Options for the remediation of water quality in Ullswater and Derwent Water

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EXECUTIVE SUMMARY

1. Ullswater and Derwent Water are two of the larger lakes in the English Lake District and both are protected by a number of statutory designations Site of Special Scientific Interest, National Nature Reserve and Special Area of Conservation. Draft favourable condition targets recently defined by the Environment Agency's Habitats Directive Technical Advisory Group for total phosphorus are 15 mg m^{-3} for Ullswater and 10 mg m^{-3} for Derwent Water. These are more stringent than the earlier 20 mg m^{-3} total phosphorus suggested by English Nature for both lakes.
2. The purpose of this report is to review whether or not Ullswater and Derwent Water currently meet these new targets, to estimate the nutrient load to the two lakes, to assess the consequence to the nutrient load of various possible management options and to model the effect these will have on water quality. Finally, a preliminary investigation will be made of the possible impact of climate change in altering the water quality in the two lakes under the different nutrient loads.
3. Ullswater is on the mesotrophic-oligotrophic boundary and, based on CEH data from 2005, has good ecological status under the terms of the Water Framework Directive. Ullswater currently complies with the target annual mean concentration of total phosporus: values were 10 and 9.8 mg m^{-3} in 2000 and 2005 respectively.
4. Derwent Water is a mesotrophic lake and, based on CEH data from 2005, has good ecological status for total phosphorus and moderate ecological status for phytoplankton chlorophyll *a*. Fortnightly data from CEH suggest that as an annual mean the concentration of total phosphorus is just below the 10 mg m^{-3} target: the mean concentration between 2000 and 2004 (inclusive) was 8.5 mg m^{-3} and the last time the 10 mg m^{-3} target was exceeded as an annual mean was in 1997.
5. Ullswater has an estimated mean discharge of 9.32 m^3 s^{-1} (equivalent to 294 10^6 m^3 y^{-1}) and a total load of 1604, 103 329 and 836 850 kg y^{-1} for soluble reactive phosphorus nitrate-nitrogen and silica respectively. The equivalent load of total phosphorus was estimated to be between 2466 and 3088 kg y^{-1}, depending on the conversions of soluble reactive to total phosphorus used. WwTWs contribute about 26% of the soluble reactive phosphorus load and between 15% and 19% of the total phosphorus load. Using an export coefficient approach that included inputs from livestock, the load of total phosphorus from the catchment was estimated to be 3738 kg y^{-1}: much more than the catchment contribution estimated from stream and discharge monitoring. The load and discharge

yield an average concentration of total phosphorus of between 8.4 and 10.5 mg m^{-3}, depending on the calculation method, lying either side of the measured concentration of 9.8 mg m^{-3}. Annual mean concentrations of 10, 15 and 20 mg m^{-3} would be achieved by annual total phosphorus loads of 2940, 4410 and 5880 kg y^{-1} respectively.

6. The hydraulic inflow to Derwent Water is more uncertain than for Ullswater because there is not a monitoring station on the outflow. Best estimates are for an average discharge of 7.27 $m^3 s^{-1}$ (229 $10^6 m^3 y^{-1}$). The total loads of soluble reactive phosphorus, nitrate-nitrogen and silica are estimated to be 848, 71037 and 511841 respectively. The total load of total phosphorus is estimated to be between 1189 and 1417 kg y^{-1} depending on the conversion factor between soluble reactive and total phosphorus. WwTW contribute about 49% of the load of soluble reactive phosphorus and between 32% and 38% of the load of total phosphorus. Using an export coefficient approach that included inputs from livestock, the catchment load was estimated to be 1746 kg y^{-1}, more than the estimate from inflow chemistry. The concentration of total phosphorus in the lake calculated from the hydrology and loads are between 5.2 and 6.2 mg m^{-3}, depending on the calculation method, being much lower than the measured concentration of 8.5 mg m^{-3}. Annual mean concentrations of 10 and 8 mg m^{-3} would be achieved by annual total phosphorus loads of 2290, and 1839 kg y^{-1} respectively and the export coefficient estimate of load is fairly similar to this.
7. Estimates of conversion of total phosphorus in phytoplankton chlorophyll a are relatively uncertain. However the best estimate is that setting an upper limit on the WwTw at Glenridding will only cause and 8% reduction on average chlorophyll *a* inUllswater whereas setting the same limit on all the WwTW in Derwent Water will cause a 24% reduction in chlorophyll *a*. This will be sufficient to return the lake to good ecological status under the Water Framework Directive and is a recommended management option.
8. PROTECH accurately simulated the phytoplankton timing, abundance and type in both lakes but suggested that the lakes are less responsive to phosphorus loading than the other ‘budget approaches’ suggest.
9. PROTECH models of the response of the lake to changing water temperature, in response to higher air temperature resulting from climate change, suggest a modest reduction in overall chlorophyll concentration in both lakes at higher temperatures but a larger shift in composition towards cyanobacteria in Ullswater. These are preliminary results and other climate change factors such as changed patterns of rainfall may have larger effects.

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1. INTRODUCTION

Ullswater and Derwent Water are two of the largest lakes in the English Lake District: Ullswater has the second largest volume and area after Windermere; and Derwent Water has the sixth-largest volume and third largest area (Table 1).

The statutory designations for the two lakes are given below:

Ullswater
Site of Special Scientific Interest (1984)
National Nature Reserve (1993)
Special Area of Conservation (SAC) under the European Habitats Directive- River Eden SAC.

Derwent Water
Site of Special Scientific Interest (1984)
National Nature Reserve (1993)
Special Area of Conservation (SAC) under the European Habitats Directive- River Derwent and Bassenthwaite Lake SAC.

Draft favourable conditions targets have been defined by English Nature for the SAC designations under the Conservation (Natural Habitats & c.) Regulations 1994. These are given in Annex A for Ullswater and Annex B for Derwent Water.

The water quality objectives for the two lakes are for an annual mean total phosphorus concentration of not more than 20 mg m^{-3} (NB equivalent to 20 µg L^{-1}; mg m^{-3} will be used in this report). The targets have, however, recently been revised by the Environment Agency's Habitats Directive Technical Advisory Group WQTAG (WQTAG, 2005). Using approaches similar to the Water Framework Directive, Ullswater has been classified as a medium alkalinity deep lake and Derwent Water has been classified as a low alkalinity shallow lake. The targets for total phosphorus using this approach are 15 mg m^{-3} for Ullswater and 10 mg m^{-3} for Derwent Water.

Table 1. Geographical and physical characteristics of the 20 lakes that comprise the CEH 'Lakes Tour' with Derwent Water and Ullswater shown in bold (after Maberly et al*., 2006).*

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

There are four objectives to the project:

1. Provide a brief synopsis of relevant previous phosphorus budget work and nutrient history from historical limnological studies for each lake.

2. Identify the nutrient loading to Ullswater and Derwent Water required to meet total phosphorus annual means of:

- Ullswater: 20, 15 and 10 mg m^{-3}
- Derwent Water: 10 and 8 mg m^{-3}

3. Calculate the change in water quality in Ullswater and Derwent Water that would result from:

- reducing the phosphorus load to achieve the targets identified in Objective 2
- Ullswater: reducing the final effluent phosphorus limit at Glenridding WwTW to 1 mg L^{-1} as an annual average (compared to the existing situation)
- Derwent Water: reducing the final effluent phosphorus limit on WwTW discharges upstream of Derwent Water to 1 mg L^{-1} as an annual average (compared to the existing situation)
- applying a 15% reduction in diffuse loading in each lake catchment
- one other scenario to be identified – assessing proportional/alone impacts using metabolic modelling or other agreed methods.

4. Undertake PROTECH modelling of Ullswater and Derwent Water for each of the scenarios identified in Objective 3 and use PROTECH to assess the potential impact of climate change on lake water quality.

2. Objective 1: Synopsis of relevant phosphorus budgets and nutrient history from limnological studies

2.1 Ullswater

Ullswater is the second largest lake in the English Lake District after Windermere in terms of area and volume and the largest if Windermere is separated into two basins. It is situated in the north-east of the English Lake District and drains eventually into the River Eden.

A comprehensive review of the ecology of Ullswater was made in 1992 (Talling *et al*., 1992). The basic information on physical characteristics and basic chemistry will be unchanged and since Ullswater is generally a relatively stable lake there are unlikely to be major changes in other aspects of its ecology. Since 1992, some more detailed studies have been carried out on Ullswater (e.g. Hall *et al*., 1999; 2000). Ullswater is also included in the regular 'Lake Tours' carried out by CEH and its predecessors and the values for 2005 are summarised in Table 2. Based on this information, Ullswater is classified as a mesotrophic lake with some features tending towards oligotrophy (Table 2). In terms of the WFD, and using current class boundaries for a deep lake with medium alkalinity, the lake has a good ecological status.

Table 2. Limnological characteristics of Ullswater in 2005 (based on CEH 'Lakes Tour data; Maberly et al*. 2006).*

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

Analysis of the 'Lakes Tour' data for Ullswater show no statistically significant long-term trends apart from an increase in concentration of silica in spring. The pattern of change for total phosphorus is shown in Figure 1. The annual mean concentration exceeded the suggested 15 mg m^{-3} favourable conservation target in 1995 but was substantially below that target in 2000 and 2005.

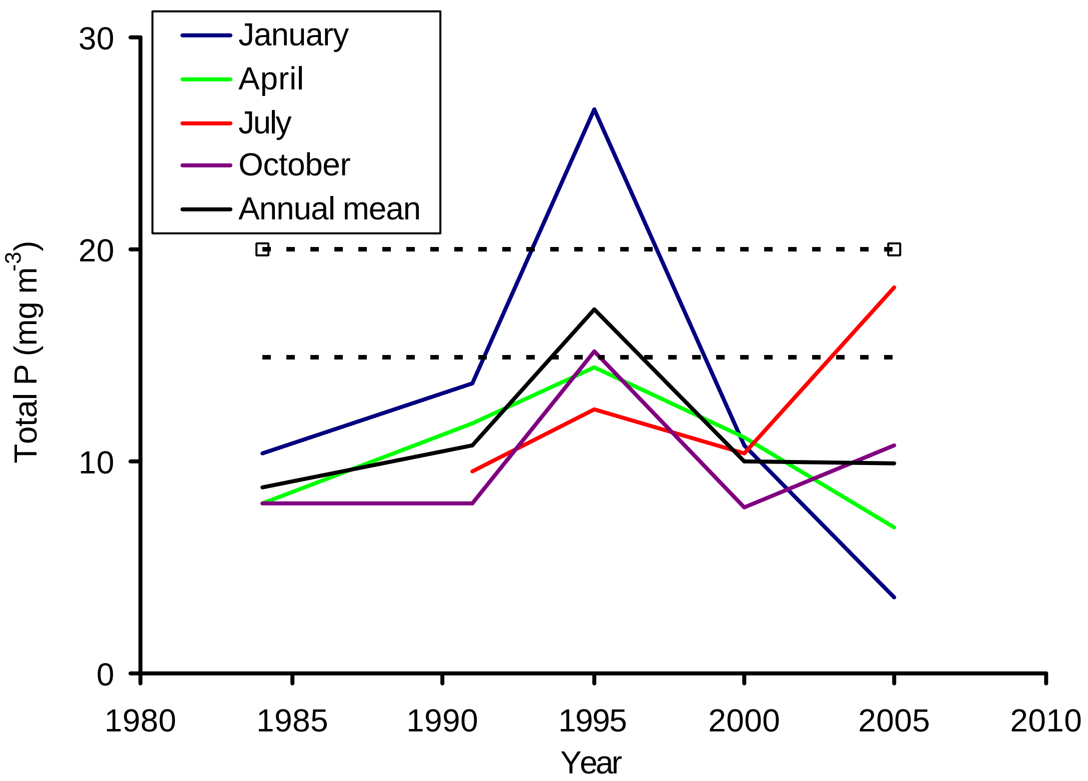


Figure 1. Long-term data derived from CEH Lakes Tour data (FBA data for 1984) on Ullswater at four seasons per year sampled over the deepest point in the lake. The horizontal lines show the suggested favourable conservation annual mean concentrations of 20 and 15 mg m^{-3}.

Data of higher frequency, and distinguishing between the North and South Basins, collected by the Environment Agency between 2000 and 2005 are shown in Figure 2. Based on these data, the annual mean in 2005 was 25 mg m^{-3} in the North Basin and 21 mg m^{-3} in the South Basin and so both are substantially higher than the conservation target of 15 mg m^{-3}. These data, however, appear to show an extremely high variability in total phosphorus, with minima below 2 mg m^{-3} and maxima reaching over 100 mg m^{-3}. This variability does not match that found by CEH in fortnightly sampling between 1997 and 1999 at a location close to the Environment Agency's North Basin (Fig. 2) and is extremely hard to rationalise scientifically since total phosphorus is normally fairly conservative, especially in a large lake with a relatively long retention time.

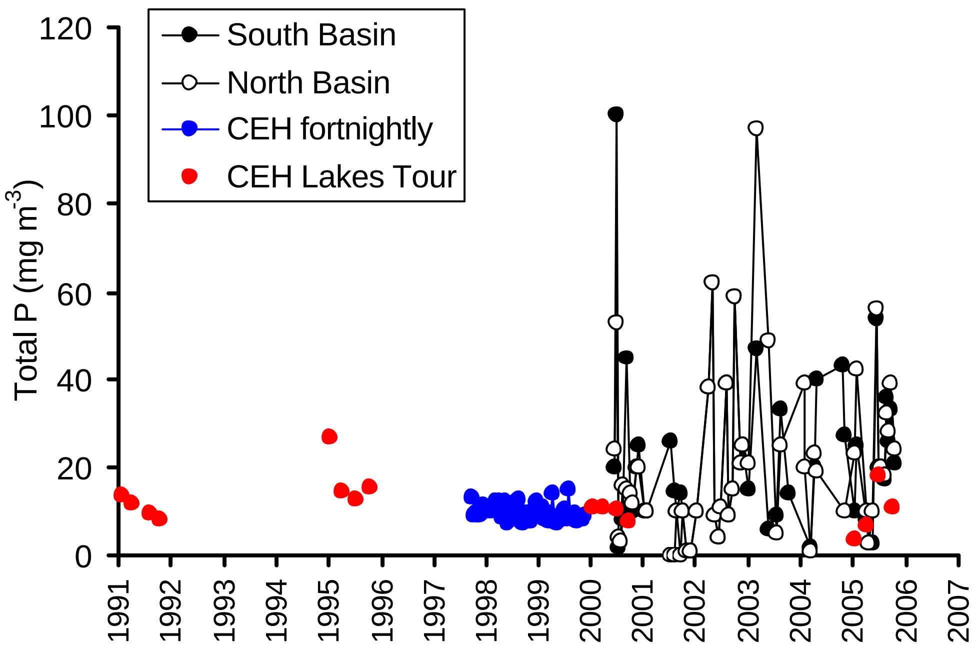


Figure 2. Environment Agency seasonal data on the North and South Basin of Ullswater from summer 2000 to autumn 2005. Also shown are CEH fortnightly data during 1997 to 1999 (Hall et al., 2000) and CEH Lakes Tour data for 1991, 1995, 2000 and 2005 (Hall et al., 1992, 1996, Parker et al., 2001, Maberly et al., 2006).

The CEH 'Lakes Tour' data which is only based on four samples per year, give a mean concentration of TP in Ullswater in 2005 of 9.8 mg m^{-3}, with a range of from 3.6 mg m^{-3} in January and 18.2 mg m^{-3} in July (Maberly *et al.*, 2006). However, fortnightly data from CEH exist between September 1997 and November 1999. This had a virtually identical mean of 9.7 mg m^{-3} (Hall *et al.*, 2000) and a limited range of between 7 and 14 mg m^{-3} (Fig. 2). The Lakes Tour data from 1995 to 2005 inclusive have a slightly higher mean value of 12.3 mg m^{-3} largely because concentrations of TP tended to be high in 1995. Averaged over all the data from 1984 onwards the mean concentration is 11.4 mg m^{-3}. Based on data from CEH, therefore, the lake appears to be complying with the WQTAG (2005) requirement for the mean concentration of TP to be below 15 mg m^{-3} but the data from the Environment Agency suggests it is not. It is currently unclear why there is a large difference between data collected by the Environment Agency and CEH.

2.2 Derwent Water

Derwent Water is situated in the north-west of the English Lake District (central National Grid Reference NY260210), near the town of Keswick. It is located in a broad basin, Borrowdale, that also contains Bassenthwaite Lake into which Derwent Water flows. The solid geology comprises the weathering-resistant Borrowdale volcanic series in the south and the softer Skiddaw slates to the north and west. The lake basin itself is situated on a mixture of morainic and alluvial material.

Derwent Water is the third largest lake in the English Lake District in terms of area (Table 1). It is also one of the shallowest with a mean depth of only 5.5 m although the maximum depth extends to 22 m. The main inflow, the River Derwent, rises in the Borrowdale fells at the southern end of the lake. A second, smaller inflow, Watendlath Beck, also enters the lake at the southern end. The lake has a relatively large catchment area in relation to lake volume and the high rainfall on much of the catchment yields an average retention time of about 55 days, which is the fourth shortest of the major Cumbrian lakes (Talling, 1999). The main outflow, the River Derwent, leaves the lake at a natural sill barrier at the Northern end, close to Keswick. It is joined by the River Greta 400 m downstream and flows 5.5 km into Bassenthwaite Lake. There is evidence for a fall in minimum lake level since the late 19th Century, caused by a reduction in the level of the sill at the outflow (Ove Arup, 1999).

The data from the CEH Lakes Tour in 2005 (Maberly *et al*., 2006) suggest that Derwent Water is mesotrophic on all measures apart from the minimum secchi depth which is on the oligotrophic/mesotrophic boundary (Table 3). Classifications for the WFD is only moderate for phytoplankton chlorophyll *a*. It should be stressed that the ecological boundaries for the WFD have not been finally decided, but if they remain as at present, Derwent Water might fail to reach good ecological status in terms of phytoplankton chlorophyll *a*. The concentrations of TP are clearly at least good and close to the high:good boundary (Fig. 6).

Table 3. Summary of limnological conditions and trophic and Water Framework Directive classifications in Derwent Water in 2005. (Based on data from CEH/EA Lakes Tour in 2005, Maberly et al. *2006).*

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

Derwent Water has been monitored by CEH (and its predecessor, the Institute of Freshwater Ecology) every fortnight as part of a long-term programme since August 1990, so the extent of data available is much greater than for Ullswater. Variables measured include depth profiles of oxygen and temperature, secchi depth, concentration of phytoplankton as chlorophyll *a*, phytoplankton species composition, concentrations of soluble reactive phosphorus (SRP), total P, NO_3-N, NH_4-N, SiO_2, alkalinity and pH. The results from this monitoring programme have been summarised in a series of reports (Jaworski *et al*., 1991, 1992, 1993, 1994; Reynolds *et al*., 1995, 1996, 1997, 1998, 1999, 2001a, 2002; Maberly et al. 2003, 2004, 2005).

Seasonal patterns of chemical changes based on these data are shown in Figure 3. The concentrations of nutrients and phytoplankton chlorophyll *a* are the variables which are most relevant to this report and will, therefore be concentrated on. There is little seasonal change in the concentration of total phosphorus or SRP: the latter is at or close to the limit of detection for much of the year. Nitrate shows a strong seasonal pattern with summer depletion to a monthly minimum in August. Ammonium is present in low concentrations for most of the year with highest concentrations in August and September, probably associated at least in part with erosion of the thermocline and entrainment of nutrient from depth. Silica shows a similar seasonal pattern to nitrate but with an earlier depletion in June and July brought about by diatom growth in spring.

On average phytoplankton increase from an overwinter concentration of chlorophyll *a* of about 3 mg m^{-3} to a spring peak of about 6 mg m^{-3} followed by a mid-summer concentration of about 6.6 mg m^{-3} declining back to overwinter concentrations in October as the lake destratifies (Fig. 3). This level of phytoplankton biomass is low compared to many other lakes in Cumbria, particularly the more productive sites within the Windermere catchment (Talling, 1993).

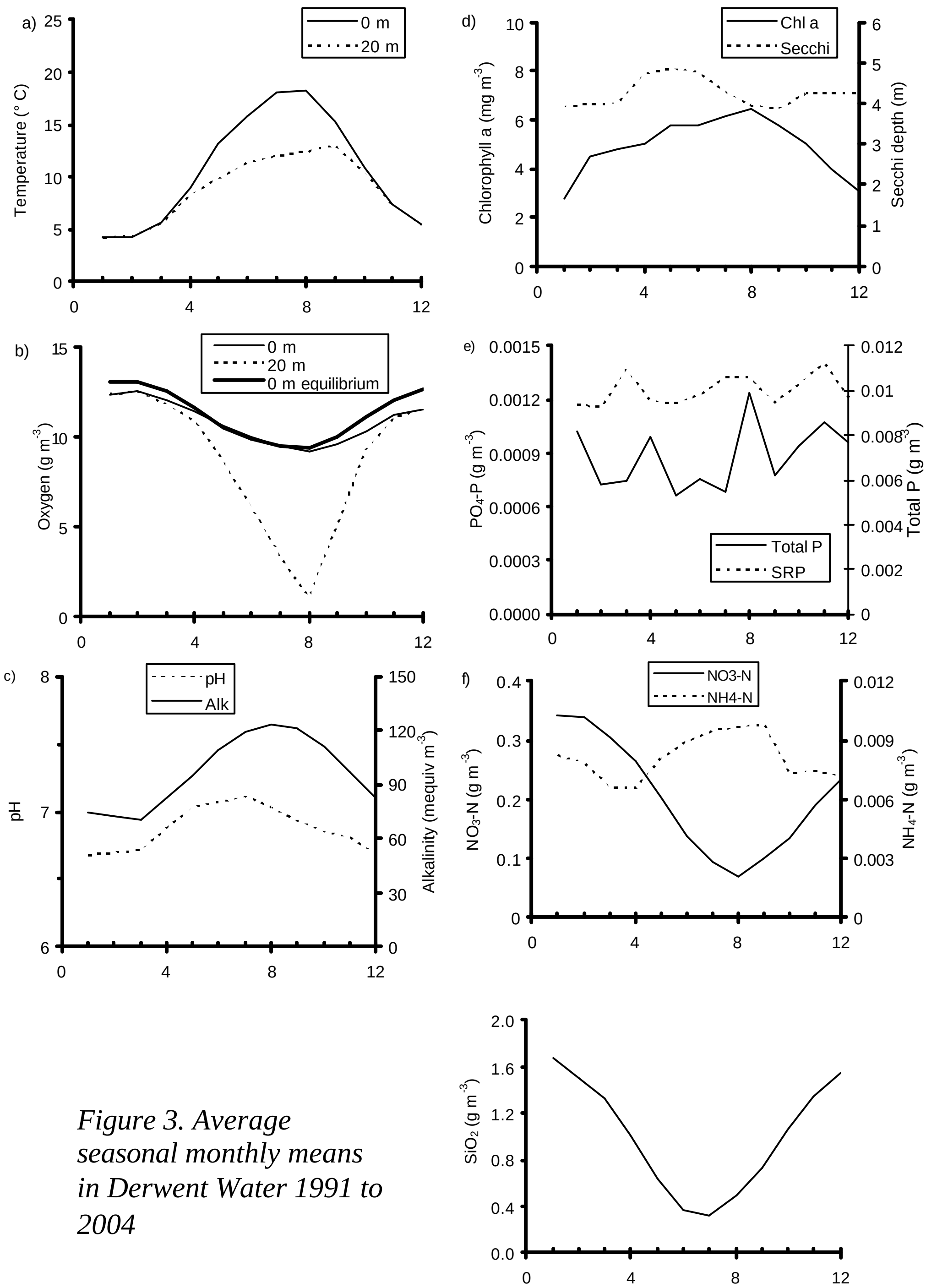


Figure 3. Average seasonal monthly means in Derwent Water 1991 to 2004

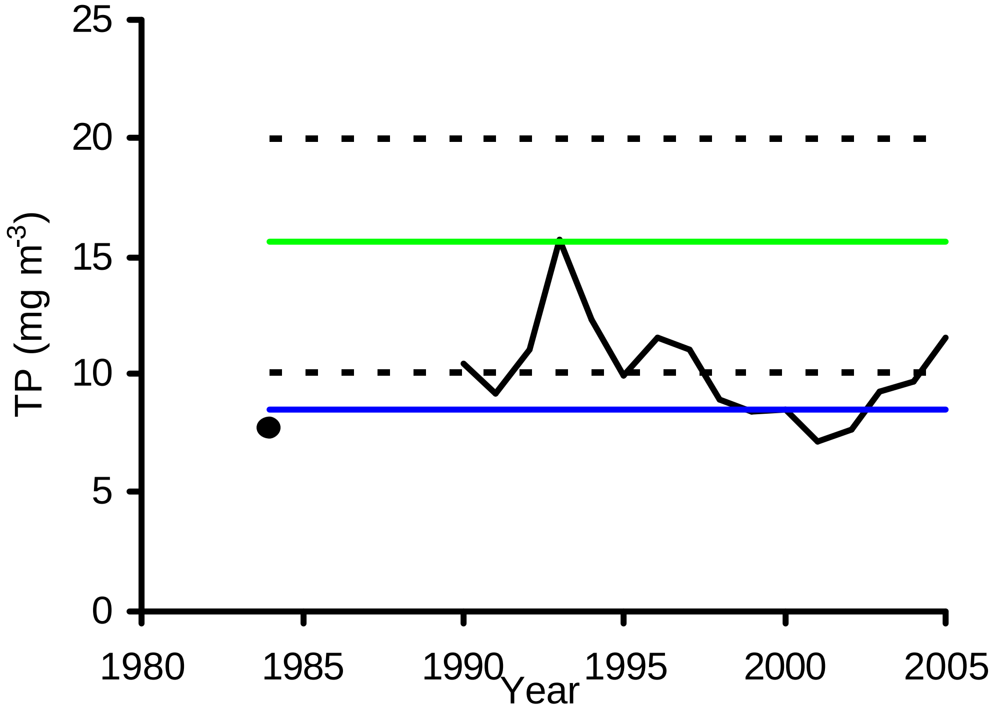


Figure 4. Long-term change in average concentration of total phosphorus in Derwent Water based on fortnightly measurement. The upper and lower black dotted lines show the 20 and 10 mg m^{-3} favourable conservation status concentrations. The green and blue lines show the site-specific mean concentration of total P for the high: good and good : moderate boundary in the Water Framework Directive (Maberly et al*., 2006). Fortnightly data from 1990 onwards, single point in 1984 based on 'Lakes Tour' data.*

Initial draft favourable condition targets for Derwent Water, defined by English Nature for the Special Area of Conservation regulations, 1994, give an annual mean total phosphorus concentration of 20 mg m^{-3}. This is clearly much higher than the concentration that has been experienced in Derwent Water recently (Fig. 4). These TP concentrations have been revised by the Environment Agency's Habitats Directive Technical Advisory Group, WQTAG (2005) which has given a target of 10 mg m^{-3} for annual mean concentration of TP. The mean value in 2004 was 9.7 mg m^{-3} and the last time that 10 mg m^{-3} was exceeded was in 1997 apart from the most recent complete year, 2005, with a mean of 11.6 mg m^{-3} (Fig. 4). The mean concentration between 2000 and 2005 (inclusive) was 9.0 mg m^{-3}, Derwent Water therefore appears to be at, or possibly just below this target but there has been a trend of increasing concentration of TP in the last five years (Fig. 4).

3. Objective 2: Nutrient loading to Ullswater and Derwent Water

This section of the report estimates loads of nutrients to Ullswater and Derwent Water and the consequent concentration of total phosphorus based on chemistry and flow data provided by the Environment Agency. The first step is to provide best estimates of loads of nutrient and hydrological discharge and then use this, in conjunction with lake volume in Table 1, to estimate annual mean concentrations. The focus will be on phosphorus, as the key nutrient limiting phytoplankton productivity in both lakes, but loads will also be estimated for nitrogen and silica as these are needed to drive the lake model PROTECH. Although for some of the sites many years of data are available, the data used here are from 1/1/1998 to 31/12/1999 inclusive to make the data comparable with the PROTECH modelling work presented later in the report. The temporal-resolution of the data is relatively crude, generally monthly, and so the load estimates will necessarily be relatively imprecise.

3.1 Ullswater

3.1.1 Hydrology

The flow data provided by the Environment Agency for Ullswater are summarised in Table 4. Of the five monitored streams, Goldrill Beck contributed the largest hydraulic load: about 52% of the streams monitored (Table 4).

The average hydraulic flow at the discharge from Ullswater at Pooley Bridge between 1/1/1998 and 31/12/1999 was 9.32 $m^3\ s^{-1}$ (294 $10^6 m^3\ y^{-1}$). This is 3.03 $m^3\ s^{-1}$ greater than that accounted for by the five inflowing streams in Table 4 which had an average discharge over the same period of 6.29 $m^3\ s^{-1}$. The unaccounted for discharge was therefore taken as 3.03 $m^3\ s^{-1}$ in the calculations of load below.

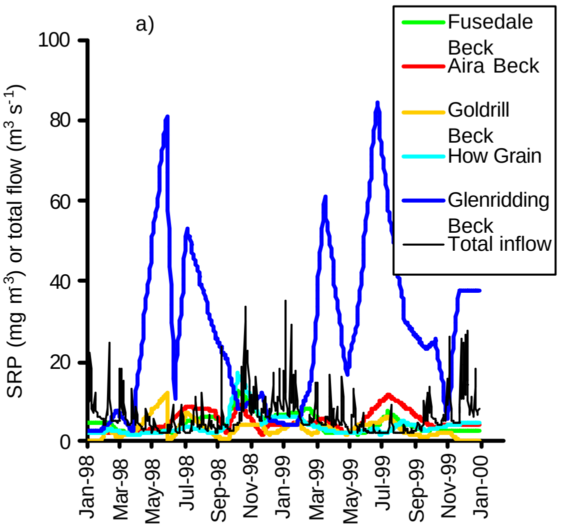
Table 4. River GQA data used in the project for Ullswater. Monthly flow estimates were produced by Low Flows 2000 software.

Site number	Site	Grid Ref	Flow data
INFLOWS			
88009739	Goldrill Beck	NY 3940 1650	Daily flow data
88006210	Glenridding Beck	NY 3870 1692	Monthly flow estimates
88006211	Aira Beck	NY 4009 1978	Monthly flow estimates
88006218	How Grain (Sandwick Beck)	NY 4239 1986	Monthly flow estimates
88006232	Fusedale Beck	NY 4435 1970	Monthly flow estimates
OUTFLOWS			
88006262	R. Eamont at Pooley Bridge	NY 4699 2445	Daily flow data

3.1.2 Nutrient loads

The average flow-weighted concentration of SRP flowing into Ullswater was quite high at 7.9 mg m^{-3} (Table 5). Glenridding Beck had the highest annual concentrations of SRP, with spot concentrations up to 83 mg m^{-3} (Fig. 5), and was also the main contributor to the load, contributing 52% despite only contributing 14% of the hydraulic load. This is presumably the result of input from the Glenridding WwTW upstream from the GQA site, which had an average daily flow of 93.75 m^3. This is supported by the seasonal changes of SRP concentration in this stream which peak around Easter and in the summer (Fig. 5). The other streams lacked a very strong seasonality and SRP concentration remained below 20 mg m^{-3} in 1998 and 1999 and at times fell to concentrations approaching the detection limit.

The loads of nitrate and silica were broadly proportional to hydraulic discharge as the range of concentrations were relatively constant, although Goldrill Beck and How Grain had slightly elevated concentrations of nitrate and silica concentrations were high in Fusedale Beck. At times the load from Goldrill Beck exceeded that from Glenridding Beck although a direct comparison is made difficult by the different temporal distribution of the hydrological data: monthly at Glenridding Beck but daily at Goldrill Beck (Fig. 6). Overall, Glenridding Beck was the major source of SRP to the lake, followed by Goldrill Beck (Table 5).



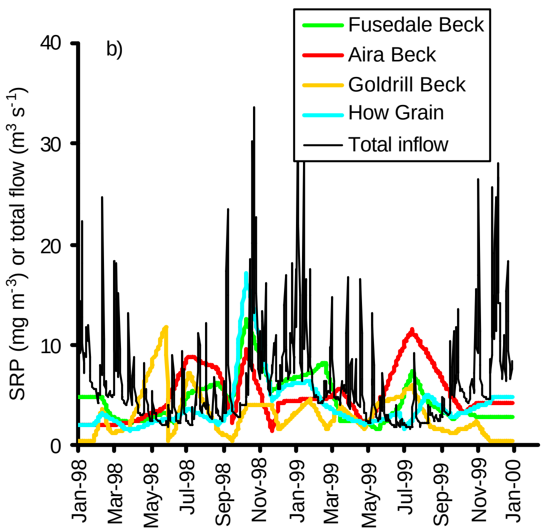


Figure 5. Seasonal changes in concentration of SRP and total inflow a) including and b) excluding Glenridding Beck.

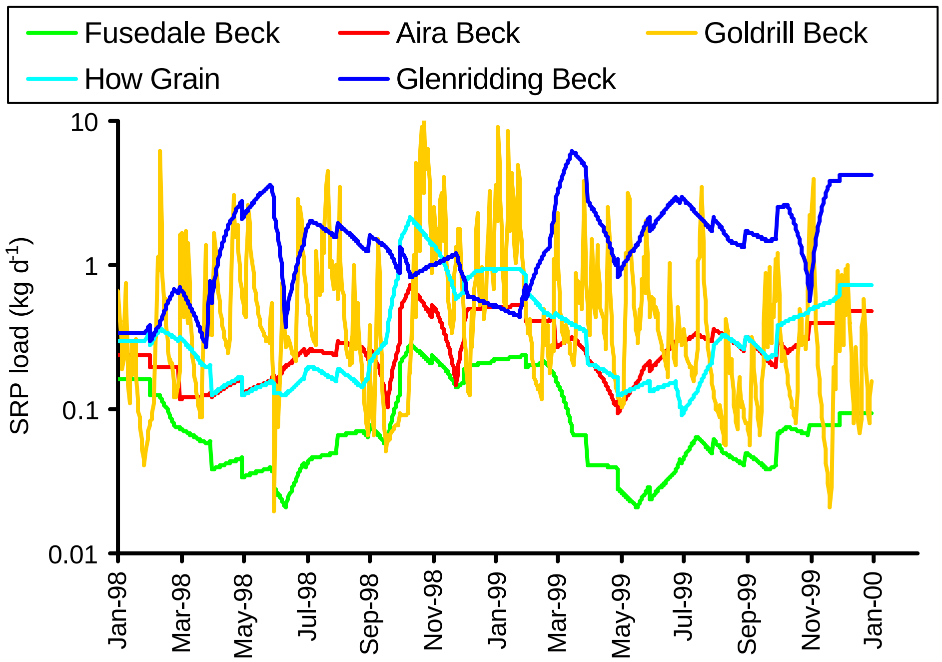


Figure 6. Seasonal patterns of SRP load to Ullswater from the five monitored inflow streams. Note the logarithmic scale on the y-axis

Table 5. Hydraulic discharge (Q), mean concentration and annual load for five streams flowing into Ullswater. Based on daily interpolated data between 1 Jan 1998 and 31 December 1999 (see text). Values in parenthesis give value as percent of the total. The TOTAL (whole catchment) is an estimate that includes the additional load of SRP from parts of the catchment that are not monitored (see text for more information).

		Mean concentration ($mg\ m^{-3}$)			Load ($kg\ y^{-1}$)		
Stream	Q ($m^3\ s^{-1}$)	SRP	NO_3-N	SiO_2	SRP	NO_3-N	SiO_2
Goldrill Beck	3.28 (52)	3.1	359	2239	275 (23)	36 840 (53)	242 534 (22)
How Grain (Sandwick Beck)	1.14 (18)	4.0	416	3424	159 (13)	17 079 (24)	124 710 (11)
Glenridding Beck	0.89 (14)	27.4	319	2865	624 (52)	8 698 (12)	80 814 (7)
Aira Beck	0.76 (12)	5.0	252	3284	107 (9)	5 827 (8)	80 905 (7)
Fusedale Beck	0.23 (4)	4.6	223	4386	34 (3)	1603 (2)	32 309 (3)
Mean (flow weighted)	-	7.9	343	2864	-	-	-
Total measured	6.29	-	-	-	1200	70 046	561 272
Unaccounted	3.03	-	-	-	354	33 253	275 578
TOTAL (whole catchment)	9.32	-	-	-	1554	103 229	836 850

There is an estimated 3.03 $m^3\ s^{-1}$ of water entering Ullswater that is not included in the monitored streams. The flow weighted concentration for all the inflowing streams to Ullswater was 7.9 $mg\ m^{-3}$ (Table 5) but this included the high concentrations from Glenridding Beck. Instead of using this concentration, a flow-weighted mean of 3.7 $mg\ m^{-3}$ was calculated for the four other streams and this was multiplied by the unaccounted for hydraulic discharge, 3.03 $mg\ m^{-3}$, to produce an additional unaccounted for load of 354 $kg\ y^{-1}$ yielding a total load of 1554 kg SRP y^{-1}. The equivalent extra loads were 33 253 and 275 578 $kg\ y^{-1}$ for nitrate and silica respectively (Table 5).

The final component of the nutrient budget is input from known WwTW that are not already included because they are upstream of the GQA sites. Table 6 gives the consented discharges to Ullswater and shows that the discharges from Brackenrigg Hotel and Leeming House Hotel discharge into surface waters while those from Bank House, Sharrow Bay and

Rampsbeck Hotel discharge to groundwater and are so assumed to be retained in the catchment.

Table 6. Discharge data used in the project for Ullswater. Discharges upstream (US) of GQA sites and therefore already included in the measured data are noted. Maximum consented volumes as dry weather flow (DWF), average daily flow (ADF) and flow to full treatment (FTFT) are given, values for ADF in parenthesis are estimated from FTFT (see text).

					Max Consented volume (m^3 d^{-1})		
Site No.	Site	Grid Ref	Receiving water	Notes	DWF	ADF	FTFT
88010791	Patterdale YHA	NY 3994 1570	Goldrill Beck	US of GQA		(10.9)	16
88006042	Patterdale WwTW	NY 3976 1592	Goldrill Beck	US of GQA		(32.2)	50
88006002	Glenridding WwTW	NY 3845 1685	Glenridding Beck	US of GQA	75	93.75	679
88021924	Bank House, Sharrow Bay	NY 4453 1998	Soakaway to groundwater			(2.25)	3.5
88021752	Sharrow Bay	NY 4550 2221	Soakaway to groundwater			(19.33)	30
88021920	Rampsbeck Hotel	NY 4520 2320	Soakaway to groundwater			(6.76)	10.5
88009708	Brackenrigg Hotel	NY 4480 2320	Unnamed beck			(3.22)	5
88021624	Leeming House Hotel	NY 4428 2175	Unnamed beck			(12.56)	19.5

Where necessary, average daily flow was estimated from full flow to treatment using the 95 percentile value and calculating the mean and standard deviation to meet this using a log-normal distribution and a coefficient of variation (standard deviation/mean) of 0.3. The discharge data from Brackenrigg House and Leeming House are sparse with only one data point (from 1995) for the former and eight data points from 2004 to 2005 for the latter. The average values and calculated loads are presented in Table 7.

Table 7. Average daily discharge, mean concentrations of SRP and nitrogen (ammonium plus oxidised nitrogen) for WwTW not already included in GQA monitoring sites on Ullswater.

Site	Discharge (m^3 d^{-1})	Mean SRP (g m^{-3})	SRP load (kg y^{-1})	Mean N (mg m^{-3})	N load (kg y^{-1})
Brackenrigg Hotel	3.22	3.05	3.6	29.1	34.2
Leeming House Hotel	12.56	9.96	45.7	14.44	66.2
TOTAL	**-**	**-**	**49**		**100**

The total estimated loads of SRP, inorganic nitrogen and silica are given in Table 8.

Table 8. *Summary of loads of SRP, inorganic nitrogen and silica to Ullswater (kg y^{-1}).*

Source	SRP	N	SiO_2
Catchment plus WwTW	**1 554**	**103 229**	**836 850**
Other WwTW	49	100	-
TOTAL	**1 603**	**103 329**	**836 850**

3.1.3 PROTECH input

For PROTECH, daily total discharge between 1/1/1998 and 31/12/1999 inclusive were calculated based on the known inflows scaled-up so the total average discharge equalled the average value of 9.32 $m^3\ s^{-1}$ estimated in Table 5.. The concentrations of SRP, nitrate-nitrogen and silica were individually up-scaled so that the annual load agreed with Table 8.

3.1.4 Catchment loads vs WwTW loads

In the calculations above, three WwTW are upstream of GQA sampling points and the loads from the streams are therefore a combination of point and diffuse sources (Table 6). An estimate is needed of the contribution of the diffuse load from the catchment compared to the point source load from WwTW in Objective 3. This was estimated by calculating the contribution of the three WwTW to the stream load from data on average daily flow and mean concentration.

Table 9. Partitioning SRP loads from point and diffuse sources to Ullswater.

Discharge	Discharge Beck	Average daily flow ($m^3\ d^{-1}$)	Average SRP ($g\ m^{-3}$)	Annual load ($kg\ y^{-1}$)
Glenridding WwTW	Glenridding	93.75	8.65	296 (47%)[b]
Patterdale YHA	Goldrill	10.9	(5)[a]	20 (7%)[b]
Patterdale WwTW	Goldrill	32.2	(5)[a]	59 (21%)[b]
Total	**-**	**-**	**-**	**375**
Load from other WwTw (Table 8)				49
Total load from WwTW				423 (26%)[c]
Total load from catchment				1 180 (74%)[c, d]
Total load to lake (Table 8)				1 603

a. No data available, concentration estimated from mean value for other WwTW.
b. Percent of load of named stream.
c. Percent of total load.
d. Calculated by difference.

Estimates in Table 9 suggest that 47% of the SRP load in Glenridding Beck is contributed by Glenridding WwTW and 28% of the SRP load in Goldrill Beck is contributed by the two point sources in Patterdale. Overall, point sources are estimated to contribute 26% and diffuse sources 74% of the total SRP load. These are rough estimates in the light of the calculations needed above and the coarse sampling frequency. Furthermore, some of the catchment load is likely to result from small point discharges, such as from septic tanks, that are not monitored.

3.1.5 Loads of SRP vs TP in Ullswater

The sections above have been concerned with loads of SRP rather than TP for two reasons. First, this is the nutrient needed to run PROTECH and secondly the data on TP are too sparse to be useful. For example, for the six point source discharges, there were 99 determinations of SRP but only one for TP. For the inflowing streams such as Goldrill Beck and Glenridding Beck there were 139 and 135 measurements of SRP above the detection limit but only 16 and 13 respectively for TP. The total number of TP measurements were 63 and 33 on Goldrill Beck and Glenridding Beck respectively, indicating the relatively large number of readings below the detection limit. While an average concentration of SRP and TP could be calculated and used to produce a ratio of SRP:TP to estimate TP loads this is likely to be biased by the values less than the detection limit. Instead, for the catchment sources the ratio of SRP:TP was assumed to be either 0.59 (Hilton *et al*., 1993) or 0.45 (May *et al.*, 1997, excluding high values indicative of point sources; May *pers. comm*.). For the WwTW the ratio of SRP:TP was assumed to be 0.91 based on values from Keswick WwTW (Maberly & Elliott, 2002).

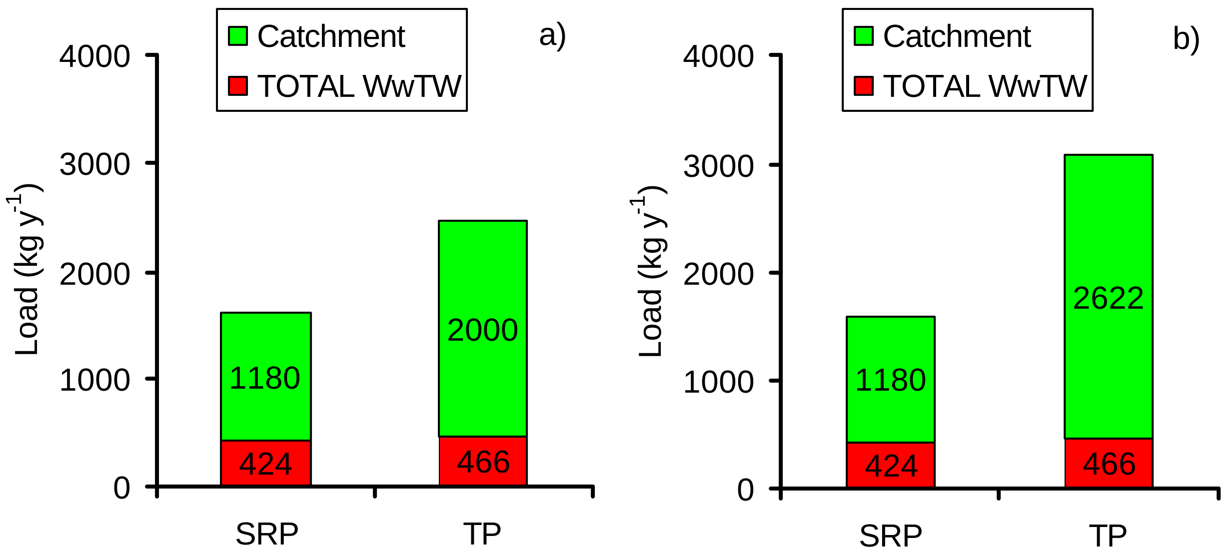


Figure 7. Estimated loads of SRP and TP to Ullswater using an SRP:TP ratio of 0.91 for the WwTW and for the catchment: a) 0.59 and b) 0.45, (see text).

Figure 7. gives the estimated loads of SRP and TP calculated as described above. The known point sources contribute 26% of the total SRP load of 1604 kg y^{-1} and 19% of the total load of TP of 2466 kg y^{-1} if the SRP:TP ratio of 0.59 is used (Fig. 7a). However the loads of TP are greater if a SRP:TP ratio of 0.45 is used (Fig. 7b): point sources then contribute 15% of the total load of TP which is estimated at 3088 kg y^{-1}. The two calculations above highlight the uncertainty in calculating nutrient loads to a lake.

It is noted here for completeness that Tipping *et al.* (1997) estimated the loads of nutrients to Ullswater based on a fortnightly monitoring exercise on seven inflow streams carried out by the Environment Agency between September 1994 and December 1995. In addition, data was collected from four WwTWs not included in the stream measurements. The load of SRP is higher than that estimated here at 2644 kg y^{-1} (compared to 1604 kg y^{-1} in this report). However, the loads of TP are extremely high at 21490 kg y^{-1} (compared to between 2466 and 3088 kg y^{-1} in this report) and seem to indicate an error in the measurement or calculations. This work will not, therefore, be considered further.

The load from the catchment divided by the catchment area (Table 1) gives an average export coefficient of 0.14 kg P ha^{-1} y^{-1} using the SRP:TP ratio of 0.59 and 0.18 kg P ha^{-1} y^{-1} using the SRP:TP ratio 0.45. This can be checked roughly using the export coefficient approach which ascribes an average TP loss per unit area for different types of land using recently revised values in Carvalho *et al*. (2003). Total load estimated using this approach (Table 10) is 1163 kg TP y^{-1} which is equivalent to an average export coefficient of 0.08 kg ha^{-1} y^{-1} in other words it is less than both of the two export coefficients estimated above from inflow data. However, in the revised approach of Carvalho *et al.* (2003) loads from pigs, sheep or cattle in the catchment are identified separately and these have been estimated for Ullswater at 2575 kg y^{-1} (Carvalho *et al*. 2003) Table 10. This gives an estimated diffuse load of 3738 kg P y^{-1}, equivalent to an average export coefficient from land and livestock of 0.25 kg ha^{-1} y^{-1} and between 43 and 87% more than the estimates from the stream load calculations. The estimate of P from people in Carvalho *et al*. (2003) of 340 kg y^{-1} is quite close to that estimated here at 466 kg y^{-1}. It should be noted that Carvalho *et al.* (2003) point out that their approach is not suitable at a site-specific level.

Table 10. Estimate of TP loads from the catchment to Ullswater based on an export coefficient approach using revised export coefficients in Carvalho et al. *(2003). The load from livestock is also taken from Carvalho* et al. *(2003).*

Land cover type	Land-cover area (km^2)	Export (kg TP ha^{-1} y^{-1})	TP Load (kg P y^{-1})
Bracken	41.6	0.02	83.1
Coniferous Woodland	0.4	0.15	6.4
Continuous Urban	0.5	0.83	43.9
Deciduous Woodland	3.3	0.02	6.5
Dense Shrub Moor	0.6	0.02	1.1
Felled Forest	0.1	0.2	2.6
Grass Heath	1.1	0.07	7.6
Inland Bare Ground	0.5	0.7	34.9
Lowland Bog	0.0	10	0.0
Meadow /Verge /Semi-natural	25.1	0.2	501.6
Moorland Grass	51.8	0.02	103.7
Mown / Grazed Turf	3.4	0.2	67.5
Open Shrub Moor	4.3	0.02	8.7
Rough / Marsh Grass	0.2	0.02	0.5
Suburban /Rural Development	0.8	0.83	65.8
Tilled Land	1.7	0.66	109.5
Unclassified	2.5	0.48	119.1
Upland Bog	0.2	0	0
Total	**146.9**	**-**	**1163**
Livestock	-	-	2575
OVERALL TOTAL	**-**	**-**	**3738**

3.1.6 Annual mean phosphorus concentration in Ullswater

The annual mean phosphorus concentration in Ullswater is related to the load and hydraulic data presented above. Mean concentrations can be calculated simply by 'diluting' the total annual load into the annual volume of water flowing into the lake. Using this approach, for a total TP load of 2466 kg y^{-1} (based on an SRP:TP ratio of 0.59) and an average annual discharge of 294 10^6 m^3 y^{-1}, the mean lake concentration of TP will be 8.4 mg m^{-3}. Using the total load of 3088 kg y^{-1} (based on an SRP:TP ratio of 0.45), the mean lake concentration of TP will be 10.5 mg m^{-3}. These two estimates fall either side of the 9.8 mg m^{-3} which is the mean of four measurements during the 2005 Lakes Tour (Table 2). The export coefficient approach for the catchment (Table 10) added to the loads from the WwTW would produce a mean TP concentration of 14.3 mg m^{-3} indicating that the export coefficient approach overestimate load of TP to Ullswater. Table 11 gives the TP loads that will produce annual mean concentrations of TP of 10, 15 and 20 mg m^{-3}.

Table 11. Annual TP loads to produce stated annual mean concentrations of TP in Ullswater.

Mean TP in lake (mg m^{-3})	Annual TP load (kg TP y^{-1})	
	Dilution calculation	OECD model
9.8 (Lakes Tour 2005)	2881	4400
10	2940	4510
15	4410	7400
20	5880	10510

An alternative way to estimate the concentration of TP to a lake resulting from a given load is to use the OECD model derived largely from the work of Vollenweider and co-workers. This takes sedimentation losses into account based on the average losses seen in a range of lakes. The values used are, therefore, not specific to a lake. The general equation used was derived by Vollenweider (1976):

$$P = \frac{L_p / q_s}{(1 + \sqrt{t_w})} \qquad \text{Equation 1}$$

Where:

P = Concentration of TP in the lake (mg m^{-3})

L_p = Annual TP loading (mg m^{-2} y^{-1})

q_s = Water discharge height (m y^{-1})

t_w = Flushing rate (y^{-1})

Equation 1 relates the average phosphorus concentration in a lake to the phosphorus loading corrected for the flushing rate of a lake. Vollenweder & Kerekes (OECD 1982) checked the applicability of the original equation by using a large dataset of north temperate lakes. They used regression analysis to produce a modified equation that fitted the data best: their equation is shown as equation 2, which is used in the report.

$$P = 1.55 \left[\frac{L_p / q_s}{(1+\sqrt{t_w})} \right]^{0.82} \qquad \text{Equation 2}$$

Using this approach the two alternative loads of 2466 and 3088 kg y^{-1} would produce in-lake concentrations of 6.1 and 7.3 mg m^{-3} respectively. A load of 4400 kg y^{-1} would be needed to account for the measured Lakes Tour value of 9.8 mg m^{-3} in 2005: this is quite close to the load estimated from the WwTW and the export coefficient approach of 4204 kg y^{-1}.

It should be noted that all the calculations above that relate load to concentration do not take the possibility of an internal load of phosphorus into account and this may, in part, account for the lower concentrations in the lake calculated from external load alone. Processes that generate an internal load of phosphorus to the lake include physical mixing of phosphate from the interstitial water of sediments into the lake, chemically-mediated release from anoxic sediments at depth and transport of phosphorus from littoral regions where it has been made available by biological processes.

3.2 Derwent Water

3.2.1 Hydrology

Table 12. gives the flow data available to the project for Derwent Water. Of the four streams with flow data available, the River Derwent flowing in at the south end of the lake contributes 82% of the discharge. Watendlath Beck contributes about 14% of the discharge but the two other streams contribute a small percent of the total discharge (Table 12).

Not all the inflows to Derwent Water are monitored: for example inflows such as Barrow Beck on the eastern side, Eller Beck to the south and the numerous small unnamed streams draining Catbells to the west. Excluding these will underestimate the hydrological and nutrient load to the lake. The magnitude of this discrepancy was quantified by calculating the difference between the sum of the inflows and the measured outflow. Unfortunately there appears not to be a good estimate of the discharge from Derwent Water. The discharge was, therefore, estimated from the daily flow data at Portinscale which includes the input from the River Greta and which averaged 14.36 m^3 s^{-1}. The contribution of the R. Greta was estimated from Low Flows 2000 monthly mean flow data which were converted to an annual mean and equalled 5.31 m^3 s^{-1}. Although this is an estimate some confidence can be placed in it because the Low Flows 2000 estimate of the River Derwent inflow to Derwent Water was very similar to that actually measured. The estimated output for Derwent Water was therefore calculated from the difference of these two flows: 9.05 m^3 s^{-1}. Of the 9.05 m^3 s^{-1}, 5.6 m^3 s^{-1} was accounted for by the measured streams and the difference, 3.45 m^3 s^{-1}, is the estimated discharge from the streams which were not monitored (Table 13). Inspection of the map of the Derwent Water catchment suggests that this is likely to be an overestimate of the contribution of the unaccounted streams, so a second estimate of the total inflow to the lake was made from the contribution of the River Derwent catchment to the total lake catchment which is 63%. On the assumption that flow is proportional to catchment area, the total discharge from the lake will be 4.58/0.63 $m^3 s^{-1}$ which equals 7.27 $m^3 s^{-1}$ (229 $10^6 m^3 y^{-1}$). This calculation suggests that the unaccounted discharge is 1.67 $m^3 s^{-1}$. This is the estimate for the unaccounted discharge used here, but the above highlights the uncertainty in the load estimates.

Table 12. River GQA data used in the project for Derwent Water. Monthly flow estimates were produced by Low Flows 2000 software.

Site number	Site	Grid Ref	Flow data
INFLOWS			
88005545	Derwent at Grange in Borrowdale	NY 2543 1749	Daily flow data
-	R. Derwent inflow	NY 2600 1880	Monthly flow estimates
88005547	Watendlath Beck	NY 2657 1910	Monthly flow estimates
88005549	Ashness Gill	NY 2678 2004	Monthly flow estimates
88005550	Brockle Beck	NY 2705 2226	Monthly flow estimates
OUTFLOWS			
751007	R. Derwent at Portinscale	NY 2502 2388	Daily flow data
-	R. Greta	NY 2570 2360	Monthly flow estimates

3.2.2 Nutrient loads

Average concentrations of SRP in the main inflow are relatively low at 2.7 mg m^{-3} (Table 13). There was a slight seasonality to the concentration with minima during winter and higher concentrations in summer (Fig. 8): a pattern that was party linked to dilution by high discharge but also probably reflected higher input of phosphate during summer because of higher numbers of tourists. Three WwTW discharge into the River Derwent upstream of the GQA sampling point (WwTW at Seatoller, Rosthwaite and Stonethwaite; Table 14) and therefore probably contribute to this phosphorus input. Concentrations of SRP in Watendlath Beck and Ashness Gill are generally low throughout the year, although Watendlath Beck also shows a tendency for higher concentrations of SRP in summer, again probably because of tourist numbers and possibly discharge from the Keswick Lodore Hotel, although this is noted as discharge to groundwater (Table 14) and may not necessarily enter Watendlath Beck. The small Brockle Beck has high average concentrations of SRP at 10.4 mg m^{-3} with a peak recorded of 57.3 mg m^{-3}. Again high concentrations occur in summer possibly as a result of tourist influence. There is likely to be a point-source input of phosphorus somewhere on this stream.

Table 13. Hydraulic discharge (Q), mean concentration and annual load for four streams flowing into Derwent Water. Based on daily interpolated data between 1 Jan 1998 and 31 December 1999 (see text). Values in parenthesis give value as percent of the total measured. The TOTAL (whole catchment) is an estimate that includes the additional load of SRP from parts of the catchment that are not monitored (see text for more information).

		Mean concentration (mg m^{-3})			Load (kg y^{-1})		
Stream	Q (m^3 s^{-1})	SRP	NO_3-N	SiO_2	SRP	NO_3-N	SiO_2
R. Derwent	4.58 (82)	2.7	325	2253	390 (87)	46 941 (87)	325 412 (82)
Watendlath Beck	0.80 (14)	0.8	137	1698	20 (4)	3 456 (6)	42 839 (12)
Ashness Gill	0.12 (2)	0.6	171	2963	2.3 (1)	647 (1)	11 213 (3)
Brockle Beck	0.11 (2)	10.4	854	4320	36 (8)	2 963 (6)	14 986 (4)
Mean (flow weighted)	-	2.5	305	2229	-	-	-
Total measured	5.60	-	-	-	448	54 007	394 450
Unaccounted	1.67	-	-	-	132	16 063	117 391
TOTAL (whole catchment)	**7.27**	**-**	**-**	**-**	**580**	**70 070**	**511 841**

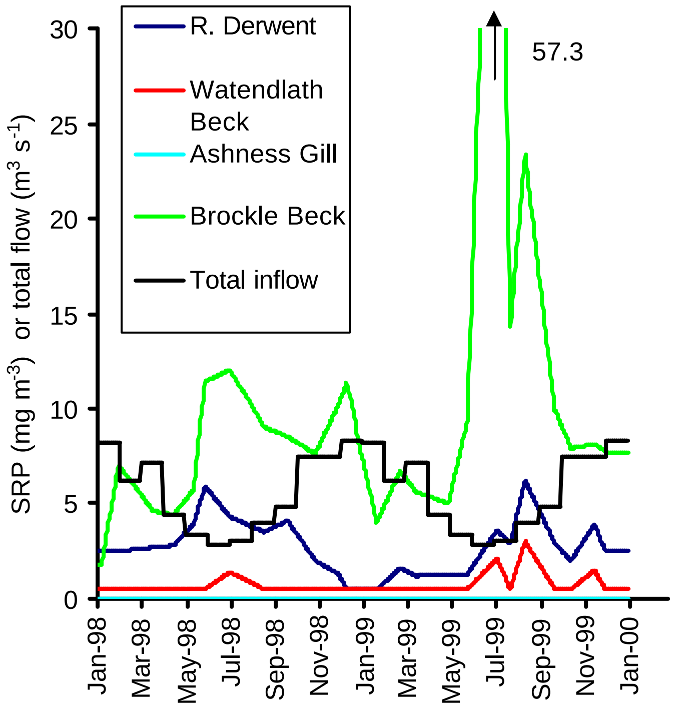


Figure 8. Seasonal concentration of SRP in the four monitored inflow streams plus total inflow from the streams.

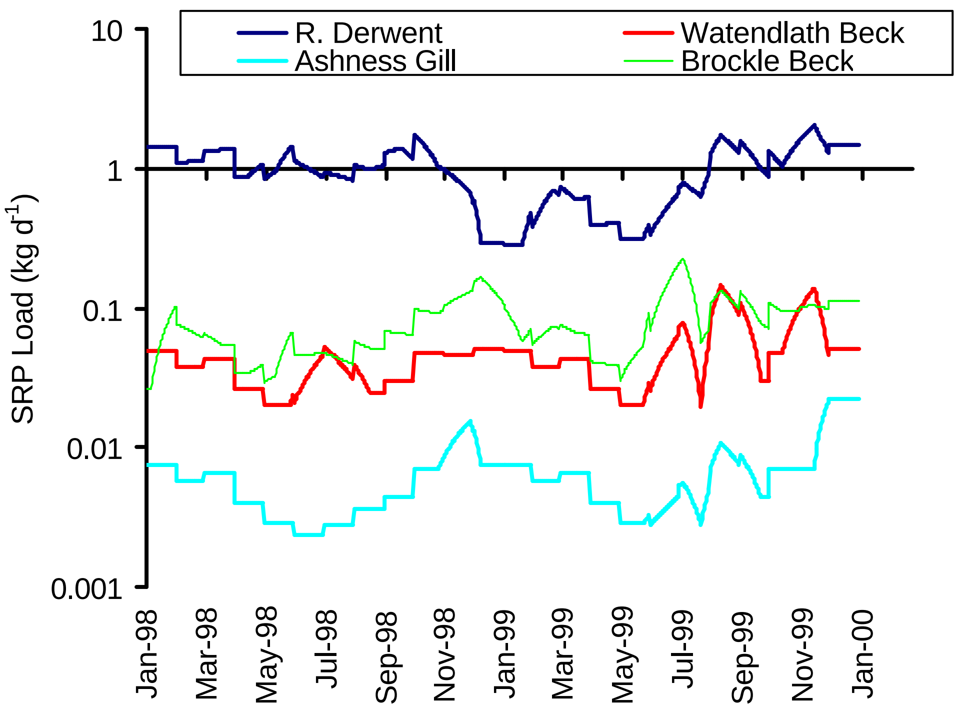


Figure 9. Seasonal patterns of SRP load to Derwent Water from the four monitored inflow streams. Note the logarithmic scale on the y-axis.

Calculations of load as the product of concentration and discharge, both interpolated to provide daily values, show that the River Derwent contributes 88% of the total measured load. Brockle Beck contributes 7% of the SRP load despite only contributing 2% of the discharge. The River Derwent is also the major contributor of the load of nitrate and silica by virtue of its high discharge, but the patterns for these nutrients are not analysed in greater detail here because they are of less importance to the ecology of Derwent Water.

Table 14. Discharge data used in the project for Derwent Water. Discharges upstream (US) of GQA sites and therefore already included in the measured data are noted. Maximum consented volumes as dry weather flow (DWF), average daily flow (ADF) and flow to full treatment (FTFT) are given, values for ADF in parenthesis are estimated from FTFT (see text).

					Max Consented volume ($m^3\ d^{-1}$)		
Site No.	Site	Grid Ref	Receiving water	Notes	DWF	ADF	FTFT
88005421	Seatoller WwTW	NY 2492 1385	River Derwent	US of GQA	26	32.5	
88005422	Stonethwaite WwTW	NY 2594 1416	River Derwent	US of GQA		(19.3)	30
88005419	Rosthwaite WwTW	NY 2548 1505	River Derwent	US of GQA	65	81.25	437
88005389	Grange in Borrowdale WwTW	NY 2568 1801	River Derwent		187	233.75	
88005450	Keswick Lodore Hotel	NY 2650 1910	Soakaway to groundwater to Watendlath Beck	US of GQA		(51.5)	80
88005452	Mary Mount Hotel	NY 2657 1918	Watendlath Beck			(5.8)	9
88021760	Greenbank Hotel	NY 2609 1798	Comb Gill			(3.2)	5
N/A	Leathes Head Hotel	NY 2587 1783	Soakaway to groundwater			(3.2)	5
88020466	Ashness Farm	NY 2714 1928	Soakaway to groundwater with hlo to trib			(3.9)	6
88020016	Hawse End	NY 2505 2132	Soakaway to groundwater			(6.4)	10
N/A	Swinside HoteL	NY 2466 2153	Soakaway to groundwater			(3.9)	6
N/A	Springs Farm	NY 2744 2263	Soakaway to groundwater			(2.6)	4

The inflow streams for which data are available do not represent the total load to the catchment as some streams have no monitoring data available. This missing load from the catchment was estimated as the product of the unaccounted hydraulic discharge (1.67 $m^3\ s^{-1}$) and the flow weighted concentration for all the inflowing streams to Derwent Water, (2.5 mg m^{-3}; Table 13). Using this calculation the additional unaccounted load was estimated to be

132 kg y^{-1} yielding a total load of 580 kg SRP y^{-1}. Similar additions for nitrate and silica produce total loads of 70 070 and 511 841 kg y^{-1} respectively (Table 13).

A final component of the nutrient loading to Derwent Water that has not yet been taken into account is the input from WwTW that have not already been included in the inflowing streams, i.e. those that are downstream of the GQA sampling points, but discharge into surface waters (nutrients from discharges to groundwater soakaways were assumed not to enter the lake). These include the major WwTW at Grange in Borrowdale and the two smaller ones at Mary Mount Hotel and Greenbank Hotel (Table 15). For the two latter WwTW, average daily flow was estimated from flow to full treatment using the 95 percentile value and calculating the mean and standard deviation to meet this assuming a log-normal distribution and a coefficient of variation (standard deviation/mean) of 0.3. To increase the accuracy of the estimates, concentrations from the full period of data available (1995 to 2005) were used since for Mary Mount Hotel, for example, there were only five data points in 1998. No silica data were available for any of the WwTW, but this is unlikely to influence the silica budget substantially.

Table 15. Average daily discharge, mean concentrations of SRP and nitrogen (ammonium plus oxidised nitrogen) for WwTW not already included in GQA monitoring sites on Derwent Water.

Site	Discharge (m^3 d^{-1})	Mean SRP (g m^{-3})	SRP load (kg y^{-1})	Mean N (mg m^{-3})	N load (kg y^{-1})
Grange in Borrowdale	233.75	2.78	237.2	10.2	866.0
Mary Mount Hotel	5.8	9.42	19.9	30.8	65.2
Greenbank Hotel	3.2	9.42	11.0	30.8	34.0
TOTAL	**-**	**-**	**268.1**	**-**	**967**

The total estimated loads of SRP, inorganic nitrogen and silica are given in Table 16.

Table 16. Summary of loads of SRP, inorganic nitrogen and silica to Derwent Water (kg y^{-1}).

Source	SRP	N	SiO_2
Catchment plus WwTW	580	70 070	513 626
Other WwTW	268	967	-
TOTAL	**848**	**71 037**	**511 841**

3.2.3 PROTECH input

For PROTECH, daily total discharge between 1/1/1998 and 31/12/1999 inclusive were calculated based on the known inflows scaled-up so the total average discharge equalled the average value of 7.27 $m^3 s^{-1}$ estimated in Table 13. The concentrations of SRP, nitrate-nitrogen and silica were individually up-scaled so that the annual load agreed with Table 16.

3.2.4 Catchment loads vs WwTW loads

In the calculations above, three WwTW are upstream of GQA sampling points and the loads from the streams are therefore a combination of point and diffuse sources (Table 14). An estimate is needed of the contribution of the diffuse load from the catchment compared to the point source load from WwTW in Objective 3. This was estimated by calculating the contribution of the three WwTW to the stream load.

Table 17. Partitioning SRP loads from point and diffuse sources.

Discharge	Discharge Beck	Average daily flow ($m^3 d^{-1}$)	Average SRP ($g m^{-3}$)	Annual load ($kg y^{-1}$)
Seatoller WwTW	River Derwent	32.5	1.93	22.9 (6%)[a]
Stonethwaite WwTW	River Derwent	19.3	0.45	3.2 (1%)[a]
Rosthwaite WwTW	River Derwent	81.3	4.11	121.9 (34%)[a]
Total	**-**	**-**	**-**	**148**
Load from other WwTw (Table 15)				268.1
Total load from WwTW				416.1 (49%)[b]
Total load from catchment				431.9 (51%)[b,c]
Total load to lake (Table 15)				848

a. Percent of load of named stream.
b. Percent of total load.
c. Calculated by difference.

Estimates in Table 17 suggest that 41% of the SRP load in the River Derwent inflow to the lake is contributed by the three WwTW upstream of the GQA sampling point. Overall, point sources are estimated to contribute 49% and diffuse sources 51% of the total SRP load. These are rough estimates in the light of the calculations needed above and the coarse sampling frequency. Furthermore, some of the catchment load is likely to result from small point discharges, such as from septic tanks, that are not monitored. The high concentrations of SRP in Brockle Beck, for example, suggests a point source or sources within the sub-catchment which are not being monitored.

3.2.5 Loads of SRP vs TP in Derwent Water

For the reasons laid out in section 3.1.5, loads have so far been calculated in terms of SRP. The factors used to convert SRP to TP in section 3.1.5 are also used here: for the catchment sources the ratio of SRP:TP was assumed to be either 0.59 (Hilton *et al.*, 1993) or 0.45 (May *et al.*, 1997) and for the WwTW the ratio of SRP:TP was assumed to be 0.91 based on values from Keswick WwTW (Maberly & Elliott, 2002).

Figure 10 gives the estimated loads of SRP and TP calculated as described above. The known point sources contribute 49% of the total SRP load of 848 kg y^{-1} and 38% of a total TP load of 1189 kg y^{-1} using a catchment SRP:TP ratio of 0.59 (Fig. 10a) and 32% of a total TP load of 1417 kg y^{-1} using a catchment SRP:TP ratio of 0.45 (Fig. 10b).

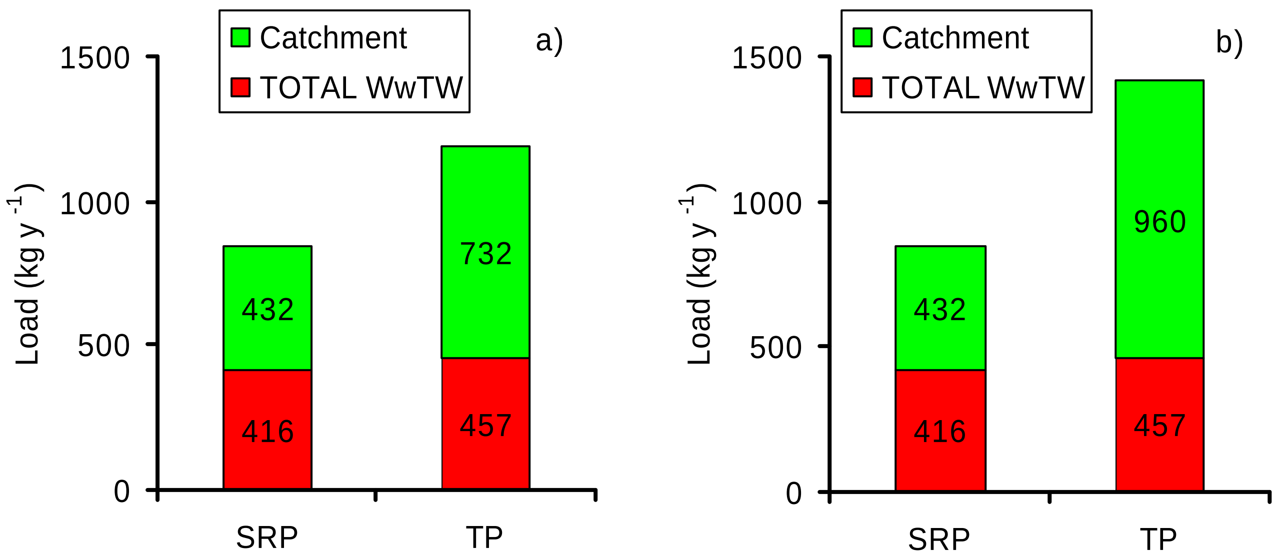


Figure 10. Estimated loads of SRP and TP to Derwent Water using an SRP:TP ratio of 0.91 for the WwTW and for the catchment: a) 0.59 and b) 0.45, (see text).

The load of TP from the catchment divided by the catchment area (Table 1) gives an average export coefficient of 0.09 or 0.11 kg P ha^{-1} y^{-1} for the two different ways of calculating TP load. This can be checked roughly using the export coefficient approach which ascribes an average TP loss per unit area for different types of land. An estimate of the total phosphorus (TP) load has been made from estimates of the landcover in the catchment and published TP export coefficients for different types of landcover for Derwent Water by May *et al.* (1997)

who give full details of the approach. Upland moorland, by virtue of its large area in the catchment (66%), is the major source of TP followed by improved pasture- and together these two landcover categories are estimated to contribute 68% of the total load (Table 18a). The total load of TP estimated in this way is 1161 kg TP y^{-1} (equivalent to an average export coefficient of 0.14 kg ha^{-1} y^{-1}), which is greater than the load estimated from stream chemistry and hydrology of 732 kg TP y^{-1} (Fig. 10). There are many possible reasons for this discrepancy, including attributing too great an input from the point sources, errors in estimates of hydrological discharge, inaccuracies resulting from load estimates made with low-frequency samples and non-sampled sources.

Table 18a. *Estimate of TP loads to Derwent Water from the catchment based on an export coefficient approach (based on May* et al., *1997).*

Landcover (excluding open water)	Area (ha)	TP export coefficient (kg $ha^{-1}y^{-1}$)	TP Load (kg y^{-1})
Upland moor	5,330	0.1	533
Improved pasture	689	0.38	261.82
Bogs & peat	86	1	86
Broadleaved forest	546	0.15	81.9
Inland bare rock	712	0.1	71.2
Urban/rural settlement (runoff only)	61	0.83	50.63
Mixed forest	250	0.15	37.5
Coniferous forest	118	0.15	17.7
Rough grazing	192	0.07	13.44
Other	49	0.1	4.9
Cleared/new forest	15	0.2	3
Total	**8048**		**1161.09**

It should be noted that all the calculations above that relate load to concentration do not take the possibility of an internal load of phosphorus into account and this may, in part, account for the lower concentrations in the lake calculated from external load alone. Processes that generate an internal load of phosphorus to the lake include physical mixing of phosphate from the interstitial water of sediments into the lake, chemically-mediated release from anoxic sediments at depth and transport of phosphorus from littoral regions where it has been made available by biological processes.

Table 18b. Estimate of TP loads from the catchment to Derwent Water based on an export coefficient approach using CEH Landcover 2000 data and export coefficients in Carvalho et al*. (2003). The load from livestock is also taken from Carvalho* et al*. (2003).*

Land cover	Area (ha)	TP export coefficient (kg ha^{-1} y^{-1})	TP load (kg P y^{-1})
Bracken	2320	0.02	46.4
Coniferous Woodland	40	0.15	6.5
Continuous Urban	40	0.83	32.6
Deciduous Woodland	490	0.02	9.8
Dense Shrub Moor	220	0.02	4.4
Felled Forest	20	0.2	3.2
Grass Heath	50	0.07	3.6
Inland Bare Ground	30	0.7	20.3
Inland Water	510		0.0
Lowland Bog	0.0	0	0.0
Meadow / Verge / Semi-natural	870	0.2	173.7
Moorland Grass	3100	0.02	62.0
Mown / Grazed Turf	50	0.2	9.1
Open Shrub Moor	570	0.02	11.4
Rough / Marsh Grass	20	0.02	0.4
Suburban / Rural Development	50	0.83	38.3
Tilled Land	110	0.66	75.5
Unclassified	40	0.48	20.1
Upland Bog	20	0	0.0
Total	**8540**	**4.44**	**517.3**
Livestock	**-**	**-**	**1238**
OVERALL TOTAL	**-**	**-**	**1755**

An alternative TP load is calculated in Table 18b using more recent land cover data from the CEH land cover database and the most recent export coefficient data (Carvalho *et al*., 2003). This gives a very different and lower estimate of the TP load to Derwent Water of 517 kg TP

y^{-1}. As noted above for Ullswater, in this approach extra loads from cattle and sheep need to be added into this load. When this is done using the estimates in Carvalho *et al*. (2003) the total diffuse input increases to 1755 kg y^{-1} (Table 18b) which is equivalent to an average export coefficient of 21 kg P ha^{-1} y^{-1} This estimated load is substantially larger than that in Table 18a and larger than that estimated from the stream load calculations of between 732 and 960 kg y^{-1} (Fig. 10). The estimated contribution of the human population of 370 kg y^{-1} in Carvalho *et al.* (2003) is similar to that estimated here (457 kg y^{-1}). As was noted for Ullswater, nutrient loads to lakes are very difficult to quantify accurately and the approach in Carvalho *et al*. (2003) is not considered to be suitable at a specific site: this approach is not, therefore, used further.

3.2.6 Annual mean phosphorus concentration in Derwent Water

The annual mean phosphorus concentration in Derwent Water is related to the load and hydraulic data presented above. The two approaches used for Ullswater are used here, namely a simple dilution approach and the OECD model of nutrient loading. First using the dilution approach, for a total TP load of 1189 kg y^{-1} or 1417 kg y^{-1} (depending on the SRP:TP ratio) and an average annual discharge of 229 10^6 m^3 y^{-1}, the mean lake concentration of TP will be 5.2 or 6.2 mg m^{-3}. Both these estimates are much smaller than the mean of fortnightly measurements between 1999 and 2004 of 8.5 mg m^{-3} (Table 3). Based on the export coefficient approach, plus the estimated load from the WwTW, the total load is 2212 kg y^{-1} which would yield an in-lake concentration of 9.7 mg m^{-3}.

The OECD approach yields similar estimates of mean lake concentration for these loads of 5.2 and 6.0 mg m^{-3} for the TP loads of 1189 kg y^{-1} or 1417 kg y^{-1} (depending on the SRP:TP ratio). Using this model, a mean in-lake concentration of TP of 8.5 mg m^{-3} would be produced by a load of 2170 kg y^{-1}. The export coefficient and WwTW total load of 2212 kg y^{-1} would produce an average in-lake concentration of 8.6 mg m^{-3} which is close to the mean from the sampling data.

Table 19 gives the TP loads that will produce annual mean concentrations of TP of 8, 8.5 (the observed mean) and 10 mg m^{-3}.

Table 19. Annual TP loads to produce stated annual mean concentrations of TP in Derwent Water.

	Annual TP load (kg TP y^{-1})	
Mean TP in lake (mg m^{-3})	Dilution calculation	OECD model
8	1832	2025
8.5 (fortnight data)	1947	2170
10	2290	2660

4. Objective 3: Changes in water quality resulting from changing nutrient load management options

Several approaches can be used to convert mean concentration of total phosphorus to concentration of mean phytoplankton chlorophyll *a*. One is that suggested in the draft implementation of the Water Framework Directive. This was produced by analysing the relationship between these variables for UK lakes. Different relationships were found for lakes of different types (Phillips, 2006). For shallow lakes (such as Derwent Water) the best fit equation is:

$$Log\ Chla = -0.512 + 1.105 * Log\ TP \qquad \text{Equation 3}$$

For deep lakes (such as Ullswater) the best fit equation is:

$$Log\ Chla = -0.220 + 0.731 * Log\ TP \qquad \text{Equation 4}$$

Where TP is the annual average concentration of TP and Chla is the annual average concentration of phytoplankton chlorophyll *a* in units of mg m^{-3} in both instances.

In addition, the reference TP concentration was determined based on the alkalinity and mean depth of a lake using the morpho-edaphic index approach, (Vighi & Chiaudani, 1984) calibrated for UK lakes which is currently the proposed way of defining reference conditions in UK lakes (Phillips, 2006). Values for Ullswater and Derwent Water were 5.3 and 6.4 mg m^{-3} respectively.

Figure 11 shows the predicted response of phytoplankton chlorophyll *a* in Ullswater and Derwent Water to different concentrations of total phosphorus. The observed concentrations of chlorophyll *a* and TP in 2005 are higher than predicted by these equations.

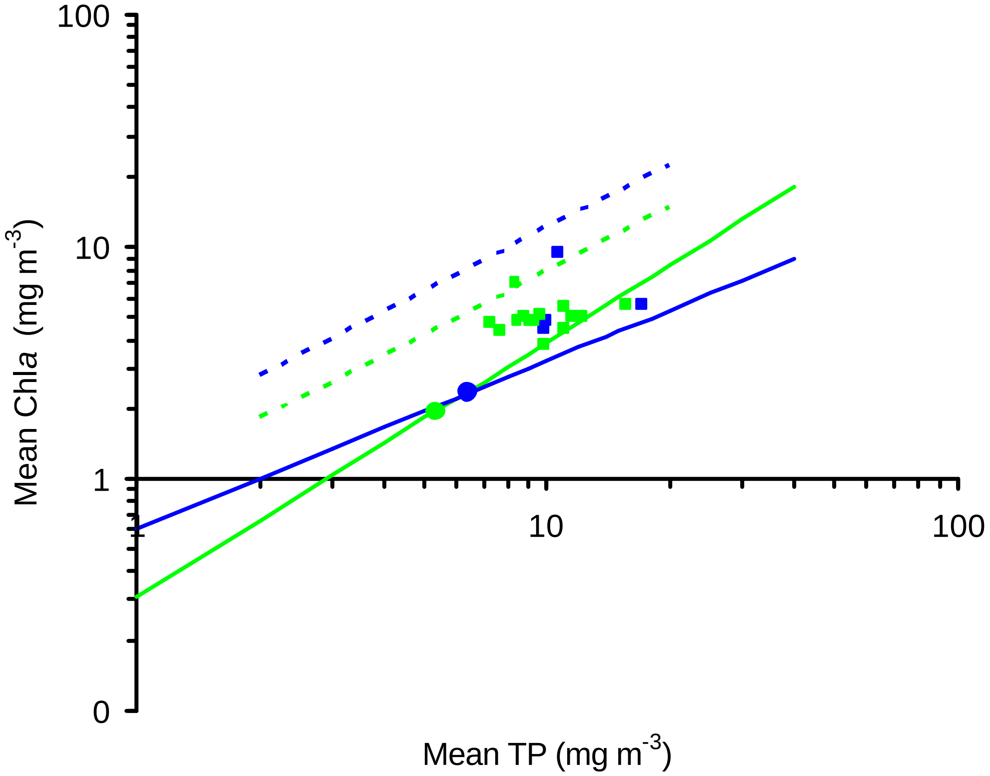


Figure 11. Response of annual mean phytoplankton chlorophyll a *to mean concentration of total phosphorus in Ullswater (blue) and Derwent Water (green). The solid lines show the WFD responses using equations 3 and 4. Also shown are predictions for the metabolic model (dashed lines). The solid circles show the reference values (see text). Squares show actual values comprising annual means based on Lake Tours in 1991, 1995, 2000 and 2005 for Ulswater and annual means based on fortnightly values between 1991 and 2004 for Derwent Water. Note that both axes are logarithmic.*

In addition to the in-lake TP concentrations identified in Table 11, the effect of three other scenarios on phytoplankton chlorophyll *a* were tested: the Glenridding WwTW had an average effluent phosphorus concentration of 1 g m^{-3}, the catchment had a 15% reduction in loading and finally both these reductions together. To simplify the number of values being presented, in all case the average of the loads based on the two SRP:TP ratios was used (ie the average of SRP:TP of 0.59 and 0.45). For the scenarios based on TP these were converted to mean concentration of chlorophyll *a* using the WFD method outlined above and the Metabolic Model approach (Reynolds & Maberly, 2002).

Table 20. Estimated effect of different scenarios for in-lake TP concentration or TP load on mean concentration of phytoplankton chlorophyll a in Ullswater. For simplicity the mean of the two SRP:TP conversion ratios for the catchment are used. Control is the current estimated load. The mean of the different estimates and percent reduction for the load scenarios are given in parenthesis. Values in parenthesis under TP load are the contributing WwTw load, catchment load.

		Mean lake TP ($mg\ m^{-3}$)		Mean phytoplankton chlorophyll *a* ($mg\ m^{-3}$)				
				WFD equation		Metabolic model		
Scenario	TP load ($kg\ y^{-1}$)	Dilution	OECD	Dilution TP	OECD TP	Dilution TP	OECD TP	**Mean**
Control	2777 (466, 2311)	9.5	6.7	3.7	2.5	11.5	8.4	**6.5**
TP = 10 $mg\ m^{-3}$	-	10		3.8		11.9		**8.1**
TP = 15 $mg\ m^{-3}$	-	15		3.9		12.2		**11.9**
TP = 20 $mg\ m^{-3}$	-	20		6.1		17.6		**15.6**
Glenridding WwTW TP = 1 $g\ m^{-3}$	2490 (179, 2311)	8.5	6.1	3.3	2.3	10.5	7.8	**6.0 (8%)**
Catchment 15% reduction	2221 (466, 1755)	7.6	5.6	2.9	2.1	9.4	7.2	**5.4 (17%)**
Both Glenridding & Catchment reduction	1934 (179,1755)	6.6	5.0	2.5	1.8	8.3	6.5	**4.8 (26%)**

As described above there is uncertainty in estimation of loads to a lake and there is another level of uncertainty in estimating the effect of load on concentration of TP in the lake and another in the conversion of this TP to phytoplankton chlorophyll *a*. To try to capture some of this uncertainty, in Table 20, mean phytoplankton chlorophyll *a* is estimated in four different ways for a given TP load and 2 different ways for a given in-lake TP concentration. The mean value perhaps gives the best estimate. It should be noted that the percent reduction in phytoplankton chlorophyll *a* estimated for the different calculation methods is slightly more consistent and this will therefore be focussed on below. Of the three nutrient load scenarios, reducing the annual mean concentration of phosphorus to an annual mean of 1 gm^{-3} will give an estimated 8% reduction in the concentration of phytoplankton chlorophyll *a* in the lake. Reducing the diffuse sources from the catchment will reduce the phytoplankton by 17% and the two measures in combination will result in about a 26% reduction in phytoplankton chlorophyll *a*.

A similar exercise was followed for Derwent Water. The in-lake TP concentration calculated simply by dilution and using the OECD model is quite similar. This suggests that Derwent Water is more similar to the mean lakes used to construct this equation than is Ullswater. Like Ullswater, however, there is quite a large difference in the estimated concentration of phytoplankton chlorophyll *a* estimated using the metabolic model and the WFD approach. Again, the percent reduction for the different scenarios is more similar than the actual values so this will be focussed on here. The reduction of WwTW concentrations of TP to 1 mg m^{-3} or below (Stonethwaite WwTW is already below this threshold, Table 17 so these values were not altered) is forecast to have a large effect on the lake, reducing the mean concentration of chlorophyll *a* by 24% (Table 21). Reducing the catchment load by 15% will only cause an average reduction in phytoplankton chlorophyll *a* by 9%, while implementing both measures will cause an estimated 34% reduction in phytoplankton.

Derwent Water currently just fails to meet good ecological status for phytoplankton chlorophyll *a* under the WFD (Maberly *et al.*, 2006). The chlorophyll *a* in Derwent Water 2005 based on the four Lakes Tour samples gave a mean of 6.9 mg m^{-3} and the fortnightly data were slightly lower at 5.9 mg m^{-3} (CEH data- Maberly *et al.*, *in prep*.). The high/moderate boundary for Derwent Water was estimated at 5.7 mg m^{-3} (Maberly *et al.*, 2006) requiring a minimum of a 3.7% reduction in the chlorophyll *a* concentration. Either of the two management scenarios in Table 21 will produce this. Reducing the input of phosphorus from the WwTW is an attractive management option since it will have a relatively large beneficial effect on water quality and should readily return the lake to good ecological status.

Table 21. Estimated effect of different scenarios for in-lake TP concentration or TP load on mean concentration of phytoplankton chlorophyll a in Derwent Water. For simplicity the mean of the two SRP:TP conversion ratios for the catchment are used. Control is the current estimated load. The mean of the different estimates and percent reduction for the load scenarios are given in parenthesis. Values in parenthesis under TP load are the contributing WwTw load, catchment load.

		Mean lake TP (mg m^{-3})		Mean phytoplankton chlorophyll *a* (mg m^{-3})				
				WFD equation		Metabolic model		
Scenario	TP load (kg y^{-1})	Dilution	OECD	Dilution TP	OECD TP	Dilution TP	OECD TP	**Mean**
Control	1303 (457, 846)	5.7	5.6	2.1	2.1	4.8	4.7	**3.4**
TP = 8 mg m^{-3}								
TP = 10 mg m^{-3}								
All WwTW TP <=1 g m^{-3}	950 (104, 846)	4.1	4.3	1.5	1.5	3.6	3.7	**2.6 (24%)**
Catchment 15% reduction	1176 (457, 719)	5.1	5.1	1.9	1.9	4.3	4.3	**3.1 (9%)**
Both WwTw& Catchment reduction	823 (104, 719)	3.6	3.8	1.3	1.3	3.1	3.3	**2.3 (34%)**

5. Objective 4: PROTECH modelling of the management options in objective 3.

5.1 Modelling approach and input data

PROTECH is a process based model that operates on a daily time step and simulates the physical structure within a lake (e.g. temperature profiles) and the growth of functional algal types in response to changing environmental conditions (Reynolds *et. al.*, 2001b).

In this project, interpolated daily hydrological input and mean inflowing concentrations of SRP, nitrate and silica were used to drive PROTECH as described in sections 3.1.3 and 3.2.3 for Ullswater and Derwent Water respectively.

Meteorological data from 1998-1999 were used from a meteorological station near Keswick and consisted of daily cloud cover, wind speed, air temperature and air humidity. At Ullswater, fortnightly observed chlorophyll measurements taken by CEH for the same period were used for validation and the simulations were primed with eight species simulating the following phytoplankton: the green alga *Chlorella*, the cryptophyte *Rhodomonas*, the cyanobacteria (blue-green algae) *Anabaena*, *Microcystis*, and *Oscillatoria* the dinoflagellate *Ceratium* and the diatoms *Asterionella* and *Fragilaria*. At Derwent Water the species complement comprised the green algae *Chlorella* and *Paulschulzia*, the cryptophyte *Cryptomonas*, the cyanobacterium *Anabaena* and the diatoms *Asterionella*, *Aulacoseira*, *Tabellaria* and *Urosolenia*. At both lakes a series of different nutrient loadings were run through PROTECH in order to produce a 'calibration curve' that related the sensitivity of the lake to SRP loading. Once produced, this can be used to determine the response of the lake to nutrient loading for any scenario within the range of loadings simulated.

5.2 PROTECH on Ullswater

5.2.1 Model validation and output

The validation required the windspeed at Keswick to be reduced by 20% and very small amounts of SRP (0.03-0.9 mg m^{-3}) to be added to the hypolimnion PROTECH layers during July to August of both years. This would be consistent with a small internal load of phosphorus in this lake. With these minor changes to the driving data PROTECH accurately simulated the magnitude of the peaks in the summer of both years as well as the overwintering concentrations (Fig. 12). The timing of growth and decline were also well simulated. The largest discrepancies were that PROTECH predicted a slightly larger phytoplankton chlorophyll *a* in spring 1998 and briefly in autumn 1999 compared to what was observed, but overall the agreement was excellent. In terms of composition, the dominant species was the pinnate diatom *Asterionella formosa* in spring with contributions from the cryptophyte *Rhodomonas* and in summer the dinoflagellate *Ceratium*. The three genera of cyanobacteria *Anabaena*, *Microcystis* and *Oscillatoria* were a minor contribution to chlorophyll *a* (Fig. 13). A detailed comparison between these predictions and counts at the time have not been made, but they species contribution appears to be broadly reasonable.

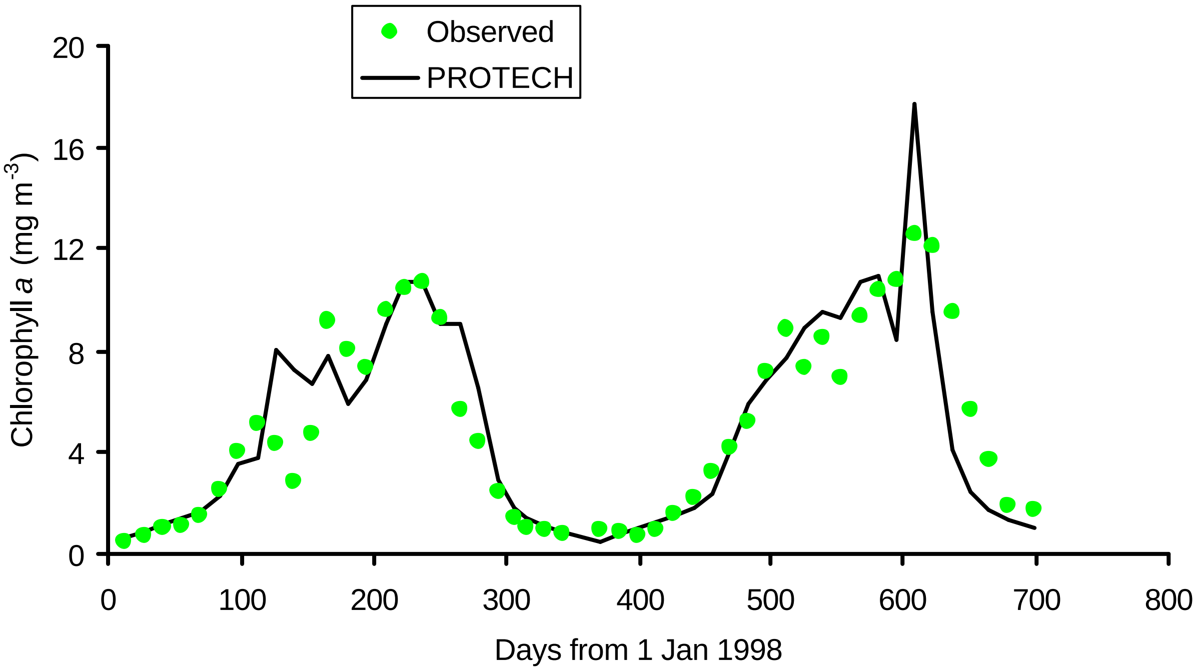


Figure 12. Observed and modelled total phytoplankton chlorophyll a *in Ullswater in 1998 and 1999.*

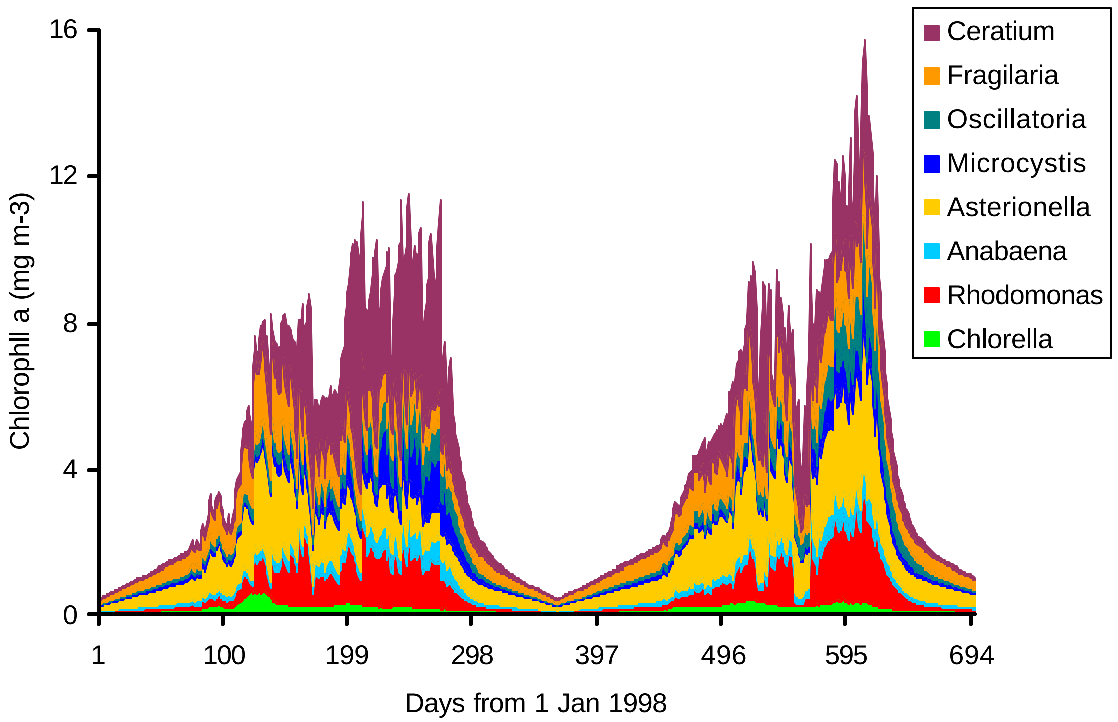


Figure 13. Seasonal changes in species complement predicted by PROTECH in Ullswater for the validation run in 1998 and 1999.

A range of different SRP loads were run in PROTECH to produce the response line in Figure 14. This shows that in Ullswater, the annual mean concentration of chlorophyll *a* increases by 0.21 mg m^{-3} for an increase of SRP load of 1000 kg y^{-1}.

Not all species respond equally to increased load of SRP. The main species that responded to increased load was the diatom *Asterionella formosa* (Fig. 15). The other representative diatom, *Fragilaria* and the cryptophyte *Rhodomonas* were also relatively responsive, while there was no change in annual chlorophyll *a* of the dinoflagellate and, interestingly, little response by the two cyanobacteria *Anabaena* and *Microcystis*. A third cyanobacterium, *Oscillatoria*, showed a slight increase with phosphorus load.

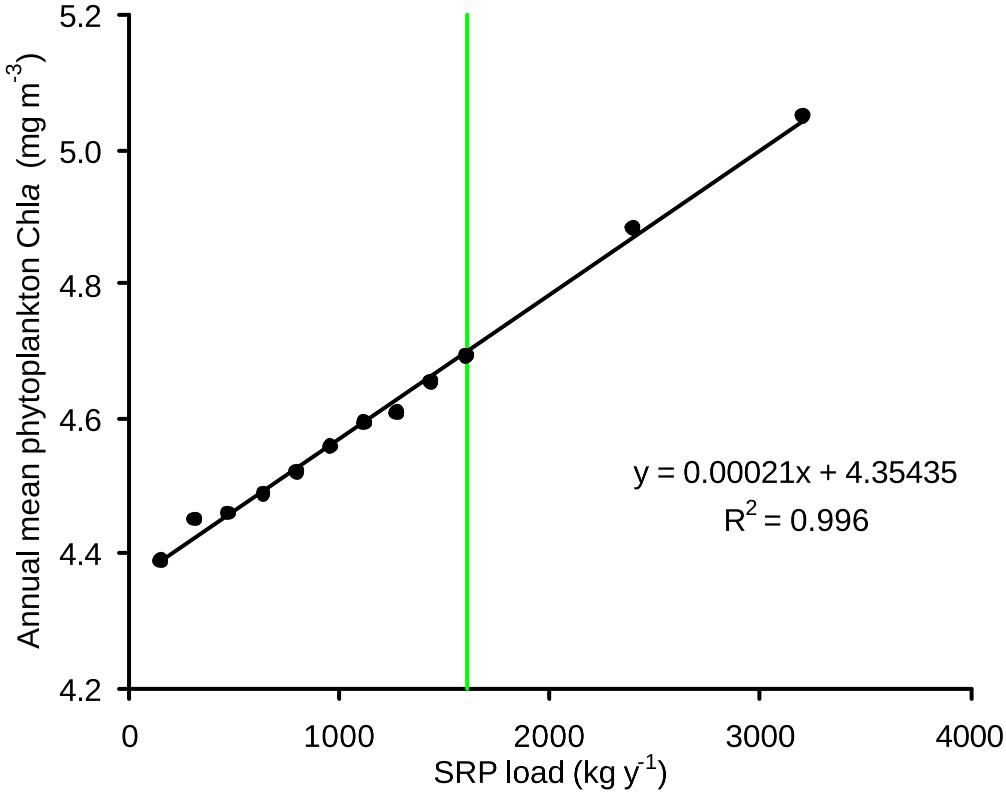


Figure 14. Response of total annual mean phytoplankton chlorophyll a in Ullswater to different external loads of SRP. The green vertical line shows the current estimated external load of SRP.

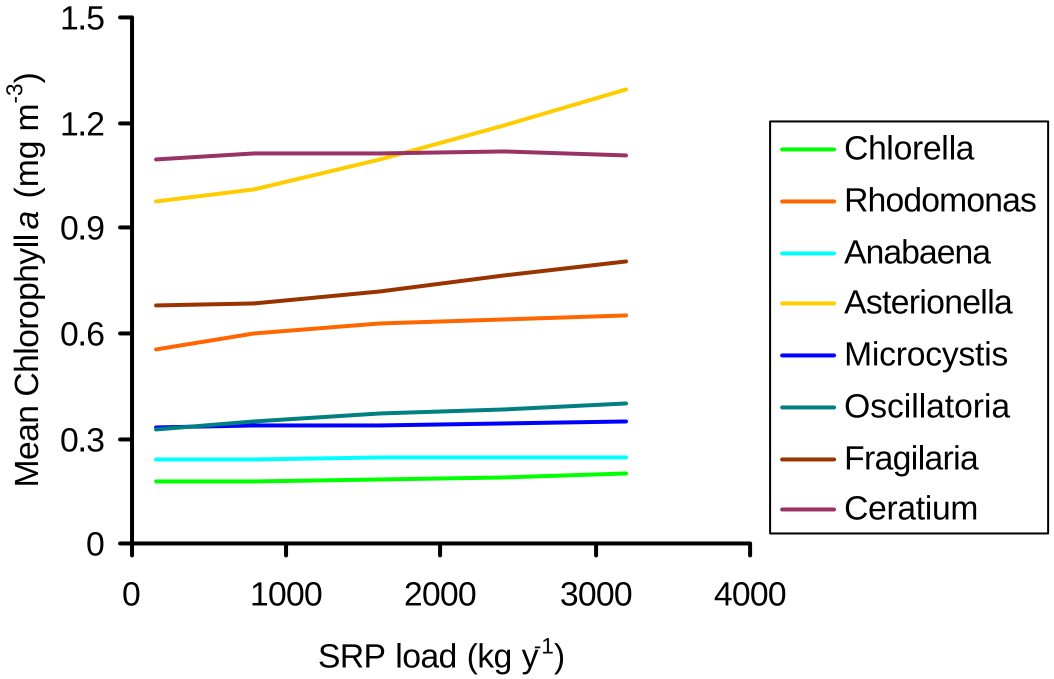


Figure 15. Response of the different species of phytoplankton to external load of SRP in Ullswater.

5.2.2 Relating management options to chlorophyll a in Ullswater

The response of phytoplankton chlorophyll a to SRP load was related to the PROTECH output in the following way. The loads of TP from the WwTW and the catchment in Table 20 were converted separately to SRP using SRP:TP ratios of 0.91 and 0.52 (average of 0.54 and 0.45) respectively. The PROTECH model suggests very modest changes in annual mean phytoplankton biomass in response to changing SRP load (Table 22). This is surprising and much less than that predicted using the metabolic model and the calculation based on the average responses of a range of lakes. This will be discussed more fully in the Conclusion section.

Table 22. Forecast effect of different SRP loading scenarios for Ullswater based on the response of the PROTECH model in Figure 14.

Scenario	SRP load (kg y^{-1})	Mean phytoplankton chl *a* (mg m^{-3})
Control	1626	4.7
Glenridding WwTW TP = 1 g m^{-3}	1365	4.6
Catchment 15% reduction	1337	4.6
Both Glenridding & catchment reduction	1076	4.6

5.3 PROTECH on Derwent Water

5.3.1 Model validation and output

The validation run on Derwent Water did not require any calibration, i.e. there were no changes needed to the meteorological or nutrient input files in order to produce a close simulation of the observed data (Fig. 16).

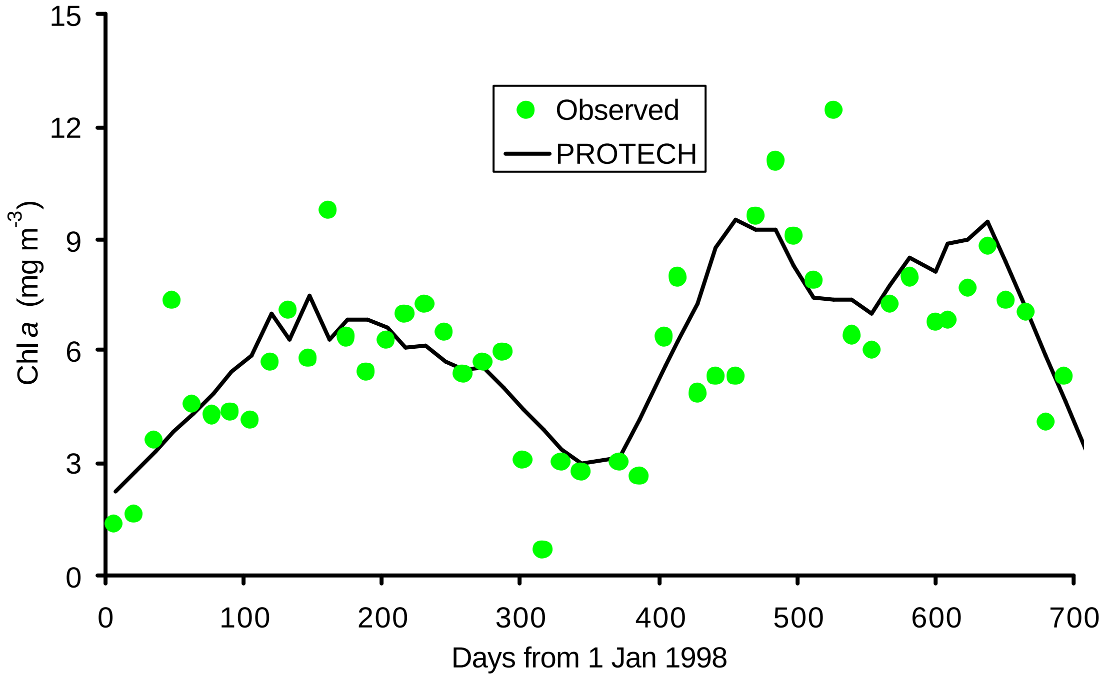


Figure 16. Observed and modelled total phytoplankton chlorophyll a *in Derwent Water in 1998 and 1999.*

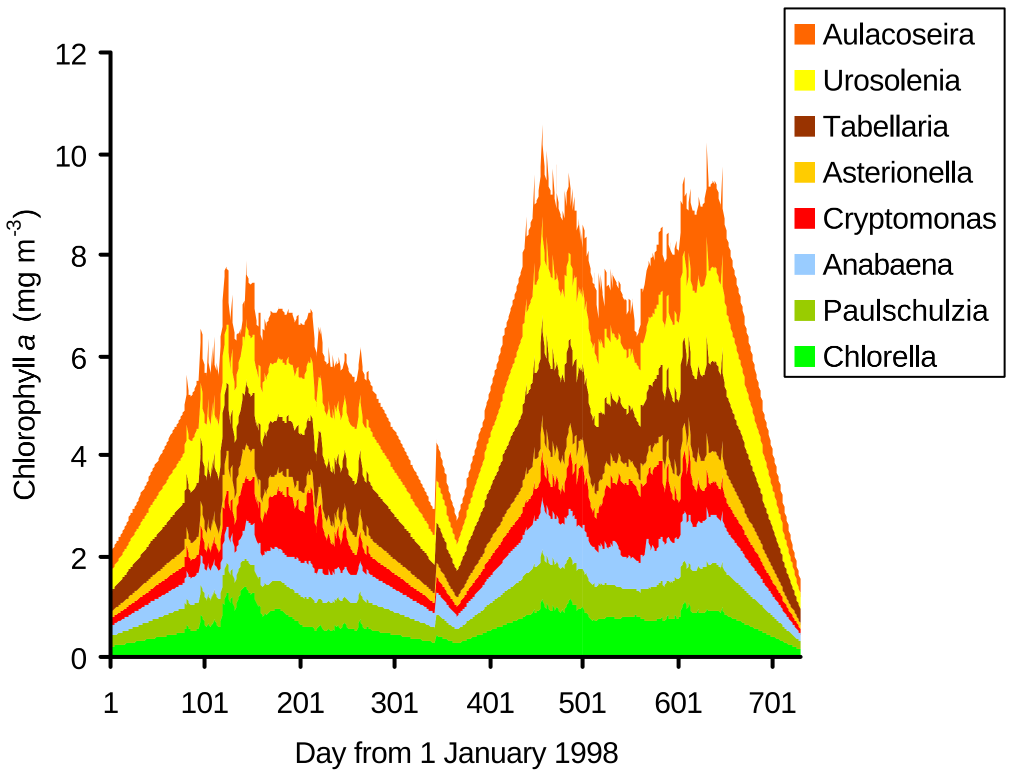


Figure 17. Seasonal changes in species complement predicted by PROTECH in Derwent Water for the validation run in 1998 and 1999.

In terms of species composition, there was a fairly even contribution of the different functional types in the model (Fig. 17). In Derwent Water fortnightly data on composition are available, so the PROTECH simulations were compared with the observations at the level of phylogentic group. Broadly, there was a good agreement (Table 23) with major periods when the lake moved away from diatom dominance usually being picked up by PROTECH.

Table 23. Comparison between the observed and simulated dominant taxa in Derwent Water, estimated from the CEH algal count data and PROTECH run, respectively. D = Diatom, G = Green, BG = Blue Green, DINO = Dinoflagellate.

Date	Observed	PROTECH	Date	Observed	PROTECH
07 January 1998	D	D	07 January 1999	D	D
21 January 1998	D	D	21 January 1999	D	D
04 February 1998	G	D	09 February 1999	D	D
18 February 1998	D	D	18 February 1999	D	D
04 March 1998	D	D	04 March 1999	D	D
18 March 1998	D	D	17 March 1999	D	D
01 April 1998	G	D	01 April 1999	D	D
15 April 1998	D	D	15 April 1999	D	D
29 April 1998	G	D	29 April 1999	D	D
13 May 1998	G	G	13 May 1999	BG	D
27 May 1998	G	G	27 May 1999	BG	D
10 June 1998	G	D	10 June 1999	D	D
24 June 1998	D	D	24 June 1999	D	DINO
08 July 1998	D	DINO	08 July 1999	D	DINO
22 July 1998	D	D	22 July 1999	D	D
05 August 1998	D	D	05 August 1999	D	DINO
19 August 1998	D	D	23 August 1999	DINO	D
02 September 1998	D	D	02 September 1999	D	D
16 September 1998	D	D	16 September 1999	D	D
30 September 1998	D	D	30 September 1999	D	D
14 October 1998	D	D	14 October 1999	D	D
29 October 1998	D	D	28 October 1999	D	D
12 November 1998	G	D	11 November 1999	D	D
25 November 1998	D	D	25 November 1999	D	D
10 December 1998	D	D	21 December 1999	D	D

A range of different SRP loads were run in PROTECH to produce the response line in Figure 18. This shows that in Derwent Water the annual mean concentration of chlorophyll *a* increases by 0.47 mg m^{-3} for an increase in SRP load of 1000 kg y^{-1}.

Not all species respond equally to increased SRP load. The two most responsive species were the two small forms the chlorophyte *Chlorella* and the cryptophyte *Cryptomonas*. The other species were relatively unresponsive.

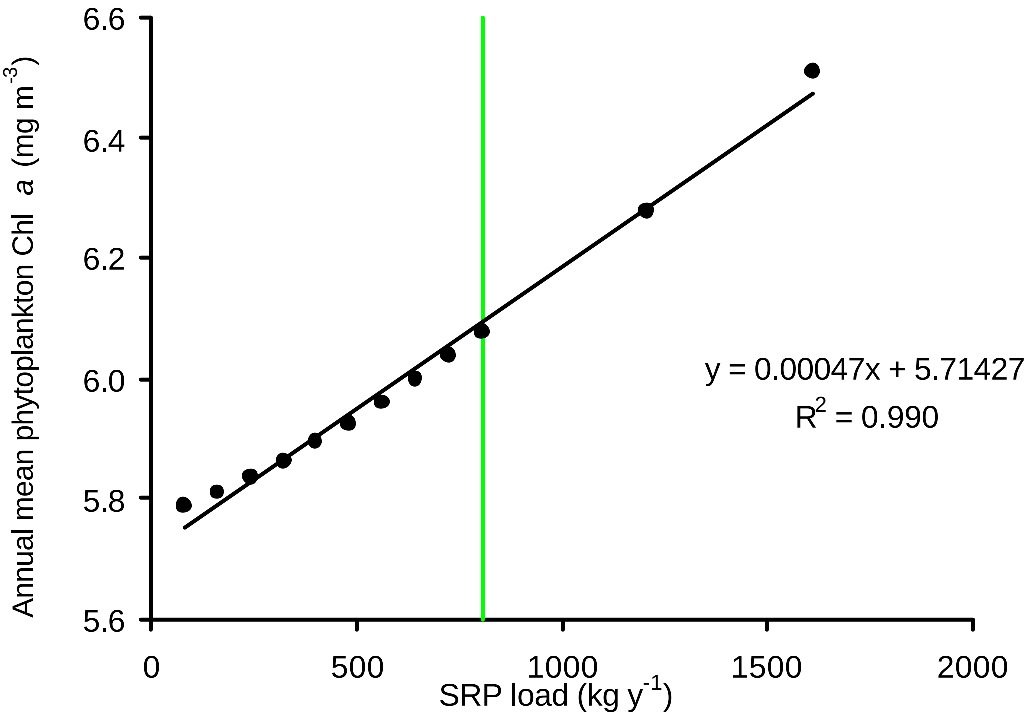


Figure 18. Response of total annual mean phytoplankton chlorophyll a in Derwent Water to different external loads of SRP. The green vertical line shows the current estimated external load of SRP.

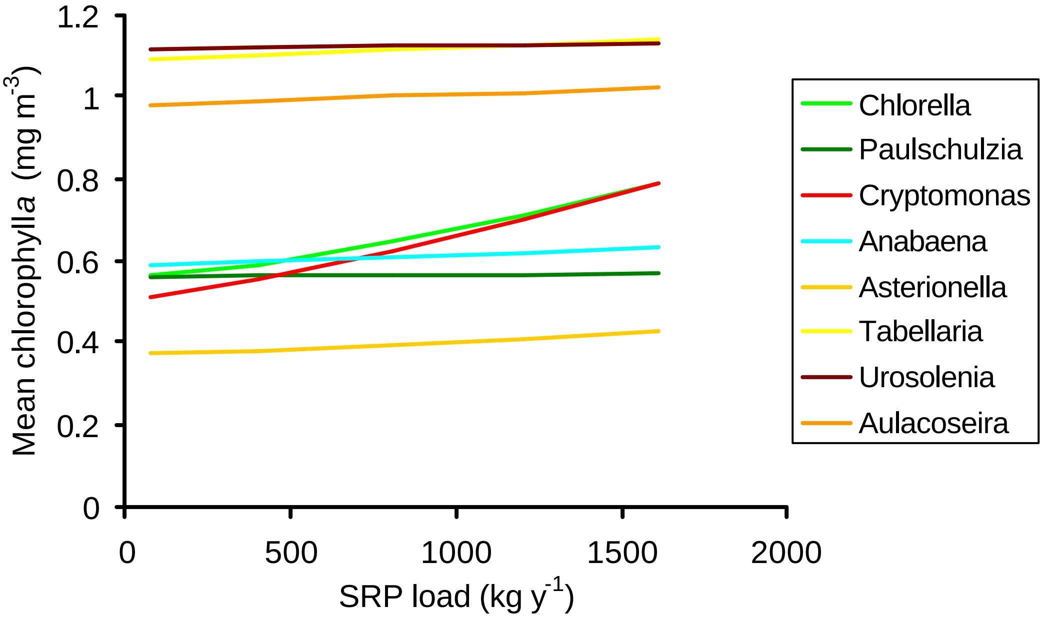


Figure 19. Response of the different species of phytoplankton to external load of SRP in Derwent Water.

5.3.2 Relating management options to chlorophyll a in Derwent Water

The approach taken to produce SRP loads used on Ullswater in 5.2.2 was used here. Table 24 shows that, like Ullswater there was a relatively modest effect of the different SRP load scenarios on the predicted mean phytoplankton chlorophyll *a* in Derwent Water. This is surprising and much less than that predicted using the metabolic model and the calculation based on the average responses of a range of lakes. This will be discussed more fully in the Conclusion section.

Table 24. Forecast effect of different SRP loading scenarios for Derwent Water based on the response of the PROTECH model in Figure 18.

Scenario	SRP load (kg y^{-1})	Mean phytoplankton chl *a* (mg m^{-3})
Control	856	6.1
All WwTW TP = 1 g m^{-3}	535	6.0
Catchment 15% reduction	790	6.1
Both WwTW & catchment reduction	469	5.9

5.4 Initial assessment of climate change on Ullswater and Derwent Water

5.4.1 Introduction and approach

Climate change is already happening and further change is inevitable even if emissions of greenhouse gases are halted this year. The consequences for lakes are likely to be large and result from changes at different levels from relatively indirect effects caused by changes of landuse in the catchment to direct effects caused by change in weather factors such as air temperature, wind speed and rainfall. The consequences for lake management and in particular the implementation of national and European directives, such as the Water Framework Directive, are only just beginning to be addressed.

It was not possible to run an extensive set of climate change scenarios within the scope of this project. Furthermore, some of the more influential weather factors that may affect lakes, such as rainfall, are relatively poorly forecast at the moment. Instead, a simple approach was taken for the most certain change in weather pattern: an increase in water temperature resulting from an increase in air temperature. The two year validation simulations for each lake were repeated but the temperature in each PROTECH layer in the model was forced to be either 1 or 2 ^{o}C cooler or 1 to 5 ^{o}C warmer in 1 ^{o}C steps. This left the thermocline structure unchanged and allowed the effect of only a change in temperature to be simulated. To drive the model with a range of different nutrient loads, the soluble reactive phosphorus inflow concentrations used for the validation run were reduced by 50% and 10% and increased by 10%, 50% and 100%. This gave a combination of 48 different simulations for each lake, including the initial validation run. A identical approach has been used in work on Bassenthwaite Lake (Elliott *et al*., 2006).

5.4.2 Ullswater response to temperature

The responses of total chlorophyll and component chlorophyll of the different functional types of phytoplankton within the PROTECH model are shown in Figure 20. The annual mean concentration is shown as a function of load of soluble reactive phosphorus to the lake with each line representing a different water temperature scenario. Total chlorophyll increases with increased nutrient load as shown before. Quite wide changes in water temperature have relatively modest effects on the amount of phytoplankton supported by the lake and the model forecasts a slight reduction in average annual chlorophyll *a* at higher water temperatures. This relatively modest response masks quite large changes in the response of individual functional types. The two diatoms *Asterionella* and *Fragilaria*, the green alga *Chlorella* and the dinoflagellate *Ceratium* are forecast to show slight reductions at higher temperatures. The cryptophyte *Rhodomonas* is forecast to show a large reduction in annual average biomass at higher temperatures and temperature appears to be a much larger factor controlling the abundance in this species than phosphate loading. In contrast, the two cyanobacteria *Anabaena* and *Microcystis* are forecast to increase in abundance at higher temperatures, while the other cyanobacterium *Oscillatoria* shows a marked increase at higher temperatures which is particularly marked at the higher phosphate loads.

5.4.3 Derwent Water response to temperature

The responses of total chlorophyll and component chlorophyll of the different functional types of phytoplankton within the PROTECH model are shown in Figure 21. The annual mean concentration is shown as a function of load of soluble reactive phosphorus to the lake with each line representing a different water temperature scenario. Total chlorophyll increases with increased nutrient load as shown before. Quite wide changes in water temperature have relatively modest effects on the amount of phytoplankton supported by the lake and the model forecasts a slight reduction in average annual chlorophyll *a* at higher water temperatures. Derwent Water is generally slightly more responsive to changes in phosphate load than is Ullswater and the negative effect of higher water temperature is slightly more marked, especially at the higher loadings. Certain species proved to be very unresponsive to nutrient load and to water temperature: these comprise the three diatoms *Aulacoseira*, *Tabellaria* and *Urosolenia*. The colonial green alga *Paulschulzia* was very slightly more responsive to both phosphate load and water temperature and performed slightly less well at the higher temperatures as did the diatom *Asterionella*. Another green alga, *Chlorella*, was slightly more responsive and was less abundant at the higher water temperature, particularly at the higher phosphate loads. The cryptophyte *Cryptomonas* was responsive to nutrient load and only slightly responsive to water temperature, with greatest abundance slightly above the current average values. The only functional type that showed a clear increase at the higher temperatures in Derwent Water was the cyanobacterium *Anabaena* although the extent of the increase was small.

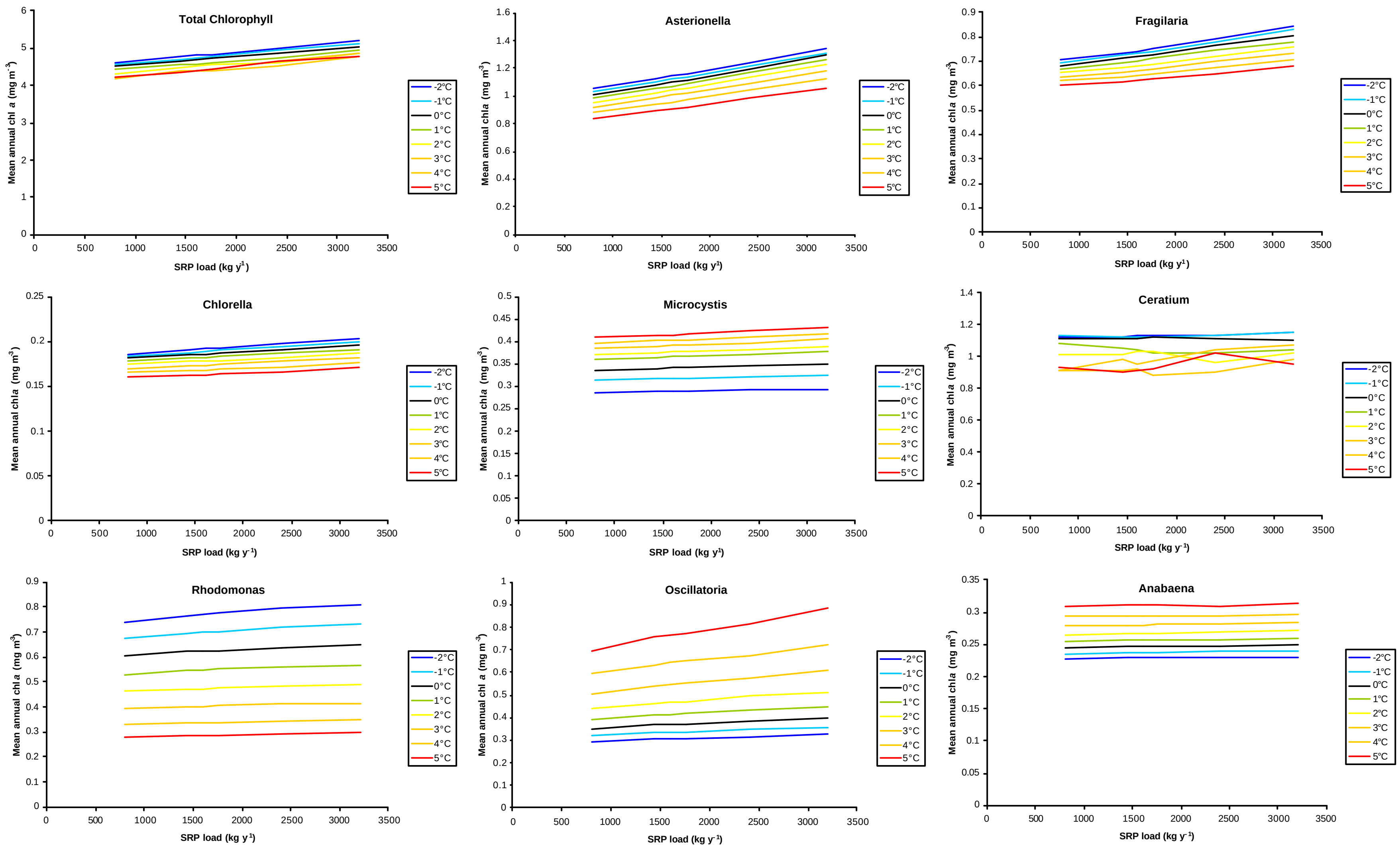


Figure 20. Response of total chlorophyll a and different species of phytoplankton to SRP load and different water temperatures in Ullswater.

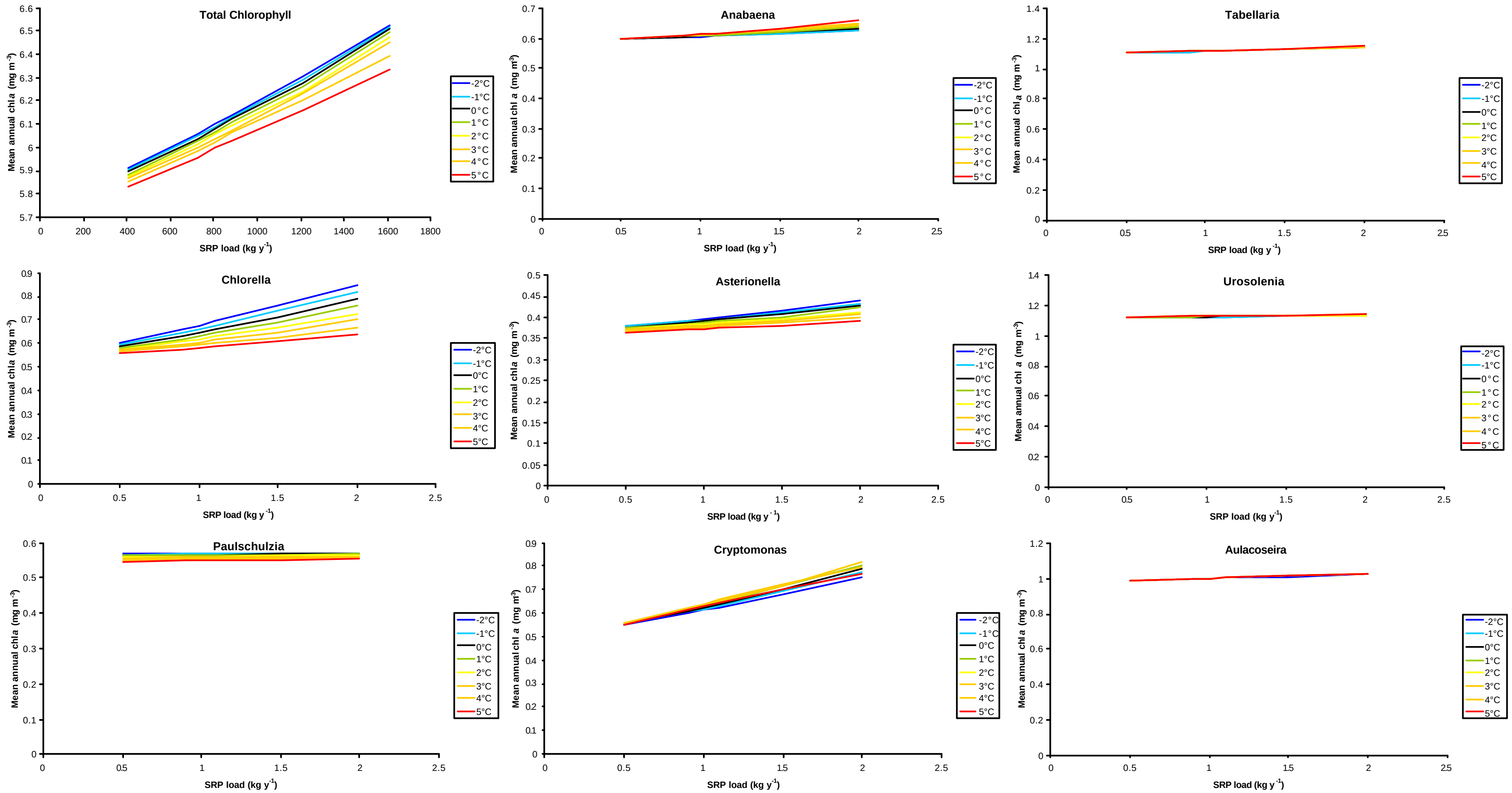


Figure 21. Response of total chlorophyll a and different species of phytoplankton to SRP load and different water temperatures in Derwent Water.

5.4.4 Preliminary conclusions on climate change

As mentioned in 5.4.1, this project was only able to study the most certain change resulting from climate change: and increase in air temperature causing an increase in water temperature. The results suggest that shifts in species composition may be greater than overall changes in chlorophyll *a*, particularly in Ullswater which appeared to be more responsive than Derwent Water. In reality, of course, other direct factors are likely to also alter. Changes in rainfall patterns are likely to have a major impact, especially in Derwent Water which is relatively rapidly flushed (Table 1). If the forecast pattern of rainfall is realised, increased storm events and higher winter rainfall may increase nutrient loading while lower summer rainfall may allow greater growth of phytoplankton by reducing hydraulic losses and this is likely also to favour the generally slower growing cyanobacteria.

6. Conclusions & Discussion

The first objective of this project was to assess whether or not Ullswater and Derwent Water currently comply with the revised draft favourable conditions targets set by the EA WQTAG of 15 and 10 mg TP m^{-3} respectively. Ullswater currently appears to comply since, based on CEH Lakes Tour data (four times per year) it had mean concentrations of 10 and 9.8 mg m^{-3} in 2000 and 2005 respectively (Table 25). However there is a small amount of uncertainty because more frequent data provided by the Environment Agency suggest that TP is much more variable than the data from CEH and suggest a higher annual mean concentration of TP of between 21 and 25 mg m^{-3}. Fortnightly data from CEH between September 1997 and November 1999 also give a mean TP concentration for Ullswater of about 9.7 mg m^{-3}. The average for Ullswater from Lakes Tour data in 1984, 1991, 1995, 2000 and 2005 is 11.4 mg m^{-3} again suggesting compliance. It is unclear at present why the data from the EA is so variable and relatively high.

Derwent Water has a lower target than Ullswater and fortnightly data from CEH suggest that it is close to this target. The target was just exceeded in 2005 (11.6 mg m^{-3}) and before that in 1997, however there appears to be a small upward trend in annual mean TP concentrations (Fig. 4) suggesting that management action is likely to be needed at Derwent Water to achieve 10 mg m^{-3} consistently.

Accurate estimates of the load of TP and SRP are intrinsically difficult to calculate because the input of these nutrients can be highly non-linear with large loads at the start of a high-flow period requiring high intensity sampling to assess load accurately. Furthermore not all the necessary data on inflow (particularly for Derwent Water) are known and very few estimates of TP exist so the relationship between SRP and TP has to be approximated. In addition there is uncertainty over the exact fate of phosphorus from septic tanks that soak away into the groundwater. Finally, there is likely to be some internal load of phosphorus within these lakes: essentially a recycling of phosphorus within the lake as a result of physical, chemical and biological processes.

Table 25. Summary of average inputs, concentrations and results of management options for Ullswater.

Characteristic	Value	Comments
INPUTS		
Mean hydrological input ($m^3\ s^{-1}$; $10^6\ m^3\ y^{-1}$)	9.32; 294	
SRP load (kg y^{-1})	1604	
NO_3-N load (kg y^{-1})	103 329	
SiO_2 load (kg y^{-1})	836 850	
TP load (kg y^{-1})	2466 - 3088	Depending on catchment SRP:TP
WwTW contribution to SRP load	26%	
WwTW contribution to TP load	15 - 19%	Depending on catchment SRP:TP
CONCENTRATIONS		
Estimated reference TP (mg m^{-3})	5.3	Morpho-edaphic index
Measured TP (mg m^{-3})	9.8	Lakes Tour 2005 (Maberly *et al.*, 2006).
TP calc. from TP load (dilution method)	8.4 – 10.5	Depending on catchment SRP:TP
TP calc. from TP load (OECD method)	6.1 – 7.3	Depending on catchment SRP:TP
Measured Chl *a* (mg m^{-3})	4.5	Lakes Tour 2005 (Maberly *et al.*, 2006).
Chl a calc. from TP load (WFD equation)	3.1	Mean of dilution and OECD TP
Chl a calc. from TP load (Metabolic model)	10.0	Mean of dilution and OECD TP
MANAGEMENT OPTIONS		
Reduction in Chl *a* if Glenridding WwTW TP = 1 g m^{-3}	8%	
Reduction in Chl *a* if catchment TP load reduced 15%	17%	
Reduction in Chl *a* if both of above	26%	

Table 26. Summary of average inputs, concentrations and results of management options for Derwent Water.

Characteristic	Value	Comments
INPUTS		
Mean hydrological input ($m^3\ s^{-1}$; $10^6\ m^3\ y^{-1}$)	7.27; 229	
SRP load ($kg\ y^{-1}$)	848	
NO_3-N load ($kg\ y^{-1}$)	71 037	
SiO_2 load ($kg\ y^{-1}$)	511 841	
TP load ($kg\ y^{-1}$)	1189 - 1417	Depending on catchment SRP:TP
WwTW contribution to SRP load	49%	
WwTW contribution to TP load	32 – 38%	Depending on catchment SRP:TP
CONCENTRATIONS		
Estimated reference TP ($mg\ m^{-3}$)	6.4	Morpho-edaphic index
Measured TP ($mg\ m^{-3}$)	8.5	Mean fortnightly data 1999-2004 (CEH).
TP calc. from TP load (dilution method)	5.2 – 6.2	Depending on catchment SRP:TP
TP calc. from TP load (OECD method)	5.2 – 6.0	Depending on catchment SRP:TP
Measured Chl *a* ($mg\ m^{-3}$)	5.9	Mean fortnightly data 2005 (CEH).
Chl a calc. from TP load (WFD equation)	2.1	Mean of dilution and OECD TP
Chl a calc. from TP load (Metabolic model)	4.8	Mean of dilution and OECD TP
MANAGEMENT OPTIONS		
Reduction in Chl *a* if all WwTW TP <= 1 $g\ m^{-3}$	24%	
Reduction in Chl *a* if catchment TP load reduced 15%	9%	
Reduction in Chl *a* if both of above	34%	

Best estimates of loads to Ullswater suggest an annual input of 1604 kg y^{-1} for SRP and between 2466 and 3088 kg y^{-1} for TP (Table 25). Using a simple dilution of load by the inflowing water, the in-lake concentration would be between 8.4 and 10.5 mg m^{-3}: quite similar to the concentration measured by CEH in the 2005 Lakes Tour of 9.8 mg m^{-3}. Using the OECD model that ascribes an average loss rate of TP to the sediment, the equivalent concentrations are rather lower at 6.1 and 7.3 mg m^{-3} (Table 25) suggesting either that the loads are too small or that this model does not work well on Ullswater. Reynolds (1992) points out that simple loading equations such as these are not very applicable to a particular lake but were intended to describe the response of a large number of lakes. WwTW contribute a relatively small amount of the TP in Ullswater: between 15 and 19%.

In Derwent Water, best estimates of annual input are 848 kg y^{-1} for SRP and between 1189 and 1417 kg y^{-1} for TP (Table 26). WwTW are responsible for a relatively large part of the total load of TP: between 32 and 38%. The dilution approach suggests a corresponding mean in-lake concentration of TP of between 5.2 and 6.2 mg m^{-3}. The OECD model gives a rather similar estimate in this case, at between 5.2 and 6.0 mg m^{-3} (Table 26). These values are rather smaller than the measured in-lake concentration between 1999 and 2004 of 8.5 mg m^{-3} which suggests that a portion of the load is not being accounted for, the hydrology is wrong, there is a substantial internal load or some combination of all three.

Converting loads or concentrations of TP to an average concentration of phytoplankton chlorophyll *a* is notoriously difficult. Two different approaches were used here: the metabolic model approach of Reynolds & Maberly (2002) and a lake-type specific equation suggested for use in the Water Framework Directive which, like the OECD model for lake TP, is based on a population of lakes rather than a specific lake. The different calculation approaches give a range of chlorophyll *a* concentrations for the different phosphorus load scenarios. However, the extent of the reduction was more consistent across the different calculations so that was used. In the case of Ullswater, the largest beneficial effect was from the 15% reduction in catchment loads to the lake which follows on from the estimated predominance of the catchment to the phosphorus load (Table 25). In the case of Derwent Water, reducing the concentration of the TP from the WwTW to 1 mg m^{-3} will have a much larger effect than a reduction in catchment load (Table 26). Given that the water quality in Derwent Water is only just complying with the 10 mg m^{-3} mean concentration of TP and failed to meet good ecological status for chlorophyll *a* in a preliminary assessment (Maberly *et al*., 2006), this would be a sensible option.

The model PROTECH performed well in both lakes, simulating the seasonal changes, quantities and types of phytoplankton in both lakes with fidelity. Model runs at different SRP loads showed an increase in phytoplankton biomass in accordance with the expectation that these are essentially phosphorus-limited lakes. However, the extent of the increase was very modest and as a result the different nutrient load scenarios did not differ substantially in forecast chlorophyll *a*. We currently do not understand the reason for this. It is possibly linked, in part, to the internal load that was needed to be added to Ullswater to match the growth of the phytoplankton since this is not an external load. However, the lack of response was also seen in Derwent Water where no internal load was needed for model runs. It should be noted that the annual mean concentration of phytoplankton chlorophyll *a* in Derwent Water between 1991 and 2004 only shows a weak relationship to the mean concentration of TP (Fig. 11) implying that physical factors such as temperature, light climate or flushing rate may be important in controlling phytoplankton populations in these lakes.

Despite this problem, an initial assessment was made with of the response of the two lakes to increased water temperature, one of the most certain effects of climate change. The results suggest that shifts in species composition is likely to be greater in extent than changes in overall biomass, particularly perhaps in Ullswater. Although in both lakes there was a slight reduction in phytoplankton biomass with increasing temperature, in Ullswater in particular there was a suggested shift towards the less preferable cyanobacteria. One other prediction for a future climate is for wetter winters and drier summers, both of which might worsen water quality by increasing loading from the catchment during the winter and reducing hydraulic losses, particularly in Derwent Water in the summer.

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Appendices