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THE ECOLOGY OF BASSENTHWAITE LAKE (ENGLISH LAKE DISTRICT)

by

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THE ECOLOGY OF BASSENTHWAITE LAKE (ENGLISH LAKE DISTRICT)

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Abstract

Bassenthwaite Lake is, in many ways, different from the other major lakes in the English Lake District: it is the most northerly, the shallowest, has the largest catchment and the shortest mean retention time. There is also considerable temporal variation in lake level. Although the lake is surrounded by rich lowland soils, the catchment is dominated by upland moor and improved pasture, underlain by four dominant soil types: well drained loam with bare rocks and scree, shallow acid upland peat, fine loam and thick acidic peat soils. The supply of catchment-derived sediment, and its recycling within the lake, has serious implications for water quality and for the ecology of this site. These sediments largely comprise inorganic material, and are believed to originate mainly from the high fells and the floodplain of the River Derwent.

Typically the lake thermally stratifies between May and September, and is mixed for the remainder of the year. However, the lake is only ever weakly stratified and episodic mixing events influence the ecology of the lake, by increasing the concentration of oxygen at depth and resuspending sediment in shallow water.

Palaeolimnological reconstruction of long-term changes in total phosphorus concentration has revealed that, over the past 250 years, the lake has become increasingly eutrophic, with increases in phosphorus concentration being particularly dramatic in the late 1970s and early 1980s. Though tertiary sewage treatment was implemented at Keswick (the principal town in the catchment) in 1995, there has been only a slight reduction in phosphorus concentration in the lake. At present, diffuse sources contribute more to the total phosphorus load than do point sources and it is suspected that efficient internal recycling of a relatively modest sediment store, caused by wind-driven resuspension, may be one reason for the limited reduction in phosphorus availability.

Unlike the other productive Cumbrian lakes, the spring phytoplankton bloom is weak, biomass tends to be highest in early autumn and individual species tend to lack a strong seasonality. There has been no detectable change in phytoplankton biomass since the implementation of tertiary sewage treatment at Keswick Wastewater Treatment Works in 1995. The macrophyte flora of the lake is diverse, and includes the rare species *Luronium natans*. The community composition has been relatively stable since around 1913. However, the invasion of non-native species such as *Elodea canadensis*, *E. nuttallii* and in particular *Crassula helmsii* is a cause of concern.

Relatively little is known of the zooplankton and benthic invertebrate communities of Bassenthwaite Lake, though the fish community has been studied intensively. The lake is one of only two in the UK which still contain a native population of the vendace (*Coregonus albula*), the UK's rarest freshwater fish. The current status of vendace is considered to be extremely poor and the survival of this species is threatened by low hypolimnetic concentrations of oxygen, siltation of spawning grounds, species introductions and climate change. In total, 10 fish species have been recorded in the lake, of which three have been introduced in recent times, i.e. roach (*Rutilus rutilus*), ruffe (*Gymnocephalus cernuus*) and dace (*Leuciscus leuciscus*).

A total of 150 bird species has been recorded at Bassenthwaite Lake since 1995. The establishment of breeding ospreys (*Pandion haliaetus*) in 2001 has arguably been the highest profile event at the lake in recent years, while significant numbers of cormorants (*Phalacrocorax carbo*) now frequent the lake which may have implications for an already struggling vendace population.

The ecological integrity of Bassenthwaite Lake is threatened by man-made changes in nutrient and sediment loading, invasion by non-native species and climate change. These threats can be minimised by science-based management informed by a thorough knowledge of the lake and its catchment: the ethos behind the Bassenthwaite Lake Restoration Programme instigated by the Lake District Still Waters Partnership. This review aims to synthesise available information on the ecology of Bassenthwaite Lake so that it is available for lake managers, scientists and naturalists. Sustainable catchment management and continued monitoring are key to the maintenance and enhancement of ecological quality at this beautiful and diverse site.

INTRODUCTION

Bassenthwaite Lake is the most northerly of the large lakes in the English Lake District (Fig. 1) and the shallowest, with a mean depth of 5.3 m. (Fig. 2; Table 1). It has the largest catchment area of any of the major lakes (Table 1) and the high average altitude of this catchment, 335 m (NERC 1999), results in a high annual precipitation. This combination of characteristics produces one of the most ecologically influential and defining features of this system: an average retention time of only 19 days (Table 1). This is the shortest of any of the major Cumbrian lakes.

Although the lake is surrounded by rich lowland soils, most of the catchment, including the subcatchments of Derwent Water and Thirlmere, is mountainous with nutrient-poor and base-poor soils. Nevertheless, Bassenthwaite Lake is relatively productive: it was the fourth most productive lake out of 20 Cumbrian lakes surveyed during 2000, based on an analysis of the annual average concentration of phytoplankton chlorophyll *a* (Parker et al. 2001). Physical processes strongly influence the ecology of Bassenthwaite Lake. Stratification is weak and readily broken down by wind mixing, as a result of the shallowness and exposed

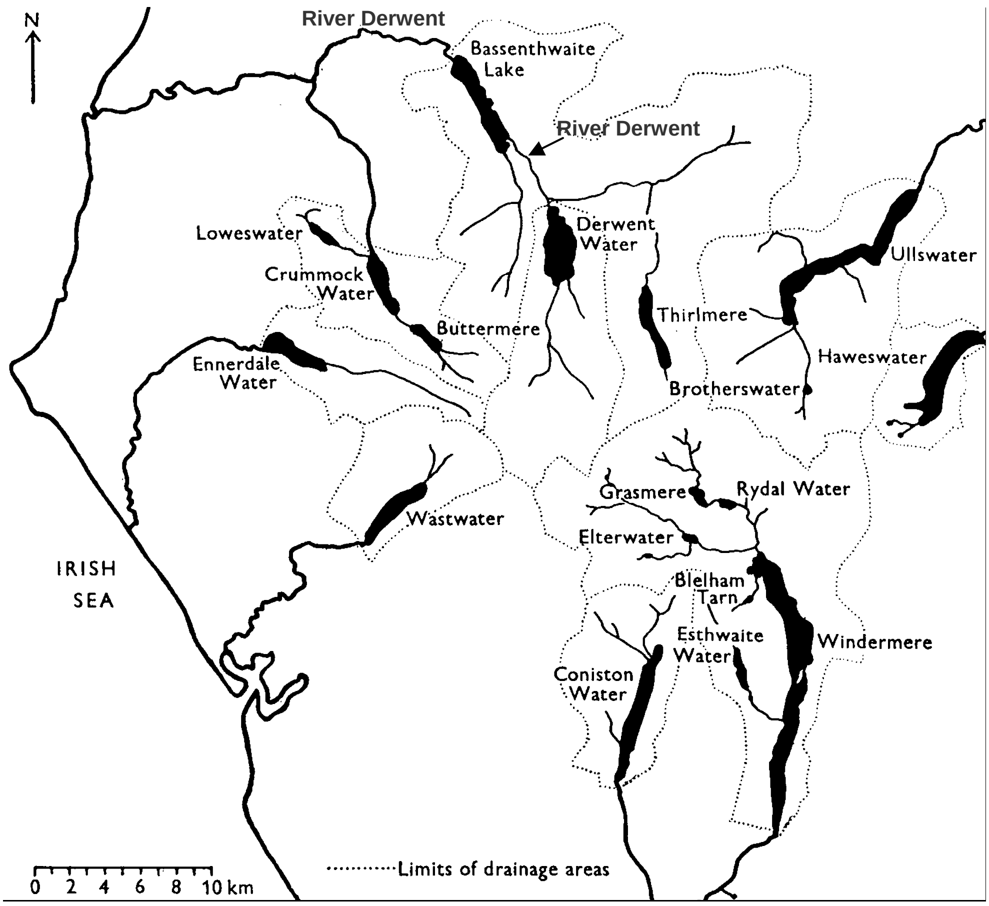
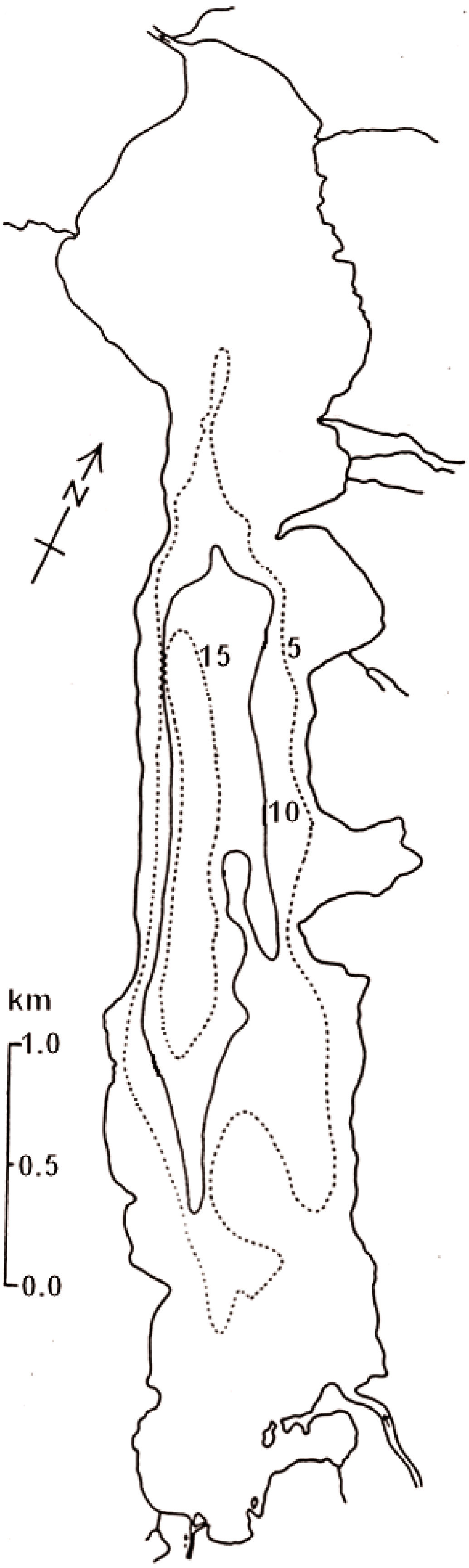


FIG. 1. Map of north-west England showing the lakes of the English Lake District and Bassenthwaite Lake.



location of the lake. For the same reason, water transparency is very variable (Talling 1999), driven by extensive resuspension of sediment during windy episodes (Parker et al. 1999).

The lake has been designated a National Nature Reserve and a Grade I Site of Special Scientific Interest. The lake also receives special protection under the Wildlife and Countryside Act, 1981. Since the Keswick Wastewater Treatment Works (WwTW) serves a population equivalent of greater than 10 000 and Bassenthwaite Lake exhibits eutrophication symptoms above the trigger thresholds, it qualifies as being a 'sensitive area' in the terminology of the European Urban Waste Water Treatment Directive (91/271/EEC). Bassenthwaite Lake is also a candidate Special Area of Conservation under the Habitats Directive.

The earliest major review of Bassenthwaite Lake was carried out in 1989 (Atkinson et al. 1989), and was followed by other reviews (Pickering et al. 1993; Jaworski et al. 1994) and a general review of the literature of the English Lakes, including Bassenthwaite Lake (Talling 1999). Since Atkinson et al. (1989) a large amount of new information has become available on Bassenthwaite Lake, largely produced by the fortnightly sampling on the lake carried out by the Centre for Ecology and Hydrology (formerly the Institute of

FIG. 2. The bathymetry of Bassenthwaite Lake, depth in m (Ramsbottom 1976).

Table 1. Basic limnological features of Bassenthwaite Lake, derived largely from Reynolds (1999). *Incorrectly given as 238 km^2 in Ramsbottom (1976) and other studies. Equals 361.3 km^2 if the three lakes (Bassenthwaite Lake, Derwent Water and Thirlmere) are added. **Derived from daily mean discharge at Ouse Bridge from 1977 to 2001. ***Calculated mean residence time from hydraulic discharge and lake volume.

Feature	Value
Altitude (m)	69
Area (km^2)	5.28
Area of lake < 5 m deep (km^2)	3.46 (66 %)
Volume of lake (10^6 m^3)	27.9
Mean depth (m)	5.3
Maximum depth (m)	20
Area of catchment (km^2)*	347.4
Mean annual hydraulic discharge (10^6 m^3 y^{-1})**	544
Mean hydraulic residence time (days)***	19

Freshwater Ecology). This monitoring has resulted in a large number of status reports (Jaworski et al. 1994; Reynolds et al. 1995a; Reynolds et al. 1996; Reynolds et al. 1997; Winfield et al. 1997; Reynolds et al. 1998; Reynolds et al. 1999; Reynolds et al. 2000; Reynolds et al. 2001; Reynolds et al. 2002; Maberly et al. 2003; Maberly et al. 2004; Winfield et al. 2004a).

There is a large amount of information available on Bassenthwaite Lake but much of this is unpublished or exists only in commissioned reports. The purpose of this review is to bring this information together and make it available to conservation bodies, managers, researchers and interested naturalists alike.

CATCHMENT CHARACTERISTICS

Land cover

May et al. (1995) used a Geographical Information System (GIS) to estimate the coverage of different land use types in the 15 subcatchments that comprise the catchment of Bassenthwaite Lake. This analysis was based upon data provided by the Lake District National Park Authority. Twelve land cover categories were identified (Fig. 3). The catchment is dominated by upland moor (53 %) and improved pasture (21 %) is the second most abundant land cover. Rough grazing, inland bare rock and coniferous bare rock each contribute about 5 % to the total land cover. Urban and rural settlements, of which the principal is the town of Keswick,

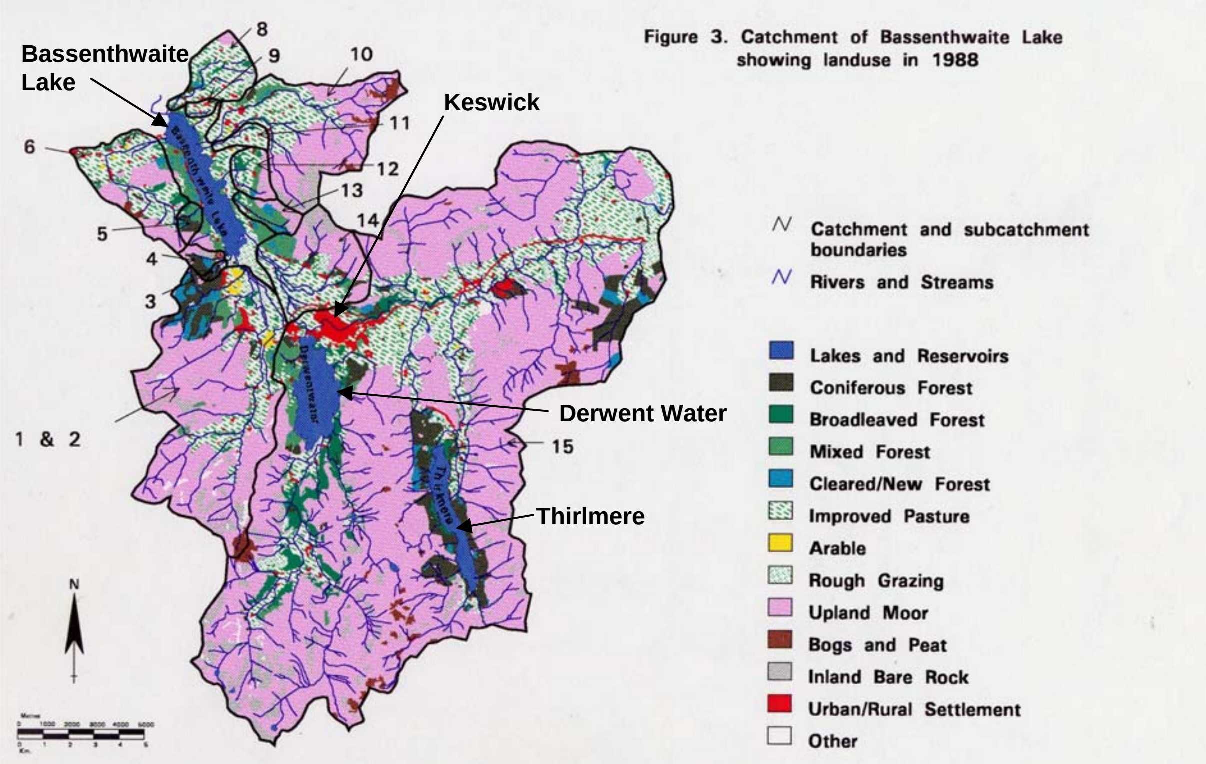


FIG. 3. Land cover in the Bassenthwaite catchment in 1988 (May et al. 1995). Land cover data supplied by the Lake District National Parks Authority.

only contribute 1.8 % of the land cover in the catchment, but nevertheless have a disproportionately large effect on the ecology of the lake.

Soils

May et al. (1996) also used a GIS approach to map the soils in the catchment using the Hydrology of Soil Types (HOST) classification of Boorman et al. (1995). Ten different soil types were found (Fig. 4), of which well drained loam with bare rocks and scree (38 %), shallow, acid upland peat (24 %), fine loam (16 %) and thick, very acid peat soils (10.8 %) were the dominant types. The shallow acid upland peat occurred mostly on the uplands, while well-drained loam with bare rocks and screes was mostly found on the lower slopes, and fine loam tended to cover the valley bottoms (Fig. 4). There was a strong correspondence between soil type and land cover. For example, 84 % of shallow upland peat was covered by upland moor and more than 65 % of well-drained loam with bare rocks and screes and of fine loam was covered by improved pasture.

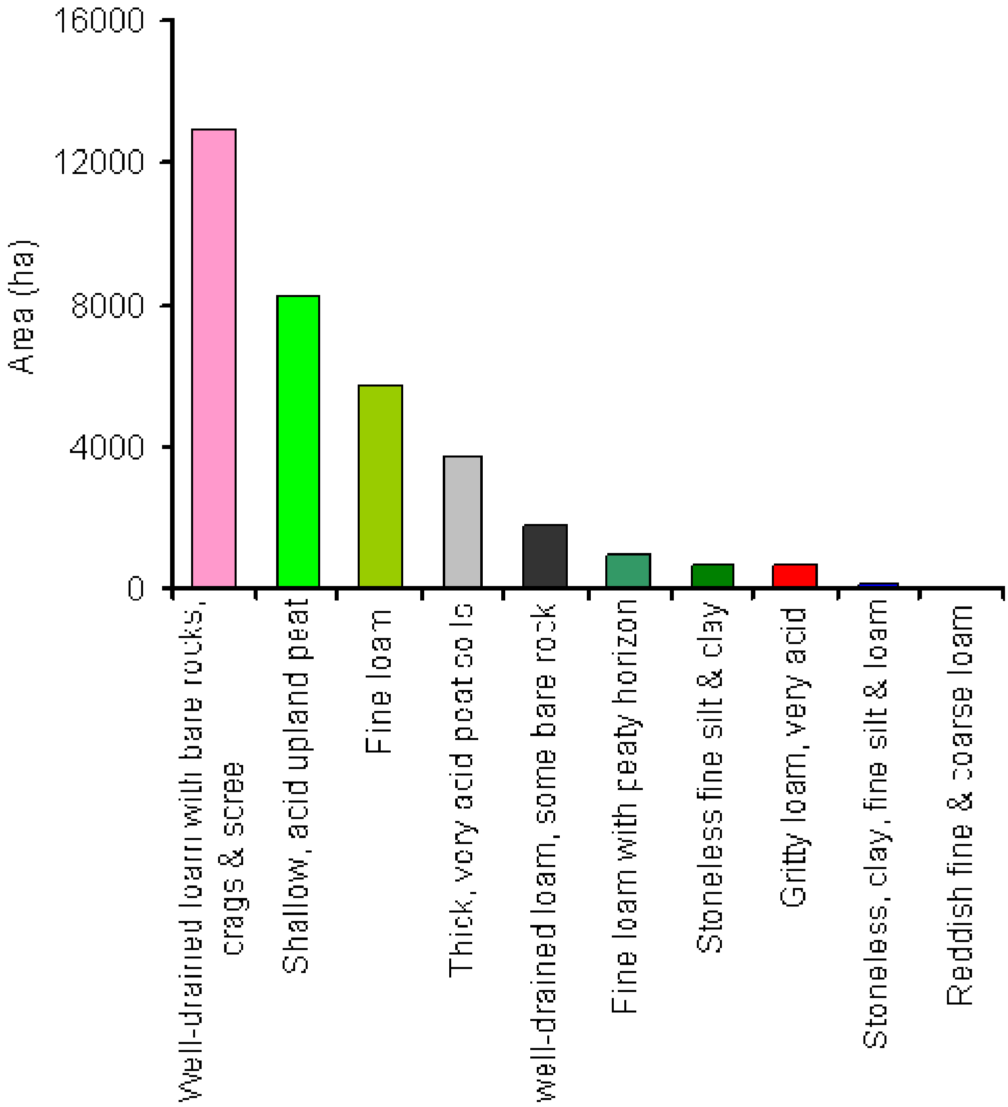


FIG. 4. Soil categories of Bassenthwaite Lake in 1988, based on data in May et al. (1996), originating from the National Soil Resources Institute. © Cranfield University 2004.

PHYSICAL CHARACTERISTICS OF BASSENTHWAITE LAKE

Hydraulic balance, flushing and water level

The catchment area of Bassenthwaite Lake is large compared to the lake volume. As a result, the hydraulic discharge from the lake is also high compared to the lake volume and the average retention time is rather short; only 19 days (Maberly & Elliott 2002). Indeed, the magnitude of this hydraulic flushing is sufficient to affect the course of the seasonal phytoplankton succession in the lake (Reynolds et al. 2002). Water level fluctuations in Bassenthwaite Lake can be dramatic, with recorded rises of up to 1 m during floods (Stokoe 1983).

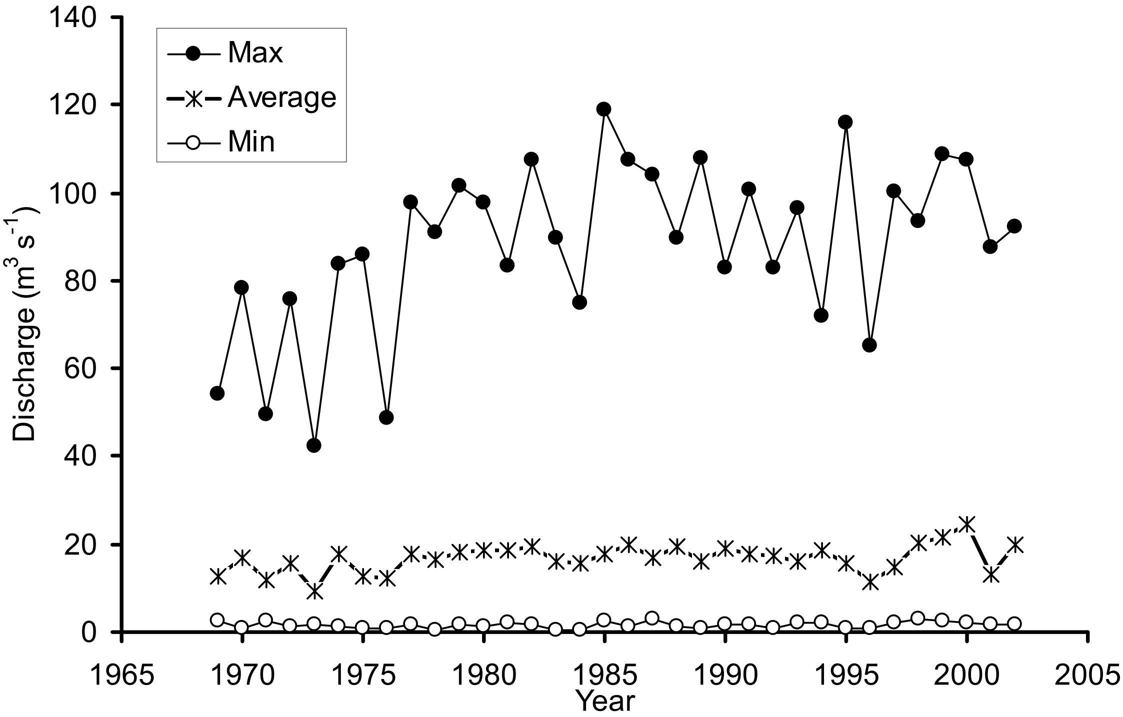


FIG. 5. Annual daily maximum, average (mean) and minimum discharge at Ouse Bridge between 1969 and 2002 (data provided by the Environment Agency).

A number of flow gauges exist in the catchment of Bassenthwaite Lake (see May et al. (1995) for map and grid references). Data from the outflow at Ouse Bridge provide an appropriate measure of the total flow through the lake. The annual average discharge is about 17 $m^3\ s^{-1}$ (Fig. 5) which equals an annual hydraulic discharge of about 527 × 10^6 $m^3\ y^{-1}$. Since 1969, there has been a statistically significant increase in annual mean ($P = 0.015$) and maximum ($P = 0.004$) discharge. The maximum discharge was on 22 December 1985 when 119 $m^3\ s^{-1}$ was recorded. This would produce an instantaneous retention time of 2.7 days between 1969 and 2002.

The seasonal pattern of average discharge shows a minimum around week 28, in early July, with maximum discharge in winter (Fig. 6). Low discharge has occurred on some occasions in winter, but the lowest discharge occurs in dry summers such as in 1976 (minimum 0.64 $m^3\ s^{-1}$) and more recently in 1995 (minimum 0.66 $m^3\ s^{-1}$). These very low discharges would produce an instantaneous retention time of about 497 days.

Linked to the changing discharge are changes in water level (Fig. 7). In winter, water level can vary by up to 2 m. In summer the variation is less marked, particularly during seasonal minima which occur during mid-summer (weeks 26 to 28). The average seasonal range of water level is about 1 m. This is not particularly large but, given the shallow nature of much of the lake (Fig. 2), low water could result in a large area becoming exposed. Low water can be found in virtually any week of the year.

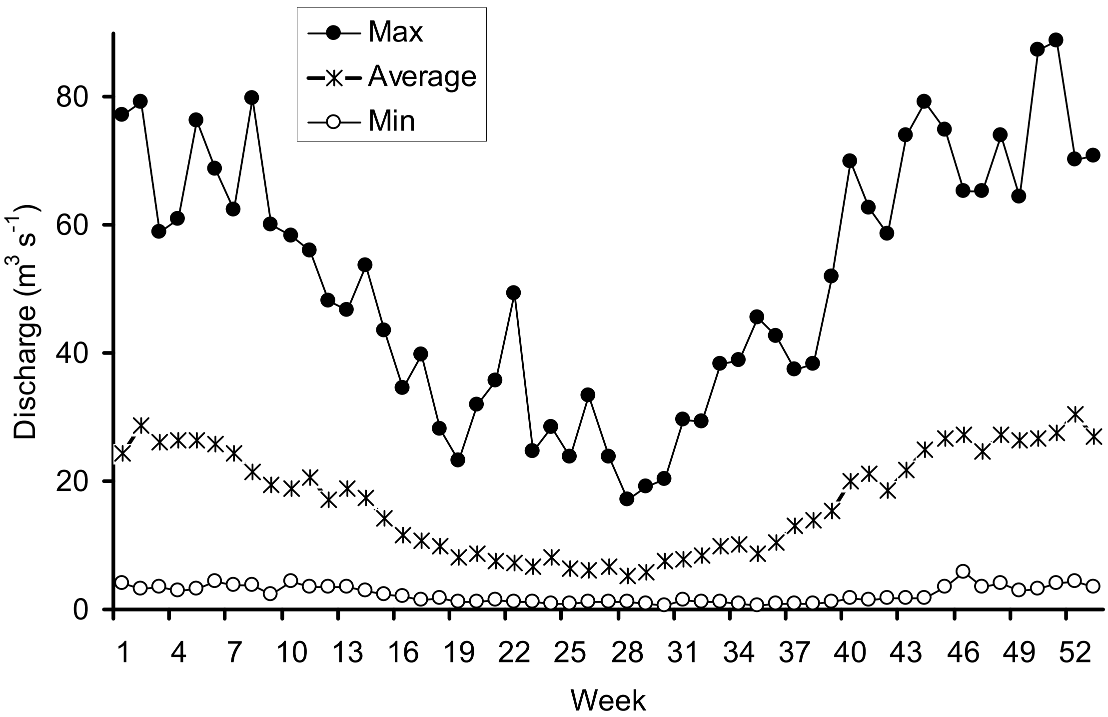


FIG. 6. Weekly maximum, average (mean) and minimum discharge at Ouse Bridge between 1969 and 2002 (data provided by the Environment Agency).

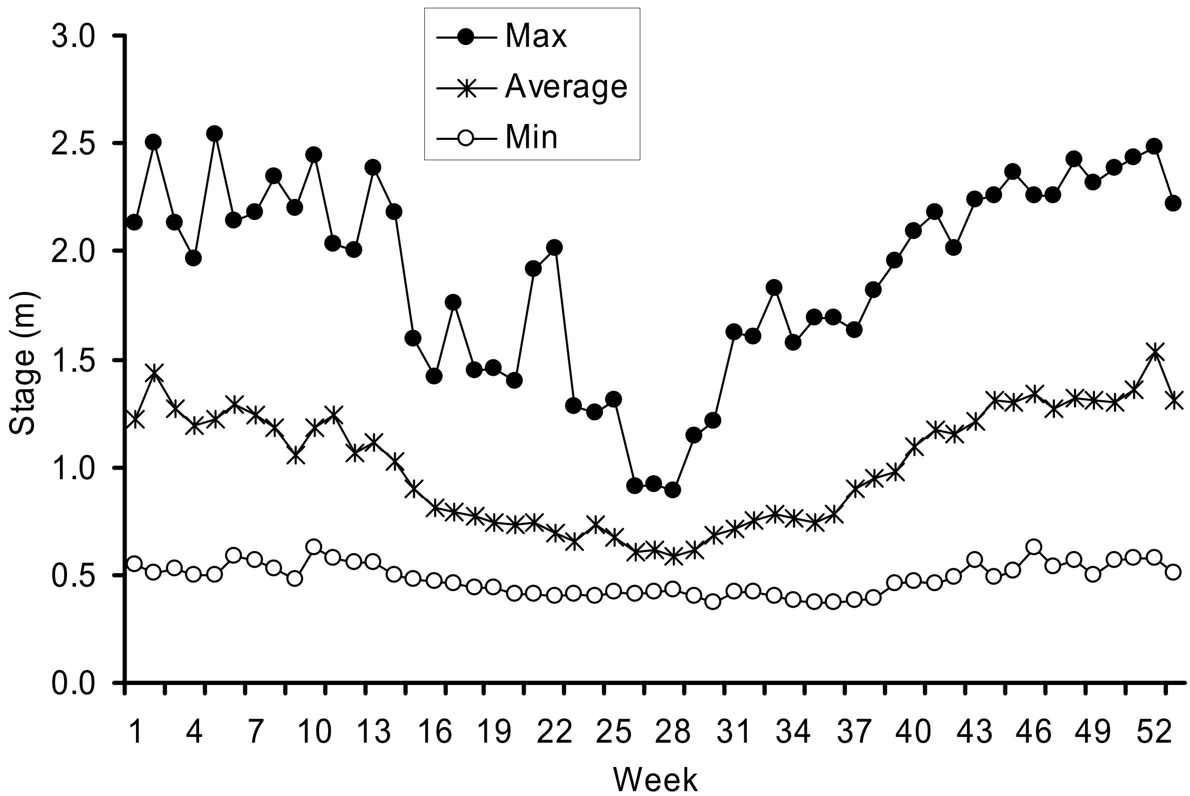


FIG. 7. Weekly maximum, average (mean) and minimum water level at Ouse Bridge between 1977 and 2003 (data provided by the Environment Agency).

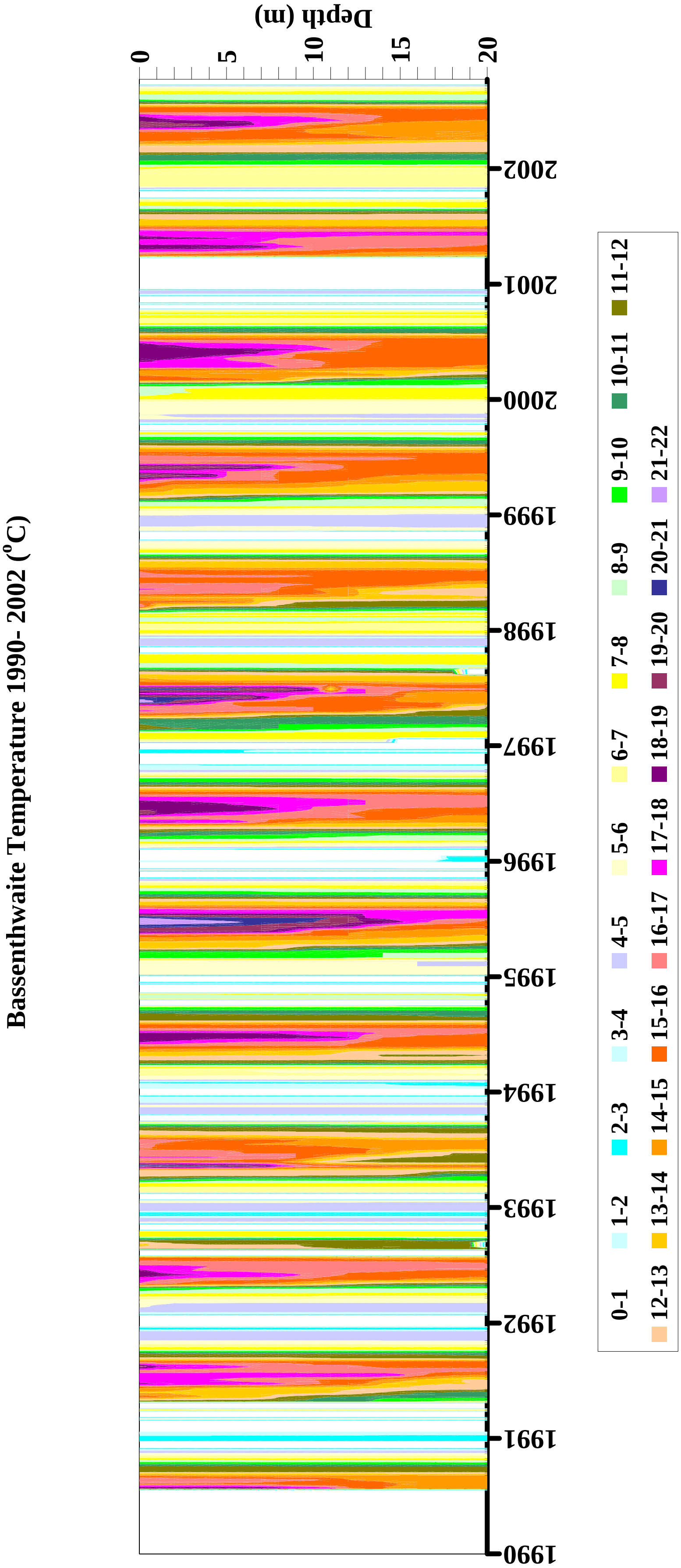


FIG. 8. Thermal structure of Bassenthwaite Lake based on fortnightly sampling between 1990 and 2002. Note gap during 2001 caused by inability to access the lake because of restrictions resulting from the outbreak of foot-and-mouth disease. (Data from CEH).

Mixing and stratification

The seasonal mixing pattern of Bassenthwaite Lake, as indicated by vertical variations in water temperature, can be characterised as 'warm monomictic'. Typically the lake is thermally stratified between May and September, and in overturn for the remainder of the year (Jaworski et al. 1994). However, during the stratified period the vertical temperature difference is relatively small, especially during wet and cloudy summers (Reynolds et al. 1999). As a result the water column structure is readily broken down under windy conditions (Maberly et al. 2003). One consequence of this is that the water temperature at depth is higher than in many other lakes of the English Lake District, with possible consequences for the ecology of the lake. The transience of this thermal structure is also of particular significance to sediment cycling in the lake since turbulent mixing energy is transmitted to the lake bed, allowing resuspension of sediment. The mixing pattern can also vary from year to year (Fig. 8). For example, stratification was weak in 1998 but very strong in 1995 during a prolonged period of summer drought.

Light penetration

Compared to the other major Cumbrian lakes, light penetration in Bassenthwaite Lake is low (Fig. 9). The earliest light attenuation measurements in the lake were made by Pearsall (1920) by measuring the liberation of iodine from acidified potassium iodide on exposure to light. Using this method, Pearsall (1920) recorded a 2 % surface light level at 3.2 m. Pearsall (1921) noted a Secchi disc depth of 2.2 m for Bassenthwaite. This is comparable to values obtained in recent times, although Pearsall used a 7 cm diameter disc while a 30 cm disc is used today so the depth values cannot be exactly equated. Secchi disc transparencies of less than 3 m are commonplace and are somewhat typical of the more productive Lake District lakes (Parker et al. 2001). This results from the presence of dissolved humic material, suspended sediments and phytoplankton (Atkinson et al. 1989). However, there is a degree of variation in this property during the course of the year as a result of temporal changes in phytoplankton biomass and of resuspension of lake sediments (Talling 1999). The resuspension of lake sediments also brings about increases in the turbidity of the outflow from the lake (Hall et al. 2001). There is some evidence for a long-term change in transparency: data from the CEH (Centre for Ecology and Hydrology) 'Lakes Tours' suggests that the mean annual Secchi depth has decreased between 1991 and 2000 (Parker et al. 2001) but although there has been a decrease in minimum and

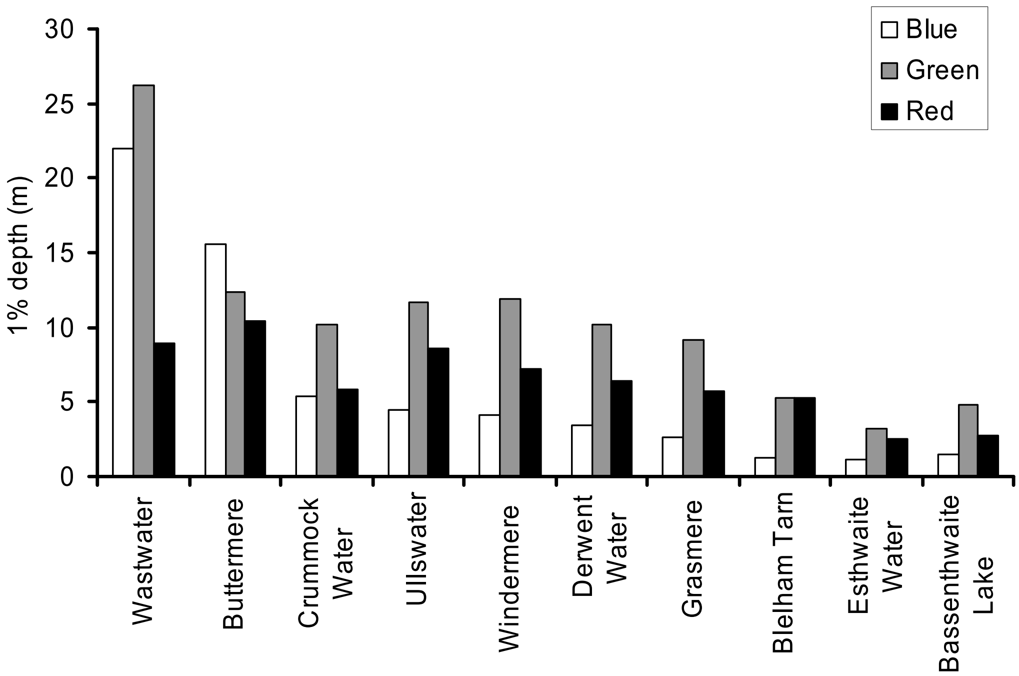


FIG. 9. Depth at which light is attenuated to 1 % of the surface level for blue, green and red wavebands in summer 1994, for 10 Cumbrian lakes (Maberly, Taylor & George, unpublished data).

average Secchi depth based on the fortnightly data between 1990 and 2003 (Fig. 10), this is not statistically significant.

Measurements of light attenuation with a spectroradiometer have shown that in Bassenthwaite Lake, as in the more productive lakes, dissolved organic carbon (DOC) causes blue wavelengths to penetrate to shallower depths than red (Fig. 9). Tipping et al. (1988) found, based on samples collected between May 1986 and April 1987, that the concentration of DOC in the lake was 1.7 mg L^{-1} on average, close to the average for the 13 English Lake District lakes they studied which varied between 0.7 mg L^{-1} (Buttermere) and 2.6 mg L^{-1} (Esthwaite Water). The most penetrating waveband in Bassenthwaite Lake, the green waveband, had a 1 % depth of 4.8 m (Fig. 9). The 1% depth roughly defines the lower limit of the euphotic zone, and a depth of 4.8 m or shallower would occur over 35 % of the lake surface area (Ramsbottom 1976).

In many lakes, Secchi disc depth is controlled by the concentration of phytoplankton chlorophyll *a*. In Bassenthwaite Lake, while high concentrations of phytoplankton chlorophyll *a* do produce shallow Secchi depths, Secchi depth can also be low when there is very little phytoplankton chlorophyll *a* (Fig. 11). This is caused by the high concentration of suspended solids, which can reach nearly 10 mg L^{-1} (measured since November 1999; Fig. 12). Therefore, in this lake, transparency is controlled by both phytoplankton and suspended solids.

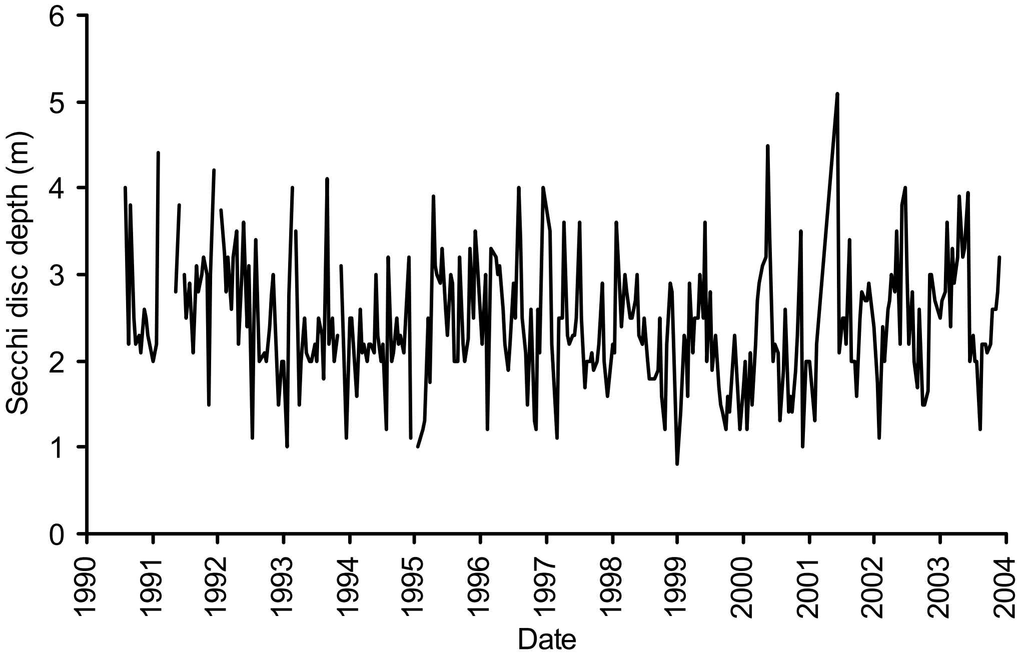


FIG. 10. Long-term changes in Secchi depth in Bassenthwaite Lake based on fortnightly sampling (data from CEH).

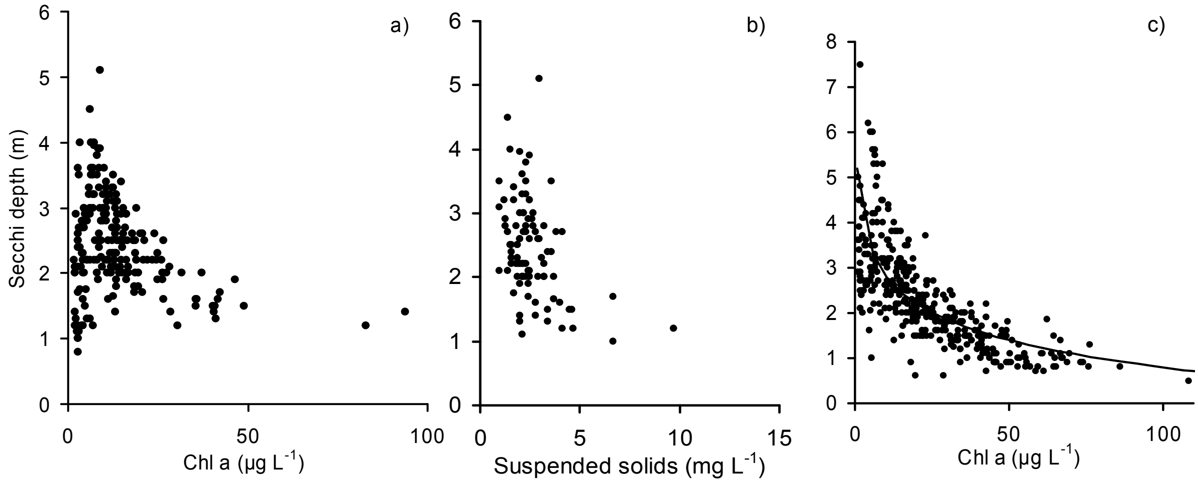


FIG. 11. Relationship between a) depth of Secchi disc and phytoplankton chlorophyll *a* (1990–2003) and b) depth of Secchi disc and suspended solids (1999–2003). For comparison, c) shows the relationship between Secchi depth and chlorophyll *a* in Esthwaite Water where the influence of suspended solids is believed to be low. (Data from CEH, and FBA for (c)).

Suspended solids are of ecological interest because they influence the underwater light climate and may also be one of the causes in the decline of a rare fish species, vendace, in the lake. The concentration of suspended solids has been very variable, with very high concentrations in December 1999 and December 2000 with a peak concentration of 9.7 mg L^{-1} on the

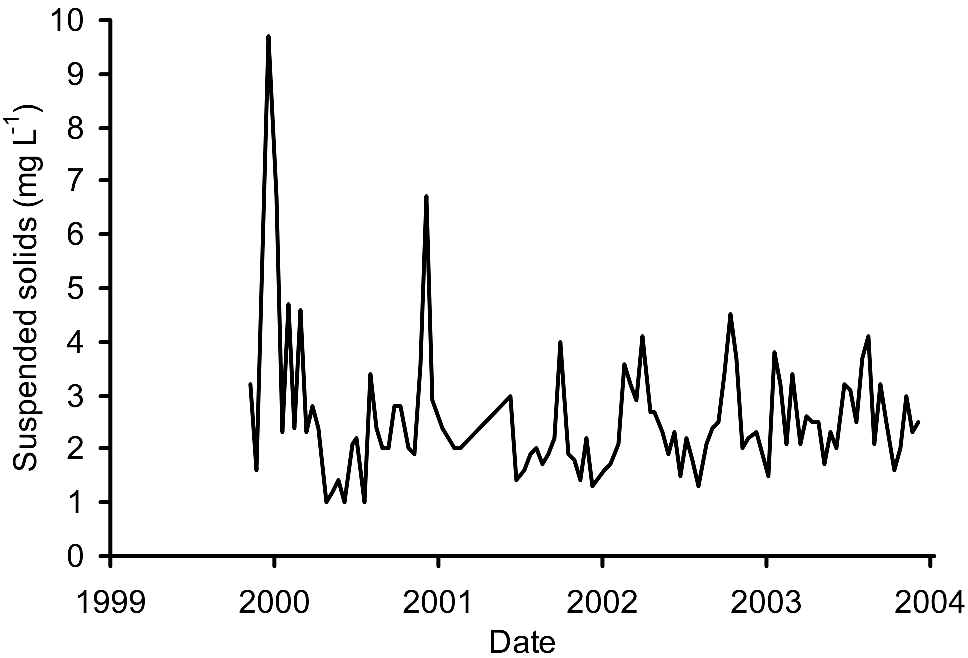


FIG. 12. Suspended solids in Bassenthwaite Lake between November 1999 and December 2004 (data from CEH).

former date (Fig. 12). Since 2000, annual maxima have been lower: typically around 4 mg L^{-1}, although it is not clear if these lower values will continue or what was the cause of the higher values at the start of the record.

WATER CHEMISTRY

Sources of nutrients to Bassenthwaite Lake

Phosphorus budgets before tertiary treatment at Keswick WwTW (pre-1995)

The first phosphorus budget for Bassenthwaite Lake was produced by Hilton et al. (1993). This was based on an eight-month campaign (January to August 1993) using data collected frequently by the then National Rivers Authority. The concentration of soluble reactive phosphorus (SRP), nitrate-nitrogen and silicate-silica (note, not total phosphorus (TP)) was measured on 11 inflowing streams, and on the River Derwent (the main inflow) where samples were taken above and below the sewage treatment works to distinguish between the load from the catchment and the treatment works. Daily estimates of load were calculated as the product of discharge and concentration. Daily discharge was produced by interpolation of measurements every two to seven days.

This work estimated a total annual load of SRP to the lake of 7846 kg (Table 2) of which 78 % was derived from the four wastewater treatment works (WwTW), and 72 % from Keswick WwTW alone. In order to obtain an estimate of the load of TP, Hilton et al. (1993) assumed a multiplier of

1.7 to convert from SRP to TP. Using this value, the total load to the lake was estimated to be 13 338 kg P y^{-1}.

May et al. (1995; 1996; 2001) estimated loads of TP to the lake on the basis of land use categories within the stream subcatchments, combined with published export coefficients for different land use types and information on human population. Using this approach and further direct measurements of total phosphorus loads (Lawlor & Tipping 1996) the best estimate of the 1993 load to Bassenthwaite Lake was 16.7 Mg y^{-1} (Reynolds 1999). Of this total load, 60 % came from the catchment and 37 % from Keswick WwTW. The best estimate of the 1993 soluble phosphorus load was 7.9 Mg y^{-1}, 22 % of which came from the catchment and 71 % of which came from Keswick WwTW.

The estimates of phosphorus load to the lake do not include loads from the storm overflow at Keswick WwTW since these data were not available. They also exclude any internal loading of phosphorus resulting from release

Table 2. External phosphorus load to Bassenthwaite Lake derived from the months from January to August 1993 (Hilton et al. 1993) and calculated for 12 months on a *pro rata* basis. Values in parentheses are the percent contribution of each source to the total load.

	SRP load (kg)	
Sources of phosphorus	8-month	12-month
WwTW		
Keswick WwTW	3757	5636 (71.8)
Thornthwaite WwTW	58.6	87.9 (1.1)
Embleton WwTW	133	199.5 (2.5)
Bassenthwaite WwTW	125.4	188.1 (2.4)
Catchments		
River Derwent (above Portinscale)	881.8	1322.7 (16.9)
Newlands Beck	119.9	179.9 (2.3)
Tributary downstream of Thornthwaite WwTW	1.8	2.7 (0.03)
Beckstones Gill	1.8	2.7 (0.03)
Beck Wythop	3.3	5.0 (0.06)
Dubwath Beck	0	0 (0)
Un-named tributary (NY208320)	53.9	80.9 (1.0)
Chapel/Halls/Dash Becks	35.8	53.7 (0.68)
Pooley Beck	9.4	14.1 (0.18)
Tributary near Bowness Bay	15.8	23.7 (0.30)
Skill Beck	26.6	39.9 (0.51)
Total WwTW	**4074**	**6111 (77.9)**
Total from catchment	**1150**	**1735 (22.1)**
Total load	**5224**	**7846**

from the sediments during wind-induced mixing. This internal load can be important in a shallow lake such as Bassenthwaite and Davies & Reynolds (1994) have shown that the retention of phosphorus in sediments from Bassenthwaite Lake is poor.

Phosphorus budgets after tertiary treatment at Keswick WwTW (post-1995)

Reynolds (1999) summarised the estimated revised load from the Keswick WwTW after phosphate stripping was implemented in 1995. During 2000 and 2001, there were only 44 dates for which both flow data and total phosphorus concentration were available. To improve the estimate of load, monthly average flow and concentration data were calculated. Their product gave an estimate of monthly load for each year. The validity of this approach is strengthened by the fact that there was no correlation between concentration and flow on the 44 occasions when both were measured on the same date. The monthly total phosphorus loads for six years, along with the monthly average, are shown in Fig. 13. The very low loads in the summer of 1995 presumably result from the relatively dry weather, although 1996 was drier but the load was higher. The high values in 2000 may reflect the relatively wet year. On average, the load of total phosphorus to the lake is fairly even month to month and averages 193 kg P month^{-1}.

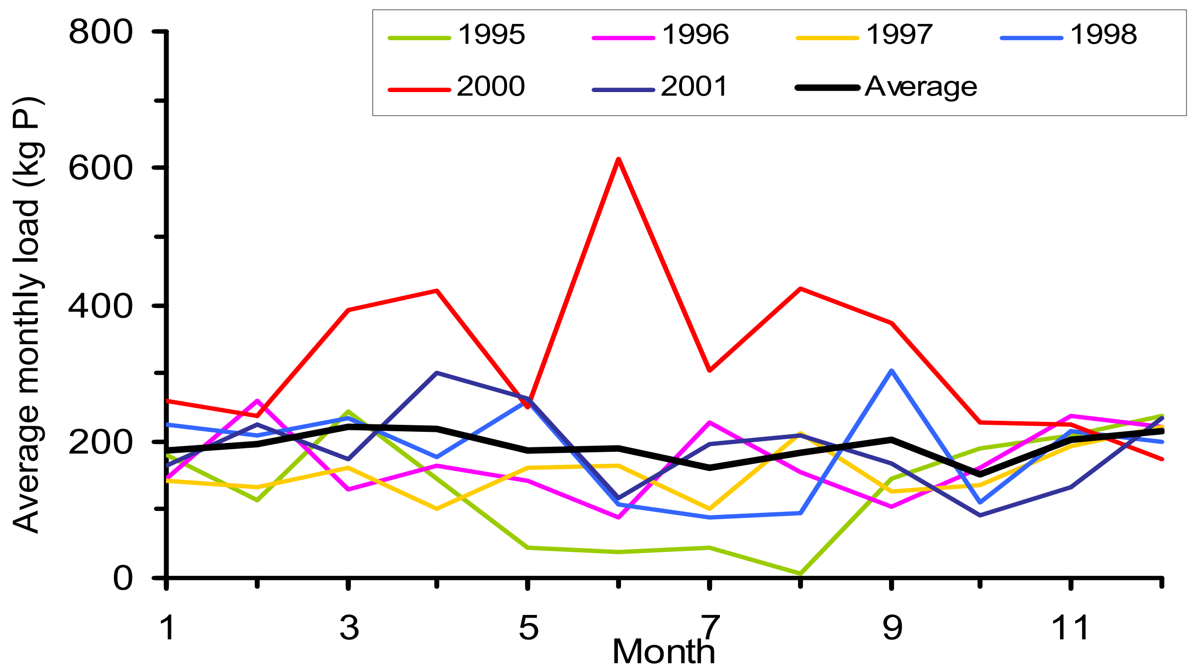


FIG. 13. Average (mean) monthly load of total phosphorus from Keswick WwTW for six years, plus the monthly average for all the years.

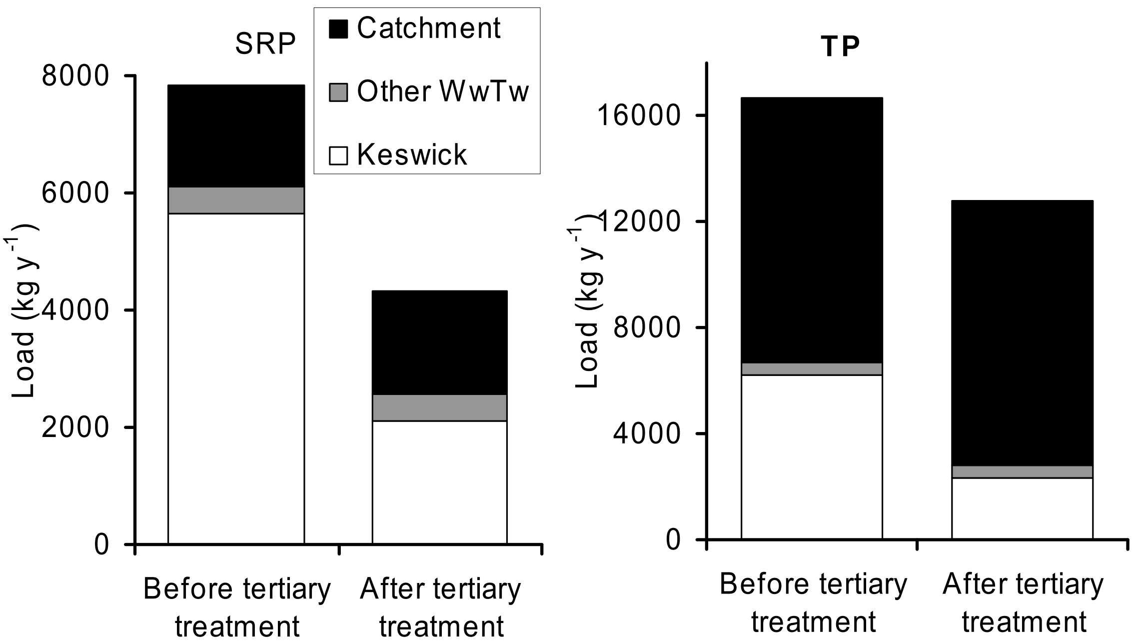


FIG. 14. Summary of external loads of soluble reactive phosphorus (SRP) and total phosphorus (TP) from the Keswick WwTW, other WwTW and the catchment before and after the installation of tertiary P-removal in 1995. Based on an analysis in Maberly & Elliott (2002).

The tertiary treatment has, on average, reduced the total phosphorus load from Keswick WwTW to 2312 kg y^{-1} from 6200 kg y^{-1} in 1993, a reduction of 3888 kg y^{-1} (63 % reduction; Fig. 14). The total load of phosphorus to the lake is calculated to have reduced from 16700 kg y^{-1} in 1993 to, on average, 12 812 kg y^{-1} after the phosphorus stripping (23 % reduction). As a result of tertiary treatment at Keswick WwTW, the proportion of the load contributed by the catchment has increased was from 22 % to 44 % for SRP load and from 60 % to 78 % for load.

Loads of other nutrients

May et al. (1995) estimated the load of NO_3-N and SiO_2 to be 128 000 and 699 000 kg y^{-1} respectively. The River Derwent accounted for 70 % of the NO_3-N load and 64 % of the SiO_2 load.

Water chemistry of Bassenthwaite Lake

Major ions

Here the term ‘major ions’ encompasses the following ions in order of decreasing concentration: chloride (Cl^-), calcium (Ca^{2+}), sodium (Na^+), bicarbonate (HCO_3^-), magnesium (Mg^{2+}) sulphate (SO_4^{2-}), nitrate (NO_3^-) and potassium (K^+). In terms of major ionic composition, Bassenthwaite Lake is similar to the more productive English Lakes such as Esthwaite

Water and Blelham Tarn (Sutcliffe et al. 1982). Although relatively rich compared to the other Cumbrian lakes, Bassenthwaite Lake has a relatively low ionic content in a world-wide context (Atkinson et al. 1989). There is evidence for a long-term increase in the conductivity of the lake water (Sutcliffe et al. 1982). The average composition of the major ions based on four samples taken in 2000 (Parker et al. 2001) is given in Table 3. There is an approximate balance between cations and anions, and the total ionic strength is 0.7 mmol L^{-1}.

Alkalinity and pH

In the English Lake District lakes, alkalinity provides a surrogate measure of the concentration of bicarbonate. Temporal variations in the alkalinity of Bassenthwaite Lake are strongly cyclical (Jaworski et al. 1994; Reynolds et al. 1998). Each year the alkalinity begins to rise in spring, reaching a maximum in mid-summer. During the autumn and winter the alkalinity falls rapidly. For the period 1990–2003, maximum values were often in excess of 250 µequiv L^{-1} (12.5 mg $CaCO_3$ L^{-1}) while minima were frequently as low as 140 µequiv L^{-1} (7 mg $CaCO_3$ L^{-1}). In 2000, the annual average was 180 µequiv L^{-1} (Parker et al. 2001; Table 3). Seasonal fluctuations of this type are frequently found in lakes. The high values are partly brought about by lower rainfall and higher evaporation during the summer months (Sutcliffe et al. 1982) but seasonal uptake of nitrate during the summer also generates alkalinity. As well as these annual fluctuations, there appears to be a trend of increasing alkalinity since about 1949 (Talling 1999). The data since 1990 also suggest that alkalinity has increased since about 2000 (Fig. 15).

Table 3. Major ionic composition in Bassenthwaite Lake based on four samples taken in 2000 (Parker et al. 2001). Records of major ion chemistry from the 1950s and 1970s are available in Sutcliffe et al. (1982) and Carrick & Sutcliffe (1982).

Anions		**Cations**	
Ion	Concentration/ µequiv L^{-1}	Ion	Concentration/ µequiv L^{-1}
Alkalinity (HCO_3^-)	180	Na^+	211
Cl^-	262	K^+	12.1
NO_3^-	23.4	Ca^{2+}	232
SO_4^{2-}	82.7	Mg^{2+}	83.2
TOTAL	**548**	**TOTAL**	**538**

Temporal variations in pH show a rather similar periodicity to that apparent in alkalinity. In winter and early spring, pH is frequently less than 7, though this often increases to pH 7.5 or, in extreme cases, pH 9 during the summer (Jaworski et al. 1994; Reynolds et al. 1998) (Fig. 16). Such elevations of pH are frequently brought about during periods of high phytoplankton productivity which results in depletion of CO_2 (Talling 1976): for example, in Esthwaite Water surface pH exceeds 10.0 for a short time in most summers (Maberly 1996). In Bassenthwaite Lake, elevated pH is less marked than in lakes of comparable productivity, probably because of high rates of CO_2-supply resulting from the large hydraulic throughput.

Nutrients

Historical changes in nutrient concentrations

Atkinson et al. (1989) summarised data from numerous sources, showing long-term variations in the surface water chemistry of Bassenthwaite Lake (Table 4).

These data, originating from a series of isolated surveys spanning the period 1920 to 2000, suggest some significant historical changes in surface water chemistry. In the second half of the 20th century there was an increase in the conductivity and alkalinity of the lake water. There was also a trend of increasing nitrate-nitrogen and total phosphorus concentrations over this time period. This suggestion of increasing eutrophication of the lake is supported by palaeolimnological data, which indicate that, though Bassenthwaite Lake has been mesotrophic for approximately 250 years, there has been a relatively recent deterioration in water quality (Bennion et al. 2000). Long-term changes in the total phosphorus concentration of the lake have been reconstructed based on fossil diatom remains. Three distinct phases of change were evident in the species composition of the fossil diatoms over the period 1710–1995. Sediments accumulated between 1710 and 1860 contained a diverse community of small centric and pennate planktonic diatoms, as well as epiphytic taxa. These taxa were characteristic of nutrient-poor waters. Sediments accumulated between 1860 and 1960 had a less diverse assemblage, the community being increasingly dominated by *Synedra nana*, *Asterionella formosa* and the mesotrophic indicator *Fragilaria crotonensis*. Subsequent to 1960, the assemblage was co-dominated by two new taxa, *Aulacoseira subarctica* and *Cyclotella pseudostelligera*. In this final period, taxa characteristic of nutrient-rich waters began to appear in the sediment record. Diatom-inferred total phosphorus reconstructions indicated worsening eutrophication post-1860. Estimated total phosphorus concentrations were

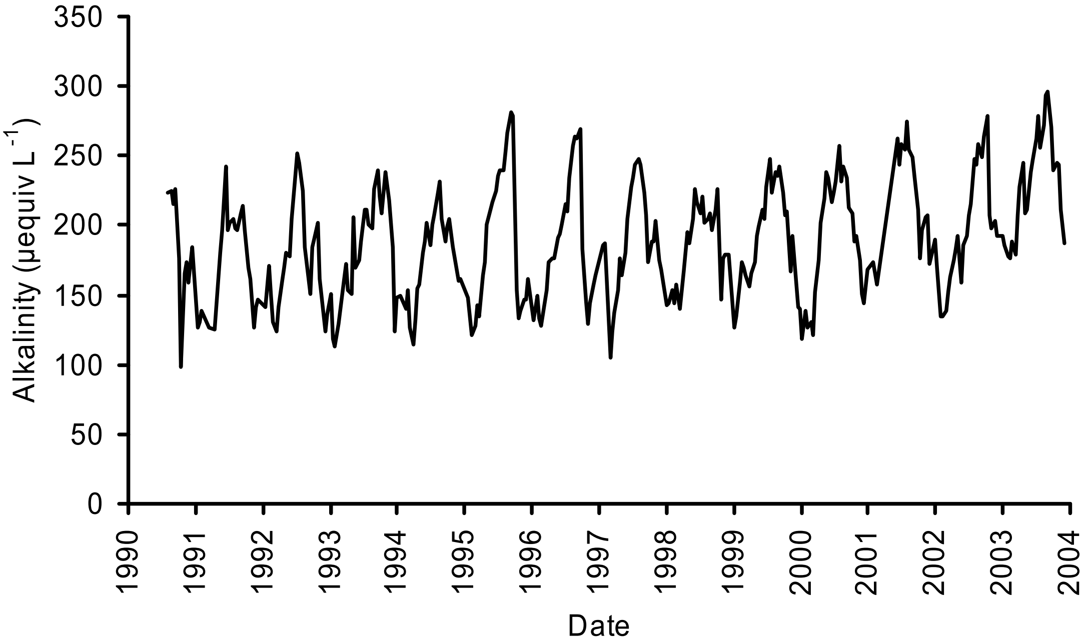


FIG. 15. Long-term changes in alkalinity in Bassenthwaite Lake based on fortnightly sampling (data from CEH).

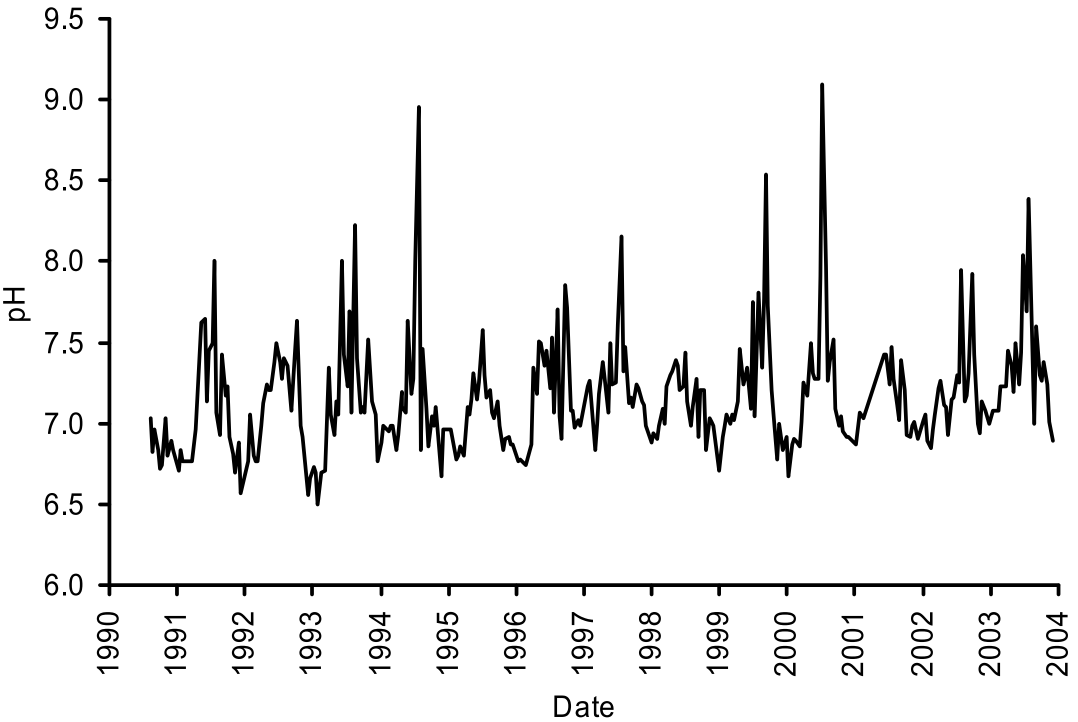


FIG. 16. Long-term changes in pH in Bassenthwaite Lake based on fortnightly sampling (data from CEH).

Table 4. Long-term records of surface water characteristics in Bassenthwaite Lake. Multiple observations are recorded as mean values with number of measurements in parentheses. Pre-1974 values of alkalinity (except b) are corrected by a factor of -20 µequiv L^{-1}, to take differing methodologies into account. Reproduced from Atkinson et al. (1989) with the Lakes Tour survey in 2000 given for comparison.

	1920[a]	1928[b]	1939-1940	1946[d]	1949[d]	1955-1956[e]	1971[f]	1974-1976[e]	1984[g]	1987-1988[h]	2000[i]
Secchi transparency (m)	2.2									2.5 (12)	2.1 (4)
Conductivity (k_{20}, µS cm^{-1})			52.0 (3)				69.3 (7)			63 (12)	66.6 (4)
Alkalinity (µequiv L^{-1})		148 (9)	137 (2)		129 (13)	168 (8)		189 (25)		188 (12)	180 (4)
pH		7.0 (9)	6.6					6.9 (22)		6.8 (12)	7.01 (4)
NO_3-N (µg L^{-1})		91.9 (9)			151 (13)	252 (8)		224 (9)	390 (4)	370 (12)	326 (4)
PO_4-P (µg L^{-1})		2.1 (9)			1 (10)		0.85 (2)		1 (4)	5.0 (12)	1.6 (4)
Total P (µg L^{-1})				14			19 (3)		23 (2)	35 (12)	20.0 (4)
SiO_2-Si (mg L^{-1})		1.2 (9)			1.9 (13)				1.22 (4)	1.54 (12)	1.34 (4)
Chlorophyll *a* (µg L^{-1})							10.6 (2)			15.3 (12)	14.5 (4)

Sources:
a Pearsall (1921)
b Pearsall (1932)
c Mortimer, unpublished notebooks (FBA)
d Mackereth & Lund, unpublished
e Sutcliffe et al. (1982)
f Jones (1972)
g FBA unpublished
h Mubamba (1989)
i Parker et al. (2001)

stable at about 20 µg TP L^{-1} pre-1860, rising steadily to around 30 µg TP L^{-1} by 1960, and rising dramatically to 50–60 µg TP L^{-1} between 1970 and 1985. After 1985, values decreased to approximately 30 µg TP L^{-1}. Although the inferred TP concentrations in recent years have been higher than actual measured concentrations, the pattern of change in inferred concentrations does agree with historical records of nutrient concentrations.

Since 1990, more intensive routine sampling has been conducted on Bassenthwaite Lake, allowing the identification of seasonal patterns and longer-term trends in nutrient chemistry over the past decade. In the following sections, the results of these routine chemical analyses will be described.

Phosphorus (pre-1995)

Prior to 1993, temporal variations in soluble reactive phosphorus (SRP) concentration were characterised by strong seasonal cycles (Jaworski et al. 1994) (Fig. 17b). Typically concentrations were highest in winter (~10 μg L^{-1}) but were quickly reduced to limiting amounts (< 1 μg L^{-1}) during the spring, as a result of uptake by phytoplankton. Concentrations remained low during the summer, until they increased during overturn. In contrast, no such seasonal periodicity was evident in total phosphorus concentration (Fig. 17a), which had shown a trend of increasing concentration between the 1970s and the 1980s (Reynolds 1999). Between 1993 and 1994, the nature of these temporal variations changed. Peak concentrations of SRP and total phosphorus were higher during this period (Figs 17a & b) and the period of SRP depletion, during the summer months, became shorter (Jaworski et al. 1994; Reynolds 1999). The reason for this is not known.

Phosphorus (post-1995)

Since the initiation of tertiary treatment at Keswick WwTW in 1995, there appears to have been a modest improvement in the water quality of Bassenthwaite Lake. Although mean annual concentrations of SRP and total P do not appear to have changed dramatically since 1995, annual maxima of both variables appear to have decreased (Maberly et al. 2003). There has also been a slight, though statistically significant, decrease in annual mean total phosphorus concentration since 1995 (Maberly et al. 2004). Also, there have been more extensive periods of reduced SRP concentrations during the summer and less of the erratic variability that characterised the early 1990s (Reynolds 1999). Pre-1993 seasonal patterns of variation in SRP appeared to have resumed (Fig. 17b) (Reynolds et al. 2001), and there has also been a modest reduction in total phosphorus concentration and in the frequency of sediment resuspension related peaks in this variable. It has been suggested that the lack of a more dramatic reduction in phosphorus concentration is a result of recycling of phosphorus from the sediments (Reynolds et al. 2000).

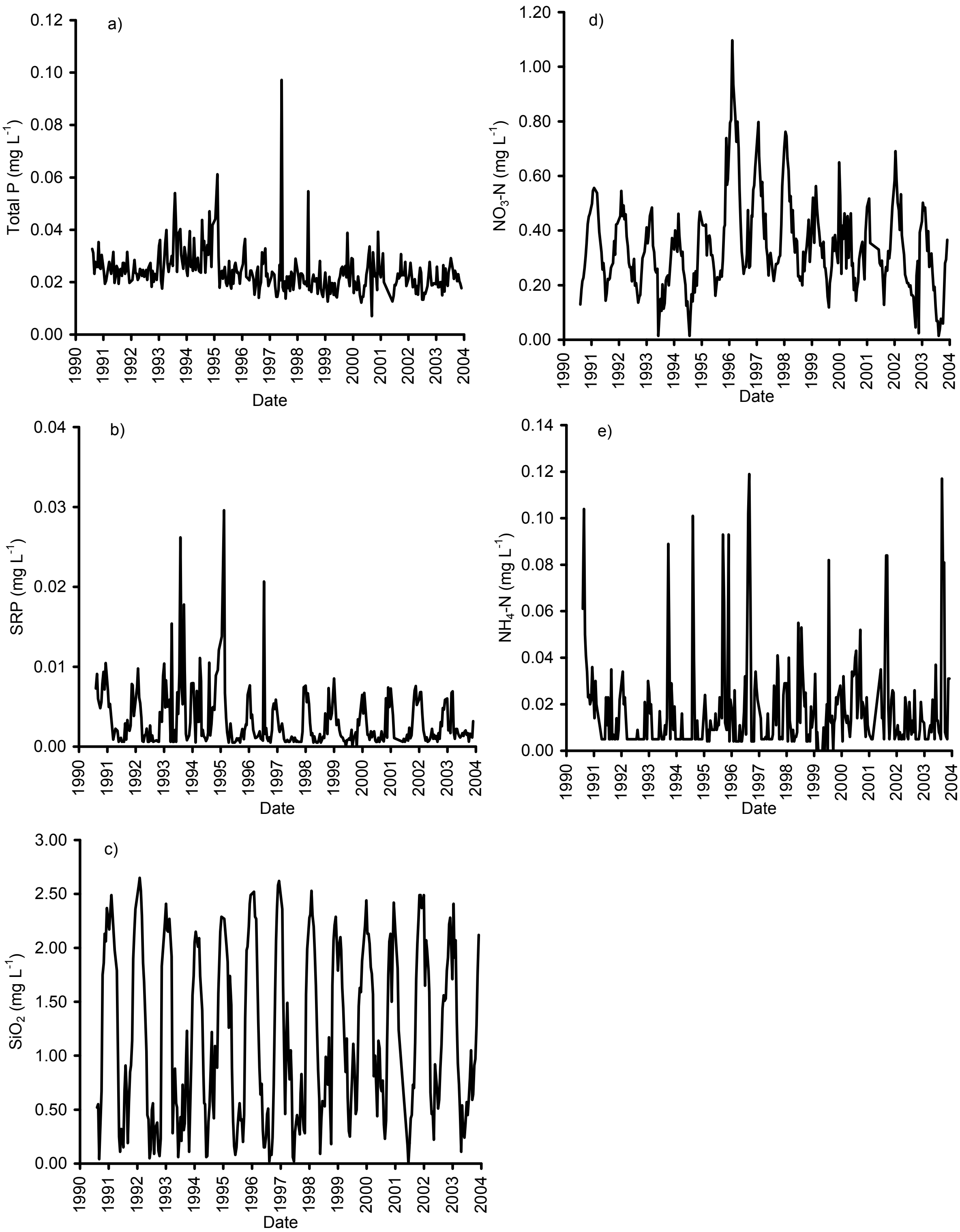


FIG. 17. Changes in Bassenthwaite Lake of: a) total phosphorus, b) soluble reactive phosphorus, c) silica, d) nitrate and e) ammonium (data from CEH).

Nitrogen

Nitrate is the dominant form of inorganic nitrogen in Bassenthwaite Lake. Concentrations of nitrate follow a consistent seasonal cycle: they typically peak in February/March and fall to a low point during the summer, before rising into the following year (Jaworski et al. 1994; Reynolds et al. 1995a; Reynolds et al. 1997) (Fig. 17d). Prior to 1995, annual maximum concentrations of nitrate-nitrogen were approximately 500 µg L^{-1}. These concentrations are substantially higher than those measured in 1928 (Table 4). Following the instigation of tertiary treatment at Keswick WwTW there was an initial marked increase in annual winter maximum and summer minimum concentrations (Fig. 17d). This is consistent with a reduced demand for nitrogen as a result of a reduction in availability of phosphorus. If this interpretation is correct, however, the data since 2002 suggest that the availability of phosphorus has substantially increased (Fig. 17d).

Concentrations of ammonium-nitrogen are typically much lower than those of nitrate-nitrogen and, apart from occasional high values, some of which are associated with periods of overturn, maximum values typically reach around 35 µg L^{-1} (Fig. 17e). There is no well defined and reproducible seasonal pattern in the concentration of ammonium-nitrogen (Jaworski et al. 1994; Reynolds et al. 1995a; Reynolds et al. 1997).

Silica

Since the initiation of routine sampling, annual variations in the concentration of silica have followed a consistent seasonal cycle (Reynolds et al. 1995a; Reynolds et al. 1997). Each year peak concentrations of 2 mg L^{-1} to 2.5 mg L^{-1} have been detected during the winter months with a rapid depletion during spring by diatom growth (Fig. 17c). During the summer and early autumn silica concentrations are low (< 0.5 mg L^{-1}), rapidly increasing in response to the termination of thermal stratification.

Dissolved oxygen

Since the initiation of a monitoring programme on Bassenthwaite Lake, seasonal variation in oxygen concentrations has been rather consistent (Fig. 18). Typically, the water column is well oxygenated until April, prior to thermal stratification. During the stratified period deep water oxygen concentrations decline, until the lower water column becomes anoxic by late August/early September (Jaworski et al. 1994; Reynolds et al. 2002; Maberly et al. 2003). Temporal changes in vertical profiles of dissolved oxygen concentration have frequently revealed wind-induced mixing events that have temporarily broken down the density stratification in the

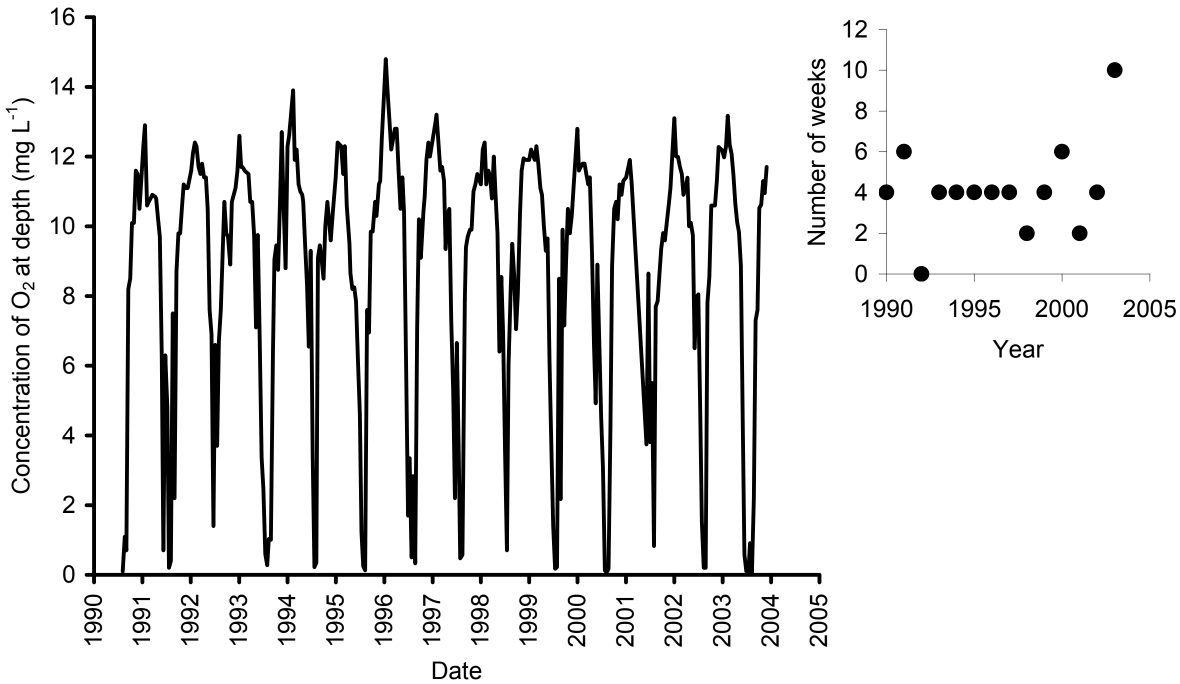


FIG. 18. Oxygen concentration at depth (20 m) in Bassenthwaite Lake between 1990 and 2004. Inset shows the number of weeks the concentration of oxygen was below 1 mg L^{-1} each year (data from CEH).

water column during the summer months (Jaworski et al. 1994; Reynolds et al. 1999). During these events the oxygen concentration of the lower water column can increase sharply due to the entrainment of oxygen-rich water from the epilimnion into the hypolimnion. As Bassenthwaite is a productive lake, oxygen depletion occurs at depth in most years (Fig. 18). As yet, there is no evidence for a long-term change in the severity of oxygen depletion at depth (Maberly et al. 2004) although in 2003 oxygen concentration at depth was below 1 mg L^{-1} for longer than had been recorded previously (Fig. 18 inset).

Sediment

Physical characteristics

Detailed analyses have revealed that much of the lake-bed sediment and settling particulate matter in Bassenthwaite Lake is inorganic (Parker et al. 1999; Bennion et al. 2000). Based on samples of surficial sediment collected in 1999, Parker et al. (1999) found that organic matter constituted only 1.5 % of the total sediment dry mass on average for lake bed sediments. The magnitude of the organic component was greater for the settling particulates caught in traps placed in the lake (23.2 % dry mass), though the material was still primarily inorganic in nature. An analysis of the organic content of a single sediment core taken in 1995 by Bennion et al. (2000), indicated that the organic content was higher (9 % – 15 % dry mass) in the upper 80 cm of the core, though still minor compared to the inorganic component.

Chemistry

Much of the work that has been done on the chemistry of Bassenthwaite Lake sediments has focussed on the content and fractionation of phosphorus. Initially, it was thought that a large internal store of phosphorus was present in the sediments. This supposition was based on the fact that in-lake nutrient concentrations did not appear adequate to support the observed phytoplankton biomass (Jaworski et al. 1994; Reynolds et al. 1995b) and that early attempts to model the phytoplankton community, using only external loads of phosphorus as a nutrient source, underestimated observed chlorophyll concentrations (Reynolds et al. 1996; Maberly & Elliott 2002). However, subsequent analyses of phosphorus content and fractionation in the sediments revealed that the internal store was not as large as initially thought, with concentrations ranging from only 1.0 mg P g^{-1} to 2.5 mg P g^{-1} dry matter (Davies & Reynolds 1994). Concentrations were lower than in the sediments of a number of other major Lake District lakes and, furthermore, these sediments were undersaturated with respect to phosphorus. It was apparent that much of this limited internal store comprised potentially mobile fractions, in particular the redox sensitive iron-bound fraction. Based on these findings, it has been suggested that the high levels of productivity in Bassenthwaite Lake are partially supported by efficient recycling of this relatively modest internal store, rather than by its quantity (Davies & Reynolds 1994; Irish & Reynolds 1995; Reynolds et al. 1995b). This efficient recycling allows the same initial loading to support several successive algal generations.

The chemical composition of the sediments is subject to a high degree of spatial heterogeneity. The total phosphorus, carbon and nitrogen content was shown by Parker et al. (1999) to be higher in deep water sites than in shallow water sites, though the reverse trend in total phosphorus concentration was shown by Reynolds et al. (1995b). There is also a high degree of spatial variation in the concentrations of organic matter and major metals (Hall et al. 2001).

After the introduction of tertiary treatment at the Keswick WwTW, attempts were made to establish whether sediments close to the treatment works had been contaminated as a result of the treatment process (Simon & Lawlor 1998). The content of iron, cobalt, cadmium and total sulphur compounds was analysed at different distances from the treatment works.

No evidence was found for contamination of the sediments in the immediate proximity of the treatment works, though the spatial extent of this survey was rather limited.

Sources

The carbon to nitrogen ratio in the sediments is typical of terrestrially derived material and suggests that sedimenting phytoplankton cells form a minor component of these sediments (Parker et al. 1999). Attempts to identify which parts of the catchment are supplying this sediment by finding a characteristic 'chemical signature' for the lake bed sediments, based on concentrations of organic matter and major metals, have been confounded by the high degree of spatial heterogeneity in these chemical properties (Hall et al. 2001).

However, areas with a high potential for erosion have been identified using a modelling approach based on consideration of soil structural and hydrological properties. Analysis of slope and vegetation cover within these areas has allowed further identification of locations that have the potential to act as sources of sediment (Fig. 19). It is thought that the greatest potential sediment sources in the catchment are patches of eroded ground on the high fells, particularly on the Skiddaw Massive (Orr et al. 2004). However, significant quantities of fine- and coarse-grain sediment are also supplied by channel erosion, particularly in the area between Derwent Water and Bassenthwaite Lake.

There are believed to be a number of other sources of sediment that are, as yet, unquantified. The poaching of field drains by livestock, particularly on the floodplain of the River Derwent, is likely to be important as is the erosion of sub-surface drains throughout the catchment and of abandoned mine waste in the Newlands valley. Further research should aim to quantify the importance of these potential sediment sources (Orr et al. 2004). Derwent Water and Thirlmere are thought to act as sediment traps so that a relatively small proportion of the catchment supplies sediment directly to Bassenthwaite Lake (Hall et al. 2001; Orr et al. 2004). Research on sediment delivery to Bassenthwaite Lake should therefore focus on streams that enter the lake directly.

Sedimentation and resuspension

Gross sedimentation rates were measured in Bassenthwaite Lake during 1999 using sediment traps. The traps were designed to prevent resuspension of trapped material and hence measure gross sedimentation (which is not equivalent to accumulation). Gross sedimentation rates were high (13.8 g to 44.6 g dry weight m^{-2} d^{-1}) and appeared to be greater than in the other Lake District lakes (Parker et al. 1999). This equated to an annual

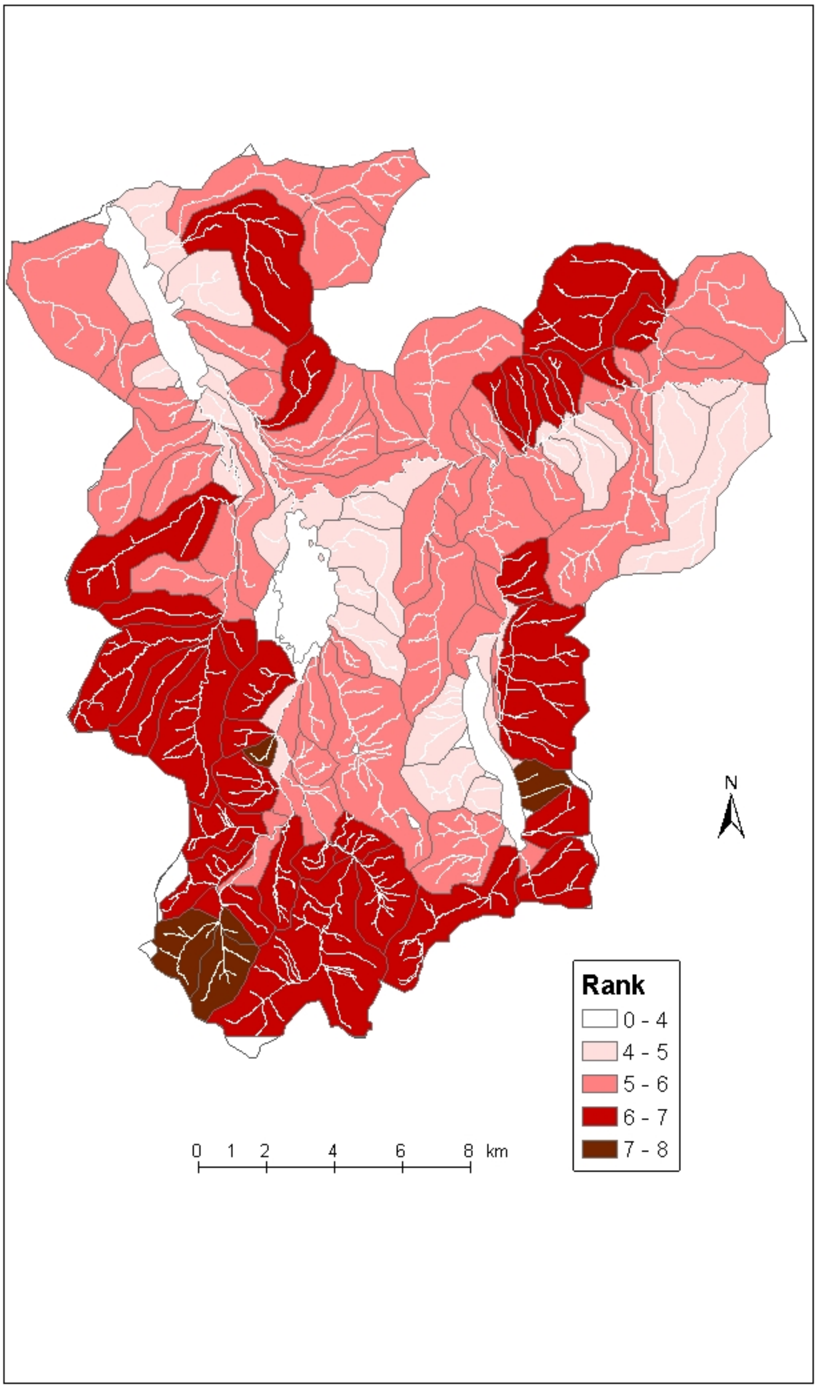


FIG. 19. Sediment supply risk ratings for the Bassenthwaite subcatchments (Orr et al. 2004). Rank 7–8 represents the highest risk.

gross sedimentation rate of 10.4 cm y^{-1}. This estimate is much higher than the measured accumulation rate of 0.66 cm y^{-1} (Cranwell et al. 1995) and 0.38 cm y^{-1} estimated from data in Bennion et al. (2000). The difference presumably reflects the extent of sediment resuspension within the lake. A sediment budget, calculated from observations between January and April

1999, indicated that a very small proportion of the sedimenting material in the lake (6 % dry mass) could be accounted for by the influx of new material in the inflow (Parker et al. 1999). This also suggests that much of the suspended material in the lake is derived from resuspension events. It has been concluded that much of the sediment present in Bassenthwaite Lake has been there for years, and is being continually resuspended by mixing events (Parker et al. 1999). Engineering works in the catchment during the latter part of the 19th century and in the 1970s, as well as land-use changes around the time of the second world war are all thought to have contributed to this sediment load. Modelling has shown that the area of the lake bed subjected to such mixing is dependent on wind speed and direction (Parker et al. 1999). Weak to moderate northerly winds generally disturb more of the bed than southerly winds of similar speeds. Though wind-induced resuspension can account for most of the suspended sediment in the lake, it must be noted that the influx of sediment in the inflows is highly episodic and that, following mass movements in the catchment, large quantities of new sediment could enter the lake (Hall et al. 2001).

The export of sediment from Bassenthwaite Lake is dependent on wind-induced resuspension. Once the sediment is resuspended, it can be flushed from the lake (Parker et al. 1999; Hall et al. 2001). It is believed that the slow pace of remediation in the lake is partially a result of the slow flushing of these sediments. Further research would be needed to quantify this sediment export rate.

Palaeolimnological data suggest that sediment accumulation rates have shown a progressive increase from approximately 0.05 g cm^{-2} y^{-1} in 1900 to 0.14 g cm^{-2} y^{-1} in 1940 (Bennion et al. 2000). The present day rate of approximately 0.1 g cm^{-2} y^{-1} is still higher than that experienced in the early 20th century. Bennion et al. did not offer an explanation for this pattern of change, but it presumably reflects changes in land-use and farming practices in the catchment. A historical study of past land-use patterns and practices would be needed to try and establish a precise cause for the changing rate of sedimentation. Given the largely inorganic nature of the sedimented material, it is unlikely to be a direct consequence of eutrophication. However, analysis of a deeper sediment core has indicated that, although organic material is a rather minor component of the sediment, there has been an increase in organic content since 1900 (Bennion et al. 2000). The possibility of increasing sedimentation in Bassenthwaite Lake, as a result of increasing phytoplankton crops, cannot therefore be completely dismissed.

PHYTOPLANKTON

Phytoplankton form the base of the food-chain in many lakes, especially larger ones, and are often one of the primary reasons for concern over water quality. They are also one of the measures of ecological quality used in the EC Water Framework Directive 2000/60/EC (European Commission 2000).

Phytoplankton biomass

The concentration of phytoplankton chlorophyll *a* is widely used as a convenient measure of phytoplankton biomass. The first seasonal time course of phytoplankton chlorophyll *a* for Bassenthwaite Lake appears to be that of Mubamba (1989) from 1987 who took monthly samples and recorded an annual maximum in October of 31.0 µg L^{-1}. This is comparable with the time-series for Bassenthwaite Lake between August 1990 and December 2003, shown in Fig. 20. During that time, the average concentration of phytoplankton chlorophyll *a* was 13.4 µg L^{-1}. Based on the boundaries recommended by the Organisation for Economic Cooperation and Development (OECD 1982) this classifies the lake as eutrophic. Between 1990 and 2003, the concentration of phytoplankton chlorophyll *a* varied between a minimum of 1.1 µg L^{-1} and a maximum of 93.9 µg L^{-1}. The year-to-year changes in maximum, minimum and average concentrations of phytoplankton chlorophyll *a* are shown in Fig. 21. There have been no long-term changes in these annual statistics between 1995 and 2003 (Maberly et al. 1994), but between 1991 and 2003 there has been a slight, but significant increase in the annual minimum concentration of chlorophyll *a* ($P = 0.03$).

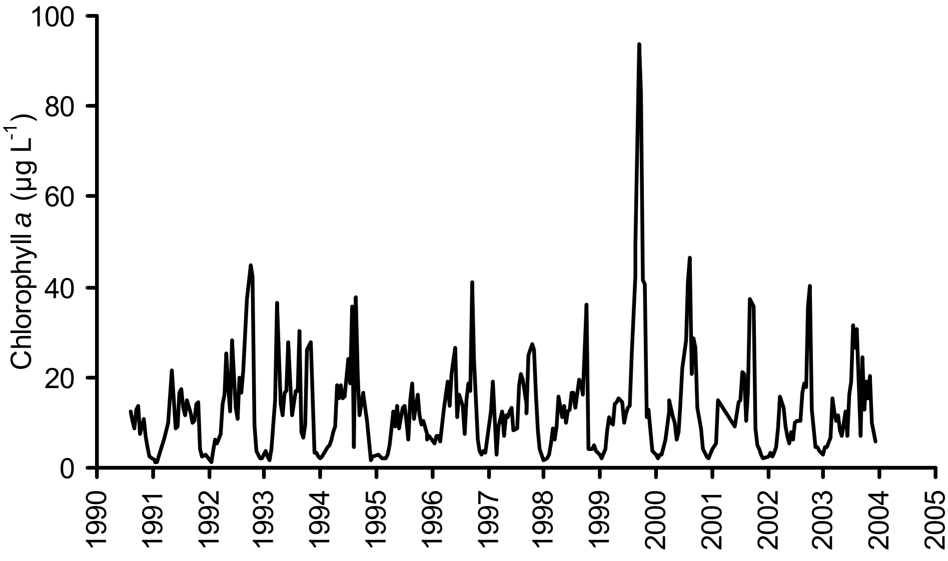


FIG. 20. Time-series of change in phytoplankton chlorophyll *a* in Bassenthwaite Lake (data from CEH).

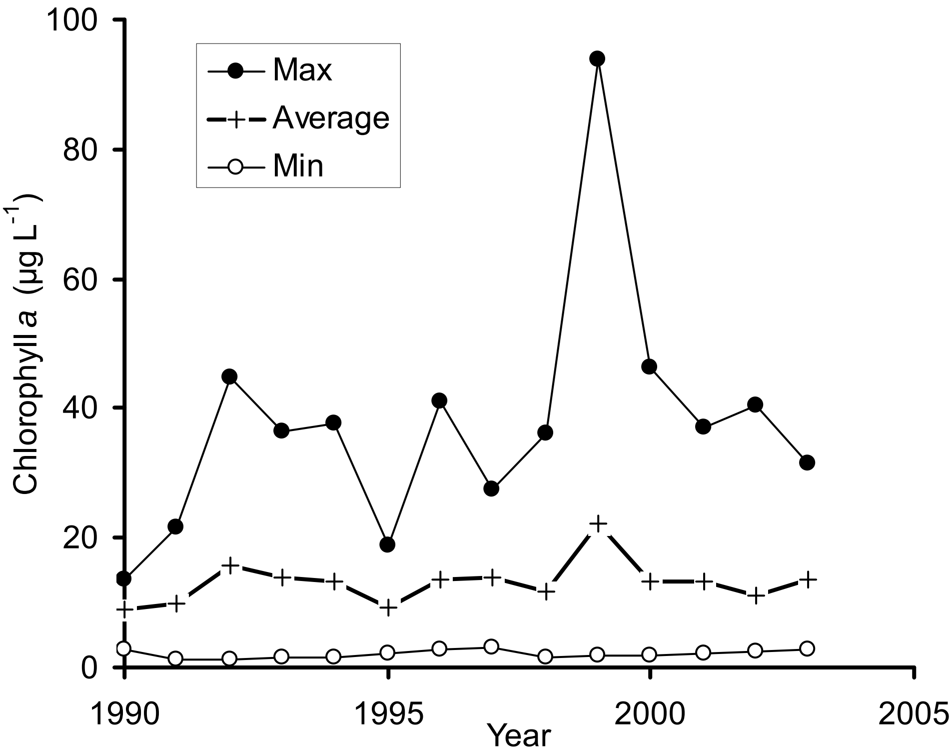


FIG. 21. Annual maximum, average (mean) and minimum concentration of phytoplankton chlorophyll *a* in Bassenthwaite Lake (data from CEH).

The seasonal pattern of phytoplankton is unlike that of the other major Cumbrian lakes as there is only a very weak spring bloom, the timing of which varies considerably from year to year. For example the spring maximum occurred on 24 May in 1995 and 5 February in 1997. On average (Fig. 22) the spring bloom occurs in the first week of April with a peak of 19 µg L^{-1}. This is followed by a slight decline, which does not constitute a ‘clear water phase’, and an increase to an annual maximum in late summer or early Autumn, usually between late August and early October, with average maxima of about 30 µg L^{-1}, but substantial populations can exist until late October.

There have been no substantial changes in the seasonality or magnitude of phytoplankton chlorophyll *a* in the time periods before and after P-stripping at the Keswick WwTW (Figs 20, 21 & 22). The average concentrations of chlorophyll *a* in the period before (1990–1994) and after (1995–2003) P-stripping were very similar, 12.8 µg L^{-1} and 13.6 µg L^{-1} respectively. This may result from the relatively modest reduction in P-load that the stripping has achieved and the likely internal recycling, but probably mainly reflects the large effect that physical factors have on the phytoplankton biomass in Bassenthwaite Lake. As mentioned above, the average retention time in the lake is only 19 days, but during high flows when a discharge of 100 m^3 s^{-1} is not uncommon, the retention time will

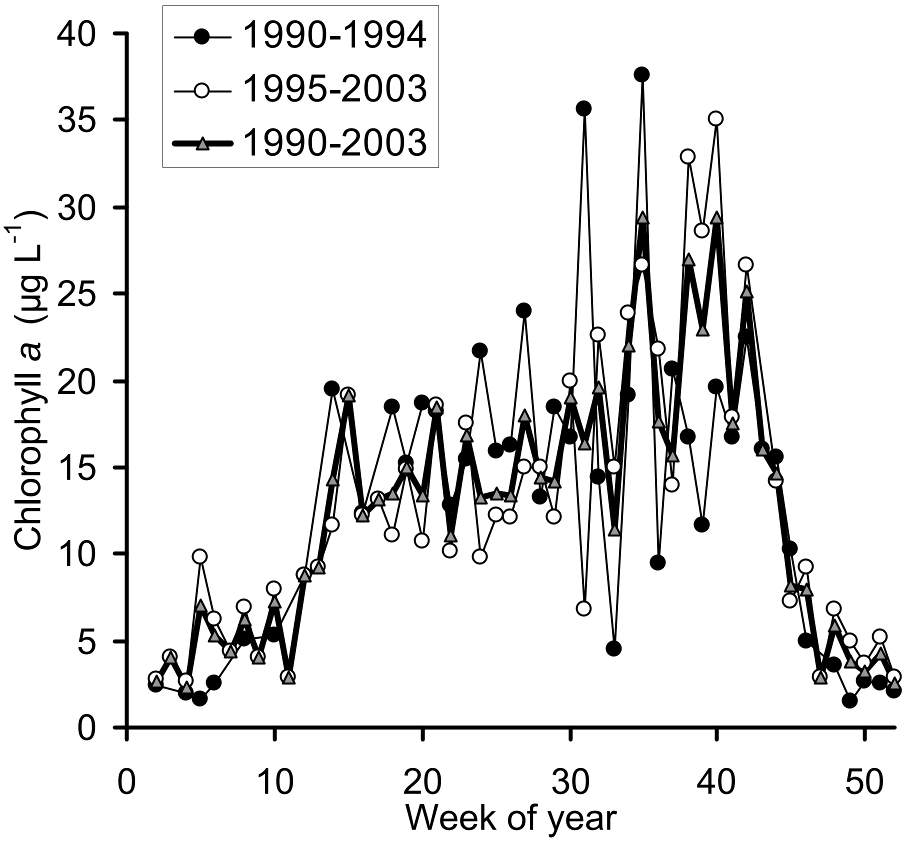


FIG. 22. Average seasonal pattern of phytoplankton chlorophyll *a* before (1990–1994) and after (1995–2003) P-stripping at the Keswick WwTW and for the whole period between 1990 and 2003. (Data from CEH).

only be 3 days. This demonstrates the potentially huge effect that hydraulic losses can have on the phytoplankton populations in the lake.

Phytoplankton species composition

The earliest records of the species composition of phytoplankton in Bassenthwaite Lake appear to be data of W. & G.S. West from 1909 noted by Pearsall (1921), to which Pearsall added observations of his own (Pearsall 1921, 1932). The most conspicuous taxa are noted but not quantified. Dominant groups included diatoms such as *Tabellaria* and *Asterionella*, desmids and the dinophyte *Ceratium hirundinella*. Pearsall (1921) also estimated the approximate percent contribution of different major groups of algae in this lake, and other major Lake District lakes, from samples collected in August 1920. Diatoms were clearly dominant (83 %), followed by Cyanobacteria (8.5 %), with small contributions from other groups. Although it is not easy to relate these two pieces of information to present day counts, the phytoplankton species composition at the turn of the 20th century and today appear to be broadly similar.

Data from the intervening years are presented in Atkinson et al. (1989) which include the data of Gorham et al. (1974) and records for 1987–1988 of Mubamba (1989). In addition, Kadiri & Reynolds (1993) present data from 'Lakes Tour' samples taken on four occasions in 1978 and 1984. In all these works the phytoplankton flora is similar to that recorded in the fortnightly sampling programme, the results of which are outlined below.

Between 6 August 1990, when CEH (then the Institute of Freshwater Ecology) first started monitoring Bassenthwaite Lake routinely, and December 2003, a total of 316 samples have been analysed and 326 taxa of phytoplankton have been recorded and enumerated. Of these, 82 have been recorded only once and a further 107 less than 10 times, and so are not an important component of the phytoplankton flora of the lake. These rare taxa may represent species washed in from the surrounding catchment, or taxa that only occasionally find an ecological niche in the lake. The 36 taxa that have been found more than 20 % of the time, and so may be considered typical of the lake, are listed in Table 5.

The seasonal patterns of cell density for the eight most frequent taxa are shown in Fig. 23. An unusual feature of these patterns is the lack of strong seasonality in many of the taxa. For example, in the South Basin of Windermere and other lakes in the Windermere catchment, *Asterionella formosa* has a marked seasonality producing a strong spring bloom, followed by a rapid reduction as stratification and silica depletion occur, and then a slow recovery in late summer and winter (Maberly et al. 1994). The lack of a strong seasonality in Bassenthwaite Lake probably results from the large variations in timing of peak biomass, as noted above for chlorophyll *a*, and the weaker effects of stratification. A number of taxa such as *Chlorella* sp., *Chrysochromulina parva*, *Rhodomonas* sp., *Fragilaria crotonensis*, *Crypotomonas ovata* and *Ochromonas* sp. show a uni-modal seasonal pattern with peak cell density in mid-summer (Fig. 23).

A relatively recent arrival to Bassenthwaite Lake is the centric diatom *Aulacoseira ambigua* (Grunow) Simonsen (Canter & Haworth 1991). This was first found in the autumn of 1990 and did not appear to be abundant in a sample taken in 1987 before the regular monitoring programme had begun (Canter & Haworth 1991) or earlier in 1978 or 1984 (Kadiri & Reynolds 1993). Before 1990, the related diatom, *A. subartica* was dominant and this species still exists in the lake, but at low cell densities. Canter & Howarth (1991) regarded the arrival of *A. ambigua* as reflecting the recent increased nutrient status in the lake, since this species is typically found in more nutrient rich lakes than *A. subarctica*. It remains to be seen whether any reduction in the nutrient status of Bassenthwaite Lake will result in a reversion to dominance by *A. subarctica* over *A. ambigua*. On a longer time-scale, *A. subarctica* is itself relatively recent: substantial records of this species were not found in sediments dated before 1950

Table 5. The most frequent 36 phytoplankton taxa recorded between 1990 and 2003. The table shows the rank, the number of occasions the taxon has been recorded and the percent of occasions recorded out of the 316 sampling dates (data from CEH).

Species	Rank	No. occasions	%
Chlorella BEIJERINCK 1890 sp.	1	302	95.6
Asterionella formosa HASSALL 1855	2	294	93.0
Aulacoseira ambigua (GRUNUW IN VAN HEURCK) SIMONSEN 1979	3	267	84.5
Chrysochromulina parva LACKEY 1939	4	231	73.1
Rhodomonas KARSTEN sp.	5	231	73.1
Fragilaria crotonensis KITTON 1869 (var. *crotonensis*)	6	195	61.7
Cryptomonas ovata EHRENBERG	7	184	58.2
Ochromonas WYSSOTSKI sp.	8	174	55.1
Pseudosphaerocystis WORONICHIN 1893 sp.	9	143	45.3
Kephyrion PASCHER 1911 sp.	10	136	43.0
Aulacoseira subarctica (O. MÜLLER) E.Y. HAWORTH 1988	11	135	42.7
Cryptomonas EHRENBERG 1838 spp. ‘medium’ (cf. *C. ovata* ?)	12	124	39.2
Tabellaria flocculosa var. *asterionelloides* (GRUNOW) KNUDSON 1952	13	119	37.7
Chlamydomonas EHRENBERG 1833 sp.	14	116	36.7
Stephanodiscus astraea (EHRENBERG) GRUNOW var. *minutula* (KÜTZING) GRUNOW	15	115	36.4
Koliella HINDÁK sp.	16	113	35.8
Monoraphidium contortum (THURET) KOMÁRKOVÁ-LEGNEROVÁ 1969	17	94	29.7
Cyclotella comensis GRUNOW IN VAN HEURCK 1882	18	92	29.1
Urosolenia eriensis (H.L. SMITH) ROUND & CRAWFORD in ROUND ET AL 1990	19	91	28.8
Dictyosphaerium pulchellum WOOD 1874	20	91	28.8
Cyclotella pseudostelligera HUSTEDT 1939	21	90	28.5
Nitzschia /Synedra (indeterminate)	22	89	28.2
Anabaena circinalis RABENHORST EX BORN. & FLAH. 1888 SENSU KOMÁREK & ETTL 1958	23	85	26.9
Cryptomonas EHRENBERG 1838 spp.(‘small’)	24	82	25.9
Rhodomonas minuta SKUJA & RUTTNER	25	77	24.4
Urosolenia ROUND & CRAWFORD	26	74	23.4
Peridinium EHRENBERG 1832 spp. (‘large’)	27	73	23.1
Cryptomonas erosa EHRENBERG	28	72	22.8
Monochrysis SKUJA sp.	29	70	22.2
Coenochloris fottii (HINDÁK) TSARENKO 1990	30	69	21.8
Staurastrum cingulum (W. & G.S. WEST) G.M. SMITH 1922	31	68	21.5
Monoraphidium KOMARKOVA-LEGENEROVA sp.	32	67	21.2
Anabaena flos-aquae BRÉBISSON EX BORN. & FLAH. 1888 SENSU KOMÁREK & ETTL 1958	33	66	20.9
Anabaena circinalis &/or *flos-aquae*	34	65	20.6
Paulschulzia SKUJA 1948 sp.	35	65	20.6
Spondylosium BREBISSON sp.	36	64	20.3

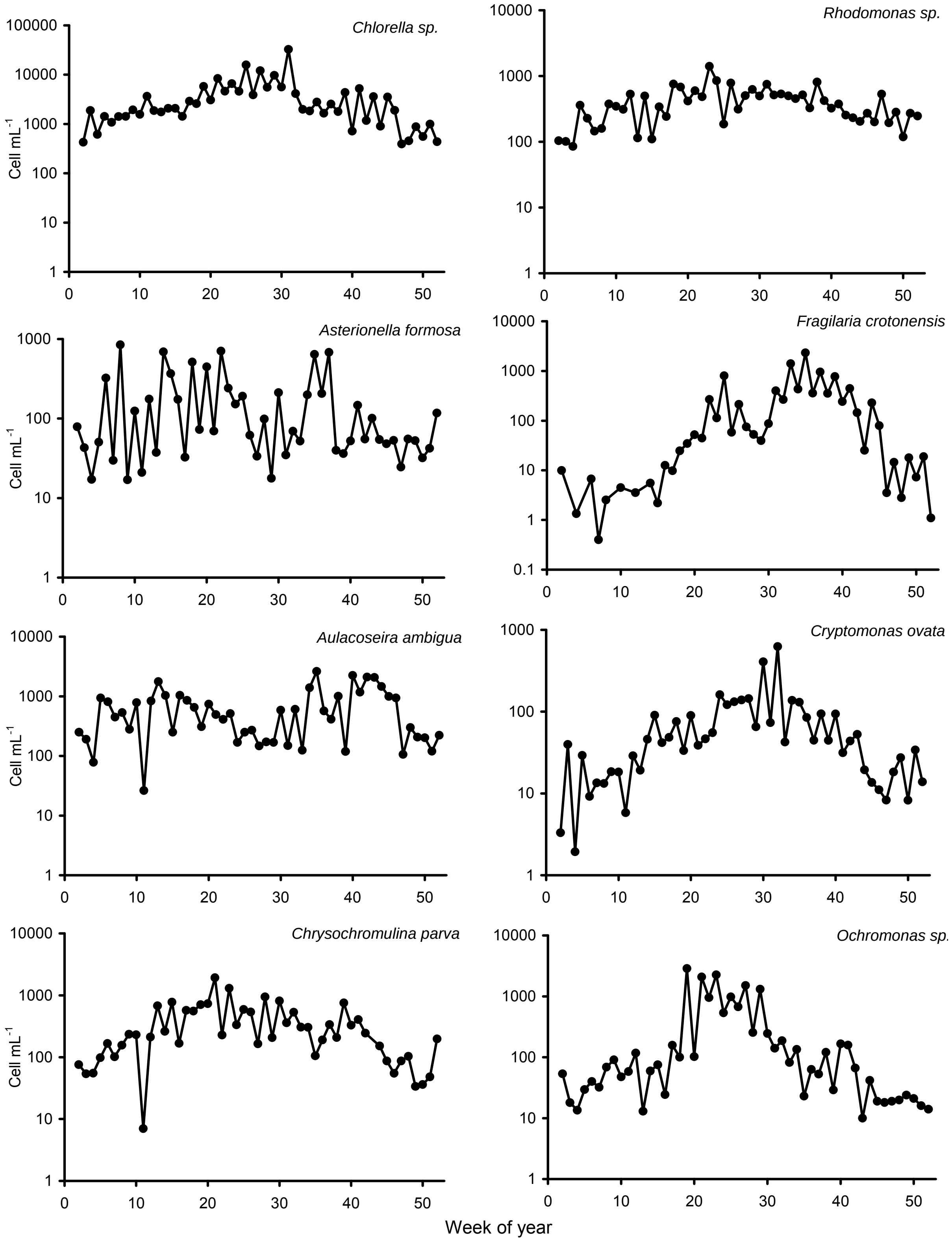


FIG. 23. Seasonal patterns of the eight most frequent taxa of phytoplankton in Bassenthwaite Lake between 1990 and 2003. Note that the y-axis is logarithmic and not the same scale for all panels (data from CEH).

Table 6. Dominant genera at time of algal maximum between 1998 and 2002. Buoyant genera are underlined. Based on analysis in Maberly (2003).

Year	Date of maximum	Dominant genera at time of maximum in approximate order of abundance
1996	25-Sept	*Tychonema, Fragilaria, <u>Anabaena</u>*
1997	29-Oct	*Aulacoseira, Chlorella, <u>Anabaena</u>*
1998	14-Oct	*Aulacoseira, Chlorella*
1999	16-Sept	*Dictyosphaerium, Aulacoseira*
2000	16_Aug	*<u>Aphanizomenon</u>, Gonatozygon, Aulacoseira*
2001	19-Sept	*Chlorella, Dictyosphaerium*
2002	16-Oct	*<u>Anabaena</u>, Aulacoseira*

(Bennion et al. 2000). The diatom flora before 1860 was similar to that in Wastwater although the diatom-inferred concentration of TP was much higher at about 20 µg L^{-1} in 1710 (Bennion et al. 2000).

On a number of occasions, most recently in autumn 2002, there has been local concern over ‘blooms’ of blue-green algae (Cyanobacteria) in certain parts of the lake, particularly near the outflow. In 2002, this did not result from a particularly high overall phytoplankton biomass, rather it resulted from the type of alga that was present. The autumn peak in 2002 was dominated by a Cyanobacterium *Anabaena solitaria* Klebahn (peak population at the centre of the lake of 108 mm filament length mL^{-1}), which is buoyant and so susceptible to movement by winds and waves that can cause it to accumulate in large numbers downwind. In previous years, the annual phytoplankton peak has tended not to be dominated by buoyant algae (Table 6) making local accumulations less likely. The Cyanobacteria tend to be slow growing and only come to dominate at the end of a period of favourable growth with low rates of loss. Bassenthwaite Lake is frequently highly flushed but the dry autumn of 2002 will have caused rates of loss to be low and may have favoured growth of Cyanobacteria.

Epilithon and other algae

Little work appears to have been carried out on the benthic algae of Bassenthwaite Lake, such as the epilithon (algae that grow on rocks). King et al. (2000) surveyed Bassenthwaite Lake and 16 other lakes in the English Lake District on three occasions between summer 1997 and autumn 1998, and identified and enumerated the species found. They found that two major environmental gradients, total phosphorus and factors related to ionic composition such as conductivity and concentration of dissolved inorganic carbon, controlled the epilithic species composition.

They did not, however, present any information specific to Bassenthwaite Lake. In a separate paper King et al. (2002) present information on epilithon species composition from samples collected at Bassenthwaite Lake in October 1998. Based on 12 replicate samples, they found that the epilithon composition was relatively uniform and comprised mainly diatoms (63 %). The dominant diatom species was *Achnanthes minutissima* (about 30 % of diatoms). Other important diatom groups were the Centrales (mainly *Aulacoseira*) and the genera *Fragilaria*, *Cymbella* and *Nitzschia*.

MACROPHYTES

Bassenthwaite Lake as an environment for macrophyte growth

Macrophytes comprise the non-microscopic algae and plants that are present submerged, floating and emergent around the margins of a freshwater body. Because of natural variation in the leaf form and height of a plant, as well as natural variation in water level, the distinction between submerged, floating and emergent is inevitably blurred and macrophytes exist in a cline from fully submerged to fully emergent. The lake level is very variable in Bassenthwaite Lake. This has potentially negative consequences for macrophyte growth as low water levels may leave submerged species exposed to air and high levels may restrict the amount of light received by plants growing at depth.

Another negative feature of Bassenthwaite Lake for macrophytes is the generally high light attenuation of the water. In the period 1995 to 2003, the average Secchi disc depth was 2.41 m. Using the rather rough relationship that the attenuation coefficient for downwelling light is approximately 1.44/Secchi depth (m), the estimated attenuation coefficient during the growing season is 0.6 m^{-1}. The suggested average depth limit for isoetid macrophytes is 16.3 % of surface light (Middelboe & Markager 1997) which, for an attenuation coefficient of 0.6 m^{-1}, would on average occur at a depth of 3.0 m. Equivalent depths for bryophytes (2.2 %), charophytes (5 %) and elodeids (12.9 %) (Middelboe & Markager 1997) are 6.4 m, 5.0 m and 3.4 m respectively.

The upper depth-limit for macrophyte colonisation will be controlled by wave exposure. Shores on the north-east and south-west central parts of the lake are relatively exposed and the upper, clear-water depths are likely to be unsuitable for elodeids and may be too disturbed even for robust species such as *Littorella uniflora*. Nevertheless, the feature of the lake that favours the growth of macrophytes is that it is generally shallow. The mean depth of the lake is only 5.3 m (Table 1) and based on the data in Ramsbottom (1976), the depth limits estimated above equate to roughly 50 % of the lake area being potentially colonisable.

Data on macrophytes in Bassenthwaite Lake

The abundance and diversity of macrophytes in Bassenthwaite Lake was one of the reasons for designating the lake a Site of Special Scientific Interest. The earliest data on macrophytes in Bassenthwaite Lake, as in many lakes in the English Lake District, derive from the surveys that Pearsall undertook between 1913 and 1920 by carrying out ‘soundings’ from a boat (e.g. Pearsall 1920; 1921). Pearsall recorded a moderately rich flora of submerged macrophytes (Table 7) although the only species of *Potamogeton* he recorded were *P. natans* and *P. pusillus*. Pearsall did not record *Elodea* in Bassenthwaite Lake: 1920 probably predates the arrival of this genus at the lake.

The next major survey of Bassenthwaite Lake appears to have been undertaken by Ralph Stokoe, an amateur botanist from Cockermouth with

Table 7. Macrophyte species presence in the different surveys on Bassenthwaite Lake, listed chronologically by year of survey. 1. Pearsall (1921) records this as *M. spictum*, presumably erroneously. 2. Noted in Stokoe’s list of littoral species.

Species	Pearsall (1920), 1913–1920	Stokoe (1980), 1979	Newbold & Palmer (unpublished), 1981	Bennion et al. (2000), 1996	Darwell & Taylor (1998), 1997	Darwell, Pitt & Tree (unpublished), 2003
Callitriche hamulata	+	+	+	+	+	+
Callitriche hermaphroditica		+	+	+	+	
Callitriche sp.					+	+
Crassula helmsii					+	+
Elatine hexandra		+	+	+	+	+
Elodea canadensis		+	+	+	+	
Elodea nuttallii		+	+	+	+	+
Fontinalis antipyretica	+	+	+	+		+
Isoetis lacustris	+	+	+	+	+	+
Juncus bulbosus	+	+			+	
Littorella uniflora	+	+	+	+	+	+
Lobelia dortmanna			+			
Luronium natans					+	+
Myriophyllum alterniflorum	+[1]	+	+	+	+	
Nitella opaca/flexilis	+	+	+	+	+	+
Nuphar lutea	+	+	+	+	+	+
Nymphaea alba	+		+	+	+	
Persicaria amphibia		+[2]	+	+	+	+
Potamogeton alpinus		+	+			
Potamogeton berchtoldii		+	+	+	+	
Potamogeton crispus		+	+	+	+	
Potamogeton gramineus		+	+	+	+	
Potamogeton natans	+	+	+	+	+	
Potamogeton perfoliatus		+	+	+	+	+
Potamogeton pusillus	+	+	+		+	
Ranunculus aquatilis			+		+	
Ranunculus peltatus	+	+	+	+		
Sparganium angustifolium		+	+		+	
Sparganium natans	+			+	+	+

Table 8. Submerged macrophytes in Bassenthwaite in 1979 based on Stokoe (1980). Frequency is the number of areas, out of 13 lake areas designated by Stokoe, in which the species was found. Abundance is scored as: A, abundant; P, plentiful; S, sparse.

	Distribution				Depth range (m)	
		Abundance				
Species	Frequency	A	P	S	Upper	Lower
Callitriche hamulata	7		6	1	0.7	3.0
Callitriche hermaphroditica	7	2		5	1.0	3.0
Elatine hexandra	2	1		1	0.5	1.0
Elodea canadensis	6		2	4	0.5	2.0
Elodea nuttallii	7	4	3		0.5	2.5
Equisetum fluviatile	4		3	1	0.5	0.5
Isoetis lacustris	11		10	1	0.5	2.0
Juncus bulbosus	4		4		1.0	1.0
Littorella uniflora	13	3	9	1	0.0	1.2
Myriophyllum alterniflorum	11	1	9	1	0.5	2.0
Nuphar lutea	2		1	1	1.0	1.0
Potamogeton alpinus	1		1		0.7	1.5
Potamogeton berchtoldii (2 forms)	2			2	0.5	1.5
Potamogeton crispus	3		2	1	1.0	1.5
Potamogeton gramineus	3	1	1	1	0.5	2.0
Potamogeton natans	1	1			0.5	0.7
Potamogeton perfoliatus	6		2	4	1.0	2.5
Potamogeton pusillus	1			1		
Ranunculus peltatus	10		7	3	0.5	2.5
Schoenoplectus lacustris	4	2	2		0.5	1.0
Sparganium angustifolium	1			1	0.5	0.5
Fontinalis antipyretica	4		4		0.5	2.2
Nitella opaca/flexilis	12	7	4	1	0.5	4.0

a passion for, and expertise on, macrophytes, who undertook a huge amount of survey work on the macrophytes in the lakes and tarns of Cumbria. Stokoe's records for Bassenthwaite Lake, compiled after his death (Stokoe 1983), are restricted to a species list. However, he also produced a report in January 1980 which presents these data in much more detail and includes information on spatial distribution (Stokoe 1980). The work was carried out between 1976 and 1979, but mainly in October of the latter year, when grapnel samples were taken during a circumnavigation by rowing-boat with records of depth for each sample in 13 different areas of the lake.

In his 1980 report, Stokoe recorded 23 species of submerged macrophytes (Table 7) plus two species of *Callitriche* (Table 8) which, while normally submerged were presumably found as terrestrial forms as they were included in Stokoe's list of 'littoral species'. He made special mention of the relatively rare *Callitriche hermaphroditica*, *Elatine hexandra* and *Juncus filiformis* in the flora and commented on the absence of *Lobelia dortmanna*, except for one specimen found in the drift, and *Nymphaea alba*. The two exotic species, *Elodea canadensis* and *E. nuttallii* were both recorded and the latter was noted as 'luxuriant'.

Stokoe (1980) included very useful information on the relative frequency and abundance of different species, as well as their depth range (Table 8). Information on distribution shows that species such as *Littorella uniflora*, which was found in all 13 areas of the lake, *Nitella flexilis/opaca*, *Isoetes lacustris*, *Ranunculus peltatus* and *E. nuttallii* were all frequent, or on occasions abundant.

Stokoe noted the presence of the main emergent species but did not survey these extensively so these are not dealt with in detail here, apart from the species list in Table 9. However, he did note that *Phragmites australis*, the common reed which is often a dominant fringing species, was infrequent and only found in two small colonies, possibly as a result of wave action.

The maximum depth of colonisation appeared to be about 3 m for the two species of *Callitriche* and extended down to 4 m for the charophyte *Nitella opaca/flexilis*. These depth limits are somewhat shallower than

Table 9. Emergent species recorded in Bassenthwaite Lake in 1979 (Stokoe 1980). Frequency is the number of areas, out of 13 lake areas, in which the species was found.

Species	Frequency	Species	Frequency
Agrostis stolonifera	6	*Lemna minor*	1
Alisma plantago	2	*Lysimachia vulgaris*	3
Callitriche platycarpa	1	*Lythrum salicaria*	5
Callitriche stagnalis	1	*Mentha aquatica*	4
Caltha palustris	4	*Montia fontana*	1
Carex nigra	3	*Myosotis caespitosa*	2
Carex vesicaria	3	*Myosotis scorpoides*	3
Eleocharis palustris	5	*Oenanthe crocata*	3
Galium palustre	4	*Phalaris arundinacea*	11
Glyceria fluitans	1	*Phragmites australis*	2
Hydrocotyle vulgaris	1	*Persicaria amphibia*	1
Iris pseudacorus	7	*Polygonum hydropiper*	1
Juncus acutiflorus	1	*Potentilla palustris*	1
Juncus articulatus	5	*Ranunculus flammula*	3
Juncus effusus	8	*Ranunculus repens*	3
Juncus filiformis	4	*Rorippa nasturtium-aquatica*	2
Juncus tenuis	1	*Scutellaria galericulata*	3

those roughly estimated from contemporary Secchi disc data (5.0 m for charophytes and 3.4 m for elodeids) but this could just be a result of the rough nature of these calculations rather than a change in water clarity. They are, however, quite similar to the depth limits noted by Pearsall as 3 m (Pearsall 1920) and 2.5 m (Pearsall 1921) suggesting that the light climate had remained essentially unaltered between 1920 and 1979.

Of particular value is the rough map that Stokoe produced (reproduced in Fig. 24) which shows the distribution of submerged macrophytes in the lake. This distribution can, potentially, be compared with more recent surveys to assess the extent of any change in macrophyte distribution over the last 20 years.

Shortly after the surveys of Stokoe, which finished in 1979, Chris Newbold and Margaret Palmer carried out a macrophyte survey in 1981 (Newbold & Palmer 1981, unpublished). The only information available is a distribution map showing transects around the lake and the distribution of species identified by number. This survey was produced either by grapnel from a boat, like that of Stokoe, or by diving. Although the area covered does not appear to be as comprehensive as the Stokoe survey it has the benefit of recording species at particular locations (Fig. 25). In contrast to Stokoe (1980), Newbold & Palmer did find *Lobelia dortmanna*, although only in Bowness Bay, and *Nymphaea alba*, although only in Broadness Bay. Stokoe (1980) did not find *N. alba*, but noted it was found in an older non-specified record, presumably that of Pearsall (1921).

Bennion et al. (2000) carried out a macrophyte survey on Bassenthwaite Lake in July 1996 by walking around the perimeter and using a grapnel to trawl perpendicular to the shore. Abundance was noted on a DAFOR scale (Dominant, Abundant, Frequent, Occasional, Rare). A large number of species were recorded and these were similar to previous surveys. The main differences were that *L. dortmanna*, *Potamogeton alpinus* and *P. pusillus* were not recorded. The former two species have not been found subsequently and so are presumably a real loss, while *P. pusillus*, which is hard to distinguish from *P. berchtoldii*, has been found in subsequent surveys.

A major, modern survey of the macrophyte flora of Bassenthwaite Lake was carried out by Angela Darwell in July and August 1997. She used snorkelling to map carefully 17 bays and score species abundance on a four-point scale – abundant, locally common, present and scarce (Darwell & Taylor 1998). The detailed bay-by-bay comparison of the 1979 and 1997 data in Darwell & Taylor (1998) is simplified here to a whole lake comparison. There appear to be a number of species that have been lost since earlier surveys. Notable among these is *Potamogeton alpinus* which was recorded by Pearsall (1921), Stokoe (1980) and Newbold & Palmer (1981). *Lobelia dortmanna* was recorded at one location by Newbold &

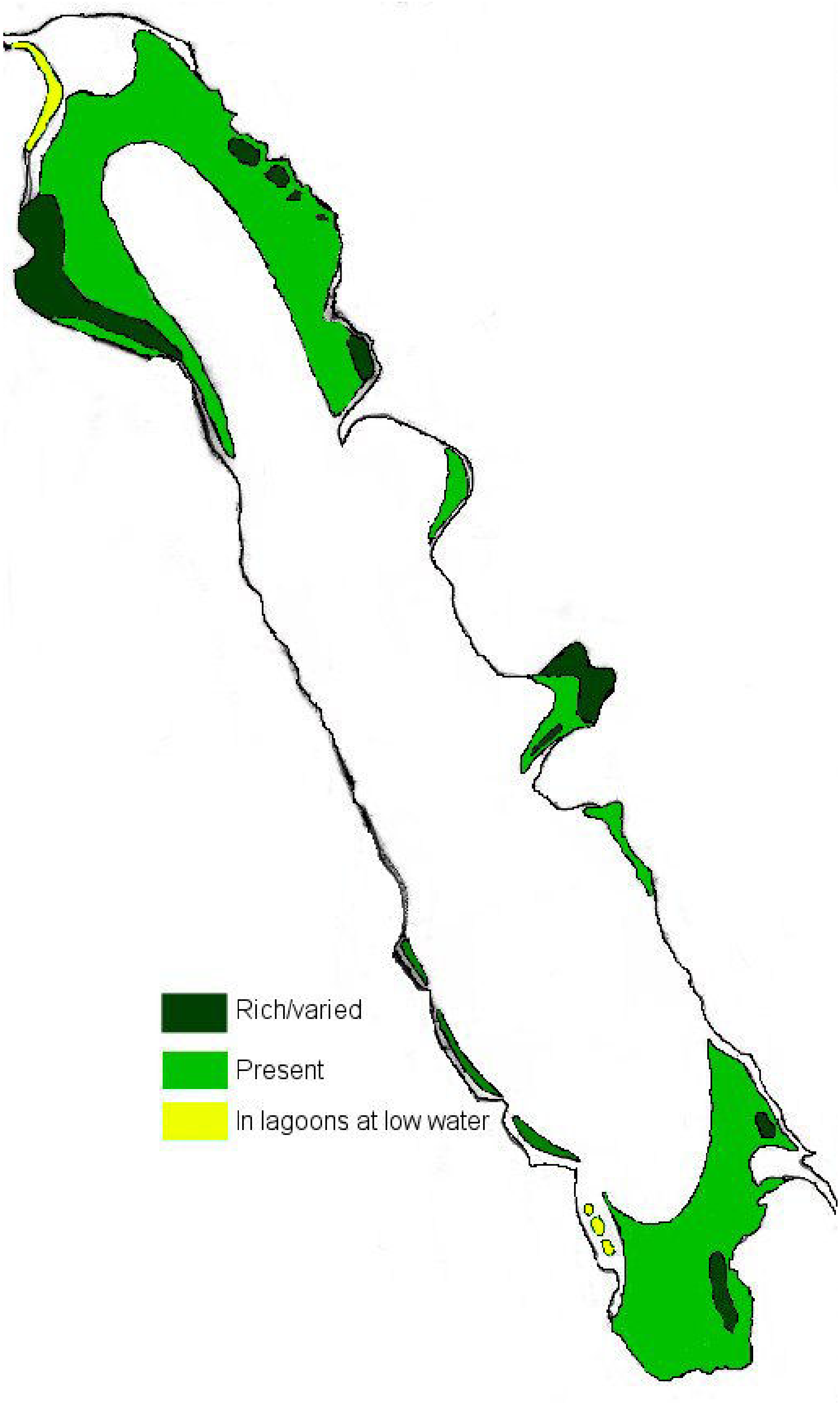


FIG. 24. Distribution map of submerged macrophytes in Bassenthwaite Lake in 1979. Redrawn from Stokoe (1980).

Palmer (unpublished), Bowness Bay, but a detailed survey of this bay by Darwell failed to find it in 1997.

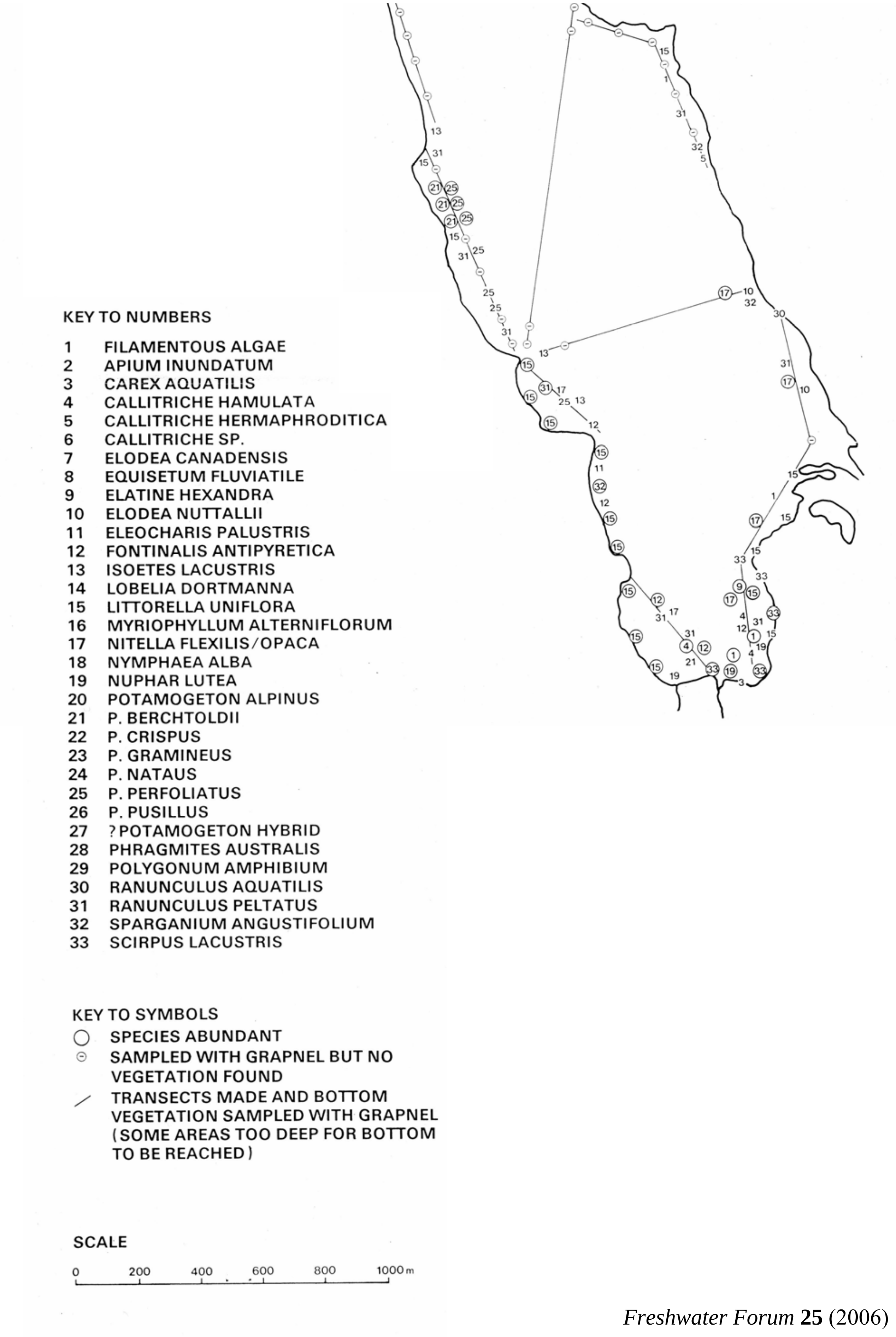
Two major new species, welcome and less-welcome, were recorded by Darwell & Taylor (1998; the third 'new' species mentioned by these authors, *Sparganium natans*, had been previously recorded by Pearsall, 1921, as *S. minimum*). The native *Luronium natans*, a nationally scarce plant listed in Schedule 8 of the Wildlife and Countryside Act (1981 plus amendments) had been found in Derwent Water in a survey in 1996 and this could have been the source of the population in Bassenthwaite, although it is equally likely that the species had been present but unnoticed in previous surveys. In 1997, *L. natans* was found in four bays in the lake, mainly at the southern end, but one site was recorded in a bay on the north-east shore. The other new record is the alien *Crassula helmsii*, the New Zealand pigmyweed. This is widespread in Derwent Water, immediately upstream from Bassenthwaite, where it was first recorded in 1992 (CEH, unpublished data). Darwell & Taylor (1998) recorded *C. helmsii* from three bays at the southern end of Bassenthwaite, two of which also contain *L. natans*. A subsequent grapnel survey of Bassenthwaite Lake in 2002 to check for the distribution of *C. helmsii* relocated it in one of the southern bays, failed to find it in two of the bays, and recorded it in a southern bay not surveyed by Darwell (P. Taylor, unpublished). Although there has not been a major spread of *C. helmsii* in Bassenthwaite between 1997 and 2002, it was, worryingly, found in a new location on the north-east shore (Broadness Bay). A series of transect surveys was carried out by R. Jerram on 25 and 26 June 2003 at Bridges Hole at the southern end of the lake (Barron 2004) to assess any interaction between *C. helmsii* and *L. natans*. This will be a useful baseline to assess whether *C. helmsii* outcompetes *L. natans*, albeit in a small area of the lake.

The most recent survey of the macrophytes in Bassenthwaite Lake was carried out by Angela Darwell, Jo-Anne Pitt and Angus Tree on 11 September 2003. This involved three 100-m shore surveys and two boat transects (Darwell, Pitt & Tree, unpublished). Not all species recorded previously were found in this rather rapid survey, but the presence of *C. helmsii* and *L. natans* was confirmed.

Concluding remarks

Bassenthwaite Lake has a diverse macrophyte flora which appears to have been relatively stable since the first recorded survey by Pearsall between 1913 and 1920. However, the lake has experienced a number of invasions by alien species. The arrival of two species of *Elodea* appears not to have had a major effect on the flora of the lake. Although *Potamogeton alpinus*

FIG. 25 *(this and facing page)*. Distribution map of submerged macrophytes in Bassenthwaite Lake in July 1981 (Newbold & Palmer, unpublished): northern bays, *this page*; southern bays, *facing page*.


KEY TO NUMBERS
1 FILAMENTOUS ALGAE
2 APIUM INUNDATUM
3 CAREX AQUATILIS
4 CALLITRICHE HAMULATA
5 CALLITRICHE HERMAPHRODITICA
6 CALLITRICHE SP.
7 ELODEA CANADENSIS
8 EQUISETUM FLUVIATILE
9 ELATINE HEXANDRA
10 ELODEA NUTTALLII
11 ELEOCHARIS PALUSTRIS
12 FONTINALIS ANTIPYRETICA
13 ISOETES LACUSTRIS
14 LOBELIA DORTMANNA
15 LITTORELLA UNIFLORA
16 MYRIOPHYLLUM ALTERNIFLORUM
17 NITELLA FLEXILIS/OPACA
18 NYMPHAEA ALBA
19 NUPHAR LUTEA
20 POTAMOGETON ALPINUS
21 P. BERCHTOLDII
22 P. CRISPUS
23 P. GRAMINEUS
24 P. NATAUS
25 P. PERFOLIATUS
26 P. PUSILLUS
27 ?POTAMOGETON HYBRID
28 PHRAGMITES AUSTRALIS
29 POLYGONUM AMPHIBIUM
30 RANUNCULUS AQUATILIS
31 RANUNCULUS PELTATUS
32 SPARGANIUM ANGUSTIFOLIUM
33 SCIRPUS LACUSTRIS
KEY TO SYMBOLS
SPECIES ABUNDANT
SAMPLED WITH GRAPNEL BUT NO VEGETATION FOUND
TRANSECTS MADE AND BOTTOM VEGETATION SAMPLED WITH GRAPNEL (SOME AREAS TOO DEEP FOR BOTTOM TO BE REACHED)
SCALE
0 200 400 600 800 1000 m

and *Lobelia dortmanna* have both disappeared this probably does not result from the arrival of these *Elodea* species. More recently another alien, *Crassula helmsii*, has appeared, probably from Derwent Water where it is now very abundant. So far this species has not spread extensively in Bassenthwaite Lake but this should be monitored as there is a possibility that it could outcompete the nationally scarce, recently recorded, but presumably always present *Luronium natans*.

The depth limit of macrophytes in Bassenthwaite Lake appears not to have changed substantially since the surveys of Pearsall, indicating that the light climate has not deteriorated over that time. What is less clear is whether there has been a reduction in the abundance of macrophytes. Taken on face value, the area of macrophytes noted by Stokoe in 1979 (Stokoe 1980) is less than noted by Newbold & Palmer (unpublished) and Darwell & Taylor (1998) but this is complicated by the different methods used. If it can be established that the area colonised by macrophytes has declined this could be a possible cause of a greater concentration of suspended sediment in the lake because macrophytes are known to stabilise sediment (e.g. James et al. 2004). This is speculation but nevertheless could be tested by paleolimnological methods. A survey of macrophyte spatial distribution, perhaps achieved most effectively using modern hydroacoustic techniques in addition to traditional techniques, would provide a useful baseline for future studies.

ZOOPLANKTON

The zooplankton represent an important link in the aquatic food web in that many of these species feed on phytoplankton and are themselves consumed by planktivorous fish such as the vendace. Microscopic crustaceans are conspicuous members of the zooplankton, though other multicellular organisms such as rotifers are often abundant.

There is a paucity of information regarding the rotifer community of Bassenthwaite Lake. Data from Mubamba (1989) show the presence of six genera in the lake: *Asplanchna*, *Keratella*, *Kellicottia*, *Filinia*, *Hexartha* and *Trichocerca*. There are, however, more data describing the crustacean zooplankton community. A series of surface plankton samples collected between 1921 and 1922 provide early data on the open-water crustacean zooplankton community of Bassenthwaite Lake (Gurney 1923). In these early surveys a total of seven species were found to be abundant in the lake, and these were considered a mixture of eurythermal and warm water species. Bassenthwaite Lake was found to be unique among the Lake District lakes in that it lacked any species of *Daphnia* or *Bosmina*. Other

Table 10. Species of crustacean zooplankton recorded in Bassenthwaite Lake by Gurney (1923), Smyly (1968) and Mubamba (1989).

Species	Gurney (1923), 1921–1922	Smyly (1968), 1961–1962	Mubamba (1989), 1986–1987
Cladocera:			
Daphnia hyalina		+	+
Bosmina obtusirostris		+	+
Sida crystallina	+		
Diaphanosoma brachyurum	+	+	+
Simocephalus vetulus		+	
Ilyocryptus sordidus		+	
Alona affinis		+	
Chydorus sphaericus		+	
Polyphemus pediculus	+		+
Bythotrephes longimanus	+	+	
Leptodora kindtii	+	+	+
Copepoda:			
Eudiaptomus gracilis	+	+	+
Mesocyclops leuckarti	+	+	+
Cyclops albidus		+	
Cyclops fimbriatus		+	
Cyclops viridis		+	
Canthocamptus staphylinus		+	
Ostracoda:			
Cypria opthalmica		+	

species of cladocerans were present, notably the herbivorous *Sida crystallina* and *Diaphanosoma brachyurum*, and the predatory *Polyphemus pediculus*, *Bythotrephes longimanus* and *Leptodora kindtii*. Only two species of copepod were found: *Eudiaptomus gracilis* and *Mesocyclops leuckarti*. During 1961 and 1962, Smyly (1968) collected further samples from the lake. The species list so produced was more extensive than that presented by Gurney (1923), with two new species found in the open water zone of the lake: *Daphnia hyalina* and *Bosmina coregoni* var. *obtusirostris*. Although this might be interpreted as evidence of colonisation by these species it is likely that this simply reflects the more extensive sampling programme employed in the second study. With the addition of these species it became apparent that the open-water crustacean zooplankton community of Bassenthwaite Lake was very similar to that in Derwent Water. By collecting water immediately above the lake bed Smyly (1968) also recorded a number of deep water species (*Cyclops albidus*, *C. fimbriatus*, *C. viridis*, *Canthocamptus staphylinus*, plus *Cypria opthalmica* and other members of the Ostracoda). A later survey by Mubamba (1989) confirmed the same dominant open-water species as the previous surveys. The species recorded in these three surveys are presented in Table 10.

More recent crustacean zooplankton samples were collected during CEH's Lake's Tour programme (Hall et al. 1992; Hall et al. 1996; Parker et al. 2001). In each of 1991, 1995 and 2000 four zooplankton samples were collected. The species recorded in these samples were essentially the same as those found in the earlier surveys. Given the temporal coverage of all of these surveys (four samples per year), it would seem likely that the true species list for the lake is somewhat larger than that presented above. We might expect that species with rather short life-cycles would have been missed at such a sampling resolution. For this reason it is also inappropriate to use these results to draw conclusions regarding the seasonal dynamics of the crustacean zooplankton community.

The overall biomass of crustacean zooplankton was found to be low in the Lakes Tour samples and so, given the concern over water quality in Bassenthwaite Lake, the decision was made to monitor more closely the crustacean zooplankton community. Therefore, since February 2001, fortnightly zooplankton samples have been collected from the lake. Some of these samples, which are currently archived at CEH Lancaster, have been counted. These data, which originate from the 2004 fortnightly sampling of Bassenthwaite Lake, give a clearer picture of the seasonal dynamics of the zooplankton community (Fig. 26a). The data show that there are dramatic seasonal variations in the abundance of zooplankton: the highest population densities being found during late April to late May and August to September. During 2004 the spring peak was dominated by *Daphnia hyalina* and *Bosmina obtusirostris*. This spring peak is rather typical of a number of Lake District lakes (CEH, unpublished data). During summer there was a second peak in *Daphnia* abundance, but the dominant species at this time was *Eudiaptomus gracilis*. Though much smaller, a peak in the abundance of *Mesocyclops leuckarti* was also evident at this time.

Although detailed species count data are only currently available for 2004, there are less quantitative, interannual data based on counts of zooplankton on the filter paper used to filter for phytoplankton chlorophyll *a*. These data show that there is a large variation in zooplankton abundance inter-annually and that Cladocera outnumber copepods by 2:1 on average (Fig. 26b). The zooplankton density assessed by this method is lower than that in Esthwaite Water, but about the same as in the South Basin of Windermere (data not shown).

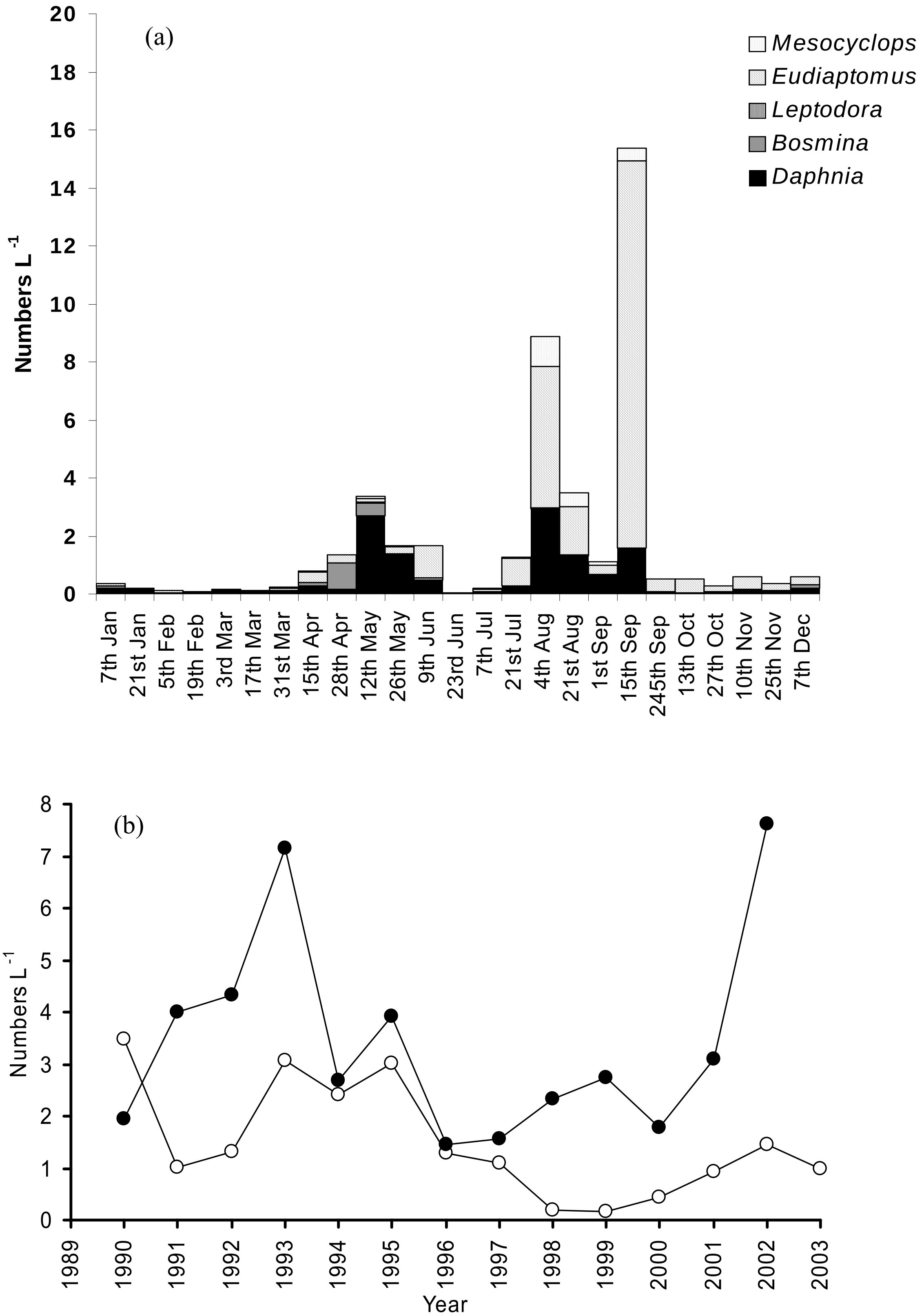


FIG. 26. a) Seasonal variations in zooplankton community composition during 2004 and b) year-to-year changes in average numbers of cladocerans (closed circles) and copepods (open circles) in Bassenthwaite Lake, based on samples between 1990 and 2003 (data from CEH).

BENTHIC INVERTEBRATES

There is relatively little quantitative information on the bottom living invertebrates in the shallow water areas of Bassenthwaite Lake (Talling 1999). At present, our knowledge of the shallow water community is largely based upon data summarised in Macan (1970), including an extensive survey in 1966 and 1967, and a more limited survey by Mubamba (1989). The community was found to be typical of the more productive lakes in the English Lake District: dominated by oligochaete worms, chironomid larvae, the crustacean *Gammarus* and pea mussels (*Pisidium* spp.). Notable species of freshwater shrimp (*Crangonyx pseudogracilis*, a north American immigrant), water louse (*Asellus meridianus*) and mayfly (*Ephemera danica*) have all been found in the lake (Atkinson et al. 1989). Conversely, some widespread species, such as *Ecdyonurus dispar* and *Heptagemia lateralis*, are not present. A species list for Bassenthwaite Lake is presented in Table 11.

In general, the species richness of the community and the overall abundance of invertebrates was found by (Mubamba 1989) to be higher in shallow sediments than in deeper sediments. In deeper sediments a number of oligochaete species have been found (Reynoldson 1990). There appears to be a degree of seasonal variation in the abundance of benthic invertebrates, the highest numbers being recorded in autumn or winter. The reason for this is unclear (Mubamba 1989).

FISH POPULATIONS

Introduction

Prior to the mid 1980s, very little was known about the fish fauna of Bassenthwaite Lake other than that it contained the UK's rarest freshwater fish, the vendace (*Coregonus albula*). A brief review of this extremely limited and largely unscientific historical background was given by Atkinson et al. (1989) and will not be repeated here because only one relevant study from this period, in the form of the unpublished undergraduate thesis of Broughton (1972) which will be considered later, has subsequently been discovered.

Scientific understanding of the fish of Bassenthwaite Lake was improved in the late 1980s Mubamba (1989). Although this study focussed on the vendace population of Bassenthwaite Lake and that of nearby Derwent Water, in 1986 and 1987, it also provided some information on other fish species. The findings of Mubamba (1989) relevant to Bassenthwaite Lake were also reviewed by Atkinson et al. (1989) and so

Table 11. Species of benthic macroinvertebrates recorded in Bassenthwaite Lake by Brinkhurst (1964), Macan (1970), Mubamba (1989) and Reynoldson (1990).

Species	Brinkhurst (1964), 1960	Macan (1970), 1966-1967	Mubamba (1989), 1986-1987	Reynoldson (1990)
Snails (Gastropoda):				
Ancylus fluviatilis		+		
Limnaea pereger		+		
Planorbis contortus		+		
Physa fontinalis		+		
Planorbis albus		+		
Valvata piscinalis		+		
Limnaea palustris		+		
Planorbis spirorbis		+		
Limnaea truncatula		+		
Mussels (*Pisidium*)			+	
Water bugs (Corixidae):				
Sigara dorsalis		+		
Sigara scotti		+		
Corixa punctata		+		
Micronecta poweri		+		
Worms (Oligochaeta):				
Limnodrilus hoffmeisteri	+			+
Limnodrilus claparedeanus				+
Limnodrilus profundicola				+
Peloscolex ferox	+			
Aulodrilus pluriseta	+			
'Tubificidae'			+	
Tubifex tubifex				+
Tubifex ignotus				+
Ilyodrilus templetoni				+
Crustaceans (Crustacea):				
Asellus meridianus		+		
Gammarus pulex		+		
Crangonyx pseudogracilis		+		
Flatworms (Platyhelminthes):				
Polycelis nigra		+		
Polycelis tenuis		+		
Dugesia lugubris		+		
Dendrocoelum lacteum		+		
Leeches (Hirudinea):				
Helobdella stagnalis		+		
Erpobdella octoculata		+		
Glossiphonia complanata		+		
Glossiphonia heteroclita		+		
Caddis flies (Trichoptera):				
Polycentropus flavomaculatus		+		
Polycentropus irroratus		+		
Agapetus fuscipes		+		
Cyrnus trimaculatus		+		
Cyrnus flavidus		+		
Tinodes waeneri		+		
Mayflies (Ephemeroptera):				
Centroptilum luteolum		+		
Ephemera danica		+		
Caenis moesta		+		
Stoneflies (Plecoptera):				
Nemoura avicularis		+		
Chloroperla torrentium		+		
Diura bicaudata		+		
Leuctra inermis		+		
Leuctra hippopus		+		
Beetles (Coleoptera):		+		
Deronectes assimilis		+		
Oulimnius tuberculatus		+		
Chironomids (Chironomidae)			+	

will not be repeated here, although they will be referred to where appropriate in the context of long-term changes in the fish populations.

In contrast to the limited research at Bassenthwaite Lake up to the late 1980s, investigations have been almost continuous since 1990 under a series of projects dealing with various aspects of the fundamental and applied ecology of the fish populations. This work, most of which has been undertaken as commissioned studies for customers (including, in alphabetical order, English Nature, Environment Agency, Scottish Natural Heritage and United Utilities) has resulted in 15 key reports and 18 scientific publications, all of which are given here in the reference list. As a result of its rarity and susceptibility to adverse environmental conditions, much of this and subsequent research has focussed specifically on the local ecology and conservation management of the vendace. However, information has also been gathered and interpreted on the lake's other fish species and a consistent attempt made to understand the ecology of the overall fish community.

There is no single correct way to review this considerable amount of scientific information on the fish of Bassenthwaite Lake. However, given that much of this work has been focussed on the vendace, this species will be predominantly used to frame the present review. In doing so, this review will draw heavily on the recent synthesis of the conservation ecology of vendace in Bassenthwaite Lake and Derwent Water by Winfield et al. (2004b). In a wider context, it is notable that the vendace also consistently features highly in discussions of water quality issues at Bassenthwaite Lake. Moreover, as this coregonid has the most exacting environmental requirements of all of the fish species occurring in the lake, it can be regarded as a sentinel species: if environmental conditions allow it to flourish, then they will also be suitable for the other components of the fish community.

Species list

Scientific sampling at Bassenthwaite Lake has recorded a total of 10 fish species: Atlantic salmon (*Salmo salar*), brown (including sea) trout (*Salmo trutta*), dace (*Leuciscus leuciscus*), eel (*Anguilla anguilla*), minnow (*Phoxinus phoxinus*), perch (*Perca fluviatilis*), pike (*Esox lucius*), roach (*Rutilus rutilus*), ruffe (*Gymnocephalus cernuus*) and vendace (*Coregonus albula*) (Winfield et al. 2004b). However, this list includes three species which are non-native and have been recorded only in recent years. These comprise roach, first recorded in 1986 (Mubamba 1989), ruffe, first recorded in 1991 (Winfield et al. 1994b; Winfield et al. 1996a; Winfield et al. 1996b), and dace, first recorded in 1996 (Winfield et al. 2002a). These species introductions, and their consequences for fish community composition and the vendace, will be considered further below.

A brief introduction to the vendace

The vendace is a medium-sized and typically lacustrine coregonid occurring principally in the northern areas of Europe, although it has also been introduced to lakes further south in mainland Europe for fisheries purposes (Lelek 1987). Like other members of the genus *Coregonus*, it prefers relatively high concentrations of dissolved oxygen (Dembinski 1971), low water temperatures (Hamrin 1986), and spawning areas with no or limited fine sediments (Wilkonska & Zuromska 1982). In addition, the vendace feeds extensively on zooplankton such as *Daphnia* spp. throughout its life cycle and reduced availability of such prey may lead to marked population declines (Auvinen 1988).

Within the UK, vendace populations have historically been recorded from four water bodies (Maitland 1966b), although the two populations of Castle Loch (N 54° 50’, W 4° 40’) and Mill Loch (N 55° 8’, W 3° 27’) near Lochmaben in south-west Scotland are now believed to be extinct (Maitland & Lyle 1990). The two remaining populations occur in Bassenthwaite Lake and Derwent Water (N 54° 34’, W 3°8’), which are connected by the River Derwent. Although Bassenthwaite Lake and Derwent Water are physically similar (Talling 1999) and lie near to each other in the same catchment, they differ significantly with respect to trophic status (considered elsewhere in this review) and fishery activities. At Bassenthwaite Lake, fishing is largely limited to non-game species, particularly pike, but at Derwent Water most activity is focussed on game angling for brown trout. Genetic investigations have shown that the vendace populations of these two lakes are not significantly differentiated, possibly as a result of gene flow from occasional accidental migrants or the relative youth of the lakes (Beaumont et al. 1995).

Due to the limited national distribution of the vendace and the loss of the two populations described above, this species is the UK’s rarest freshwater fish species and is consequently protected under nature conservation legislation in the form of the Wildlife and Countryside Act, 1981. It is also a UK Biodiversity Action Plan species. There are thus no current commercial or recreational fisheries for this species, and indeed the only record of this fish being exploited in the UK was for the now extinct populations of Lochmaben (Maitland 1966a).

More extensive reviews of vendace biology in mainland Europe are given in Winfield et al. (1994a). Unless specified otherwise, the remainder of this review relates specifically to the vendace of Bassenthwaite Lake.

Historical and current status of vendace

In the absence of historical or current fisheries, this otherwise common source of status information is unavailable for the vendace population of

Bassenthwaite Lake. No information at all is available for the pre-1960s period and only three brief studies were subsequently made prior to the 1990s. Maitland (1966b) obtained a sample in the mid 1960s, as did Broughton (1972) in the early 1970s and Mubamba (1989) in the mid-1980s. Since the early 1990s, the vendace population has been studied intensively, including an annual monitoring programme involving a combination of hydroacoustics and gill netting running from 1996 (Winfield et al. 1997) to the present (Winfield et al. 2004a). The estimated population sizes of fish of the size of post-juvenile vendace in the deep-water areas of Bassenthwaite Lake from 1995 to 2003 are shown in Fig. 27, which is taken from Winfield et al. (2004a) where a full statistical assessment is presented. Although these numbers are now compromised to some extent by the introduced species which cannot be differentiated from vendace by hydroacoustic techniques, as discussed in detail in Winfield et al. (2004a), even with the introduction of this uncertainty the hydroacoustic data still indicate a significant fall in absolute vendace abundance in recent years.

Based on samples collected in 1965, Maitland (1966b) described the vendace population as 'thriving', i.e. of good status. Limited data presented in Broughton (1972) from sampling in 1972 also indicated a good population status, but by 1987 the samplings and examinations carried out by Mubamba (1989) showed that the status had declined to poor as a result of inconsistent recruitment. This situation persisted into the early 1990s (Winfield et al. 1994b; Winfield et al. 1996b), after which it declined even further with continued inconsistent recruitment and consequently reduced population abundance (Winfield et al. 2004a).

It is notable that no vendace have been taken in the survey gillnets of the monitoring programme since 2000, although they have been detected in small numbers by other sampling activities (Winfield & Fletcher 2002). Consequently, the current status of vendace in Bassenthwaite Lake is considered to be extremely poor (Winfield et al. 2004a).

Reviews of generic threats facing the conservation of fish communities in the UK and European studies on the vendace, by Winfield (1992) and Winfield et al. (1994a) respectively, were used to frame the approach of the 1990s and subsequent investigations, augmented by other studies of local environmental conditions and unpublished data held by CEH. This resulted in the initial identification of four broad threats, i.e. eutrophication, sedimentation, species introductions and climate change, which have subsequently been assessed by a combination of field and laboratory studies. These are discussed further in the section on ecological pressures.

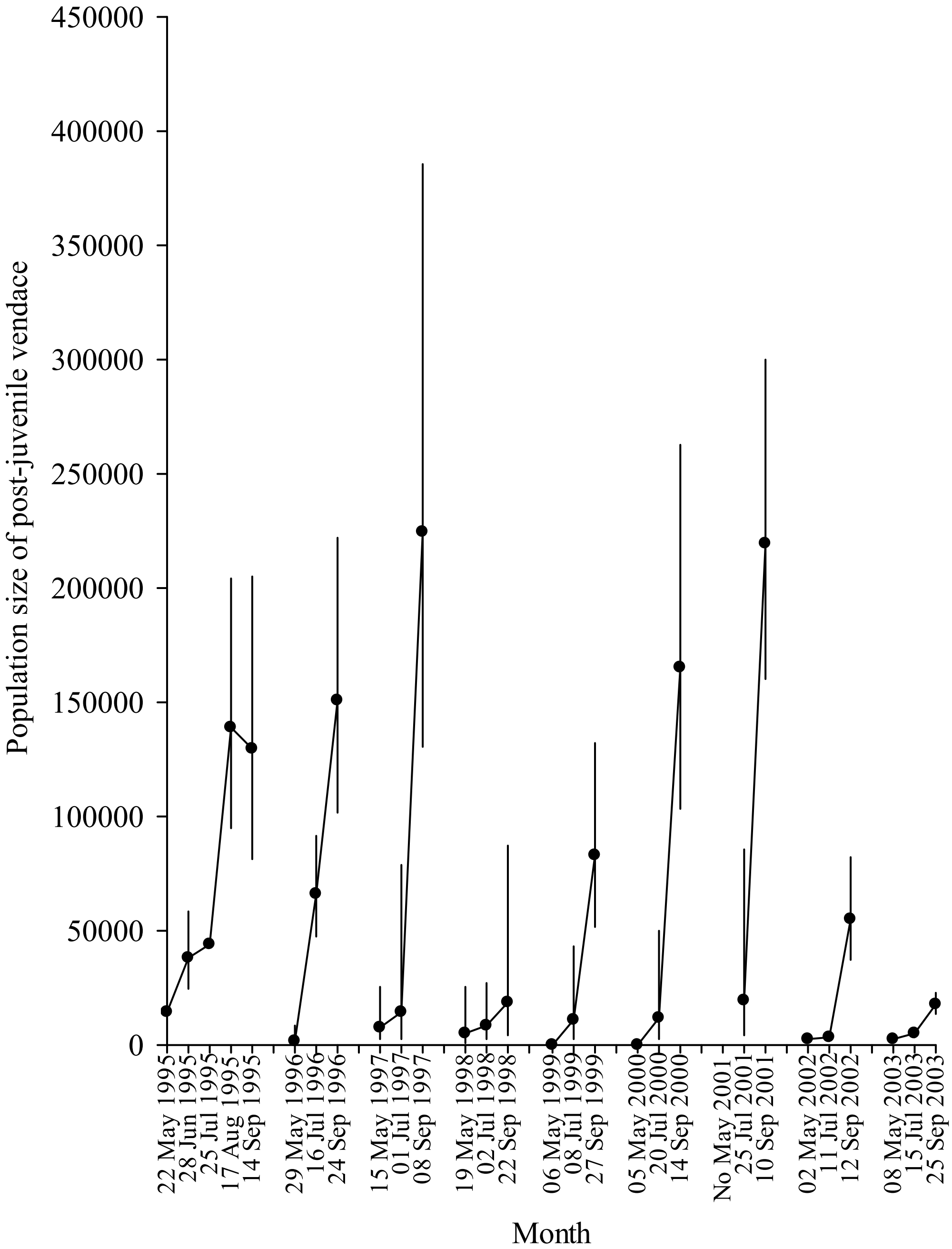


FIG. 27. Estimated population sizes (geometric means with 95 % confidence limits) of putative post-juvenile vendace during 28 echo-sounding surveys in Bassenthwaite Lake from 1995 to 2003. Note that only one transect was carried out in July 1995 due to dangerous weather conditions and so confidence limits could not be calculated, no survey was possible in May 2001 due to access restrictions because of foot-and-mouth disease, and calculations assume that a medium size class (100 mm to 249 mm) of fish was largely vendace. However, netting exercises have demonstrated that this assumption is now invalid because of recent species introductions. Reproduced from Winfield et al. (2004a).

Fish communities

Initial full assessments of the fish community of Bassenthwaite Lake, including both deep-water and shallow-water sampling locations, were undertaken in the early 1990s as described by Winfield et al. (1994b). From the mid-1990s, more extensive sampling involving hydroacoustics and gill netting has been continued to monitor the fish communities as described by Winfield et al. (1998a), Winfield et al. (2002a) and Winfield et al. (2004a). In addition to confirming the establishment of the roach population discovered in 1986 by Mubamba (1989), this work also produced the first records of ruffe and dace in Bassenthwaite Lake in 1991 and 1996 respectively (Winfield et al. 2002a). Species composition by numbers of the fish community at representative shallow-water and deep-water sites during this period are shown in Fig. 28. The shallow-water fish community began the period dominated by the introduced roach population, which subsequently decreased in relative importance as that of the introduced ruffe and native perch populations increased. Community composition was also highly variable at the deep-water site, with the most obvious and alarming change being the relative demise of the vendace population to the extent that it has not been detected by the monitoring programme since 2000. It is also notable that the introduced ruffe population is important in the deep-water fish community, although the introduced dace population remains a minor component at both sites.

Concluding remarks

The fish community of Bassenthwaite Lake is currently in a highly volatile state as a result of species introductions and other environmental problems. In fish conservation terms, impacts on the vendace population are by far the most important issue to be tackled in the management of this lake, although the importance of the lake as a migratory route for Atlantic salmon and brown (including sea) trout should not be neglected.

The degrees of threats to the conservation of vendace in Bassenthwaite Lake and Derwent Water diverged dramatically in the latter years of the 20th century following the development of eutrophication and sedimentation issues at the former lake and its more developed history of introduced species. Prospects for the local survival of vendace in Bassenthwaite Lake now give great cause for concern, although they may be marginally enhanced by the possible occasional natural input of individuals from the more robust and less threatened population of Derwent Water by passage along the connecting River Derwent.

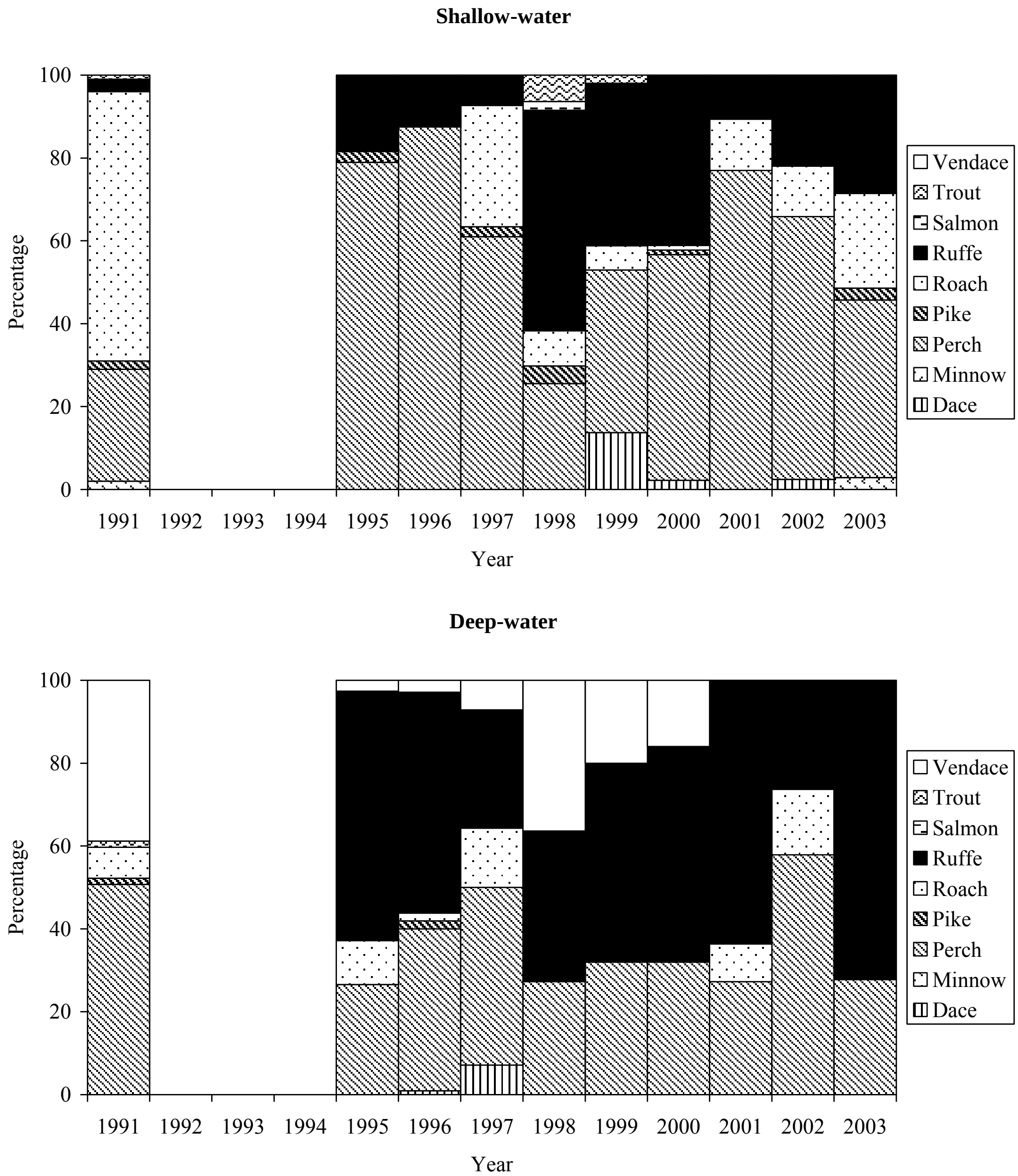


FIG. 28. Species composition by numbers of the shallow-water (total sample size for all years 564 individuals) and deep-water (total sample size for all years 419 individuals) fish communities of Bassenthwaite Lake in 1991 and from 1995 to 2003. Data originate from two of five sites sampled within the monitoring programme described by Winfield et al. (2004b) and the earlier studies of Winfield et al. (1998a) and Winfield et al. (1994a).

Finally, attention is drawn to the potential value of the fish of Bassenthwaite Lake to act as flagship species in the management of this lake. Although not having the charisma of mammals or birds as far as most members of the general public are concerned, freshwater fish species have an established public interest base through the widespread recreational activity of angling. This is certainly the case for Bassenthwaite Lake and its river system where both game and coarse angling are frequently practised. The importance for environmental management of harnessing such public opinion through increased awareness and understanding is now widely appreciated in many countries (e.g. Cambray & Pister 2002) and it is suggested that the vendace in particular could play an invaluable and unique role in this context in the management of Bassenthwaite Lake. Possible strategies for the management of Bassenthwaite Lake, aimed at improving the status of the vendace population, are outlined in the later section on ecological pressures.

BIRDS

Introduction

A brief review of the birds of Bassenthwaite Lake, with a focus on those species closely associated with water, was given by Atkinson et al. (1989). Apparently, no scientific research on these fauna has been conducted subsequently, although extensive monitoring and management activities have been progressed by the Lake District National Park Authority. The latter are reviewed by Barron (2004), which has been used as the principal source of information for the present description.

Species

Barron (2004) reported a total of 127 bird species recorded at Bassenthwaite Lake in 2003, with a further 23 species having been recorded not during that specific year but since 1995.

Most of these species have little contact with the lake itself and so their further consideration is not relevant to the present review, but at least 22 species may be regarded as being aquatic to a greater or lesser extent and occurring in significant numbers or having significant conservation interest. These are great crested grebe (*Podiceps cristatus*), cormorant (*Phalacrocorax carbo*), great egret (*Ardea alba*), grey heron (*Ardea cinerea*), mute swan (*Cygnus olor*), Canada goose (*Branta canadensis*), wigeon (*Anas penelope*), teal (*Anas crecca*), garganey (*Anas querquedula*), mallard (*Anas platyrhynchos*), pochard (*Aythya ferina*), tufted duck (*Aythya fuligula*), goldeneye (*Bucephala clangula*), red breasted merganser (*Mergus serrator*), goosander (*Mergus merganser*), ruddy duck (*Oxyura jamaicensis*), osprey (*Pandion haliaetus*), water rail (*Rallus aquaticus*), moorhen (*Gallinula chloropus*), coot (*Fulica atra*), kingfisher (*Alcedo atthis*) and dipper (*Cinclus cinclus*). Among the latter species, the osprey and cormorant are worthy of further consideration here for very different reasons.

Arguably the highest profile event at Bassenthwaite Lake in recent years has been the establishment of breeding ospreys, which were first recorded in 2001. A detailed account of this welcome development is given by Barron (2004), in which reference is made to the fact that perch is the fish species most regularly caught. This observation is consistent with the recent resurgence in the perch population reported elsewhere in this review. Note that it is highly unlikely that ospreys will ever predate vendace because the latter species spends daylight hours at depths in excess of 10 m, well beyond the diving range of this bird.

Barron (2004) reported a significant number of cormorants, amounting to 57 individuals, on Bassenthwaite Lake in March 2003. In contrast to the osprey, this species is capable of diving to the depths inhabited by vendace. Furthermore, elsewhere in the Lake District, predation pressure from cormorants on the closely related whitefish (*Coregonus lavaretus*) of Haweswater has given considerable cause for concern and precipitated management action on a newly established local breeding colony of this bird (Winfield et al. 2003). Given the general increase in the numbers of cormorants overwintering on UK fresh waters in recent years (e.g. Russell et al. 1996), it would be prudent to investigate the local feeding ecology of this bird at Bassenthwaite Lake in the context of potential impacts on the vendace and other fish populations.

Birds generally enjoy a very high public profile and this is certainly the case for Bassenthwaite Lake as evidenced by the great interest recently aroused by the local breeding activities of ospreys. Although the local fundamental ecology of these fauna has not been studied, their conservation importance and consequent public profile are such that they should not be neglected in the future management and public interpretation of this lake.

MAMMALS

There do not appear to be any studies relating specifically to the mammalian fauna of the catchment of Bassenthwaite Lake. However, unpublished post-1980 mammal records are available (Tullie House Museum, unpublished data) and the species list derived from these records is reproduced in Table 12.

In terms of influencing the functioning of Bassenthwaite Lake, arguably the most significant members of the mammalian community are the sheep (*Ovis aries*) and cows (*Bos taurus)* that graze in the catchment. These animals are likely to make a significant contribution to the diffuse nutrient inputs to the lake and, by poaching catchment soils, are likely to increase the sediment load to the lake.

Table 12. Species list for the non-domesticated mammalian fauna of the Bassenthwaite Lake catchment, based upon unpublished records from the Tullie House Museum, Carlisle.

American mink (*Mustela vison*)	Otter (*Lutra lutra*)
Badger (*Meles meles*)	Pine marten (*Martes martes*)
Bank vole (*Clethrionomys glareolus*)	Pipistrelle (*Pipistrellus pipistrellus*)
Brown hare (*Lepus capensis*)	55 kHz Pipistrelle (*Pipistrellus pipistrellus*)
Brown long-eared bat (*Plecotus auritus*)	Polecat (*Mustela putorius*)
Common shrew (*Sorex araneus*)	Polecat-ferret (*Mustela furo* × *putorius*)
Daubenton's bat (*Myotis daubentoni*)	Pygmy shrew (*Sorex minutes*)
Field vole (*Microtus agrestis*)	Rabbit (*Oryctolagus cuniculus*)
Fox (*Vulpes vulpes*)	Red deer (*Cervus elaphus*)
Grey squirrel (*Sciurus carolinensis*)	Red squirrel (*Sciurus vulgaris*)
Hedgehog (*Erinaceus europaeus*)	Roe deer (*Capreolus capreolus*)
Mole (*Talpa europaea*)	Stoat (*Mustela erminea*)
Natterer's bat (*Myotis nattereri*)	Weasel (*Mustela nivalis*)
Noctule (*Nyctalus noctula*)	Wood mouse (*Apodemus sylvaticus*)

ECOLOGICAL PRESSURES FACED BY BASSENTHWAITE LAKE

Nutrient loads

The magnitude of algal populations is a major water quality issue. Using ecological models, Reynolds & Maberly (2002) and Maberly & Elliott (2002) have shown that phosphorus has a large controlling effect on phytoplankton biomass in Bassenthwaite Lake and that, only at very high TP or SRP loads, does another factor become limiting. This highlights the likely sensitivity of phytoplankton biomass to future changes in phosphorus loading. However, the models underestimated the amount of phytoplankton chlorophyll *a* that has been observed in Bassenthwaite Lake, when driven purely by external nutrient loads from the inflows and WwTW. Additional nutrients needed to be added during the simulations in order to produce phytoplankton standing crops of similar magnitude to those observed. This provided further evidence that an additional source of phosphorus exists, of which the most likely is internal release of phosphorus by wind-disturbance of the sediment, or mixing from depth. These additional sources of phosphorus have the potential to reduce the efficacy of controlling water quality by reducing nutrient loadings from Keswick WwTW. Futhermore, the models predicted that potentially toxic Cyanobacteria would constitute a larger proportion of the total phytoplankton biomass at higher nutrient concentrations. This work has therefore highlighted the potential deleterious effects of further increases in nutrient loading on the water quality of Bassenthwaite Lake, via increases in the supportable phytoplankton biomass and in the proportion of that biomass comprised of Cyanobacteria.

The ultimate way of controlling eutrophication of lakes is to control the load of external nutrients. Control of nutrient point sources is already underway at Bassenthwaite Lake with the implementation of the tertiary treatment plant at the Keswick WwTW. Annual mean total phosphorus concentrations have declined from 25 mg P m^{-3} in 1995 to 22 mg P m^{-3} in 2003 (Maberly et al. 2004). The reduction in annual maximum SRP concentration has been more dramatic, declining from approximately 30 mg P m^{-3} to less than 10 mg P m^{-3} over the same time period. There has been little evidence for a concomitant reduction in the standing crop of phytoplankton since 1995 (Maberly et al. 2004). However, the species composition of the community appears to have changed in that opportunistic species have been suppressed in favour of species which perennate within the lake (Reynolds et al. 2000).

A number of explanations have been offered for the slow speed of remediation. It is possible that phosphorus removal at Keswick WwTW is inadequate to bring about a more dramatic reduction in phytoplankton biomass. This could be because the efficiency of phosphorus removal is strongly dependent on the volume of water passing through the treatment works. Although removal efficiency is typically somewhere between 60 and 80 %, it may be as low as 22 % in very wet weather (Reynolds 1999). However, enhancing the removal efficiency of the treatment process at Keswick WwTW is not necessarily the only answer. Currently, point sources are estimated to account for 60 % of the soluble reactive phosphorus in the lake and only 22 % of the total phosphorus (Maberly & Elliott 2002). The majority of the total phosphorus fraction is derived from diffuse sources (Hall et al. 2000). Thus, a further increase in the efficiency of phosphate stripping would only have a modest effect on phosphorus concentrations in the lake and in fact it has been estimated that the lake concentration of TP would be 19 mg m^{-3} even if the Keswick WwTW achieved 100 % phosphorus removal (Maberly & Elliott 2002).

The remediation of Bassenthwaite Lake by the control of external nutrient loads is likely to be slow because the high mobility of sediment derived phosphorus is believed to result in a large internal loading of phosphorus. This mobility allows the same initial loading to support successive generations of phytoplankton within the lake. This effect is exacerbated by the low rate of removal of sediment from the lake. Although sediments are frequently disturbed by wind-induced mixing events, bulk settling of these particles ensures that much of this material (and the phosphorus associated with it) remains in the lake.

The issue of eutrophication in Bassenthwaite Lake is particularly important given its significance as one of the only two remaining UK sites at which the vendace can be found. The adverse effects of advanced eutrophication on coregonids, such as the vendace, are now well known from many case histories throughout Europe, e.g. Wilkonska & Zuromska (1982), with the two primary impacting mechanisms being low hypolimnetic dissolved oxygen concentrations and the siltation of spawning grounds. Although historical measurements of dissolved oxygen concentrations are unavailable, profiles from the 1990s (Fig. 18) show that summer concentrations may fall below 3 mg L^{-1} in the hypolimnion, which is the concentration about which vendace distribution begins to be impacted (Dembinski 1971). Clearly, any further reductions in dissolved oxygen concentrations could have significant adverse impacts on the vendace population of the lake. The issue of siltation, which may also have origins unassociated with eutrophication, is considered below. In addition to impacts through oxygen availability and deterioration of spawning grounds, the observed eutrophication of Bassenthwaite Lake may also impact the vendace population indirectly by shifting potential competitive conditions from those favouring coregonids and salmonids, to those favouring cyprinids (see Persson 1991).

Sediment loads

Another major issue for the ecology of Bassenthwaite Lake is the progressive increase, since 1900, of sedimentation rates in the lake. The problem is further exacerbated by frequent wind-induced resuspension events and the limited flushing of sediments from the system. These sediments may act as a source of readily bioavailable phosphorus to the water column, supporting algal productivity and therefore affecting water quality, but are also extremely detrimental to the breeding success of the lake's vendace population. The problem of sedimentation on vendace spawning grounds in Bassenthwaite Lake undoubtedly has a eutrophication-induced component, but it is apparently dominated by processes beginning elsewhere in the catchment. Furthermore, there remain a number of unquantified sediment sources in the catchment. As a result, the management of this problem is both complex and difficult. Moreover, even if the source or sources in the catchment were to be identified and completely rectified, it would probably take a considerable period of time before natural flushing removed those problem sediments already in the lake. Another rather speculative possibility is that if the area of the lake colonised by macrophytes can be increased, especially in the northern and southern bays, sediment stability would be enhanced resulting in less sediment resuspension. A programme of sustainable catchment management is certainly required to control the input of nutrients and sediment to Bassenthwaite Lake. This would not only eventually reduce the phosphorus concentration and phytoplankton biomass within the lake, but also enhance the reproductive success of the UK's rarest freshwater fish.

In view of the severity of the sedimentation problem with respect to incubating vendace eggs, an assessment of the feasibility of introducing clean artificial spawning substratum has been made (Winfield 1999). Mats of a type of plastic grass that had previously been successfully used as an incubation substratum for whitefish eggs (Winfield et al. 2002b) were introduced on a spawning ground on 5 December 1998, but on their subsequent inspection only 15 days later they had become heavily sediment-laden and thus inhospitable to vendace eggs as illustrated in Fig. 29. Subsequent chemical analysis has shown that this material is largely inorganic and appears to be the result of erosion in the lake's catchment. Similar investigations at Blelham Tarn in another catchment of

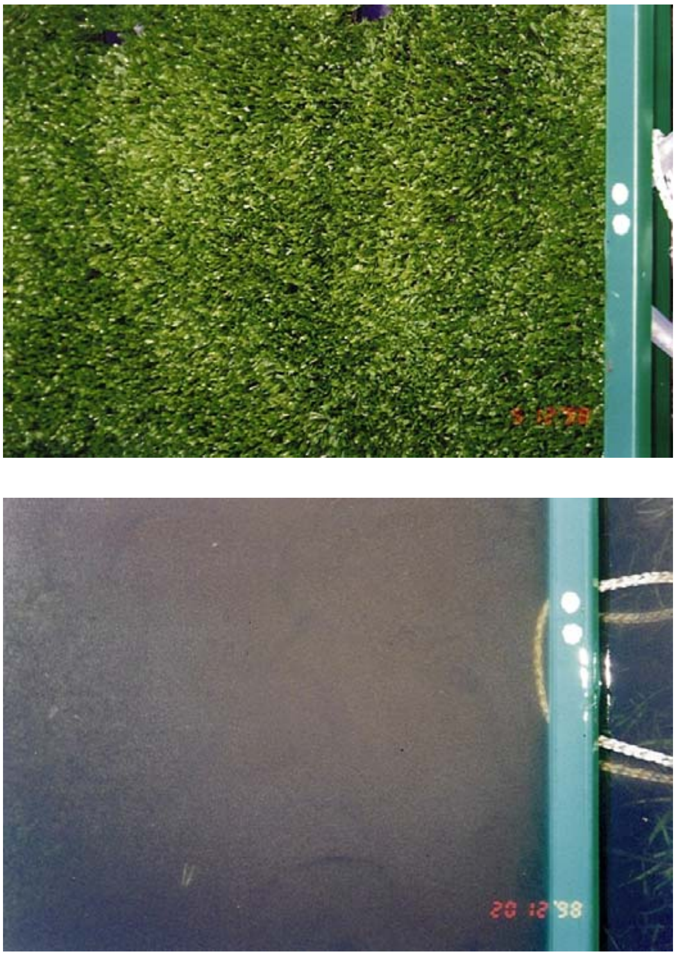


FIG. 29. One of three trays of artificial spawning substratum before (above) and after (below) its installation in Bassenthwaite Lake between 5 and 20 December 1998, illustrating the degree of the problem of the local deposition of fine sediments. Reproduced from Winfield (1999).

the English Lake District have shown increased sedimentation to be attributable to surface soil erosion, largely as a direct response to increased pressure from sheep grazing (van der Post et al. 1997). Whatever its origins, the presence of this material in Bassenthwaite Lake makes successful vendace egg incubation extremely unlikely. The frequent resuspension of exogenous fine sediments currently presents an insurmountable problem for the successful incubation of vendace eggs on any benthic spawning substratum in Bassenthwaite Lake. Moreover, like eutrophication, this threat is likely to remain a problem for a considerable number of years.

If vendace eggs cannot currently be incubated successfully in Bassenthwaite Lake over at least the next few years due to siltation on all known spawning grounds, then a potential emergency measure is to bypass these conditions by stripping fish and incubating their eggs in captivity before returning the resulting young to the lake. Although such capture and husbandry presented no technical problems in the late 1990s (e.g. Lyle et al. 1998), attempts to catch appropriate broodstock in 2000 and 2001 were unsuccessful due to very low population abundance of vendace (Winfield & Fletcher 2002).

All of the above measures share the aim of conserving one of the last two UK vendace populations in its natural habitat of Bassenthwaite Lake. However, given the poor status of vendace in this lake and the low probability of an improvement in local environmental conditions in the near future, the establishment of refuge populations has also been examined as an alternative approach to the conservation of this species.

In order to avoid the generation of new conservation problems at the recipient site, introductions to establish refuge populations are rightly subject to the same conditions and safeguards as those required for fisheries purposes. A feasibility study was consequently undertaken on 87 potential recipient sites in or near the English Lake District (Lyle & Winfield 1999). During a desk study, 34 sites were eliminated on the above grounds as they or their catchments contained other rare fish species, while another 49 sites were deemed not to meet the habitat requirements of vendace. The remaining four sites were evaluated by field visits including the sampling of their fish communities, but none were found to be ideal. Consequently, a refuge population of vendace from Bassenthwaite Lake has not yet been established in England.

However, such a refuge population has now been established in south-west Scotland as a result of attempts to reintroduce vendace to this area using donor material from Bassenthwaite Lake and Derwent Water. The donor eggs were stripped from adults during three spawning seasons from 1996 to 1998 and the resulting eggs incubated in captivity. As the original vendace localities of Castle Loch and Mill Loch remain environmentally unsuitable, newly-hatched larvae from Bassenthwaite Lake were introduced to Loch Skeen (N 55° 26’, W 3° 18’) and those from Derwent Water to Daer Reservoir (N 55° 21’, W 3° 37’) after appropriate evaluation and consultation (Lyle et al. 1998; Lyle et al. 1999a; Lyle et al. 1999b). The success of the introduction of Bassenthwaite Lake vendace to Loch Skeen was demonstrated in 2003 by Maitland et al. (2003) through the capture of 55 individuals, the majority of which belonged to year classes subsequent to those of the initial introductions. Further assessment of the biological characteristics of these fish and a comparison with equivalent data collected from Bassenthwaite Lake in the early 1990s was subsequently carried out by Winfield et al. (2004d). No evidence for a similar success of the introduction of Derwent Water vendace to Daer Reservoir has yet been recorded).

Species introductions

Species introductions represent a major threat to the biodiversity of aquatic ecosystems. Alien species can adversely affect their recipient communities by competing with native species, predating them, introducing new diseases and pathogens, and modifiying habitat structure. In Bassenthwaite Lake, this issue is particularly pertinent for both the aquatic macrophyte and fish communities.

In terms of the macrophyte community, the most significant invading species are *Elodea canadensis*, *E. nuttallii* and *Crassula helmsii*. The former two species arrived some time between 1920 and 1979, while the latter arrived in more recent years. The arrival of *C. helmsii* is particularly alarming, and there is a possibility that it could outcompete native species, including the nationally scarce *Luronium natans*. The precise habitat requirements of the very adaptable *C. helmsii* are not precisely known and physical removal of this species is very difficult. The extent of *C. helmsii* and *L. natans* needs to be monitored carefully.

UK lakes are particularly susceptible to the threat posed by the introduction of non-native fish species because their fish communities are naturally species-poor as a result of the last glaciation ca. 10 000 years ago. Roach, ruffe and dace have all been introduced into the fish community of Bassenthwaite Lake. Assessments of the likely impacts of the above introductions on the vendace population have been made by a combination of diet and distribution studies of young and adult life stages in the context of roach as a potential competitor and ruffe as a potential predator of vendace eggs (Winfield et al. 1998a). While it was concluded that there was no evidence that roach posed a threat, a conclusion subsequently supported by an independent study of roach and vendace in Sweden (Beier 2001), there remains concern over the impact of ruffe. Vendace eggs were absent from the diet of this percid in the vendace spawning period of 1995 (Winfield et al. 1998b), but were subsequently found in varying amounts in the corresponding periods of 1996 and 2001 (Winfield et al. 2004b). As well as ruffe representing a predation pressure for vendace eggs, this species may also indirectly affect the vendace by aiding sediment resuspension and nutrient release, thus stimulating eutrophication (Tarvainen et al. 2005).

Although similar introductions of ruffe elsewhere in the UK, mainland Europe and North America have been recognised as a threat to coregonid populations for many years (Winfield et al. 1998b), intensive attempts to find appropriate biological, chemical or molecular control measures have been unsuccessful (Gunderson et al. 1998). Moreover, for the specific case of Bassenthwaite Lake, the observed spatial overlap between ruffe and vendace in the deep-water areas means that any physical removal method such as trawling or gill netting would produce a significant and counter-productive lethal bycatch of vendace.

An assessment was also made by Winfield et al. (2004c) of the impact of the introduced ruffe and roach on the native perch of Bassenthwaite Lake, given that competitive interactions between ruffe and perch have frequently been recorded elsewhere (e.g. Bergman & Greenberg 1994). However, changes in perch population biology recorded over the period from 1991 to 2002 showed no signs of adverse impacts by the introduced ruffe population through competition or any other means. Thus, the two percids appear to be able to coexist in Bassenthwaite Lake even though the latter species now accounts for up to 53 % of the inshore fish community. Moreover, a low diet overlap observed between the two species also refutes a competition hypothesis. In contrast, diet overlap was very high between perch and introduced roach due to common consumption of *Daphnia* spp. Competition for cladoceran prey between the latter two species under eutrophic conditions often results in depressed perch populations (Persson 1991). It was concluded by Winfield et al. (2004c) that such a situation probably existed in Bassenthwaite Lake in the early 1990s, resulting in a depressed growth rate and truncated length distribution of perch. During the later 1990s, competitive pressure from roach may have been reduced, allowing the recovery of the perch to conditions present in the mid-1980s before the significant development of introduced populations as recorded by Mubamba (1989). The immediate mechanism for this hypothesized reduction in competitive pressure may have been a series of years of very poor or no roach recruitment (CEH, unpublished data), although the reason for this observed recruitment failure is unknown.

The threat of species introductions still needs to be addressed on a more generic level by preventing the introduction of further potentially problematical species. There is strong evidence that such introductions, along with some introductions to other water bodies of the English Lake District, are attributable to discarded or escaped live-bait used by anglers during recreational angling for pike (Winfield & Durie 2004). This situation led the Environment Agency to propose a change to local fisheries byelaws to ban the use or possession with intended use of any dead or alive freshwater fish, salmonids or eel as bait in 14 named lakes of the English Lake District, including Bassenthwaite Lake and Derwent Water (Winfield & Durie, *op. cit.*). This proposal successfully passed through an extensive public consultation phase and has subsequently been approved by the Department of the Environment, Food and Rural Affairs with effect from 26 July 2002.

Climate change

We know that the climate is changing and we also know that lakes can be strongly influenced by weather and hence climate change. This influence operates through a number of interlinked processes ranging from land-use change resulting from socio-economic factors driven by climate change, delivery of materials including sediment and nutrients from the catchment, in-lake processes, hydraulic flushing, and long-term change in water temperature. This all makes prediction of a response very difficult and we will only deal here, briefly, with the most likely responses, particularly as we know that different lakes respond to climate change in different ways (e.g. George et al. 2004).

It is likely that the hydraulic discharge of Bassenthwaite Lake will be affected by climate change. Predictions suggest that winter rainfall levels may increase and summer rainfall levels may decrease. Both are potential threats to the lake. Higher winter rainfall could lead to greater erosion and transport of sediment from exposed soil in the catchment into the lake, with knock-on effects for the already stressed vendace population. In Bassenthwaite Lake, the phytoplankton populations are partly kept in check by the relatively high rates of loss caused by hydraulic flushing. Drier summers and autumns are likely to allow algal blooms to develop, and these may comprise species that are ‘undesirable’ such as Cyanobacteria as occurred in the autumn of 2002. Projected temperatures rises, under existing future climate scenarios, have already been shown to have the potential to increase both phytoplankton biomass, and the biomass of Cyanobacteria in particular, in Bassenthwaite Lake (Elliott et al. 2005a; Elliott et al. 2005b). As a result, it is likely that management measures in a future climate will need to be even more stringent than they are today in order to protect the lake from deteriorating further and to improve its present ecological status.

As well as possible effects on sediment transport, future climatic changes might affect vendace survival via other mechanisms. The biology of aquatic poikilotherms is greatly affected by variations in water temperature, and so temperature changes associated with climate change have the potential to influence the ecology of many lacustrine fish species. In Bassenthwaite Lake, surface water temperatures might rise above the thermal tolerances of the vendace, thus restricting the habitat volume available to this species. However, such effects are not always directly attributable to temperature itself and, for example, a study of growth in non-coregonid species in North America has shown the importance of associated changes in the timing of lake stratification (King et al. 1999). For coregonids, which generically share a requirement for relatively low temperatures and high dissolved oxygen levels, global warming may present the additional complication of causing mortalities as a result of oxygen depletion. This has been predicted for the whitefish (*Coregonus lavaretus*) in Lake Constance, Europe (Trippel et al. 1991) and could occur if phytoplankton blooms become more severe in the future, thus producing elevated quantities of settling algal detritus and greater rates of aerobic decomposition. The result would be enhanced microbial oxygen consumption and more prolonged periods of anoxia in the deep water zone. This would further confine the volume of habitat available to the vendace. In Bassenthwaite Lake, temperature and dissolved oxygen conditions are already periodically near the tolerance limits of vendace and so any further deterioration could have catastrophic consequences.

CONCLUSIONS

Lakes are sensitive and complex ecosystems. They integrate local processes that occur in their catchment, that alter the delivery of nutrients or sediment to the lake, and also respond to regional and global influences such as deposition of nitrogen or sulphur in ‘acid rain’ or increases in air-temperature as a result of climate change. These processes in themselves are complex because they can interact and different lakes vary in their sensitivity to these different pressures. A further complexity occurs when lakes are further perturbed by the introduction of non-native species that can alter the balance of trophic interactions within the lake. These species introductions may result simply from human introductions, but may also be made possible, or their impacts amplified, by the changed conditions in the lake caused by climate change or nutrient enrichment.

Bassenthwaite Lake has been particularly affected by most of these processes. It has a large catchment and so its responses can be strongly catchment-driven. Because of its large catchment and relatively small lake volume it may be particularly sensitive to changes in rainfall: a weather variable that is forecast to alter under a future climate. Bassenthwaite Lake has also experienced invasions of non-native species, with potentially adverse effects on the native species. Lake managers need to understand all these processes and have the tools to ameliorate the most urgent pressures. Long-term consistent records of many physical, chemical and biological variables in a lake such as Bassenthwaite Lake are an invaluable resource to provide that understanding. In turn the understanding will provide the tools, such as lake models, that can be used for the informed management of a lake.

The EC Water Framework Directive requires that water bodies are brought to good ecological status by 2015. Phytoplankton biomass as chlorophyll *a* is one of the criteria used to assess that status. For a shallow lake such as Bassenthwaite (mean depth between 3 m and 15 m) with a low alkalinity (less than 200 μequiv L^{-1}) the currently suggested annual mean concentration of phytoplankton chlorophyll *a* at the good to moderate ecological boundary is 5 μg L^{-1}. At present, annual mean concentrations in Bassenthwaite Lake are about twice these (mean 12 μg L^{-1}) between 2000 and 2004) so there is clearly some way to go to improving the water quality of this lake.

Scientifically-based management is increasingly important because there is almost always a social and economic cost to conservation or improving water quality. This cost is ultimately paid for by the local or national population. It may be the cost of improved water treatment works, or the cost to farmers of changing the way they manage their land. In either case, these costs need to be balanced against the benefits of a diverse and healthy lake ecosystem that provides goods and services to society and can be used and enjoyed by local people and visitors alike. This represents the ethos behind the Bassenthwaite Lake Restoration Programme instigated by the Lake District Still Waters Partnership, a collaboration between the Centre for Ecology and Hydrology, English Nature, The Environment Agency, The Freshwater Biological Association, The Lake District National Park Authority, The National Trust and United Utilities. This initiative provides the framework for sound, scientifically-based sustainable management of the catchment. This review will hopefully be a valuable resource that provides the current scientific information to underpin this initiative.

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