A survey of the lakes of the English Lake District:

The Lakes Tour 2005

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Centre for Ecology & Hydrology
Lancaster Environment Centre
Library Avenue
Bailrigg
Lancaster
LA1 4AP

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EXECUTIVE SUMMARY

1. This report presents information resulting from a survey of the limnology of the 20 major lakes and tarns in the English Lake District based on samples taken in January, April, July and October 2005. This ‘Lakes Tour’ supplements similar tours in 1984, 1991, 1995 and 2000.
2. On each sampling occasion depth-profiles were collected of water temperature and oxygen concentration and Secchi depth was measured. An integrated water sample was analysed for pH and alkalinity, major cations and anions, plant nutrients, phytoplankton chlorophyll *a* and species composition and zooplankton abundance and species composition.
3. The lakes had a range in tendency to stratify in summer with the weakest stratification in large, relatively shallow and exposed lakes such as Bassenthwaite Lake. During summer stratification oxygen-depletion at depth was only found in the more productive lakes.
4. Water clarity, assessed by Secchi disc, varied between about 10 m in clear unproductive lakes such as Wastwater to less than 2 meters in the more productive lakes during summer.
5. Major ion composition varied with geology and altitude. Lakes on the Silurian slates (those in the Windermere and Coniston Water catchments) tended to have anions dominated by alkalinity (bicarbonate) and cations dominated by calcium whereas the other lakes tended to have anions dominated by chloride and cations dominated by sodium.
6. Availability of phosphorus is the main factor that affects lake productivity. Lowest concentrations were found in Wastwater and highest in Blelham Tarn and Esthwaite Water. Nitrate was the dominant form of nitrogen: concentrations of ammonium only exceeded that of nitrate in one sample and was often below the limit of detection. Nitrate concentrations tended to be lowest in July and seasonal fluctuations were most marked in the productive lakes. Silica, an essential nutrient for diatoms, showed a similar seasonal pattern to nitrate but the depletion was more marked in April because the spring bloom is typically dominated by diatoms. In unproductive lakes such as Wastwater and Ennerdale Water concentrations of silica did not vary seasonally.

7. The concentration of chlorophyll *a* was used as a measure of phytoplankton abundance. Comparisons across lakes showed low concentrations all the year in the unproductive lakes and seasonally high concentrations in the more productive lakes.
8. The species composition varied seasonally in all the lakes, even unproductive ones with limited seasonal changes in nutrient concentrations, underlying the sensitivity of phytoplankton to environmental conditions. Overall, diatoms dominated in January and April, cyanobacteria were dominant in July and diatoms, cyanobacteria and cryptophytes dominated in October.
9. Zooplankton abundance was greatest in the productive lakes and seasonally abundance tended to be greatest in July and October. Twenty species of zooplankton were recorded in total. The unproductive lakes tended to be dominated by *Eudiaptomus gracilis* and this species dominated most of the lakes in January. At other times of the year the more productive lakes *Daphnia* spp. were often important. Another cladoceran, *Bosmina* spp. was only present in appreciable numbers in January and April.
10. The current state of each lake were summarised in terms of key limnological variables, trophic state and ecological status under the current definitions of the Water Framework Directive.
11. The lakes in the English Lake District are extremely valuable ecologically as they are highly diverse. This was illustrated by showing the link between catchment altitude (as a proxy for land use and soil type) and a range of water chemistry variables and the relationship between phytoplankton chlorophyll *a* and total phosphorus which shows that the productivity of these lakes is controlled by phosphorus. The magnitude of the seasonal changes in silica and nitrate is positively linked to lake productivity. Secchi depth is negatively correlated with phytoplankton, but in January Secchi depth is less for a given chlorophyll, probably because of attenuation by dissolved organic carbon and particulate material brought in to the lakes by winter rains. Minimum oxygen concentration at depth is also negatively related to phytoplankton chlorophyll *a* while phytoplankton diversity is greater in more productive than in less productive lakes.
12. Long-term change from1984 to 2005 (1991 to 2005 for some variables) were analysed. There has been a general increase in alkalinity in many sites and site-specific changes in total P and chlorophyll *a* at a number of sites.

13. The ecological status of each lake under the Water Framework was assessed based on the current draft site-specific ecological boundaries. Based on total P, Wastwater was at High status, Derwent Water, Brothers Water, Rydal Water, Loweswater, Bassenthwaite Lake, Loughrigg Tarn, Elterwater, Ennerdale Water, Buttermere, Thirlmere, Coniston Water, Haweswater and Ullswater were at Good status and Grasmere, Blelham Tarn, Esthwaite Water, Crummock Water and North and South Basins of Windermere were at Moderate status. Based on phytoplankton chlorophyll *a*, Wastwater, Ennerdale Water, Buttermere, Thirlmere and Brothers Water were at High ecological status, Crummock Water, Haweswater and Ullswater were at Good ecological status, Derwent Water, Loweswater, Bassenthwaite Lake, Blelham Tarn, Esthwaite Water and Coniston Water were at Moderate ecological status, Grasmere, Rydal Water, Loughrigg Tarn, Elterwater, and the North and South Basins of Windermere were at Poor ecological status. Overall, 70% of the lakes have High or Good ecological status based on TP but only 40% have High or Good ecological status based on phytoplankton chlorophyll *a*. If the current ecological boundaries remain then many of the lakes in the English Lake District will need to be managed more carefully if Good ecological status is to be achieved by 2015 as laid down by the Water Framework Directive.
14. It is suggested that more work is needed at lakes which have failed Good ecological status and at Wastwater and Rydal Water which have not been studied comprehensively before and where there is evidence of long-term change.

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1. Introduction

The lakes that form the English Lake District have been sampled by the Freshwater Biological Association, the Institute of Freshwater Ecology and its successor the Centre for Ecology and Hydrology, since the 1920s. At about this time Pearsall (1921) arranged some of these lakes in an order corresponding to trophic status, which he recognised was related to their surrounding geology and land use. The lakes range from the unproductive, e.g. Wastwater, which are situated in mountainous regions on hard volcanic rocks to the more productive e.g. Esthwaite Water, which lie on softer rocks usually situated in fertile valleys with deep alluvial soils. The English Lake District is unique, certainly in the UK, in having this wide range of lake types.

Since the 1920s a number of surveys of the English Lakes have been carried out (Pearsall, 1932; Gorham *et al.*, 1974; Jones *et al.*, 1979; Kadiri & Reynolds, 1993). Some of these data were reviewed by George (1992) and Talling (1999) summarised what is known for some of these lakes. The current form of the 'Lakes Tour' started in 1984, although not all the current determinands were measured, and has been repeated in 1991, 1995 and 2000 (Hall *et al.*, 1992, 1996; Parker *et al.*, 2001). The scheme is of a low intensity: samples are only taken four times per year, but nevertheless provides a robust and fairly comprehensive picture of how lakes have responded to environmental pressures.

The English Lake District is one of the most popular tourist regions in the UK because of its relatively unspoilt and dramatic landscapes, of which the lakes form an integral part. This popularity, along with an increasing local population, increased agricultural use of fertilisers, climate change and introduction of alien species by Man's activities has put large ecological pressures on the lakes. Recent legislation originating from the European Commission, The Water Framework Directive (WFD; 2000/60/EC), places a legal duty on the Environment Agency to manage inland, estuarine and coastal water, including lakes, to prevent further deterioration and to improve their ecological quality. Quality or ecological status is determined not just by water chemistry but also by a range of ecological characteristics including the composition and abundance of phytoplankton. The data from the Lakes Tour have already been used to help determine various ecological quality boundaries for the implementation of the WFD. The Lakes Tour also serves to identify lakes that may be showing signs of deterioration and which deserve further more detailed study, and documents the recovery of lakes that have already been subject to management.

2. Materials & Methods

2.1 Sites

The location of the twenty lake basins (Windermere is treated as two basins: North and South) sampled in this work is shown in Figure 2.1 and their geographical and physical features are recorded in Table 2.1.

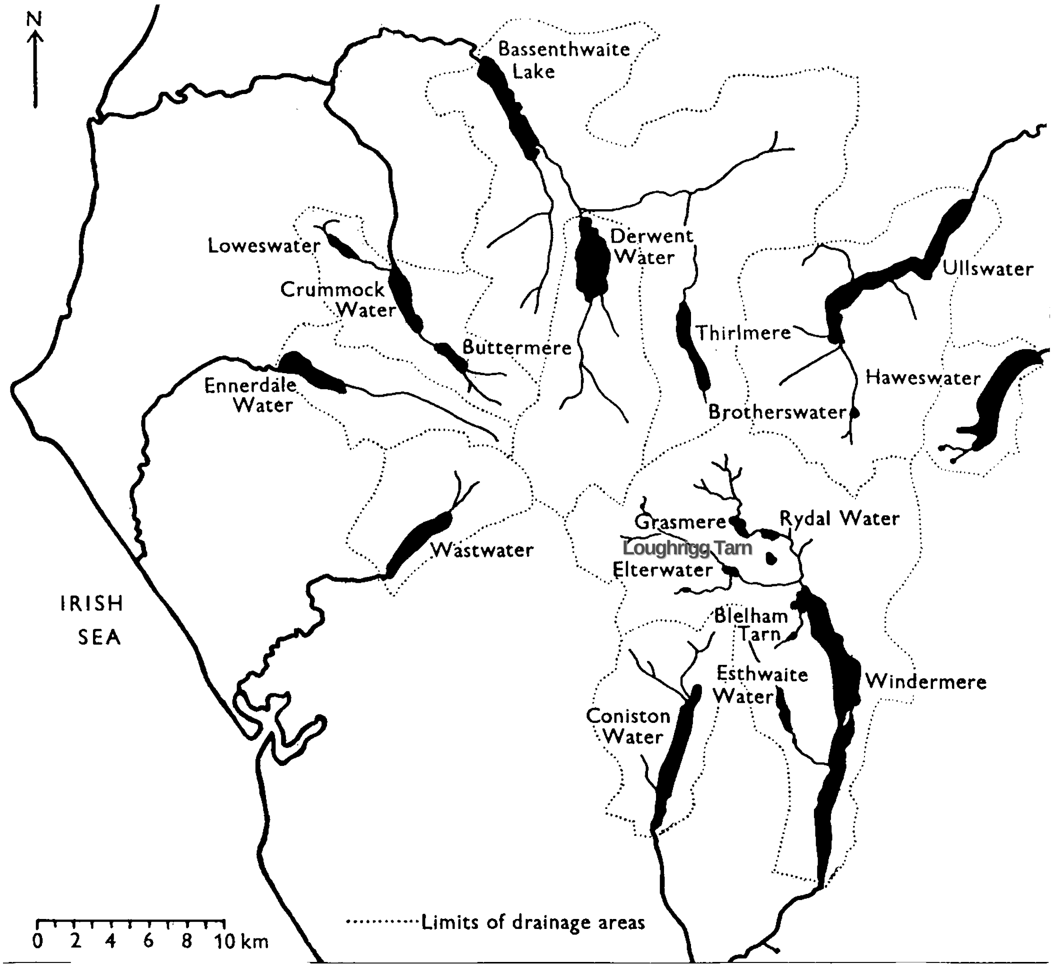


Figure 2.1. The English Lake District showing the 20 lake basins surveyed in this study (based on Knudsen, 1954).

Table 2.1. Geographical and physical characteristics of the 20 lakes basins in the Lakes Tour.

Lake	Catchment area (km^2)	Mean catchment altitude (m)	Lake length (km)	Max. width (km)	Area (km^2)	Volume (m^3 x 10^6)	Mean depth (m)	Max. depth (m)	Approx. mean retention time (days)
Bassenthwaite Lake	360	333	6.2	1.10	5.3	27.9	5.3	19.0	30
Blelham Tarn	4.3	105	0.67	0.29	0.1	0.7	6.8	14.5	50
Brothers Water	13.2	437	0.60	0.40	0.2	1.5	7.2	15.0	21
Buttermere	18.7	377	2.0	0.54	0.9	15.2	16.6	28.6	140
Coniston Water	62.5	227	8.7	0.73	4.9	113.3	24.1	56.1	340
Crummock Water	62.7	327	4.0	0.85	2.5	66.4	26.7	43.9	200
Derwent Water	85.4	354	4.6	1.91	5.4	29.0	5.5	22.0	55
Elterwater	1.0	108	1.0	0.4	0.03	0.1	3.3	7.0	20
Ennerdale Water	43.5	374	3.8	1.10	3.0	53.2	17.8	42.0	200
Esthwaite Water	17.0	148	2.5	0.62	1.0	6.4	6.4	15.5	100
Grasmere	30.2	328	1.6	0.60	0.6	5.0	7.7	21.5	25
Haweswater	32.3	463	6.9	0.90	3.9	76.6	23.4	57.0	500
Loughrigg Tarn	0.95	175	0.4	0.3	0.07	0.5	6.9	10.3	117
Loweswater	8.2	243	1.8	0.55	0.6	5.4	8.4	16.0	150
Rydal Water	33.8	312	1.2	0.36	0.3	1.5	4.4	18.0	9
Thirlmere	53.8	398	6.0	0.78	3.3	52.5	16.1	46.0	280
Ullswater	147	393	11.8	1.02	8.9	223.0	25.3	63.0	350
Wastwater	42.5	385	4.8	0.82	2.9	115.6	40.2	76.0	350
Windermere North Basin	175	231	7.0	1.6	8.1	201.8	25.1	64.0	180
Windermere South Basin	250	231	9.8	1.0	6.7	112.7	16.8	42.0	100

2.2 Sampling

2.2.1 Location and dates

Each lake was sampled from approximately the deepest point, the location of which is shown in Table 2.2. The aim of the protocol is to collect all samples within a 2-week period, weather allowing. In 2005, the sample period was sixteen days in January, fifteen days in April, nine days in July and seven days in October (Table 2.2) so this criteria was not quite met on the first two sampling occasions because of bad weather but was achieved in July and October: a time of year where the lakes are changing most rapidly and so timing of sampling is probably more critical. The date each lake was sampled is given in Table 2.2.

Table 2.2. Sampling location and dates for the Lakes Tour 2005.

Lake	Sampling location (NGR)	January	April	July	October
Bassenthwaite Lake	NY214295	26-Jan	06-Apr	13-Jul	05-Oct
Blelham Tarn	NY366006	13-Jan	04-Apr	11-Jul	03-Oct
Brothers Water	NY403127	28-Jan	07-Apr	14-Jul	06-Oct
Buttermere	NY188154	29-Jan	13-Apr	19-Jul	06-Oct
Coniston Water	SD298935	21-Jan	18-Apr	20-Jul	10-Oct
Crummock Water	NY158192	19-Jan	13-Apr	19-Jul	06-Oct
Derwent Water	NY267207	26-Jan	06-Apr	13-Jul	05-Oct
Elterwater	NY329043	20-Jan	08-Apr	15-Jul	07-Oct
Ennerdale Water	NY103153	27-Jan	14-Apr	18-Jul	04-Oct
Esthwaite Water	SD358972	13-Jan	05-Apr	12-Jul	04-Oct
Grasmere	NY340064	13-Jan	04-Apr	11-Jul	03-Oct
Haweswater	NY478139	20-Jan	12-Apr	11-Jul	03-Oct
Loughrigg Tarn	NY344044	20-Jan	08-Apr	15-Jul	07-Oct
Loweswater	NY127215	19-Jan	13-Apr	19-Jul	06-Oct
Rydal Water	NY358063	20-Jan	11-Apr	15-Jul	07-Oct
Thirlmere	NY318154	28-Jan	07-Apr	14-Jul	06-Oct
Ullswater	NY400190	20-Jan	12-Apr	12-Jul	07-Oct
Wastwater	NY160058	27-Jan	14-Apr	18-Jul	04-Oct
Windermere North Basin	NY383006	13-Jan	19-Apr	12-Jul	05-Oct
Windermere South Basin	SD382914	13-Jan	19-Apr	12-Jul	05-Oct

2.2.2 *Oxygen and temperature profiles in the water column*

Oxygen and temperature profiles were measured with a Wissenschaftlich-Technische Werstätten (WTW) Oxi 340i meter fitted with a combination thermistor and oxygen electrode (WTW TA197) at the deepest point in the lake. This was also the location for all of the limnological measurements and sampling.

2.2.3 *Secchi disc transparency*

A white painted metal disc, 30 cm in diameter, was lowered into the water until it disappeared from view. The disc was then raised slightly until it reappeared and that depth was noted.

2.2.4 *Water samples*

An integrated sample of surface water was taken using a weighted 5 m long plastic tube (except on the two basins of Windermere where a 7 m long tube was used). The tube was lowered until vertical in the water column, the upper end was then sealed, and the tube recovered. Replicate samples were dispensed into a previously rinsed 5 dm^3 plastic bottle. After mixing thoroughly, the water was decanted into: -

a) two disposable 500 cm^3 plastic bottles, for nutrient analysis.

b) a 500 cm^3 plastic bottle containing 2.5 cm^3 of Lugols iodine for subsequent enumeration and identification of algal populations (Lund *et al.,* 1958). The iodine was added to the algal cells to preserve them and increase their rate of sedimentation during subsequent processing in the laboratory.

The remainder of the water sample was used for the determination of chlorophyll *a* concentration in the phytoplankton.

A small glass bottle with a ground glass stopper was completely filled with lake water by submerging it just below the water surface and inserting the stopper so that no air was trapped within the bottle. This sample was used to determine the pH and alkalinity of the sample.

2.2.5 *Nutrient and chemical analysis*

Nitrate, chloride, sulphate, sodium, calcium, magnesium and potassium concentrations were determined by ion chromatography using a Metrohm ion chromatograph. Ammonia, dissolved reactive silicate, total phosphorus, soluble reactive phosphate, alkalinity and pH were determined as described in Mackereth *et al.* (1978).

2.2.6 Algal pigments and populations

The concentration of algal pigments was determined using a boiling methanol extraction procedure as described by Talling (1974). A known volume of water was filtered through a Whatman GF/C filter, the pigments extracted and analysed spectrophotometrically.

A 300 ml sub-sample of the iodine-preserved water sample was concentrated to 5 cm^3 by sedimentation. A known volume of the concentrated sample was transferred to a counting chamber and the algae were enumerated as described by Lund *et al.* (1958). Microplankton and nanoplankton were counted at x100 magnification and x400 magnification respectively.

2.2.7 Zooplankton populations

A standard zooplankton net (mesh size 250 µm, mouth diameter 0.3 m) was lowered to 2/3 the maximum depth of the water column and then hauled steadily to the surface. The contents of the net were emptied into a bottle, and immediately fixed by adding ethanol. In the laboratory the samples were concentrated by filtration and stored in labelled vials in 70% ethanol. The zooplankton were identified and enumerated under a stereozoom microscope, according to Scourfield & Harding (1966) and Gurney (1931-1933). The counts were then converted to numbers per dm^3.

3. Results

3.1 Weather during 2005

The weather during 2005 in relation to the sampling periods is illustrated using data from Esthwaite Water (Fig. 3.1). The January survey took place during a relatively mild spell before a colder period in February. The April samples were taken during windy weather with periods of heavy rain. The samples in July were taken during a period of hot, dry, sunny weather. The October samples were collected during a calm spell of weather when the air temperature was relatively high for the time of year.

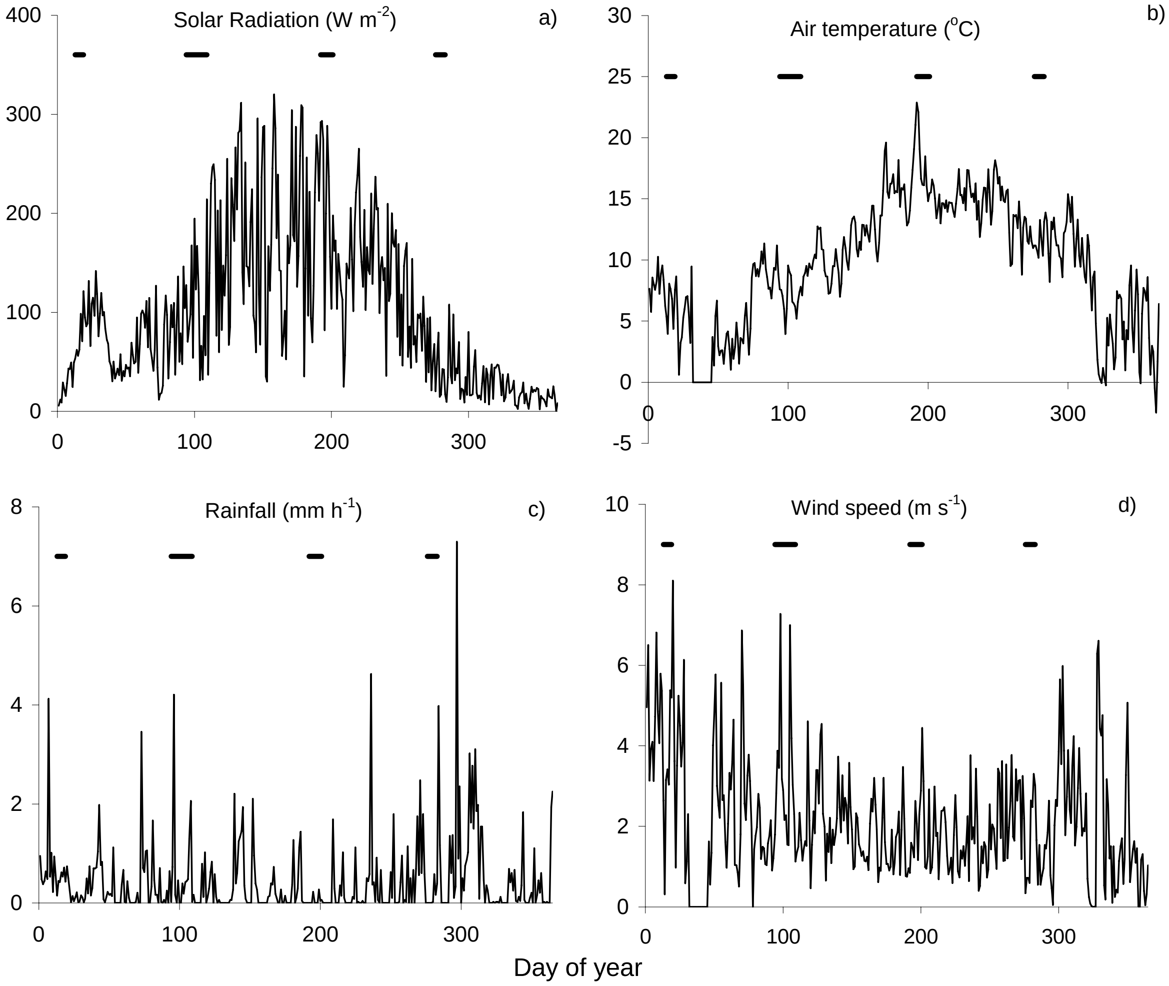


Figure 3.1. Daily mean meteorological data for Esthwaite Water during 2005 comprising: a) total solar radiation; b) air temperature; c) rainfall and d) wind speed. The horizontal bars on each panel show the extent of the sampling periods, derived from Table 2.2.

3.2 Description of the limnology in 2005

3.2.1 Depth-profiles of temperature and oxygen concentration

All the lakes showed a seasonal temperature cycle typical of temperate lakes (Fig. 3.2). All the lakes were fully mixed in January. In April some of the smaller lakes, such as Blelham Tarn, had a weak stratification but the lakes with a large volume, and hence large heat capacity, such as Wastwater or Ullswater, had not stratified. All the lakes had stratified to some extent in July (Fig. 3.2). However, shallow lakes with a relatively large surface area, such as Bassenthwaite Lake (Table 2.1), tended to have a much weaker stratification than a small relatively deep lake such as Brothers Water. In October, stratification persisted in some of the lakes but had broken down in others. The raw temperature data are given in Appendix 1.

In a very unproductive lake the concentration of oxygen will approach 100% equilibrium at all times and depths. This is approximately the pattern in Wastwater (Fig. 3.3) where the slight reduction in concentration at the surface in summer (the orthograde oxygen distribution which is a classical feature of oligotrophic lakes) is a result of lower oxygen solubility in the warmer surface waters. An approximately uniform concentration of oxygen in stratified lakes in summer is also seen in Ennerdale Water, Haweswater and Thirlmere which are also unproductive lakes. Slight oxygen depletion at depth during stratification results from decomposition processes in the hypolimnion and sediments consuming oxygen faster than it can be replaced from the epilimnion by mixing processes. This pattern is seen to a slight extent in lakes such as Coniston Water and Crummock Water and to a slightly greater extent in lake such as Derwent Water and Brothers Water (Fig. 3.3). In the most productive lakes, such as Blelham Tarn or Elterwater, oxygen becomes completely depleted at depth: ie. the lower layers of the lake become anoxic (Fig. 3.3). This can have severe ecological consequences as is discussed in Section 4.1 and is a symptom of extreme eutrophication.

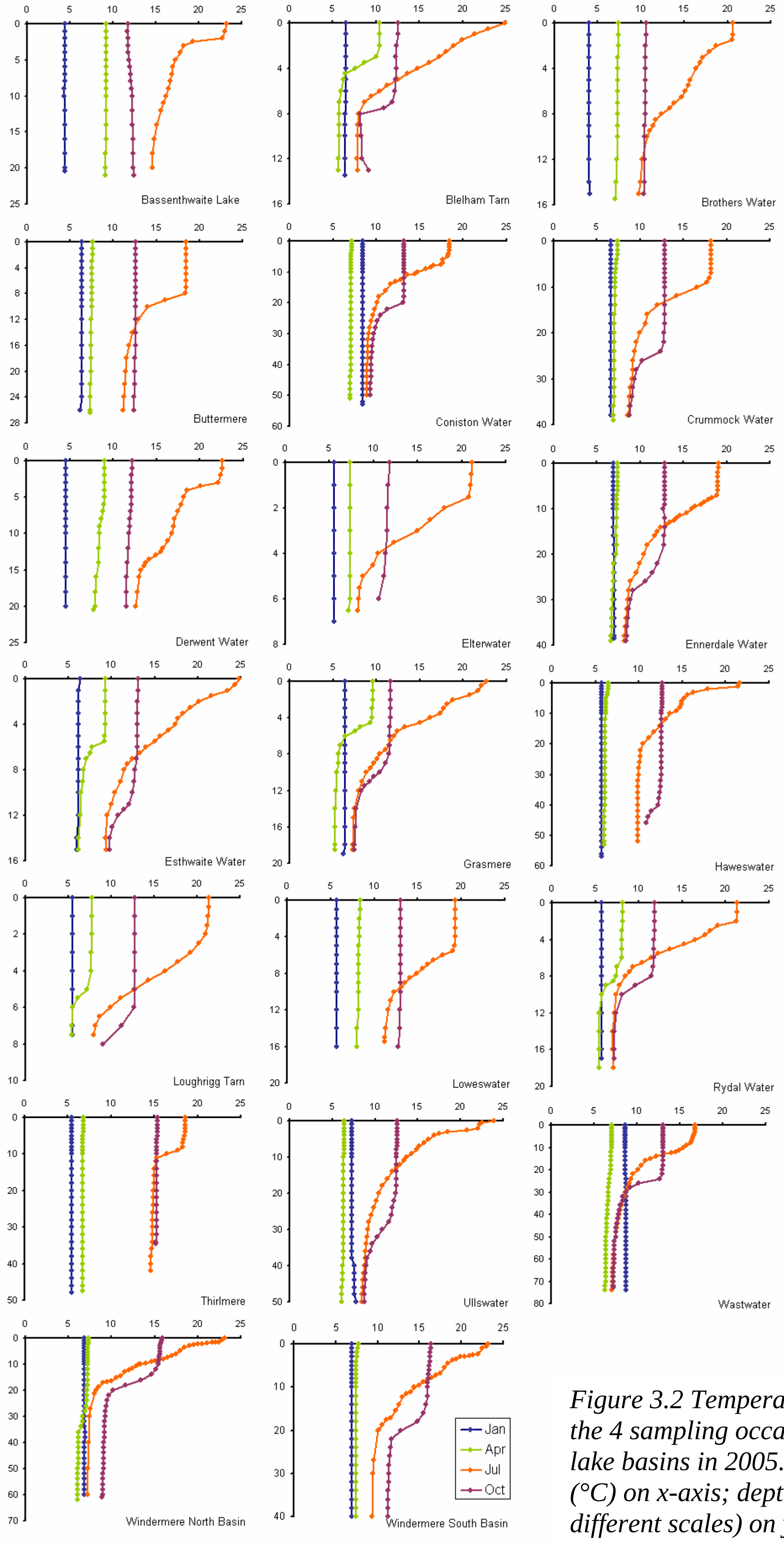


Figure 3.2 Temperature profiles on the 4 sampling occasions for the 20 lake basins in 2005. Temperature (°C) on x-axis; depth (m; note different scales) on y-axis.

Table 3.1 gives the minimum concentration of oxygen recorded at depth in each lake. This usually occurred in July but in some lakes occurred in October. It should be noted that in some lakes where fortnightly data are available, such as Bassenthwaite Lake and the South Basin of Windermere, substantially greater oxygen depletion was recorded between July and October, so this coarse sampling does not necessarily capture the true extent of oxygen-depletion in a lake. The raw oxygen concentration profile data are presented in Appendix 1.

Table 3.1. Annual minimum concentrations of oxygen at depth in 2005. The annual minimum at depth was found in the July or October sample.

Lake	Minimum oxygen concentration at depth ($g\ m^{-3}$)
Bassenthwaite Lake	0.58
Blelham Tarn	0.10
Brothers Water	0.16
Buttermere	7.36
Coniston Water	5.89
Crummock Water	6.18
Derwent Water	1.81
Elterwater	0.00
Ennerdale Water	8.92
Esthwaite Water	0.08
Grasmere	0.16
Haweswater	8.22
Loughrigg Tarn	0.10
Loweswater	0.01
Rydal Water	0.10
Thirlmere	9.51
Ullswater	4.54
Wastwater	9.84
Windermere North Basin	7.85
Windermere South Basin	5.09

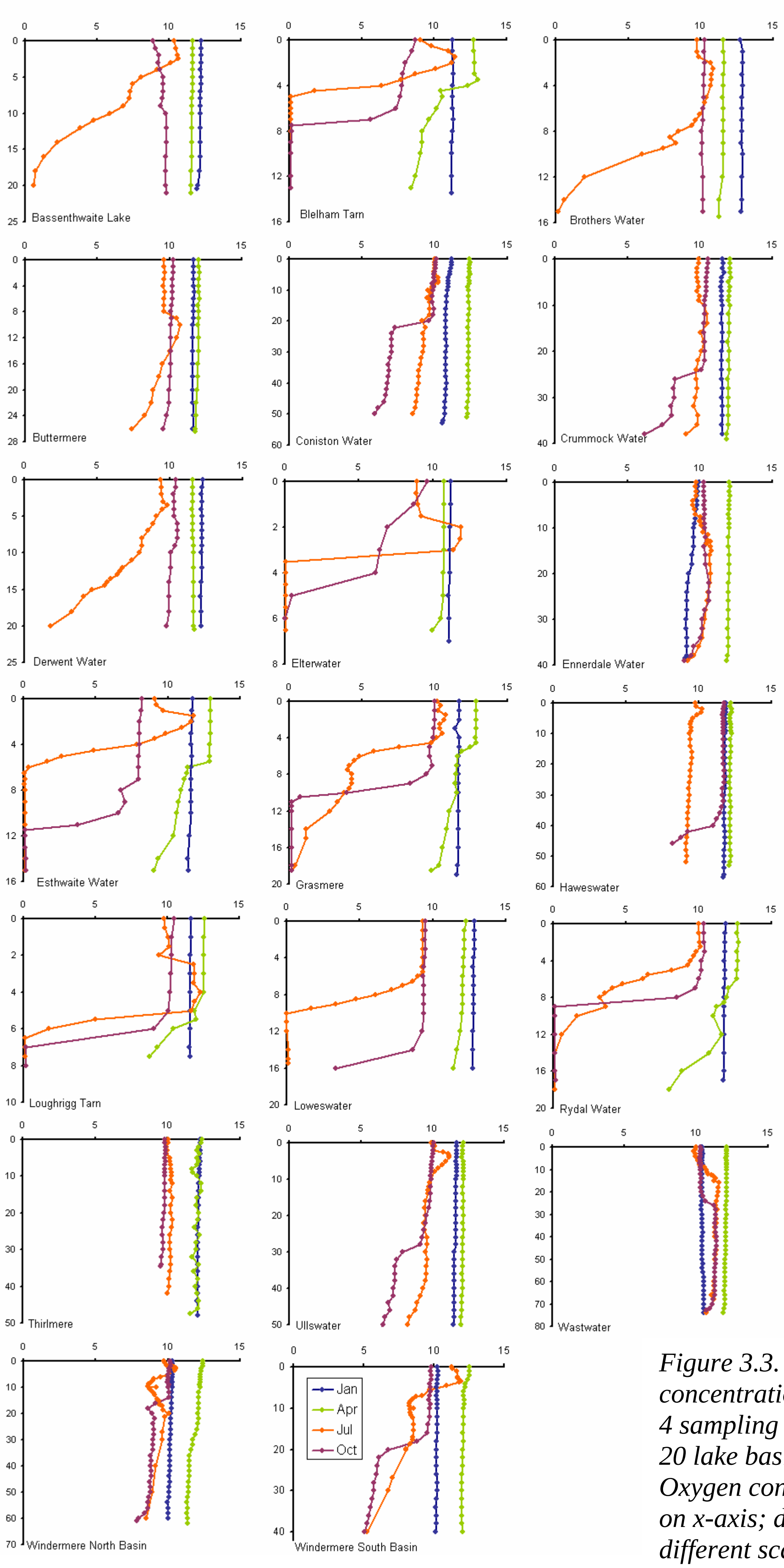


Figure 3.3. Oxygen concentration profiles on the 4 sampling occasions for the 20 lake basins in 2005. Oxygen concentration (g m^{-3}) on x-axis; depth (m; note different scales) on y-axis.

3.2.2 Secchi disc transparency

The depth of the Secchi disc is a rough but convenient measure of water transparency. Figure 3.4 shows that in very unproductive lakes, such as Wastwater and Ennerdale Water, the Secchi depth is visible down to about 9 to 10 m and there is very little seasonal variation. In contrast, in productive lakes such as Bassenthwaite Lake, the Secchi depth is between 2 and 3 m and again did not vary very much seasonally. In lakes such as Elterwater and Rydal Water there were quite substantial seasonal fluctuations in Secchi depth with lowest transparency in spring, probably as a result of spring phytoplankton blooms (Fig. 3.4). The relationship between phytoplankton and the depth of the Secchi disc is discussed in Section 4.1. Buttermere and Crummock Water had the clearest water in July and relatively shallow Secchi depths in January. Since there are small populations of phytoplankton in these lakes, the seasonal pattern is presumably the result of particulate material brought in by winter rainfall. The raw Secchi disc data are presented in Appendix 2.

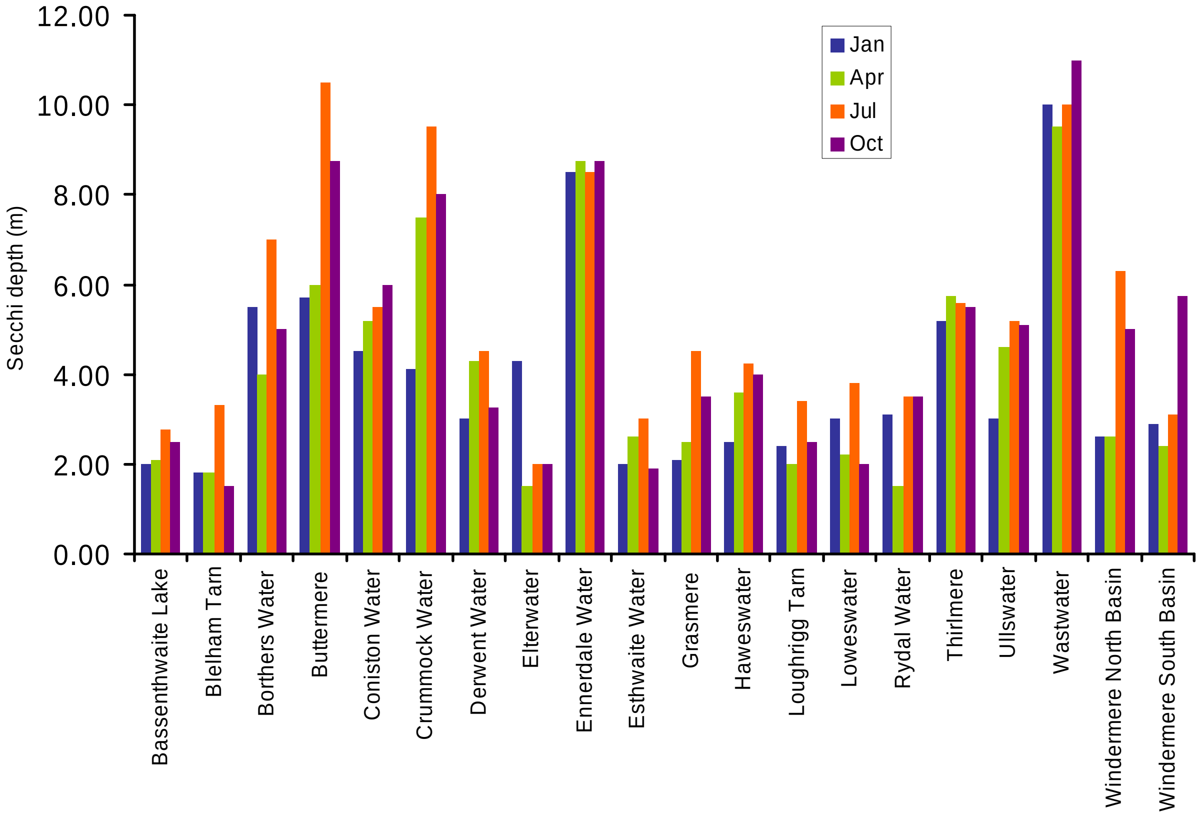


Figure 3.4. Seasonal changes in Secchi disc transparency in the 20 lake basins during 2005.

3.2.3 Major ions

The ionic composition of the major lakes and tarns of the English Lake District has been widely studied (e.g Sutcliffe *et al*. 1982, Sutcliffe 1998). Although there is seasonal variation in ionic composition of the major ions, caused partly by seasonal changes in input via precipitation and partly by differential dilution resulting from evapo-transpiration, ionic composition is relatively conservative and presented here as an annual average. The raw seasonal data are given in Appendix 3. The underlying geology (Fig. 3.5) has a large effect on the composition of the lake water. The annual average data are shown in Figure 3.6 ordered by the main underlying geological rock type and by altitude within each category following Sutcliffe (1998).

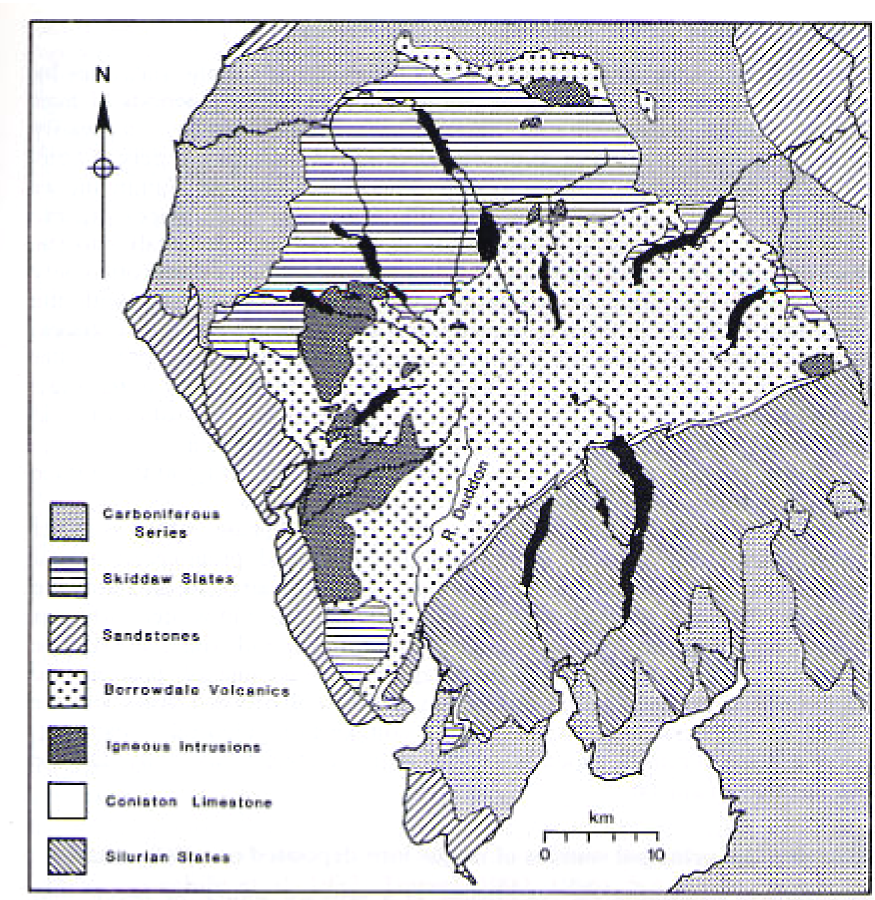


Figure 3.5. The underlying geology of the English Lake District (based on Sutcliffe, 1998).

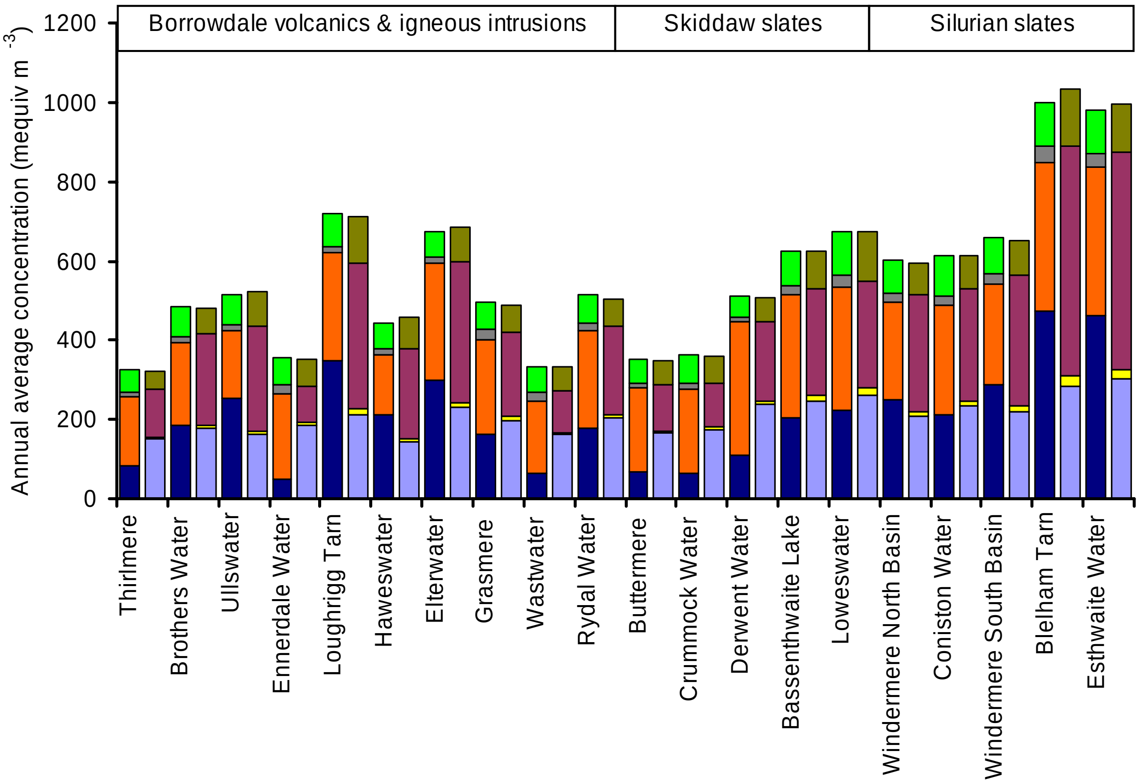


Figure 3.6. Annual average concentration of major anions (first column) and cations (second column) for the 20 lake basins in 2005. Lakes are ordered by underlying geology and then by decreasing catchment altitude following Sutcliffe (1998). Anions are: alkalinity (dark blue), chloride (orange), nitrate (grey) and sulphate (green). Cations are: sodium (light blue), potassium (yellow), calcium (purple) and magnesium (olive green).

In all lakes there is a good balance between cation and anion concentrations which shows that the analysis has been carried out accurately (Fig. 3.6). In lakes on the Silurian slates bicarbonate (alkalinity) tends to be the dominant anion, but chloride has a higher concentration in many of the lakes on the Borrowdale volcanics and Skiddaw slates. This is largely because alkalinity tends to be lower on the Borrowdale and Skiddaw series while chloride concentrations are fairly similar across the 20 lakes. A similar difference is seen in the cations with the balance between calcium and sodium: calcium tends to be the dominant cation in lakes on the Skiddaw slates but sodium tends to dominate in lakes on the two other geologies.

Data on alkalinity, as well as pH, are shown in more detail in Figure 3.7. There is a large range of alkalinities from Ennerdale Water and Wastwater with very low alkalinities to Blelham Tarn and Esthwaite Water with high alkalinities. None of these major lakes have a negative alkalinity (i.e. a net acidity). Almost all the lakes show a weak seasonality in alkalinity with lowest values in January and highest values in July or October. This probably results largely from changing hydrology and evapo-transpiration. The pH varied between 6.5 and 9.1 (Fig. 3.7b). Seasonal variation was mainly apparent in the more productive lakes where it will result from depletion of carbon dioxide as a result of rapid photosynthesis by the phytoplankton. More detailed records (ie. 15-minutely) have shown even more extreme pH variation: for example the pH exceeds 10 in Esthwaite Water in most years (Maberly 1996).

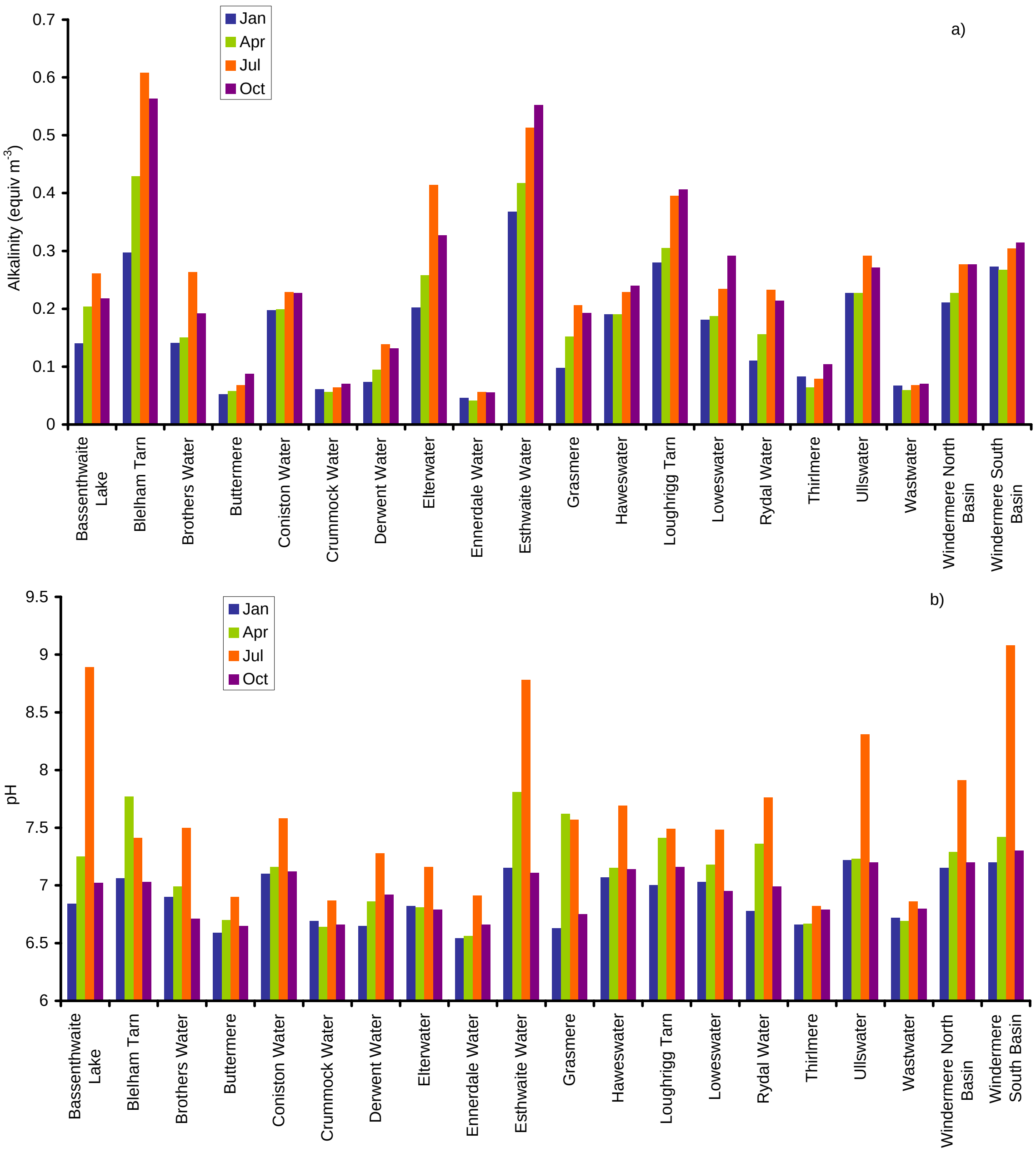


Figure 3.7. Seasonal changes in a) alkalinity and b) pH in the 20 lake basins during 2005.

3.2.4 Nutrient chemistry

Phosphorus, nitrogen and silicon are the three elements required in large amounts by some or all phytoplankton. Each of these is discussed in turn and the raw results are presented in Appendix 4. The productivity of the major English lakes is primarily controlled by the concentration of phosphorus, the limiting nutrient. The concentration of total phosphorus (TP) represents the total concentration of the element in dissolved and particulate fractions including inorganic and organic forms. While not all this TP is available to phytoplankton, it does indicate the trophic status of a lake. Overall concentrations range from very low concentrations in Wastwater, to consistently high concentrations in Esthwaite Water and Belham Tarn (Fig. 3.8). Seasonal change in concentration of TP was most marked in Elterwater where the concentration of TP was much higher in July than January. On average over all 20 lake basins, concentrations of TP were highest in July and lowest in January (Fig. 3.9).

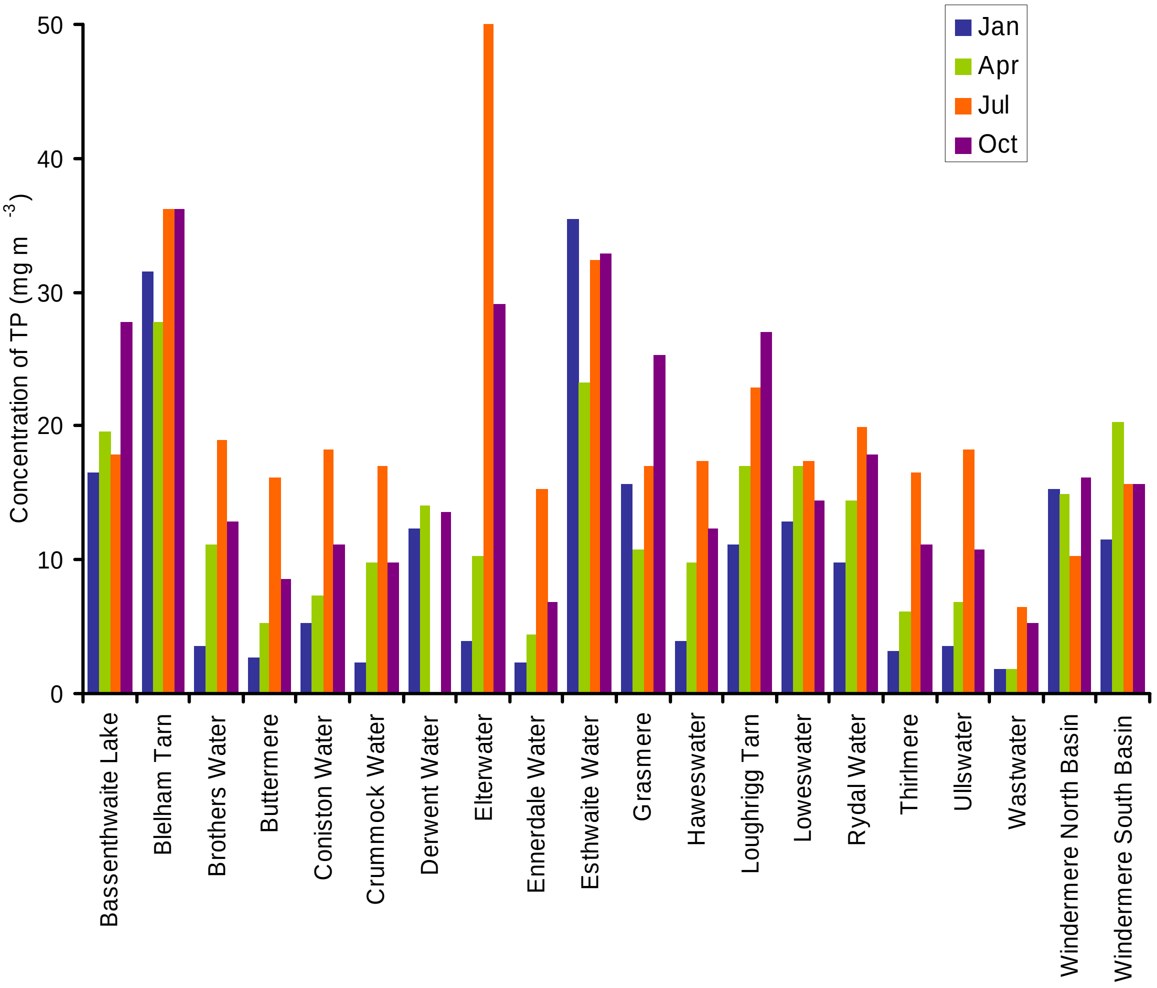


Figure 3.8. Seasonal changes in the concentration of total phosphorus in the 20 lake basins during 2005.

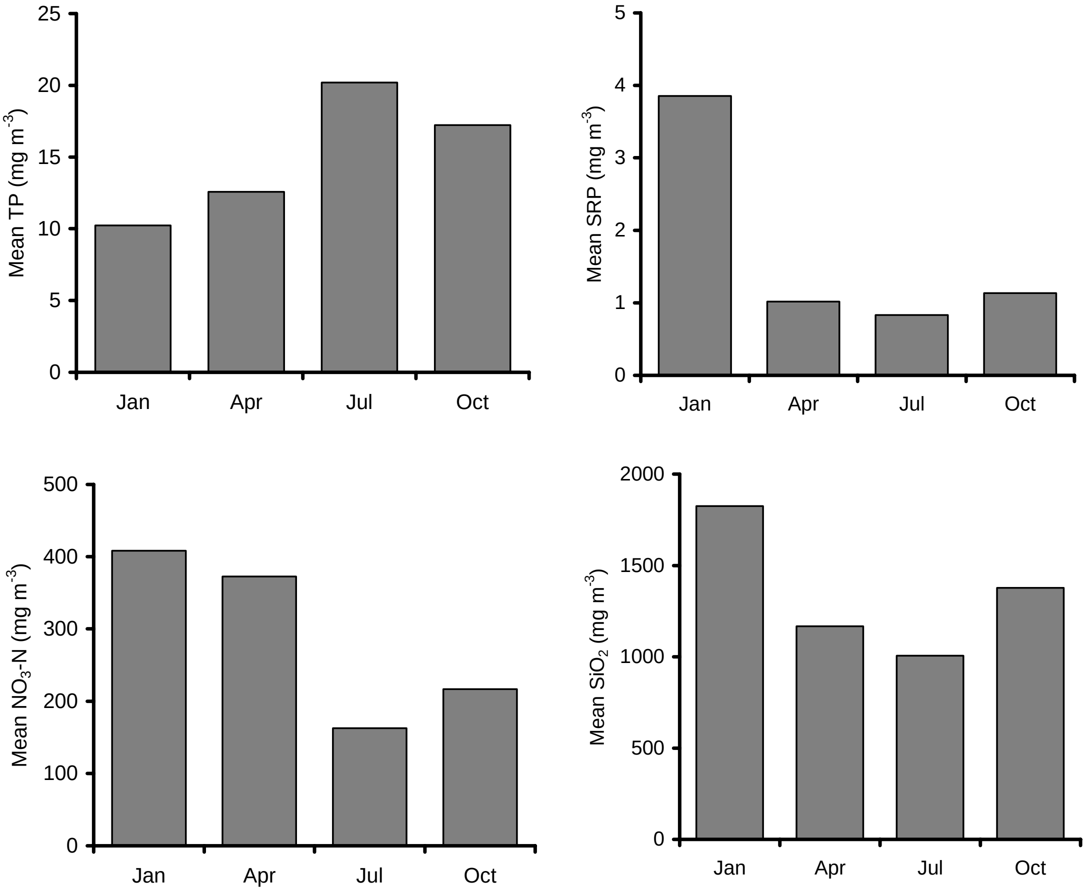


Figure 3.9. Average seasonal concentrations of total phosphorus, soluble reactive phosphorus, nitrate-nitrogen and silica in the 20 lake basins during 2005.

Soluble reactive phosphorus (SRP), the analysed form of phosphorus which most closely reflects that available to microbes including the phytoplankton, showed large seasonal changes and differences among the 20 lake basins (Fig. 3.10). In contrast to TP, concentrations of SRP were highest in January (Fig. 3.9) when demand by phytoplankton was reduced because temperature and daily light levels were low. Ennerdale Water had an elevated concentration of SRP in July: this could be real but could also result from contamination of the sample.

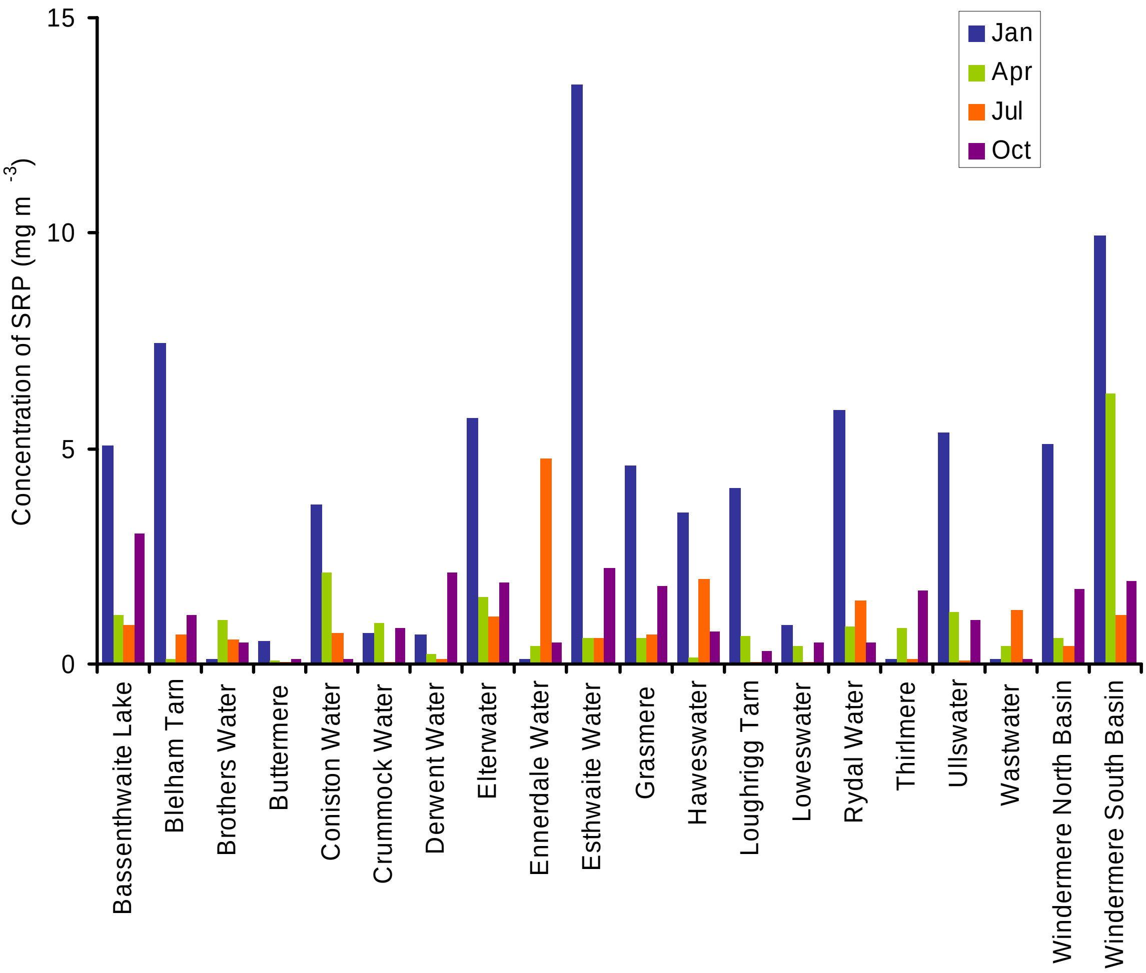


Figure 3.10. Seasonal changes in the concentration of soluble reactive phosphorus in the 20 lake basins during 2005.

Nitrate is usually the main form of nitrogen available to phytoplankton. Like SRP, concentrations were highest in January but the concentration in April was only slightly depleted in contrast to SRP in April where concentrations were already strongly depleted (Fig. 3.9). Thirlmere had the lowest concentrations of nitrate of the 20 lakes and other unproductive lakes such as Wastwater and Ennerdale Water also had low concentrations (Fig. 3.11) but the difference in nitrate concentration among lakes was less than for TP or SRP.

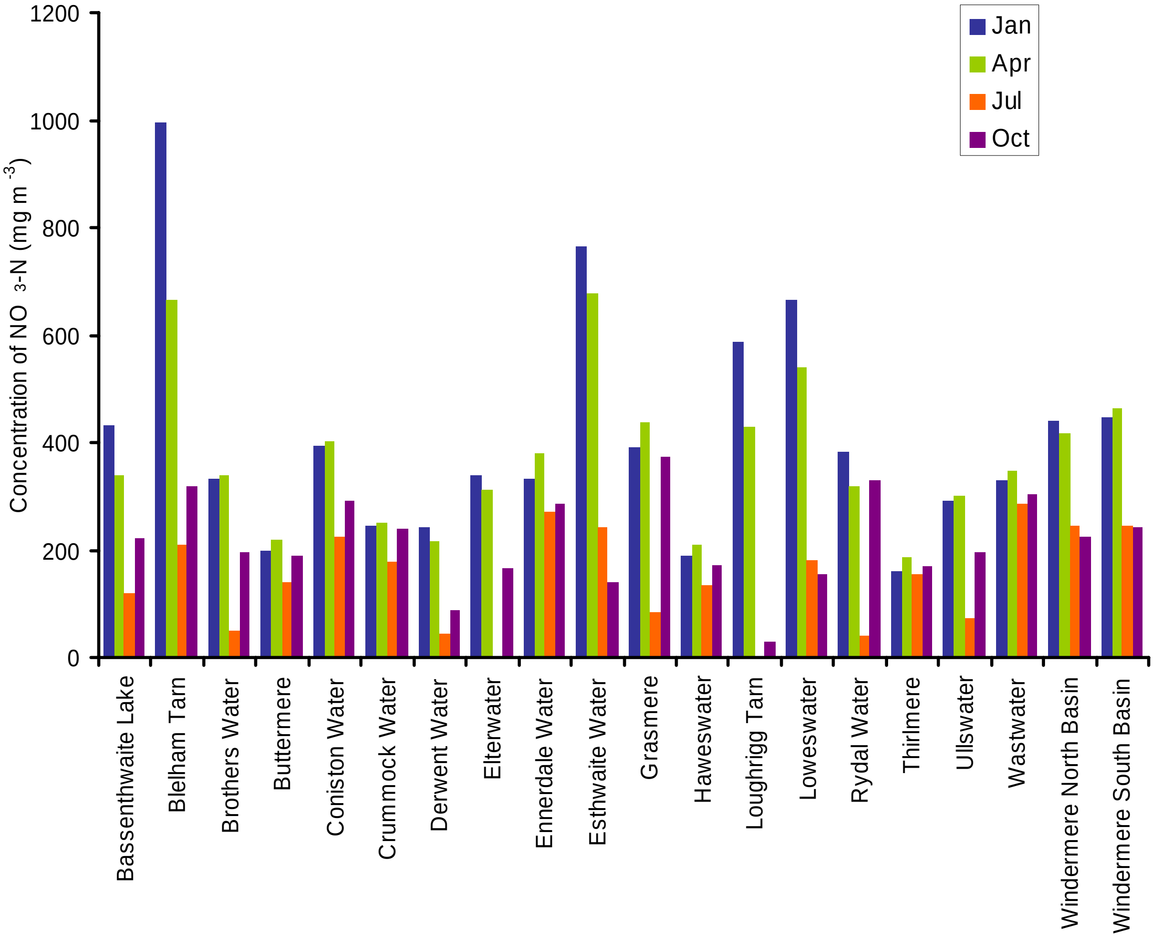


Figure 3.11. Seasonal changes in the concentration of nitrate-nitrogen in the 20 lake basins during 2005.

Ammonium was generally present in very low concentrations (Fig. 3.12): most samples were below the detection limit of 5 mg m^{-3}. Concentrations of nitrate exceeded those of ammonium in all samples apart from the October sample from Loweswater. Elterwater was the only lake where ammonium was detectable in all samples, albeit at low concentration. Some of the more productive lakes such as Eshwaite Water, Loughrigg Tarn and Loweswater showed relatively high concentrations of nitrate in October possible as a result of entrainment of ammonium into surface waters from depth.

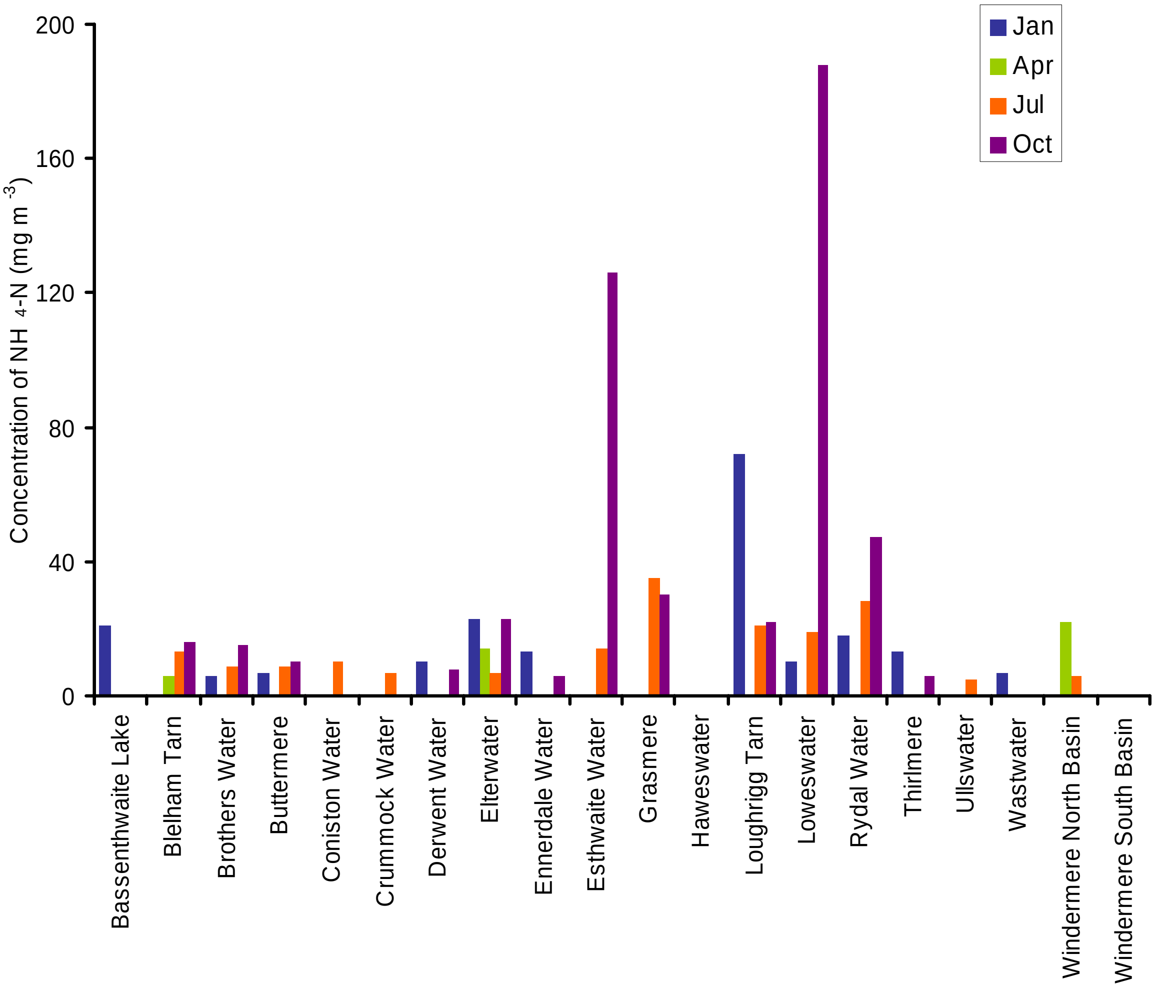


Figure 3.12. Seasonal changes in the concentration of ammonium-nitrogen in the 20 lake basins during 2005.

Silicon is used by a number of groups of phytoplankton, such as the chrysophytes, but is an essential major nutrient for the diatoms. The average seasonal pattern of change of silica is rather similar to that of nitrate (Fig. 3.9) although the depletion of concentration in April is slightly more marked for silica since spring is usually a major period of diatom growth. Ennerdale consistently had the highest concentration of silica and there was very little seasonal variation in concentration (Fig. 3.13). Similar low seasonal variation was also found in other productive lakes such as Wastwater, Crummock Water and Thirlmere. This pattern contrasts with productive lakes such as Grasmere and the South Basin of Windermere where diatom growth reduced concentrations of silica to low concentrations. In Loweswater and Ullswater silica depletion only occurred late in the year in the October sample. In the case of Loweswater this was the result of the spring bloom being dominated by cyanobacteria rather than diatoms (see Section 3.2.6).

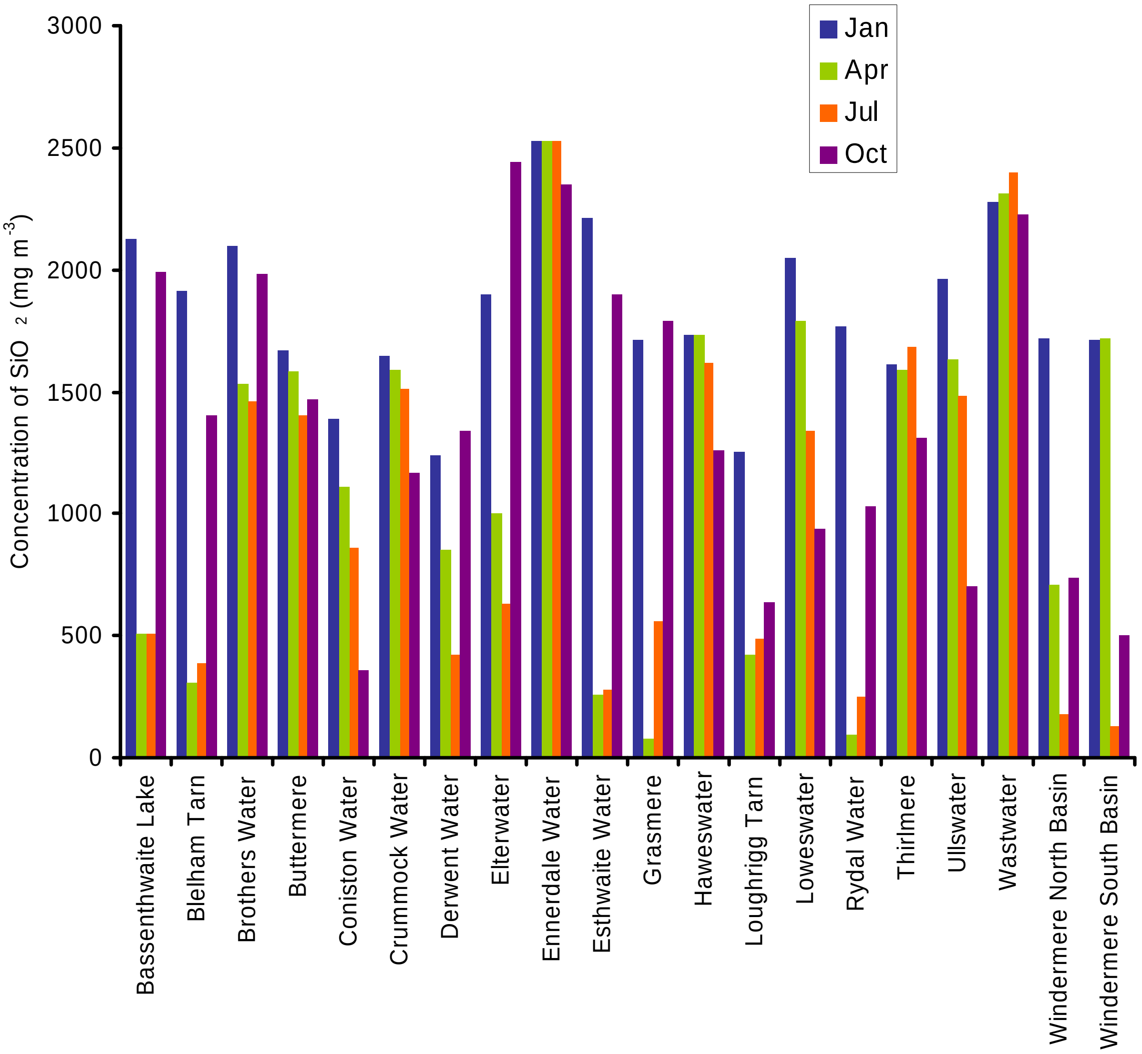


Figure 3.13. Seasonal changes in the concentration of silica in the 20 lake basins during 2005.

3.2.5 *Phytoplankton chlorophyll* **a** *concentration*

Phytoplankton biomass is estimated here using the concentration of the photosynthesis pigment chlorophyll *a*. Figure 3.14 shows the large variation in the concentration of chlorophyll *a* both among lakes and at different times within a lake. In 2005, the concentration of chlorophyll *a* varied between 0.52 mg m^{-3} in Wastwater in January and 72.3 mg m^{-3} for Elterwater in July. The pattern of phytoplankton chlorophyll *a* is broadly the inverse of that for Secchi depth with low concentrations in lakes like Wastwater and Buttermere, and high concentrations in lakes like Elterwater and Loughrigg Tarn (see Section 4.1). In January, concentrations were generally low in all lakes since there is little phytoplankton growth at this time of year because of low temperature and light availability, made worse by full mixing of cells throughout the lake depth as the lakes are not stratified. Furthermore, especially for the more rapidly flushed lakes (Table 2.1), washout of phytoplankton by hydraulic discharge is likely to be particularly rapid. Many of the lakes, such as Grasmere and Loweswater showed an annual maximum concentration of chlorophyll *a* in April corresponding to the spring bloom (Fig. 3.14). In others, such as Elterwater the maximum occurred in July and in Loughrigg Tarn, unusually, the maximum was in October. The raw data on phytoplankton chlorophyll *a* are presented in Appendix 2.

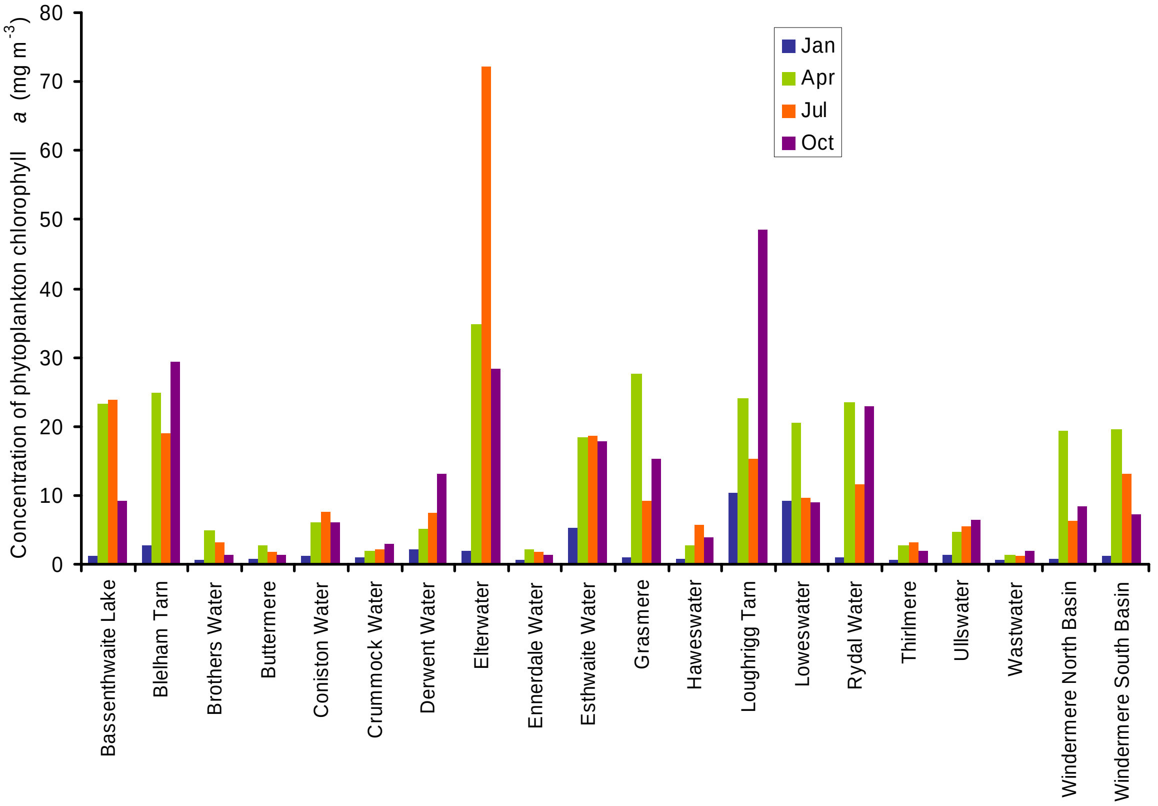


Figure 3.14. *Seasonal changes in the concentration of phytoplankton chlorophyll* a *in the 20 lake basins during 2005.*

3.2.6 Phytoplankton species composition

The phytoplankton are a sensitive and responsive component of the biology of a lake and one of the key ecological characteristics used by the Water Framework Directive to assess the ecological status of a lake. The raw data on phytoplankton species composition are recorded in Appendix 5.

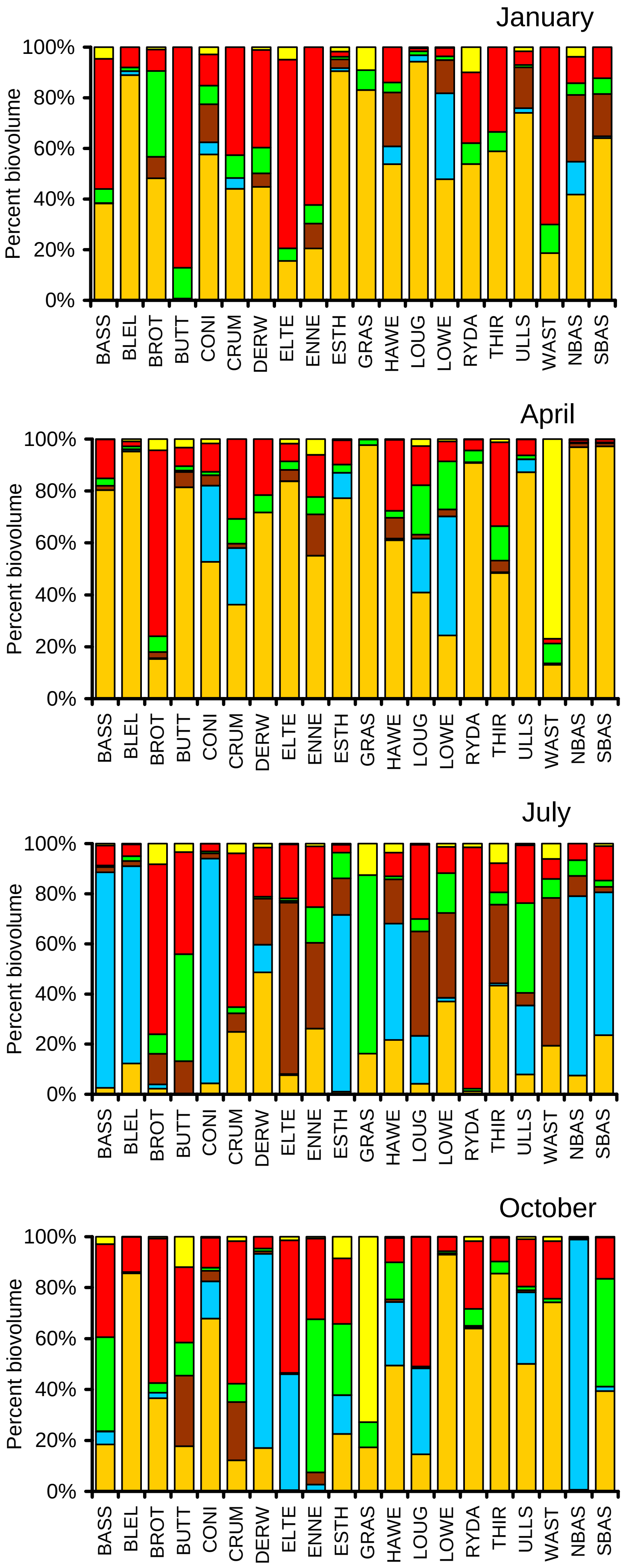


Figure 3.15. Composition of the major groups of phytoplankton in the 20 lake basins during 2005. Diatoms (gold); cyanobacteria (blue); dinoflagellates (brown); euglenophytes (dark green); chlorophytes (green); cryptophytes (red); chrysophytes (yellow).

There was a clear seasonality in all the lakes, even unproductive ones such as Wastwater where nutrient chemistry was relatively constant (Fig. 3.15). In January, diatoms were dominant in most lakes apart from in the more unproductive and low alkalinity lakes such as Buttermere, Ennerdale Water and Wastwater where cryptophytes particularly *Rhodomonas* and *Cryptomonas* dominated. Cryptophytes were also the dominant group in Elterwater in January. Over all the 20 lakes in January the most important taxa were the diatoms *Tabellaria flocculosa* var *asterionelloides*, *Aulacoseira subarctica* and *Cyclotella* sp. In April, diatoms tended to be even more dominant in many of the lakes as this month coincides more or less with the 'spring bloom' which is often dominated by diatoms. Notable exceptions to this pattern were Wastwater which changed from cryptophyte to chrysophyte dominance, particularly *Chrysococcus* sp. and Loweswater which was dominated by cyanobacteria (blue-green algae) mainly *Planktothrix mougeotii*. Crummock Water also had a reasonably large population of cyanobacteria, but it is possible that this derives from inflow of phytoplankton from Loweswater. Over all 20 lakes in April, the most important taxa were the diatoms *Synedra* sp. and *Asterionella formosa* and the cyanobacterium *Anabaena circinalis*. Phytoplankton populations were very diverse in July (Fig. 3.15). Cyanobacteria were dominant in Bassenthwaite Lake, Blelham Tarn, Coniston Water, Esthwaite Water and the two basins of Windermere. Diatoms were dominant in Derwent Water and Thirlmere. Green algae (chlorophytes) mainly *Chlorella* sp. were dominant in Grasmere and Buttermere. Dinoflagellates were dominant in Elterwater, Loughrigg Tarn and Wastwater and cryptophytes were dominant in Brothers Water and Rydal Water. Over all 20 lakes in July the most important taxa were the cyanobacterium *Aphanothece clathrata* and *Anabaena circinalis* and the dinoflagellate *Peridinium* sp. In October, diatoms were slightly more important again but in Windermere North Basin, Derwent Water and Elterwater cyanobacteria were dominant and chlorophytes dominated in Ennerdale. Chrysophytes were dominant in Grasmere. Over the 20 lakes in October the most important taxa were cyanobacterium *Aphanothece clathrata,* the cryptophyte *Cryptomonas* sp. and the chrysophyte *Synura* sp.

3.2.7 Zooplankton populations

The abundance of planktonic crustacea varied greatly among lakes and seasons (Fig. 3.16). Zooplankton densities ranged between <1 individual per dm^3 of lake water to over 170 individuals per dm^3 (in Esthwaite Water in April). The highest zooplankton densities were recorded in Esthwaite Water, Blelham Tarn and Loughrigg Tarn. Very small populations of zooplankton were found in Elterwater. Although the pattern of seasonal change in abundance varied somewhat among lakes, in most of the basins zooplankton abundance was highest in July or October.

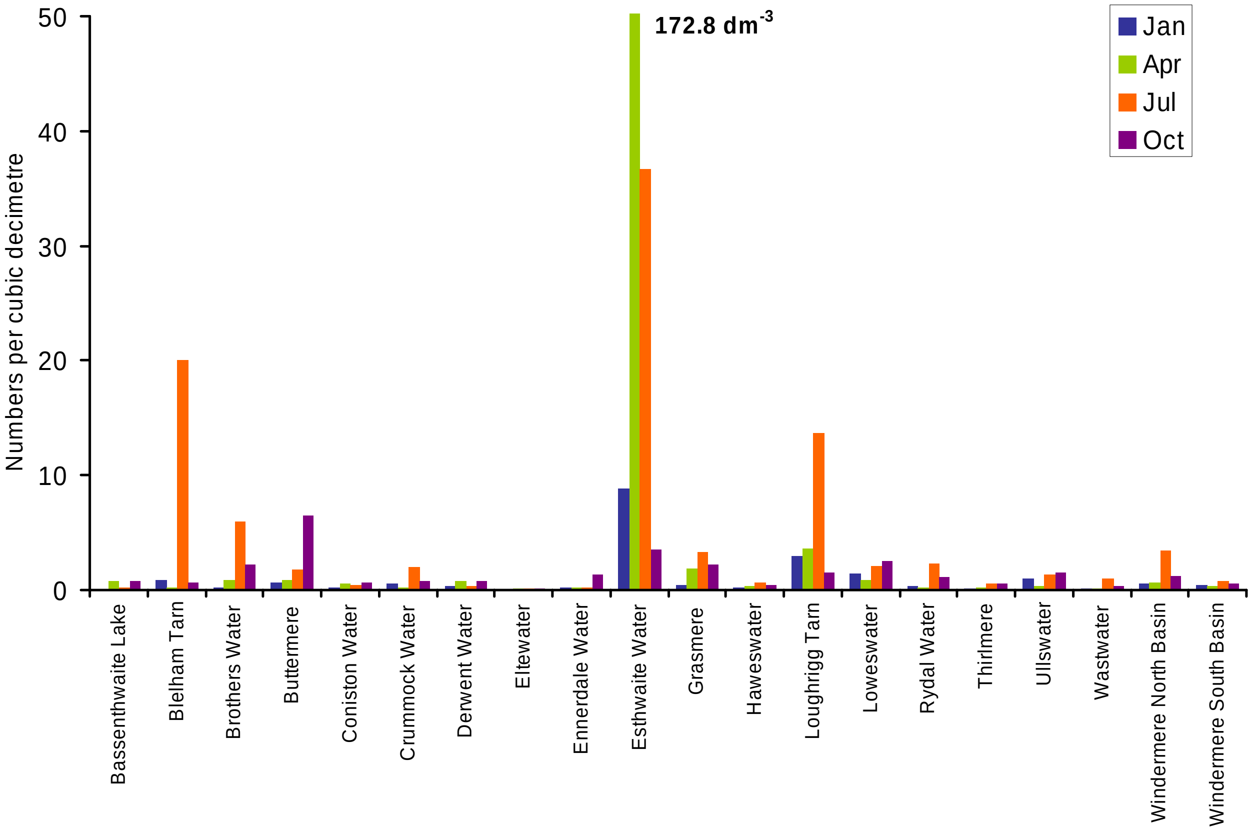
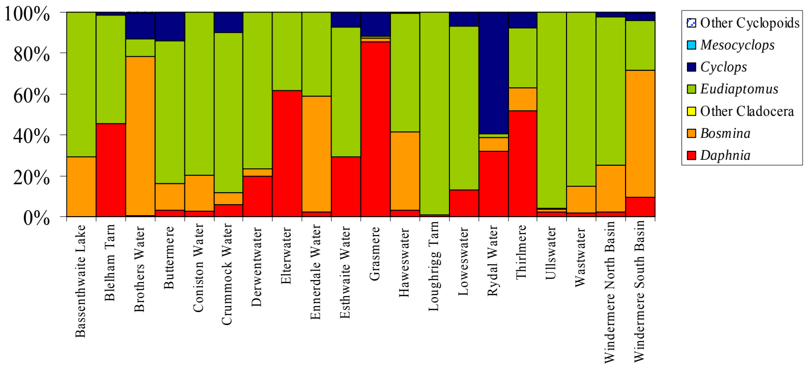


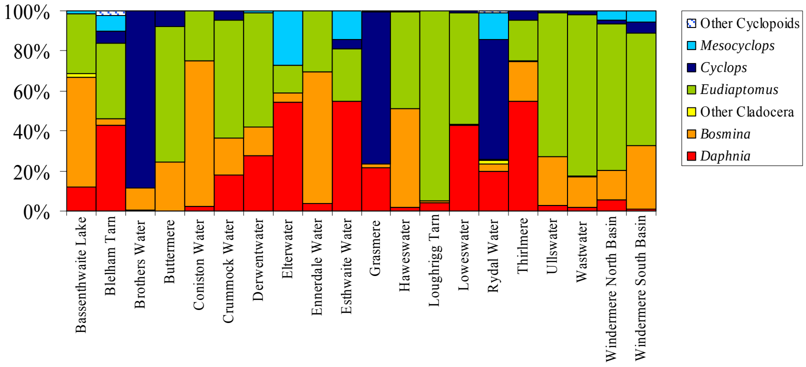
Figure 3.16. *Seasonal changes in the abundance of planktonic crustacea in the 20 lake basins during 2005.*

In total, 20 species of planktonic crustacea were recorded during the 2005 sampling programme. Population densities of these species may be found in Appendix 6. In order to simplify these data, ecologically similar species have been grouped together for the following description. It is apparent, from an examination of the taxonomic composition of the zooplankton community that there were considerable variations in the diversity of the zooplankton assemblage among sites and seasons (Figure 3.17).

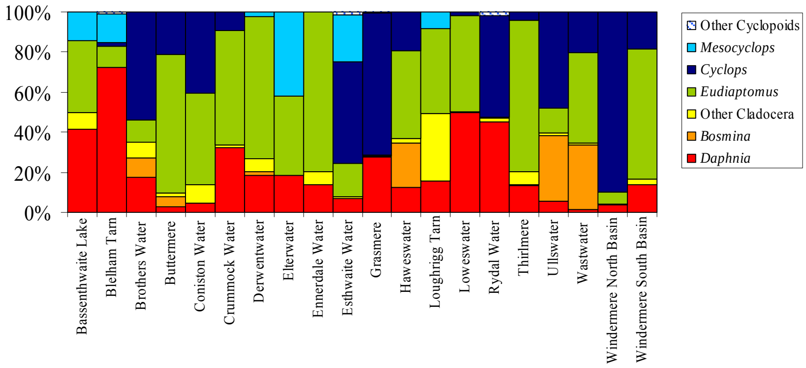
January



April



July



October

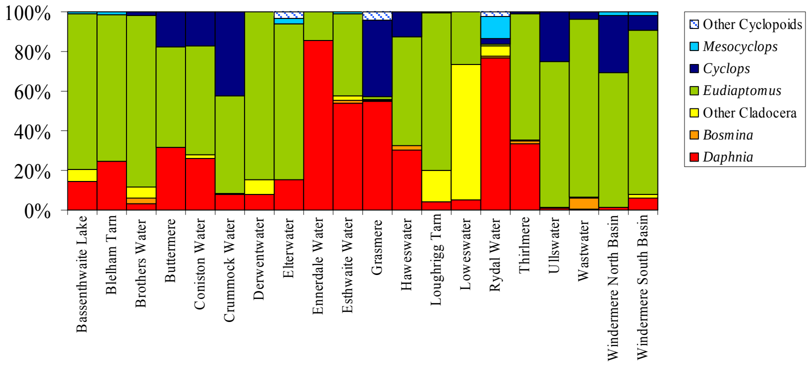


Figure 3.17. The taxonomic composition of the open water zooplankton community of the 20 lake basins, on the 4 sampling occasions during 2005. Each taxon is represented by a percentage contribution to total zooplankton numbers.

During January, the community of most of the lake basins was dominated by the calanoid copepod *Eudiaptomus gracilis* and by cladocera from the genus *Bosmina*. Both taxa feed upon phytoplankton and are known to feed efficiently at low food concentrations. The former even shows a degree of resilience to starvation. They are therefore well adapted to life at a time of year when phytoplankton densities are low (Fig. 3.14). Despite this widespread dominance, in a number of the more productive lakes (including Blelham Tarn, Elterwater and Grasmere) *Daphnia* spp. constituted a considerable proportion of the community. During April, *Daphnia* spp., *Bosmina* spp. and *E. gracilis* still constituted much of the community, although the latter appeared less dominant than in January. At this time, the cyclopoid copepod *Mesocyclops leuckarti* was also found in significant numbers in a number of the more "productive" lake basins i.e. those with comparatively high chlorophyll *a* concentrations. This species feeds efficiently upon smaller zooplankton, including rotifers. It is possible that the emergence of this species was supported by spring increases in rotifer numbers in these lake basins. This species was still found in some of the same lake basins in July. However, at this time, other cyclopoid copepods from the genus *Cyclops* (primarily *Cyclops strenuus abyssorum*) constituted a large part of the community of many lake basins. Adults of this species feed upon large phytoplankton and herbivorous zooplankton. It is likely that these individuals began life during the spring phytoplankton bloom, feeding on the abundant algae, and were now predators of species such as *E. gracilis*. By summer, *Bosmina* spp. no longer formed a large part of the community. This situation persisted into October, when communities were once again dominated by *E. gracilis* and *Daphnia* spp.. During the final two sampling occasions other cladocerans appeared within the communities, including the predatory *Bythotrephes longimanus* and *Leptodora kindtii* and herbivorous species such as *Ceriodaphnia quadrangula* and *Diaphanosoma brachyurum*.

3.3 Current status of the English Lakes

This section assesses the current status of each of the 20 lakes basins surveyed in 2005 on a lake-by-lake basis. In addition to a general assessment, each lake is categorised according to its trophic state and likely ecological status in the terms of the EC Water Framework Directive. The OECD (1982) boundaries for trophic state based on concentration of total phosphorus and chlorophyll *a* and Secchi depth are shown in Table 3.2.

Table 3.2. OECD (1982) boundaries for lake trophic status.

Trophic category	Mean annual TP (mg m^{-3})	Mean annual Chl *a* (mg m^{-3})	Max Chl *a* (mg m^{-3})	Mean annual Secchi (m)	Min Secchi (m)
Ultra-oligtrophic	≤4	≤1	≤2.5	≥12	≥6
Oligotrophic	4 < 10	1 < 2.5	2.5<8	12 > 6	6> 3
Mesotrophic	10< 35	2.5 < 8	8 < 25	6 > 3	3 ≤ 1.5
Eutrophic	35 < 100	8 <25	25 < 75	3 > 1.5	1.5 ≤ 0.7
Hypertrophic	≥100	≥ 25	≥ 75	≤ 1.5	≤0.7

The classification boundaries for the implementation of the Water Framework Directive (WFD) are presently being defined. The first task in implementing this approach is to categorise lakes according to ecological type. Two features are relevant here, the alkalinity and depth. The lakes in the Lakes Tour fall, fairly equally, into four types: low alkalinity-shallow lakes (LA-S), low alkalinity-deep lakes (LA-D), medium alkalinity-shallow lakes (MA-S) and medium alkalinity-deep lakes (MA-D; Table 3.3).

Table 3.3 Classification of lakes of the Lakes Tour in the Water Framework Directive typology for standing waters (UK TAG 2006) and the current Ecological Quality Ratios (EQRs) for different ecological boundaries for the different categories, see text. Note none of the lakes have a mean depth less than 3 m and thus none fall into the very shallow category.

	Low alkalinity (<200 mequiv m^{-3})		Medium alkalinity (200 <1000 mequiv m^{-3})	
Depth	Lake	EQRs	Lake	EQRs
Shallow (mean depth < 15 m)	Brothers Water, Derwent Water, Grasmere Rydal Water	H:G 0.74 G:M 0.41 M:P 0.22 P:B 0.12	Bassenthwaite Lake Blelham Tarn Elterwater Esthwaite Water Loughrigg Tarn Loweswater	H:G 0.58 G:M 0.32 M:P 0.18 P:B 0.10
Deep (mean depth > 15 m)	Buttermere Crummock Water Ennerdale Water Thirlmere Wastwater	H:G 0.80 G:M 0.48 M:P 0.29 P:B 0.17	Coniston Water Haweswater Ullswater Windermere North Basin Windermere South Basin	H:G 0.60 G:M 0.41 M:P 0.28 P:B 0.19

This report uses the currently suggested approach and boundaries. In summary, the reference TP concentration for a lake is determined as a site-specific feature based on the alkalinity and mean depth of a lake using the morpho-edaphic index approach, (Vighi & Chiaudani, 1984) calibrated for UK lakes. The annual mean concentration is compared with this and a ratio (the EQR) calculated as: EQR = Ref TP: Site TP. The currently suggested EQRs for the different lakes types are shown in Table 3.3.

Reference mean (growing season, ie April to September) concentration of phytoplankton chlorophyll *a* is calculated from the reference concentration of TP calculated above using the currently recommended regression equations (Phillips, 2006; all concentrations in mg m^{-3}).

Shallow lakes: $Log\,Chla = -0.512 + 1.105 * Log\,TP$

Deep lakes: $Log\,Chla = -0.220 + 0.731 * Log\,TP$

The high:good and good:moderate ecological boundaries for phytoplankton chlorophyll *a* are set using the EQR values (site mean chlorophyll *a*: reference chlorophyll *a*) set out above. Note that annual mean phytoplankton chlorophyll *a* is used for the site data rather than data from April to September as implied in the current WFD protocols. Note also that type-specific values for phytoplankton chlorophyll *a* rather than site-specific values may also be used.

Table 3.4 gives the estimated values for total phosphorus and phytoplankton chlorophyll *a* at various WFD states. These are compared with values in each lake in the sections below.

Table 3.4. Site-specific annual mean concentrations of total phosphorus and phytoplankton chlorophyll a *at the High:Good (H:G), Good:Moderate (G:M) and Moderate:Poor (M:P) boundary for the 20 sites in the Lakes Tour.*

	Total phosphorus (mg m^{-3})				Phytoplankton chlorophyll *a* (mg m^{-3})			
Lake	H:G	G:M	M:P	P:B	H:G	G:M	M:P	P:B
Bassenthwaite Lake	13.2	23.9	42.6	76.6	5.0	9.1	16.2	29.2
Blelham Tarn	15.4	28.0	49.8	89.6	6.0	10.8	19.3	34.7
Brothers water	9.3	16.8	31.3	57.3	3.3	6.0	11.2	20.6
Buttermere	5.2	8.7	14.4	24.5	2.1	3.6	5.9	10.1
Coniston Water	8.6	12.6	18.4	27.2	3.1	4.6	6.7	9.9
Crummock Water	4.5	7.5	12.5	21.3	1.9	3.2	5.3	9.1
Derwent Water	8.7	15.6	29.1	53.4	3.2	5.7	10.6	19.5
Elterwater	16.6	30.1	53.4	96.2	6.5	11.7	20.8	37.5
Ennerdale Water	4.7	7.9	13.1	22.3	2.0	3.3	5.5	9.4
Esthwaite Water	15.6	28.3	50.2	90.4	6.0	11.0	19.5	35.1
Grasmere	8.8	15.9	29.6	54.2	3.2	5.8	10.8	19.7
Haweswater	8.7	12.7	18.6	27.3	3.2	4.6	6.8	10.0
Loughrigg Tarn	14.1	25.6	45.6	82.1	5.4	9.8	17.5	31.5
Loweswater	11.9	21.6	38.5	69.3	4.5	8.2	14.5	26.1
Rydal Water	10.5	18.9	35.2	64.6	3.6	6.6	12.2	22.4
Thirlmere	5.6	9.3	15.4	26.2	2.2	3.7	6.2	10.6
Ullswater	8.9	13.0	19.1	28.1	3.4	5.0	7.3	10.8
Wastwater	4.1	6.9	11.4	19.4	1.8	3.0	5.0	8.5
Windermere North Basin	8.9	13.0	19.0	28.0	3.2	4.8	7.0	10.3
Windermere South Basin	10.3	15.0	22.0	32.5	3.8	5.6	8.2	12.1

3.3.1 Bassenthwaite Lake

Bassenthwaite Lake is a large shallow lake in the north-west of the English Lake District (Fig. 2.1). It has a very short average retention time for a lake of this size because it has a large catchment area (Table 2.1). Derwent Water and Thirlmere lie within the catchment of Bassenthwaite. Key limnological features in 2005 are shown in Table 3.5. A comprehensive review of the ecology of Bassenthwaite Lake has recently been published (Thackeray *et al*., 2006).

Bassenthwaite Lake from Winlatter Pass. (Photo: M.M. De Ville).

Bassenthwaite Lake appears to be on the meso-eutrophic boundary: The mean concentration of TP, maximum concentration of chlorophyll *a* and minimum Secchi depth are within the mesotrophic range, while the mean concentration of phytoplankton chlorophyll *a* and mean Secchi depth suggest the lake is eutrophic (Table 3.5). In terms of the current WFD classification boundaries, Bassenthwaite Lake is categorised as being in a Good ecological state for TP and a Moderate ecological state for phytoplankton chlorophyll *a*. Bassenthwaite Lake is studied fortnightly as part of the CEH long-term monitoring programme that started in 1990 on this lake.

Table 3.5. Summary of limnological conditions and trophic and Water Framework Directive classifications in Bassenthwaite Lake in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	206		
Mean pH (geometric mean)	7.1		
Mean total phosphorus (mg m^{-3})	20.4	Mesotrophic	Good
Mean soluble reactive phosphorus (mg m^{-3})	2.5		
Mean nitrate-nitrogen (mg m^{-3})	279		
Mean silica (mg m^{-3})	1285		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	14.4	Eutrophic	Moderate
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	23.9	Mesotrophic	
Mean Secchi depth (m)	2.3	Eutrophic	
Minimum Secchi depth (m)	2.0	Mesotrophic	
Minimum oxygen concentration (mg m^{-3})	0.6		

3.3.2 Blelham Tarn

Blelham Tarn is a small lake that drains into the North Basin of Windermere (Table 2.1, Fig. 2.1). It had the highest alkalinity and highest annual mean concentration of TP and nitrate of any of the 20 lakes studied here. It also had the highest concentration of calcium, chloride and magnesium of any of the lakes in 2005 and the third highest concentration of phytoplankton chlorophyll *a* (Table 3.25).

Blelham Tarn (Photo: S.C. Maberly)

Blelham Tarn appears to be on the meso-eutrophic border (Table 3.6) but it suffers severe oxygen depletion at depth during summer. The ecological state in terms of WFD classification is Moderate.

Blelham Tarn is studied fortnightly as part of the CEH long-term monitoring programme that was started by the Freshwater Biological Association in 1945 and continued by CEH since 1989.

Table 3.6. Summary of limnological conditions and trophic and Water Framework Directive classifications in Blelham Tarn in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	474		
Mean pH (geometric mean)	7.2		
Mean total phosphorus (mg m^{-3})	33.0	Mesotrophic	Moderate
Mean soluble reactive phosphorus (mg m^{-3})	2.3		
Mean nitrate-nitrogen (mg m^{-3})	547		
Mean silica (mg m^{-3})	1003		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	19.0	Mesotrophic	Moderate
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	29.4	Eutrophic	
Mean Secchi depth (m)	2.1	Eutrophic	
Minimum Secchi depth (m)	1.5	Meso/Eutrophic	
Minimum oxygen concentration (mg m^{-3})	0.1		

3.3.3 Brothers Water

Brothers Water is a small lake with a fairly high-altitude catchment that drains into the southern end of Ullswater (Table 2.1). Of the 20 lakes in the Lakes Tour it had the third highest annual mean concentration of silica and one of the lowest concentrations of potassium (Table 3.25).

Brothers Water from Kirkstone Pass (Photo: M.M. De Ville).

Brothers Water is close to the mesotrophic-oligotrophic boundary. Surprisingly for such a lake, there is quite a substantial oxygen depletion at depth (Table 3.7) but mainly in the bottom water which may reflect the fact that the water at depth is isolated in a fairly small volume because of the lake. Nevertheless when mapping onto the WFD classification it is classified as being at Good or High status (Table 3.7).

Table 3.7. Summary of limnological conditions and trophic and Water Framework Directive classifications in Brothers Water in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	187		
Mean pH (geometric mean)	6.9		
Mean total phosphorus (mg m^{-3})	11.6	Mesotrophic	Good
Mean soluble reactive phosphorus (mg m^{-3})	0.5		
Mean nitrate-nitrogen (mg m^{-3})	230		
Mean silica (mg m^{-3})	1768		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	2.5	Oligo/Mesotrophic	High
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	4.9	Oligotrophic	
Mean Secchi depth (m)	5.4	Mesotrophic	
Minimum Secchi depth (m)	4.0	Oligotrophic	
Minimum oxygen concentration (mg m^{-3})	0.16		

3.3.4 Buttermere

Buttermere (Photo J.B. James).

Buttermere is a moderately sized lake in the north-west of the English Lake District that drains into Crummock Water (Fig. 2.2). The Secchi depth is surprisingly variable (Fig. 3.4) with low transparency in January despite phytoplankton chlorophyll *a* also being low. This suggests input of particulate or dissolved coloured material during winter rains. A similar feature is also seen in Crummock Water. It had the third lowest concentration of TP of any of the lakes in the Lakes Tour (after Wastwater and Ennerdale Water) and low concentrations of sulphate and potassium and alkalinity (Table 3.25).

All the measures suggest that Buttermere is oligotrophic (Table 3.8). This is also reflected in the limited oxygen depletion at depth. In terms of the WFD, Buttermere is classified as being at Good or High ecological status.

Table 3.8. Summary of limnological conditions and trophic and Water Framework Directive classifications in Buttermere in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	66		
Mean pH (geometric mean)	6.7		
Mean total phosphorus (mg m^{-3})	8.2	Oligotrophic	Good
Mean soluble reactive phosphorus (mg m^{-3})	0.2		
Mean nitrate-nitrogen (mg m^{-3})	187		
Mean silica (mg m^{-3})	1530		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	1.7	Oligotrophic	High
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	2.7	Oligotrophic	
Mean Secchi depth (m)	7.7	Oligotrophic	
Minimum Secchi depth (m)	5.7	Oligotrophic	
Minimum oxygen concentration (mg m^{-3})	7.4		

3.3.5 Coniston Water

Coniston Water is the fifth largest lake in the study in terms of area and the fourth largest in terms of volume.

Coniston Water (Photo L. King).

Various measures suggest that it is mesotrophic and this is reflected in the slight oxygen depletion at depth (Table 3.9). In terms of the WFD the ecological status is Good in terms of TP but the mean chlorophyll *a* value in 2005, was just above the site-specific Good:Moderate boundary of 4.8 mg m^{-3} for this lake and so fell into the Moderate category. A review of the ecology of Coniston Water was carried out by Maberly *et al*. (2003).

Table 3.9. Summary of limnological conditions and trophic and Water Framework Directive classifications in Coniston Water in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	213		
Mean pH (geometric mean)	7.2		
Mean total phosphorus (mg m^{-3})	10.5	Oligo/Mesotrophic	Good
Mean soluble reactive phosphorus (mg m^{-3})	1.7		
Mean nitrate-nitrogen (mg m^{-3})	328		
Mean silica (mg m^{-3})	930		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	5.3	Mesotrophic	Moderate
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	7.7	Oligotrophic	
Mean Secchi depth (m)	5.3	Mesotrophic	
Minimum Secchi depth (m)	4.5	Oligotrophic	
Minimum oxygen concentration (mg m^{-3})	5.9		

3.3.6 Crummock Water

Crummock Water receives water from Buttermere to the south and Loweswater to the north-west (Fig. 2.2). It had the second lowest alkalinity and pH (Table 3.10) of the 20 lakes. Like Buttermere, Secchi transparency was low in January despite phytoplankton chlorophyll *a* also being low suggesting it results from particulate or dissolved organic material originating in the catchment.

Crummock Water, looking north. (Photo: S.C. Maberly).

Its trophic status is essentially oligotrophic although the mean concentration of TP was on the oligotrophic/mesotrophic boundary, largely because of a fairly high concentration of TP measured in July (Fig. 3.8). The lack of a substantial depletion of oxygen at depth is consistent with its oligotrophic status. Its ecological status in terms of the WFD is Good for phytoplankton chlorophyll *a*, but only Moderate for TP. The mean concentration of TP (Table 3.10), however, was only slightly above the Good:Moderate boundary for a lake of this type.

Table 3.10. Summary of limnological conditions and trophic and Water Framework Directive classifications in Crummock Water in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	63		
Mean pH (geometric mean)	6.7		
Mean total phosphorus (mg m^{-3})	9.7	Oligo/Mesotrophic	Moderate
Mean soluble reactive phosphorus (mg m^{-3})	0.6		
Mean nitrate-nitrogen (mg m^{-3})	228		
Mean silica (mg m^{-3})	1480		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	2.0	Oligotrophic	Good
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	3.0	Oligotrophic	
Mean Secchi depth (m)	7.3	Oligotrophic	
Minimum Secchi depth (m)	4.1	Oligotrophic	
Minimum oxygen concentration (mg m^{-3})	6.2		

3.3.7 Derwent Water

Derwent Water (Photo: M.M. De Ville).

Derwent Water lies in the north of the English Lake District within the catchment of Bassenthwaite Lake (Fig. 2.2). It is relatively shallow but has some deep water down to 22 m (Table 2.1). Of the 20 lakes in the Lakes Tour, Derwent Water had the lowest annual average concentration of nitrate and sulphate and low concentrations of magnesium, but the third highest concentration of chloride (Table 3.25).

Its trophic status is clearly mesotrophic but there is a quite substantial oxygen depletion, but this is restricted to the deep water (Fig. 3.3). The status in terms of the WFD was 'Good' for the annual mean concentration of TP. In contrast, the mean phytoplankton chlorophyll *a* at 6.9 mg m^{-3} was quite substantially over the G:M threshold of 4.8 mg m^{-3}. It should be noted that, based on fortnightly data, in five of the last ten years the mean phytoplankton chlorophyll *a* concentration has just fallen below the G:M threshold (Maberly *et al*., 2005) placing Derwent Water on the G:M boundary. Derwent Water is studied fortnightly as part of the CEH long-term monitoring programme that began on this lake in 1990.

Table 3.11. Summary of limnological conditions and trophic and Water Framework Directive classifications in Derwent Water in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	109		
Mean pH (geometric mean)	6.9		
Mean total phosphorus (mg m^{-3})	13.3	Mesotrophic	Good
Mean soluble reactive phosphorus (mg m^{-3})	0.8		
Mean nitrate-nitrogen (mg m^{-3})	148		
Mean silica (mg m^{-3})	963		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	6.9	Mesotrophic	Moderate
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	13.0	Mesotrophic	
Mean Secchi depth (m)	3.8	Mesotrophic	
Minimum Secchi depth (m)	3.0	Oligo/Mesotrophic	
Minimum oxygen concentration (mg m^{-3})	1.8		

3.3.8 Elterwater

The inner basin of Elterwater is the smallest of the 20 lakes studied here in terms of area and volume and also has the second shortest average retention time. The July sample had the highest concentration of TP and chlorophyll *a* of any samples collected in 2005 (Figs 3.8 and 3.14) but as an annual average Elterwater had the third highest concentration of TP of the 20 lakes but the highest concentration of phytoplankton chlorophyll *a* (Table 3.25).

Elterwater viewed from Loughrigg Fell (Photo: M.M. De Ville).

Elterwater can be allocated to a range of trophic categories depending on the feature used (Table 3.12). Thus it is classified as mesotrophic based on TP but hypertrophic based on mean chlorophyll *a*. The other measures give a eutrophic assessment which is probably the fairest category and consistent with the complete oxygen depletion at depth. In terms of the WFD, Elterwater is categorised as Good for TP but only Poor for phytoplankton chlorophyll *a*, so again there is a disparity in classification. The annual mean chlorophyll *a* concentration in Elterwater is actually close to the Poor:Bad boundary.

Table 3.12. Summary of limnological conditions and trophic and Water Framework Directive classifications in Elterwater in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	300		
Mean pH (geometric mean)	6.9		
Mean total phosphorus (mg m^{-3})	23.3	Mesotrophic	Good
Mean soluble reactive phosphorus (mg m^{-3})	2.6		
Mean nitrate-nitrogen (mg m^{-3})	273		
Mean silica (mg m^{-3})	1493		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	34	Hypertrophic	Poor
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	72	Eutrophic	
Mean Secchi depth (m)	2.5	Eutrophic	
Minimum Secchi depth (m)	1.5	Meso/Eutrophic	
Minimum oxygen concentration (mg m^{-3})	0.0		

3.3.9 *Ennerdale Water*

Ennerdale Water is a moderate-sized lake in the west of the English Lake District (Fig. 2.2). It had the lowest alkalinity and concentration of calcium and the highest concentration of silica of any of the studied lakes and the second lowest concentration of TP after Wastwater (Figs 3.7, 3.13; Table 3.25).

Ennerdale Water(Photo: S.C. Maberly).

Ennerdale Water is a very unproductive lake, classified as oligotrophic or even ultra-oligotrophic for maximum phytoplankton chlorophyll *a* and minimum Secchi depth (Table 3.13). There is very little evidence for oxygen depletion at depth. In terms of the WFD it is close to reference state, i.e. High ecological status for a lake of this type.

Table 3.13. Summary of limnological conditions and trophic and Water Framework Directive classifications in Ennerdale Water in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	50		
Mean pH (geometric mean)	6.6		
Mean total phosphorus (mg m^{-3})	7.2	Oligotrophic	Good
Mean soluble reactive phosphorus (mg m^{-3})	1.4		
Mean nitrate-nitrogen (mg m^{-3})	319		
Mean silica (mg m^{-3})	2485		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	1.5	Oligotrophic	High
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	2.1	Ultra-oligotrophic	
Mean Secchi depth (m)	8.6	Oligotrophic	
Minimum Secchi depth (m)	8.5	Ultra-oligotrophic	
Minimum oxygen concentration (mg m^{-3})	8.9		

3.3.10 Esthwaite Water

Esthwaite Water looking north. (Photo: Freshwater Biological Association).

Esthwaite Water is a small to moderate sized lake that drains into the South Basin of Windermere via the Cunsey Beck. It was classified as the most productive lake in the English Lake District when Pearsall made his original trophic classification (Pearsall, 1921). In 2005 Esthwaite had the second highest annual average concentration of TP, nitrate, sulphate, chloride and calcium and alkalinity and the highest concentration of potassium (Table 3.25).

Different ways of assessing trophic status give categories of mesotrophic or eutrophic, although the mean concentration of TP is only just in the mesotrophic category (Tables 3.4 and 3.14). The mean phytoplankton chlorophyll *a* and mean Secchi depth both suggest the lake is eutrophic and this is probably the correct classification given the pronounced oxygen depletion at depth (Table 3.14). WFD criteria suggest that the lake is at Moderate ecological status.

Esthwaite Water is studied fortnightly as part of the CEH long-term monitoring programme that was started by the Freshwater Biological Association in 1945 and continued by CEH since 1989.

Table 3.14. Summary of limnological conditions and trophic and Water Framework Directive classifications in Esthwaite Water in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	463		
Mean pH (geometric mean)	7.4		
Mean total phosphorus (mg m^{-3})	31.0	Mesotrophic	Moderate
Mean soluble reactive phosphorus (mg m^{-3})	4.2		
Mean nitrate-nitrogen (mg m^{-3})	456		
Mean silica (mg m^{-3})	1163		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	15.0	Eutrophic	Moderate
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	18.7	Mesotrophic	
Mean Secchi depth (m)	2.4	Eutrophic	
Minimum Secchi depth (m)	1.9	Mesotrophic	
Minimum oxygen concentration (mg m^{-3})	0.1		

3.3.11 Grasmere

Grasmere from Loughrigg Terrace. (Photo: M.M. De Ville).

Grasmere is a fairly small lake with a short retention time at the northern end of the Windermere catchment (Fig. 3.2).

Its trophic status is somewhere on the mesotrophic to eutrophic boundary but it experiences quite pronounced oxygen depletion at depth (Table 3.15). In terms of the WFD, its ecological status is only Moderate for TP but Poor for phytoplankton chlorophyll *a*. In fact, the annual mean phytyoplankton chlorophyll *a* is close to the Bad ecological boundary. A recent review of Grasmere can be found in Reynolds *et al.* (2001).

Grasmere is studied fortnightly as part of the CEH long-term monitoring programme that was started by the Freshwater Biological Association in 1969 and continued by CEH since 1989.

Table 3.15. Summary of limnological conditions and trophic and Water Framework Directive classifications in Grasmere in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	162		
Mean pH (geometric mean)	6.9		
Mean total phosphorus (mg m^{-3})	17.2	Mesotrophic	Moderate
Mean soluble reactive phosphorus (mg m^{-3})	1.9		
Mean nitrate-nitrogen (mg m^{-3})	322		
Mean silica (mg m^{-3})	1035		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	13.3	Eutrophic	Poor
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	27.7	Eutrophic	
Mean Secchi depth (m)	3.2	Mesotrophic	
Minimum Secchi depth (m)	2.1	Mesotrophic	
Minimum oxygen concentration (mg m^{-3})	0.2		

3.3.12 Haweswater

Haweswater is the fifth largest lake in terms of volume and fourth deepest (Table 2.1) It is the most easterly of those studied here (Fig. 2.1). It is a reservoir and was greatly increased in size in about 1930 by the construction of a dam at the north-east end of the lake. As an annual mean it had the third lowest concentration of nitrate and the lowest concentration of chloride of any of the 20 lakes in 2005 (Table 3.25).

Haweswater looking north-east towards the dam. (Photo: S.C. Maberly).

Its trophic status is mesotrophic, tending towards oligotrophic which is consistent with the minimal oxygen depletion at depth (Table 3.16). In terms of the WFD, it has Good ecological status for both TP and phytoplankton chlorophyll *a*.

Table 3.16. Summary of limnological conditions and trophic and Water Framework Directive classifications in Haweswater in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	212		
Mean pH (geometric mean)	7.2		
Mean total phosphorus (mg m^{-3})	10.9	Oligo/Mesotrophic	Good
Mean soluble reactive phosphorus (mg m^{-3})	1.6		
Mean nitrate-nitrogen (mg m^{-3})	176		
Mean silica (mg m^{-3})	1585		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	3.3	Mesotrophic	Good
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	5.8	Oligotrophic	
Mean Secchi depth (m)	3.6	Mesotrophic	
Minimum Secchi depth (m)	2.5	Mesotrophic	
Minimum oxygen concentration (mg m^{-3})	8.2		

3.3.13 Loughrigg Tarn

Loughrigg Tarn. (Photo: M.M. De Ville).

Loughrigg Tarn is in the Windermere catchment. It is the second smallest lake studied here in terms of both area and volume. It has a relatively long retention time for a lake of its size (Table 2.1). It had the third highest annual mean alkalinity and calcium concentration, the lowest mean concentration of silica and the second highest concentration of phytoplankton chlorophyll *a* of any of the 20 lakes in 2005 (Table 3.25).

Loughrigg Tarn has a range of trophic state assessments depending on which feature is used. The mean TP concentration and minimum Secchi depth suggest that the tarn is mesotrophic whereas other measures suggest the tarn is eutrophic and the mean phytoplankton chlorophyll *a* concentration is just below the hypertrophic category (Table 3.17). On balance, Loughrigg Tarn is probably eutrophic and this is consistent with the substantial oxygen depletion at depth (Table 3.17). The classification in terms of the WFD suggests that Loughrigg Tarn has Moderate ecological status in terms of TP but only Poor ecological status for phytoplankton chlorophyll *a*. This may be result from the relatively long retention time that reduces hydraulic loss of phytoplankton populations.

Table 3.17. Summary of limnological conditions and trophic and Water Framework Directive classifications in Loughrigg Tarn in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	347		
Mean pH (geometric mean)	7.2		
Mean total phosphorus (mg m^{-3})	19.5	Mesotrophic	Moderate
Mean soluble reactive phosphorus (mg m^{-3})	1.3		
Mean nitrate-nitrogen (mg m^{-3})	348		
Mean silica (mg m^{-3})	700		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	24.6	Eu /Hypertrophic	Poor
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	48.5	Eutrophic	
Mean Secchi depth (m)	2.6	Eutrophic	
Minimum Secchi depth (m)	2.0	Mesotrophic	
Minimum oxygen concentration (mg m^{-3})	0.1		

3.3.14 Loweswater

Loweswater is a moderate to small lake in the north-west of the English Lake District that drains into Crummock Water Fig. 2.1). It has a relatively long retention time for a lake of its size (Table 2.1). As annual mean for the 20 lakes in 2005, Loweswater had the highest concentration of sulphate and third highest concentration of nitrate (Table 3.25).

Loweswater. (Photo M.M. De Ville).

Loweswater is close to the mesotrophic-eutrophic boundary, probably tending to be eutrophic given the complete oxygen depletion at depth (Table 3.18). In terms of the WFD, Loweswater is classified as Moderate for TP but 'Poor' for phytoplankton chlorophyll *a*. This may result from the relatively long retention time for a lake of this size (Table 2.1) which favours the development of phytoplankton biomass by reducing hydraulic loss.

The results of a 12-month study of Loweswater are given in Maberly *et al*. (2006).

Table 3.18. Summary of limnological conditions and trophic and Water Framework Directive classifications in Loweswater in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	223		
Mean pH (geometric mean)	7.1		
Mean total phosphorus (mg m^{-3})	15.4	Mesotrophic	Good
Mean soluble reactive phosphorus (mg m^{-3})	0.6		
Mean nitrate-nitrogen (mg m^{-3})	386		
Mean silica (mg m^{-3})	1530		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	12.1	Eutrophic	Moderate
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	20.6	Mesotrophic	
Mean Secchi depth (m)	2.8	Eutrophic	
Minimum Secchi depth (m)	2.0	Mesotrophic	
Minimum oxygen concentration (mg m^{-3})	0.0		

3.3.15 Rydal Water

Rydal Water is a small lake that receives water from the slightly larger Grasmere less than 1 km upstream (Fig. 2.1). Rydal Water eventually flows into the River Rothay and thence into the North Basin of Windermere. As an annual mean Rydal Water had the second lowest concentration of silica (Table 3.25).

Rydal Water. (Photo: M.M. De Ville).

Rydal Water appears to be on the mesotrophic/eutrophic boundary, probably tending towards eutrophic since there is a substantial depletion of oxygen at depth (Table 3.19). In terms of the WFD, the mean concentration of TP just falls into the Good category, but the annual mean concentration of phytoplankton chlorophyll *a* just falls into the Poor ecological state. It is not clear why there should be such a disparity between these two measures, but one possibility is that algae that have grown in Grasmere may be transported into Rydal Water.

Table 3.19. Summary of limnological conditions and trophic and Water Framework Directive classifications in Rydal Water in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	178		
Mean pH (geometric mean)	7.1		
Mean total phosphorus (mg m^{-3})	15.5	Mesotrophic	Good
Mean soluble reactive phosphorus (mg m^{-3})	2.2		
Mean nitrate-nitrogen (mg m^{-3})	268		
Mean silica (mg m^{-3})	785		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	14.7	Eutrophic	Poor
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	23.5	Meso/Eutrophic	
Mean Secchi depth (m)	2.9	Eutrophic	
Minimum Secchi depth (m)	1.5	Meso/Eutrophic	
Minimum oxygen concentration (mg m^{-3})	0.1		

3.3.16 Thirlmere

Thirlmere (Photo: V. Moss).

Thirlmere is a moderate sized lake in the centre of the English Lake District and is part of the Bassenthwaite catchment (Fig. 2.1). It is dammed at its northern end and is a reservoir and as a result experiences quite marked changes in water level. As an annual mean in 2005, Thirlmere had the second lowest concentration of nitrate and sulphate and the lowest concentration of magnesium and potassium (Table 3.25).

Thirlmere is clearly oligotrophic (Table 3. 20) although the mean Secchi depth just falls into the mesotrophic category. This may be the result of dissolved organic carbon as some streams, from the coniferous plantations on the western side, have quite high concentrations of dissolved organic carbon (S.C. Maberly, unpublished). Tipping *et al.* (1988) found that Thirlmere had a relatively high absorbance at 340 nm compared to other unproductive lakes in the region. There is virtually no oxygen depletion at depth which is consistent with its oligotrophic nature. The ecological status in terms of the Water Framework Directive suggests that Thirlmere is in a Good or High ecological state (Table 3.20).

Table 3.20. Summary of limnological conditions and trophic and Water Framework Directive classifications in Thirlmere in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	83		
Mean pH (geometric mean)	6.7		
Mean total phosphorus (mg m^{-3})	9.2	Oligotrophic	Good
Mean soluble reactive phosphorus (mg m^{-3})	0.7		
Mean nitrate-nitrogen (mg m^{-3})	167		
Mean silica (mg m^{-3})	1548		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	2.1	Oligotrophic	High
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	3.1	Oligotrophic	
Mean Secchi depth (m)	5.5	Mesotrophic	
Minimum Secchi depth (m)	5.2	Oligotrophic	
Minimum oxygen concentration (mg m^{-3})	9.5		

3.3.17 Ullswater

Ullswater is the second largest lake in the English Lake District after Windermere in terms of area and volume and the largest if Windermere is separated into two basins. It is situated in the north-east of the English Lake District and drains eventually into the River Eden (Fig. 2.1). As an annual mean, Ullswater had the second lowest concentration of chloride of any of the 20 lakes (Table 3.25).

Ullswater (Photo D.R. Harvey).

Ullswater is a mesotrophic lake with some features tending towards oligotrophic (Table 3.21). In terms of the WFD, the lake has a Good ecological status.

Table 3.21. Summary of limnological conditions and trophic and Water Framework Directive classifications in Ullswater in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	254		
Mean pH (geometric mean)	7.3		
Mean total phosphorus (mg m^{-3})	9.8	Oligo/Mesotrophic	Good
Mean soluble reactive phosphorus (mg m^{-3})	1.9		
Mean nitrate-nitrogen (mg m^{-3})	216		
Mean silica (mg m^{-3})	1442		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	4.5	Mesotrophic	Good
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	6.4	Oligotrophic	
Mean Secchi depth (m)	4.5	Mesotrophic	
Minimum Secchi depth (m)	3.0	Oligo/Mesotrophic	
Minimum oxygen concentration (mg m^{-3})	4.5		

3.3.18 Wastwater

Wastwater. (Photo: I.J. Winfield).

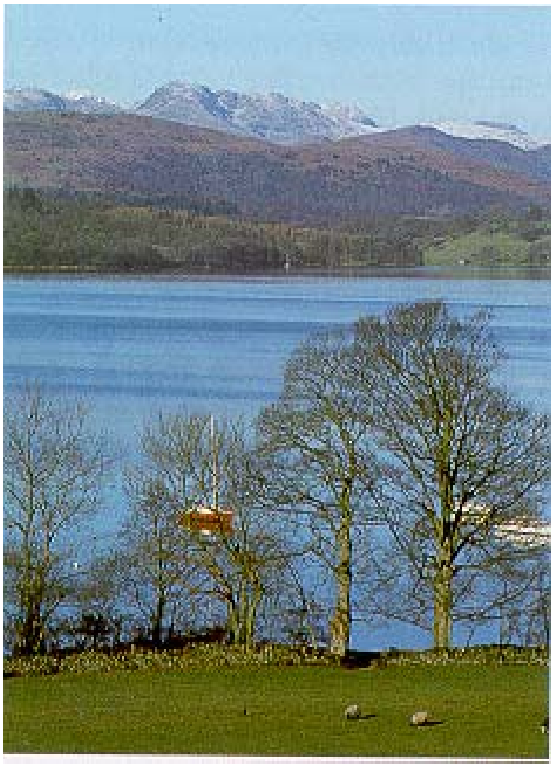
Wastwater is the third largest lake in the English Lake District in terms of volume, but only the tenth largest in terms of area. The difference results from the great average depth of the lake with the greatest mean depth (40 m) and maximum depth (76 m; Table 2.1) of any lake in the English Lake District. Wastwater is also the prime example of an oligotrophic lake in the region and was the most unproductive lake in the lake series devised by Pearsall (1921). As an annual mean in 2005, Wastwater had the lowest concentration of TP, the third lowest alkalinity and the second highest concentration of silica (Tables 3.22, 3.25).

It is ultra-oligotrophic or oligotrophic in terms of its trophic state and both TP and phytoplankton chlorophyll *a* indicate it is in a High ecological state in terms of the WFD.

Table 3.22. Summary of limnological conditions and trophic and Water Framework Directive classifications in Wastwater in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	66		
Mean pH (geometric mean)	6.8		
Mean total phosphorus (mg m^{-3})	3.9	Ultra-oligotrophic	High
Mean soluble reactive phosphorus (mg m^{-3})	0.5		
Mean nitrate-nitrogen (mg m^{-3})	318		
Mean silica (mg m^{-3})	2305		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	1.2	Oligotrophic	High
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	1.3	Ultra-oligotrophic	
Mean Secchi depth (m)	10.1	Oligotrophic	
Minimum Secchi depth (m)	9.5	Ultra-oligotrophic	
Minimum oxygen concentration (mg m^{-3})	9.8		

3.3.19 Windermere North Basin



The North Basin of Windermere (Photo: The Freshwater Biological Association).

Windermere is the largest lake in the English Lake District and the largest natural lake in England. Limnologically it is divided into a larger North Basin and a slightly smaller South Basin, separated by shallow water and islands. The North Basin has the second-largest maximum and mean depth, area and volume of any of the lakes studied.

The North Basin is mesotrophic with a slight hint of it being meso/eutrophic (Table 3.23). There is a relatively modest oxygen depletion, consistent with its mesotrophic status. In terms of the WFD however, the annual mean concentration of TP is only Moderate and categorised by phytoplankton chlorophyll *a* it is only Poor.

The North Basin of Windermere is studied fortnightly as part of the CEH long-term monitoring programme that was started by the Freshwater Biological Association in 1945 and continued by CEH since 1989.

Table 3.23. Summary of limnological conditions and trophic and Water Framework Directive classifications in the North Basin of Windermere in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	248		
Mean pH (geometric mean)	7.3		
Mean total phosphorus (mg m^{-3})	14.2	Mesotrophic	Moderate
Mean soluble reactive phosphorus (mg m^{-3})	2.0		
Mean nitrate-nitrogen (mg m^{-3})	332		
Mean silica (mg m^{-3})	838		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	8.6	Eu/Mesotrophic	Poor
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	19.3	Mesotrophic	
Mean Secchi depth (m)	4.1	Mesotrophic	
Minimum Secchi depth (m)	2.6	Mesotrophic	
Minimum oxygen concentration (mg m^{-3})	7.9		

3.3.20 Windermere South Basin

The South Basin of Windermere is about half the volume and 80% of the area of the North Basin. In addition to receiving water from the North Basin, Esthwaite Water flows into the South Basin via Cunsey Beck.

South Basin of Windermere. (Photo: Freshwater Biological Association).

The South Basin of Windermere is more productive than the North Basin and is categorised as somewhere between mesotrophic and eutrophic. Its status in terms of the WFD is, like the North Basin, only Moderate and Poor for concentration of TP and phytoplankton chlorophyll *a* respectively.

The South Basin of Windermere is studied fortnightly as part of the CEH long-term monitoring programme that was started by the Freshwater Biological Association in 1945 and continued by CEH since 1989.

Table 3.24. Summary of limnological conditions and trophic and Water Framework Directive classifications in the South Basin of Windermere in 2005.

Characteristic	Value	Trophic	WFD
Mean alkalinity (mequiv m^{-3})	290		
Mean pH (geometric mean)	7.4		
Mean total phosphorus (mg m^{-3})	15.8	Mesotrophic	Moderate
Mean soluble reactive phosphorus (mg m^{-3})	4.8		
Mean nitrate-nitrogen (mg m^{-3})	350		
Mean silica (mg m^{-3})	1015		
Mean phytoplankton chlorophyll *a* (mg m^{-3})	10.2	Eutrophic	Poor
Maximum phytoplankton chlorophyll *a* (mg m^{-3})	19.6	Mesotrophic	
Mean Secchi depth (m)	3.5	Mesotrophic	
Minimum Secchi depth (m)	2.4	Mesotrophic	
Minimum oxygen concentration (mg m^{-3})	5.1		

3.3.21 Summary of the lakes in 2005.

The annual mean (for oxygen minimum at depth) values for each lake in 2005 are summarised in Table 3.25. Raw values are given in the appendices.

Table 3.25. Annual mean (oxygen minimum at depth) for the 20 lakes of the Lakes Tour in 2005. Note that pH is calculated as the geometric mean.

Lake	TP (mg m^{-3})	SRP (mg m^{-3})	NO_3-N (mg m^{-3})	SiO_2 (g m^{-3})	Chl *a* (mg m^{-3})	Secchi (m)	Min O_2 (g m^{-3})	pH	Alk (mequiv m^{-3})	Cl (mequiv m^{-3})	SO_4 (mequiv m^{-3})	Na (mequiv m^{-3})	K (mequiv m^{-3})	Ca (mequiv m^{-3})	Mg (mequiv m^{-3})
Bassenthwaite Lake	20.4	2.5	279	1.29	14.4	2.3	0.58	7.1	206	311	89.8	246	13.9	272	92.9
Blelham Tarn	33.0	2.3	547	1.00	19.0	2.1	0.10	7.2	474	375	109.8	285	24.0	579	146.3
Brothers Water	11.6	0.5	230	1.77	2.5	5.4	0.16	6.9	187	206	74.5	178	6.0	231	65.6
Buttermere	8.2	0.2	187	1.53	1.7	7.7	7.36	6.7	66	213	58.8	165	5.7	116	60.8
Coniston Water	10.5	1.7	328	0.93	5.3	5.3	5.89	7.2	213	274	101.7	234	13.1	281	84.0
Crummock Water	9.7	0.6	228	1.48	2.0	7.3	6.18	6.7	63	213	71.1	174	7.7	108	69.4
Derwent Water	13.3	0.8	148	0.96	6.9	3.8	1.81	6.9	109	339	53.6	237	8.1	203	58.1
Elterwater	23.3	2.6	273	1.49	34.4	2.5	0.00	6.9	300	294	66.1	231	11.8	354	88.5
Ennerdale Water	7.2	1.4	319	2.49	1.5	8.6	8.92	6.6	50	217	67.0	186	7.8	90	70.0
Esthwaite Water	31.0	4.2	456	1.16	15.0	2.4	0.08	7.4	463	375	110.9	302	25.6	547	121.7
Grasmere	17.2	1.9	322	1.04	13.3	3.2	0.16	6.9	162	241	70.3	197	9.8	215	64.4
Haweswater	10.9	1.6	176	1.59	3.3	3.6	8.22	7.2	212	152	65.3	143	8.1	228	78.5
Loughrigg Tarn	19.5	1.3	348	0.70	24.6	2.6	0.10	7.2	347	273	80.5	211	17.4	367	116.0
Loweswater	15.4	0.5	386	1.53	12.1	2.8	0.01	7.1	224	312	112.3	261	19.1	270	124.8
Rydal Water	15.5	2.2	268	0.79	14.7	2.9	0.10	7.1	178	245	70.2	204	9.8	224	67.9
Thirlmere	9.2	0.7	167	1.55	2.1	5.5	9.51	6.7	83	174	55.9	150	5.2	122	44.4
Ullswater	9.8	1.9	216	1.44	4.5	4.5	4.54	7.3	254	171	75.3	161	9.4	266	85.4
Wastwater	3.9	0.5	318	2.31	1.2	10.1	9.84	6.8	66	181	65.0	162	6.5	105	59.4
Windermere North Basin	14.2	2.0	332	0.84	8.7	4.1	7.85	7.3	248	246	85.5	208	13.2	292	81.0
Windermere South Basin	15.8	4.8	350	1.02	10.2	3.5	5.09	7.4	290	252	90.6	221	14.9	329	86.7

4. Discussion

4.1 Spatial patterns of change across lakes

The English Lake District is unusual as, among the major lakes, there is a large range of lake types in terms of depth, size, hydrology, basic water chemistry and trophic state within a small geographic area. This results essentially from the varied geology in the region (Fig. 3.5) but also from the varied land-use and the altitude and morphology of individual catchments. Furthermore it is extremely fortunate that these 20 lakes have been studied in a reasonably consistent way since 1984, and some for much longer, so that comparisons can be made across years as well as types. This gives an excellent opportunity to analyse and illustrate the inter-relationships between various limnological variables to help understand how lakes function and respond to environmental perturbation.

An example of the importance of the catchment in determining the ecology of the lake is shown in Figure 4.1 where concentrations of potassium, alkalinity, total phosphorus and nitrate all decline with altitude. All the correlations are significant at $P<0.001$. Altitude is not likely to be the direct cause of the relationship but is probably correlated with erodability of rock, accumulation of ions because the water has travelled through more geology and soil and changes in land-use and soil types. Water chemistry has been shown to be closely linked to land-use in small upland tarns (Maberly *et al.*, 2003) and it likely that this is a key factor in these larger lakes as well.

Although nitrogen may be an equally important limiting nutrient in certain types of lakes (Maberly *et al.* 2002; James *et al.*, 2003) in the large lakes of the English Lake District studied here, phosphorus is the key nutrient limiting phytoplankton production. This is apparent from Figure 4.2 where concentrations of phytoplankton chlorophyll *a* are closely linked to the concentration of total phosphorus.

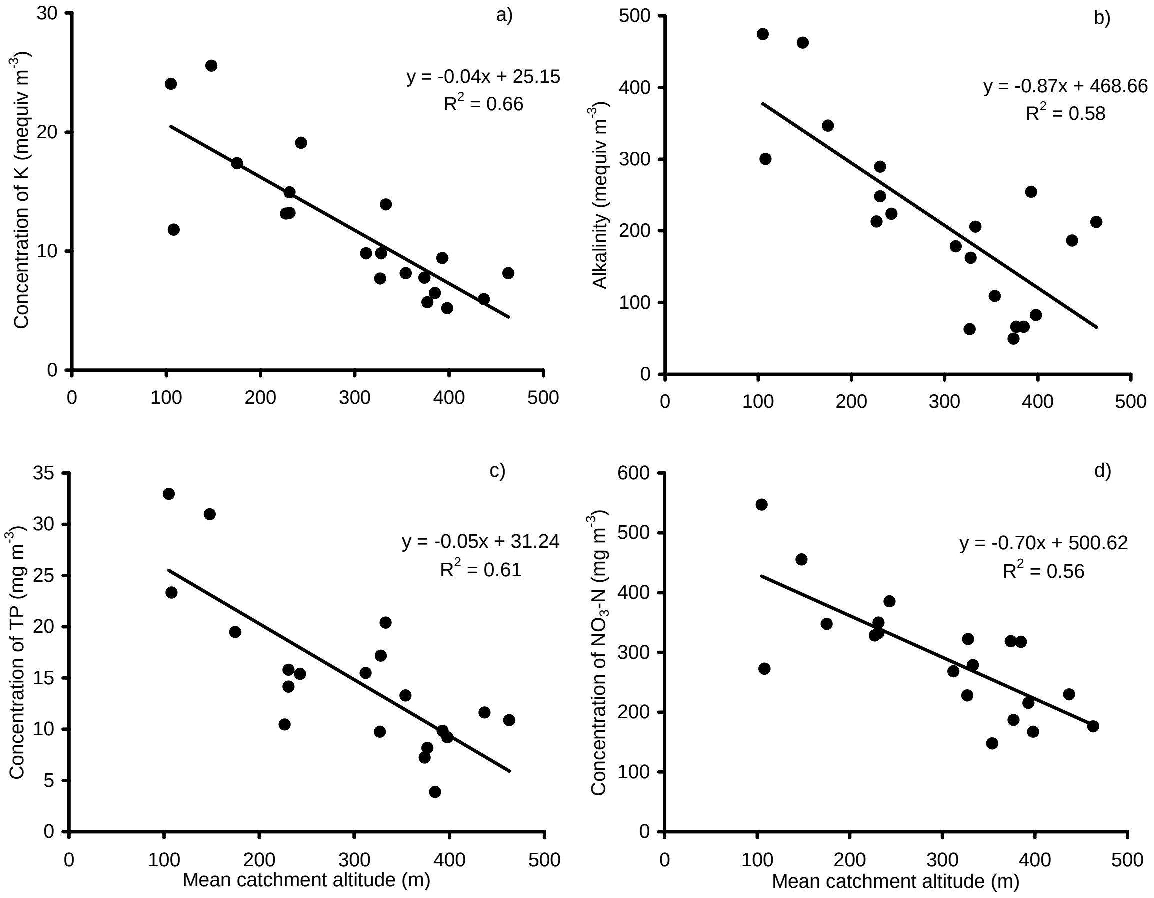


Figure 4.1. Relationships between annual mean concentrations of two major ions (a) potassium; (b) alkalinity) and two nutrients (c) total phosphorus and (d) nitrate-nitrogen) and mean catchment altitude. Data from 2005 Lakes Tour.

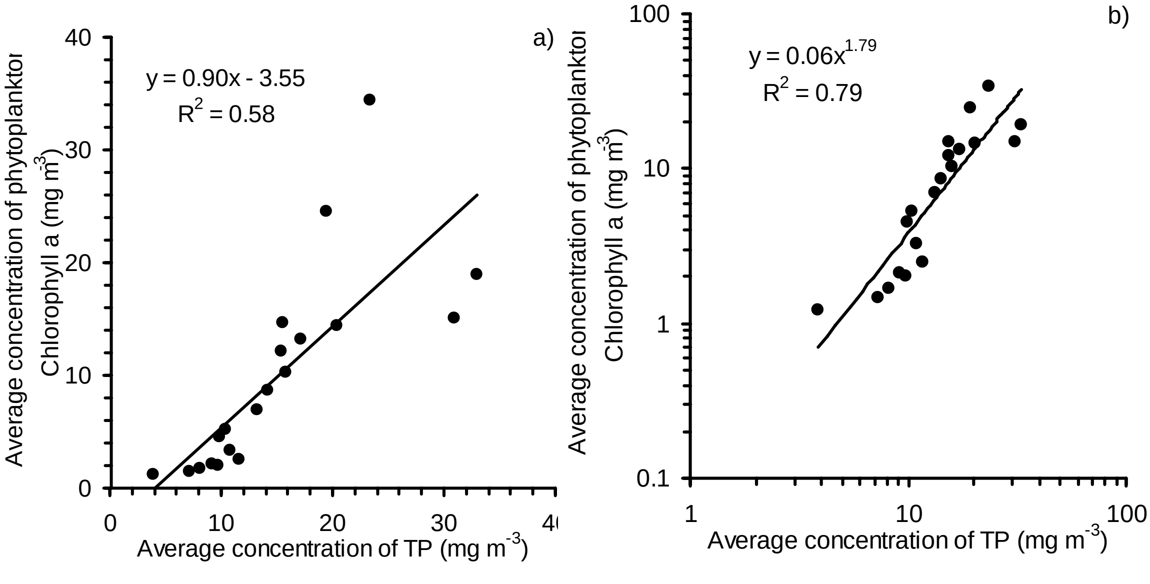


Figure 4.2. Relationship between average concentration of phytoplankton chlorophyll a *and total phosphorus plotted on: a) a linear scale and b) a log-log scale. Data from 2005 Lakes Tour.*

The greater the productivity of lake, the greater effect the biology has on the seasonal dynamics of a lake. Figure 4.3 shows how in productive lakes, with high concentrations of TP such as Esthwaite Water and Blelham Tarn, seasonal changes in concentrations of nitrate and silica are great, but in unproductive lakes such as Waterwater and Ennerdale Water there is very little seasonal change in these two other nutrients. Analysing seasonal changes in concentrations of nitrate and silica is, therefore, a useful additional method to describe the productivity of a lake. They have the advantage of being more conservative than soluble reactive phosphorus and being closer to an available nutrient than TP which often shows relatively little seasonal change.

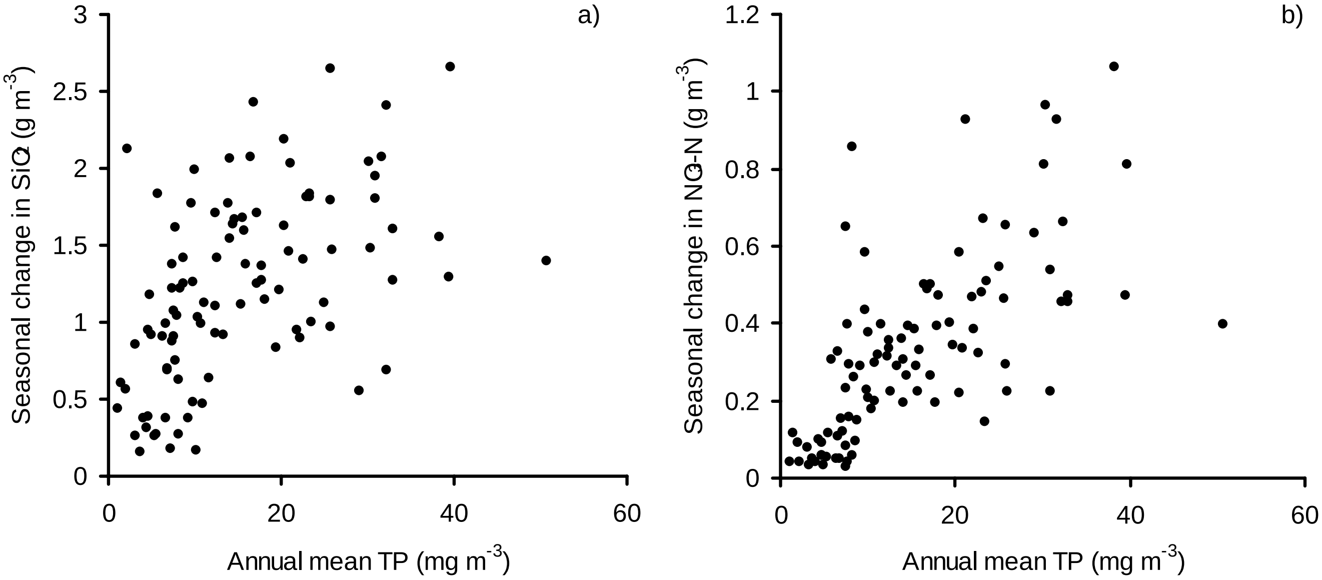


Figure 4.3. Seasonal change (annual maximum minus annual minimum) in concentration of: a) silica and b) nitrate as a function of annual mean TP for the 20 lakes in 1984, 1991, 1995, 2000 and 2005.

The greater productivity of these types of lakes is expressed in the amount of phytoplankton chlorophyll *a* (Fig. 4.2). This in turn has a number of consequences for the limnology of a lake. An obvious consequence is that a large population of phytoplankton reduces water clarity. Figure 4.4 show this relationship for data between 1991 and 2005 categorised per month. The responses in April, July and October are very similar and clearly dominated by the phytoplankton with the proportion of the variance accounted for (R^2) ranging from 0.59 to 0.80. There was a slightly different response in January: Secchi depth tended to be lower for the same concentration of phytoplankton chlorophyll *a* and the proportion of the variance accounted for (R^2) was lower at 0.29. This is probably the result of non-phytoplankton material such as suspended solids, being relatively more important in January than in the three other months, because phytoplankton populations are generally at their lowest and winter rains will bring in suspended solids from the catchment.

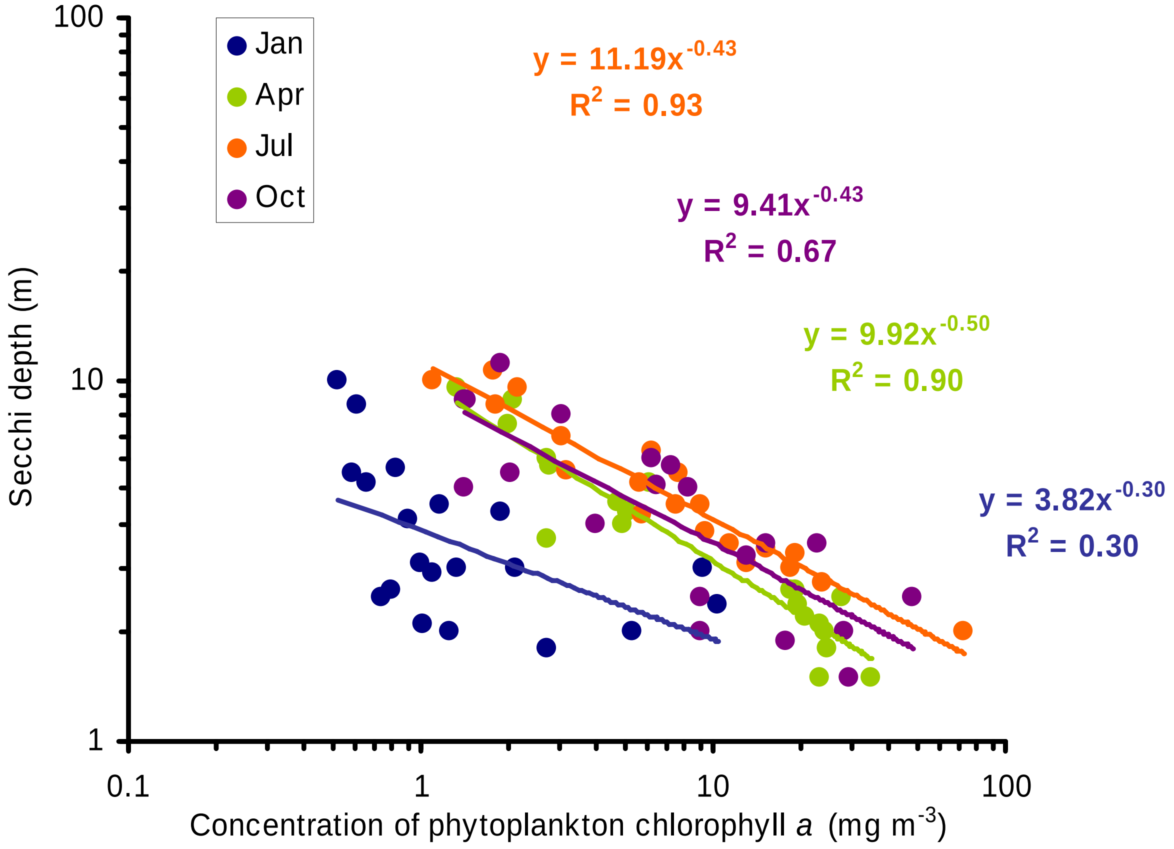


Figure 4.4. Relationship between Secchi depth and concentration of phytoplankton chlorophyll a *in the 20 lake basins and four sampling occasions in the 1991, 1995, 2000 and 2005 Lakes Tours. The best-fit line is shown with equation and proportion of the variance accounted for (R^2).*

Another consequence of increased phytoplankton productivity is the depletion of oxygen at depth in the summer. This has a number of potentially undesirable consequences. First, oxygen depletion changes the redox potential at the sediment surface converting iron from ferric (Fe^{3+}) which is able to bind phosphorus to ferrous (Fe^{2+}) which does not, and as a result phosphorus bound in the sediment may be released into the water column causing a positive feedback increasing nutrient enrichment.

Secondly, low oxygen concentrations at depth will limit the ability of fish to exploit these depths and this, in turn, may have negative consequences, especially for fish, such as the Charr, which are believed to avoid the higher water temperatures in the epilimnion. It is important to note that the peak oxygen depletion in the lakes of the English Lake District typically occurs between the end of August and the start of October, depending on when exactly stratification breaks down. Therefore the oxygen minimum measured in the Lakes Tour, which generally occurred in July, but occasionally in October, will be an underestimate of the true extent of oxygen depletion. Furthermore, a number of factors other than productivity will influence the extent of oxygen depletion such as the ratio of volume of water in the epilimnion to the volume of water in the hypolimnion. Nevertheless, the data show a clear negative relationship between oxygen concentration at depth and phytoplankton chlorophyll *a*. Lakes with very little phytoplankton have oxygen minima which approach those at air-equilibrium and lakes where the annual mean concentration of phytoplankton *a* exceeds about 10 mg m^{-3} have complete oxygen depletion at depth (Fig. 4.5a). The relationship is improved further if the mean concentration in the hypolimnion is estimated, rather then the annual minimum (Fig. 4.5b). The concentration was estimated as the unweighted mean oxygen concentration in the hypolimnion, the depth of which was determined subjectively by inspection of the temperature-depth plots in Figure 3.2).

Finally, there also appears to be a link between the species richness of phytoplankton and lake productivity, at least over the range of lakes studied here. Figure 4.6 shows a statistically significant increase in phytoplankton species richness with the annual mean concentration of TP.

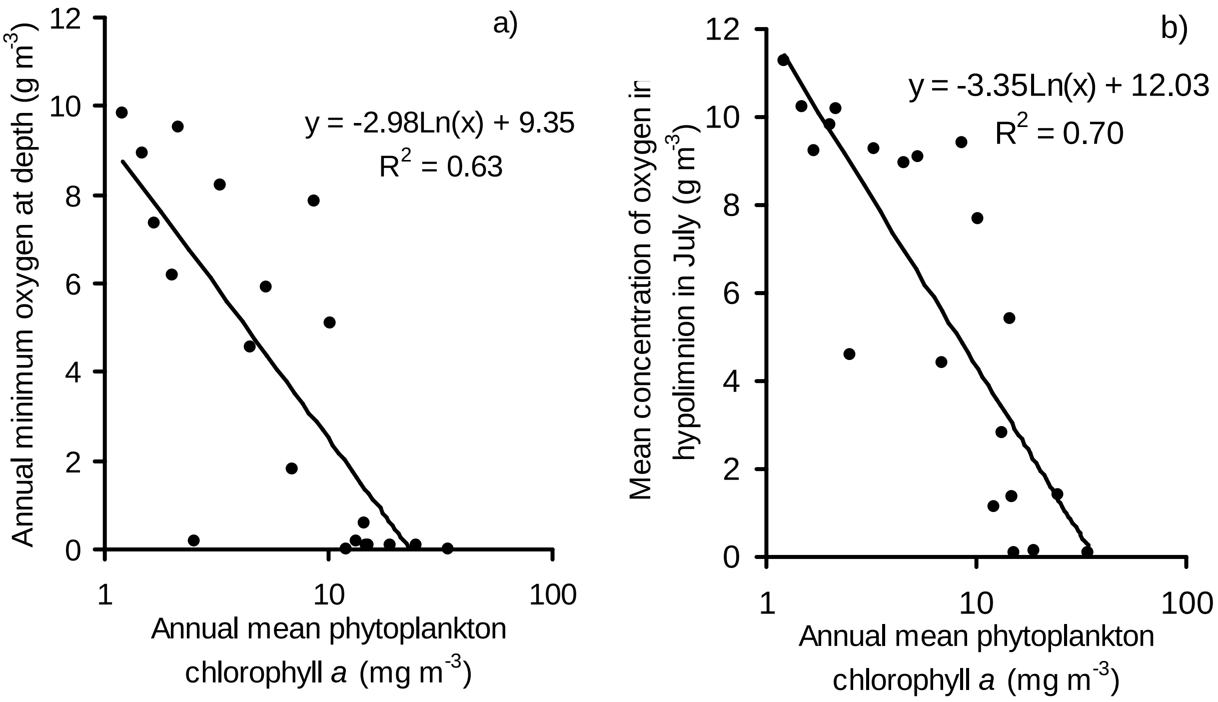


Figure 4.5. Data for 2005 showing the relationship between concentration of oxygen at depth and the annual mean phytoplankton chlorophyll a *(on a log scale). The data are plotted: a) as the annual minimum and b) as an average for the hypolimnion (see text).*

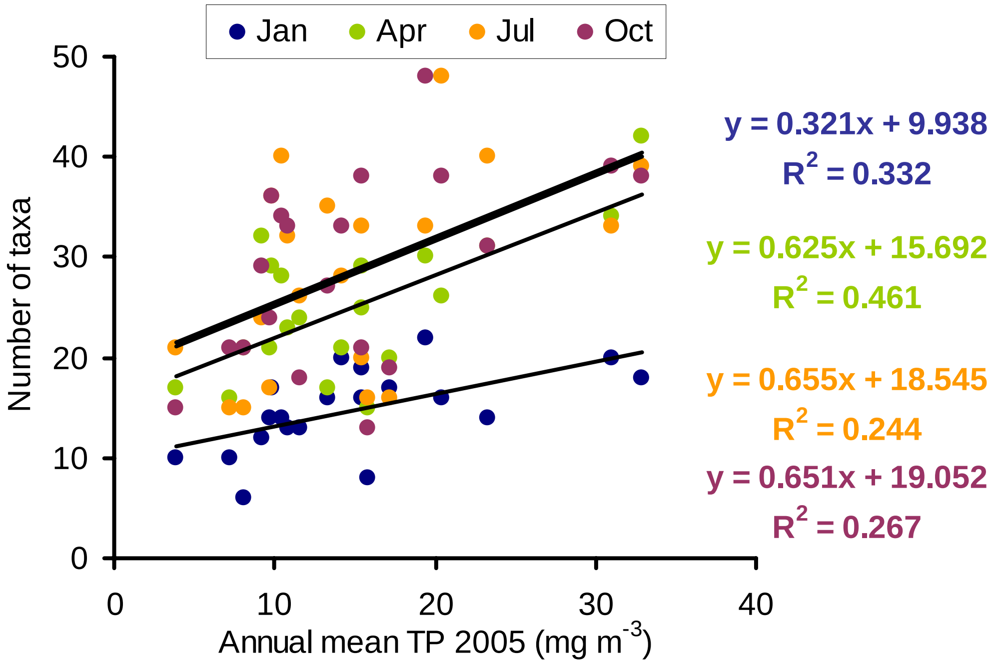


Figure 4.6. Relationship between number of phytoplankton taxa and annual mean concentration of TP as a measure of phytoplankton for the four seasons on 2005.

4.2 Temporal records of change

In this section, records from the 2005 Lakes Tour are compared to those in 1991, 1995 and 2000 and, where possible, 1984 to assess the extent of any change in these lakes. For all the correlations, especially for chlorophyll *a* and Secchi depth where values were not recorded in 1984, there are a limited number of degrees of freedom and so some changes may be real even though they are not statistically significant. Changes are reported on a lake-by-lake basis.

4.2.1 Bassenthwaite Lake

The only statistically significant changes in nutrient chemistry are a decrease in winter concentrations of nitrate and silicate (Table 4.1). During this period there have been small detectable changes in the lake resolved by more-detailed fortnightly sampling (Thackeray *et al*., 2004, 2006). None of the major ions or pH showed a statistically significant change in concentration (Table 4.2). There has not been a statistically significant increase in concentration of phytoplankton chlorophyll *a*, but there are indications of an increase in April and July and of a decline in Secchi depth in April (Table 4.3). Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.2 Blelham Tarn

There have been no statistically significant changes in nutrient chemistry, major ions, phytoplankton chlorophyll *a* or Secchi depth in Blelham Tarn apart from a significant decrease in concentration of potassium in summer (Tables 4.1, 4.2, 4.3). The lake therefore appears to be fairly stable. Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.3 Brothers Water

The only significant change in the nutrient chemistry of Brothers Water has been an increase in summer concentration of silica (Table 4.1). Summer concentrations of silica in 2005 were particularly high (Appendix 4) and this probably resulted from the very few diatoms present (Fig. 3.15). There was a significant increase in alkalinity in Brothers Water (Table 4.2) but this is probably not ecologically significant because alkalinity in Brothers Water is very variable, probably because it is rapidly flushed and may receive water from different hydrological paths. Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.4 Buttermere

There are no statistically significant changes in nutrient chemistry in Buttermere (Table 4.1) however there has been a statistically significant increase in pH in spring and autumn and an increase in alkalinity in autumn (Table 4.2). Concentrations of calcium have declined in winter and the Secchi transparency has declined in summer (Table 4.3). The changes in alkalinity and pH may reflect a decline in the impact of acid-deposition, particularly that caused by sulphur deposition which has declined and caused a reversal of acidification in some poorly buffered lakes and streams in Cumbria (Tipping *et al.*, 1998). Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.5 Coniston Water

Coniston Water has experienced a slight but not statistically significant increase in concentration of TP between 1984 and 2005 (Table 4.1). There has been a marked increase in alkalinity in three seasons apart from winter (Table 4.2). Concentrations of phytoplankton chlorophyll *a* have increased in autumn (Table 4.3) Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.6 Crummock Water

There has been a slight but not statistically significant increase in TP concentration between 1984 and 2005 (Table 4.1). The significant increase in ammonium concentration has no ecological significance and concentrations are low or below the detection limit in all samples analysed. There have been no changes in concentration of phytoplankton chlorophyll *a* or Secchi depth. (Table 4.3). Crummock Water appears to be in a stable state. Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.7 Derwent Water

Derwent Water is basically stable but there are some hints of slight nutrient enrichment. Concentrations of TP have increased slightly, but not statistically significantly, in spring and autumn (Table 4.1). This is consistent with a decline in concentrations of nitrate in the spring as a result of increased demand and an increase in summer and autumn alkalinity as uptake of nitrate by phytoplankton causes alkalinity to increase. However the magnitude of alkalinity increase is greater than be explained by the decline in nitrate so other factors such as reduced acid-deposition may also be responsible for the increase in alkalinity in Derwent Water. Further evidence for a slight nutrient enrichment in Derwent Water is given by the slight but statistically significant increase in phytoplankton chlorophyll *a* in summer from 5.1 mg m^{-3} in 1991 to 7.5 mg m^{-3} in 2005 (Table 4.3). Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.8 Elterwater

The only statistically significant long-term change in Elterwater is a decline in the concentration of silica in summer (Table 4.1). However, it is clear from the long-term changes in annual mean concentration of TP that there has been a marked reduction in concentration since a peak in 1995 of 138 mg m^{-3}: the concentration in 2005 is now lower than the first record in 1984 (Fig. 4.7). At least some of this improvement will have resulted from re-routing the sewage outfall from Elterwater to the River Brathay below the lake. Although the concentration of TP has decreased, this does not appear to have resulted in a decline in concentration of phytoplankton chlorophyll *a* (Fig. 4.8) although Secchi depth was marginally greater in 2005 than in 2000 (Fig. 4.9). Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

Table 4.1 Correlation coefficient of change in nutrient chemistry for the 20 lake basins and four seasons between 1984 and 2005. Significant correlations are shown in bold and shaded green when $P<0.05$, yellow when $P<0.01$ and orange when $P<0.001$. Data below detection limit, so not analysed, indicated by '-'.

Lake	Season	Total P	SRP	NO_3-N	NH_4-N	SiO_2
Bassenthwaite Lake	Winter	-0.09	0.24	**-0.91**	-0.53	**-0.97**
	Spring	0.47	-0.78	-0.42	-	0.08
	Summer	-0.72	0.84	-0.37	-	0.21
	Autumn	0.46	0.53	-0.69	-	0.11
Blelham Tarn	Winter	-0.02	0.63	-0.79	-	-0.10
	Spring	0.03	-0.33	-0.87	-0.47	-0.21
	Summer	0.70	0.05	-0.79	-0.64	-0.49
	Autumn	0.64	0.34	0.15	-0.70	-0.64
Brothers Water	Winter	-0.02	-0.04	0.26	-0.55	0.48
	Spring	0.78	0.51	-0.23	-	0.38
	Summer	0.58	0.14	-0.49	0.23	**0.90**
	Autumn	0.20	-0.78	-0.38	0.32	-0.32
Buttermere	Winter	-0.78	0.03	0.72	-0.53	-0.53
	Spring	-0.44	-0.34	0.12	-	-0.76
	Summer	0.59	-0.69	-0.58	0.71	-0.64
	Autumn	0.87	-0.28	0.35	0.78	-0.55
Coniston Water	Winter	-0.05	0.87	0.09	-	0.76
	Spring	-0.04	0.70	-0.58	-	0.52
	Summer	0.74	0.41	-0.52	0.87	0.78
	Autumn	0.71	-0.61	-0.82	-	-0.70
Crummock Water	Winter	-0.48	-0.09	0.73	-	0.33
	Spring	0.05	0.22	0.68	-	0.32
	Summer	0.76	-0.69	-0.32	**0.90**	0.68
	Autumn	0.60	0.69	0.74	-	-0.14
Derwent Water	Winter	0.14	-0.13	-0.70	0.61	-0.87
	Spring	0.53	-0.69	**-0.98**	-	-0.84
	Summer	-	-0.61	-0.84	-	0.15
	Autumn	0.79	0.69	0.38	0.36	-0.00
Elterwater	Winter	-0.20	0.17	-0.52	0.00	0.43
	Spring	-0.04	0.03	0.24	0.57	-0.36
	Summer	-0.31	0.08	-	-0.79	**-0.97**
	Autumn	-0.06	0.00	-0.03	-0.37	0.69
Ennerdale Water	Winter	-0.36	-0.56	0.19	0.62	-0.51
	Spring	-0.32	-0.11	-0.66	-	-0.70
	Summer	0.79	0.80	**-0.92**	-	-0.11
	Autumn	0.67	-0.77	**-0.89**	0.75	-0.71
Esthwaite Water	Winter	0.69	0.62	**-0.93**	-	0.43
	Spring	0.25	0.02	-0.77	-	0.71
	Summer	0.53	-0.20	-0.78	0.62	0.02
	Autumn	0.27	-0.74	**-0.98**	-0.61	-0.11
Grasmere	Winter	-0.17	0.50	0.50	-	-0.51
	Spring	-0.41	0.39	0.50	-	-0.07
	Summer	0.17	0.34	0.19	**0.91**	0.20
	Autumn	-0.59	-0.45	0.43	-0.56	-0.20
Haweswater	Winter	-0.17	0.18	0.41	-	-0.61
	Spring	0.25	-0.03	-0.81	-	0.71
	Summer	0.43	**0.95**	-0.81	-	0.26
	Autumn	0.56	-0.58	-0.59	-	-0.19
Loughrigg Tarn	Winter	-0.16	0.43	-0.22	0.38	0.54
	Spring	-0.57	-0.19	0.81	-	-0.25
	Summer	-0.84	-0.50	-	-0.40	-0.08
	Autumn	-0.28	-0.79	-0.58	-0.73	-0.59
Loweswater	Winter	0.11	0.30	-0.67	0.30	0.66
	Spring	**0.91**	-0.27	-0.71	-	0.85
	Summer	0.79	-0.69	**-0.96**	0.71	0.66
	Autumn	0.87	0.33	**-0.91**	**0.96**	0.18
Rydal Water	Winter	0.27	0.74	-0.45	-0.65	-0.05
	Spring	-0.53	-0.17	-0.31	-	**-0.92**
	Summer	**0.99**	0.69	-0.11	-0.19	0.14
	Autumn	0.00	**-0.91**	0.27	-0.54	-0.68
Thirlmere	Winter	-0.87	-0.61	0.67	0.34	-0.70
	Spring	0.47	0.19	-0.81	-	-0.76
	Summer	0.63	-0.62	-0.79	-	0.19
	Autumn	0.64	0.68	0.06	-0.40	-0.82
Ullswater	Winter	-0.29	0.29	-0.53	-	0.75
	Spring	-0.11	0.14	-0.15	-	**0.98**
	Summer	0.79	0.05	-0.82	0.80	0.83
	Autumn	0.25	-0.06	-0.26	-	0.46
Wastwater	Winter	-0.08	-0.66	0.78	0.45	-0.31
	Spring	-0.24	-0.02	-0.47	-	0.74
	Summer	0.71	0.86	-0.56	-	-0.19
	Autumn	0.40	-0.86	-0.36	-	-0.45
Windermere North Basin	Winter	0.30	-0.40	-0.72	-	0.11
	Spring	-0.87	0.28	-0.07	0.85	0.22
	Summer	-0.60	0.13	-0.12	0.28	0.20
	Autumn	-0.04	0.75	0.22	-	-0.19
Windermere South Basin	Winter	-0.76	-0.58	-0.67	-	-0.03
	Spring	-0.76	0.47	0.18	-	0.71
	Summer	-0.36	0.01	0.65	-	-0.81
	Autumn	-0.73	-0.12	0.24	-	-0.41

Table 4.2 Correlation coefficient of change in nutrient chemistry for the 20 lake basins and four seasons between 1984 and 2005. Significant correlations are shown in bold and shaded green when P<0.05, yellow when P<0.01 and orange when P<0.001. Data below detection limit, so not analysed, indicated by '-.'

Lake	Season	pH	Alky	SO4	Cl	Ca	Mg	Na	K
Bassenthwaite Lake	Winter	0.78	0.66	-0.84	-0.52	-0.83	-0.71	-0.54	-0.72
	Spring	0.76	0.58	-0.76	-0.80	-0.48	-0.57	-0.55	-0.79
	Summer	-0.67	-0.32	-0.60	-0.80	-0.69	-0.74	-0.69	-0.60
	Autumn	0.33	0.36	-0.55	-0.67	-0.42	-0.75	-0.73	-0.67
Blelham Tarn	Winter	0.37	-0.22	-0.75	-0.09	-0.77	-0.28	-0.20	0.14
	Spring	0.67	0.73	-0.61	-0.36	-0.07	0.15	-0.19	-0.74
	Summer	-0.72	-0.19	-0.49	-0.20	0.03	0.54	-0.39	**-0.98**
	Autumn	-0.69	0.53	-0.46	-0.34	0.20	0.26	-0.58	-0.81
Brothers Water	Winter	0.60	0.56	-0.83	-0.37	-0.65	-0.49	-0.47	-0.81
	Spring	0.03	0.04	-0.74	-0.45	-0.54	-0.40	-0.22	-0.11
	Summer	-0.11	0.47	-0.62	-0.84	-0.72	-0.70	-0.77	-0.80
	Autumn	0.01	**0.92**	-0.67	-0.87	-0.36	-0.60	-0.86	-0.88
Buttermere	Winter	0.59	0.74	-0.70	-0.51	**-0.96**	-0.77	-0.67	-0.16
	Spring	**0.88**	0.64	-0.66	-0.73	-0.62	-0.73	-0.80	0.38
	Summer	0.84	0.61	-0.43	-0.63	-0.68	-0.74	-0.71	-0.80
	Autumn	**0.93**	**0.92**	-0.63	-0.60	-0.34	-0.70	-0.77	-0.67
Coniston Water	Winter	0.88	0.80	-0.83	-0.62	-0.74	-0.64	-0.68	-0.64
	Spring	0.57	**0.94**	-0.69	-0.59	-0.59	-0.86	-0.17	-0.19
	Summer	0.42	**0.90**	-0.59	-0.41	**-0.90**	-0.64	-0.43	-0.01
	Autumn	0.85	**0.92**	-0.68	-0.50	-0.47	-0.45	-0.36	0.12
Crummock Water	Winter	0.54	0.45	-0.71	-0.73	-0.60	**-0.93**	-0.65	-0.14
	Spring	**0.99**	0.87	-0.57	-0.74	-0.61	-0.83	-0.51	0.19
	Summer	0.62	0.65	-0.56	-0.67	-0.71	-0.71	-0.62	-0.60
	Autumn	**0.95**	**0.99**	-0.50	-0.37	-0.54	-0.82	-0.50	-0.76
Derwent Water	Winter	0.60	0.15	-0.83	-0.28	-0.79	-0.63	-0.26	-0.19
	Spring	0.04	0.86	-0.77	-0.72	-0.58	-0.72	-0.66	-0.52
	Summer	0.80	**0.96**	-0.68	-0.78	-0.82	-0.88	-0.76	-0.87
	Autumn	0.75	**0.91**	-0.64	-0.06	0.76	-0.64	-0.18	0.27
Elterwater	Winter	0.45	0.71	-0.66	0.17	0.64	0.87	0.03	0.33
	Spring	-0.12	0.41	-0.77	-0.31	0.17	0.30	-0.30	-0.66
	Summer	-0.19	0.63	-0.74	-0.26	0.67	0.74	-0.27	-0.54
	Autumn	-0.62	0.67	-0.66	-0.30	0.56	0.47	-0.36	-0.13
Ennerdale Water	Winter	-	0.71	-0.75	-0.45	**-0.92**	-0.85	-0.63	-0.38
	Spring	0.50	**0.92**	-0.63	-0.64	-0.48	-0.79	-0.65	-0.40
	Summer	0.81	**0.97**	-0.33	-0.78	-0.80	-0.85	-0.65	-0.87
	Autumn	0.84	**0.99**	-0.67	-0.57	-0.71	-0.81	-0.58	0.14
Esthwaite Water	Winter	0.69	0.79	**-0.89**	-0.41	-0.84	-0.36	-0.28	0.05
	Spring	0.47	**0.89**	-0.76	-0.48	-0.31	-0.30	-0.08	-0.32
	Summer	0.72	0.70	-0.69	-0.59	-0.25	-0.26	-0.42	-0.70
	Autumn	0.45	**0.94**	-0.73	-0.53	0.22	-0.16	-0.51	-0.52
Grasmere	Winter	0.34	0.40	-0.87	-0.41	-0.73	-0.36	-0.50	0.26
	Spring	0.63	0.35	-0.78	-0.77	-0.49	-0.35	-0.66	-0.37
	Summer	-0.27	0.31	-0.68	-0.81	**-0.89**	-0.78	-0.86	-0.75
	Autumn	0.19	0.86	-0.70	-0.77	-0.33	-0.72	**-0.88**	-0.81
Haweswater	Winter	0.79	0.87	-0.76	-0.70	-0.87	-0.88	-0.71	-0.70
	Spring	0.10	0.80	-0.78	**-0.92**	-0.73	-0.84	-0.71	-0.33
	Summer	0.57	**0.92**	-0.71	**-0.97**	-0.76	-0.73	-0.87	-0.74
	Autumn	**0.89**	**0.91**	-0.78	**-0.93**	-0.45	-0.71	**-0.88**	-0.02
Loughrigg Tarn	Winter	0.53	0.76	-0.83	-0.37	**-0.92**	-0.02	-0.41	-0.13
	Spring	-0.64	0.46	-0.79	-0.62	-0.42	0.11	-0.69	-0.09
	Summer	-0.22	0.88	-0.66	-0.70	-0.33	0.19	-0.56	-0.54
	Autumn	-0.73	0.80	-0.70	-0.49	-0.10	0.26	-0.53	0.20
Loweswater	Winter	0.69	0.82	-0.82	-0.46	-0.85	-0.45	-0.60	0.15
	Spring	0.67	**0.95**	-0.65	-0.56	-0.68	-0.44	-0.25	0.53
	Summer	0.45	**0.91**	-0.59	-0.40	**-0.90**	-0.27	-0.43	-0.30
	Autumn	0.16	**0.98**	-0.66	-0.66	-0.57	-0.17	-0.47	0.07
Rydal Water	Winter	0.80	0.74	-0.72	-0.44	-0.79	-0.34	-0.57	-0.48
	Spring	0.57	0.46	-0.74	-0.64	-0.53	-0.58	-0.59	-0.43
	Summer	0.52	**0.90**	-0.60	-0.84	**-0.93**	-0.85	-0.84	**-0.93**
	Autumn	0.82	**0.92**	-0.71	-0.81	-0.09	-0.38	-0.81	0.05
Thirlmere	Winter	0.84	**0.96**	-0.74	-0.66	**-0.99**	-0.70	-0.73	**-0.91**
	Spring	0.72	**0.97**	-0.77	-0.81	-0.84	**-0.90**	-0.70	0.34
	Summer	0.05	0.78	-0.65	-0.74	**-0.92**	**-0.92**	-0.80	-0.81
	Autumn	0.86	**0.98**	-0.71	-0.86	-0.15	**-0.94**	-0.74	0.06
Ullswater	Winter	**0.90**	0.80	-0.67	-0.66	-0.84	-0.80	-0.66	-0.87
	Spring	0.00	0.29	-0.68	-0.75	-0.54	-0.73	-0.57	-0.20
	Summer	0.59	0.78	-0.60	**-0.89**	-0.85	-0.49	-0.66	-0.70
	Autumn	0.85	**0.91**	-0.67	-0.85	-0.40	-0.60	-0.87	-0.76
Wastwater	Winter	-	-	-0.76	-0.65	-0.07	-0.55	-0.81	-0.84
	Spring	0.70	-	-0.63	-0.75	-0.47	-0.40	-0.62	-0.21
	Summer	0.86	0.61	-0.30	**-0.93**	-0.60	**-0.95**	-0.72	**-0.91**
	Autumn	**0.89**	0.78	-0.64	-0.67	-0.37	-0.86	-0.73	0.23
Windermere North Basin	Winter	0.76	0.88	-0.86	-0.60	-0.86	-0.60	-0.74	0.37
	Spring	-0.62	0.88	-0.63	-0.72	-0.57	-0.31	-0.75	-0.17
	Summer	-0.07	**0.95**	-0.61	-0.74	-0.64	-0.48	-0.74	-0.73
	Autumn	0.83	0.75	-0.69	0.26	-0.11	0.45	-0.11	-0.13
Windermere South Basin	Winter	0.84	**0.94**	-0.88	-0.71	-0.81	-0.45	-0.61	0.31
	Spring	-0.61	**0.94**	-0.76	-0.75	-0.60	-0.55	**-0.89**	0.02
	Summer	-0.17	0.88	-0.68	-0.67	-0.84	-0.40	-0.69	-0.06
	Autumn	0.64	**0.98**	-0.64	-0.70	-0.61	-0.62	-0.67	-0.66

Table 4.3 Correlation coefficient of change phytoplankton chlorophyll a, *Secchi depth and minimum oxygen concentration at depth for the 20 lake basins and four seasons between 1984 and 2005 (1991 and 2005 for oxygen minimum). Significant correlations are shown in bold and shaded green when $P<0.05$, yellow when $P<0.01$ and orange when $P<0.001$. Data below detection limit or not relevant, so not analysed, indicated by '-'.*

Lake	Season	Chla	Secchi	O_2-min
Bassenthwaite Lake	Winter	-0.153	0.142	-
	Spring	0.924	-0.883	-
	Summer	0.801	-0.066	0.321
	Autumn	-0.798	-0.358	-
Blelham Tarn	Winter	-0.576	-0.309	-
	Spring	0.559	0.248	-
	Summer	-0.628	0.753	0.795
	Autumn	0.581	-0.507	-
Brothers Water	Winter	0.177	-0.391	-
	Spring	0.852	-0.057	-
	Summer	-0.474	0.926	-0.831
	Autumn	0.175	-0.829	-
Buttermere	Winter	-0.739	-0.548	-
	Spring	-0.039	-0.183	-
	Summer	0.545	-0.981	-0.141
	Autumn	0.140	-0.726	-
Coniston Water	Winter	-0.874	-0.376	-
	Spring	0.628	-0.692	-
	Summer	0.935	-0.062	-0.697
	Autumn	0.964	-0.452	-
Crummock Water	Winter	-0.603	-0.667	-
	Spring	-0.348	0.329	-
	Summer	0.235	0.031	-0.632
	Autumn	0.550	-0.602	-
Derwent Water	Winter	0.647	-0.485	-
	Spring	0.767	-0.477	-
	Summer	0.999	0.241	-0.802
	Autumn	0.881	-0.729	-
Elterwater	Winter	0.580	-0.688	-
	Spring	0.813	-0.943	-
	Summer	0.866	0.293	0.000
	Autumn	0.843	0.182	-
Ennerdale Water	Winter	-0.924	0.100	-
	Spring	0.818	-0.849	-
	Summer	0.775	-0.616	0.468
	Autumn	-0.643	-0.783	-
Esthwaite Water	Winter	0.604	-0.955	-
	Spring	-0.435	0.773	-
	Summer	-0.775	0.820	0.795
	Autumn	0.432	-0.240	-
Grasmere	Winter	0.860	-0.905	-
	Spring	0.628	-0.467	-
	Summer	0.236	0.838	0.795
	Autumn	0.805	0.095	-
Haweswater	Winter	-0.777	-0.328	-
	Spring	-0.146	-0.195	-
	Summer	-0.178	-0.212	-0.686
	Autumn	0.900	0.072	-
Loughrigg Tarn	Winter	-0.458	-0.406	-
	Spring	-0.539	-0.287	-
	Summer	-0.027	-0.209	-0.229
	Autumn	0.992	-0.598	-
Loweswater	Winter	0.173	-0.285	-
	Spring	0.892	-0.848	-
	Summer	0.626	-0.177	0.795
	Autumn	0.321	-0.839	-
Rydal Water	Winter	0.997	-0.998	-
	Spring	0.509	-0.733	-
	Summer	0.776	-0.044	0.795
	Autumn	0.812	-0.025	-
Thirlmere	Winter	0.317	0.474	-
	Spring	0.853	-0.677	-
	Summer	0.286	-0.793	0.089
	Autumn	0.744	-0.486	-
Ullswater	Winter	-0.605	-0.840	-
	Spring	-0.896	0.215	-
	Summer	-0.581	-0.384	-0.882
	Autumn	0.884	-0.643	-
Wastwater	Winter	-0.091	-0.933	-
	Spring	0.956	-0.905	-
	Summer	0.456	-0.674	-0.895
	Autumn	0.914	0.088	-
Windermere North Basin	Winter	-0.373	-0.973	-
	Spring	0.569	-0.205	-
	Summer	-0.564	0.863	-0.142
	Autumn	0.326	0.597	-
Windermere South Basin	Winter	-0.114	-0.983	-
	Spring	-0.134	-0.135	-
	Summer	-0.553	-0.162	-0.156
	Autumn	0.006	-0.284	-

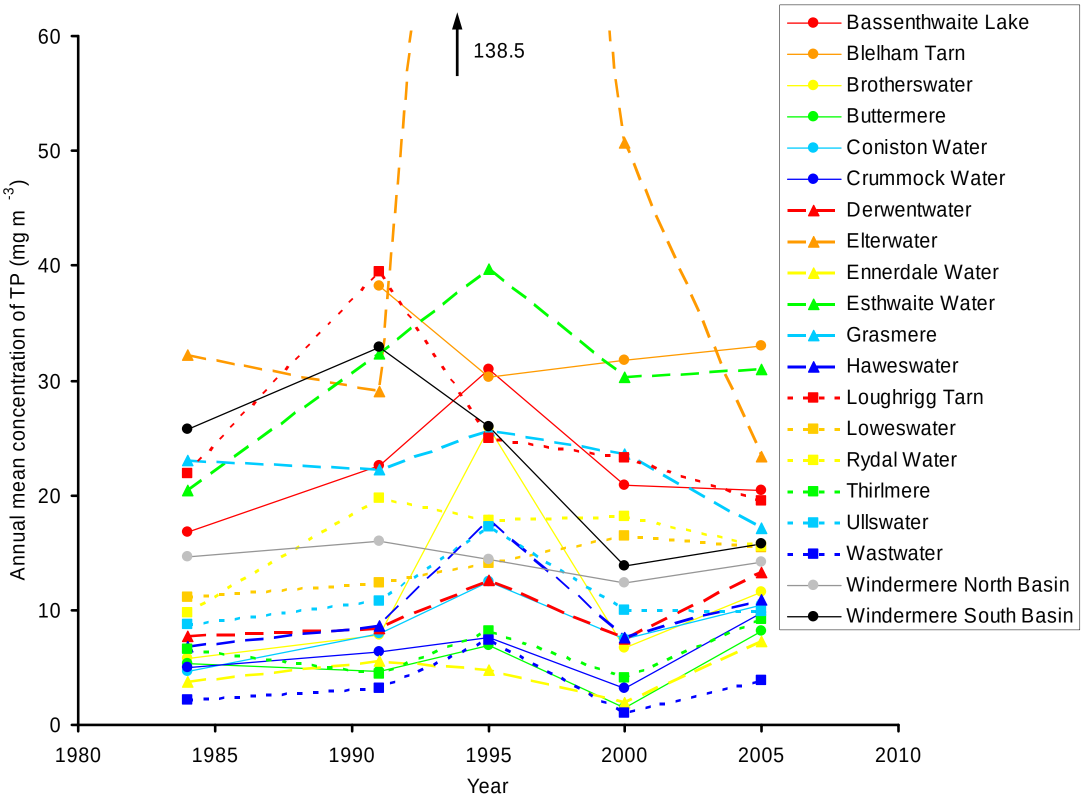


Figure 4.7. Long-term change in the annual average concentration of TP from 1984 to 2005.

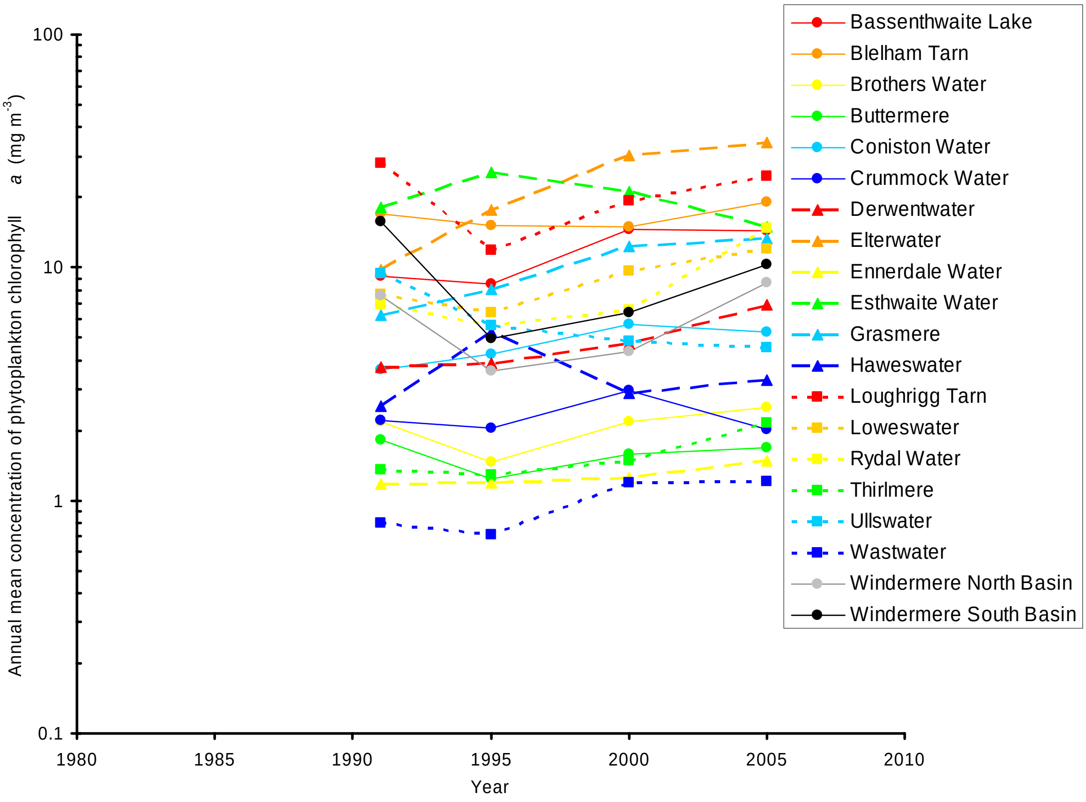


Figure 4.8. Long-term change in the annual average concentration of phytoplankton chlorophyll a *from 1991 to 2005.*

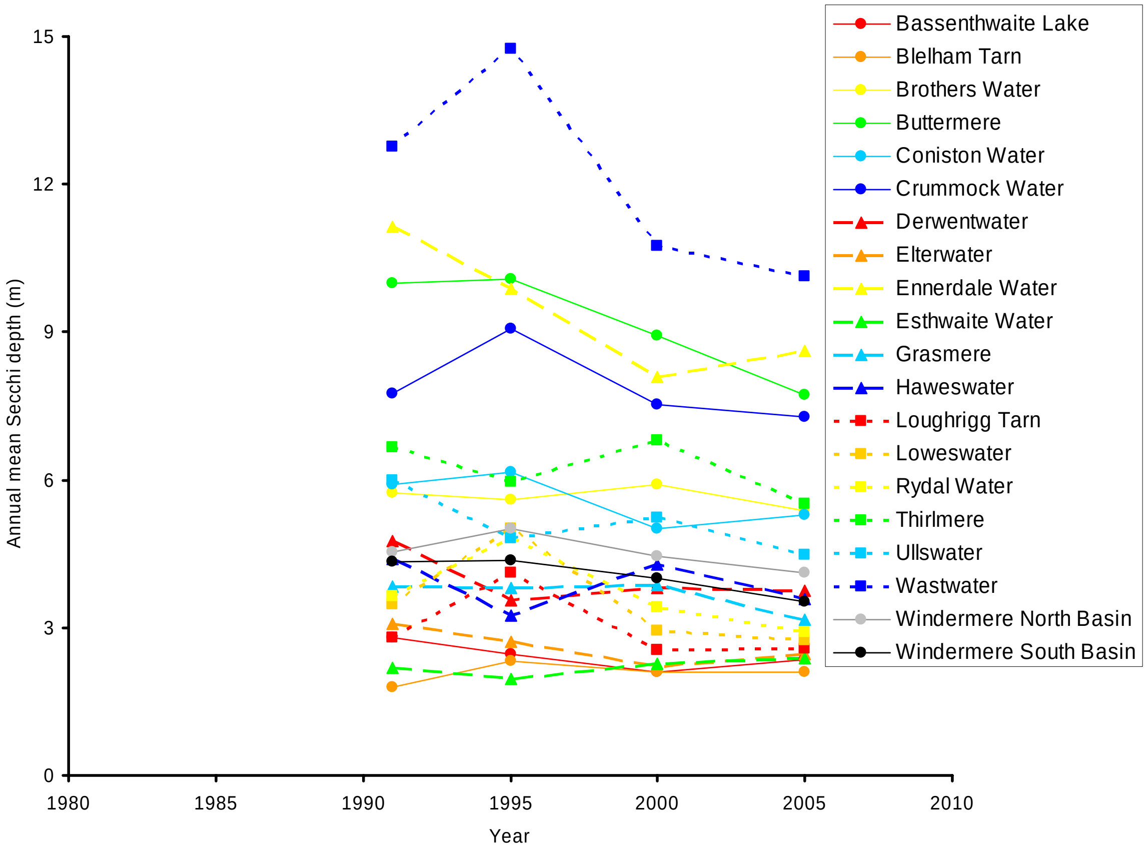


Figure 4.9. Long-term change in the annual average Secchi depth from 1991 to 2005.

4.2.9 Ennerdale Water

The pattern of change in Ennerdale Water is somewhat similar to that in Derwent Water. There is a hint that the concentration of TP has increased in summer although the change is not quite significant (Table 4.1). However, there has been a reduction in concentration of nitrate in spring which might indicate increased demand by the phytoplankton and there has been a slight increase in spring phytoplankton from 1.05 to 2.06 mg m^{-3} in 1991 and 2005 respectively. This is associated with a spring decline in Secchi depth from 10.0 m in 1991 to 8.8 m in 2005. Neither the increase in concentration of phytoplankton nor the decrease in Secchi depth are quite significant (Table 4.3) but they provide some indication for increased productivity in the lake. Ennerdale has also shown a clear increase in alkalinity in spring, summer and autumn: for example in summer alkalinity has increased from 33 to 56 mequiv m^{-3} between 1984 and 2005 (Table 4.2). Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.10 Esthwaite Water

There has been little change in the nutrient chemistry in Esthwaite Water. The concentration of nitrate has declined in winter and autumn but unlike, for example, Derwent Water this does not seem to be associated with an increase in nutrient availability. Alkalinity has increased in spring and autumn (Table 4.2). Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.11 Grasmere

There have been no important statistically significant changes in Grasmere (Tables 4.1, 4.2, 4.3). However there are some slight indications that the lake is getting more nutrient enriched since the correlations of change for chlorophyll *a* are positive in each of the four months but this change is not evident in any of the nutrients measured (Table 4.1). Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.12 Haweswater

There is a clear indication that the alkalinity of Haweswater has increased since 1984. The correlations of change in alkalinity are positive in all four seasons and significant in summer and autumn (Table 4.2). Furthermore, there has been a significant increase in pH in autumn. Concentrations of chloride declined statistically significantly in spring, summer and autumn and concentrations of sodium declined in autumn (Table 4.3). Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.13 Loughrigg Tarn

The most striking change in Loughrigg Tarn has been the statistically significant increase in autumn phytoplankton (Table 4.3). In 1991, autumn phytoplankton chlorophyll *a* was only 11 mg m^{-3} and this increased in the succeeding surveys and in 2005 was 49 mg m^{-3}. The causes of this increase are not immediately apparent since there have been no statistically significant increases in nutrients over this period. Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.14 Loweswater

Loweswater shows clear evidence for nutrient enrichment. There has been a significant increase in concentration of TP in the spring and the decline in concentrations of nitrate in summer and autumn is probably as a consequence of the increased demand (Table 4.1). Ammonium has increased in the autumn and this may also be a result of nutrient enrichment. There has been a striking increase in phytoplankton chlorophyll *a* in the spring, and a decline in oxygen concentration at depth, but neither is quite statistically significant (Table 4.3). Like many lakes in the region alkalinity has increased and this has been statistically significant between spring and autumn. Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.15 Rydal Water

Rydal Water shows mild signs of nutrient enrichment. There has been a highly significant increase in TP in summer but not other times of year (Table 4.1) and a decline in spring silica which indicates increased demand by phytoplankton. Phytoplankton have increased, but only in winter, but this is linked to a concomitant decline in Secchi depth. The absolute increase in phytoplankton chlorophyll *a* is actually rather small. Identifying the causes of changes in Rydal Water is more difficult than in many of the other lakes because it is highly influenced by changes in the larger Grasmere immediately upstream. Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.16 Thirlmere

There have been no significant changes in nutrient concentrations in Thirlmere between 1984 and 2005 (Table 4.1) although there is very weak evidence that TP may have increased slightly. Alkalinity has increased and calcium decreased, but this is a fairly common pattern in several lakes (Table 4.2). There has been an increase in phytoplankton chlorophyll *a* in spring which, while not quite statistically significant, is linked to a decrease in Secchi depth so is possibly real although slight in absolute magnitude (Table 4.3). Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.17 Ullswater

There is no evidence for changing nutrient status in Ullswater apart from an increase in silica in spring (Table 4.1). Major ions are also stable apart from the slight increase in alkalinity noted for other lakes (Table 4.2). There have been no significant changes in phytoplankton chlorophyll *a*, Secchi depth or oxygen depletion (Table 4.3). Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.18 Wastwater

Wastwater appears to be relatively stable. There have been no changes in concentration of nutrients, a slight increase in autumn pH, possibly associated with a tendency for a slightly higher alkalinity in summer and autumn (Tables 4.1, 4.2). There has been a decline in summer concentrations of chloride, magnesium and potassium but this is quite a common pattern across all the lakes. The only very slightly worrying response is evidence for a small increase in spring chlorophyll *a* and a decline in Secchi depth (Table 4.3). Although neither are significant this warrants further investigation given that Wastwater is the premier oligotrophic lake in the region. Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.19 Windermere North Basin

Windermere North Basin shows no indication for changes in nutrient concentrations based on Lakes Tour data (Table 4.1). Fortnightly data have shown, however, some more subtle changes (see e.g. Maberly *et al*., 2005). Of the major ions the only significant change is an increase in alkalinity in summer (Table 4.2). There has been a marked decline in Secchi depth but this does not appear to be wholly linked to changes in winter phytoplankton chlorophyll *a* (Table 4.3). Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.2.20 Windermere South Basin

Like the North Basin, there are no significant changes in nutrient concentration in the South Basin of Windermere although the more detailed fortnightly data do reveal slight long-term changes (Maberly *et al*., 2005). Alkalinity has increased significantly (Table 4.2). Like the North Basin, there is a significant decline in Secchi depth in winter that does not appear to be related to an increase in phytoplankton chlorophyll *a* (Table 4.3). Long-term changes in the annual mean of TP, chlorophyll *a* and Secchi depth are shown in Figures 4.7 to 4.9.

4.3 Summary of ecological status of the lakes under the WFD

Figure 4.10 summarises the ecological status of the 20 lakes based on TP and phytoplankton chlorophyll *a*, grouped according to their ecological type. It is important to note again that the ecological boundaries are still being fine-tuned and the ones used here were correct at the date of writing but may change slightly in the future.

The critical ecological boundary for the Water Framework is the Good:Moderate boundary, because at lakes that are only Moderate or worse, measures will need to be put in place to improve ecological status. Of the four lakes in the low alkalinity shallow category all have Good ecological status in terms of concentration of TP but in terms of chlorophyll *a*, Brothers Water has High ecological status while Derwent Water, Grasmere and Rydal Water have Moderate or Poor ecological status (Fig. 4.10).

There are six lakes in the medium alkalinity shallow category. Of these Loweswater, Bassenthwaite Lake, Loughrigg Tarn and Elterwater have Good ecological status based on TP but Blelham Tarn and Esthwaite Water just slip into the Poor category. This type of lake has the worst ecological status of any of the lake-types for chlorophyll *a*: all fall below the Moderate ecological status (Fig. 4.10).

In contrast, the low alkalinity deep lakes have generally High or Good ecological status (Wastwater, Ennerdale Water, Buttermere and Thirlmere) while Crummock Water has Moderate ecological status for TP but High ecological status for phytoplankton chlorophyll *a* (Fig. 4.10).

The five medium alkalinity deep lakes are also of generally Good or Moderate ecological status. Based on TP, all lakes are of good ecological status, but the North Basin of Windermere is only just in this category. In terms of phytoplankton chlorophyll *a* however, while Haweswater is on the High: Good boundary and Ullswater has Good ecological status, Coniston Water is on the Good:Moderate boundary and the North and South Basins of Windermere are on the Moderate:Poor boundary (Fig. 4.1).

Overall, 70% (14 lakes) have an ecological status of Good or High for TP but only 40% (8 lakes) have an ecological status of Good or High for phytoplankton chlorophyll *a* (Fig. 4.11). The medium alkalinity, shallow lakes tend to have the worst ecological status of any lake type.

Many of the major lakes in the English Lake District are currently not at Good ecological status and therefore stringent management plans need to be drawn up to produce measures that will achieve Good ecological status by 2015 as required by the Water Framework Directive.

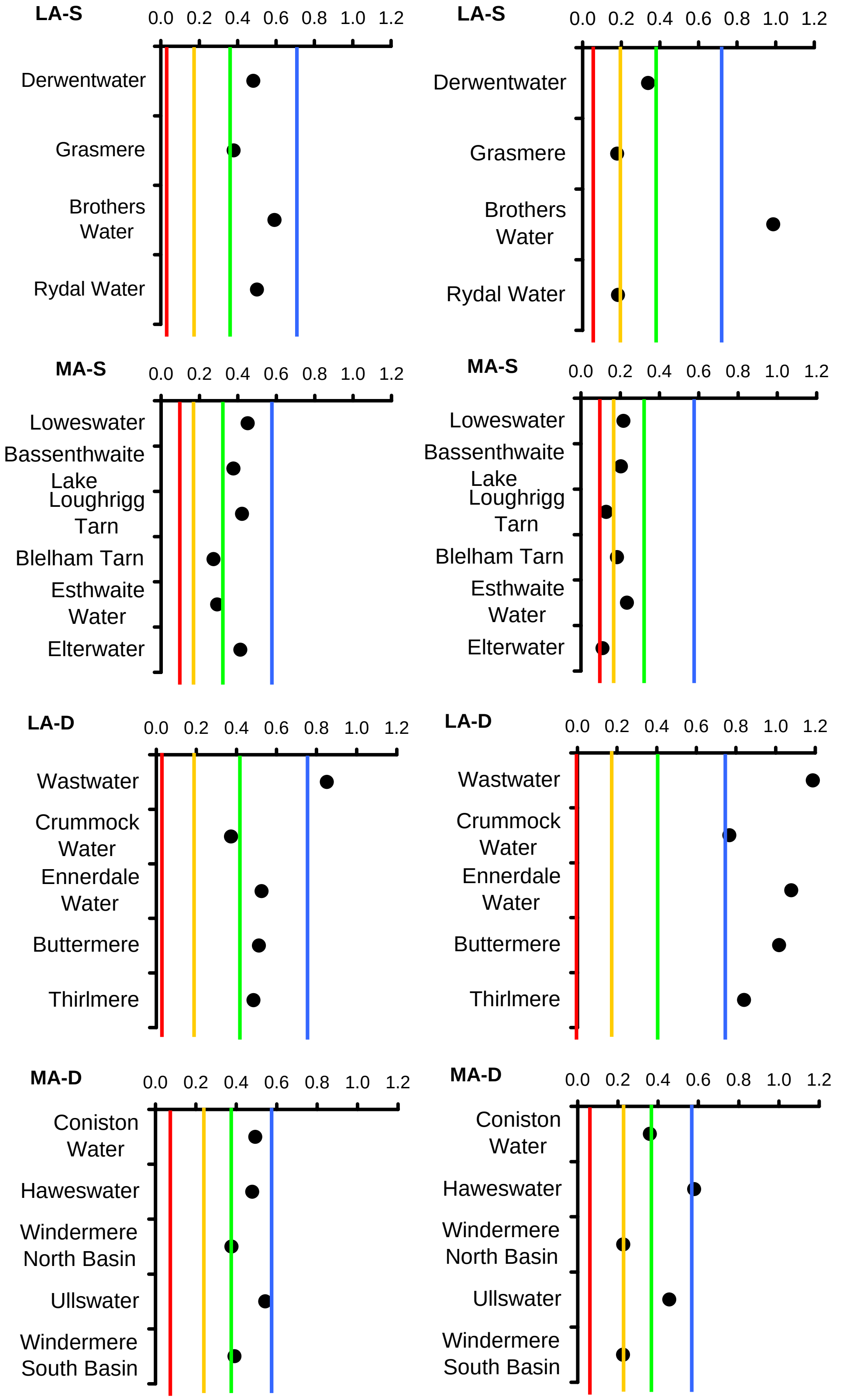


Figure 4.10 Ecological quality ratios for the 20 lakes categorised according to ecological type (LA = low alkalinity, MA = moderate alkalinity, S = shallow and D = deep) for TP (left-hand column) and chlorophyll a *(right-hand column) in relation to current ecological boundaries: Blue – High:Good; green = Good:Moderate; orange = Moderate: Poor; red = Poor:Bad.*

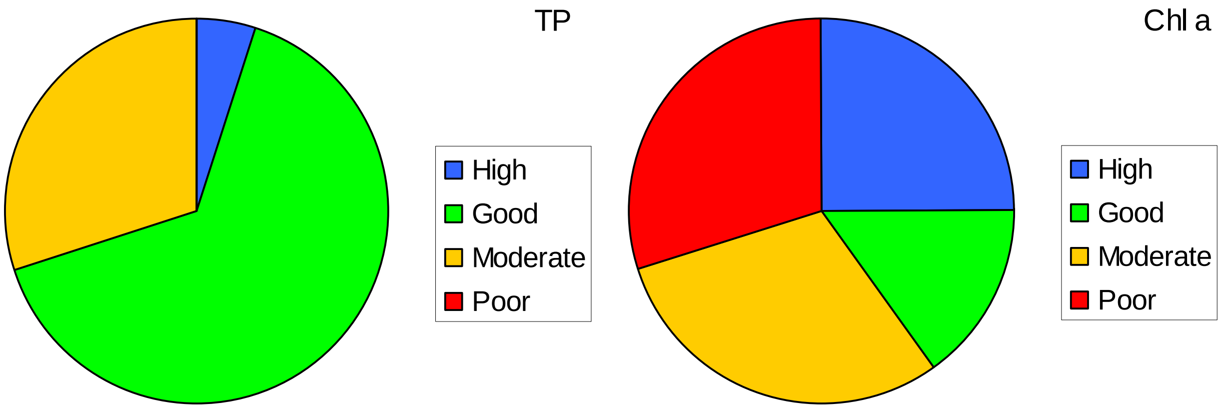


Figure 4.11 Summary of overall ecological status for the 20 lakes according to TP or phytoplankton chlorophyll a.

4.4 Suggestions for further work

Based on the work reported here, the lakes that fail Good ecological status will need some remedial work and therefore probably a better understanding of their limnology and the reasons for poor water quality which will probably require a better understanding of the sources of nutrients to the lake. The current scheme for assessing ecological status based on phytoplankton chlorophyll *a* suggests that Derwent Water, Grasmere, Rydal Water, Bassenthwaite Lake, Blelham Tarn, Elterwater, Esthwaite Water, Loughrigg Tarn, Loweswater and the two Basins of Windermere are not at Good ecological status.

Wastwater is the premier oligotrophic lake in England. It is clearly still at reference condition with High ecological status. However, there are some signs that water quality is deteriorating, albeit slightly. Spring chlorophyll *a* concentrations are getting higher and Secchi depth is getting shallower (Table 4.3) and annual mean phytoplankton chlorophyll *a* and annual mean Secchi depth are also deteriorating (Figs 4.8 and 4.9). There has not been a comprehensive limnological survey of Wastwater, and this combined with signs of change in the lake need to be investigated.

Rydal Water shows signs of deteriorating water quality. Assessing the causes of these changes is complicated because it is closely-linked to Grasmere upstream. However, since CEH monitor Grasmere fortnightly, a fortnightly survey of Rydal Water would greatly aid our understanding of the limnology of the lake and the possible causes of nutrient enrichment. So far as I am aware, there has never been a comprehensive limnological survey of Rydal Water.

5. References

George D.G. (1992). A general assessment of long-term change in the larger lakes of the English Lake District. Report to the National Rivers Authority.

Gorham E., Lund J.W.G., Sanger J.E. & Dean W.E. (1974). Some relationships between algal standing crop, water chemistry and sediment chemistry in the English Lakes. *Limnology & Oceanography* 19: 601-617.

Gurney R. (1931-1933) British freshwater copepoda I-III. London: Ray Society.

Hall G.H., Allen P.V., Corry J.E., Cranwell P.A., Haworth E.Y., Hewitt D.P. & Jaworski G.H.M (1992). The Lakes Tour 1991: a survey of the Cumbrian Lakes. Report to the National Rivers Authority.

Hall G.H., Cubby P., De Ville M.M., Fletcher J., Haworth E.Y., Hewitt D., Hodgson P., James J.B., Lawlor A.J., Parker J.E., Rigg E. & Woof C. (1996). A survey of the limnological characteristics of the lakes of the English Lake District: Lakes Tour 1995. Report to the Environment Agency, North West Region.

James C., Fisher J. & Moss B. (2003). Nitrogen driven lakes: The Shropshire and Cheshire meres? *Archiv für Hydrobiologie* 158: 249-266.

Jones J.G., Orlandi M.J.L.G. & Simon B.M. (1979). A microbiological study of sediments from the Cumbrian Lakes. *Journal of General Microbiology* 115: 37-48.

Kadiri M.O. & Reynolds C.S. (1993) Long-term monitoring of the conditions of the lakes: the example of the English Lake District. *Archiv für Hydrobiologie* 129: 157-178.

Knudsen B.M. (1954). The ecology of the diatom genus *Tabellaria* in the English Lake District. *Journal of Ecology* 42: 345-358.

Lund J.W.G., Kipling C. & LeCren E.D. (1958). The inverted microscope method of estimating algal numbers and the statistical basis of estimations by counting. *Hydrobiologia* 11: 143-170.

Maberly S.C. (1996). Diel, episodic and seasonal changes in pH and concentrations of inorganic carbon in a productive English Lake, Esthwaite Water, Cumbria. *Freshwater Biology* 35: 579-598.

Maberly S.C., King L., Dent M.M., Jones R.I. & Gibson C.E. (2002) Nutrient limitation of phytoplankton and periphyton growth in upland lakes. *Freshwater Biology* 47: 2136-2152.

Maberly S.C., Dent M.M., Parker. J.E., Lawlor, A.J. & Vincent C. (2003). Coniston Water: Investigation into apparent nutrient enrichment and copper toxicity. Final report commissioned by the Environment Agency and United Utilities. 48pp.

Maberly S.C., King L., Gibson C.E., May L., Jones R.I., Dent M.M. & Jordan C. (2003) Linking nutrient limitation and water chemistry in upland lakes to catchment characteristics. *Hydrobiologia* 506: 83-91.

Maberly S.C., De Ville M.M., Elliott J.A., Fletcher J.M., James J.B., Reynolds C.S., Thackeray S.J. & Vincent C. (2005). The Urban Waste Water Treatment Directive: Observations on the water quality of Windermere, Grasmere, Derwent Water and Bassenthwaite Lake, 2004. Final report commissioned by the Environment Agency. 25 pp.

Maberly S.C., Norton L., De Ville M.M., Elliott J.A., Thackeray S.J., Groben R. & Carse F. (2006). An investigation in the potential impacts of farming practices on Loweswater. Final report to the Rural Development Service and the National Trust. 79pp.

Mackereth, F.G.H., Heron, J. & Talling, J.F. (1978) *Water analysis: some revised methods for limnologists*. Freshwater Biological Association. Scientific Publication No. 36.

Organisation for Economic Co-operation and Development - OECD (1982). Eutrophication of Waters: Monitoring, assessment and control. Paris, France.

Parker J.E., Abel D., Dent M.M., Fletcher J.M., Hewitt D.P., James J.B., Lawlor A.J., Lofts S., Simon B.M., Smith E.J. & Vincent C.D. (2001). A survey of the limnological characteristics of the lakes of the English Lake District: The Lakes Tour 2000. Report to the Environment Agency. 60 pp.

Pearsall W.H. (1921). The development of vegetation in the English Lakes, considered in relation to the general evolution of glacial lakes and rock basins. *Proceedings of the Royal Society (B)* 92: 259-284.

Pearsall W.H. (1932). Phytoplankton in the English Lakes II. The composition of the phytoplankton in relation to dissolved substances. *Journal of Ecology* 20: 241-262.

Phillips G. (2006). Draft chlorophyll *a* classification boundaries for intercalibration. Draft report to the UK Lakes Task Team.

Reynolds C.S., Maberly S.C., Irish A.E. & Elliott J.A. (2001) Options for the remediation of water quality in Grasmere with special reference to uncertainties concerning loading. Report commissioned by the Environment Agency. 27pp.

Scourfield D. J. & Harding J. P. (1966) *A key to the British species of freshwater Cladocera*. Freshwater Biological Association Scientific Publication. Ambleside: Freshwater Biological Association.

Sutcliffe D.W., Carrick T.R., Heron J., Rigg E., Talling J.F., Woof C. and Lund J.W.G. (1982). Long-term and seasonal changes in the chemical composition of precipitation and surface waters of lakes and tarns in the English Lake District. *Freshwater Biology* 12: 451-506.

Sutcliffe D.W. (1998). The ionic composition of surface waters in the English Lake District, related to bedrock geology, with some singular facts and speculation on the existence of mineral-rich groundwaters. *Freshwater Forum* 11: 30-51.

Talling J.F. (1974). In standing waters. In: *A Manual on Methods for Measuring Primary Production in Aquatic Ecosystems,* Ed. R.A. Vollenweider (IBP Handbook No. 12, 2nd edn), pp. 119-123. Oxford, Blackwells.

Talling, J.F. (1999) Some English Lakes as diverse and active ecosystems: a factual summary and source book. FBA, 80 pp.

Thackeray S.J., Maberly S.C. & Winfield I.J. (2004). The present state of Bassenthwaite Lake: implication and opportunities for future management strategies. Final report commissioned by Unitied Utilities, English Nature and the Lake District National Park Authority, 94pp.

Thackeray S.J., Maberly S.C. & Winfield I.J. (2006). The Ecology of Bassenthwaite Lake (English Lake District). *Freshwater Forum* 25: 1-80.

Tipping E., Carrick T.R., Hurley M.A., James J.B., Lawlor A.J., Lofts S., Rigg E., Sutcliffe D.W. & Woof C. (1998) Reversal of acidification in upland waters of the English Lake District. *Environmental Pollution* 103:143-151.

Tipping E., Hilton J. and James B. (1988). Dissolved organic matter in Cumbrian lakes and streams. *Freshwater Biology* 19: 371-378.

UKTAG (2006).UK environmental standards and conditions (Phase 1): Draft provided to groups and organisations for review and comment (SR1-2006).

Vighi M. & Chiaudani M. (1984). A simple method to estimate lake phosphorus concentrations resulting from natural, background, loadings. *Water Research* 19: 987-991.

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7. Appendices

Appendix 1: Temperature and oxygen profiles

Appendix 2: Phytoplankton chlorophyll a and Secchi depth

Appendix 3: Major ions and pH.

Appendix 4: Nutrients.

Appendix 5: Phytoplankton counts.

Appendix 5: Zooplankton abundance.