 12.19 individuals 100 m^{-2}, together with non-quantified numbers of eel, minnow, perch, stone loach and rainbow trout. In 2008 and 2010, electrofishing at three sites in Cunsey Beck (Cunsey Bridge, Ees Bridge and Eel House Bridge) produced brown trout, eel, minnow, perch, pike, stone loach and roach. The brown trout were present as fry at population densities between 0.94 and 6.77 individuals 100 m^{-2} and as parr at population densities between 0.50 and 7.65 individuals 100 m^{-2}. The gill-net survey comprised single survey gill nets set at inshore, offshore surface and offshore bottom locations and resulted in the sampling of a total of 192 fish, comprising 6 brown trout (fork length range 290 to 380 mm), 124 perch (fork length range 45 to 310 mm), 6 rainbow trout (fork length range 370 to 415 mm) and 56 roach (fork length range 50 to 325 mm). The inshore gill net contained 112 perch and 36 roach, while the offshore surface net contained 6 brown trout, 12 perch, 6 rainbow trout and 20 roach. The offshore bottom gill net, which was set in a typical offshore area of low oxygen availability at depth, did not catch any fish.

13.3 Amphibians

CWT data from 1984 to 1999 covering 'significant species' recorded at Esthwaite Water and its immediate environment include no records for amphibians (C. Cornish, CWT, *pers. comm.*). However, the habitats offered by Esthwaite Water and its immediate environment are very likely to be good for such species, with common frog (*Rana temporaria*) and common toad (*Bufo bufo*) almost certainly present (S. Griffin, Hesketh Ecology, *pers. comm.*).

13.4 Reptiles

CWT data from 1984 to 1999 covering 'significant species' recorded at Esthwaite Water and its immediate environment include no records for reptiles (C. Cornish, CWT, *pers. comm.*). However, the habitats offered by Esthwaite Water and its immediate environment are very likely to be good for such species and records of grass snake (*Natrix natrix*) do exist for the general area (S. Griffin, Hesketh Ecology, *pers. comm.*). Furthermore, adder (*Vipera berus*), common lizard (*Lacerta vivipara*) and slow worm (*Anguis fragilis*) may be expected in suitable habitat and CWT will be co-ordinating a grass snake survey of Cumbria in 2011 which may also generate information on other reptile species (S. Griffin, Hesketh Ecology, *pers. comm.*).

13.5 Birds

Very old records exist for crossbills (*Loxia curvirostra*) (Armitt, 1894a) and pied flycatcher (*Ficedula hypoleuca*) and wood warbler (*Phylloscopus sibilatrix*) (Armitt, 1894b) in the vicinity of Esthwaite Water, while the roosting of house martins (*Delichon urbica*) in a reed bed at the lake was observed by Airey (1951). Atkinson (1977) briefly reviewed the ornithological importance of Esthwaite Water's waterfowl in national and international contexts, while Delaney (1993) reported the local presence of pink-footed goose (*Anser brachyrhynchus*) in the summer of 1991. CWT data from 1984 to 1999 covering 'significant species'of birds recorded at Esthwaite Water and its immediate environment include the cuckoo (*Cuculus canorus*), curlew (*Numenius arquata*), dunnock (*Prunella modularis*) and tree pipit (*Anthus trivialis*) (C. Cornish, CWT, *pers. comm.*). CBDC records for birds specifically at Esthwaite Water comprise bittern (*Botaurus stellaris*), dipper (*Cinclus cinclus*), golden eagle (*Aquila chrysaetos*), great crested grebe (*Podiceps cristatus*), mallard (*Anas platyrhynchos*), mute swan (*Cygnus olor*), pochard (*Aythya ferina*), teal (*Anas crecca*) and whooper swan (*Cygnus cygnus*) (M. Grose, CBDC, *pers.*

comm.). Species monitored by the Wetland Bird Survey from 1960 to the present (D. Shackleton, Cumbria Bird Club, *pers. comm.*) include as regular records, great crested grebe, grey heron (*Ardea cinerea*), greylag goose (*Anser anser*), lapwing (*Vanellus vanellus*), lesser black-backed gull (*Larus fuscus*), little grebe (*Tachybaptus ruficollis*), mallard, moorhen (*Gallinula chloropus*), mute swan, oystercatcher (*Haematopus ostralegus*), pochard, teal and tufted duck (*Aythya fuligula*), as sporadic species they include common sandpiper (*Actitis hypoleucos*), curlew, great black-backed gull (*Larus marinus*), pink-footed goose (*Anser brachyrhynchus*), and red-breasted merganser (*Mergus serrator*), and as rare species they include barnacle goose (*Branta leucopsis*), gadwall (*Anas strepera*), herring gull (*Larus argentatus*), kingfisher (*Alcedo atthis*), long-tailed duck (*Clangula hyemalis*), shelduck (*Tadorna tadorna*), snipe (*Gallinago gallinago*), water rail (*Rallus aquaticus*), white-fronted goose (*Anser albifrons*), whooper swan and wigeon (*Anas penelope*). Finally, an osprey (*Pandion haliaetus*) has been observed hunting at Esthwaite Water (S.C. Maberly, *pers. obs.*).

Talling (1999) provides a useful summary of information on Esthwaite Water's water birds, for which the lake is designated as an internationally important Ramsar site, based on records of winter and summer birds maintained since 1967. Winter duck numbers have increased since 1981, when fish farming began and, presumably coincidentally, the lake level was lowered by approximately 0.5 m. This trend has been particularly marked for tufted duck, which Talling (*op. cit.*) suggested was the result of birds benefitting from an increase in food supply associated with the operations of the fish farm. Cormorants have also increased and individuals now frequent the lake all year round, whereas in earlier years they were usually only winter visitors. Finally, Talling (*op. cit.*) comments that great crested grebes have one of their Cumbrian strongholds on Esthwaite Water.

13.6 Mammals

CWT data from 1984 to 1999 covering ‘significant species’ recorded at Esthwaite Water and its immediate environment include the mammals otter (*Lutra lutra*), pipistrelle bat (*Pipistrellus pipistrellus*) and red squirrel (*Sciurus vulgaris*) (C. Cornish, CWT, *pers. comm.*). CBDC records for mammals specifically at Esthwaite Water include American mink (*Mustela vison*), coypu (*Myocaster coypus*) and red squirrel, with otter reported for nearby Priest Pot (M. Grose, CBDC, *pers. comm.*). In addition, Ellison (1968) also makes reference to the presence of coypu at Esthwaite Water.

14. Weather & climate change

14.1 Introduction

Palaeolimnological records have shown the general sensitivity of Esthwaite Water to broad changes in climate (e.g. Dong *et al.* 2011a, Dong *et al.* 2011b) and analysis of historical records for lakes around Europe also show that lakes are generally sensitive to climate (George 2010). Part of the sensitivity derives from the potential for short-term weather events to have a long-term effect on a lake. For example, in a stratified lake, a period of cold and windy weather can break down, or markedly reduce, summer stratification with major consequences for the light climate, mixing nutrients (including CO_2) regenerated at depth into the epilimnion and resetting the succession of phytoplankton back to a composition more typical of late spring (Reynolds 1997).

14.2 Regional weather patterns

Analysis of lake response to weather patterns gives insights into how lakes may respond to climate change. Esthwaite Water has been shown to respond to a variety of regional weather patterns such as the North Atlantic Oscillation Index (NAOI). This primarily affects winter weather in western Europe. When the NAOI is positive, there is a strong flow of air from the Atlantic bringing mild, wet and windy winters. In contrast, in years with a negative NAOI, there is a stronger flow of air from continental Europe bringing cold, dry and relatively calm winter weather. Long-term data (de-trended to remove long-term changes) from Esthwaite Water as well as nearby Blelham Tarn and the two basins of Windermere were analysed by George *et al.* (2004). Water temperature was strongly positively correlated with the NAOI in all four lake basins as a result of the changing air-temperature. Winter nitrate concentration was strongly negatively correlated with the NAOI in all four basins, and was also linked to winter air-temperature, possibly indicating lower uptake by the catchment during cold winters. Winter phosphate responded differently to the NAOI in the four basins. There was no significant effect in the two basins of Windermere but in Esthwaite Water and Blelham Tarn concentrations were higher in a positive NAOI winter and the response was positively correlated with rainfall. Winter concentrations of phytoplankton chlorophyll *a* were also affected by the NAOI in the two smaller basins and the driving meteorological link appeared to be high rainfall diluting winter phytoplankton populations. The sensitivity of Esthwaite Water and Blelham Tarn to rainfall was linked to their shorter average retention time, (91 and 42 days respectively) compared to the North and South Basins of Windermere with average retention times of 100 and 185 days respectively. This example shows that different types of lakes will be sensitive to different aspects of changing weather and climate: Esthwaite Water will be sensitive to changes in rainfall affecting flushing rate. George (2000) showed that the winter abundance of the calanoid copepod *Eudiaptomus gracilis* was linked to the NAOI via an effect on water temperature with higher abundance in warmer winters where the NAOI was positive. The proposed mechanisms is that warm winters disfavour competing *Daphnia* as they cannot obtain enough food to support their metabolism.

Linked to the NAOI is the position of the Gulf Stream in the north Atlantic. George & Taylor (1995) showed that the position of the north-wall of the Gulf Stream had a teleconnexion via weather patterns that influenced stratification and zooplankton strength in Windermere. George (2000) showed that the summer abundance of *Daphnia* is also affected by the position of the Gulf-Stream in Esthwaite Water. George (2002) showed that the position of the Gulf Stream also affected the magnitude of the summer concentration of chlorophyll *a* in Esthwaite Water. This correlation is caused by the effect of the Gulf Stream on the depth of the thermocline. Years where the thermocline is deep are associated with higher concentrations of phosphate entrained from the hypolimnion (Section 8.9) and hence higher phytoplankton biomass.

Another weather pattern, but of generally shorter duration, that also can control lake characteristics is the pressure distribution schemes devised by Lamb (1972). There is a strong relationship between surface temperature and Lamb weather patterns (George *et al.* 2010a) and nitrate concentrations and Lamb weather patterns (George *et al.* 2010b).

Recently, a larger-scale weather phenomenon linked to the position of the jet stream, Rossby breaking waves, has been shown to strongly control the surface temperature of the Cumbrian lakes, including Esthwaite Water (Strong & Maberly 2011). Unlike the NAOI, this effect operates in all seasons. It is probably the factor that controls the NAOI and the position of the Gulf Stream.

As mentioned above, short term changes in weather conditions can alter lake conditions and phytoplankton biomass. They have also been shown to alter phytoplankton species richness. Madgwick *et al*. (2006), showed that species richness was negatively correlated with the strength of stratification, estimated as the Schmidt stability.

14.3 Long-term change linked to climate

One of the most frequently observed biological responses to climate change is an altered phenology, the seasonal timing of life-history events. Thackeray *et al*. (2010) showed that phenological change is widespread in marine, freshwater and terrestrial environments in spring and summer. In freshwaters across all taxa, the average rate of advancement from 1976 to 2005 has been about 0.25 day y^{-1} for phytoplankton, 0.35 day y^{-1} for invertebrates and 0.45 day y^{-1} for vertebrates. The difference in rate of advance leads to the possibility of 'trophic-mismatch', i.e. production of a consumer population before or after its food is fully available. While the high-level of consistency in advancement across disparate organisms and environments is consistent with the effect of large-scale drivers, such as climate change, not all the changes appear to be linked to climate. For example, Thackeray *et al*. (2008) showed that of two spring-blooming diatoms in Windermere, earlier growth of *Cyclotella* spp. was linked to water temperature and earlier stratification while earlier growth of *Asterionella formosa* was linked to phosphorus enrichment and lake warming. This conclusion has been reinforced by recent work, including phytoplankton from Esthwaite Water. Feuchtmayr *et al*. (2011).

14.4 Responses to future climates

Currently, forecasts of future UK weather have been generated by UKCP09 http://ukclimateprojections.defra.gov.uk/content/view/12/689/. For 2050 and the medium greenhouse gas emissions scenario, the north west of England is forecast to have warmer air temperatures in winter and summer, by about 1^{o}C, slightly wetter winters (up to a 10% increase on current conditions) and substantially drier summers (about 30% decrease on current conditions).

George *et al.* (2007) combined data from the long-term measurements on Esthwaite Water with outputs from Regional Climate Models, produced before the UKCP09 outputs mentioned above, but with similar general trends, again using 2050 as the date in the future. Average water temperature in Esthwaite Water was forecast to increase in the winter from 4.00 ^{o}C in 1961 – 1990 to 5.09 ^{o}C in 2050 (an increase of 1.09 ^{o}C) and in the summer from 17.2 ^{o}C to 19.37 ^{o}C (an increase of 2.17 ^{o}C). Retention time in winter was projected to decrease from 63.4 days in 1961 – 1990 to 58.4 days in 2050 (an increase of 5 days) and in summer to increase from 151.6 to 156.0 days (a reduction of 4.4 days).

A process-based model, PROTECH (Elliott *et al*. 2010) has recently been applied to Esthwaite Water to investigate the likely effects of future changes in retention time and water temperature on the production of phytoplankton and, especially, cyanobacteria (Elliott 2010, 2011). The model suggested that the vernal bloom, dominated by the diatom *Asterionella formosa*, was relatively unaffected by the two variables. This is perhaps unsurprising as light availability is a key factor determining growth rate and success at the start of the year and daylength is constant and cloud cover was not modelled to vary (and is unlikely to vary- see Section 2.2). In contrast, in the summer two genera of cyanobacteria, *Anabaena* and *Aphanizomenon* responded to the climate-change drivers. The combination of low flow and higher temperatures, in particular, promoted the abundance of these species, both in absolute terms and as a proportion of the total phytoplankton biomass. Part of the ecological advantage of *Anabaena* and *Aphanizomenon* is that they are favoured by the strong stratification that often develops when surface water temperature is high, but also have the ability to fix atmospheric nitrogen and so are at an advantage in late summer if inorganic nitrogen as nitrate of ammonium is depleted (See Section 8.6). Another characteristic of some cyanobacteria, such as *Anabaena* and *Aphanizomenon* is the ability to produce toxins. The World Health Organisation (WHO) has set precautionary guidelines for 'safe' cyanobacterial levels at 10 mg chlorophyll *a* m^{-3}. The modelling work suggests that the threshold was exceeded for more days a year under low flow (Elliott 2010).

15. Lake management

15.1 Management change & lake response

In the United Kingdom, lake management has largely taken the approach of reducing input of nutrients from diffuse catchment sources (agricultural fertilizers and septic tanks) and point sources such as WwTWs and, in this lake, the fish farm. Much work has already been undertaken in the Esthwaite Water catchment to reduce nutrient load, the major points of which are detailed in Section 2.3.

United Utilities, and their predecessor North West Water, have instigated a number of changes to reduce the impact of nutrients from waste water on the lake. These include:

- introducing tertiary treatment at the Hawkshead WwTW in 1986 to reduce concentrations of phosphate in the final effluent to 1.5 g m^{-3} as an annual average
- redirecting the output from the Near Sawrey WwTW to Cunsey Beck below Esthwaite Water which removed a load of about 70 kg P y^{-1} to the lake;

United Utilities are in the process of implementing further improvement to their waste-handling procedures. These include:

- In March 2010, two small feeder pumping stations within the catchment at Esthwaite Lodge and Foldgate were upgraded. This is estimated to remove 4 kg P y^{-1}.
- Upgrading the tertiary treatment at the Hawkshead WwTW from a final effluent phosphate concentration of 1.5 g m^{-3} to 1 g m^{-3} as an annual average and to increase the capacity to treat high flows. This should be completed in early 2012.
- Upgrading the Hawkshead pumping station to transfer more flow to the WwTW. This should be completed in early 2012. Currently 90 kg P y^{-1} reaches the lake from the intermittent discharges resulting from low capacity from the pumping stations and the wwTW.

In addition, Natural England bought out the fish-farm activities of the trout farm located in the southern basin of the lake in autumn 2009. The enterprise where caged rainbow trout were reared in the lake and then sold for consumption has now ceased and with it the major phosphorus input from fish feed (see Section 5.3).

In a recent report to Natural England, Maberly *et al*. (2011) reviewed the water quality in Esthwaite Water in comparison to the previous decade. There is evidence for an improvement in water quality between 2008 and 2010 (Fig. 15.1). Concentrations of TP, SRP and nitrate were below the annual average for 2000 to 2009 in 2009 and 2010 and chlorophyll *a* concentrations and Secchi depth showed an improvement compared to the 2000 to 2009 average in 2009. While this is very welcome, it must be borne in mind that the long-term record shows that year-to-year variation, driven by variability in the weather can be substantial. Furthermore, it is not clear exactly what the cause of this apparent improvement might be. The chemical improvement pre-dates any management actions at the waste water handling plants and pre-dates the closure of the fish-farm. However Mr Nigel Woodhouse, the owner of the fish-farm, has mentioned that the fish-farm activities were being reduced from 2007 to 2009 so this could be cause of the improved water quality. Unfortunately, no records of fish-food application now exist to test this contention.

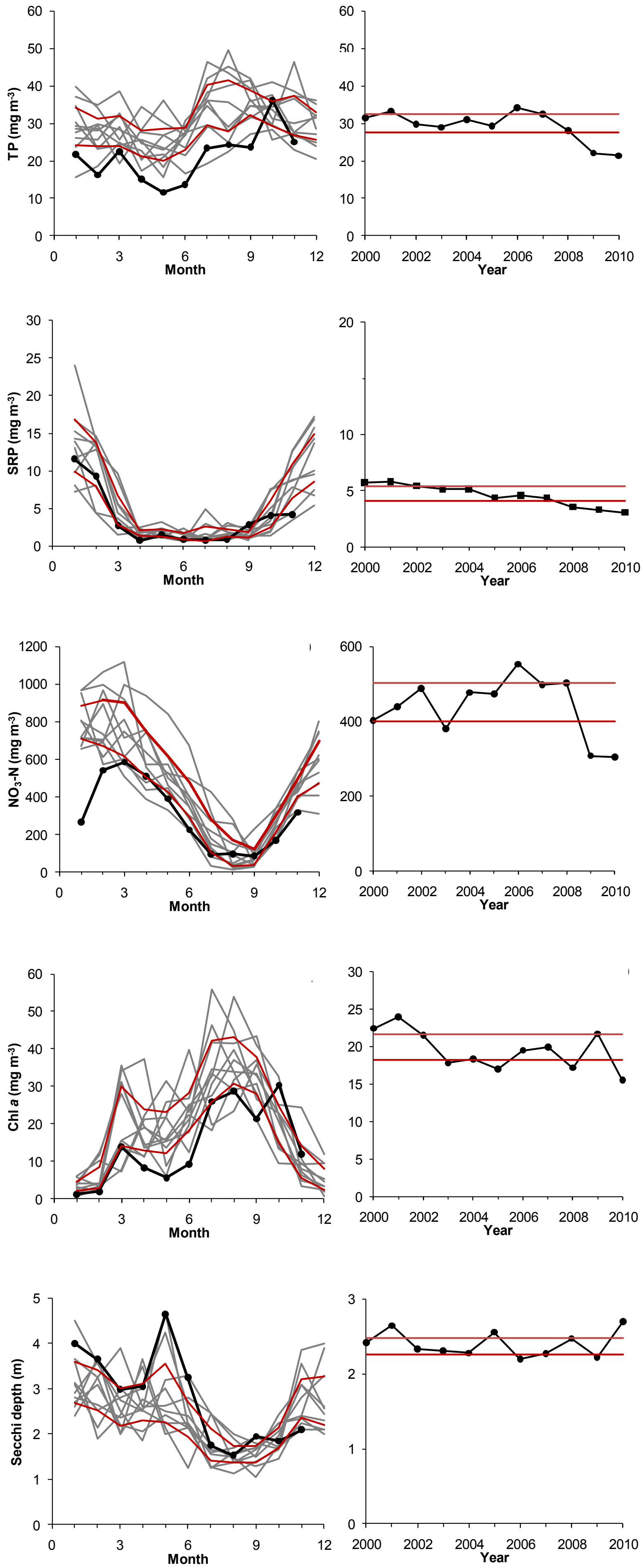

Figure 15.1. Seasonal and annual comparison of conditions in Esthwaite Water in 2010 compared with the previous ten years. left-hand columns, seasonal comparison, individual years 2000 to 2009 shown in grey, 2010 shown in black with dots for monthly mean and the red lines show the upper and lower 95% confidence intervals for 2000 to 2009 based on monthly means; right-hand columns, long term annual trend; the red lines showing the upper and lower 95% confidence intervals based on annual means (Maberly et al. *2011).*

15.2 Water quality and the Water Framework Directive

The Water Framework Directive is the key legislative driver to improve the water quality of the lake. The long-term annual averages for TP and chlorophyll *a* are shown in Figure 15.2, based on the current boundaries for Esthwaite Water. Even at the start of the record in 1945 (TP) and 1964 (Chl *a*), Esthwaite Water would only have been at Moderate ecological state, although TP was close to the Good:Moderate boundary. Both TP and chlorophyll *a* showed a deteriorating ecological state with the worse water quality for both measures in the 1990s when the ecological state was firmly in the poor category. There was an improvement in ecological status in the early 2000s bringing Esthwaite to around the Moderate:Poor boundary and the improvement noted above for recent years places the lake in Moderate ecological status.

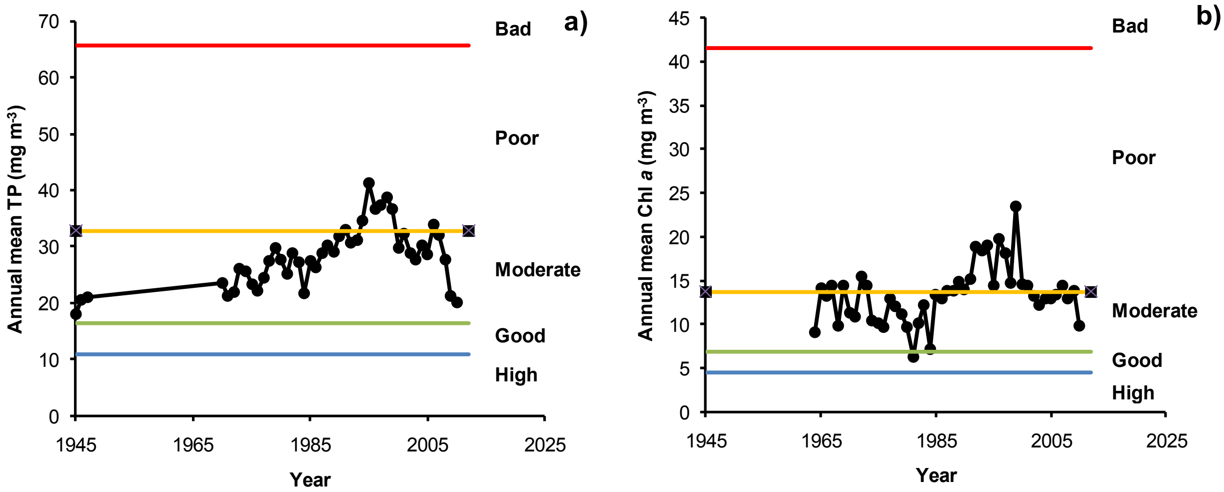

Figure 15.2. Long-term change in annual (geometric) average concentrations of total phosphorus (TP) and phytoplankton chlorophyll a *(Chl* a*) in comparison to current Water Framework Directive boundaries for ecological status.*

Clearly Esthwaite Water is not at the good ecological status that the Water Framework Directive requires it to be by 2015. The effect of the improvements to the WwTW need to be monitored and inputs quantified as precisely as possible. Inputs of phosphorus from septic tanks on the non-sewered parts of the catchment could be evaluated and improved where found to be sub-standard.

The current macrophyte flora is dominated by *Elodea nuttallii.* A greater diversity of species will hopefully be promoted as the nutrient levels decrease. The return of *Najas flexilis* will probably require active intervention. It is possible that seed are still present in sediment at locations where the historic populations grew and, these could be collected and germinated to see if viable and then re-established in the lake.

Current and future climate change is likely to make the management of Esthwaite Water more difficult as the forecast higher summer temperatures and lower summer rainfall are likely to promote the development of cyanobacteria, which have the potential to be toxic. The local actions that can be taken to counter the effects of climate change are to reduce the phosphorus load entering the lake. Some of these actions are underway, but their effect is unlikely to be immediate as several years, or more, may be needed before a sustained improvement is realised.

16. Suggestions for future research

Esthwaite Water is one of the best-studied lakes in the world, but nevertheless more research is needed to understand fully how the lake will respond to future change at the local, regional and global scale. Some of the key areas of work to take into the future include:

- Continued detailed fortnightly monitoring of the lake to establish extent and rate of improvement to management carried out in the catchment;
- Detailed survey of the fish populations in the lake;
- Improved external phosphorus budget for the lake;
- Greater quantification of internal phosphorus sources to the phytoplankton from entrainment from depth and input from the littoral region;
- Investigation of the feasibility of recovering native populations of *Najas flexilis* from seedbanks within the sediment;
- Studies to understand better the role of zooplankton in the trophic web of the lake and their control on water quality;
- Modelling studies to elucidate further the possible response of Esthwaite Water to drier and warmer summers;
- Development of short-term forecasts of water quality using data from the automatic water quality monitoring station, medium-range weather forecasts and PROTECH.

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