Environmental Change Research Centre

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Feasibility Studies on the Restoration Needs of Four Lake SSSIs
Final Report to English Nature Contract No EIT 30-05-005

B.J. Goldsmith, S.J. Luckes, H. Bennion, L. Carvalho,
M. Hughes, P.G. Appleby, and C.D. Sayer

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ECRC

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Executive Summary

This report sets out to determine the feasibility of restoration work on four lake Sites of Special Scientific Interest (SSSIs) in the north-west of England: Elterwater, Hawes Water, Sunbiggin Tarn and Thurstonfield Loch. These sites have been flagged by English Nature as being at possible risk from eutrophication.

Elterwater had a sewage discharge diverted away from its poorly flushed, Inner Basin in 1999 and chemical monitoring has shown significant reductions in the nutrient status of the Inner Basin following this work. Macrophyte surveys show the Outer Basin to be of good ecological status and nutrients are low. Although the Inner Basin currently contains approximately 2000 m^3 of black, nutrient rich sediments (derived from the sewage discharge between 1970-1999) external nutrient sources are low and thus no remedial action is recommended for Elterwater other than allowing natural recovery to take place. Regular monitoring of water quality and 3-5 year plant surveys should be implemented..

At **Hawes Water t**he loss of plant species and a reduction of charophyte cover have given reason for concern. Chemical monitoring showed the inflow of the lake to be high in phosphorus (P) but this was not reflected in the lake P concentration, perhaps as a result of the high precipitation rate of P in marl lakes. External nutrient sources from agriculture were identified as being the most likely cause of eutrophication at Hawes Water. The possibility of poor sewerage facilities at a nearby holiday development should also be investigated. It is recommended that a catchment plan is implemented to control external nutrient sources and regular monitoring of water quality and 3-5 year plant surveys are implemented. Palaeoecological work is also recommended to establish base-line conditions at Hawes Water.

Sunbiggin Tarn supported 25,000 breeding pairs of black-headed gulls in the 1980's and concerns arose over the possible effects that this has had on nutrient enrichment at the site. Diatom analysis of a sediment core showed no conclusive evidence of gull-induced eutrophication but shifts in the diatom species did suggest a high level of physical disturbance at Sunbiggin Tarn. Gull numbers are now negligible and a macrophyte survey revealed no overall degradation in species composition at the site compared to surveys conducted prior to the expansion of the gull colony. Regular monitoring of water quality and 3-5 year plant surveys are recommended, and external sources of nutrients from agricultural practice should be controlled.

Thurstonfield Lough is a very shallow, eutrophic lake. Its conservation interest focuses on the marginal vegetation and surrounding woodland. Sedimentation rates are high and the area of open water is shrinking due to in-filling and reed bed expansion. Nutrients and sediment loads from agriculture sources pose a significant threat to the lough through further eutrophication and siltation. Because of the low conservation interest of the open water habitat it is recommended that this site be dredged for the express purpose of increasing the quality and extent of the open water. This remedial action must only be taken in conjunction with efforts to reduce the currently high loads of sediment and nutrients from the catchment. Good agricultural practice should be encouraged in the catchment. The construction of a silt trap and the extension of reed bed areas on the inflow will help reduce and manage inputs. Palaeoecological work is recommended prior to dredging in order to establish restoration base-lines for Thurstonfield Lough. Following restoration work it is important to implement water quality monitoring and macrophyte surveys to assess the success of remedial action and plant re-colonisation.

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1
Introduction

English Nature has notified approximately two hundred lakes as Sites of Special Scientific Interest (SSSIs) for their aquatic interest. Of these, it is thought that at least half are deteriorating due to the effects of eutrophication, a situation which has prompted English Nature to initiate a restoration programme in collaboration with the Environment Agency. The aim of the programme is to attempt to establish the potential and feasibility of management actions which can restore sites back to their characteristic biodiversity.

This study investigates four contrasting lake SSSIs in north-west England which have all been identified as being under what is thought to be a threat from eutrophication. The overall aims are to assess the current chemical and biological status of these sites, try to gain an understanding of why they might be becoming more eutrophic and then, via an assortment of different means, establish how they might best be returned to what we perceive to be their ideal state.

In reality there is no real “ideal state” for a lake, as no lake system exists as a boundary unit, completely separate from the surrounding land. One of the quintessential elements of any lake is that it is part of a surrounding ecosystem that is constantly changing through natural and anthropogenic influence. What we are trying to achieve as managers, therefore is an “acceptable state” within the natural and cultural environment. Once we decide that this state is unacceptable, due to the loss of a particular plant or animal for example, we must act – or restore – in order to return it to our preferred state. At the start of any restoration project, the reasons for remediation should be carefully set down and clear evidence of negative change or potential damage must be gained before any action is taken. This will be the basis for justifying the funds to be spent and help to settle any conflicts of interest with other land users (Moss *et al.* 1996).

Only by careful examination of a site can adequate justification be achieved for the implementation of expensive (and potentially ineffectual) management action, such as sediment removal or biomanipulation. The setting of appropriate targets for a lake is thus complex, needing a compromise between what is desirable and what, in practical, political and economic terms, is feasible. All of these aspects must be considered together because, without clearly agreed objectives for the desired condition and use of the lake, it could be at risk from adverse change in the future.

This study aims to address these issues at four SSSI lakes: Elterwater (Cumbria), Hawes Water (Lancashire), Sunbiggin Tarn (Cumbria) and Thurstonfield Lough (Cumbria). A wide range of methods are used including chemical monitoring, biological surveys, palaeoecological studies, catchment reconnaissance, catchment modelling and literature reviews.

2
Methods

2.1 Water Analysis

Water samples were analysed for the following: conductivity, pH, chlorophyll *a*, soluble reactive phosphorus (SRP), total phosphorus (TP), nitrate (as nitrogen, NO_3^--N), potassium (K^+), sodium (Na^+), calcium (Ca^{2+}), iron (Fe-TS) and chloride (Cl^-).

2.1.1 Sample Collection

Polyethylene sample bottles and laboratory glassware were pre-washed by soaking for at least 24 hours in 2% hydrochloric acid, followed by thorough rinsing in distilled then de-ionised distilled water (three times with each). Sample bottles were also rinsed in the field with the water sample being collected. Water samples were collected in 500 ml (unfiltered) and 100 ml polyethylene bottles (filtered). Water was filtered in the field for soluble reactive phosphorus and nitrate analysis using glass fibre filters with a pore size of 0.7 µm (Whatman GF/F), under vacuum. The maximum amount of water was filtered that would pass though one filter, the volume was noted and the filter retained in the dark (preserved with magnesium carbonate) for chlorophyll *a* analysis. Measurements of pH and conductivity were performed in the field.

2.1.2 Soluble Reactive Phosphorus (SRP)

Determination of SRP was done spectrophotometrically. The principle of the analysis involves the phosphate reacting with molybdate, in a suitably acidified solution, to form molybdo-phosphoric acid. This product is then reduced by ascorbic acid to give a molybdenum blue complex. The intensity of the blue complex is directly proportional to the concentration of SRP in the sample and can be determined on a spectrophotometer at 885 nm against a set of standards of known concentration (Wetzel & Likens 1991).

In this study the concentrations of SRP and TP are expressed as micro-grams per litre (µgl^{-1}) of phosphorus (i.e. PO_4^{3-}-P) rather than phosphate (PO_4^{3-}). The conversions between these two forms are:

$$PO_4^{3-}\text{-P} = PO_4^{3-} \times 0.326$$
$$PO_4^{3-} = PO_4^{3-}\text{-P} \times 3.065$$

2.1.3 Total Phosphorus (TP)

The determination of TP requires organic P to be oxidised to an inorganic form so it can be analysed as above. This process is achieved by a persulphate digestion using a microwave digester to apply controlled heat and pressure for the reaction (Wetzel & Likens 1991, Johnes & Heathwaite 1992). Once cooled the samples are analysed in the same way as SRP using standards exposed to the same treatment.

2.1.4 Nitrate (NO_3^--N)

Nitrate analysis was performed by a spectrophotometric method utilising the production of an azo dye. The principle of the method involves the reduction of nitrate to nitrite in the presence of a cadmium catalyst (spongy cadmium), which has been shown to give almost complete conversion (APHA 1989). The resultant sample contains all the oxidised nitrogen as nitrite. The nitrite is then determined by diazotizing the sample with sulphanilamide and then coupling with N,1-naphthylethelene diamide to give an intense crimson azo dye. The absorbance is measured on a spectrophotometer at 543 nm and compared to a set of standards of known concentration.

Nitrate concentrations in this study are expressed as nitrate-nitrogen (NO_3^--N) rather than nitrate (NO_3^-) the conversion between these being:

$$NO_3^-\text{-N} = NO_3^- \times 0.226$$
$$NO_3^- = NO_3^-\text{-N} \times 4.429$$

2.1.5 Chlorophyll *a*

The filter from a measured volume of lake water was ground with fine sand using a pestle and mortar, transferred into a glass centrifuge tube and immersed in a 90% acetone solution. The samples are then allowed to stand for at least two hours in a refrigerator. The chlorophyll *a* pigment is thus extracted by the acetone to yield a green solution, the intensity of which is proportional to the amount of pigment present and can be determined using a spectrophotometer.

The concentration of chlorophyll *a* in the original lake water sample is given by the equation:

$$\mathbf{[chlorophyll\ \mathit{a}]\mu g.l^{-1} = (O.D._{665} - O.D._{750}) \times ((13.9 \times v)/(V \times l))}$$

where:

$O.D._{665}$ = absorbance at 665nm, a distinctive peak for chlorophyll *a.*
$O.D._{750}$ = absorbance at 750nm, a correction for any background turbidity.
v = the volume of the extract in ml.
V = the volume of water filtered in litres
l = the path-length of the cuvettes used in cm.
13.9 = an absorption coefficient (a constant) for chlorophyll *a.*

2.1.6 Anions and Cations

Anion and cation analysis was performed at the Fisheries Research Services Laboratories (Pitlochry) by Ron Harriman.

2.2 The Environment Agency GB Lakes Inventory

A georeferenced inventory of standing waters in Great Britain has recently been developed along with a risk based prioritisation protocol based on the three properties, importance, hazard and sensitivity, to identify waters at risk of eutrophication and acidification, and to assess their potential for restoration (Bennion *et al*., 2003). The inventory contains basic physical characteristics for all standing waters in Great Britain derived from the 1:50 000 ordnance survey panorama digital dataset. For those water bodies >1 hectare, catchment boundaries were generated and associated attribute data were derived, to allow implementation of the risk protocol.

In the eutrophication risk protocol, loadings of the nutrient phosphorus (P) were chosen as the relevant parameter to define the level of hazard that each lake is exposed to. Current nutrient loads were estimated from GIS-derived catchment land use and population data. The total P discharged to each lake from agricultural loss and from humans was estimated using a simplified set of P export coefficients from the literature (Hilton *et al*., 1999). Current total P load (expressed as kg/yr) was then calculated for each lake catchment by summing the total contribution from land use, animals and people. The land cover data are from the CEH Land Cover Map of Great Britain (1990), the livestock statistics are from the MAFF 5km gridded data (1999) and the population data are from the SURPOP 1991 dataset. Historical P concentrations for the year 1931 were hindcast using the University of Reading export-coefficient model for lakes in England and Wales (Johnes *et al*., 1996). The export-coefficients used to calculate the current and hindcast total P loads are different but nevertheless a comparison of the two values for any lake gives a broad indication of changes in nutrient loads between 1931 and 1999. The total P loads were then converted into in-lake annual mean P concentrations ($\mu g\ l^{-1}$) using the relevant OECD regression equations which take account of retention time (OECD, 1982).

Retention times were estimated using annual mean run-off data taken from the Centre of Ecology and Hydrology dataset for 1995-1997. Where possible, measured mean lake depth data were used in the calculation of retention time but, owing to the lack of bathymetric surveys for the majority of lakes, mean depth was based on modelled data in most cases. The use of modelled depth data introduces possible errors into the calculations. Owing to the large number of lakes in the inventory (total=43,738; > 1 ha=c.14 000) and the need for nationally available datasets for running the risk based protocol, data resolution problems inevitably occur. This should be considered when interpreting any of the data from the inventory and it is recommended that information for specific lakes is checked at the local level.

2.3 Macrophyte Surveys

The macrophyte surveys were carried out using a boat, rowing around the entire littoral area of the lakes. Transects at right angles to the shore were carried out at regular intervals. All submerged, floating leaved and emergent plants were recorded. Plant distributions and localised abundance (DAFOR scale) were assessed through a combination of visual inspection, aided by a bathyscope, and throws of a grapnel (double-headed rake). The whole site abundance of each species recorded was also estimated at the end of the survey, using the semi-quantitative DAFOR scale:

D	=	Dominant
A	=	Abundant
F	=	Frequent
O	=	Occasional
R	=	Rare

The prefix “L” for “locally” was used to qualify some DAFOR ratings.

An assessment of status of the macrophyte assemblage for the whole site in terms of Trophic Rank Scores (Palmer, Bell & Butterfield, 1992) was carried out. Secchi depth and maximum depth of macrophyte growth were also recorded during the site visit.

Specimens of charophytes were sent to Nick Stewart for verification.

2.4 Sediment Volume

Sediment volumes were determined at two sites, Elterwater and Thurstonfield Lough. The objectives differed slightly and thus the methods used to determine the depth of sediment at the two sites were slightly different. At Elterwater only the depth of the enriched sediment layer was required, whereas at Thurstonfield Lough the entire sediment depth was measured (see below). Once the depth of sediment had been determined, however, the method for calculating the volume was the same. Sediment depth values were interpolated using an inverse distance weighted algorithm to estimate the total volume of sediment in the lake. The inverse distance weighted algorithm uses a fixed radius to interpolate a value for cells in a matrix (a 10 m cell size was used for this exercise). If there are no or few actual measurements within the search radius then the estimate will be poor. The closer a cell is to an actual measurement the better the estimate will be. Nearby values are given a greater weighting than those further away – hence the name inverse distance weighted. This method is used because it is assumed that sediment depths show a large degree of auto-correlation i.e. the sediment depth at one point is closely correlated with the sediment depth at a nearby point. A mean sediment depth can therefore be determined and multiplied by the lake area to calculate the volume.

2.4.1 Elterwater

At Elterwater the aim was to estimate the volume of the dark layer of sediment in the Inner Basin identified by Haworth *et al.* (1997) as being enriched and due to the sewage outflow. The depth of this layer was obtained by taking multiple cores in a grid pattern across the lake using a gravity corer (Glew corer), and measuring the visible black layer from each core. These data, combined with an accurate location for each point taken with a GPS, were then used to estimate the volume.

2.4.2 Thurstonfield Lough

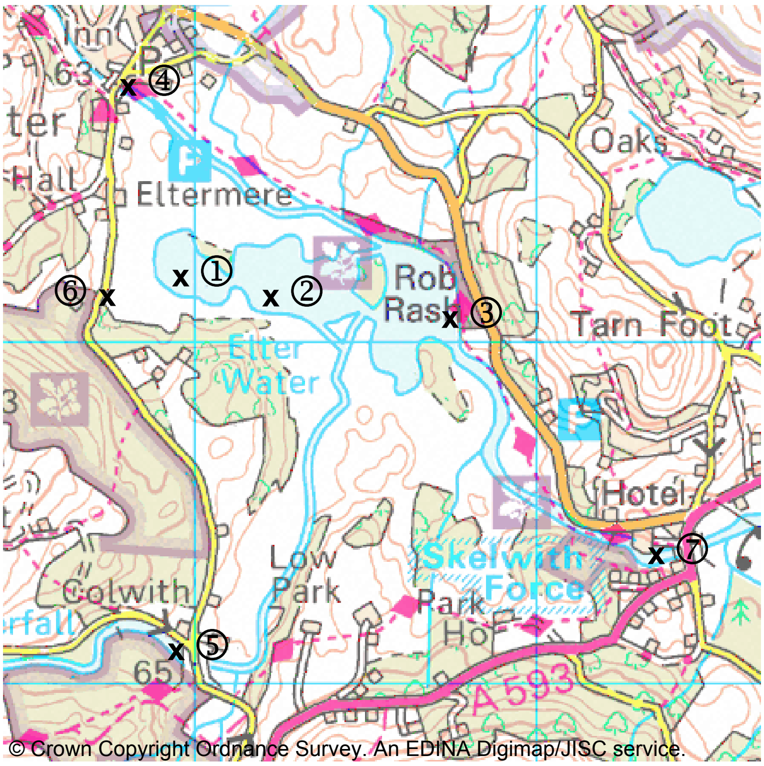
At Thurstonfield Lough the entire sediment volume was needed and thus a different method was used to determine the depth. With a boat, a series of transects were made across the lake and at points along each transect the water depth was noted and a 3 m long metal pole pushed down into the sediment until the hard basal clay was hit. This point was easily judged by an inability to push the pole any further into the sediment. The sediment depth was therefore equal to the distance the pole was inserted from the water surface, minus the water depth. Each point was marked with an accurate GPS reading.

3
Elterwater

3.1 Description

Elterwater lies in the heart of the English Lake District, Britain's largest National Park. Since 1951, when the park was established, the importance of this area has continued to grow both nationally and internationally. The outstanding natural beauty of the area is responsible for attracting in the region of 12 million visitors each year, with the lakes themselves being the fundamental attraction (EA 2000). In addition to its amenity use however, the Lake District remains a living and working countryside and farming and industry continues to play a major role in how the Lake District looks visually and prospers economically (EA 2000). Elterwater is situated near the village of Elterwater in the Langdale Valley and forms part of the catchment draining to Lake Windermere.

Figure 3.1 Map of Elterwater showing water sampling locations



1. Water Sample: Inner Basin
2. Water Sample: Middle Basin
3. Water Sample: Outer Basin
4. Inflow 1: Great Langdale Beck
5. Inflow 2: Little Langdale Beck
6. Inflow 3: Stream inflow to Inner Basin
7. Outflow: River Brathay

Table 3.1 Catchment and lake characteristics for Elterwater.

Catchment	
Location	Langdale Valley, west of Ambleside
Nat. Grid. Ref.	NY334042
Altitude	55 m A.O.D
Total Area	54 km^2
Geology	Borrowdale Volcanics
Land Use	Farm land with fells beyond, slate quarrying
Lake (Inner Basin)	
Maximum depth	7.4 m
Mean depth	2.3 m
Area	3.586 ha
Volume	86,000 m^3
Retention Time	Unknown
Lake (Middle Basin)	
Maximum depth	6 m
Mean depth	2.3 m
Area	7.407 ha
Volume	151,000 m^3
Retention Time	Unknown
Lake (Outer Basin)	
Maximum depth	7.5 m
Mean depth	2.5 m
Area	8.326 ha
Volume	205,000 m^3
Retention Time	Unknown
Current Water Quality Classification	
Outer Basin	Mesotrophic
Middle Basin	Eutrophic
Inner Basin	Hyper-eutrophic
Monitoring	Continuous monitoring since 1995 (Inner Basin)
Nature Conservation Designations	SSSI and within The Lake District National Park.

Elterwater is one of the smaller lakes in the Lake District being approximately 900 m in total length. The underlying geology of the area is predominantly base-poor rock of volcanic origin overlain by nutrient-poor soils and peat (Bennion *et al.* 2003). This geology coupled with high precipitation (c. 2200 $mmyr^{-1}$), results in a natural surface run-off of very low nutrient content (Talling & Heaney 1988). The land use in the Elterwater catchment is predominantly fell, with only a small area of improved grazing and very light industrial usage (minor slate quarrying). Table 3.1 summarises the catchment characteristics.

The lake is distinctive, having three relatively small inter-connected shallow basins known as the Inner, Middle and Outer Basins (EA 2000). There are two main inflows: an artificial channel diverts the Great Langdale Beck from the Middle to the Outer Basin, and Little Langdale Beck now discharges from the south at the junction of the Middle and Outer Basins (Stockdale 1991). During heavy flooding it will overbank into the Middle Basin as will Great Langdale Beck to some extent (Stockdale 1991). Several other small tributaries enter around the lake. The outflow, the River Brathay, flows from the outer most basin and eventually into Lake Windermere. The hydrology of the lake is very unusual in that both major inflows and the only outflow are associated with the Outer Basin, resulting in the Inner (and to some extent the Middle) Basin being very poorly flushed. The Middle Basin only receives sporadic in-flow from the Outer Basin during times of flood, and flow from the Inner Basin is very rare. A total of ninety eight percent of the hydraulic load to Elterwater by-passes the Inner Basin of the lake (EA 2000).

3.1.1 Conservation Interest - Reason for Notification

Elterwater is a designated SSSI and presents one of the most extensive and least disturbed examples of lakeshore wetlands within the region. The site covers a diverse series of habitats ranging from open water, swamp and fen to marshy grasslands, willow and alder carr and drier oak woodland (NCC 1983a). The form of the lake, with three connecting basins of different nutrient chemistry from enriched to more nutrient poor conditions is interesting, and the aquatic flora is varied. The following information is a summary of the "Reasons for Notification" for SSSI status, 1983.

In particular, species included in the site designation are water lobelia (*Lobelia dortmanna*), alternate water-milfoil (*Myriophyllum alterniflorum*), floating bur-reed (*Sparganium angustifolium*), yellow and white water lilies (*Nuphar lutea* and *Nymphaea alba*) and notably the uncommon six-stamened waterwort (*Elatine hexandra*) (NCC 1983a).

The lake margins are of particular interest at Elterwater presenting one of the best lake-shore transition habitats in Cumbria (NCC 1983a). Much of the lake is surrounded by common reeds (*Phagmites communis*), reed canary grass (*Phalaris arundinacea*) and sedges (*Carex* spp.). In association with these there is a diverse flora including gypsy wort (*Lycopus europeaus*), skull-cap (*Scutellaria galericulata*), common valerian (*Valeriana officianalis*) and the local northern sedge (*Carex aquatilis*). Around much of the shoreline these communities grade into marshy grasslands or willow and alder carr. The wet grassland areas are characterised by tufted hair-grass (*Deschampsia caespitose*), purple moor-grass (*Molinia caerulea*), and several species of rush (*Juncus* spp.). In association with these grows a number of herbaceous species, including: greater burnet (*Sanguisorba officianalis*), meadow sweet (*Filipendula ulmeria*) and devil's bit scabious (*Succisa pratensis*). Areas of fen around the lake support bog myrtle (*Myrica gale*), marsh cinquefoil (*Potentilla palustris*), bog bean (*Menyanthes trifoliata*), heath spotted orchid (*Dactylorhiza ericetorum*) and *Sphagnum* moss. The willow and alder carr also supports a rich flora, including shrubs such as guilder rose (*Vibernum opulus*) and bird cherry (*Prunus padus*), and a ground flora including water avens (*Geum rivale*), marsh hawk-bit (*Crepis paludosa*) and the rare touch-me-not balsam (*Impatiens noli-*

tangere). Above the willow and alder carr the drier areas support stands of predominantly oak woodland.

This diverse flora and rich array of habitats provides for a wealth of varied fauna. The site is reported to be rich in invertebrates and provides good undisturbed habitat for wild fowl, including whooper swans during the winter. Reed warbler and nuthatch both occur at Elterwater which is close to their northern limit in the UK.

In addition to important species and habitat at Elterwater the site is also unusual (for The Lake District) because of very low levels of disturbance. The lake is privately owned and the principal land owner is very sympathetic to the conservation needs of the area. Fishing and boating are not permitted on the lake and the only public access to the shore is at the eastern end via a path running around the north-east side of the Outer Basin and thus human disturbance is minimal.

Elterwater supports low numbers of pike, perch and brown trout, but does have large numbers of eels (EA 2000). No migratory salmonids use this lake due to the impassable falls on the Brathay at Skelwith Bridge (EA 2000). The fisheries habitat, within and around Elterwater, appears to sustain reasonable fish populations with many areas of reed bed and overhanging vegetation providing cover (EA 2000).

3.1.2 Reasons for Concern - Nutrient Sources

The three basins of Elterwater are chemically very distinct. The nutrient levels differ between basins, with greater concentrations usually observed in the Inner Basin followed by the Middle and then the Outer. Based on the OECD boundaries and UWWTD criteria for trophic categories, Elterwater was classified as oligotrophic/mesotrophic in the Outer Basin, mesotrophic/eutrophic in the Middle Basin, and hypertrophic in the Inner Basin (Zinger-Gize 1991).

The problems at Elterwater are well documented (*c.f.* EA 2000, Zinger-Gize *et al*. 1999,Talling & Heaney 1988, Stockdale 1991, Haworth *et al*. 1997) and focus on the post 1970’s eutrophication of the Inner Basin brought about by the discharge of treated sewage effluent from the Great Langdale sewage treatment works. Historically, there has been little industry in the catchment, which would influence the lake water quality directly. Notably, juniper and alder was cut near Elterwater to supply charcoal for the Elterwater gunpowder works from circa 1764 until 1930, when the site was converted to a hotel, and finally to holiday timeshare flats in 1981 (Haworth *et al.* 1997). During the nineteenth century linen was produced by John Ruskin at St Martins (Heaton Cooper 1966 cited in Stockdale 1991, Haworth et al. 1997) and copper was mined in the catchment at Greenburn (Haworth et al. 1997). The pale green chlorite-rich slate of the Borrowdale series has long been quarried above Elterwater village and at Hodge Close (draining into Pierce Howe Beck and hence to the Brathay). Although quarrying remains, run-off no longer appears to affect the surface water, although it is possible that past reports of ‘milky’ spate water may have been due to fines from stone cutting (Stockdale 1991). Most of the waste from the catchment would have made its way into Great Langdale Beck and thus, due to the unique hydrology of Elterwater, by-passed the lake.

Anthropogenic sources of nutrients, and especially phosphorus, have increased considerably in the post war period in the Lake District. The causes vary between the different sub-catchments but since 1946 many new sewage treatment plants were set up in the region and effluent was discharged directly to the surface waters, e.g. on the inflows to Blelham Tarn (1962), Esthwaite Water (1973), Grasmere (1971), Windermere South Basin (1964) and Elterwater (1973) (Talling & Heaney 1988).

The Great Langdale sewage treatment works was commissioned in 1973 to treat the domestic sewage for a population of 1752 people (Elterwater village and Chapel Stile), a figure which did not take into account increases during the summer months due to tourism. The treated sewage effluent passed into a stream, which enters into the innermost basin of Elterwater. Apart from field drains this is the only inflow to the western end of the site (Lund 1981). In addition Great Langdale STW has suffered from ground water infiltration problems, resulting in poor efficiency of the plant and the discharge of storm sewage to Great Langdale Beck, leading occasionally to aesthetic pollution problems (EA 2000).

This discharge to the Inner Basin accounted for 43-66% of the mean daily hydraulic input to the Inner Basin, and 70 percent of ortho-phosphate input from the catchment (Stockdale, 1991). The huge increase in nutrients, combined with the poor flushing of the Inner Basin, resulted in hyper-eutrophic conditions and this was also beginning to affect the Middle Basin (EA 2000).

In the two years following the installation of the Great Langdale STW Lund (1981) reported blooms of the motile green algae *Volvox*, an abundance of small centric diatoms (*Cyclotella* spp.) and the cyanobacteriun *Anabaena*. This community was described by Lund as being "similar to that common in sewage-oxidation ponds", while at the same time the communities in the Outer Basin were typical of oligotrophic waters (Lund 1981). In the hot summer of 1976 chlorophyll *a* concentrations exceeded 300 μgl^{-1} on three occasions (Lund 1981). The situation was found to be similar 20 years later by Zinger-Gize *et al*. (1999) with a possible increase in enrichment according to the presence of more cyanobacterial species in the Inner Basin. Chlorophyll *a* concentrations showed a gradient from Inner to Outer Basins with 30–70 μgl^{-1} in the Inner, 10-20 μgl^{-1} in the Middle and <5 μgl^{-1} in the Outer Basin (Zinger-Gize 1991). The progressive increase in phytoplankton chlorophyll between the basins was supported by a parallel increase in numbers and biovolume (Zinger-Gize 1991).

In addition to changes in the algal composition, concerns have also been expressed about the effect of eutrophication on the aquatic macrophytes. In a plant survey conducted by Newbold (1997) concern arose from the low abundance and frequency of some species in the Inner Basin. In addition increased accretion of nutrient rich sediment was highlighted as a potential problem due to an increase in turbidity from algal biomass. The switch to more pollution tolerant macrophyte species was considered likely. Similarly the presence of six-stamened waterwort (*Elatine hexandra*) in the Middle Basin suggests some enrichment. The Outer Basin was more typical of an upper-oligotrophic, lower-mesotrophic community. The low density of water lobelia (*Lobelia dertmanna*) was ascribed to possible stress. Whether such stress was due to increased nutrients and/or increases in sediment accretion was unclear.

Both temperature and oxygen have shown marked stratification. In 1994 and 1995 deoxygenation in the hypolimnion (below 3m depth) was recorded from May to September in the Inner and Middle Basins (Zinger-Gize 1999). During the summer months, when the lake was stratified, a de-oxygenated hypolimnion layer covered over 50% of the basin, but extensive de-oxygenation near the surface was also reported (Zinger-Gize 1999). The Inner Basin differed from the other two basins with de-oxygenation below 4m depth in the Middle Basin, and only a slight decrease in dissolved oxygen levels in the Outer Basin. The temperature difference between the surface and the bottom exceeded 10 ^oC in the Inner Basin, and was about 7 ^oC in the Middle Basin (Zinger-Gize 1999).

Specific conductivity has been shown to differ between each basin with 80 μScm^{-1} in the Inner Basin, 55 μScm^{-1} cm in the Middle and 40 μScm^{-1} in the Outer surface waters. Below the thermocline, conductivity increased to 140 μScm^{-1} in the Inner Basin (Zinger-Gize 1999).

Analyses during the first year of entry of sewage, made at a point a few metres away from its entry into the Inner Basin between April 1974 and May 1975, gave values for SRP, TP, nitrate and ammonium nitrate of 187–7480; 45–8950; 269–3172 and undetectable to 33,600$\mu g\ l^{-1}$, respectively (Lund 1981). Nutrient measurements in the summer of 1994 and 1995 from the centre of the three basins recorded TP values of 145-288 μgl^{-1}, 31-63 μgl^{-1} and 18-33 μgl^{-1} from Inner to Outer. Transparency also increased progressively, from the Inner Basin (1 m) to the Outer Basin (up to 5 m) (Zinger-Gize 1999).

The difference between the three basins is due to the hydrology. Flows were measured in the interconnected narrows from September 1994 to December 1995 (Zinger-Gize 1999). Most flow between basins occurred only as distinct 'events' with the observed flow directions going from the Inner to the Middle Basins (occasionally) or from the Middle to the Outer Basin (Zinger-Gize 1999). Other surveys have shown that floodwater may pass from the Middle Basin into the Inner Basin, which may increase the Inner Basin's volume by over 18% (Stockdale, 1991). These flow events are limited and thus transfer of water between the basins (particularly the Inner Basin) is rare.

Previous monitoring at Elterwater is scarce. There is little available data on chemical analysis of the water and where data exist, it is not always stated from which of the three basins the sample was taken (Stockdale 1991). Figures are given for the nitrate concentration in the River Brathay before entry into Windermere for 1938 by Mortimer (1938, cited in Stockdale 1991). Round (1957 a,b,c cited in Stockdale 1991) included the Inner Basin in his "Study on bottom living algae in some lakes of the English Lake District". He notes anomalies in phytoplankton population (*Euglena mutabilis* and *E.oxyuris* – 1957c) and describes the site as having "abnormally" high productivity (Stockdale 1991). Lund (1981) describes Elterwater as originally oligotrophic. It was known to be similar to lakes Grasmere and Rydal prior to the installation of an STW in 1973 (Stockdale 1991). These one-off measurements and descriptions give us an insight into changes at Elterwater, but cannot confirm the point of change.

In a study on the sediment geochemistry, Haworth *et al*. (1997) found elevated concentrations of P in the upper 15 cm (post-1970) of sediment in the Inner Basin and to a lesser extent in the Middle Basin. This was accompanied by changes in the fossil diatom assemblages. The main results of the study were:

- A sharp increase in organic matter post-1973, with carbon rising from 6–13 mg cm^{-2} yr^{-1}.
- Increases in total phosphorus from 150–650 µg $cm^{-2}yr^{-1}$ especially since c.1970.
- Increases in the planktonic diatom taxa from 6-35% especially since c.1970, a clear shift towards diatom indicators of nutrient rich waters.
- An increase in diatom-inferred pH from c.6.4 to c.7.2 in the late 1960's confirmed by comparison with measurements in the 1950's and 1990's (Haworth et al. 1997).
- A decline in the C/N ratio above 15 cm (c.1968) indicative of increasingly autochthonous algal inputs.

(Haworth *et al*. 1997)

When compared to other Lake District sites (and other lowland sites in the UK) the sediment P concentrations of the Elterwater Inner Basin were very high (Table 3.2, Haworth *et al*. 1997).

Table 3.2 Surface sediment phosphorus at Elterwater compared with the ranges found in other lakes (Haworth *et al*. 1997).

Lake	Total P mg/g	Soluble P mg/g
Elterwater Inner Basin	10.2	3.8
Elterwater Middle Basin	4.8	1.5
Elterwater Outer Basin	3.5 – 14	
Windermere North Basin	4.0	
Esthwaite Water	3.5 – 8.1	
Grasmere	2.7 – 5.9	
Coniston	2.2 – 4.4	0.9
Ullswater	2.4 – 4.7	
Derwentwater	1.9 – 4.3	
Brotherswater	1.9 – 4.6	
Bassenthwaite Lake	2.7 – 3.6	1.0
Slapton Ley, Devon	6 – 16	
Bosherton L, S. Wales	0.8	

From the sediment geochemistry and associated data from the Elterwater Inner Basin, it is clear that the most marked changes date from the early 1970's and coincide with the installation of the discharge from the sewage treatment works. This has led to the greatly increased accumulation of phosphorus and organic matter in the sediments and an accelerated rate of sediment accumulation (Haworth *et al*. 1997). In view of the evidence reviewed, there can be little doubt that it is the increased inputs of phosphorus that are primarily responsible for increased biological production and its consequences at Elterwater.

3.1.3 A Response to the Concern

Concern for the deterioration of water quality in the still waters of the Lake District led in 1990 to a strategy of the National Rivers Authority (NRA) North-West Region, which contained the following recommendations (Zinger-Gize 1999).

1. To protect, and where necessary improve, the existing environmental status of all still waters within the region. The principle objectives are to conserve and enhance the natural beauty and amenity value of these habitats and to safeguard the flora and fauna that are dependent on these water bodies.
2. To increase understanding of how these systems function by developing and maintaining an effective monitoring programme for these aquatic environments.
3. To give particular consideration to those waters which contain rare or endangered species, or possess other special habitat characteristics.
4. To have regard to the use and potential use of still waters as a water resource and to consider the effects of water abstraction on the environment.
5. To take into account the value of still waters as a recreational resource.
6. To improve, where appropriate the visual appearance and general aesthetic quality of still waters.

In 1993, a series of water quality surveys were organised and continuous monitoring established on a number of lakes which were identified as being at risk of water quality deterioration, including Elterwater (Zinger-Gize 1999). A two-year project was initiated in 1994, concentrating on eight key lakes in Cumbria; Bassenthwaite, Derwent Water, Ullswater, Brotherswater, Esthwaite, Coniston, Grasmere and Elterwater (Zinger-Gize 1999).

Due to the minimal water exchange between the Inner and the Outer Basins, it was thought unlikely that improved treatment of Langdale STW, i.e. nutrient removal, would prove effective in the long term (Zinger-Gize 1999). At Elterwater the findings outlined above, led the Environment Agency, in agreement with North West Water plc (now United Utilities), English Nature and other environmental bodies, to propose the diversion of the sewage effluent into the River Brathay downstream of Elterwater, thus eliminating the discharge into the lake itself (Zinger-Gize 1999). The proposals were as follows:

- Relocation of the Great Langdale STW outfall to the River Brathay downstream of Elterwater.
- Studies looking at infiltration are underway.
- New rising main from Great Langdale village to the STW is planned.
- New pumping station.
- Great Langdale pumping station storm overflow is included as an unsatisfactory storm overflow in the AMP III submissions.

(EA 2000)

This diversion scheme was completed by the end of 1999 and a programme to monitor improvements to the Inner Basin and the impact on the River Brathay has subsequently been designed (Zinger-Gize 1999). The current monitoring hopes to assess the natural remediation processes and timescales within the Inner Basin and the possible need for intervention due to high internal loadings from past inputs.

Deposits of phosphorus rich organic sediments remain within the Inner Basin, and due to a lack of flushing are likely to take many years to revert to pre-inflow conditions. Many scenarios have been suggested to tackle this problem, including isolation and nutrient precipitation (Carvalho & Moss, 1995), sediment removal and diverting Great Langdale Beck into the Inner Basin combined with widening the channel between the Inner and Middle Basins to improve the flushing rate. All of these solutions are expensive.

3.2 Aims and Methodology

Four principal aims were identified for investigation at Elterwater. These are outlined below along with a summary of the methods used.

3.2.1 Aim 1:

To estimate the present inputs and outputs of total phosphorus and SRP.

Methodology:

- Quarterly measurements of TP and SRP for the 3 lake basins, 3 inflows and the outflow (Figure 3.1 & Table 3.3).
- Quarterly measurements of nitrate-nitrogen, pH and conductivity, for the 3 lake basins, 3 inflows and the outflow.
- Quarterly measurements of chlorophyll *a* concentrations for the 3 lake basins.
- Quarterly measurements of potassium, calcium, iron, sodium and chloride for the Inner and Middle Basins.

Table 3.3 Location of sampling points in the Elterwater catchment

Site	Location	Os Grid Ref.
Inner Basin	South side of Inner Basin at end of fence. Sampled from beyond reeds.	NY3300,0415
Middle Basin	South side of Middle Basin. Sampled from beyond reeds.	NY3320,0410
Outer Basin	E. side of Outer Basin between inflow and outflow.	NY3375,0415
Inflow 1	Great Langdale Beck - Bridge in the village of Elterwater.	NY3282,0474
Inflow 2	Little Langdale Beck – Colwith Bridge at Wrynose Pass turning.	NY3300,0307
Inflow 3	Below Great Langdale STW - east of the road at edge of woodland.	NY3283,0408
Outflow	River Brathay at Skelwith Bridge - south of craft centre.	NY3436,0339

3.2.2 Aim 2:

To estimate the volume (depth × area) of sediment in the lake.

Methodology:

- According to the palaeolimnological study by Haworth *et al*. (1997), the upper 15 cm of sediment represents the post-1970 enrichment phase in the Inner Basin. A notable colour change to black sediment and marked increase in planktonic diatom taxa were also noted at the 15 cm level in the dated core. Based on these findings, a series of short sediment cores were taken in May 2002 with a Glew corer along transects in order to estimate the depth of this black, nutrient rich layer, across the lake basin and thus provide an estimate of the sediment volume accumulated since c. 1970.

- A similar methodology was applied in the Middle Basin
- This methodology assumes that only the volume of the black, nutrient-rich sediment is required, i.e. that which has accumulated since the start of sewage effluent discharge to the lake in 1973. Given the current oligo-mesotrophic nature of the Outer Basin, sediment volume estimates were not assessed.

3.2.3 Aim 3:

To survey the present aquatic vegetation

Methodology:

- A macrophyte survey was carried out in August 2002 according to contract specifications.

3.2.4 Aim 4:

To prepare a feasibility study for the restoration of the three basins, discussing the merits and approximate costs of mud pumping having made an assessment of how long it would take for natural flushing rates to reduce the sediment input of P to achieve a target level of 30 $\mu g l^{-1}$ TP.

Methodology:

- The results from 1, 2 and 3 above, additional lake and catchment studies, data from the Environment Agency and research on mud pumping were consulted.
- Flow data were not routinely collected as part of the quarterly surveys owing to difficulties and inaccuracies associated with sampling the large outflow. However, existing data on flows and retention times were used and nutrient loads were estimated from the EA GB Lakes Inventory (Bennion *et al.* 2003).

3.3 Results

3.3.1 Water Quality Analysis and Catchment Sources

The results of the quarterly chemistry sampling are presented in Table 3.4. The full data are presented in Appendix I. The reason for the high total phosphorus concentration of the Inner Basin in May 2002 (see Appendix 1) is unclear. Comparisons of the chlorophyll *a* concentrations between the Inner and Middle Basins suggest that some of this was due to higher algal biomass in the Inner Basin, possibly associated with internal P sources released during the spring turnover of the basin. In addition the water levels were high at Elterwater during the May sampling and thus the high value may also be a result of in-washed particulate P from the catchment.

Table 3.4 Mean annual chemistry (2002) for the sampling sites at Elterwater (Inflow 1 = Great Langdale Beck, Inflow 2 = Little Langdale Beck, Inflow 3 = Below STW)

	Inner Basin	Middle Basin	Outer Basin	Inflow 1	Inflow 2	Inflow 3	Outflow
pH	6.95	6.99	6.96	6.56	6.59	6.70	6.56
Conductivity (μScm^{-1})	65	69	52	41	40	85	46
NO_3^--N (mgl^{-1})	0.160	0.098	0.164	0.279	0.175	0.33	0.243
SRP (μgl^{-1})	4.8	4.2	3.4	4.3	2.6	3.6	4.2
TP (μgl^{-1})	31.2	36.6	6.8	10.2	18.1	10.9	23.3
Chl *a* (μgl^{-1})	6.61	5.47	0.92	-	-	-	-
Potassium - K^+ (mgl^{-1})	0.48	0.45	-	-	-	-	-
Calcium - Ca^{2+} (mgl^{-1})	6.57	5.02	-	-	-	-	-
Sodium - Na^+ (mgl^{-1})	4.15	3.77	-	-	-	-	-
Iron – Fe TS (μgl^{-1})	140	160	-	-	-	-	-
Chloride - Cl^- (mgl^{-1})	6.51	5.87	-	-	-	-	-

Following the redirection of treated sewage effluent to below Elterwater a significant decline in nutrient inputs since 2000 would be expected.. Using land cover and stocking data from the EA Lakes Inventory (Bennion *et al.* 2003) the current catchment sources can be identified (Tables 3.5, 3.6 & 3.7). The inventory can also be used to gain an estimate of a target baseline for P at Elterwater using a hind-casting model based on historical data for the year 1931(Johnes *et al.* 1996).

Table 3.5 Catchment and lake characteristics (source: EA GB Lakes Inventory, Bennion *et al.* 2003)

Catchment area (ha)	5129.5
Lake surface area (ha)	18.23
Lake:catch ratio	0.00355
Mean depth (m) (modelled)	2.94
Max. depth (m)	7.5
Total Volume (m^3) (modelled)	535937
Retention time (yrs) (modelled)	0.004
Mean runoff (mm) (CEH data 1995-7)	2561
Stratification class (modelled)	5 (stratified)

Table 3.6 Land use estimates for the Elterwater catchment (source: EA GB Lakes Inventory, Bennion *et al.* 2003)

Land Use Class	Catchment Cover (%)
Moorland grass	45.96
Bracken	27.56
Meadow	11.23
Deciduous woodland	4.22
Open shrub moorland	3.63
Tilled land	1.50
Rural development	0.84
Dense shrub moor	0.82
Grazed turf	0.70
Unclassified	0.64
Urban	0.45
Rough grass	0.42
Water	0.42
Grass heath land	0.41
Bare ground	0.38
Upland bog	0.36
Felled forest	0.34
Coniferous woodland	0.10

Table 3.7 Modelled phosphorus data for Elterwater, including a hindcast estimate for the site from 1931 using export coefficients (Johnes *et al.* 1996) (source: EA GB Lakes Inventory, Bennion *et al.* 2003). * The figures for human impact although underestimated for 1991, are now assumed to be negligible due to diversion of sewage effluent.

Model Component	Model Value
No. Cattle	354.34
No. Sheep	14096.5
No. Pigs	0
No. People (1991)	422*
Land cover P (kg/yr)	343.46
Cattle P (kg/yr)	79.14
Sheep P (kg/yr)	644.0
Human P (kg/yr)	160.36*
Current P load total (kg/yr)	1226.96*
Current P load excluding human P (kg/yr)	1066.6
Current Modelled P conc. Exc. human P (µg/l)	9.0
Hind-cast P load (kg/yr)(1931) Reading model	827
Hind-cast Lake P conc. (µg/l) Reading model	7.0

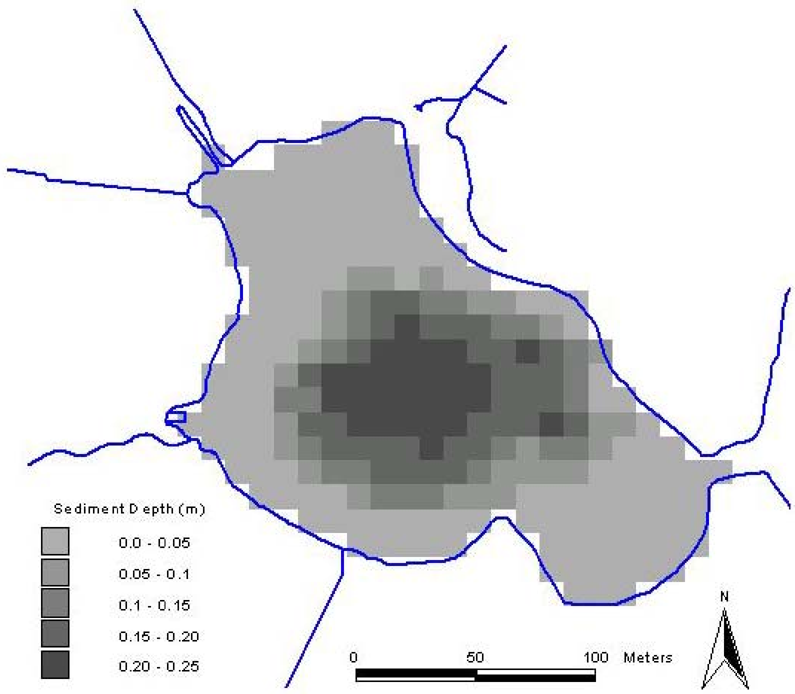
The EA GB Lakes Inventory (Bennion *et al.* 2003) identifies agriculture as the principal source of P to Elterwater. The lake has a relatively large catchment area (5130 ha) with extensive agriculture and thus reasonablh high diffuse inputs are to be expected. The inclusion of human P is no longer applicable since the diversion of the effluent discharge, and although some dwellings in the catchment are likely to remain without mains sewerage the effect will be relatively negligible. Without human P included, the inventory places the current loading at approximately 25 percent higher than the figure for 1931.

3.3.2 Lake Sediment Volume

Elterwater – Inner Basin

Sediment depth was measured at 20 points in the Inner Basin using a Glew gravity corer. Estimates were made of the depth of the organic black layer because the palaeolimnological study of Haworth *et al*. (1997) showed that this layer represents the post-1970 enrichment phase. Below this layer, the sediment was more consolidated and dark brown in colour and the total length of the Glew core at each point was recorded. The depth of the organic black layer ranged from 0 to 24 cm, and the overall length of the Glew cores ranged from 0 (where no mud was retrieved) to 33 cm. The water depth at the sampling points ranged from very shallow areas of less than 1 m in the margins to 8.0 m in the deep central basin. The organic black sediment was found mainly in the deepest, central part of the lake (> 6 m water depth). There was no significant relationship between lake depth and the total sediment depth recovered, except that there was very little sediment in the littoral zone (depth < 1 m). Figure 3.2 shows the depth distribution of the organic black sediments in the Inner Basin.

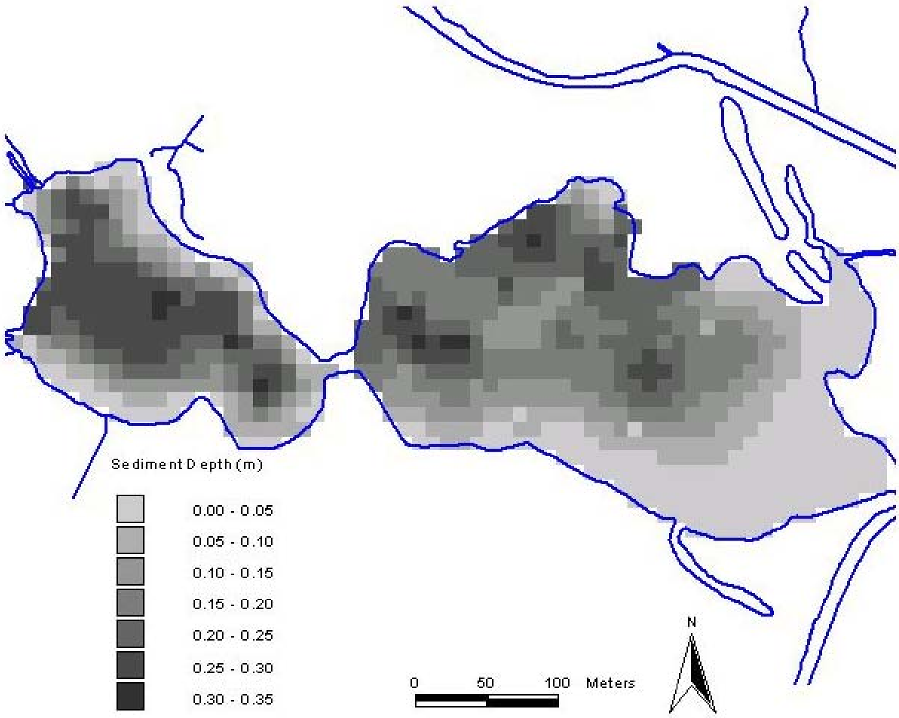
Figure 3.2 Depth distribution of the black organic sediments in Elterwater Inner Basin



Sediment depth values were interpolated from the 20 measured points using an inverse distance weighted algorithm to estimate the total volume of sediment sampled from the lake (based on the total length of the Glew cores) and the portion of that which was black and organic. The estimated mean total sediment depth was 0.17 m. Using a surface area of 2.6 ha (derived from OS Land Line® data), the estimated total sediment volume was 4,300 m^3. The total volume of black organic sediment was estimated at 2,000 m^3.

Elterwater – Middle Basin
Sediment depth was measured at 15 points in the Middle Basin using the same method as for the Inner Basin and Glew core length ranged from 0 to 35 cm. Water depth at the sampling points ranged from 0 to 7.3 m and generally longer cores were obtained from the deeper parts of the basin. There was no distinct black organic layer in the Middle Basin.

Figure 3.3 Depth distribution of the samples of sediment obtained from Elterwater Inner and Middle Basin. No black organic sediments were found in the Middle Basin.



Sediment depth values were interpolated using an inverse distance weighted algorithm to estimate the total volume of sediment in the lake, as for the Inner Basin. Note that there was poor coverage of the lake due to difficulty in collecting data and therefore the uncertainty of the estimate is greater than for the Inner Basin. The estimated mean total sediment depth was 0.12 m. Using a surface area of 5.7 ha (derived from OS Land Line® data), the estimated total sediment volume was 7,200 m^3. The sediment depths sampled from the two basins are shown in Figure 3.3. It should be stressed however that these are not total sediment depths but simply the sample depths obtained using the Glew gravity corer, i.e. Glew core length. It is likely that both basins contain many metres of sediment below that sampled by the Glew corer.

Uncertainty

These estimates should be used only as a rough guide since they are based on a limited set of measurements and there are errors associated with the interpolation procedure, which cannot be avoided. Depending on the density of actual measurements taken, the margin of error for sediment estimates can be as great as 50%. The inverse distance weighted algorithm uses a fixed radius to interpolate (estimate) a value for cells in a matrix (a 10 m cell size was used for this exercise). If there are no or few actual measurements within the search radius then the estimate will be poor. The closer a cell is to an actual measurement the better the estimate will be. Nearby values are given a greater weighting than those further away – hence the name inverse distance weighted. This method is used because it is assumed that sediment depths show a large degree of auto-correlation, i.e. the sediment depth at one point is closely correlated with the sediment depth at a nearby point.

It is important to reiterate that sediment depth is recorded as the depth of sediment retrieved in a Glew core tube when dropped into the sediment using the weight of the gravity corer alone. The depth that the corer will penetrate is largely a function of sediment density, penetration being less in more dense (sticky) sediment. The sediment volumes are subsequently based on the Glew core length data and are not, therefore, intended to give any idea of total sediment volume in the basins. However, given that the interest of this study is in the volume of enriched material which has accumulated during the eutrophication phase (represented by the black organic layer), the method used here should provide a satisfactory estimate of the volume of mud that would need to be removed to reduce internal loading problems.

3.3.3 Macrophyte Survey

Survey Date: *14 August 2002*
Surveyors: *Laurence Carvalho, Carl Sayer, Helen Bennion and Jane Knott*
Secchi depth: *2.2 m (Inner Basin)*

Inner Basin

The submerged vegetation of Elterwater Inner Basin was dominated by *Elodea nuttallii* around the south and east littoral margins present down to about 1.3 m depth. The only other submerged plant was a small localised patch of *Potamogeton obtusifolius* near the outflow to the Middle Basin. In terms of floating-leaved macrophytes, there were large beds of the yellow water lily (*Nuphar lutea*) and occasional patches of *Callitriche obtusifolius* and *Callitriche stagnalis*. The site is most closely associated with a eutrophic/base-rich Type 9 or 10 according to Palmer *et al*. (1992) and has a Trophic Ranking Score of 8.40

In terms of fringing emergent vegetation, the shorelines were dominated by extensive beds of *Phalaris arundinacea* with smaller beds of *Phragmites australis* and *Carex rostrata*, with occasional plants of *Equisetum fluviatile* present in the *Carex* beds by the boathouse on the north shore.

A full species list with DAFOR abundance ratings is given in Table 3.8 and a vegetation distribution map is presented in Figure 3.4. DAFOR ratings for the Newbold (1997) study have mainly been estimated from the survey map for comparison. Newbold only applied the DAFOR rating if species were rare.

Comparing this 2002 survey with an earlier survey by Chris Newbold (1997), it appears that there has been little change in the Inner Basin. The small patch of *Nymphaea alba* recorded in 1997 amongst a *Nuphar* bed, was not recorded in 2002, but it may have been overlooked. There is also little change in site TRS scores. The only significant change is in Secchi disc readings, with a great improvement in water clarity from 1.0 m recorded in August 1997 to 2.2 m recorded in August 2002, suggesting a fairly significant recovery following the stoppage of sewage effluent discharge into the basin.

Figure 3.4 Vegetation distribution map for Elterwater Inner Basin - August 2002

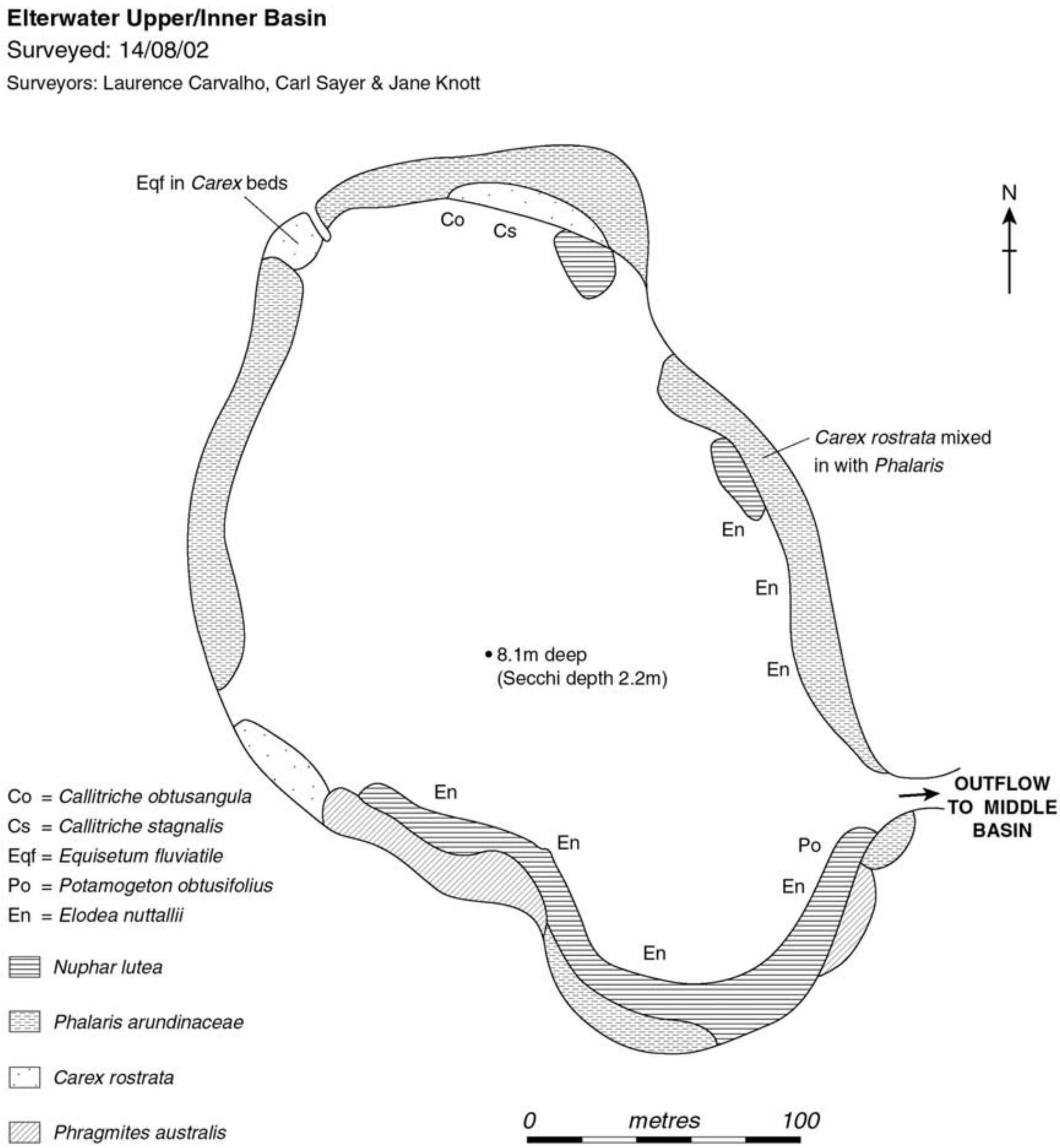


Table 3.8 Species list and DAFOR abundance rating for Elterwater Inner Basin macrophyte surveys

Submerged and floating-leaved species	TRS	14-Aug-97 Newbold	14-Aug-02 Carvalho & Sayer
Nymphaea alba	6.7	O (LA)	
Potamogeton obtusifolius	7.3	LF	LF
Callitriche stagnalis	7.7	O	LF
Callitriche obtusangula	8.5	O (LD)	LF
Nuphar lutea	8.5	A (LD)	A (LD)
Elodea nuttallii	10.0	LF	A
Species richness		**6**	**5**
Site TRS		**8.12**	**8.40**
Palmer Type		**5, 9 or 10**	**9 or 10**
Secchi depth (m)		**1.00**	**2.20**
Emergent species		**14-Aug-97**	**14-Aug-02**
Carex rostrata		F (LD)	F (LD)
Equisetum fluviatile		LF	LF
Phalaris arundinaceae		A (LD)	A (LD)
Phragmites australis		F (LD)	F (LD)

Middle Basin

The submerged vegetation of the Middle Basin was fairly sparse. *Elodea nuttallii* was recorded frequently around the basin in depths down to 1.5 m. Additionally, in the north-east corner of the basin specimens of *Littorella uniflora*, *Fontinalis antipyretica*, *Callitriche hamulata* and *Nitella flexilis* agg[1]. were recovered. Small beds of *Potamogeton obtusifolius*, *Juncus bulbosus* and *Nitella flexilis* agg. were also recorded in the inflow, the Upper Brathay, as it entered the lake.

In terms of floating-leaved macrophytes, there were large beds of the yellow water lily (*Nuphar lutea*) along the south and north banks. The site is a mixture of community types, with a few species typical of oligotrophic/base-poor lake types (Types 2 or 3) and a few species typical of eutrophic/base-rich lake types (Types 9 or 10), according to Palmer *et al*. (1992). The site had a Trophic Ranking Score of 6.22

In terms of fringing emergent vegetation, the shorelines were dominated by beds of *Phalaris arundinaceae*, *Phragmites australis* and *Carex rostrata*, with flushes of *Typha latifolia* and *Equisetum fluviatile* present along the east shore.

A full species list with DAFOR abundance ratings is given in Table 3.9 and a vegetation distribution map is presented in Figure 3.5.

[1] *Nitella flexilis is* referred to as *N. flexilis* agg. as it is an aggregate that is likely to include several varieties and probably at least 2 species. *N. opaca* is the other usually indistinguishable species from *N. flexilis*.

Figure 3.5 Vegetation distribution map for Elterwater Middle Basin - August 2002

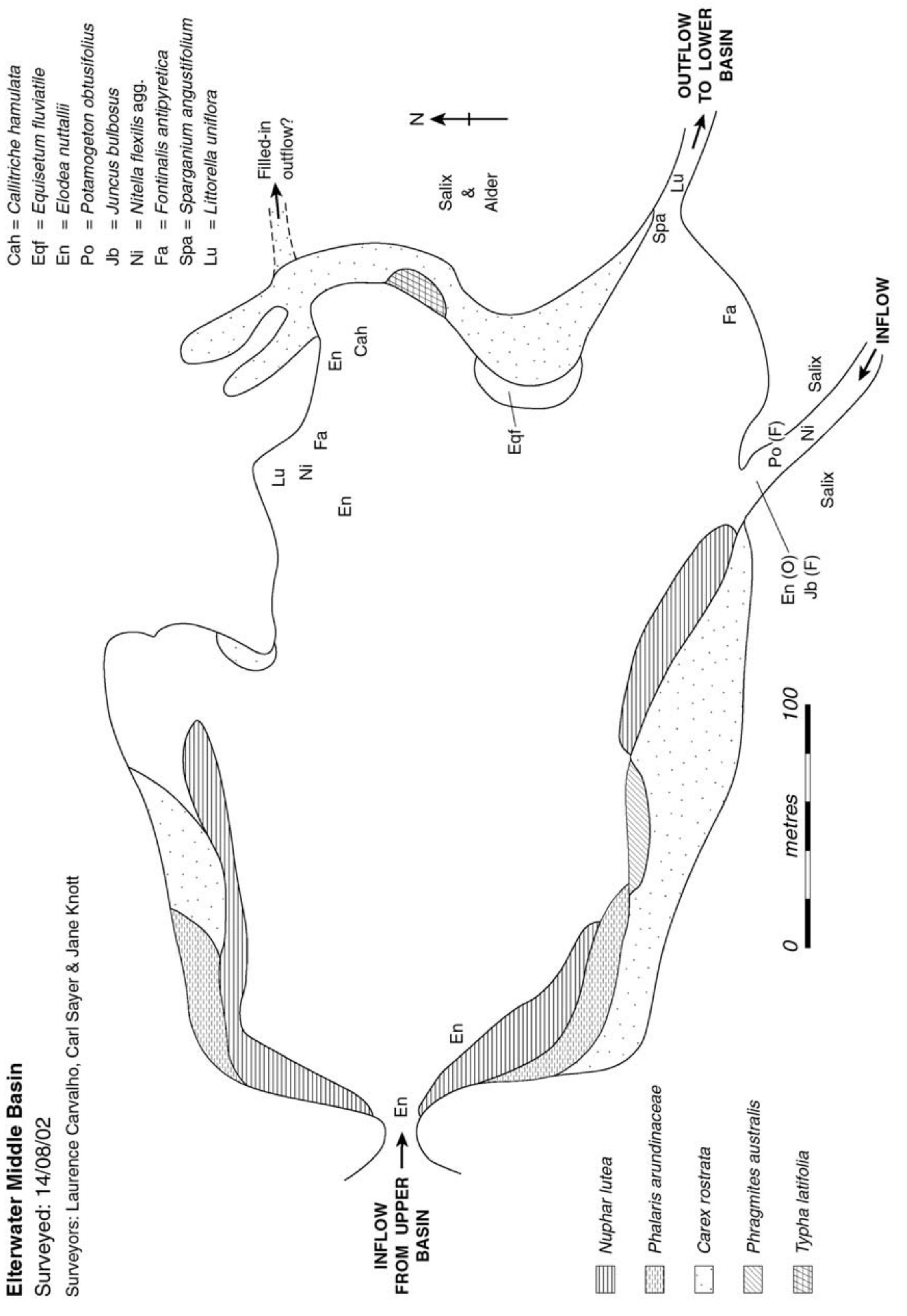


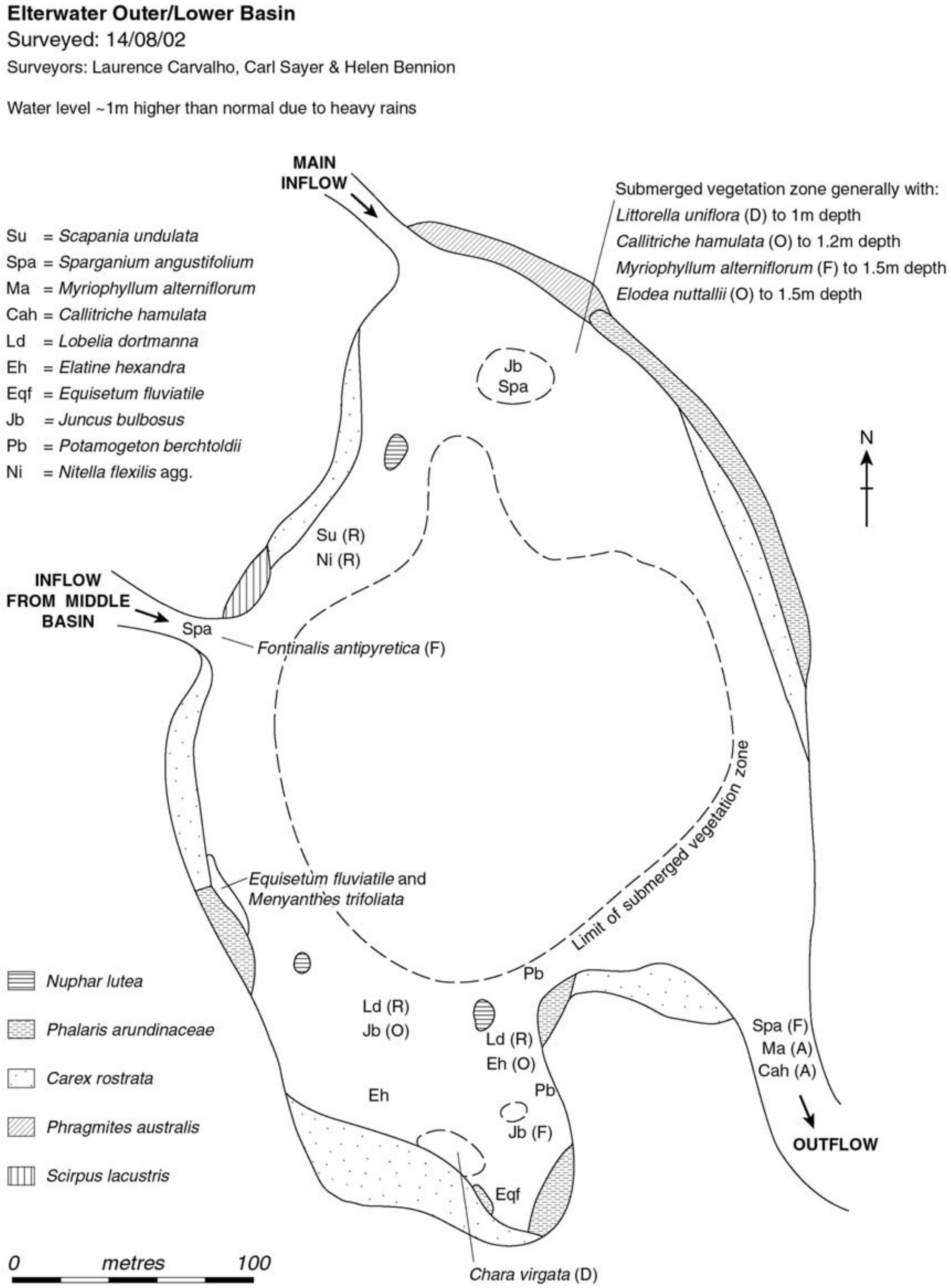
Table 3.9 Species list and DAFOR abundance rating for Elterwater Middle Basin macrophyte surveys

Submerged and floating-leaved species	TRS	14-Aug-97 Newbold	14-Aug-02 Carvalho & Sayer
Sparganium angustifolium	3.0	LF	LF
Juncus bulbosus	3.7		LF
Callitriche hamulata	5.0		LO
Myriophyllum alterniflorum	5.5	LO	
Nitella flexilis agg.	5.5		O
Elatine hexandra	6.0	LO	
Fontinalis antipyretica	6.3		O
Littorella uniflora	6.7	O	O
Nymphaea alba	6.7	LA	
Potamogeton obtusifolius	7.3	LF	LF
Nuphar lutea	8.5	A (LD)	A (LD)
Elodea nuttallii	10.0	F	F
Species richness		**8**	**9**
Site TRS		**6.71**	**6.22**
Palmer Type		**2, 3, 5, 9 or 10**	**2, 3, 5, 9 or 10**
Secchi depth (m)		**1.5 - 1.75**	

Emergent species		14-Aug-97	14-Aug-02
Carex paniculata		F (LD)	
Carex rostrata		A (LD)	F (LD)
Equisetum fluviatile		O (LA)	LA
Phalaris arundinaceae		A (LD)	F (LD)
Phragmites australis			O (LD)
Typha latifolia			O (LD)

Comparing this 2002 survey with an earlier survey in 1997 by Chris Newbold (Unpublished report in EN files), there has been little change in the more abundant species, but there appear to have been changes in the rarer species (Table 3.9). *Myriophyllum alterniflorum* and *Elatine hexandra* recorded in 1997 were not found in 2002, although there were new records in 2002 for *Juncus bulbosus*, *Callitriche hamulata*, *Nitella flexilis* agg. and *Fontinalis antipyretica*. Whether these were real changes or simply reflect the common problem associated with recording rare species is not known. The small patch of *Nymphaea alba* recorded in 1997, was not recorded in 2002, but again may have been overlooked. Because of these macrophyte changes there is a little change in the site TRS scores, reducing from 6.7 in 1997 to 6.2 in 2002, suggesting slight recovery from enrichment.

Figure 3.6 Vegetation distribution map for Elterwater Outer Basin - August 2002



Outer Basin

The submerged vegetation of the Outer Basin was much more species rich and abundant than the other two basins. The shallow littoral zone generally contained *Littorella uniflora*, *Myriophyllum alterniflorum*, *Callitriche hamulata* and *Elodea nuttallii* in depths down to 1.5 m. Additional species recorded occasionally included *Lobelia dortmanna*, *Elatine hexandra*, *Potamogeton berchtoldii* and *Juncus bulbosus* with small localised patches of the bryophytes *Fontinalis antipyretica* and *Scapania undulata* and charophytes *Chara virgata* and *Nitella flexilis* agg.

In terms of floating-leaved macrophytes, there were a few small beds of the yellow water lily (*Nuphar lutea*) and occasional plants of *Sparganium angustifolium*. The site is a clear mixture of community types, with many species typical of oligotrophic/base-poor lake types (Types 2, 3 or 4) and a few species typical of eutrophic/base-rich lake types (Types 5A or 10B), according to Palmer *et al*. (1992). The site had a Trophic Ranking Score of 6.23

Table 3.10: Species list and DAFOR abundance rating for Elterwater Outer Basin macrophyte surveys

Submerged and floating-leaved species	TRS	14-Aug-97 Newbold	14-Aug-02 Carvalho & Sayer
Potamogeton polygonifolius	3.0	R	
Sparganium angustifolium	3.0	LF	LF
Juncus bulbosus	3.7	O	O
Lobelia dortmanna	5.0	F	O
Callitriche hamulata	5.0		O
Myriophyllum alterniflorum	5.5	O	F
Nitella flexilis agg.	5.5		R
Elatine hexandra	6.0		O
Fontinalis antipyretica	6.3		LF
Littorella uniflora	6.7	F	A (LD)
Potamogeton natans	6.7	LO	
Potamogeton berchtoldii	7.3		LO
Chara virgata	8.5		LA
Nuphar lutea	8.5	O	O
Elodea nuttallii	10.0		O
Species richness		**8**	**13**
Site TRS		**5.26**	**6.23**
Palmer Type		**2, 3 or 4**	**2, 3 or 4 and 5A/10B**
Secchi depth (m)		**2.5**	

Emergent species		14-Aug-97	14-Aug-02
Carex rostrata		A (LD)	A (LD)
Equisetum fluviatile		LF	LF
Menyanthes trifoliata			LF
Phalaris arundinaceae		A (LD)	A (LD)
Phragmites australis		LD	LD
Scirpus lacustris		LD	LD

In terms of fringing emergent vegetation, the shorelines were dominated by beds of *Phalaris arundinaceae*, *Phragmites australis* and *Carex rostrata*, with smaller patches of *Scirpus lacustris, Equisetum fluviatile* and *Menyanthes trifoliata* recorded.

A full species list with DAFOR abundance ratings is given in Table 3.10 and a vegetation distribution map is presented in Figure 3.6.

Comparing this 2002 survey with an earlier survey in 1997 by Chris Newbold (Unpublished report in EN files) suggests some degree of change, although mainly in the less abundant species (Table 3.10). *Potamogeton polygonifolius* and *Potamogeton natans* recorded in 1997 were not found in 2002, although there were new records in 2002 for *Callitriche hamulata*, *Nitella flexilis* agg., *Elatine hexandra*, *Fontinalis antipyretica*, *Potamogeton berchtoldii*, *Chara virgata* and *Elodea nuttallii*. A small patch of *Menyanthes trifoliata* was also newly recorded in 2002. These macrophyte changes appear to represent further enrichment of the site, with an increase in site TRS scores from 5.3 recorded in 1997 to 6.2 in 2002.

Elterwater Summary

The three basins represent a clear gradient in the response to nutrient enrichment. The TRS scores decline from the Inner Basin to the Outer Basin, and submerged/floating-leafed macrophyte species richness increases, from 5 species in the Inner Basin, to 9 in the Middle Basin and 13 in the Outer Basin, with increasing numbers of typically oligotrophic species. Improvements in water clarity in the Inner Basin and a decrease in the site TRS score in the Middle Basin (indicative of enhanced water quality) are, however, balanced by an apparent deterioration in the Outer Basin, which has shown a fairly clear shift towards species more typical of eutrophic lakes. The explanation for this is unclear. Current water chemistry results show the Outer basin to be low in nutrients and therefore water quality is unlikely to account for macrophyte changes. The observed results may represent an increase in the sedimentation rates, brought about by enrichment of the Inner Basin or simply be a factor of the natural dynamics of the system. Inherent problems with the survey methods may also result in species being missed. Changes in the TRS are perhaps also questionable because all species of charophytes have a value 8.5 (Palmer *et al.* 1992) and thus the occurrence of *Chara virgata* increases the TRS score but is not necessarily indicative of a deterioration in the water quality.

3.3.4 Summary of Results

- Diversion of the sewage outflow away from the Inner Basin has removed the main external source of P.
- Other external P loadings are relatively low.
- Inner Basin TP concentrations have fallen considerably since diversion of the sewage effluent.
- In-lake TP concentrations in the Middle and Outer Basins have remained largely unchanged since the sewage diversion.
- The Inner Basin has approximately 2,000 m^3 of black organic sediments, identified by Haworth *et al.* (1997) as having accumulated since 1970 and being high in P.
- Black organic sediments were not found in the Middle Basin and thus internal P loads are likely to be lower than in the Inner Basin.
- Aquatic macrophytes show a slight deterioration in the Outer Basin, but more time is needed to ascertain if recovery will be brought about following effluent diversion.

3.4 Feasibility Study

Comparison of post-diversion figures with data collected prior to the diversion shows a marked reduction in the total phosphorus concentrations of the Inner Basin, while the Middle and Outer Basins have remained largely unchanged (Table 3.11). This may herald a stabilisation of trophic conditions, where any further release in P from the sediments will abate over time. There are possible problems however with the long residence time of the Inner Basin. Furthermore the estimation of the actual turnover time is not straight forward.

Table 3.11 A comparison of phosphorus concentrations in the three Elterwater Basins between 1974 and 2002 (All figures are in $\mu g l^{-1}$).

Date	Inner Basin		Middle Basin		Outer Basin		Source
	TP	SRP	TP	SRP	TP	SRP	
1974	32-206	4-123	19-94	0.8-17	7-21	0.2-21	Lund 1982
1975	28-176	0.8-140	12-84	0.6-47	5-27	0.3-3.7	
Jun. ’91	420	150	50	<10	20	<10	Stockdale 1991
Jun. ’91	170	160	30	<10	65	<10	
Jul. ’91	150	140	30	<10	10	<10	
Jul. ’94	-	111	-	2.37	-	2	Zinger-Gize *et al.* 1999
Sep. ’94	288	108	63	22	33	2	
Jul. ’95	145	82	31	7	18	1	
Treated sewage diverted away from Inner Basin in 1999							
Nov. ’00	17.5	12.9	-	-	-	-	This study 2003
Nov. ’01	57.5	4.3	52.5	1.4	37.5	0.0	
Feb. ’02	20.0	3.3	46.0	1.7	2.0	1.7	
May. ’02	72.5	3.3	40.0	3.3	10.0	3.3	
Aug. ’02	17.2	4.0	11.3	3.3	8.3	3.5	
Nov. ’02	15.0	8.3	49.0	8.3	7.0	5.0	
Mean ’02	31.2	4.8	36.6	4.2	6.8	3.4	

It is well known that the hydraulic residence time of the Inner Basin is high compared to the Outer Basins (Zinger-Gize *et al.* 1999). Stockdale (1991) estimated an average daily influx of 1703 m^3, but 790 m^3 of this was due to the sewage treatment works. Calculating the residence time was not possible during this study and the EA GB Lake Inventory (Bennion *et al.* 2003) does not take into account the unusual hydrology of the three lake basins, thus it greatly under-estimates the retention period of the Inner Basin, being based on the hydraulic loadings of the two main inflows to the overall system. Using Stockdale’s figures, with the STW outflow volume removed, the mean daily input to the Inner Basin is 913 m^3. Assuming a lake volume of 96,822 m^3 (SA x Mean depth, Zinger-Gize 1999) then the best estimate for retention time is approximately 0.29 yrs (106 days).

The first objective of any lake restoration programme is to remove the major external nutrient sources (Moss *et al.* 1996). This condition has clearly been met in the case of Elterwater and thus one must consider what further action needs to be taken to ensure that recovery continues. The simplest option would be to do nothing and continue to monitor for improvements. The concern in this case however is the large volume of phosphorus rich sediment which has built up in the Inner Basin (Figure 3.2), coupled with slow flushing rates. Although chemical recovery is already apparent, no change has yet been seen in the macrophytes. This would suggest that natural biological recovery might take a long time due to internal loadings. This is illustrated by the study of Lake Veluwe in The Netherlands, a heavily impacted lake.

A Brief Case Study - Lake Veluwe

Lake Veluwe, an artificial lake in The Netherlands, received treated sewage for over fifteen years, resulting in a switch from clear, plant dominated oligotrophic to turbid hyper-eutrophic conditions (Secchi disc extinction of <20 cm) (van der Molen *et al*, 1998). It took eleven years for the lake to return to a clear-water system following the diversion of the phosphorus source in 1979. This process was aided by winter flushings of nutrient poor waters which decreased the retention time of the lake and so helped to further decrease the internal P loading (van der Molen *et al*, 1998). The role of aquatic macrophytes in stabilising the sediments in the shallow areas of the lake was also found to be beneficial. Thus, as the lake improved and macrophytes re-established, the rate of change back to clear water was accelerated.

The Lake Veluwe study has many parallels with Elterwater, although Elterwater did not reach such an advanced state of eutrophication and it still has an established submerged and emergent macro-flora. Furthermore, although the hydraulic residence time of the Inner Basin is high, what inflow water it does receive is very nutrient poor (mean annual TP <10 $\mu g l^{-1}$). Nevertheless, it is encouraging to see the recovery of a seriously impacted Hyper-eutrophic lake without the implementation of drastic and expensive management plans, e.g. dredging or changing the overall hydrology. From the results of van der Molen *et al*. (1998) it would appear that if any extra flushing of the system is achievable it will "not have any negative effect on lake recovery". Thus benefit could be gained if even relatively small quantities of Beck water were to be re-directed into the Inner Basin (see below).

The diversion of the sewage effluent was very recent (1999) and the changes observed in the macrophyte communities in this study cannot be attributed to this with any certainty. The increase in Secchi disk depth for the Inner Basin is consistent with the lower TP concentrations recorded and this is particularly encouraging. The Middle Basin too has shown some improvement in the plant communities. Of greater concern is that the macrophyte changes in the Outer Basin appear to represent further enrichment of this basin, e.g. the loss of *Potamogeton polygonifolius* and *Potamogeton natans* and the appearance of *Callitriche hamulata*, *Nitella flexilis* agg., *Elatine hexandra*, *Fontinalis antipyretica*, *Potamogeton berchtoldii*, *Chara virgata* and *Elodea nuttallii*. The establishment of these "new" species could be a result of natural dynamics, but it does suggest an environment more conducive to competitive, nutrient tolerant plants. Continued monitoring is recommended to assess any further changes.

According to the EA GB Lakes Inventory (Table 3.7) a pre-war base-line figure for TP is 7.0 $\mu g l^{-1}$. Under present catchment usage this figure is unlikely to be achieved for all three basins, although currently the Outer Basin does have a mean annual TP of 6.8 $\mu g l^{-1}$. The internal loading of P in the upper sediments of the Inner Basin is likely to elevate the P concentrations of the Inner Basin, and possibly the Middle Basin, for many years to come, but due to low levels of P loadings in the inflow waters this situation is unlikely to get worse. It is vital under such a low impact management scheme to implement a structured, long term monitoring programme to ensure improvement continues and to have a contingency should any deterioration be observed.

3.4.1 Recommendations

Prior to this study, there was evidence of a slow but marked decline (with respect to trophic ranking) in the macrophyte communities of Elterwater, following the introduction of sewage effluent. The macrophyte survey in this study is the first full study since the effluent was diverted in 1999 and thus it is too early to say what affect, if any, the diversion has had on the lake ecology. Clearly, improvements have been noted in the TP concentrations of the Inner Basin, but the response of the Middle and Outer Basins is less clear. Due to the considerable volume (2,000 m^3 Figure 3.2) of enriched sediments in the Inner Basin and low flushing rates of the Middle and Inner Basins one might expect the speed of recovery to be slow. In light of these results the following recommendations are made for Elterwater.

Dredging is not considered as an option for Elterwater for the following reasons: i.) the situation is not currently considered bad enough to warrant dredging, ii.) sediment disposal within the area would be unacceptable, iii.) disturbance would be very damaging at Elterwater iv.) the site is already beginning to show signs of recovery (TP and water clarity in the Inner Basin), v.) the high cost is not considered efficient use of resources. Dredging is considered only as a last resort (Moss *et al.* 1996) and therefore is not a viable option at Elterwater.

Under the current conditions of low external nutrient loadings it is recommended that Elterwater be left to recover naturally. This study shows that the water quality has already improved with respect to nutrients and water clarity, since the sewage effluent was diverted. Increased water clarity should help in the recovery of macrophytes and increase the coverage of benthic algae, which should in turn aid further recovery through sediment stabilisation and a reduction in internal loadings.

Under this low intervention management scheme it is vital to implement a monitoring programme to assess chemical and biological change. We recommend that chemical monitoring of the outflow, all three basins and the two main inflows as well as the minor inflow to the Inner Basin, be carried out at a minimum of quarterly intervals. Monthly sampling over the spring and summer months would be beneficial to assess the conditions favourable for macrophyte and algal growth.

Modelled P loads from the EA GB Lakes Inventory (Bennion *et al.* 2003) infer a hind-cast lake TP concentration of 7.0 $\mu g l^{-1}$. This is similar to the current concentration in the Outer Basin (2002 annual mean TP = 6.8 $\mu g l^{-1}$) but it is unlikely that such low concentrations will be achieved in the Inner Basin for many years due to internal loadings of P and very slow flushing rates. The EN target for the Inner Basin has been set at 30 $\mu g l^{-1}$, which according to the results of this study, have almost been met (2002 annual mean TP = 31.2 $\mu g l^{-1}$), suggesting that chemical recovery is well under way, and that the Inner Basin will further benefit over time without remedial action.

An increase in flushing rates could be achieved by the diversion of water from Upper Langdale Beck into the Inner Basin. Under current conditions this is not considered necessary and may be unwise due to the risk of nutrients being washed into the Outer Basin, which currently has low nutrient status and good macrophyte communities.

Regular monitoring of the lake flora is vital to assess any directional changes in the community structure. With regular chemical monitoring in place, aquatic macrophyte surveys need only be undertaken every 3-5 years (July/August) unless any clear deterioration is seen in the water quality. As with any long term monitoring programme it will be necessary to ensure improvements continue and to have a contingency plan should any further deterioration be observed. It is strongly recommended that very careful methodologies are followed for plant surveys to ensure consistency. Ideally personnel should remain consistent too, or at least be involved in training replacement surveyors on site.

It is considered that regular monitoring will be the principal action at Elterwater in the first instance, and that recovery is the likely outcome. It is important to maintain monitoring until a satisfactory end point is reached and the system has stabilised. In light of this study, the target of 30 μgl^{-1} set by EN for the Inner Basin should be revised to bring it in line with the potential levels calculated from the EA GB Lakes Inventory (Bennion *et al.* 2003). The inventory figure modelled from current land-use and hydrological data predicts a potential lake TP concentration of approx. 9 μgl^{-1} but this does not differentiate between the three basins. We recommend, therefore, a revised TP concentration of 15 μgl^{-1} as a realistic target for the Inner and Middle Basins and a value of < 10 μgl^{-1} to be retained for the Outer Basin. If these values are reached and the plant communities remain stable, the level of site assessment can be reduced to six year intervals between surveys to ensure chemical and ecological stability.

3.4.2 Summary of Recommendations

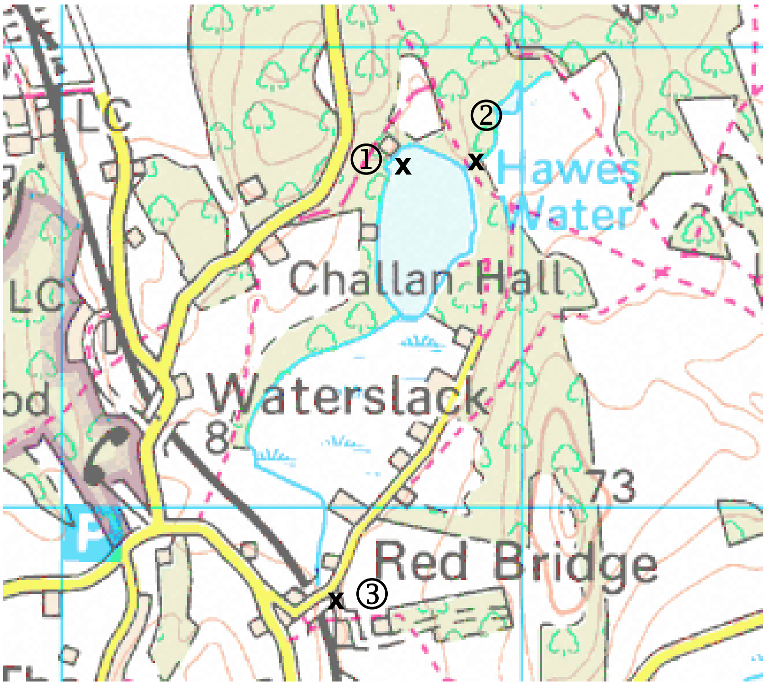
- Following the removal of external P loads Elterwater should be left to recover naturally.
- Monitor water quality of all three basins at a minimum of every three months.
- Monitor the aquatic flora of all three lake basins at a minimum of every five years.
- A new target concentration of 15 μgl^{-1} TP should be set for the Inner and Middle Basins.
- Continue monitoring until a satisfactory end point is reached and the site has stabilised.
- Following stabilisation, water quality and aquatic macrophyte surveys should be conducted every six years to ensure the ecological integrity remains.

4
Hawes Water

4.1 Description

Hawes Water, Lancashire, is located within the Arnside-Silverdale Area of Outstanding Natural Beauty (AONB) and part of the lake basin falls within the Gait Barrows National Nature Reserve (Petley-Jones *et al.* 1996). As well as being a SSSI, the lake is also important as it is one of only two natural water bodies in Lancashire (Marton Mere SSSI near Blackpool being the other) and one of only three marl lakes in the north of England (the others being Sunbiggin Tarn in Cumbria and Malham Tarn in North Yorkshire) (Petley-Jones *et al.* 1996).

Figure 4.1 Map of Hawes Water showing water sampling locations



1. Water Sample: Lake
2. Water Sample: Inflow
3. Water Sample: Outflow

Hawes Water is a low lying lake (6 m above sea level) with a total area of 5.69 ha and a catchment of approximately 124 ha. The underlying catchment geology is of carboniferous limestone series (Tournaisian and Visean) with some alluvial drift (Bennion *et al.* 2003). The lake itself lies on deep marl deposits of up to 7.5m deep which are overlain in places by peat (Newbold 1999). A bathymetric survey showed the lake to have a steeply shelving shoreline to 4 m, with much of the lake exceeding 6 m down to a maximum depth of 12.2 m (Raley, 1993 cited in Newbold 1999). The actual drainage area for the lake is complicated by the limestone geology making an accurate determination of the water sources feeding into Hawes Water very difficult. Springs and groundwater have certainly been identified as contributing to the lake (Marshall *et al.* 2002). In addition to ground water, there is one main surface inflow at the north-eastern edge. This is an artificial inlet formed by the breaching of the limestone cliff (White Scar) to aid drainage of Little Hawes Water in the 19th century (Petley-Jones, *pers. comm.*). The outflow is less well defined, but water leaves the lake via Myers Dyke at the southern end to flow around Hawes Water Moss, an area of extensive reed bed. The moss is artificial, having been drained for agricultural use in the 19th century, but through neglect water levels have returned to near their former level, and natural seral succession to reed beds and alder carr is still underway (Petley-Jones *et al.* 1996).

The land use within the physical catchment boundary, comprises mainly of extensively managed grassland and deciduous woodland. The catchment land use is summarised in Table 4.1. In addition to the recognition of this site as a nationally important marl lake, there are also two nationally restricted types of woodland; fen, carr, and grassland habitats including three rare plant communities; a number of plant and animal species which are notably scarce both in Lancashire and at a national level and the extensive reed bed which is of high ornithological importance (see Petley-Jones *et al.* (1996) for a full description of the conservation importance of Hawes Water).

Table 4.1 Catchment and lake characteristics for Hawes Water.

Catchment	
Location	Silverdale, Lancashire
Nat. Grid. Ref	SD 475765
Altitude	6 m A.O.D
Total Area	124 ha
Geology	Carboniferous limestone
Land Use	Farm land, Woodland
Lake	
Maximum depth	12.2 m
Mean depth	4.17* m
Area	5.69 ha
Volume	237381* m^3
Retention Time	0.32* Years
Current Water Quality Classification	Oligo-Mesotrophic
Nature Conservation Designations	SSSI, SAC, AONB
	* = Modelled data from EA GB Lakes Inventory (Bennion *et al.* 2003)

4.1.1 Conservation Interest – Reasons for Notification

Hawes Water, represents a high quality example of a "hard oligo-mesotrophic water with benthic vegetation of *Chara* formations" as defined under Annex I of the EC Habitats Directive (JNCC 2002). The following information on the site's specific importance is summarised from Petley-Jones *et al.* (1996).

Hawes Water is characterised by base-rich waters containing low to moderate levels of plant nutrients (nitrogen and phosphorus). This results in low phytoplankton production but provides good growth conditions for mainly rooted aquatic higher plants such as shining and broad-leaved pondweeds (*Potamogeton lucens* and *P. natans*), bladderwort (*Utricularia* spp.) and yellow and white water lilies (*Nuphar lutea* and *Nymphaea alba*) which have been found in the beds of stoneworts (*Chara* spp.). Four species of stonewort have been recorded at the site and include the rare *Chara rudis* and scarce species *C. hispida, C. aspera* and *C. pedunculata*, although the latter two species were not found by Newbold in the 1999 survey. These plant communities provide a rich habitat for invertebrates and Hawes Water supports seven species of snails, including *Bithynia tentaculata*, which is restricted to hard water. Eight species of mayflies have been recorded including *Caenis robusta*, which has previously only been found as far north as the West Midlands (Petley-Jones *et al.* 1996).

The lake margins are dominated by common reed (*Phragmites australis*) or great fen sedge (*Cladium mariscus*), growing in association with common club rush (*Schoenoplectus lacustris*), common reedmace, (*Typha latifolia*), marestail (*Hippuris vulgaris*), bur-reed (*Sparganium spp.),* two highly localised sedges, *Carex pseudocyperus* and *C. vesicaria* and the blunt-flowered rush (*Juncus subnodulosus).* This is the only location in Lancashire for the fen-sedge and the community dominated by it is rare at a national level. The blunt-flowered rush is also rare in Lancashire, the only other known occurrences in the county being at Crag Bank SSSI, Robert Hall Moor SSSI and a few other scattered locations (Petley-Jones *et al.* 1996).

On the western shore the fringe grades into alder and willow carr containing shrubs such as guelder-rose (*Viburnum opulus*) and alder buckthorn (*Frangula alnus*) with an under-storey of shorter fen vegetation including marsh marigold (*Caltha palustris*), water mint (*Mentha aquatica*), yellow iris (*Iris pseudacorus*), meadowsweet (*Filipendula ulmaria*), hemp-agrimony (*Eupatorium cannabinum*) and bittersweet (*Solanum dulcamara*). This mixed scrub community on fen peat is reported to be rare at a national level. Beyond the shrub community is a band of marshy grassland on peaty soil dominated by purple-moor grass (*Molinia caerulea*) and accompanied in places by abundant carnation sedge (*Carex panicea*) and numerous other species including angelica (*Angelica sylvestris*), meadowsweet, devil's bit scabious (*Succisa pratensis*), yellow iris, marsh bedstraw (*Galium palustre*), marsh thistle (*Circium palustre*), betony (*Stachys officinalis*) and common, glaucous and false fox sedge (*Carex nigra*, *C. flacca* and *C. otrube*). This species rich *Molinia* community is nationally rare and concern has been raised over the rapid encroachment by willow and alder scrub from the adjacent carr woodland (Petley-Jones *et al.* 1996).

Beyond the reeds to the north and north-east of the lake, a low cliff of shell marl forms a shelf between the water's edge and the former lake margin at the foot of the surrounding cliff. The peaty soil overlying the marl is dominated by *Molinia* but where the marl is exposed blue-moor grass (*Sesleria albicans*) occurs, growing in association with a number of species which are restricted in Lancashire, notably grass-of-Parnassus (*Parnassia palustris*), fragrant orchid (*Gymnadenia conopsea*) and black bog-rush (*Schoenus nigricans*) for which this is the only location in the county. This area also supports birds-eye primrose (*Primula farinosa*) and dark-red helleborine (*Epipactis atrorubens*), two other rare plant species (Petley-Jones *et al.* 1996).

To the south of Hawes Water lies Hawes Water Moss, a mossland once reclaimed for agriculture but now reverted to *Phragmites* reed bed with extensive invasion of willow scrub in recent years. This extensive reed bed, along with one at nearby Leighton Moss SSSI, represent the only two such areas in Lancashire and form an important breeding habitat for reed, sedge and grasshopper warblers. This area is also used by bittern, a nationally rare species and it is hoped that the moss will also encourage marsh harrier and bearded tit, both of which breed at Leighton Moss (Petley-Jones *et al.* 1996).

Among the twenty species of butterfly that have been recorded from the site high brown fritillary, northern brown argus and pearl-bordered fritillary are notable rarities, and the site holds the most northerly record for the silky Wainscot. The wide range of habitats at Hawes Water, support an outstanding bird community and otters have also been reported on the lake (Petley-Jones *et al.* 1996).

4.1.2 Reasons for Concern - Nutrient Sources

An aquatic plant survey was carried out during 1999, showing the *Chara* beds to be much more restricted than during a previous survey in 1984 (Newbold 1999). There was a clear reduction in the size and depth range of the *Chara* beds and two species of *Chara* present in 1984 (*C. aspera* and *C. pedunculata*) were not found in the 1999 survey. Similarly, three species of pondweed recorded in the 1984 survey could not be found in 1999 (*Potamogeton natans, P. friesii* and *P. obtusifolius*) while increases in the more typically eutrophic species of filamentous algae *Cladophora glomerata* and Nuttall's pondweed (*E. nuttallii*) were observed. These observations are consistent with the effects of eutrophication, but without supporting monitoring data this could not be confirmed (Newbold, 1999). Recently, algal blooms have been reported on the eastern shore of the lake during the summer months (Petley-Jones, *pers. comm.*), suggesting higher levels of nutrients.

In addition to changes in the flora, there has also been an apparent four-fold increase in the sedimentation rate at Hawes Water since about 1970. This was reported by Appleby (1998) from the lead^{210} dating of a 1993 sediment core taken from the lake. The reason for this increase is unclear but if it is due to increases in internal production it would point clearly to eutrophication. Alternatively there has been erosion in the catchment since the 1970's, causing sediments to be washed in to the lake or extensive re-working of older sediments from the lake itself. There is no obvious reason for the latter scenarios and thus this gives cause for concern and further justification for monitoring at Hawes Water. The effects of eutrophication can be delayed in hard-water marl lakes due to the co-precipitation of P with calcite (see text box below).

The precipitation of dissolved phosphate with calcite ($CaCO_3$) has been observed in many marl lakes (Danen-Louwerse *et al.* 1995, Dittrich & Koschel 2002). The process involves the preferential binding of ortho-phosphate to particles of calcite in the water column. These calcite particles then sink to become part of the sediments and therefore reduce the availability of P to algae and higher plants. Thus a marl lake which receives high phosphorus loads can effectively buffer P by calcite precipitation and reduce the effects of eutrophication by enhancing the internal phosphorus sink. The extent to which this process is reversible is less well understood and thus it cannot be relied upon to reduce P concentrations indefinitely (Dittrich & Koschel 2002). The control of external sources of nutrients should always be addressed.

Eutrophication would undoubtedly be a threat to this site, and if a problem is evident this could also have knock-on effects for Leighton Moss RSPB reserve (SSSI and SAC), which has recently been identified as having elevated nutrient loadings (Petley-Jones *pers. comm.*). There are no documented sources of nutrient pollution to Hawes Water, but diffuse inputs cannot be ruled out. Agriculture in the catchment is relatively extensive with the majority of the pasture being used for sheep grazing. Pasture on the Gait Barrows receives no artificial fertilizer but small areas within the catchment have occasionally received applications of farm slurry in the past 10 years and a local farmer reported that he thought a pig slurry holding tank may have been leaking (Petley-Jones, *pers. comm.*). There are very few dwellings in the catchment but none are on mains sewerage. Those on Moss Lane are unlikely to contribute waste to the lake, but rather in to Hawes Water Moss below. Of more concern however is Challan Hall to the west of the lake. Over the past fifteen years the out-buildings of Challan Hall have been developed into time-share apartments with a current capacity for approximately 50 people. It is not known to what extent the septic tank for this development has been improved to cope with a ten-fold increase in population nor where the current outfall is. This has however been raised as an issue of concern by the site manager.

4.2 Aims and Methodology

Four principal aims were identified for investigation at Hawes Water. These are outlined below along with a summary of the methods used.

4.2.1 Aim 1:

To measure the present levels of total phosphorus (TP) and soluble reactive phosphorus (SRP) and any significant variation within the lake.

Methodology:

- Quarterly measurements of TP and SRP for the lake, the inflow and outflow (Figure 4.1 and Table 4.2).
- Quarterly measurements of nitrate-nitrogen, pH, conductivity for the lake, the inflow and the outflow.
- Quarterly measurements of chlorophyll *a* concentration for the lake.
- Quarterly measurements of potassium, calcium, iron, sodium and chloride for the lake.

- Variations in nutrient concentrations within the lake were broadly assessed based on comparison of the lake, inflow and outflow nutrient chemistry data.

Table 4.2 Location of sampling points in the Hawes Water catchment

Site	Location	Os Grid Ref.
Lake	North-west side of the lake close to old stone boat-house beyond fence. Sampled from beyond reeds.	SD4775,7679
Inflow 1	Small stream between Little Hawes Water and main lake - sampled by foot path	SD4788,7674
Outflow	Myers Dyke to the south of Hawes Water at Red Bridge	SD4756,7580

4.2.2 Aim 2:

To ascertain the source of enrichment.

Methodology:

- A catchment reconnaissance and a literature survey were carried out to identify potential nutrient sources.
- Nutrient export from catchment uses was calculated using the data held in the EA Lakes Inventory (Bennion *et al.* 2003).

4.2.3 Aim 3:

If economically possible, to determine the residence time of the water at Hawes Water.

Methodology:

- Flow data have not been routinely collected as part of the quarterly surveys because this was not detailed in the original contract specification. However, an estimation of retention time has been made using the EA GB Lakes Inventory (Bennion *et al.* 2003).

4.2.4 Aim 4:

To prepare a feasibility study for the restoration of water quality at Hawes Water.

Methodology:

- The results of 1-3 above, a literature search and liaison with the site manager have been employed to prepare a feasibility study for Hawes Water.

4.3 Results

4.3.1 Water Quality Analysis

The results for the quarterly chemistry sampling are presented in Table 4.3 as annual mean data based on the four 2002 measurements. The full data-set appears in Appendix I. Estimates for lake retention times are based on the modelled catchment and hydrological data from the EA GB Lakes Inventory (Bennion *et al.* 2003).

Table 4.3 Mean annual chemistry (2002) for the three sampling sites at Hawes Water.

	Lake	Inflow	Outflow (Red Bridge)
pH	7.94	7.57	7.62
Conductivity (μScm^{-1})	396	467	408
NO_3^--N (mgl^{-1})	0.260	0.547	0.320
SRP (μgl^{-1})	3.9	36.5	10.8
TP (μgl^{-1})	21.1	87.4	32.2
Chl *a* (μgl^{-1})	3.59	-	-
Potassium - K^+ (mgl^{-1})	0.66	-	-
Calcium - Ca^{2+} (mgl^{-1})	75.13	-	-
Sodium – Na^+ (mgl^{-1})	10.00	-	-
Iron - Fe^{3+} (μgl^{-1})	30.0	-	-
Chloride - Cl^- (mgl^{-1})	17.07	-	-

Lake TP concentrations of 21.1 μgl^{-1} are lower than were expected, given the concern that the site was suffering from eutrophication. High TP concentrations in the inflow stream however are less encouraging and suggest nutrients may well be a problem at the site. There are several possible reasons why these high concentrations in the inflow were not reflected in the lake. First, marl lakes present an alkaline environment and are very efficient at precipitating phosphorus (see p. 39). Second, the inflow may only have sporadic pulses of high nutrient water due to catchment management practices (agricultural input) and these coincided with our sampling visits. Third, whilst the overland flow is nutrient rich, sources of low nutrient ground water may dilute the effect in the lake. A fourth possibility is that high plant biomass in the lake is acting as a nutrient sink and thus is masking the true "total" phosphorus concentration (Moss *et al.* 1996). The first scenario is considered to be most likely at Hawes Water.

4.3.2 Catchment Nutrient Sources

There are no obvious catchment sources of nutrients beyond those already mentioned above. Agricultural inputs are without doubt a factor and the possibility of seepage from the Challan Hall cesspit cannot be discounted, but at this stage it is not known to what extent these may be affecting the site, if at all. It is however, possible to use the EA Lakes Inventory (Bennion *et al.* 2003) to estimate current loadings based on catchment land-use and stocking densities (Tables 4.4 & 4.5). Export coefficients can also be applied to historical land-use data to calculate hind-cast P loadings for the year 1931 (Johnes *et al.* 1996) (Table 4.6).

Table 4.4 Catchment and lake characteristics (source: EA GB Lakes Inventory, Bennion *et al.* 2003)

Catchment area (ha)	123.5
Lake surface area (ha)	5.69
Lake:catch ratio	0.04609
Mean depth (m) (modelled)	4.17
Max. depth (m) (modelled)	10.6
Volume (m^3) (modelled)	237381
Retention time (yrs) (modelled)	0.32
Mean runoff (mm) (CEH data 1995-7)	551
Stratification class (modelled)	5 (stratified)

Table 4.5 Land use estimates for the Hawes Water catchment (source: EA GB Lakes Inventory, Bennion *et al.* 2003)

Land Use Class	Catchment Cover (%)
Deciduous woodland	28.80
Meadow/semi-natural	27.53
Tilled land	18.93
Grass Heath	9.62
Grazed turf	7.64
Water	3.54
Unclassified	1.92
Bare ground	1.11
Rural development	0.51
Coniferous woodland	0.25
Urban	0.15

Table 4.6 Modelled phosphorus data for Hawes Water, including a hindcast estimate for the site from 1931 using export coefficients (Johnes *et al.* 1996) (source: EA GB Lakes Inventory, Bennion *et al.* 2003).

Model Component	Model Value
No. Cattle	52.76
No. Sheep	262.51
No. Pigs	0
No. People (1991)	0
Land cover P (kg/yr)	27.85
Cattle P (kg/yr)	11.61
Sheep P (kg/yr)	11.81
Human P (kg/yr)	0 (?)
Current P load total (kg/yr)	51.27
Current Modelled P conc. Exc. human P (µg/l)	34.5
Hind-cast P load (kg/yr)(1931) Reading model	49
Hind-cast Lake P conc. (µg/l) Reading model	33

The modelled P loads at Hawes Water assume that there is no human derived P entering the lake (NB the population data in the EA GB Lake Inventory are from 1991). The contribution of people to the total load is currently unknown, but if the Challan Hall holiday lets do input effluent to the lake this could have significant implications for water quality. It is thought that the current capacity of the complex is in the region of fifty people (Petley-Jones *pers. comm.*). Assuming the complex runs at an annual capacity of 50 percent, this would contribute an additional 9.5 $kgyr^{-1}$ of P to the lake (i.e. 18 % of the current estimate for catchment loadings). Accurate figures on the Challan Hall complex were not obtained, but even if 9.5 $kgyr^{-1}$ of P is an over-estimate, the possibility remains that some waste from the Hall is entering the lake. Furthermore, if the cesspit is not of sufficient capacity, the quality of the waste may be extremely low.

4.3.3 Determination of Residence Time

There were two major problems encountered in the determination of residence time for Hawes Water. The first was the difficulty in obtaining flow data from the inlet and outlet. In order to accurately determine the input and output of water, measurements need to be made at a much greater frequency than was possible in this study. Secondly, due the nature of the limestone geology, water is very likely to be entering and leaving the lake via subsurface flows (Marshall *et al*. 2002), thus making any surface flow data unreliable. More detailed hydrological studies would be needed to measure the actual residence time of this lake.

The alternative method is to use the EA GB Lakes Inventory (Bennion *et al.* 2003) which calculates the residence time according to catchment area, modelled lake volume and mean annual run-off based on meteorological data (Table 4.4). Although this is a rather generalist method, it does provide the best possible figure available i.e. a residency of 0.32 years (117 days).

4.3.4 Summary of Results

- Current lake TP concentration is 20 μgl^{-1}, which is considered good for this site.
- High concentrations of TP in the feeder stream are a cause for concern.
- Nutrient sources are most likely to be from agriculture but the Challan Hall complex needs further investigation.
- If Challan Hall is contributing to the lake, this may account for approximately 18% of the estimated total P load of the lake. This is unconfirmed.
- A residence time of 117 days is the best estimate from the EA GB Lakes inventory (Bennion *et al.* 2003).

4.4 Feasibility Study

The management recommendations from this study are very much limited by the nature of the results. Changes in the plant communities and an apparent increase in the sedimentation rate shown by previous studies suggest that the site has become more eutrophic, but this does not constitute conclusive evidence. The changes seen in the plant community may simply be due to natural dynamics of the system and not trophic changes. Similarly the increased sedimentation may be from external sources and although this gives cause for concern in its own right it will not necessarily affect the trophic status. Without detailed hydrological data and past water chemistry data, it is beyond the scope of this study to determine if there has been any categorical change in the trophic status of Hawes Water.

Based on the EA GB Lakes Inventory data (Bennion *et al*. 2003) the current P load (excluding people) would give an estimated lake TP concentration of 35 μgl^{-1}. The measured TP in the lake showed seasonal variability with high summer concentrations recorded, but the annual mean for 2002 was only 21 μgl^{-1}, considerably lower than the model would suggest. The concentrations of TP in the inflow however are high, with a maximum value of 211 μgl^{-1} being recorded and an annual mean of 87 μgl^{-1}. It is possible that these high input values are not reflected in the lake water due to the increased rates of P precipitation in marl lakes **(**Danen-Louwerse *et al.* 1995, Dittrich & Koschel 2002). Concern must be flagged at these results however, and further investigations into catchment sources are recommended.

The most likely source of these nutrients in the inflow stream is agricultural. Although most of the catchment is extensively managed there is a need to ensure codes of good agricultural practice (DEFRA 2002) are being adhered to. Slurry applications have been recorded in the catchment and inadequate slurry storage has also been suggested (Petley-Jones *pers. comm.*). A recommendation of this study is to continue to communicate with local farmers and ensure codes of good practice are followed. It would be beneficial to liaise with the Environment Agency to develop a diffuse pollution management plan for the Hawes Water catchment.

The other potential problem at Hawes Water is the development at Challan Hall. No records could be found of cesspit improvements at the hall since the start of holiday-let developments on the site, and thus it is very likely that an extra contribution of P is reaching the lake from the Hall. The local planning authority and Environment Agency should be able to help with information on the exact location and type of sewerage system currently in use and advise on action to take if the lake is receiving this effluent. No evidence of any out-fall was found during the catchment reconnaissance and further investigation is recommended.

4.4.1 Recommendations

From the evidence examined and collected, there is reasonable cause to suspect that Hawes Water is becoming more eutrophic. Most importantly, therefore, a reduction in catchment sources (as outlined above) should be a priority at the site. Regular monitoring of water chemistry and catchment practices will be essential to establish any directional changes and identify with greater confidence the possible nutrient sources. The interval of monitoring would ideally be higher than in this study. Greater cost efficiency would be gained by monthly monitoring during the spring and summer months with fewer samples taken in the winter. Monitoring of ground water quality in the catchment is also recommended to aid in the identification of nutrient sources.

Regular monitoring of the lake flora is necessary in order to assess any directional changes in the community structure of the lake. With regular chemical monitoring in place, aquatic macrophyte surveys need only be carried out every 3-5 years (July/August) unless any clear deterioration is seen in the water quality. As with any long term monitoring programme it will be necessary to ensure improvements continue and to have a contingency plan should any further deterioration be observed. It is strongly recommended that very careful methodologies are followed for plant surveys to ensure consistency. Ideally personnel should remain consistent too, or at least be involved in training replacement surveyors on site.

It is recommended that palaeoecological studies on plant macrofossils, *Cladocera* and diatoms be carried out to help provide information on past community dynamics and allow for past environmental conditions to be inferred. Such information would help to establish base-line conditions for the lake and assess any significant biological change during its recent history (e.g. post-industrialisation).

The importance of the conservation of Hawes Water is increased by its proximity to the RSPB reserve of Leighton Moss (SSSI and SPA). Water quality monitoring at Leighton Moss has shown some areas of open water on the reserve to have TP concentrations in excess of 100 $\mu g l^{-1}$ and corresponding problems have been identified, with possible knock on effect to the breeding success of bitterns at the site (Petley-Jones *pers. comm.*). This study has also identified catchment sources as a major concern suggesting that the problem of diffuse pollution is wide spread in the area and should be addressed. Without action, the possibility exists that both of these valuable sites will continue to deteriorate. Because of the proximity of these two sites it is recommended that monitoring be combined to reduce costs and provide better use and understanding of the data.

4.4.2 Summary of Recommendations

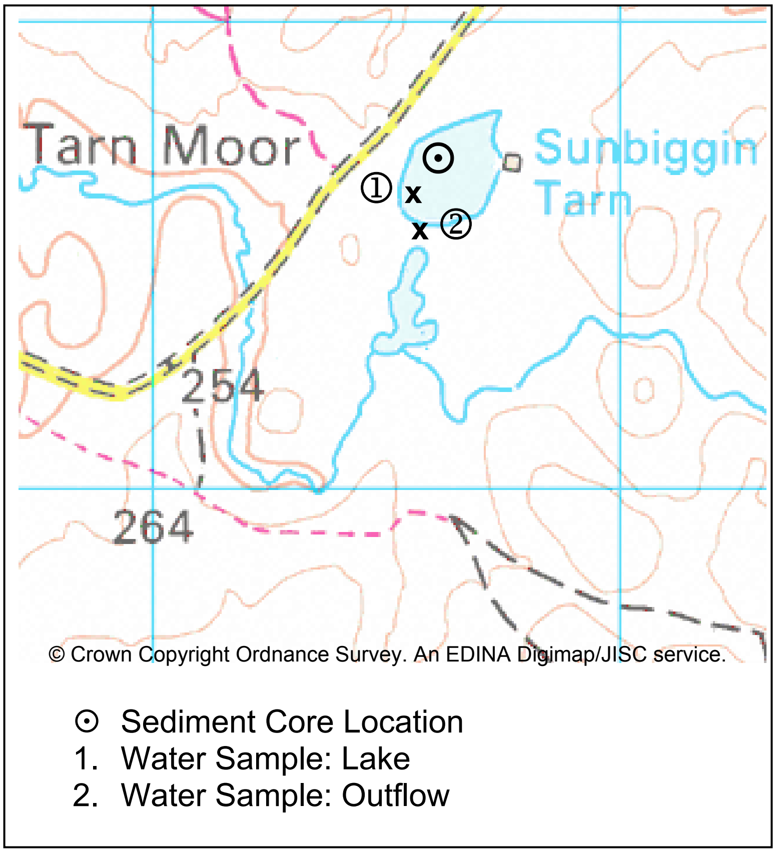
- Liaise with local farmers to assess possible nutrient sources and ensure codes of good agricultural practice are being followed.
- Establish a diffuse pollution management plan for the catchment.
- Identify the exact nature of the Challan Hall sewerage facility and ensure improvements are made if necessary.
- Implement regular chemical monitoring to assess the nature of any changes in the lake and its catchment; a minimum of quarterly sampling is recommended.
- Continue to monitor the aquatic flora of the lake to assess any directional changes; a minimum survey interval of three years is recommended at Hawes Water.
- Carry out multi-proxy palaeoecological studies to assess past biological and environmental changes.
- Combine monitoring with Leighton Moss Reserve to reduce sampling costs and amalgamate data.

5
Sunbiggin Tarn

5.1 Description

Sunbiggin Tarn is a small upland marl lake surrounded by rich fen which, along with Cow Dub Tarn, forms a wetland area of outstanding biological interest. The tarn lies to the east of the English Lake District National Park, five kilometres east of the village of Orton. The catchment is of considerable geological and botanical interest with scars and limestone pavements surrounded by heath, acid grassland and areas of acid and calcareous mire. The site is a SSSI and is of outstanding importance for its range of habitats and for the important flora and fauna they support. Along with Hawes Water (Lancashire) and Malham Tarn (N. Yorkshire) Sunbiggin Tarn is one of only three marl lakes in northern England (EN 1994).

Figure 5.1 Map of Sunbiggin Tarn showing sampling locations



Sunbiggin Tarn is a relatively small water body (3.71 ha) situated within an upland area of Carboniferous limestone. To the north of the tarn, large areas of base rock outcrop on the higher ground but the lower plateaux area of the catchment is of moderate relief and covered by boulder clays and acidic glacial till. The tarn lies at an altitude of 253 m and is approximately 225 m long and 180 m wide. Although much of the basin is relatively shallow (less than 3 m) there is a deeper area towards the western shore which drops to approximately 11 m. Sunbiggin Tarn is the larger of two water bodies on an extensive fen area. The smaller, irregularly shaped pool to the south is known as Cow Dub and is about 200 m long but less than 50 m wide. There is no permanent surface inflow to Sunbiggin Tarn, but springs do rise near the northern shore and it can be assumed that the tarn is almost entirely ground-water fed. The outflow is a narrow channel to the south of the tarn which runs through the fen and feeds into Cow Dub. From Cow Dub the water drains to the west into Tarn Sike and forms the upper reaches of the River Lune.

The catchment of Sunbiggin Tarn covers approximately 123 hectares (Bennion *et al.* 2003), but due to the complex limestone geology this may not be entirely representative of the hydrological sources of the tarn. Catchment land-use is predominantly low intensity, rough grazing, and the vegetation a mixture of acid heathland species and calcareous mire and meadow species. Annual rain fall in the region is approximately 1300 mm yr^{-1} (Fourt 1956).

Table 5.1 Catchment and lake characteristics for Sunbiggin Tarn.

Catchment	
Location	Westmorland, 5 Km E of Orton, Cumbria
Nat. Grid. Ref	NY677077
Altitude	253 m A.O.D
Total Area	123.5 ha
Geology	Carboniferous limestone
Land Use	Rough grazing
Lake	
Maximum depth	11 m
Mean depth	4.48* m
Area	3.7 ha
Volume	166,275* m^3
Retention Time	0.14 years*
Current Water Quality Classification	Mesotrophic/Eutrophic
Monitoring	No continuous monitoring
Nature Conservation Designations	SSSI
	* = Modelled data from EA GB Lakes Inventory (Bennion *et al.* 2003)

5.1.1 Conservation Interest - Reason for Notification

Sunbiggin Tarn is included within a designated SSSI which covers not just the tarn, but also the surrounding wetlands, moors and the exposed areas of bed rock in the north of the catchment (Little Asby Scars). The SSSI covers a total of 997 hectares and is recognised as a site of outstanding importance for its range of habitats and for the flora and fauna they support (EN 1994). In addition to the Ling (*Calluna vulgaris*) dominated moorland, parts of the area have been found to support fine stands of a herb-rich type of heath more typical of the "chalk heaths" of southern England. Due to the combination of calcareous bed rock and acid drift, there are many species of plant found, which do not normally co-occur with heather (EN 1994). The flora of the region has come under extensive scrutiny with detailed records going back to the early 1950's (Holgate 1955). The following information is a summary of the "Reasons for Notification" for SSSI status (EN 1994) and concentrates mainly on the aquatic and wetland habitats.

The open water of Sunbiggin Tarn supports relatively few submerged higher plants, which has been attributed to increased planktonic algal growth fed by nutrients from the droppings of the large black-headed gull colony. However, aquatic species are found in Tarn Sike and the basic flushes, the curled and broadleaved pondweeds (*Potamogeton crispus* and *P. natans*) tend to predominate but other species include opposite-leaved and small pondweeds (*Groenlandia densa* and *Potamogeton berchtoldii*), spiked water-milfoil (*Myriophyllum spicatum*), stonewort (*Chara* sp.), common water-crowfoot (*Ranunculus aquatilis*) and greater bladderwort (*Utricularia vulgaris*).

The margins of the tarn have extensive swamp and fen communities. Common reed (*Phagmites communis*) dominates a large zone from the shore to approximately 1 m water depth. The bottle sedge (*Carex rostrata*) dominates in a more mixed fen along with slender and greater tussock sedge (*Carex lasiocarpa* and *C. paniculata*), common spike-rush (*Eleocharis palustris*), soft rush (*Juncus effusus*), water horsetail (*Equisetum fluviatile*), bog bean (*Menyanthes trifoliata*) and mares tail (*Hippuris vulgaris*). Rarities at the site include tufted sedge (*Carex elata*) which is a very local and mainly an eastern species in Britain and great fen-sedge (*Cladium mariscus*) which is uncommon in the north-west and occurs here at its highest altitude in Britain.

Other areas are dominated by meadowsweet (*Filipendula ulmaria*) or marsh-cinquefoil (*Potentilla palustris*) which occurs with the nationally rare lesser tussock-sedge (*Carex diandra*). Around the fen, numerous calcareous flushes support tufa-forming mosses and many notable calcifuge species. Slender green feather-moss (*Drepanocladus* (*Hamatocaulis*) *vernicosus*) is an Annex II species and is noted as one of the best sites in Britain for this rare moss (JNCC 2002).

In addition to the flora, the calcareous waters are also rich in invertebrates; the flushes support two very rare snail species that are relicts from the last Ice Age (see text box below): Geyer's whorl snail (*Vertigo geyeri*) at one of only very few known British localities and the sandbowl snail (*Catinella arenaria*) which is known only from three other British sites (EN 1994). Sunbiggin Tarn supports a large population of this species in upland calcareous flushes with a rich assemblage of arctic-alpine plants (JNCC, 2001).

Rare Molluscs

Catinella arenaria

This elusive snail is very difficult to find; the animal camouflages its shell by coating itself with mud or faeces while senescent. The population density is very variable depending on season and weather conditions; it usually reaches its peak in August, in a good season the population may 'explode', they are then easier to find. *Catinella arenaria* is known from only three sites in England, including Sunbiggin Tarn and Braunton Burrows in Devon. It was also previously known from Glamorgan, Wales. The management of the Sunbiggin area seems to have suited this species so far. Grazing by cattle and horses prevents scrub growth which would threaten the habitat as would overgrazing, trampling and drainage.

(EN 1994, JNCC 1991)

Vertigo geyeri

This tiny species of snail is an Annex II species under the EU Habitats Directive and Sunbiggin Tarn represents one of only a few highly disjunct populations in the UK. *V. geyeri* flourished in post-glacial environments, but climatic change led to a dramatic contraction of its range, and the species is vulnerable to drainage of the sites where it survives. In addition to calcareous conditions the snails require a dense cover of low-growing grasses and sedges relatively free from *Sphagnum* and other mosses.

(JNCC 2002)

The entire site is of considerable ornithological importance with the extensive areas of calcareous open water and fen, as well as the heathland, providing suitable cover and food for many breeding, passage and wintering bird species. In the past the most obvious has been the nationally important colony of over 12,000 breeding pairs of black-headed gulls around Sunbiggin Tarn (exceeding 1% of the British breeding population). In recent years however, these numbers have declined dramatically (see below). The site is used as a breeding ground by several species of waterfowl, including wigeon, teal, tufted duck, gadwall and mallard. Other breeding species of note include little grebe, sedge warbler and water rail, with the wetlands also being used by lapwing, curlew, redshank and snipe. The area is important for wintering wildfowl and other birds and also seems to be a well used resting place for many species on passage, including goldeneye, pochard, whooper swan, goosander, whimbrel, merlin, peregrine, raven and short-eared owl (EN 1994).

5.1.2 Reasons for Concern - Nutrient Sources

Although Sunbiggin Tarn is currently a site of great conservation importance, there have been recent concerns regarding a possible deterioration in water quality. Very little water chemistry data exist for the site and no reliable records of TP concentrations are recorded in the literature. However, several aquatic vegetation surveys carried out over the past fifty years (Holdgate 1955, Welsh 1982, Stokoe 1983, EN 1994) suggest a possible decline in species richness and this has been attributed to eutrophication (EN 1994). There appear to be no direct anthropogenic inputs of nutrients to Sunbiggin Tarn and grazing is relatively extensive, although some concern exists for possible overgrazing on the common land (JNCC 2002). The most obvious major impact at Sunbiggin Tarn therefore, is thought to have come from the huge breeding colony of black-headed gulls (*Larus ribibundus*) which established at the site (Coulson 1988, EN 1994).

Black-headed gulls at Sunbiggin Tarn

The first record of black-headed gulls at Sunbiggin Tarn was in 1891 by the well-known naturalist Rev. H. A. Macpherson (Robson 1980). Between then and the mid 1980's the records have shown a slow but steady increase in gull numbers. In recent years the colony of breeding gulls has attracted public interest in the tarn, and even before the bird numbers swelled to the 1980's maximum, the site became a very popular attraction with sightseers. In 1961, Robson noted "*The colony of breeding Black-headed gulls attracts many visitors to the tarn. Many go to satisfy a pure interest in natural history, which can be well rewarded by merely seeing the chicks and eggs or watching the behaviour of the parent birds*". The gull numbers appeared relatively stable until the latter quarter of the twentieth century when the population exploded to over 12,500 pairs in 1988 and led to the recognition of the importance of this site for gull conservation, particularly in light of the disappearance of other black-headed gull colonies in the north, e.g. Ravenglass and many sites in Cumbria, Solway, inland Northumberland and as far north as Loch Lomond (Coulson 1988).

Table 5.2 History of black-headed gull numbers at Sunbiggin Tarn

Date	Approx. Gull Numbers	Source
2002	?	No Data
2000	46 pairs	EN notes
1996	1400 pairs	EN notes
1990	9000 pairs	EN notes
1988	12500 pairs	Mark and recapture experiments (Coulson 1988)
1960-80	Increases observed	EN notes
1959	1000 pairs	EN notes
1958	1500 pairs	EN notes
1956-57	1000 pairs	EN notes
1955	800 pairs	EN notes
1949-54	600 pairs	EN Notes
1946	500 pairs	EN notes
1943	500 pairs	Eggs collected by a local farmer for calf fodder (EN)
1938	400-1500 pairs	EN notes
1929	600+ gulls	Local schoolmaster, Mr. T.B. Wright (Robson 1980)
1924	549-600 gulls	Local schoolmaster, Mr. T.B. Wright (Robson 1980)
1917	"Small colony"	Local schoolmaster, Mr. T.B. Wright (Robson 1980)
1891	Gulls present	Rev. H. A. Macpherson (Robson 1980)
1777	?	Gull not mentioned in a natural history paper of the region (Robson 1980)

The reason for the dramatic rise in numbers recorded by Coulson in 1988 is unclear. Improved site management and a reduction in the predation of eggs by man may be factors, but there is no conclusive evidence to support this. What is clear however, is that by 1988 the huge size of the colony was becoming unsustainable. Coulson (1988) recorded weight loss in many birds over the breeding season suggesting there was an increased strain on food resources for the gulls. Furthermore, Coulson found only very few one year old birds on the tarn, which is unusual in gull colonies and led him to conclude that either these birds were not returning to Sunbiggin until one year later due to very high competition for nest-sites and food, or that this observation heralded a future crash or dispersion of the population. It would appear that the latter was the case. By 1990 only 9000 pairs were recorded and the most recent record taken in 2000 found only 46 pairs at the site. The reasons for the sudden decline are, like the gulls appearance, unclear, but with the perceived gull problem now gone the effect of the gulls on Sunbiggin Tarn remains to be established. Certainly while the birds were present, Coulson noted the damage being done to the submerged flora of the tarn. This study hopes to identify what, if any, long-term effects so many birds have had on the tarn and to estimate their contribution to the nutrient load.

Guanotrophy - nutrient enrichment from avian sources

Traditionally one looks at hydrological inputs of nutrients to a water body via lake inflows, point sources and diffuse runoff. At a remote site like Sunbiggin Tarn however, such sources are unlikely to be significant and thus any increase in nutrient loadings must be from an external source. The contribution of birds to lake eutrophication (guanotrophy) has been investigated at a number of UK sites (e.g. Loch Eye (Bailey-Watts 1991), Hickling Broad (Moss & Leah, 1982) and Loch Ussie (May & Gunn 2000)) and where large numbers of birds congregate this has been shown to have an effect. At Hickling Broad in Norfolk, excreta from a winter roost of over 50,000 black-headed gulls contributed to the sediments and was found to account for daily TP concentrations in the lake of over 100 $\mu g\ l^{-1}$ in the summer when it started to decompose at the sediment surface (Moss & Leah, 1982). Thus it is possible that birds can have a significant influence on TP concentrations if they are present in sufficient numbers. Furthermore, because of the ranging feeding behaviour of black-headed gulls, much of the P contribution from gull excreta is sourced from outside the catchment area.

The total contribution of P from the gull colony at Sunbiggin Tarn can only be estimated from existing published figures. Spanns (1971) calculated the daily average P export of one herring gull (*Larus argentatus*) to be 64.8 mg. Herring gulls are approximately four times the weight of black-headed gulls (Iceland Worldwide 2000), and thus a crude assumption is made for this study that the daily P export for one black-headed gull is one quarter that of a herring gull, i.e. 16.2 mg. The gulls are only resident at Sunbiggin Tarn during the breeding season and while resident spend much of their foraging time away from the catchment. A further assumption is therefore made that the gulls only spend 66 percent of their time in the catchment while resident and that they spend three months per year on site. This provides a total P contribution to Sunbiggin Tarn of approximately 976 $mg^{-1} gull^{-1} yr^{-1}$. These figures are used below to estimate total P loadings.

5.2 Aims and Methodology

Four principal aims were identified for investigation at Sunbiggin Tarn. These are outlined below along with a summary of the methods used.

5.2.1 Aim 1:

To estimate the present inputs and outputs of total phosphorus and SRP from catchment use and the black-headed gull colony.

Methodology:

- Quarterly measurements of TP and SRP for the lake and the outflow (Figure 5.1, Table 5.3). A permanent surface inflow is absent and therefore water samples could not be collected.
- Quarterly measurements of nitrate-nitrogen, pH and conductivity for the lake and the outflow.
- Quarterly measurements of chlorophyll *a* concentration for the lake.
- Quarterly measurements of potassium, calcium, iron, sodium and chloride for the lake sample.
- Quarterly flow measurements for the lake outflow.
- Calculation of nutrient export values for gulls and catchment uses.

Table 5.3 Location of sampling points in the Sunbiggin Tarn catchment

Site	Location	Os Grid Ref.
Lake	South-western side of lake. Sampled from beyond reeds.	NY6755,0760
Outflow	South of tarn, channel between tarn and Cow Dub.	NY6768,0755

5.2.2 Aim 2:

To establish the nutrient history of the tarn and to estimate the time scale involved in effecting a meaningful ecological reduction of total phosphorus (TP) in the water column. A target level of 40 $\mu g l^{-1}$ has been set by English Nature.

Methodology:

- Diatom analysis of a dated sediment core from the lake to provide estimates of historical P concentrations (using a transfer function approach) and sediment accumulation rates. See below for detailed methods.

5.2.3 Aim 3:

To survey the present aquatic vegetation

Methodology:

- A macrophyte survey was carried out in August 2002 according to contract specifications.

5.2.4 Aim 4:

To prepare a feasibility study for the restoration of the tarn, assessing the merits of mud pumping in the light of the possibility that natural flushing rates will in time sufficiently reduce the sediment input of P.

Methodology:

- The results from 1, 2 and 3 above, additional lake and catchment studies, literature on the impacts of guanotrophy and information on mud pumping were consulted.

5.3 Results

5.3.1 Water Quality Analysis, Catchment Sources and Gulls

The results for the quarterly chemistry sampling are presented in Table 5.4 as annual mean data based on the four 2002 measurements. The full data-set appears in Appendix I. Attempts to measure flow in the tarn outflow for the purpose of estimating lake retention times, were largely unsuccessful due to a lack of obvious water movements through the fen areas and therefore estimates for lake retention times are based on the modelled catchment and hydrological data from the EA GB Lakes Inventory (Bennion *et al.* 2003).

Table 5.4 Mean annual chemistry (2002) for Sunbiggin Tarn and its outflow.

	Lake	Outflow
pH	7.80	7.86
Conductivity (μScm^{-1})	258	269
NO_3^--N (mgl^{-1})	0.11	0.11
SRP (μgl^{-1})	13.1	12.8
TP (μgl^{-1})	36.4	48.6
Chl *a* (μgl^{-1})	2.8	-
Potassium - K^+ (mgl^{-1})	0.65	-
Calcium - Ca^{2+} (mgl^{-1})	58.45	-
Sodium - Na^+ (mgl^{-1})	4.36	-
Iron - Fe^{3+} (μgl^{-1})	66.6	-
Chloride - Cl^- (mgl^{-1})	7.17	-

Using land cover and stocking data from the EA Lakes Inventory (Bennion *et al.* 2003) the current sources of P can be identified. The inventory can also be used to gain an idea of a target baseline for P at Sunbiggin Tarn using a hind-casting model (Johnes *et al.* 1996) based on historical data for 1931. In addition to the inventory results, estimations of the gull P contribution have also been made. Tables 5.5 and 5.6 outline the EA GB Lake Inventory (Bennion *et al.* 2003) characteristics for Sunbiggin Tarn and Table 5.7 shows the calculated loadings of P based on export coefficients (Johnes *et al.* 1996).

Table 5.5 Catchment and lake characteristics for Sunbiggin Tarn (source: EA Lakes Inventory (Bennion *et al.* 2003).

Catch area (ha)	123.5
Lake surface area (ha)	3.71
Lake:catch ratio	0.03005
Mean depth (m) (modelled)	4.48
Max. depth (m)	11
Total Volume (m^3) (modelled)	166275
Retention time (yrs) (modelled)	0.14
Mean runoff (mm) (CEH data 1995-7)	1044
Stratification class (modelled)	5 (stratified)

Table 5.6 Land use estimates for the Sunbiggin Tarn catchment (source: EA Lakes Inventory (Bennion *et al.* 2003).

Land Use Class	Catchment Cover (%)
Moorland grass	49.90
Meadow/semi-natural	30.62
Bracken	9.82
Open shrub moor	5.57
Dense shrub moor	1.82
Tilled land	0.86
Grass heath	0.56
Upland bog	0.51
Water	0.20
Grazed turf	0.10
Rough grass	0.05

Table 5.7 Modelled phosphorus data for Sunbiggin Tarn, including a hindcast estimate for the site from 1931 using export coefficients (source: EA GB Lakes Inventory (Bennion *et al.* 2003).

Model Component	Model Value
No. Cattle	39.72
No. Sheep	550.41
No. Pigs	0
No. People (1991)	0
Land cover P (kg/yr)	9.96
Cattle P (kg/yr)	8.74
Sheep P (kg/yr)	24.77
Human P (kg/yr)	0
Current P load total (kg/yr)	43.47
Current Modelled P conc. (µg/l)	25.5
Hind-cast P load (kg/yr)(1931) Reading model	29
Hind-cast Lake P conc. (µg/l) Reading model	17

The current P load total shown in Table 5.7 is based only on the coefficients from land cover and animals and therefore ignores the contribution of the gulls to the TP load. Following the disappearance of the gulls this is likely to be a good estimate of present conditions but it would not have been reflective of the P loadings for the tarn at the height of the gull numbers. Using the basic calculations and gull export coefficients outlined above the extra load contributed by 25,000 gulls (P_{gulls} in $kg^{-1}yr^{-1}$) would have been:

$$P_{gulls} = n_{gulls} \text{ x } Pexp_{gulls} \text{ x } 91$$

Where: n_{gulls} is the number of birds
$Pexp_{gulls}$ is the daily export coefficient per gull adjusted for 33% away from site (kg)
91 being an estimated number of days on site per year

$$P_{gulls} = 25000 \times 10.692 \times 10^{-6} \times 91 = 24.32\ kg^{-1}yr^{-1}$$

$$P_{gulls} + P_{catchment} = 67.79\ kg^{-1}yr^{-1}$$

Thus the total annual P load during the mid-eighties is likely to have been in the region of 68 kg yr^{-1}. Using the modelled volume and retention time for the tarn this would return an annual TP concentration of approximately 40 μgl^{-1}. It should be stressed that this value is only a crude estimate due to the assumptions mentioned above, but it does appear to be indicative of the present conditions in the tarn where the annual mean TP for 2002 was measured at 36.4 μgl^{-1}. Without the gull input the modern mean annual TP concentration is estimated at 25.5 μgl^{-1}. One area of the model which may cause a further error is the modelled retention time. The estimated value of 0.14 yrs (51 days) appears rather low and thus the modelled TP values might be expected to be even higher if the tarn was less well flushed than suggested by the modelled data. Without better hydrological data this cannot be corrected.

5.3.2 Sediment Core Study

5.3.2.1 Methods

Sediment Coring
A sediment core was taken by colleagues from the University of Liverpool on 22-May-2000 from the deep basin towards the north-western side of the lake in a water depth of 8.5 m (Figure 5.1), using a mini-Mackereth piston corer. This type of coring device is able to retrieve sediment cores of between 50 to 100 cm in length which, in a lake of this type, is expected to represent the last 100 years or more.

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

Diatom Analysis
Diatoms (*Bacillariophyceae*: unicellular, siliceous algae) were selected as the most appropriate microfossil group for inferring environmental trends in the tarn. Diatoms are good indicators of past nutrient concentrations and can also be used for inferring changes in available lake habitats.

Seventeen levels from the sediment core were prepared and analysed for diatoms using standard techniques (Battarbee, 1986). The levels were taken at 2 cm intervals from 0-10 cm of the core and at 5 cm intervals thereafter to the core base. At least 300 valves were counted from each sample using a Leitz research microscope with a 100 x oil immersion objective and phase contrast. Principal floras used in identification were Krammer & Lange-Bertalot (1986, 1988, 1991a, b). All slides are archived at the ECRC. The data were expressed as percentage relative abundance. Information on the life-form preference of each taxon was obtained from both the literature and personal observations in order to describe each of the common species as either predominantly epiphytic (associated with plant substrates), benthic (associated with sediment substrates) or planktonic (living in the open water). Cluster analysis was performed on the percentage diatom data to facilitate description by zones, using CONISS (Grimm, 1987), implemented by TILIA and TILIAGRAPH (Grimm, 1991). CONISS is a program for stratigraphically constrained cluster analysis by the method of incremental sum of squares.

Diatom Transfer Functions
A quantitative approach to environmental reconstruction has been developed based on a predictive equation, or transfer function, that models the relationship between

diatom assemblage composition and lake-water chemistry. The transfer function is generated using a calibration, or training, data set of modern surface-sediment diatom samples and contemporary water chemistry data from a large number of lakes spanning the environmental gradient of interest. Once calibrated, it is then applied to the sediment-core fossil diatom assemblages to provide quantitative inferences of past water chemistry.

This approach, using the method of weighted averaging (WA) regression and calibration (e.g. ter Braak & van Dam, 1989), has been used successfully to provide quantitative reconstructions of a range of hydro-chemical variables, including total phosphorus (TP) (e.g. Bennion *et al*., 1996). Modern diatom TP optima are calculated for each diatom taxon based on their distribution in the training set, and then past TP concentrations are derived from the weighted average of the optima of all diatoms present in a given fossil sample. These transfer functions are able to provide estimates of baseline TP concentrations in lakes, and coupled with radiometric dating of sediment cores, enable the timing, trends and rates of enrichment to be assessed for a particular site.

A diatom-TP transfer function has been generated from 152 relatively small, shallow (< 10 m maximum depth), productive lakes in six regions of Northwest Europe (south-east England, the Cheshire and Shropshire meres, Northern Ireland, Denmark, Sweden and Wales) which is able to reconstruct epilimnetic TP concentrations with reasonable accuracy (Bennion *et al*., 1996). Annual mean TP concentrations in the training set range from 5-1200 μg TP l^{-1}, with a median value for the dataset of 104 μg TP l^{-1}. The transfer function was developed using the method of WA partial least squares (WA-PLS) which is an extension of WA that uses the residual correlation in the diatom data to improve the predictive power of the WA regression coefficients (ter Braak & Juggins, 1993). This is done through the selection of a small number of components, the optimum number of components being estimated by jack-knifing cross-validation. The optimum number of components in the TP model used here was two (see Bennion *et al*., 1996 for further details). However the two component model (WA-PLS2) only slightly improves on the one component model (WA-PLS1) (which equates to simple WA) and, therefore, both are applied in this study. The one and two component models were applied to the core data following taxonomic harmonization between the training set and core species data. The TP data used in the models were $\log_{10}$-transformed annual mean concentrations. The reconstructions were implemented using CALIBRATE (Juggins & ter Braak, 1993).

The performance statistics of the models are shown in Table 5.8. The strength of the relationship between diatom-inferred TP (DI-TP) and measured values is described by the coefficient of determination known as r^2 (0 = no fit; 1 = perfect fit). The errors of the models are described by the root mean square error (RMSE) which essentially summarises the difference between the measured values for the training set of lakes and the diatom inferred values generated by the model. These are calculated based on the original training set (the apparent RMSE) and more realistically on a cross-validated test set (the RMSE of prediction or RMSEP). The lower the error, the better the model performs. Table 5.8 shows that both models perform well and have relatively low errors of prediction.

Table 5.8 Summary statistics of the diatom models for reconstructing TP

	WA-PLS1	*WA-PLS2*
Number of lakes	152	152
Number of diatom taxa	298	298
Range	5-1190 μg TP l^{-1}	5-1190 μg TP l^{-1}
Median	104 μg TP l^{-1}	104 μg TP l^{-1}
Apparent r^2	0.85	0.91
Apparent RMSE	0.19 log_{10} μg TP l^{-1}	0.15 log_{10} μg TP l^{-1}
Predicted r^2	0.80	0.82
Predicted RMSEP(jack)	0.22 log_{10} μg TP l^{-1}	0.21 log_{10} μg TP l^{-1}

5.3.2.2 Results

Lithostratigraphy and Dating

The mini-Mackereth core (SUNB1) was 67 cm in length and had a high marl content ranging from approximately 75% in the lowermost part of the core (below 50 cm) to around 50% in the upper section. High calcium carbonate content is typical of sediments accumulating in a marl system. There were no other unusual features or notable stratigraphic changes.

Lead-210 Activity

Total ^{210}Pb activities were very low, particularly in the top 20 cm of the core where they were scarcely in excess of the supporting ^{226}Ra (Figure 5.2a). Unsupported ^{210}Pb activity (Figure 5.2b) varied irregularly with depth and the generally higher values in the lower half of the core suggest that the base of the core at 67 cm is well short of the ^{210}Pb equilibrium depth.

Artificial Fallout Radionuclides

^{137}Cs activities (Figure 5.2c) increased steadily with depth in the lower half of the core, reaching a maximum value just above the core base. Since this peak almost certainly records the 1963 fallout maximum from the atmospheric testing of nuclear weapons it appears that the entire core only spans a period of about 40 years.

Core Chronology

Figure 5.3 shows CRS model ^{210}Pb dates (Appleby & Oldfield, 1978) calculated using the 1963 depth determined from the ^{137}Cs record as a reference point. In view of the abbreviated ^{210}Pb record, ^{210}Pb dates calculated using the CRS model alone were relatively meaningless. The results suggest that sedimentation rates have varied dramatically during the past 40 years, with episodes of very rapid accumulation superimposed on a baseline value of 0.28 g cm^{-2} yr^{-1} (~1 cm yr^{-1}). The first of these episodes was a very brief event c.1980, but episodes during the past decade appear to have been more sustained and are presumably associated with events giving rise to the layer of dense sediment in the top 10 cm of the core. A detailed chronology based on these results is given in Table 5.10.

The ^{210}Pb flux to the core is calculated to be 149 ± 17 Bq m^{-2} yr^{-1}. This is a little higher than the atmospheric flux but not inordinately so. The ^{137}Cs inventory (10260 ± 94 Bq m^{-2}) is however abnormally high, suggesting significant transport from the catchment. One possible explanation for the high $^{137}Cs/^{210}Pb$ inventory ratio is greater mobility of ^{137}Cs in a catchment with relatively acid peaty soils.

Table 5.9 Fallout radionuclide concentrations in Sunbiggin Tarn sediment core SUNB1

Depth		^{210}Pb						^{137}Cs	
		Total		Unsupported		Supported			
cm	g cm^{-2}	Bq kg^{-1}	±	Bq kg^{-1}	±	Bq kg^{-1}	±	Bq kg^{-1}	±
0.5	0.2	29.4	6.6	3.2	6.8	26.2	1.7	20.7	1.6
4.5	2.6	32.6	9.1	8.4	9.3	24.2	2.0	28.3	2.0
8.5	4.3	36.7	9.0	7.5	9.2	29.2	2.1	29.9	2.4
12.5	5.7	50.1	16.2	7.9	16.6	42.2	4.0	30.2	4.2
16.5	6.9	38.3	7.8	8.3	8.1	30.0	2.0	36.4	2.0
18.5	7.6	33.4	10.1	4.8	10.5	28.6	2.9	31.3	2.2
20.5	8.2	47.5	8.7	20.3	9.0	27.2	2.2	31.4	1.9
24.5	9.5	47.4	7.9	20.9	8.3	26.5	2.3	28.0	1.8
28.5	11.0	29.4	6.6	3.4	7.8	26.0	4.2	26.5	3.8
32.5	12.2	69.9	10.4	44.4	10.8	25.5	2.6	44.3	2.4
36.5	13.4	50.3	11.0	20.5	11.5	29.9	3.1	50.4	2.9
40.5	14.5	52.1	10.4	31.1	10.6	21.0	1.9	43.4	2.1
44.5	15.6	51.5	9.0	25.6	9.3	26.0	2.3	57.5	2.8
48.5	16.8	42.0	12.4	3.7	12.9	38.3	3.5	73.2	3.5
53.5	18.2	60.2	13.4	26.0	13.9	34.1	3.8	75.4	3.6
56.5	19.0	51.7	13.5	27.2	13.7	24.5	2.4	71.9	2.8
60.5	20.2	51.0	9.8	19.6	10.1	31.4	2.5	90.4	2.7
64.5	21.4	42.7	9.9	24.1	10.1	18.6	1.7	80.5	2.4
66.5	22.1	53.1	7.3	27.5	7.5	25.6	1.9	86.8	2.4

Table 5.10 ^{210}Pb chronology of Sunbiggin Tarn sediment core SUNB1

Depth		Chronology			Sedimentation Rate		
		Date	Age				
cm	g cm^{-2}	AD	yr	±	g cm^{-2} yr^{-1}	cm yr^{-1}	± (%)
0.0	0.0	2000	0	0			
4.5	2.6	1999	1	1	2.04	4.5	88.1
8.5	4.3	1998	2	2	2.00	4.6	81.0
12.5	5.7	1997	3	2	1.79	5.3	71.7
16.5	6.9	1997	3	3	1.67	5.2	98.3
20.5	8.2	1996	4	3	0.64	2.6	99.9
24.5	9.5	1994	6	3	0.58	2.3	77.0
28.5	11.0	1992	8	3	1.92	2.9	58.4
32.5	12.2	1991	9	3	0.25	1.6	53.2
36.5	13.4	1987	13	4	0.49	1.2	42.5
40.5	14.5	1984	16	4	0.29	1.2	35.7
44.5	15.6	1981	19	4	0.32	1.5	38.1
48.5	16.8	1979	21	4	1.70	2.3	39.7
52.5	17.9	1977	23	4	0.56	1.5	51.3
56.5	19.0	1974	26	4	0.24	1.0	50.9
60.5	20.2	1969	31	4	0.29	0.8	52.0
64.5	21.4	1964	36	5	0.20	0.6	41.9
66.5	22.1	1960	40	5	0.16	0.5	28.1

Figure 5.2 Fallout radionuclides in the Sunbiggin Tarn sediment core SUNB1 showing (a) total and supported ^{210}Pb, (b) unsupported ^{210}Pb, (c) ^{137}Cs concentrations versus depth.

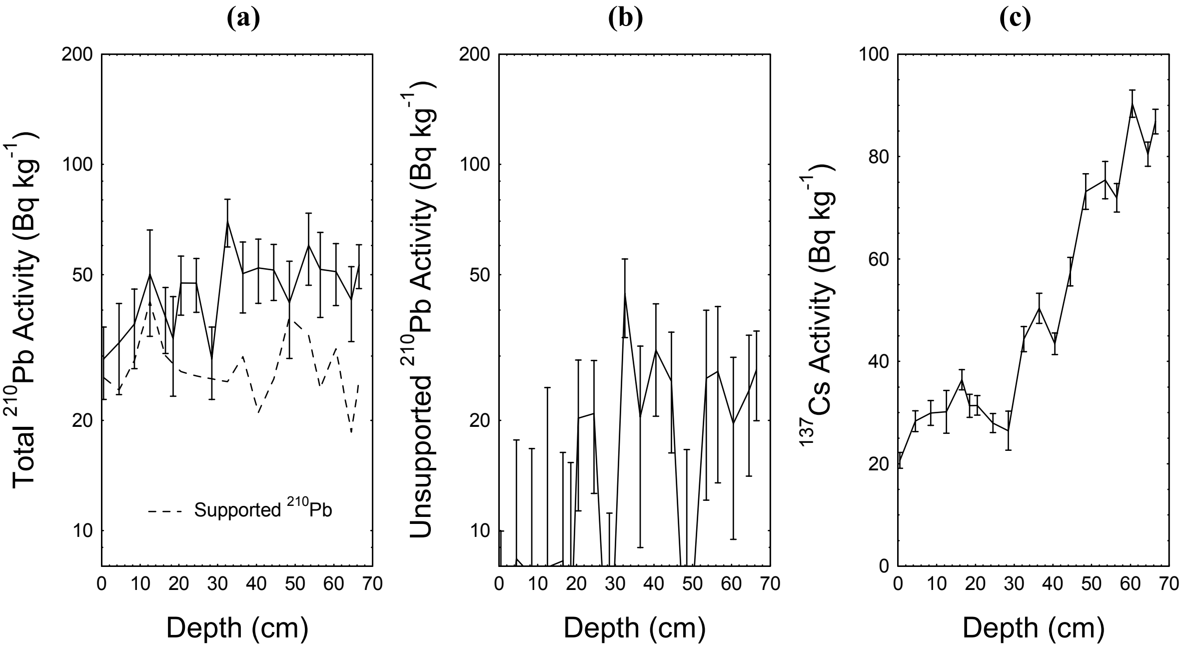
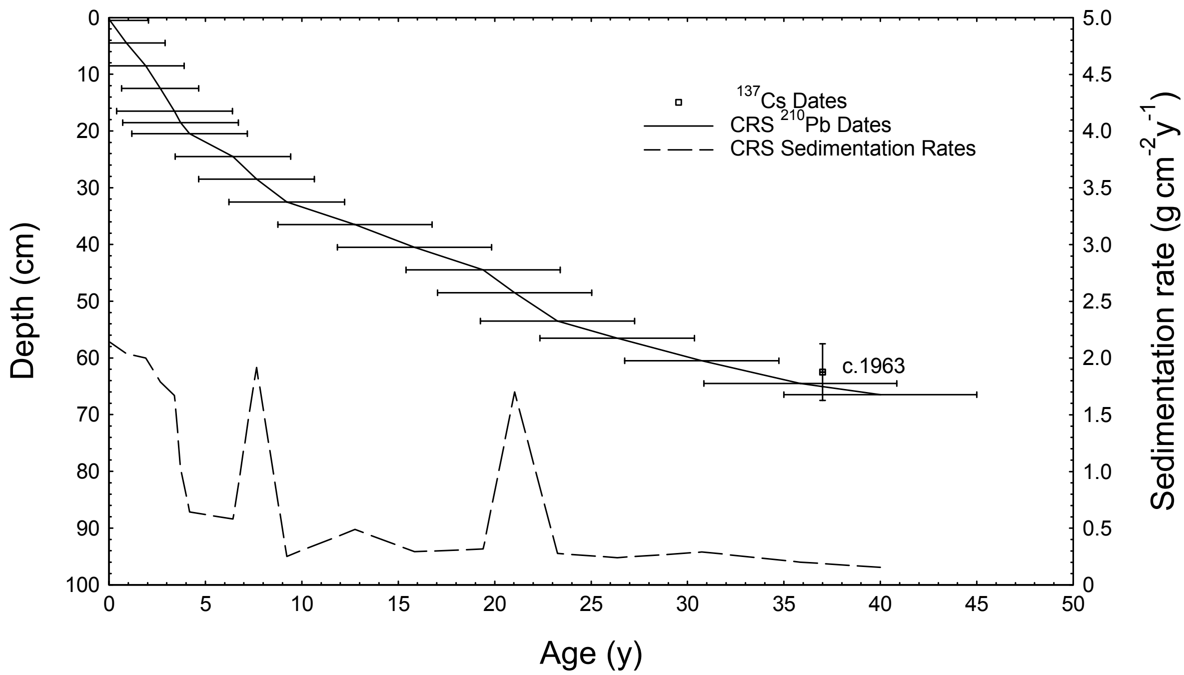


Figure 5.3 Radiometric chronology of Sunbiggin Tarn sediment core SUNB1 showing the CRS model ^{210}Pb dates and sedimentation rates and the 1963 depth determined from the ^{137}Cs stratigraphy.



Diatom Stratigraphy

The seventeen stratigraphic levels selected for diatom analysis from the core, SUNB1, were 0-1, 2-3, 4-5, 6-7, 8-9, 10-11, 15-16, 20-21, 25-26, 30-31,35-36, 40-41, 45-46, 50-51, 55-56, 60-61, 65-66 cm. Preservation was good throughout the core. The fossil data contained 63 diatom taxa, only 30 of which were present in relative abundances of > 1% in at least two samples. A full taxa list with authorities is given in the Appendix Table II and the percentage data for each sample for the 30 main taxa are given in the Appendix Table III. A summary diatom diagram is shown in Figure 5.5.

The diatom assemblages were dominated by non-planktonic taxa throughout the core with planktonic taxa always comprising less than 5% of the total assemblage. The diatom assemblages had low diversity; the number of taxa observed in the samples ranging from 22-34 and diversity was similar throughout the core. The diatom record exhibited gradual changes with shifts in the relative abundances of the common species but there were no major losses or new appearances of taxa. Using the chronology given in Table 5.10, sediment accumulation rates have been rapid and the whole core represents only the last forty years (c. 1960-2000 AD). The cluster analysis defined two major diatom assemblage zones.

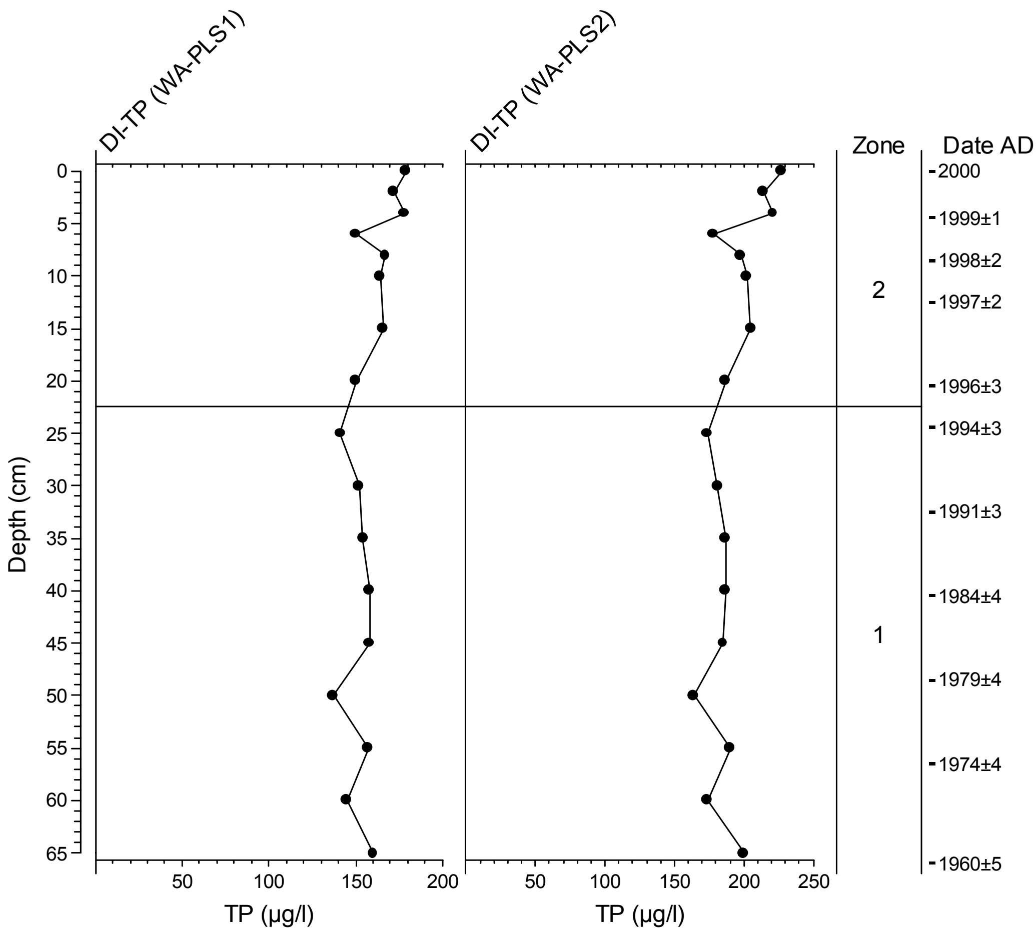
Zone 1 (66-22 cm; c. 1960-1995) was characterised by a largely benthic *Fragilaria* flora including *F. pinnata*, *F. brevistriata*, *F. construens* and *F. construens* var. *venter* which comprised 60-70% of the total assemblage. *Amphora pediculus* was also important in this zone with relative abundances of around 10% and *Cocconeis pediculus* was present at abundances of around 5% in the lowermost section below 50 cm. *Achnanthes minutissima* occurred in relatively low numbers but was at its highest relative abundances in this zone (3-5%). Only one planktonic diatom, *Stephanodiscus parvus*, occurred in relative abundances of > 1%.

Zone 2 (22 - 0 cm; 1995-2000) was characterised by a marked increase in *Amphora pediculus* to a maximum relative abundance of approximately 40%, at the expense of the benthic *Fragilaria* taxa which declined to around 20-30% of the total assemblage. There were notable increases in a number of *Navicula* taxa including *N. utermoehlii, N. graciloides, N. subrotundata, N. recens* and *N. minima.* Three *Achnanthes* taxa, *A. lanceolata, A. conspicua* and *A. ziegleri* also increased in relative abundances in this zone, as did *Cymbella reichardtii*.

Total Phosphorus Reconstruction

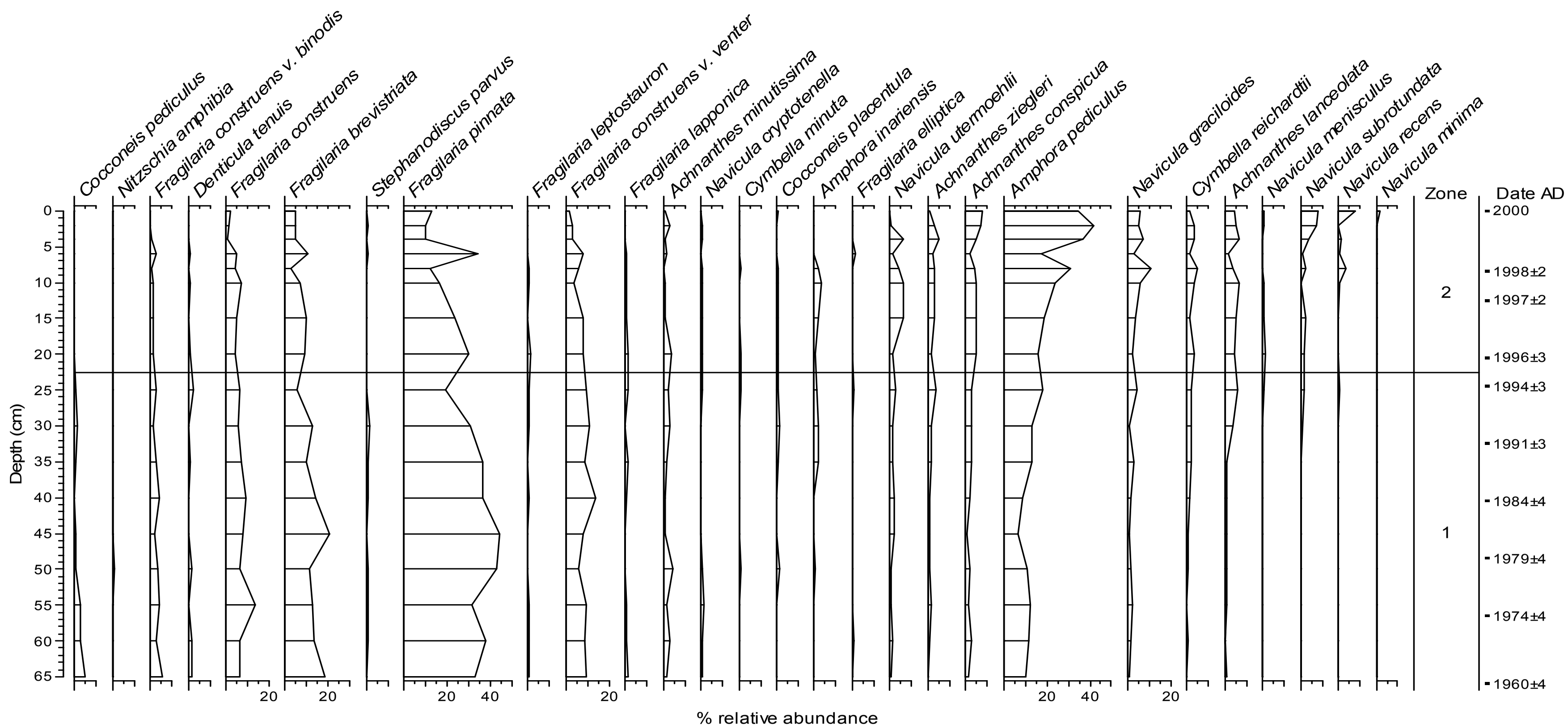
Of the 63 taxa observed in the diatom record, 45 were present in the training set. Over 90% of the fossil assemblage was represented by the training set in most samples and, therefore, there were no major analogue problems when applying the transfer functions. However, four taxa that were relatively abundant in the upper samples, *Navicula utermoehlii*, *Achnanthes ziegleri*, *Cymbella reichardtii* and *Navicula recens*, were not present in the training set and, therefore, analogues for these samples were slightly lower at around 80-85%. The DI-TP reconstruction (Figure 5.4) using both the WA-PLS1 and WA-PLS2 models followed the same trend although values produced by the latter were slightly higher. In Zone 1, DI-TP values fluctuated but displayed no clear trend and the WA-PLS1 and WA-PLS2 models gave values of approximately 140-160 μg TP l^{-1} and 170-200 μg TP l^{-1}, respectively. In Zone 2, DI-TP values displayed a general increase to maximum values at the core surface of c. 180 μg TP l^{-1} and 225 μg TP l^{-1} for WA-PLS1 and WA-PLS2, respectively. There was, however, a single lower value for the sample at 6-7 cm owing to the relatively high percentage of *Fragilaria pinnata*.

Figure 5.4 Diatom-inferred total phosphorus reconstruction for Sunbiggin Tarn



The DI-TP values were always in excess of 130 µg l^{-1} whereas measured TP during our water sampling programme never exceeded 70 µg TP l^{-1} and the annual mean TP for 2002 was approximately 40 µg TP l^{-1}. The modelled concentrations were, therefore, considerably higher than the monitored values.

Figure 5.5 Summary diatom diagram for the Sunbiggin Tarn core (showing all taxa >1% in at least 2 samples)



5.3.2.3 Discussion

All of the common taxa in the diatom assemblages are typically associated with shallow, relatively nutrient-rich, freshwater alkaline lakes and have been observed in many other shallow, productive, temperate waterbodies. The diatom flora was dominated by non-planktonic taxa throughout the core reflecting the shallow, plant-rich nature of the waterbody and indicating that the lake has probably always been a clear-water, plant dominated system with no evidence of turbid, phytoplankton dominated phases. There were remarkably few planktonic species, which appears to reflect the "mesotrophic" nature of the tarn. Almost all of the water entering the tarn is base-rich but is likely not to be excessively high in nutrients given that it is derived from calcareous springs. Whilst the surrounding moorland is used predominantly for sheep grazing and to a lesser extent for cattle, the estimated external nutrient load is relatively low (43 kg yr^{-1} according to Bennion *et al*. 2003) and appears to be insufficient to support large populations of planktonic diatoms that require high nutrient concentrations for growth. Indeed during our water sampling programme, chlorophyll *a* concentrations in the tarn were always low (< 8 µg l^{-1}) and during the macrophyte survey in August 2002 light conditions were reasonably good with a Secchi depth of 2.2 m.

There were, however, a number of marked changes in the core. One of the most striking changes was the relative increase in *Amphora pediculus* and the decrease in the benthic *Fragilaria* taxa from the mid-1990s. Furthermore, in Zone 2 of the core, a number of small *Navicula* and *Achnanthes* taxa, as well as *Cymbella reichardtii*, were present in higher amounts than had been previously observed in the record. When the diatom-TP transfer function was applied, these species shifts resulted in an increase in DI-TP values, suggesting that TP concentrations in the tarn rose over the period from around 1995 to 2000 by approximately 30-40 µg TP l^{-1}. It is important to note that the changes in the DI-TP values are within the error of the models and, therefore, must be treated with caution.

A number of authors (e.g. Anderson *et al*., 1993; Bennion, 1995; Bennion & Appleby, 1999; Sayer, 2001; Bennion *et al.* 2001) have discussed potential problems in applying diatom-TP inference models to shallow lakes where non-planktonic taxa dominate the diatom assemblages. Changes in planktonic diatom assemblages in sediment cores are often relatively easy to explain because the taxa are directly affected by conditions in the water column. Shifts, however, in diatom assemblages comprised principally of non-planktonic taxa, such as those in the Sunbiggin Tarn profile, are much more difficult to interpret. One difficulty is that there are numerous factors such as light, substrate and top-down effects that can control periphytic (attached) algae and, therefore, species shifts may not simply reflect changes in lake water chemistry. Secondly, there are uncertainties over the ecological affinities of most periphytic taxa which cause real problems in interpreting palaeolimnological records (e.g. Stoermer *et al*., 1992; Bennion *et al*., 2001). For instance, *Fragilaria* spp. can be found in epiphytic, epipelic, epilithic or epipsammic communities (Round, 1981; Sayer, 2001) and have been observed in a wide range of aquatic environments. In Sunbiggin Tarn, given the relatively good light conditions, it is most likely that *Fragilaria* spp. would have been growing *in-situ* on the sediment surface. A further problem occurs when taxa exhibit a wide tolerance to nutrient concentrations, making them poor indicators of lake trophic status. The distribution of the *Fragilaria* spp., in particular, along the whole length of the TP gradient in the northwest European training set illustrates that they have wide ecological tolerances and that their TP optima essentially lie in the centre of the sampled environmental gradient. The training set used here spans a long TP gradient of c. 5 to c. 1000 μg TP l^{-1} with a dataset median of 104 μg TP l^{-1} (Bennion *et al*., 1996) and thus the TP optima for these taxa tend to be approximately at, or greater than, 100 μg TP l^{-1}, often leading to over-estimation of DI-TP in samples where they dominate. This explains why the modelled concentrations were considerably higher than the monitored values for Sunbiggin Tarn.

Available ecological information on the major taxa in the Sunbiggin Tarn core were collated from a number of key studies and are presented in Table 5.11. The table shows the TP optima from the Bennion *et al*. (1996) transfer functions applied in this study. The *Fragilaria* taxa that dominate the early part of the core all have similar TP optima of c. 120-150 μg TP l^{-1} which is indeed also similar to that of *Amphora pediculus*, the dominant taxon in the upper part of the core. A number of taxa that increase towards the top of the core, however, have slightly higher TP optima of c. 160-200 μg TP l^{-1}. This explains the inferred increase in TP in recent years. The study of epilithic diatoms by King *et al*. (2000) also demonstrated that the *Fragilaria* taxa have similar TP optima both to each other and to *Amphora pediculus*. Kelly & Whitton (1995) assigned sensitivity values of between 1 and 5 to epilithic diatoms in UK rivers where 1 indicates taxa most abundant in nutrient poor conditions [soluble reactive P (SRP) < 10 μg l^{-1}] through to 5 which is given to taxa found to be most abundant in highly nutrient rich waters (SRP > 300 μg l^{-1}). These preferences were used to develop a Trophic Diatom Index (TDI) for monitoring eutrophication in rivers. The values agree to some extent with the Bennion *et al.* (1996) diatom TP optima in that the *Fragilaria* taxa tend to prefer lower P concentrations than the small *Navicula* taxa and *Achnanthes lanceolata* (see Table 5.11). Kelly & Whitton (1995) observed that many of the small *Navicula* taxa were tolerant of organic pollution in rivers and thus their increase in the top of the Sunbiggin Tarn core could be interpreted as an increase in organic pollution.

Trophic classifications based on two other studies (van Dam *et al*., 1994; Fore & Grafe, 2002) which have attempted to ascribe ecological indicator values to diatoms are also shown in Table 5.11. These suggest that the *Fragilaria* taxa generally require high oxygen concentrations but that some species have a broad tolerance to nutrient levels whilst others have a preference for moderate to high nutrient concentrations. Similarly some of the taxa that become important in the upper part of the Sunbiggin Tarn diatom record are classified as eutrophic requiring only moderate oxygen (e.g. *Achnanthes lanceolata*) whilst others require high oxygen (e.g. *Navicula graciloides*) or prefer mesotrophic conditions (e.g. *Navicula utermoehlii*). These examples serve to illustrate the wide range of classification methods and the often conflicting ecological indicator values derived from different schemes. In the absence of supporting documentary evidence or other long-term chemical and ecological data, it is therefore difficult to interpret the diatom record in terms of changes in water quality with any confidence.

Table 5.11 Comparison of ecological information from a number of studies for the main diatom taxa in the Sunbiggin Tarn core

Taxon name	Bennion *et al.* (1996) TP optimum µg l^{-1}	King *et al.* (2000) TP optimum ln(x+1) µg l^{-1}	Kelly & Whitton (1995) Sensitivity value in TDI	Van Dam et al. (1994)	Fore & Grafe (2002)
Fragilaria construens	121	2.89	2	High oxygen; Meso-eutrophic	High oxygen
Fragilaria brevistriata	136	2.74	2	High oxygen; Broad trophic tolerance	High oxygen
Fragilaria pinnata	142	2.85	4	High oxygen; Broad trophic tolerance	High oxygen
Fragilaria construens var. *venter*	94	na	2	High oxygen; Meso-eutrophic	High oxygen
Amphora pediculus	147	2.84	5	Fairly high oxygen; Eutrophic	Eutrophic
Achnanthes lanceolata	198	2.60	5	Moderate oxygen; Eutrophic	Eutrophic
Achnanthes conspicua	182	na	3	Fairly high oxygen; Broad trophic tolerance	na
Navicula graciloides	103	na	4	High oxygen; Eutrophic	na
Navicula utermoehlii	na	na	5	Mesotrophic	na
Navicula subrotundata	164	na	5	Na	na
Navicula recens	na	Na	4	Eutrophic	na
Cymbella reichardtii	na	Na	2	Na	High oxygen

The shift from a predominantly benthic *Fragilaria* to an *Amphora-Navicula-Achnanthes* community could be viewed as an indication of changes in habitat availability in the tarn, or disturbance rather than a direct response to enhanced nutrient concentrations. *Fragilaria* spp. commonly grow in long chains, loosely attached to reed stems or on the surface of consolidated sediments (Hughes 2002). This can be seen in *situ* as a "woolly" fringe around the submerged base of reeds. *Amphora pediculus* and *Achnanthes* spp. are firmly attached taxa forming a tight bond to reed stems (and other hard substrates). This microscopic film is often likened to a forest, with long chain forming taxa like *Fragilaria* akin to the tall trees and small attached species like *Achnanthes* and *Amphora pediculus*, the low growing under-storey (Round 1993). In a high disturbance environment the loosely attached canopy species (*Fragilaria*) would be dislodged leaving the firmly "rooted" under-storey to flourish in the increased light. Small *Navicula* spp. are motile and live within loose, mobile sediments and are considered as being indicative of organically enriched areas (Kelly & Whitton 1995). Again the increase in these taxa suggest an environment where sediments are regularly disturbed and organic enrichment is present (avian faecal material?). The observed shifts in the diatom communities may well have been due to the gulls, but this is thought to be more likely a result of disturbance rather than a direct effect of any change in trophic status.

Any changes in the structure of the plant community could explain some of the species shifts as the nature of the plant substrate can influence diatom composition (e.g. Blindow, 1987; Otten & Willemse, 1988; Goldsborough & Hickman, 1991; Shamsudin & Sleigh, 1995; Hughes, 2002). An increase in submerged macrophytes provides a greater surface area on which diatoms can grow, and in particular this will favour epiphytic species such as *Achnanthes* spp. The results of the macrophyte survey for this study (see section 5.3.3 below) indicate that there has been an increase in the overall macrophyte cover during the past two decades and thus this may explain the increase in *Achnanthes* spp. towards the top of the core. Hence, changes in nutrient concentrations may affect the diatom communities indirectly by altering the structure of the plant community on which they grow.

The slight enrichment inferred from the diatom record appears initially to support the general concerns over enrichment of the tarn. Despite the increase in plant biomass there has been an apparent deterioration in the species richness of the submerged macrophyte communities in recent decades, especially charophytes which are known to perform badly in nutrient-rich waters (e.g. Haycock & Duigan, 1994, van den Berg, 1999, van Nes *et al.* 2002), and a large black-headed gull colony is thought to be potentially the main factor giving rise to these conditions. However, the timing of the enrichment is not as expected if this were so. The period of greatest increase in black-headed gull numbers was from the 1950s to the late 1980s when numbers increased from 500 pairs in 1950 to around 12,500 pairs in 1988, yet there was very little change in the diatom assemblages in the core over this time. The diatom species shifts and consequent DI-TP increase did not occur until the mid 1990s, the period during which gull numbers have declined, with 1400 pairs in 1996 having decreased to only 46 pairs by 2000. It is difficult to explain the mismatch between the timing of the diatom inferred enrichment and that expected from the gull data with any certainty. Thus, here we propose a number of possible reasons for the observations:

i) The chronology of the sediment core may be incorrect. The dates assigned to the core are based on the ^{210}Pb and ^{137}Cs inventories both of which were rather unusual in the Sunbiggin Tarn core. Whilst some bank-side erosion is likely owing to the exposed nature of the site and consequent wave action, the estimated sediment accumulation rates based on this chronology were unexpectedly high for an upland marl system, with episodes of very rapid accumulation superimposed on a baseline value of around 1 cm yr^{-1}. An independent method of dating is required to ascertain whether the radiometric dates are reliable. It is interesting to note that the diatom changes observed in this study were largely the same as those seen in another core from the tarn (Thorne 2001). Even allowing for some misidentifications by the student, Thorne (2001) observed the same relative shift from a *Fragilaria* dominated flora to *Amphora-Achnanthes-Navicula* dominated assemblages. Unfortunately the student core was not dated but the strong similarity of the two profiles suggests that the sediment record is in tact.

ii) The DI-TP reconstruction may be misleading and the recent diatom shifts may be due to factors other than enrichment. Given the problems associated with application of transfer functions to non-planktonic diatom sequences discussed above, it is possible that the DI-TP values are unreliable and that the species shifts are controlled by factors such as light, physical disturbance, substrate and grazing.

iii) High black-headed gull numbers throughout the 1960s to 1980s are likely to have increased the internal P load locked up in the sediments (an additional input of up to 24.32 $kg^{-1}yr^{-1}$ of P). It is possible that the lake has been slow to respond and that this load is only now manifesting itself as a problem in the tarn. However, this explanation is not supported by a preliminary analysis of the geochemical P profile (i.e. P concentrations in the sediment itself) as there are no clear changes in concentrations over time (J. Boyle, personal communication). Further chemical studies of the sediment are required to characterise the internal loading history more fully.

iv) Black-headed gulls may not be the main factor giving rise to the recent enrichment. It is possible that land use or land management changes in the catchment are responsible for the inferred enrichment. A strong odour of slurry was noted during August 2002 and thus diffuse agricultural sources of P may be entering the lake. The export-coefficient model of Johnes *et al*. (1996) was applied to Sunbiggin Tarn and gave a modelled hind-cast P load of 29 kg yr^{-1} for the year 1931 compared to a current P load of 43 kg yr^{-1} modelled using data in the Great Britain Lakes Inventory (Bennion *et al.,* 2003). In terms of in-lake TP concentrations, the hind-cast TP value (1931) for Sunbiggin Tarn is 17 µg TP l^{-1} compared to the current measured TP of 36 µg TP l^{-1}. These calculations are based on P contributions modelled from land cover, cattle and sheep and do not include the P load from birds. The data suggest a 50% increase in P load from the catchment and a doubling of in-lake P concentrations since the 1930s, and thus the catchment may be a larger contributor of P to the lake than previously thought.

5.3.2.4 Summary of the Sediment Core Analysis

Interpretation of the diatom species shifts in the core was difficult owing to the uncertainties regarding the factors which determine the composition of non-planktonic communities. Nevertheless, we propose changes in the nutrient concentrations, physical disturbance, shifts in habitat availability and plant community structure as possible explanations. The DI-TP reconstructions suggest slight enrichment since the mid-1990s but this post-dates the period during which large numbers of black-headed gulls were present on the tarn. A number of possible explanations for the mismatch between the timing of the diatom inferred enrichment and that expected from the wildfowl data are proposed including uncertainties surrounding the core chronology and the diatom reconstructions, delayed lake response to the nutrient load from the gulls and alternative catchment sources of P to the lake.

Complexities in the relationship between nutrients and ecological change in shallow lakes reduce the ability of simple transfer functions to reconstruct the nutrient history of systems such as Sunbiggin Tarn. Given the uncertainties surrounding the modelled data, the TP reconstructions should not be taken as clear evidence of enrichment. The radiometric data were also somewhat problematic in that the profiles were rather irregular. An independent set of dates would be useful to assess the reliability of the chronology generated here. Assuming that the estimated sediment accumulation rates are correct, a marked increase in sedimentation rates to values as high as 5 cm yr^{-1} occurred in the late 1990s. The source of this sediment is unclear and further studies, both catchment and in-lake based, are required to elucidate the causes of the increased sedimentation rates.

5.3.3 Macrophyte Survey

Survey Date: *13 August 2002*
Surveyors: *Laurence Carvalho and Carl Sayer*
Secchi depth: *2.2 m*

The submerged vegetation of Sunbiggin Tarn was dominated by *Chara vulgaris* var. *contraria* (Moore 1986) (confirmed by Nick Stewart and also known as *C. contraria* (Allen 1950)) and *Zannichellia palustris* around the entire littoral margin present from about 0.3 m to about 2.5 m depth. There was, however, a more species-rich patch of submerged and floating-leaved species present along the south-eastern corner, dominated by *Potamogeton crispus*. A full species list with site-abundance ratings is given in Table 5.12 and a vegetation distribution map is presented in Figure 5.6.

The site is most closely associated with a calcareous-rich Type 10B according to Palmer *et al*. (1992) and has a Trophic Ranking Score of 8.66

In terms of fringing emergent vegetation, the sand and gravel substrates of the south and west shorelines were dominated by *Phragmites australis* with the occasional plants of *Menyanthes trifoliata*, *Mentha aquatica* and *Myosotis* sp., the north by *Carex rostrata*, *Filipendula ulmaria* and *Juncus effusus* and the rocky east shoreline was largely bare with the odd plant of *Caltha palustris, Callitriche stagnalis* and *Juncus effusus*. A relatively small bed of *Typha latifolia* was present in the south-west corner and flushes of *Equisetum fluviatile* and *Menyanthes trifoliata* were present around the small inflow in the south.

Although the DAFOR scale can be rather subjective it does allow for a species list to be made with some certainty, and a comparison of this 2002 survey with an earlier survey in 1982 by Charter and Welsh (Unpublished report in EN files), reveals little change in the macrophyte species present at the site. The exception is the loss of the duckweed *Lemna minor*. There is, therefore little change in site TRS scores. Other unpublished English Nature file notes suggest that the abundance of species has, however, changed. In one file note *Myriophyllum spicatum* and *Potamogeton pusillus* (probably *P. berchtoldii*) have been recorded as more abundant in the past, whilst *Zannichellia palustris* and *Chara vulgaris* var. *contraria* appear to have increased in abundance. In another 1982 boat survey by Welsh, only scraps of unhealthy *Chara* and no higher plants were recorded, with much *Aphanizomenon* (Cyanobacteria) washed up along the shore. If anything the changes since the early eighties suggest an improvement in the site condition of Sunbiggin Tarn, which now has a fairly healthy flora typical of an upland marl lake, albeit of fairly low diversity.

Figure 5.6 Vegetation distribution map for Sunbiggin Tarn August 2002

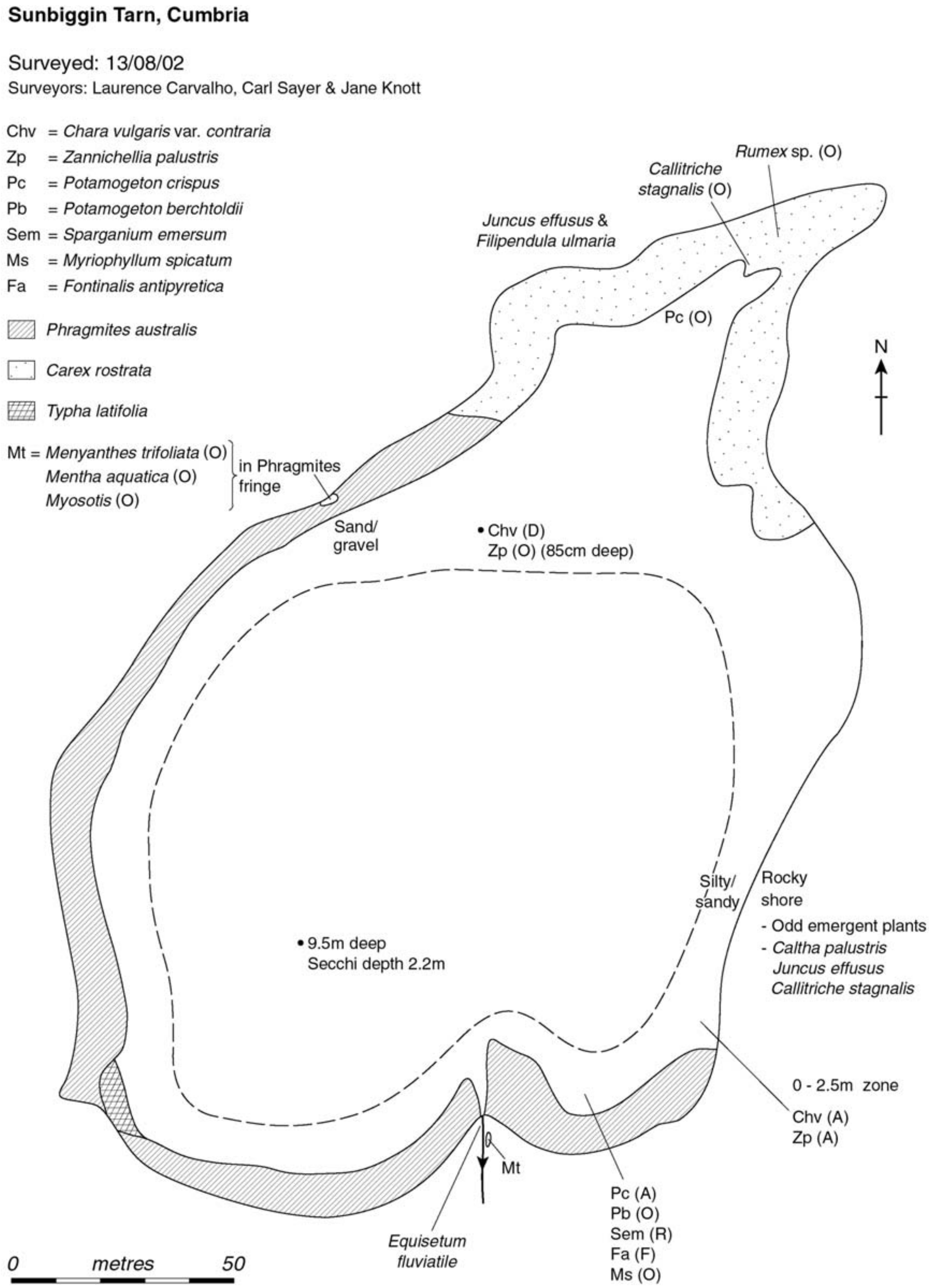


Table 5.12 Species list and DAFOR abundance rating for Sunbiggin Tarn macrophyte surveys

Submerged and floating-leaved species	TRS	1982 Charter & Welsh	2002 Carvalho & Sayer	Notes
Fontinalis antipyretica	6.3	?	O (LF)	In inflow stream in earlier survey
Potamogeton berchtoldii/pusillus	7.3	O (LF)	LO	Recorded as *P. pusillus* in other surveys
Chara connivens				Charter notes Stokoe record is from springs
Chara virgata				Charter notes Stokoe record of *C. delicatula* (synonym) from springs
Chara curta				Charter notes Stokoe record of *C. desmacantha* (synonym) from springs
Chara vulgaris var. *contraria*	8.5	?	A (LD)	
Potamogeton crispus	8.5	O (LF)	LA	common in inflow stream in earlier survey
Lemna minor	9.0	O (LF)		
Myriophyllum spicatum	10.0	?	LO	
Sparganium emersum	10.0	LF	R	
Zannichellia palustris	10.0	F	A (LD)	
Species richness		**8**	**7**	
Site TRS		**8.70**	**8.66**	
Palmer Type		**10A**	**10A**	
Max.recorded growing depth (m)		**1.4***	**2.5**	*4ft 6in. recorded in "Ref 14", but may not have been actively sought

"?" is listed if DAFOR abundance rating was not recorded

Emergent species		1982	2002	
Callitriche stagnalis			O	In 2002, on east shore
Caltha palustris		F	O	In 2002, on east shore
Carex rostrata		A (LD)	A (LD)	
Equisetum fluviatile		R	LA	In 2002, by inflow
Menyanthes trifoliata		F	F	
Phragmites australis		A	A (LD)	
Typha latifolia		LF	LD	In 2002, SW corner only

5.3.4 Summary of the Results

- Current mean annual TP concentrations of the tarn are 36.4 μgl^{-1}, and within the English Nature target set at 40 μgl^{-1}.
- Some release of P from sediments is likely but no evidence for major eutrophication was found.
- No clear nutrient signal was found from the sediment core analysis.
- Diatom species shifts in the sediment core indicate possible changes in the habitats available for diatom colonisation and increased physical disturbance in recent years.
- Aquatic macrophytes show very little change in species composition at the site over two decades.
- Moderate increases of agricultural use in catchment are now likely to be the major source of P to the tarn - currently this level appears to be “acceptable”.

5.4 Feasibility Study

The results suggest that Sunbiggin Tarn has undergone only minor trophic change due to the black-headed gull colony. The extent of this impact does not appear to have been great with respect to TP alone and any past losses of aquatic macrophytes cannot simply be attributed to increased nutrients. Physical disturbance must have been considerable during the peak of the gull colony, and this too could have affected the plants. The SSSI notification sheet for Sunbiggin (EN1994) and other English Nature file notes, state that the open water areas supported very few higher plants. This study appears to demonstrate an improvement in the macrophyte assemblages since the early eighties although this may only have occurred since the decline of the gull colony.

The sediment core analysis did not provide conclusive evidence of a shift in trophic status. One of the changes seen in the diatom communities was an increase in motile taxa and those tolerant of organic pollution. Small motile diatoms are found where soft sediments collect and are often the most common form found in disturbed environments (Goldsmith 2002), their motility enabling them to move to the sediment surface. This and other changes in the diatom assemblages towards the top of the core were more indicative of habitat change and disturbance than of trophic changes *per se*.

Contemporary water chemistry data indicate that current TP concentrations already meet the proposed English Nature target of 40 µg TP l^{-1} (annual mean TP in 2002 was 36.4 $\mu g l^{-1}$). It was not possible to calculate flushing rates in the present study but the lake retention time has been modelled using data held in the Great Britain Lakes Inventory (Bennion *et al*., 2003). The models are not without error but a retention time of 0.14 years (i.e. approximately 50 days to turn over completely) is estimated for the tarn (see comments on p 56). This is a relatively short retention time and reflects the somewhat small lake to catchment area ratio and the high mean annual run-off. These data suggest that the tarn should be able to sustain relatively high nutrient loadings without large increases in epilimnetic P concentrations or development of dense planktonic algal populations. This may explain why non-planktonic diatoms dominate the fossil assemblages in the sediment core. With careful catchment management a mean annual TP concentration of below 30 $\mu g l^{-1}$ should be achievable and might perhaps be considered as a new target figure. The control of grazing in the catchment would be a vital part of this management. Overgrazing has already been highlighted as a possible problem at the site (JNCC 2002) and it is recommended that continued efforts to address this should be pursued.

5.4.1 Recommendations

One of the objectives of the project was to make recommendations for the future management of the site but this has proved difficult due to the various problems in interpreting the palaeolimnological data, discussed above. Specifically, the project aimed to make informed decisions with regard to sediment removal. In light of the water quality results and the apparently healthy macrophyte assemblages in the tarn, sediment removal is not considered to be a viable option. Aside from cost, disturbance is a major factor at Sunbiggin due to the rich array of habitats and many protected species therein. Furthermore, due to the conservation importance of the immediate area around the tarn, the local disposal of dredged sediments is considered inappropriate. It is therefore recommended that no action be taken with respect to sediment removal. If catchment sources of P are controlled and the gull populations remain low, the tarn is unlikely to become more eutrophic.

Given the high conservation status of the tarn and concerns over the impact of enrichment on its ecology, the current lack of detailed chemical monitoring at the site should be addressed to make future management decisions more robust. It is vital to implement regular monitoring of both water quality and the biology in order to establish any direction of change. Water quality monitoring should be carried out at a minimum interval of every three months and plant surveys at 3-5 year intervals, dependent on signs of any water quality change. As with any long term monitoring programme it is essential to ensure improvement continues and to have a contingency should any deterioration be observed. Water quality monitoring can be stopped if no deterioration is seen over four consecutive years, and future surveys of water quality and macrophytes should continue at six-yearly intervals to ensure the ecological integrity of the site remains intact.

Annual monitoring of the gull numbers should also be maintained and action taken to prevent any future colonies establishing in such high numbers. It is considered likely that the physical damage from the large population of birds was greater than the effect of increased P brought about by their presence (The total weight of 50,000 black-headed gulls is 12.5 tonnes!).

In addition to monitoring, site management could be enhanced by a greater understanding of the past lake ecology. Further studies of the tarn sediments using different species groups found at the site should also be considered (see below). A combination of detailed catchment based studies of land-use change and within-lake geochemical analysis of sediment cores may be useful for identifying sediment sources.

5.4.2 Summary of Recommendations

- Liaise with local farmers and monitor stocking densities. Overgrazing is potentially damaging to the terrestrial ecology and will elevate the P load of the Tarn.
- Implement quarterly chemical monitoring to assess the nature and any changes in the lake water quality.
- Continue to monitor the aquatic flora of the lake at three yearly intervals to assess any directional.
- Continue to monitor bird numbers at the site, and prevent over population at the site.
- Stop monitoring if no significant changes are seen after a four-year period but continue chemical and biological surveys at six-yearly intervals.

A number of additional studies are recommended to support the palaeolimnological data generated in the present study.

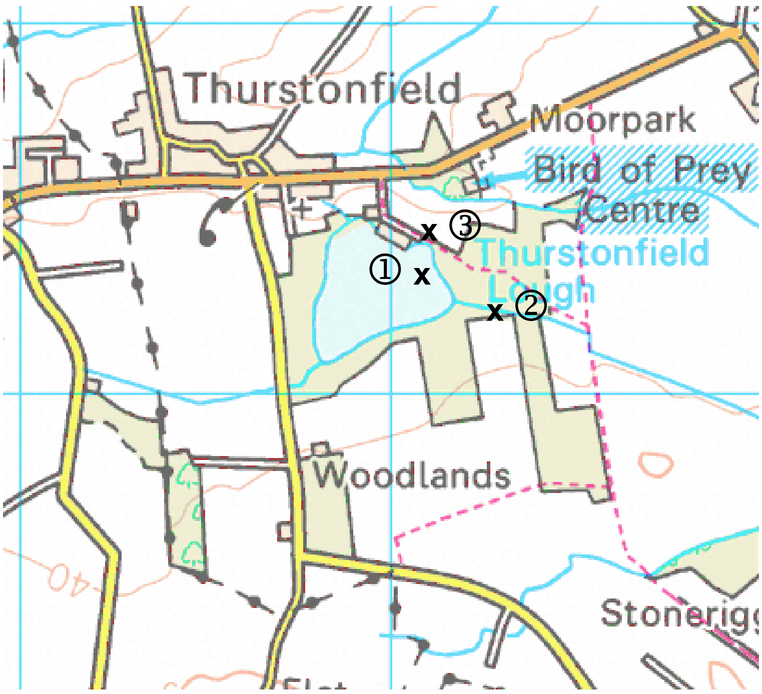
- In order to aid interpretation of the diatom record, contemporary diatom analysis is recommended to determine the habitat preferences of the principal species. This should include collection of benthic and epiphytic samples from the tarn, preferably on two to four occasions throughout the year.
- *Cladocera* analysis is recommended to provide additional information on habitat changes in the lakes. *Cladocera* are microscopic crustaceans (zooplankton) and are represented in lake sediments by a variety of body parts. They are an important component of the trophic food web in shallow lakes and could be used to infer changes in plant structure and abundance, as well as shifts in fish predation.
- To complement the diatom analysis, macrofossil analysis of a new sediment core is recommended.
- Radiometric dating of a new core or spheroidal carbonaceous particle analysis of the existing core is advised to corroborate the rapid sedimentation rates indicated by the dating results in this study.
- Detailed geochemical analysis of a sediment core is recommended to enhance the preliminary analyses carried out at the University of Liverpool.

6
Thurstonfield Lough

6.1 Description

Situated eight kilometres west of Carlisle on the Solway Plain, Thurstonfield Lough is the largest area of shallow open water in the lowlands of north-east Cumbria. The site is fringed with a range of different vegetation types, including marshland, reed beds and wet alder and sallow woodlands, making it a valuable site for conservation. The site is a SSSI and currently managed for tourism under private ownership with a view to maintaining and enhancing the local environment.

Figure 6.1 Map of Thurstonfield Lough showing water sampling locations



1. Water Sample: Lake
2. Water Sample: Inflow
3. Water Sample: Outflow

Thurstonfield Lough is a natural water body (approx. 8 ha) in a shallow depression in the boulder clays overlying Lower Lias geology. Although natural, the site has been modified over the last few centuries by the damming of the outflow streams in order to raise the level to provide water to the local flourmill in Thurstonfield village (Anon 1996). The current condition of the lough is maintained by a single sluice on the outflow (Bramble Beck) on the north east shore. There is one main feeder stream, Aikrigg Sough, in the east, and another that rarely flows in at the southwest corner. The lough lies at an altitude of 32 m and is very shallow, averaging less than 1 m. Water has a very short retention time for much of the year, particularly in periods of heavy rain when the lough is flushed through in about 50 hours (Carvalho & Moss 1994). The catchment of Thurstonfield Lough covers approximately 264 hectares (Bennion *et al.* 2003) and land-use ranges from wet alder and sallow woodland, through improved grazing to improved arable land. Annual rainfall in the region is approximately 1000 $mmyr^{-1}$ (Met. Office 2003).

Table 6.1 Catchment and lake characteristics for Thurstonfield Lough.

Catchment	
Location	Thurstonfield, 8 km W of Carlisle, Cumbria
Nat. Grid. Ref	NY320563
Altitude	32 m A.O.D
Total Area	264 ha
Geology	Lower Lias
Land Use	Mixed woodland/grazing/arable
Lake	
Maximum depth	1.2 m
Mean depth	0.47* m
Area	8.45 ha*
Volume	39,713* m^3
Retention Time	0.04 years*
Current Water Quality Classification	Eutrophic
Monitoring	No continuous monitoring. Periodic plant surveys and water chemistry
Nature Conservation Designations	SSSI
	* = Modelled data from EA GB Lakes Inventory (Bennion *et al.* 2003)

6.1.1 Conservation Interest - Reason for Notification

Thurstonfield Lough was designated as a SSSI due to it being the largest species rich area of open water in this region of lowland Cumbria (NCC 1983). At the time of designation (1983) the site was said to support some of the best examples of submerged aquatic vegetation, fringing marsh and wet sallow and alder woodland within the area (NCC 1983). In particular three nationally scarce species: six stamened waterwort (*Elatine hexandra*), autumnal water star-wort (*Callitriche hermaphroditica*) and needle spike-rush (*Eleocharis acicularis*) were recorded, but at the time Canadian pondweed (*Elodea canadensis*) was dominant in the lough (NCC 1983).

The emergent, fringing vegetation around much of the lake includes areas dominated by common reed (*Phragmites australis*), water horsetail (*Equisetum fluviatile*), Common club-rush (*Scirpus lacustris*), bulrush (*Typha latifolia*) and bottle sedge (*Carex rostrata*). Beyond this, to the west of the site wet woodlands contain grey willow (*Salix cinerea*), alder (*Alnus glutinosa*), silver birch (*Betula pendula*), oak (*quercus robar*), ash (*Fraxinus excelsior*), interspersed with rowan (*Sorbus aucuparia*) and hawthorn (*Crataegus oxyacantha*) (NCC 1983).

Over 50 species of water beetle have been recorded at Thurstonfield including several rarities (NCC 1983). The diving beetle *Hydroporus rufifrons* has been recorded at the site but recent surveys have not found it here (JNCC 2001). The site is locally important for breeding waterfowl, including large numbers of coot and tufted duck and also teal, mallard, pochard, mute swan and little grebe (NCC 1983). In addition, other notable species have been recorded at the site by the owner, including otters and red squirrels. The owner hopes to develop the existing areas of Scots pine (*Pinus sylvestris*) in order to encourage red squirrels.

In the past, the site has been stocked with rainbow trout and was managed as a commercial day fishery (Owner pers. comm.). The current owner now only allows fishing by visitors to the six chalets on site. It is thought that fish numbers are now very low, following an outbreak of the fish parasite *Argulus* sp. which prompted the owner to drain the lake in 2001. All fish were removed and the lake left dry for three weeks before being allowed to refill naturally. Current stocking rates of rainbow trout are low. There are no records of other fish in the lough.

6.1.2 Reasons for Concern - Nutrient Sources

Early records of the aquatic flora at Thurstonfield Lough show that species characteristic of relatively low nutrient conditions, such as *Littorella uniflora* and *Myriophyllum alterniflorum*, where once present in the lough (Carvalho & Moss 1994). These species have not been found in more recent plant surveys (e.g. NCC 1983, Hickson 1987, Newbold *et al.* 1998) and instead Canadian pondweed (*Elodea canadensis*), an introduced species indicative of more nutrient rich waters, is now common. A further decline in rare species seems to have occurred over the past two decades. The most recent plant survey (Newbold *et al.* 1998) failed to find either six-stamened waterwort (*Elatine hexandra*) or needle spike rush (*Eleocharis acicularis*), both of with occur on the SSSI notification sheet (NCC 1983). This suggests that there may be a possible eutrophication problem at the site.

In recent years there have been very heavy growths of *Elodea* which may be affecting the other, more notable, species present (Carvalho & Moss 1994). The performance of this species seems to vary between years at the site, being abundant in the 1983 and 1987 surveys (NCC 1983, Hickson 1987) but described as rather rare in 1998 (Newbold *et al.* 1998). Although a plant survey was not conducted for this study, *Elodea canadensis* was very evident during visits to the site in 2002. An additional problem reported by the owner is the excessive blanket weed (*Cladophora* sp.) which can cover large areas of the lough during warm stable conditions although this weed problem was not evident in August 2002.

There are no long-term historical records of water chemistry for Thurstonfield Lough but samples taken by North West Water between February 1984 and December 1987 (NWW 1987) show that levels of dissolved nutrients in the lake have been high since the mid-eighties (Tables 6.2 and 6.3).

Table 6.2 Dissolved phosphorus (SRP) values measured at Thurstonfield Lough by North West Water (1987) (values in μgl^{-1})

Date	Inflow	Lake (near outflow)	Lake (north end)
17/02/84	60	100	-
11/03/85	1410	340	-
25/11/86	200	190	70
12/02/87	20	40	50
15/05/87	30	20	20
15/06/87	30	50	40
07/08/87	70	10	5
16/10/87	160	60	10
16/12/87	90	60	40
1987 mean	**67**	**40**	**28**

Table 6.3 Nitrate (as N) values measured at Thurstonfield Lough by North West Water (1987) (values in mgl^{-1})

Date	Inflow	Lake (near outflow)	Lake (north end)
17/02/84	3.10	2.70	-
11/03/85	4.55	1.10	-
25/11/86	3.40	2.50	2.25
12/02/87	1.78	1.24	1.18
15/05/87	0.05	0.05	0.05
15/06/87	2.90	0.05	0.05
07/08/87	-	0.05	0.05
16/10/87	1.73	1.72	0.24
16/12/87	2.25	1.09	0.95
1987 mean	**1.74**	**0.70**	**0.42**

The source of these relatively high concentrations of nutrients is most likely to be agricultural. Farming practice in the catchment has intensified over the past few decades with substantial areas of woodland being cleared to make way for improved pasture (Carvalho & Moss 1994). Applications of liquid fertilizers and slurry in the catchment are regularly witnessed by the site owner (pers. comm.). Sewage used to enter the lough via the now dry western inflow, but this stopped in the 1950s and is not considered as part of the current problem (Carvalho & Moss 1994).

In addition to the apparent enrichment of the lough, it is also very shallow and the marginal vegetation has encroached considerably into areas that were once open water. Information from the site owner indicates that the area of open water has decreased by up to 50 percent in 50 years (Owner, pers. comm.). This is possibly a factor of lower water levels, as well as reed encroachment, but the owner also reported that siltation is a problem, with the feeder stream bringing in large sediment loads during high flow episodes, causing the entire lake to turn a "muddy brown" colour. The source of sediment is likely to be the same as the nutrients, i.e. from surrounding farmland. Field drains feed directly into the inflow, and some of the plots adjacent to the feeder stream are cultivated, thus increasing soil erosion. With average water depths of less than 50 cm, another problem which is likely to occur is the re-suspension of enriched sediments by wave action, thus enhancing the availability of internal nutrient sources.

The principal problems faced at Thurstonfield Lough appear to be an increase in nutrients with resultant species losses and increased plant productivity causing a decline in the area of clear open water. The problem is exacerbated by the lough being very shallow and thus at high risk of internal nutrients being released during sediment disturbance. The possibility that increased sediment loads are entering the site via the feeder stream is also considered likely.

6.2 Aims and Methodology

Five principal aims were identified for investigation at Thurstonfield Lough. These are outlined below along with a summary of the methods used.

6.2.1 Aim 1:

To measure the present phosphorus levels in the water column, and identify possible catchment sources.

Methodology:

- Quarterly measurements of TP and SRP for the lake, the inflow and outflow (Table 6.4 & Figure 6.1).
- Quarterly measurements of nitrate-nitrogen, pH, conductivity for the lake, the inflow and the outflow.
- Quarterly measurements of chlorophyll *a* concentration for the lake.
- Quarterly measurements of potassium, calcium, iron, sodium and chloride for the lake.
- Calculation of nutrient export coefficients for catchment land use.

Table 6.4 Location of sampling points in the Thurstonfield Lough catchment

Site	Location	Os Grid Ref.
Inflow	East of lake 100 m up-stream. Sampled from footbridge.	NY3227,5625
Lake	North-eastern side of lake. Sampled from boat jetty.	NY3213,5633
Outflow	North-eastern outflow (Bramble Beck) sampled below sluice near bridge.	NY3209,5639

6.2.2 Aim 2:

To estimate the volume (depth × area) of sediment in Thurstonfield Lough.

Methodology:

- This was carried out in May 2002 by inserting steel rods into the sediment at 40 points along transects covering the entire lake. The total sediment depth and water depth was measured at each point. The shallow nature of the lake and firm boulder clay bottom made this simple methodology possible.
- Sediment depth values were interpolated using an inverse distance weighted algorithm to estimate the total volume of sediment in the lake.

6.2.3 Aim 3:

To assess the impact of angling on the site, including the stocking regime.

Methodology:

- A series of interviews with the landowner, Richard Wise, on the stocking system and the role of fishing at the lake were conducted. However, the site is no longer used intensively by anglers and stocking rates are low.

6.2.4 Aim 4:

To assess the impact of fish cages on the nutrient status of the Lough

- Fish cages are no longer in place at Thurstonfield Lough.

6.2.5 Aim 5:

To prepare a feasibility study for the restoration of the Lough, discussing the merits and approximate costs of sediment removal, having made an assessment of how long it would take for natural flushing rates to reduce the sediment input of P to achieve a target level of 100 $\mu g\ l^{-1}$ TP.

Methodology:

- The results of 1-4, additional lake and catchment studies, information on sediment removal and discussions with the landowner were used to produce a feasibility study for Thurstonfield Lough.
- Flow data were not routinely collected as part of the quarterly surveys, and therefore data on retention times and nutrient exports from catchment uses were modelled using the EA GB Lakes Inventory (Bennion *et al*. 2003).

6.3 Results

6.3.1 Water Quality Analysis and Catchment Sources

The results of the quarterly chemistry sampling are presented in Table 6.5 as annual mean data based on the four 2002 measurements. The full data-set appears in Appendix I. Estimates for lake retention times are based on the modelled catchment and hydrological data from the EA GB Lakes Inventory (Bennion *et al.* 2003).

Table 6.5 Mean annual chemistry (2002) for Thurstonfield Lough.

	Lake	Inflow	Outflow
pH	7.86	7.20	7.39
Conductivity (μScm^{-1})	256	317	260
NO_3^--N (mgl^{-1})	1.16	1.77	1.18
SRP (μgl^{-1})	28.4	94.6	24.2
TP (μgl^{-1})	69.0	161.0	61.3
Chl *a* (μgl^{-1})	8.0	-	-
Potassium - K^+ (mgl^{-1})	6.02	-	-
Calcium - Ca^{2+} (mgl^{-1})	33.88	-	-
Sodium - Na^+ (mgl^{-1})	10.95	-	-
Iron - Fe^{3+} (μgl^{-1})	240	-	-
Chloride - Cl^- (mgl^{-1})	21.88	-	-

Using land cover and stocking data from the EA Lakes Inventory (Bennion *et al.* 2003) the current sources of P can be identified by the use of export coefficients calculated from catchment land use. The inventory can also be used to gain an idea of a target baseline for P at Thurstonfield Lough by using a hind-casting model (Johnes *et al.* 1996) based on historical catchment data for the year 1931. Tables 6.6 and 6.7 outline the EA GB Lake Inventory (Bennion *et al.* 2003) characteristics for Thurstonfield Lough and Table 6.8 shows the calculated loadings of P using recognised export coefficients (Johnes *et al.* 1996).

Table 6.6 Catchment and lake characteristics for Thurstonfield Lough (source: EA Lakes Inventory (Bennion *et al.* 2003).

Catch area (ha)	264
Lake surface area (ha)	8.45
Lake:catch ratio	0.03201
Mean depth (m) (modelled)	0.47
Max. depth (m)	1.2
Total Volume (m^3) (modelled)	39713
Retention time (yrs) (modelled)	0.04
Mean runoff (mm) (CEH data 1995-7)	385
Stratification class (modelled)	1 (unstratified)

Table 6.7 Land use estimates for the Thurstonfield Lough catchment (source: EA Lakes Inventory (Bennion *et al.* 2003).

Land Use Class	Catchment Cover (%)
Meadow/semi-natural	45.45
Grazed turf	32.91
Grass heath	8.95
Tilled land	5.82
Deciduous Woodland	3.72
Water	1.42
Rural Development	0.85
Unclassified	0.28
Lowland bog	0.28
Coniferous Woodland	0.14
Urban	0.12
Bracken	0.05

Table 6.8 Modelled phosphorus data for Thurstonfield Lough, including a hindcast estimate for the site from 1931 using export coefficients (source: EA GB Lakes Inventory (Bennion *et al.* 2003).

Model Component	Model Value
No. Cattle	568.66
No. Sheep	191.51
No. Pigs	0
No. People (1991)	0
Land cover P (kg/yr)	54.69
Cattle P (kg/yr)	125.11
Sheep P (kg/yr)	8.62
Human P (kg/yr)	0
Current P load total (kg/yr)	188.41
Current Modelled P conc. (µg/l)	105.8
Hind-cast P load (kg/yr)(1931) Reading model	114.0
Hind-cast Lake P conc. (µg/l) Reading model	64.0

The measured lake TP is lower than that estimated from the catchment land usage (Bennion *et al.* 2003). Using the standard OECD regression equations to convert P loads in to P concentrations a modelled catchment load of 188.41 kg P yr^{-1} gives a lake TP concentration of approximately 105 μgl^{-1}, compared to a measured annual mean TP of 69 μgl^{-1}. This situation is reversed when the inflow chemistry is considered, the feeder stream having an annual TP of 161μgl^{-1}. The catchment is therefore clearly responsible for high nutrient loadings but this does not appear to be reflected in high lake TP concentrations when only looking at annual mean data. Concentrations of TP in the lough have been measured as high as 337 μgl^{-1} (Nov. 2000, Appendix I), but these very high levels were not observed in 2002. The reason for this disparity is unclear and further chemical analysis would be needed to determine if the 2002 mean figure for TP is representative of the lough's current nutrient status.

Although the exact nature of the nutrient conditions is unclear, the results do suggest that the feeder stream is bringing in high loads of both phosphorus and nitrogen to the lough. In addition to nutrients there is an obvious problem with high sediment loads in the feeder stream. In many places the inflow is heavily silted and a small sediment trap installed by the owner is already full. Fields to the east of the lough which slope down to the feeder stream (marked as woodland on 1983 OS map) have been cultivated to within 1 m from the bank. With the inflow of agriculturally derived material, lies the risk of other pollutants to the lough. The possibility that the observed reduction in open water species is due to herbicide pollution can not be ruled out. For species already under stress from higher nutrient loadings, reduced water clarity and high competition, even small amounts of in-washed herbicides could be responsible for their final disappearance from a site (Moss *et al*. 1996). Herbicides are rarely persistent in water and thus the effects are almost impossible to detect.

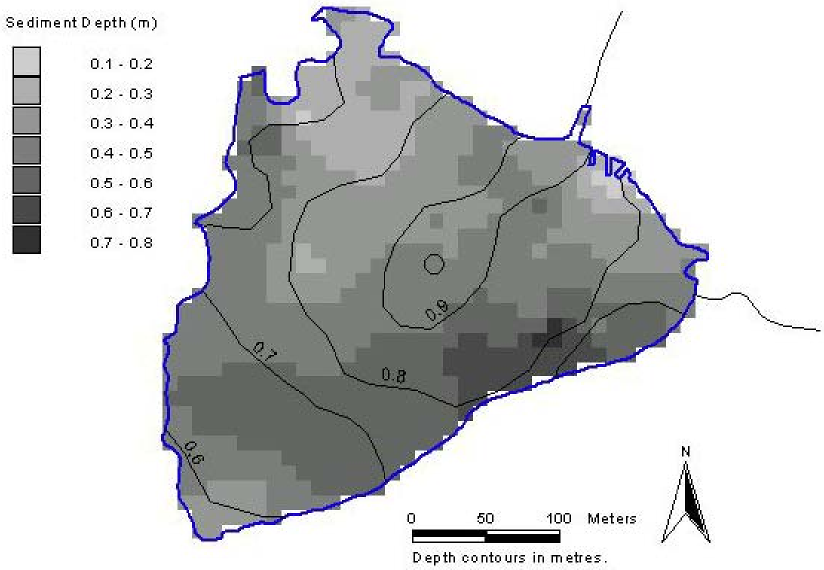
The possibility of a target concentration of lake TP at 100 μgl^{-1} is difficult to ascertain in light of both the current results and the management changes outlined below. The 2002 results suggest that the target has already been reached, but this is not altogether corroborated by the high values of TP in the inflow stream and previously high concentrations being recorded in the lough in this study. Management suggestions are outlined below, which will attempt to reduce the catchment sources of P and thus future monitoring of water quality will be required to assess the success of these projects.

6.3.2 Estimation of Sediment Volume in Thurstonfield Lough.

Sediment depth measured at 40 points ranged from 10 to 90 cm, and water depth ranged from 0.6 to 1.0 m. There was no significant relationship between lake depth and sediment depth.

Sediment depth values were interpolated using an inverse distance weighted algorithm to estimate the total volume of sediment in the lake. The lake sediment has a mean depth of 0.46 m. Based on a surface area of 7.49 ha (derived from OS Land Line® data) the total volume is estimated to be 35,000 m^3. The deepest sediments are along the south-east side of the lough.

Figure 6.2 Depth distribution of the sediment in Thurstonfield Lough



These estimates should be used only as a rough guide since they are based on a limited set of measurements and there are errors associated with the interpolation procedure, which cannot be avoided. Depending on the density of actual measurements taken the margin of error for sediment estimates can be as great as 50%. The inverse distance weighted algorithm uses a fixed radius to interpolate (estimate) a value for cells in a matrix (a 10 m cell size was used for this exercise). If there are no or few actual measurements within the search radius then the estimate will be poor. The closer a cell is to an actual measurement the better the estimate will be. Nearby values are given a greater weighting than those further away – hence the name inverse distance weighted. This method is used because it is assumed that sediment depths show a large degree of auto-correlation, i.e. the sediment depth at one point is closely correlated with the sediment depth at a nearby point.

6.3.3 Fisheries and Angling at Thurstonfield

Following a series of interviews with the site owner it has been established that there is no longer an intensive fishery at the site. The previous owner ran a day fishery and it is thought that the lough was stocked at very high densities, but there are no figures to support this. Following a persistent outbreak of an *Argulus* sp. parasite at the site the lough was drained by the current owner in January 2001 and left dry for three weeks. Since then the lake has been restocked with rainbow trout at a density of 50 fish per acre (approx. 125 fish per hectare). There is no evidence to suggest that the fish are impacting on the water quality.

Fish Cages

Prior to this study it was reported that fish cages were in place on the lough and that these might be having a detrimental effect on the nutrient status of the water body. These fish cages are no longer in place and the owner reports this to have been the case for at least five years. There is no documented information on these cages and thus any possible impact can not be determined.

6.3.4 Summary of Results

- The mean annual TP concentration for the lough in 2002 was 69 μgl^{-1}.
- The mean annual TP concentration for the inflow in 2002 was 161 μgl^{-1}.
- Catchment sources (agriculture) are recognised as contributing the majority of the P load.
- In addition to nutrients, high external loads of suspended solids are a problem.
- The lough is very shallow, and siltation and reed encroachment are a major threat to open water survival.
- Sediment depths of up to 90 cm were measured, with a mean sediment depth of 46 cm.
- The deepest sediments are towards the south-east edge of the lough.
- The estimated volume of sediment covering the entire lough is 35,000 m^3.
- Fish stocks are currently low (<125 fish (rainbow trout) per hectare) and the fishery is no longer managed intensively.
- Fish cages have not been in the lough for at least 5 years.

6.4 Feasibility Study

Thurstonfield Lough would appear to be threatened from two possible sources: eutrophication and in-filling. Without any historical water quality data it is not possible to ascertain any directional changes in P concentrations. The lake is currently eutrophic however and recent plant surveys suggest sensitive species are being lost and the growth of more generalist plant species such as *Elodea canadensis* and *Cladophora* sp. are becoming a problem (Newbold *et al.* 1998). What is evident from this study is that there are considerable nutrient loadings coming from the catchment and there is a strong likelihood that these are from agricultural sources. The problem of in-filling is exacerbated by several factors. First the high nutrient levels favour the increased production of marginal vegetation and rapid encroachment of the reed beds has been reported by the site owner. Increased areas of marginal vegetation act to stabilise water movement and thus speed up the sedimentation rate. Secondly, high nutrient conditions accelerate autochthonous (within-lake) production and account for high sedimentation rates. Rates in excess of 1 cm per year are not unusual in enriched lowland lakes (Bennion 1994) and in this case would reduce the life-span of the lough to only 50 years. Added to this is a third problem, that of external sediment loadings. The lough appears to be receiving high levels of eroded material via the feeder stream. Combined, these three factors could cause the site to change from its current open water status to a fen community over a relatively short time period. This would change the function of the site and if the open water was lost altogether, detract from the original value denoted by its SSSI assignation (NCC 1983).

Due to the conditions outlined above, the lough is considered to be at high risk of losing its current status and suffering further decline if remedial action is not taken. Two possible levels of action are recommended, the first concentrates on a decrease in catchment derived nutrient and sediment sources and the second on sediment removal from the lough. It should be stated here that sediment removal is recognised as a last resort technique (Moss *et al.* 1996) and is only considered applicable to this site because it is <u>so</u> shallow. Furthermore it should be stressed that there is little advantage in removing sediments without also attempting to significantly reduce the nutrient sources (Moss *et al.* 1996), and in this case the external sediment loads.

6.4.1 Control of External Nutrient and Sediment Loads

Promote good agricultural practice
Ideally the first step would be to attempt to reduce the problem at source by liaising with local farmers and ensuring best practise is followed in the use of chemical and slurry applications and cultivation. Timing of fertilizer applications to avoid water logged or frozen soil ensures more efficient take-up and is thus beneficial to the farmer as well as the environment. This is of particular concern to the area of farmland lying south-east of the lough. All farmers are encouraged to follow a voluntary code of good practice with respect to field run-off (DEFRA 2002) and it is particularly important to promote this within the Thurstonfield Lough catchment.

Promote the introduction of buffer zones
The introduction of buffer zones between farmland and the receiving waters, has proved effective in the control of nitrogen and sediments (including particulate P)

elsewhere, but does not have any significant capability to control soluble P (Moss *et al.* 1986). In this case any reduction in either sediment or nutrient load would be an advantage and although the long-term effectiveness of buffer zones to control nitrogen is questionable (Moss *et al.* 1986) they can provide good wildlife corridors. If the areas either side of the inflow stream immediately beyond the eastern edge of the woodland could be buffered with a minimum of 10 m strips, this would act as an initial barrier to particulates and provide additional wildlife habitat. Buffer strips beyond the eastern woodland may not be appropriate in this case because of the need to gain consent from land owners beyond the site boundaries

Increase the potential of wet woodland
The area of wet woodland to the east of the lough already provides some buffering capacity but this could be increased. The recognition that this wooded area is a valuable wildlife resource in its own right, necessitates careful extra management. The feeder steam is currently canalised through part of the wooded area and the channel would benefit from being more open here in order to slow the water down and provide a more extensive wetland area. East Cumbria Countryside Project (1995) has already stated the advantage of coppicing small areas of the alder and birch. As well as increasing the woodland edge effect and habitat diversity, the re-growth of well managed coppice stands would provide a greater nutrient sink than older stands of woodland (Moss *et al.* 1996). ECCP (1995) recommended a 20 year cycle for coppice coupes.

Increase wetland area at lake inflow
In addition to increasing the area of wet woodland, there is also potential for an increased area of reed bed at the lake inflow. Reeds provide a very effective means of controlling nitrogen and sediment and to a much lesser extent phosphorus (Moss *et al.* 1996). A maximum area of reed bed could be achieved by clearing a delta into the scrub/woodland area behind the existing line of the lough shore and planting with reeds (*Phagmites australis*) to fill the deepened delta. The reed bed could also be encouraged to encroach into the lake. Such an area is only likely to work effectively under carefully controlled conditions, and should not be allowed to dry out. The LIFE Reed Project (Mills *et al.* 1998) gives detailed information on the construction and management of reed beds as filters (and wildlife habitat). A long-term management strategy is likely to involve a necessity to harvest the reeds periodically to maintain performance. Clear objectives need to be established prior to planting, including a means of harvesting (by hand is manageable on a small scale like this) and removal from or use at the site (potentially reeds could be used to thatch the on-site chalets and to construct wildlife hides around the lough).

The construction of such a reed bed would be best done in conjunction with other recommended work to minimise disturbance at the site. A clearing and deepening of the area could be done at the same time as the digging of a settling pond and diversion stream. Alternatively it may best be achieved when the lake is drained to remove sediment. If the latter is done, reed rhizomes could be re-located from areas of the lough which are being cleared and deepened.

Settlement pond and diversion of feeder stream
The site owner has pursued the idea of installing a settlement pond before the feeder stream reaches the lough in order to reduce the sediment loading. This would require access to facilitate sediment removal when full. In addition plans have been drafted to allow for the diversion of the feeder stream between the settlement pond and the outflow. This would be operated by a simple sluice and used to direct water away from the lough during periods of high flow when sediment transport and nutrient run-off are usually highest. The estimated cost of this scheme, including the installation of access routes and footpath maintenance, was quoted as being £6,330 by an Environment Agency approved contractor (Ken Hope Ltd., Westmoor).

If carefully managed, this scheme could help to alleviate the high intensity episode of external loadings to the lough by diverting the water but also help to reduce the amount of suspended material entering the system. The creation of a settlement pond may also be managed to facilitate amphibian breeding at the site. Although common toads (*Bufo bufo*) have been reported in high numbers at the site (owner, pers, comm.), Hickson (1987) identified amphibians (and reptiles) as being notable by there absence and recommended the construction of a small breeding pond. No doubt fish predation on the larval stages, was a major factor when the lough was heavily stocked with trout, and it is unclear what effect the lower stocking rates have had on amphibian success more recently. Nevertheless, a small sheltered pond could provide good additional breeding habitat for frogs, toads and newts. If such a pond was managed in this way, removal of settled sediments would need to be restricted to late autumn or winter.

6.4.2 Sediment Removal

The principal threats to this site are considered to be eutrophication and in-filling. The combined effects of these processes are further enhancing the rapid expansion of marginal vegetation, resulting in a reduction in the surface area of open water. Without any remedial action the current conservation value of the lough is very likely to decline. Even if reductions in sediment in-wash and nutrients are achieved, the future of open water surviving at the site in the long-term is not considered likely. It is therefore recommended that sediment be removed from Thurstonfield Lough in order to prolong the life of the system and enhance the existing habitats to provide more open water. This is considered as a last resort but essential action to maintain the lough. The removal of sediment will also eliminate the current internal loadings of the lough and therefore reduce the effects of eutrophication and perhaps restrict the growth of *Elodea canadensis* which derives the majority of its phosphorus from the sediments (Carvalho & Moss 1994).

Method of sediment removal
Because the lake is so shallow and the water level is maintained via a sluice, it is very easy to drain the lake. Once drained, the site will be easily accessible to heavy plant equipment, and thus forgo the need for specialised mud pumping equipment to be brought in. Enquiries made by the site owner suggest this to be the cheapest and most effective plan. A brief summary of the work is outlined below.

Sediment removal
- Drain the lake.
- Form a clay bund with a digger along the north-western side of the lake at a point approximately 50 metres into the current line of the reeds. The reeds and willows are over 100m deep at this point.
- A sump hole would then be dug at the lowest point in the lake bed which is the north-east corner of the lake between the feeder stream and the outflow.
- A bulldozer (with low ground pressure tracks) would then skim the sediment towards the sump and this sediment "soup" can then be pumped with a conventional "mud pump" across the lake bed and over the clay bund to the settlement lagoon.
- It is recommended that some shallower marginal regions are left completely undisturbed as refugia for plant species. The identification of where particular species occur should be made.
- Removal of the majority of rich black sediment (averaging 46 cm) from beyond the reed margin is recommended and the possible deepening of a small area has been proposed by the land owner (dependent on basal material).
- Whilst the lake is dry and the machinery present the reed beds along the lake perimeter can be reduced and some of the rhizome rich material used to develop the reed beds around the inflow (see above).
- Pumped sediments would have high water content and need to be settled behind the bund before allowing the water to filter away into the diversion beck on the west side of the lake.
- It is advisory not to let this water flow back through the lough (Moss *et al.* 1996).
- Once dry, the site owner is keen to develop the area behind the bund for terrestrial conservation (possibly planting Scots pine to encourage red squirrels)
- The lake would be allowed to refill naturally.
- Freshly pumped sediment poses an extreme risk of drowning to both humans and animals and, therefore, the area must be fenced and clear warning notices erected during the work.
- Following the work, the average lake depth would be in the region of 90 cm, with a maximum of 1.5 m. Scope does exist to deepen an area up to 2.5 m.

Approximate costs
Two contractors have already given outline quotes for this work of less than £20,000. Further quotes would be needed now that the sediment depth and volume is known and for extra woodland management outlined above. The overall cost of this work including the additional recommendations (not including buffer strips) is estimated to be in the region of £30,000. Additional costs will be incurred for future work and are in part dependent on the success of the project (see below).

Timing
The optimal timing for this work to be undertaken needs to be carefully assessed. Logistically the work would be easier in the summer, when the inflow stream is at its lowest, and soils dry for heavy machinery. Warmer weather would also facilitate evaporation and aid the sediment drying process. Late summer would give the maximum chance for birds to have finished breeding. For the plant communities early spring would be more suitable, giving them the maximum chance to propagate from freshly disturbed root fragments and from seed, but plant propagules will also

re-establish in late summer although not with the same vigour (Hearn *et al*. 2002). Spring work would also be detrimental to amphibian communities and thus it is advised against for this project. Damage to invertebrate communities is inevitable. If areas of marginal vegetation are left undisturbed it is hoped adequate number will survive.

The contractors estimate that the work to form the bund and clear the lake bed would take one month to six weeks, with additional time required dependent on other work being conducted on the site (outlined above).

Re-establishment of plant communities

Sediment removal is obviously a very high disturbance operation and it will result in temporary loss of habitat and species at the site. The major problem at Thurstonfield Lough will be the loss of submerged macrophytes. Many submerged and floating leaved species spread via vegetative means and thus can re-establish quite quickly after disturbance if fragments are preserved or remain at the site (Moss *et al.* 1996). The protection of some marginal areas can aid this process. There should be no hurry to re-plant the site, but rather assess natural re-establishment of species in the first year. Species which no longer occur at the site can be artificially established once other management criteria have been met. There is no point in trying to re-establish populations of *Elatine hexandra* for example, if clear-water conditions are not established. If, however, re-growth of the more common species is sufficient, the re-introduction of other plants is desirable. It is not considered good practice to introduce plant species which have never been recorded at the site (Moss *et al.* 1996). It will be essential to monitor the chemistry of the site in the first few years and implement a longer-term monitoring scheme for the biology from the outset.

It should be stressed here that the natural re-colonisation of plants is not guaranteed. If this were to be the case extra financial input would be necessary over several years in order to provide plants for re-planting and create suitable habitat. Physical disturbance by wave action or birds can be a major problem in the establishment of submerged plants and necessitates the construction and installation of protective structures (Moss *et al.* 1996).

It will be important to maintain clear water. Initially it is vital to ensure reasonable levels of zooplankton (particularly large *Daphnia* sp.) which are effective grazers of planktonic algae and thus any introduced fish must not be zooplanktivorus (i. e. not bream or carp). This is particularly important in shallow lakes where plants are not yet established because of a lack of refugia from predatory fish. If fish are re-introduced, piscivorous fish are preferable with perhaps the native brown trout being favoured over rainbow trout.

Other financial considerations

An important consideration in the planning of this scheme is that it will need continued financial support for the foreseeable future. This should be directed in three main areas. i.) The removal of sediment from the inflow settlement pond which will need to be done when full. There is no set time interval for this but autumn would subject the least disturbance if amphibians are known to use the pond. ii.) Reed cutting. This should not need to be annual and the small area will mean it can be done by hand. Provision needs to be made for the use or disposal of the reeds. iii.)

Biological and chemical monitoring will be vital to ensure the success of the work. Water quality monitoring should be carried out at a minimum interval of every three months and plant surveys at 3-5 year intervals, dependent on signs of any water quality change. As with any long term monitoring programme it is essential to ensure improvement continues and to have a contingency should any deterioration be observed. Water quality monitoring can be stopped if no deterioration is seen over four consecutive years, and future surveys of water quality and macrophytes should continue at six-yearly intervals to ensure the ecological integrity of the site remains intact. In addition, some financial reserve should be made for the replanting of the lough if natural re-colonisation is not successful.

Legal implications
The legal implications of dredging need careful consideration and the Environment Agency should be contacted before any work is done. If spoil is to be disposed of on site as a land reclamation scheme an exemption from a Waste Management Licence (under the terms of the Environmental Protection Act 1990) should be granted (Hearn *et al.* 2002). It is vital that expert advice be sort to ensure dredging and sediment disposal remains within the law. This should also include the analysis of sediments for toxicity prior to dredging to ensure down-stream contamination does not occur. Heavy penalties are in place if any down-stream disturbance is caused. A discharge consent will be required for water and particulates draining from the lagoon, and expert advice will be needed to ensure the correct procedures are put in place and followed.

6.4.3 Summary of Remedial Action

- Encourage good agricultural practice in the catchment.
- If possible, the introduction of buffer zones beyond the eastern woodland.
- Increase the potential of the woodland as a nutrient and sediment sink.
- Create a settling pond to enhance sediment removal.
- Divert feeder stream during high flow events.
- Replace area of wet scrub and woodland behind lake inflow with reed beds.
- Remove sediment to deepen the lough and remove internal P loadings. This is only feasible if action is taken to control external sources of sediments and nutrients.
- Monitor the chemistry of the site in the first few years (quarterly) and implement a longer-term monitoring scheme for the biology from the outset (3-5 years).

6.4.4 Additional Recommendations

- Prior to any work being conducted sediment cores should be taken for the analysis of plant macrofossils. This would hopefully allow for the determination of past plant communities and thus a better understanding of restoration targets.
- *Cladocera* analysis is also recommended to provide additional information on habitat changes in the lakes. They are an important component of the trophic food web in shallow lakes and could be used to infer changes in plant structure and abundance, as well as shifts in fish dynamics.

- Analysis of sediment toxicity would also be necessary to ensure compliance with discharges from the site during dredging. Material from sediment cores could be shared to minimise costs.
- All remedial works should be conducted at the same time to minimise disturbance to the site and also to reduce costs.
- The diversion of the feeder stream may not be necessary if a reed bed is put in place by the lake inflow.

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Appendix I

Water chemistry for Elterwater - see Figure 3.1 for location of sampling points

Date	Sample site	Site code	pH	Cond. μScm^{-1}	Nitrate mgl^{-1}	SRP μgl^{-1}	TP μgl^{-1}	Chl. *a* μgl^{-1}
22/11/00	4	Inflow 1	7.10	41	0.20	5.7	7.5	n/a
28/02/01	4	Inflow 1	7.29	75	0.19	1.4	42.5	n/a
28/02/02	4	Inflow 1	6.33	44	0.46	1.7	10.0	n/a
28/05/02	4	Inflow 1	6.33	38	0.38	6.7	15.0	n/a
22/08/02	4	Inflow 1	6.45	33	0.09	3.7	6.9	n/a
19/11/02	4	Inflow 1	7.12	48	0.19	5.0	9.0	n/a
2002 Mean	**4**	**Inflow 1**	**6.56**	**41**	**0.28**	**4.3**	**10.2**	**n/a**
22/11/00	5	Inflow 2	7.26	58	0.27	4.3	7.5	n/a
28/02/01	5	Inflow 2	7.23	130	0.25	0.0	35.0	n/a
28/02/02	5	Inflow 2	6.35	44	0.32	1.7	5.0	n/a
28/05/02	5	Inflow 2	6.21	37	0.21	1.7	52.5	n/a
22/08/02	5	Inflow 2	6.73	31	0.04	2.1	5.9	n/a
19/11/02	5	Inflow 2	7.05	46	0.13	5.0	9.0	n/a
2002 Mean	5	Inflow 2	**6.59**	**40**	**0.18**	**2.6**	**18.1**	**n/a**
28/02/01	6	Inflow 3	7.06	no data	0.37	1.4	25.0	n/a
28/02/02	6	Inflow 3	6.49	no data	0.59	1.7	10.0	n/a
28/05/02	6	Inflow 3	6.03	80	0.57	5.0	17.5	n/a
22/08/02	6	Inflow 3	7.15	93	0.11	2.9	6.0	n/a
19/11/02	6	Inflow 3	7.14	83	0.03	5.0	10.0	n/a
2002 Mean	6	Inflow 3	**6.70**	**85**	**0.33**	**3.6**	**10.9**	**n/a**
28/02/01	3	Outer Basin	7.51	100	0.27	0.0	37.5	8.3
28/02/02	3	Outer Basin	6.33	49	0.28	1.7	2.0	0.0
28/05/02	3	Outer Basin	8.00	49	0.06	3.3	10.0	1.3
22/08/02	3	Outer Basin	6.60	62	0.08	3.5	8.3	1.8
19/11/02	3	Outer Basin	6.92	48	0.24	5.0	7.0	0.6
2002 Mean	3	Outer Basin	**6.96**	**52**	**0.16**	**3.4**	**6.8**	**0.9**
28/02/01	2	Middle Basin	7.69	100	0.27	1.4	52.5	7.1
28/02/02	2	Middle Basin	6.29	49	0.06	1.7	46.0	1.1
28/05/02	2	Middle Basin	8.42	60	0.17	3.3	40.0	8.1
22/08/02	2	Middle Basin	6.65	52	0.03	3.3	11.3	6.7
19/11/02	2	Middle Basin	6.58	115	0.14	8.3	49.0	6.0
2002 Mean	2	Middle Basin	**6.99**	**69**	**0.10**	**4.2**	**36.6**	**5.5**
22/11/00	1	Inner Basin	7.14	78	0.24	12.9	17.5	0.9
28/02/01	1	Inner Basin	7.76	105	0.19	4.3	57.5	10.8
28/02/02	1	Inner Basin	6.30	49	0.29	3.3	20.0	0.7
28/05/02	1	Inner Basin	8.12	76	0.17	3.3	72.5	11.9
22/08/02	1	Inner Basin	6.60	67	0.01	4.0	17.2	10.7
19/11/02	1	Inner Basin	6.76	68	0.18	8.3	15.0	3.1
2002 Mean	1	Inner Basin	**6.95**	**65**	**0.16**	**4.8**	**31.2**	**6.6**
22/11/00	7	Outflow	7.24	48	0.30	7.1	10.0	n/a
28/02/01	7	Outflow	7.06	105	0.13	5.7	37.5	n/a
28/02/02	7	Outflow	6.60	49	0.39	3.3	11.0	n/a
28/05/02	7	Outflow	6.13	50	0.28	3.3	67.5	n/a
22/08/02	7	Outflow	6.46	34	0.09	5.2	8.5	n/a
19/11/02	7	Outflow	7.03	50	0.21	5.0	6.0	n/a
2002 Mean	7	Outflow	**6.56**	**46**	**0.24**	**4.2**	**23.3**	**n/a**

Appendix I

Water chemistry for Hawes Water - see Figure 4.1 for location of sampling points

Date	Sample Site	Site code	pH	Cond. μScm^{-1}	Nitrate mgl^{-1}	SRP μgl^{-1}	TP μgl^{-1}	Chl. *a* μgl^{-1}
22/11/00	2	Inflow	7.69	497	0.32	5.7	10.0	n/a
28/02/01	2	Inflow	No data	No data	No data	No data	No data	n/a
28/02/02	2	Inflow	7.95	390	1.51	1.7	2.0	n/a
28/05/02	2	Inflow	7.33	460	0.15	15.0	130.0	n/a
22/08/02	2	Inflow	7.45	515	0.13	124.4	211.4	n/a
18/11/02	2	Inflow	7.53	504	0.40	5.0	6.0	n/a
2002 Mean	2	Inflow	**7.57**	**467**	**0.55**	**36.5**	**87.4**	**n/a**
22/11/00	1	Lake	7.93	450	0.13	4.3	7.5	1.5
28/02/01	1	Lake	No data	No data	No data	No data	No data	n/a
28/02/02	1	Lake	8.03	360	0.63	1.7	2.0	2.9
28/05/02	1	Lake	8.12	410	0.26	6.7	65.0	3.3
22/08/02	1	Lake	7.73	400	0.04	2.4	10.2	5.4
18/11/02	1	Lake	7.88	412	0.12	5.0	7.0	2.7
2002 Mean	1	Lake	**7.94**	**396**	**0.26**	**3.9**	**21.1**	**3.591**
22/11/00	3	Outflow	7.69	471	0.25	8.6	12.5	n/a
28/02/01	3	Outflow	No data	No data	No data	No data	No data	n/a
28/02/02	3	Outflow	7.90	350	0.99	5.0	8.0	n/a
28/05/02	3	Outflow	7.47	400	0.15	11.7	57.5	n/a
22/08/02	3	Outflow	7.48	430	0.03	21.5	57.1	n/a
18/11/02	3	Outflow	7.61	451	0.11	5.0	6.0	n/a
2002 Mean	3	Outflow	**7.62**	**408**	**0.32**	**10.8**	**32.2**	**n/a**

Water chemistry for Sunbiggin Tarn - see Figure 5.1 for location of sampling points

Date	Sample site	Site code	pH	Cond. μScm^{-1}	Nitrate mgl^{-1}	SRP μgl^{-1}	TP μgl^{-1}	Chl. *a* μgl^{-1}
22/11/00	1	Lake	7.72	326	0.19	17.1	22.5	0.7
28/02/01	1	Lake	no data	no data	no data	no data	no data	n/a
28/02/02	1	Lake	7.95	240	0.35	6.7	18.0	0.7
28/05/02	1	Lake	7.26	210	0.00	10.0	70.0	1.1
22/08/02	1	Lake	7.99	290	0.01	25.6	47.4	8.0
19/11/02	1	Lake	7.99	292	0.09	10.0	10.0	1.6
2002 Mean	1	Lake	**7.80**	**258**	**0.11**	**13.1**	**36.4**	**2.8**
22/11/00	2	Outflow	7.86	321	0.27	17.1	25.0	n/a
28/02/01	2	Outflow	no data	no data	no data	no data	no data	n/a
28/02/02	2	Outflow	7.91	250	0.36	6.7	8.0	n/a
28/05/02	2	Outflow	7.68	220	0.00	11.7	115.0	n/a
22/08/02	2	Outflow	7.75	295	0.01	26.0	47.3	n/a
19/11/02	2	Outflow	8.11	310	0.07	6.7	24.0	n/a
2002 Mean	2	Outflow	**7.86**	**269**	**0.11**	**12.8**	**48.6**	**n/a**

Appendix I

Water chemistry for Thurstonfield Lough - see Figure 6.1 for location of sampling points

Date	Sample site	Site code	pH	Cond. μScm^{-1}	Nitrate mgl^{-1}	SRP μgl^{-1}	TP μgl^{-1}	Chl. *a* μgl^{-1}
22/11/00	2	Inflow	7.28	395	2.15	95.7	147.5	n/a
28/02/01	2	Inflow	7.32	450	1.08	77.1	207.5	n/a
28/02/02	2	Inflow	6.80	225	2.54	101.7	226.0	n/a
28/05/02	2	Inflow	6.91	325	1.84	196.7	302.5	n/a
22/08/02	2	Inflow	7.60	380	1.31	11.6	24.6	n/a
22/11/02	2	Inflow	7.50	339	1.40	68.3	91.0	n/a
2002 Mean	**2**	Inflow	**7.20**	**317**	**1.77**	**94.6**	**161.0**	**n/a**
22/11/00	1	Lake	7.36	550	3.00	40.0	337.5	0.9
28/02/01	1	Lake	7.17	485	0.88	42.9	135.0	2.0
28/02/02	1	Lake	6.85	210	3.70	48.3	120.0	3.7
28/05/02	1	Lake	9.15	205	0.00	5.7	32.5	0.9
22/08/02	1	Lake	7.93	290	0.06	26.5	65.3	25.8
22/11/02	1	Lake	7.52	318	0.88	32.9	58.0	1.6
2002 Mean	**1**	Lake	**7.86**	**256**	**1.16**	**28.4**	**69.0**	**8.0**
22/11/00	3	Outflow	7.44	328	1.55	35.7	70.0	n/a
28/02/01	3	Outflow	7.35	475	1.45	34.3	122.5	n/a
28/02/02	3	Outflow	6.79	220	3.38	51.7	136.0	n/a
28/05/02	3	Outflow	7.30	220	0.35	13.3	50.0	n/a
22/08/02	3	Outflow	7.91	290	0.07	15.3	25.2	n/a
22/11/02	3	Outflow	7.55	310	0.90	16.7	34.0	n/a
2002 Mean	**3**	Outflow	**7.39**	**260**	**1.18**	**24.2**	**61.3**	**n/a**

Appendix II

List of all diatom taxa (total 63) observed in the Sunbiggin Tarn core.

Achnanthes lanceolata (Breb. Ex Kutz.) Grun. in Cleve & Grun. 1880
Achnanthes clevei Grun. in Cleve & Grun. 1880
Achnanthes oestrupii (A. Cleve-Euler) Hust. 1930
Achnanthes exigua Grun. in Cleve & Grun. 1880
Achnanthes minutissima Kutz. 1833
Achnanthes conspicua A. Mayer 1919
Achnanthes ziegleri Lange-Bertalot 1991
Achnanthes sp.
Amphora veneta Kutz. 1844
Amphora thumensis (Mayer) A. Cleve
Amphora pediculus (Kutz.) Grun.
Amphora inariensis Krammer
Cymbella sinuata Greg. 1856
Cymbella microcephala Grun. in Van Heurck 1880
Cymbella minuta Hilse ex Rabenh. 1862
Cymbella silesiaca Bleisch ex Rabenh. 1864
Cymbella reichardtii Krammer 1985
Cymbella sp.
Cocconeis placentula Ehrenb. 1838
Cocconeis pediculus Ehrenb. 1838
Cyclotella radiosa (Grunow) Lemmerman 1900
Denticula tenuis Kutz. 1844
Diploneis sp.
Eunotia sp.
Fragilaria pinnata Ehrenb. 1843
Fragilaria construens (Ehrenb.) Grun. 1862
Fragilaria construens var. binodis (Ehrenb.) Grun. 1862
Fragilaria construens var. venter (Ehrenb.) Grun. in Van Heurck 1881
Fragilaria brevistriata Grun. in Van Heurck 1885
Fragilaria lapponica Grun. in Van Heurck 1881
Fragilaria leptostauron (Ehrenb.) Hust. 1931
Fragilaria elliptica Schum. 1867
Fragilaria intermedia Grun. in Van Heurck 1881
Fragilaria sp.
Gomphonema acuminatum Ehrenb. 1832
Gomphonema parvulum (Kutz.) Kutz. 1849
Gomphonema minutum (Ag.) Ag. 1831
Gomphonema sp.
Gyrosigma sp.
Meridion circulare (Grev.) Ag. 1831
Navicula radiosa Kutz. 1844
Navicula hungarica Grun. 1860
Navicula seminulum Grun. 1860
Navicula cryptocephala Kutz. 1844
Navicula cryptocephala var. veneta (Kutz.) Rabenh. 1863
Navicula pupula Kutz. 1844
Navicula menisculus Schum. 1867
Navicula cocconeiformis Greg. ex Greville 1855
Navicula minima Grun. in Van Heurck 1880
Navicula graciloides A. Mayer 1919
Navicula bacillum Ehrenb. 1840
Navicula tripunctata (O.F. Mull.) Bory 1822
Navicula subrotundata Hust. 1945
Navicula utermoehlii Hust. 1943
Navicula capitoradiata Germain 1981
Navicula recens (Lange-Bertalot)LB 1985
Navicula [cryptotenella [var. 1]] Ballestera (JR) 1994
Navicula [cf. seminulum] NJA+HB, Eutrophic sites 1992
Navicula sp.
Neidium sp.
Nitzschia amphibia Grun. 1862
Stephanodiscus hantzschii Grun. in Cleve & Grun. 1880
Stephanodiscus parvus Stoermer & Hakansson 1984

Appendix III

Percentage relative abundance data for the main diatom taxa (all those occurring in >1% in at least 2 samples) in the Sunbiggin Tarn core

	Sample Depth (cm)																
Taxa	**0-1**	**2-3**	**4-5**	**6-7**	**8-9**	**10-11**	**15-16**	**20-21**	**25-26**	**30-31**	**35-36**	**40-41**	**45-46**	**50-51**	**55-56**	**60-61**	**65-66**
Cocconeis pediculus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.9	0.3	0.0	0.3	0.3	2.9	2.6	4.7
Nitzschia amphibian	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.1	0.0	0.5	0.0
Fragilaria construens v. *binodis*	0.0	0.0	0.3	3.0	0.3	1.2	1.5	1.3	2.4	1.6	2.9	4.1	2.1	3.5	4.3	2.9	5.4
Fragilaria sp.	0.6	0.3	0.3	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.6	1.6	0.0	1.1	0.0	0.9
Denticula tenuis	0.3	0.0	0.0	1.4	0.0	0.9	0.6	1.0	2.4	0.6	1.2	0.0	0.3	1.6	0.3	2.1	1.6
Fragilaria construens	1.8	1.1	0.6	5.2	4.5	6.9	4.9	4.2	6.4	5.9	6.9	9.5	7.6	6.7	13.3	6.5	6.6
Fragilaria brevistriata	4.8	4.9	4.9	10.7	2.9	6.6	9.4	8.7	5.5	12.8	9.8	14.2	20.4	11.2	12.5	13.0	18.3
Stephanodiscus parvus	0.0	0.6	0.0	0.8	0.0	0.3	0.0	0.3	0.3	1.6	0.6	0.6	0.3	0.8	0.5	1.0	0.3
Fragilaria pinnata	12.7	9.7	10.1	33.9	12.3	16.6	23.7	29.6	19.5	30.9	36.7	36.1	44.5	42.8	31.2	37.8	33.1
Fragilaria leptostauron	0.0	0.0	0.0	0.0	0.3	0.3	0.0	1.3	0.6	0.3	0.0	0.6	0.0	0.0	0.3	0.5	0.3
Fragilaria construens v. *venter*	1.8	2.6	3.1	7.7	6.1	3.9	7.6	7.7	9.5	10.9	8.4	13.3	7.9	5.6	9.3	8.6	9.1
Fragilaria lapponica	0.0	0.0	0.0	0.8	0.3	0.3	0.3	1.0	0.9	0.0	1.4	0.6	0.0	0.0	0.5	0.3	0.9
Achnanthes minutissima	0.9	3.1	0.6	1.7	0.0	0.6	0.6	3.9	2.1	3.1	1.4	0.6	1.0	4.8	1.6	3.1	1.9
Navicula cryptotenella	0.0	0.6	0.3	0.0	0.3	0.3	0.3	0.6	0.6	0.0	0.0	0.0	0.0	0.5	1.3	0.3	0.3
Cymbella minuta	0.6	0.0	0.0	0.3	0.6	0.6	0.0	0.6	0.9	0.3	0.3	0.3	0.0	1.1	0.5	0.0	0.0
Cocconeis placentula	0.9	0.3	0.3	0.3	1.0	0.9	0.6	1.0	0.6	1.3	0.6	0.3	0.3	1.1	0.0	0.3	0.3
Amphora inariensis	0.0	0.0	0.0	0.0	1.6	3.0	1.5	0.6	1.2	1.6	1.7	0.0	0.0	0.5	0.0	0.0	0.0
Fragilaria elliptica	0.0	0.3	0.0	1.4	0.0	0.3	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0
Navicula utermoehlii	0.0	0.3	6.4	1.4	3.9	6.3	6.4	1.0	2.4	1.3	1.2	1.8	1.6	0.3	0.8	1.0	0.3
Achnanthes ziegleri	1.2	2.9	4.9	2.5	2.9	3.3	3.3	1.9	3.7	1.6	1.7	1.2	1.0	1.1	1.3	0.8	0.3
Achnanthes conspicua	7.5	7.1	4.6	2.2	3.9	4.5	5.2	4.5	3.0	2.8	2.9	2.1	0.8	1.9	1.1	2.9	1.6
Amphora pediculus	34.3	41.7	36.8	17.1	31.3	24.1	18.5	15.8	18.0	13.1	13.0	9.2	6.8	11.0	12.5	11.7	10.4
Navicula graciloides	5.7	5.1	7.1	3.0	11.0	5.7	4.0	2.3	4.3	0.6	3.2	1.5	1.0	1.3	2.4	1.8	0.6
Cymbella reichardtii	1.2	3.1	3.7	1.4	4.8	3.0	1.5	3.5	1.8	1.6	2.0	1.2	0.5	0.3	0.0	0.5	0.0
Achnanthes lanceolata	4.5	4.9	6.4	1.9	3.5	6.6	4.9	4.2	6.1	3.8	0.9	1.2	0.5	0.8	0.5	0.0	0.6
Navicula sp.	0.6	1.1	0.9	0.0	0.6	0.3	0.3	0.3	0.6	0.0	0.0	0.3	0.5	0.3	0.0	0.0	0.0
Navicula menisculus	0.6	0.6	0.0	0.0	0.0	0.6	0.6	1.6	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Navicula subrotundata	8.1	7.4	4.0	0.8	2.3	0.0	2.1	1.6	1.5	1.3	0.3	0.0	0.5	0.3	0.3	0.0	0.3
Navicula recens	7.5	0.3	1.5	0.6	3.2	0.3	0.0	0.0	0.9	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Navicula minima	1.8	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0