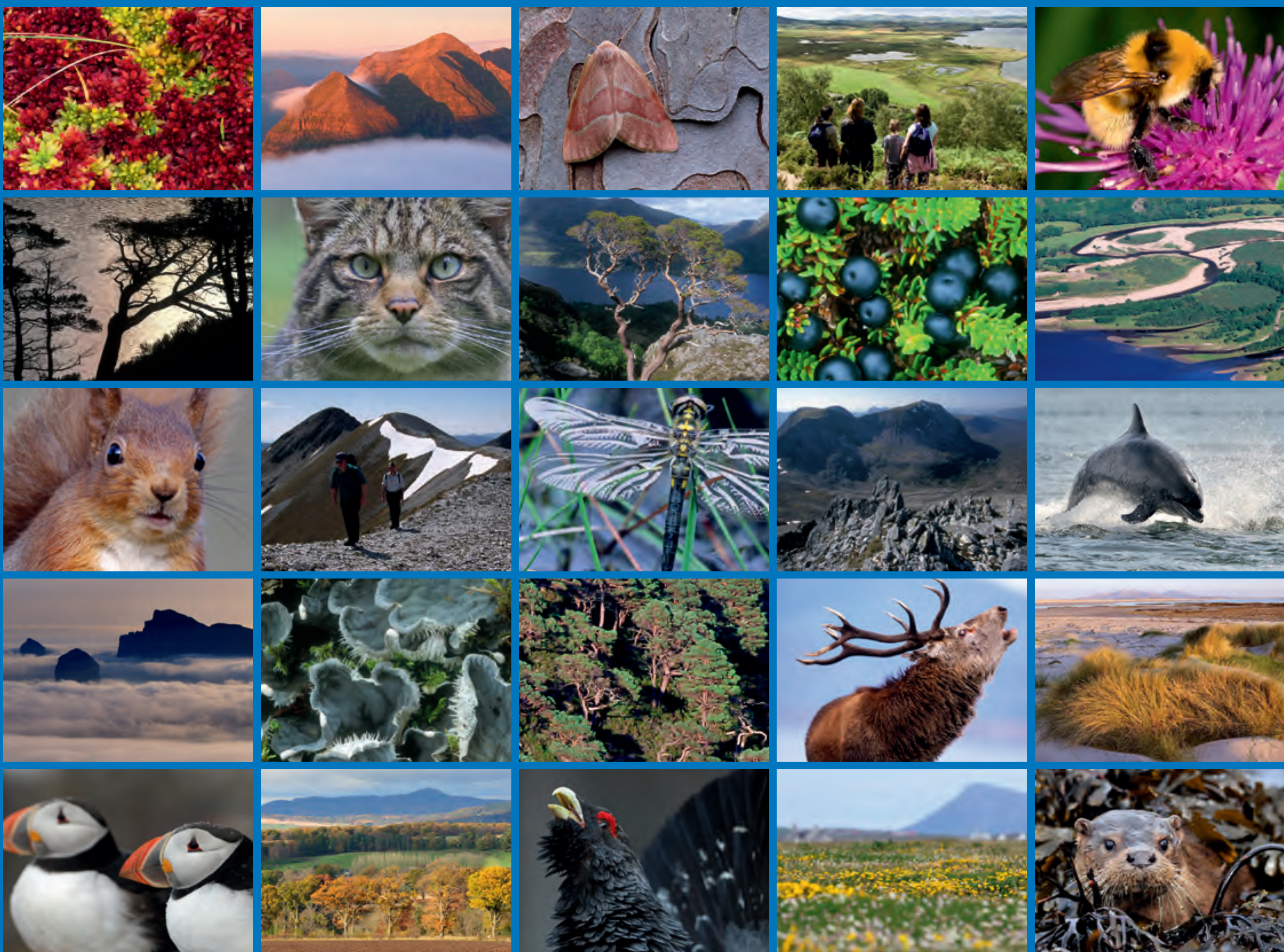
Scottish Natural Heritage
Commissioned Report No. 511

Water quality monitoring at Loch Leven 2008-2010 – Report of results



COMMISSIONED REPORT

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Water quality monitoring at Loch Leven 2008-2010 – Report of results

For further information on this report please contact:

Denise Reed
Scottish Natural Heritage
Great Glen House
INVERNESS
IV3 8NW
Telephone: 01738 458582
E-mail: Denise.Reed@snh.gov.uk

This report should be quoted as:

Dudley, B.J., Spears, B.M., Carvalho, L., Gunn, I.D.M. & May, L. (2012). Water quality monitoring at Loch Leven 2008-2010 – Report of results*. Scottish Natural Heritage Commissioned Report No.* 511

COMMISSIONED REPORT

Summary

Water quality monitoring at Loch Leven 2008-2010 – Report of results

Commissioned Report No. 511 (Project No. 12642)
Contractor: Centre for Ecology & Hydrology
Year of publication: 2012

Background

Long-term monitoring of the water quality in Loch Leven has been undertaken by the Natural Environment Research Council since the late 1960s. Over this period, the lake has suffered serious degradation due to the combined effects of eutrophication, pesticide pollution and climate change. However, following management intervention aimed at improving the water quality of the lake, the site has undergone a slow recovery. The recovery trajectory has, at times, seemed somewhat erratic. This is due to inter-annual variations in the in-lake processes that regulate the release of legacy phosphorus from the sediments following a reduction in catchment sources, although climatic variation and changes in biological interactions have also affected the recovery in the loch.

This report summarises the findings of 40 years of research on Loch Leven and presents additional data from samples collected between 2008 and 2010. The more recent data are integrated into the longer term perspective to provide an indication of whether the recovery of Loch Leven is continuing. The implications of the findings from the Loch Leven Long Term Monitoring Project, in terms of achieving water quality targets and providing key ecosystem services, are discussed.

Main findings

- Water quality improvements, which we attribute to catchment load reductions since the 1980s, have been sustained in 2008-2010.
- Cyanobacterial blooms continue to occur in late-summer and autumn in Loch Leven. During these blooms, cyanobacterial abundance often exceeds the WHO ‘Low Risk’ threshold, but has not exceeded the ‘Medium Risk’ threshold since 2004.
- A spring clear-water phase continues to be a characteristic feature of Loch Leven. This is encouraging for the further colonisation of macrophytes. The clear-water phase may be related to an increase in zooplankton diversity that may, in turn, be associated with increased grazing pressure on phytoplankton.
- In contrast to the 1970s and 1980s, when catchment sources dominated, the main source of phosphorus to the water column during the growing season has been the sediment of the lake (internal loading) since at least 2000.

- The magnitude of internal loading is related to multiple factors, including catchment pressures, weather and water quality. Climate change is likely to affect many of these factors, but the net effect of climate change on the lake's water quality is unknown.

- The intensity of internal loading in summer determines total phosphorus and chlorophyll concentrations, and water clarity (through a feedback loop). Since 2005, it appears that the magnitude of internal loading has reduced.

- Nitrogen limitation appears to be playing a larger role in restricting phytoplankton abundance in summer. The management of nitrogen in the catchment should be considered.

- Further research is required to determine the impact that existing catchment management activities have had on nutrient inputs to the loch. This would be achieved by a nutrient loading survey that incorporates detailed source apportionment.

- Further research is required to understand the interacting roles of biology, climate and nutrient supply on water quality in Loch Leven.

For further information on this project contact:
Denise Reid
Tel: 01738 458594
For further information on the SNH Research & Technical Support Programme contact:
DSU (Policy & Advice Directorate), Scottish Natural Heritage, Great Glen House, Inverness, IV3 8NW.
Tel: 01463 725000 or pads@snh.gov.uk

Table of Contents **Page**

List of Figures **Page**

List of Tables **Page**

Acknowledgements

We thank the Kinross Estate Company for their continuing in-kind support for the Loch Leven long term monitoring program, and all of the field workers and research scientists who have contributed to the project over the years. We are also grateful to SEPA, SNH, RSPB, and others for their continuing cooperation and support.

This project was co-funded by Scottish Natural Heritage and the NERC Centre for Ecology & Hydrology.

1 INTRODUCTION

Long-term monitoring of the water quality in Loch Leven has been undertaken by the Natural Environment Research Council (NERC, the parent body of the Centre for Ecology & Hydrology, CEH, and it predecessors) since the late 1960s. Over this period, the lake has suffered serious degradation due to the combined effects of eutrophication, pesticide pollution and climate change. However, following management intervention to improve the water quality of the lake to achieve 'good ecological status', the site has undergone a slow and, in the case of some variables, unstable recovery trajectory. This erratic recovery has mainly occurred as a result of in-lake processes regulating sediment phosphorus (P) release from sediments following a reduction in catchment P sources, although climatic variation and changes in biological interactions have also affected the recovery in the loch.

In this report, we summarise the findings of 10 papers on Loch Leven that were recently published in a Special Issue of the journal Hydrobiologia (Introduction) and we present data from samples collected between 2008 and 2010 (Results). The more recent data are integrated into the longer term perspective to provide an indication of whether the recovery of Loch Leven is continuing (Discussion). The implications of the findings from the Loch Leven Long Term Monitoring Project for the achievement of water quality targets and the provision of key ecosystem services are discussed.

1.1 Background

Loch Leven is a large, shallow lake in lowland Scotland, UK, that has local, national and international importance in terms of its conservation value. The site was made a Site of Special Scientific Interest (SSSI) in 1956 for its large number of wildfowl, and the outstanding number of higher plant species and rare insects that it supported. In 1964, the loch was designated as a National Nature Reserve to "maintain its attraction for all species of wildfowl and to safeguard all the characteristics that go to form this unique habitat". In 1971, it was also declared a Ramsar site, in recognition of its importance as a naturally eutrophic loch and wetland system that has international importance as an overwintering site for waterfowl. More recently, the loch became a Natura 2000 site, forming part of an international network that aims to protect natural habitats and rare species.

The loch supports a world famous trout fishery that began in the 1800s. Loch Leven trout are world famous for their quality, especially their dark colouration and unusually pink flesh. There are many historical records of them being exported to fisheries around the world in the late 1800s and early 1900s. At that time, such translocations were positive, newsworthy, events, as reported in the New York Times, 23 December 1884:

> "The Anchor Line steamer Furnessia, which sailed from Glasgow yesterday for New York, has on board 100,000 Loch Leven trout ova, which are a present from Sir Gibson Maitland to the American Fisheries Commission, to be hatched in Michigan and introduced into the Great Lakes of America"

In the mid 1830s, the loch underwent a range of hydrological modifications to provide a more reliable water supply to downstream industry. These modifications included straightening of the outflow to lower the level of the loch and the installation of sluice gates to control discharge (Munro, 1994). These engineering works, which cost the equivalent of about £40,000 at the time (approximately £2.3M by present day values), aimed to:

1. increase water availability by reducing the surface area and, consequently, evaporative losses
2. create more farmland around the shoreline
3. control the outflow to meet the water supply needs of downstream industry

The physical impact of these engineering works is well documented (Munro, 1994). The loch depth was reduced by 1.4 m and the surface area by 4.5 km^2. The shoreline was moved by up to 500m. In addition, the four existing islands were enlarged and three new islands were created. The final dimensions of the loch and its catchment are summarised in Table 1.

Changes in the hydromorphology of the system affected the ecology of the loch. The installation of the sluice gates prevented salmon, sea trout and charr from entering via the outflow. The lowering of the water level destroyed established macrophyte beds, adversely affecting the habitat and food supply of the fish community (Fleming, 1836). However, the associated 25% increase in flushing rate may have helped reduce the likelihood of algal blooms in later years, when the loch started to become more eutrophic (May & Spears, 2012a). Overall, the results of the hydromorphological modification were disappointing; the reclaimed land was poor, the value of the fishery was reduced by 1/3rd (Fleming, 1836) and the final cost was nine times that originally estimated (Munro, 1994). This demonstrates how important it is to underpin lake management with evidence based decision making (May & Spears, 2012c).

Table 1 - Morphometric data for Loch Leven after the completion of engineering works in the mid 1830s (after Smith, 1974).

Attribute	Value
Mean depth	3.9 m
Maximum depth	25.5 m
Surface area	13.3 km^2
Volume	52.4 m^3
Length	5.9 km
Breadth	2.3 km
Length of shoreline	18.5 km
Shoreline development	1.43

Loch Leven has been the subject of scientific research for almost 200 years and more than 140 articles have been published about the loch over that period. Early works characterised the biodiversity and physical characteristics of the loch, providing an important historical baseline for future research. The more structured Loch Leven long-term monitoring programme that is in place today began in 1968 under the umbrella of the International Biological Programme (IBP). Data collected from 1968 onwards now comprise one of the longest and most comprehensive limnological datasets in the world for shallow freshwater lakes, covering more than 500 physical, chemical and biological variables.

The NERC Centre for Ecology and Hydrology (CEH) have recently reviewed these data, focusing on the links between pollution, climate change and ecological responses. The results, which are published in detail in a special issue of the journal Hydrobiologia entitled “Loch Leven: 40 years of scientific research” (May & Spears, 2012b), are summarised below.

1.2 Changes in external phosphorus loading

Loch Leven has a long, and well documented, history of water quality problems resulting from eutrophication, including an increase in the frequency of cyanobacterial blooms at the site that was noted as early as 1963 (Morgan, 1974; Holden & Caines, 1974). By the mid 1980s, these had become so serious that they threatened the high conservation, recreational and economic value of the loch (May & Spears, 2012c). The observed degradation in water quality and amenity value was attributed to increases in agricultural runoff, discharges from waste water treatment works (WWTWs) and effluent from industrial sources (Holden & Caines, 1974; Bailey-Watts & Kirika, 1987).

By the 1980s, phosphorus (P) had been identified as the main nutrient limiting algal growth and biomass accumulation within Loch Leven. This is illustrated by the close, positive relationship between algal abundance and total phosphorus (TP) concentration recorded there between 1964 and 1985 (Figure 1). It was, therefore, concluded that it was necessary to reduce phosphorus inputs to the lake to improve water quality. This became the focus of a range of catchment management measures undertaken in the late 1980s and early 1990s (LLCMP, 1

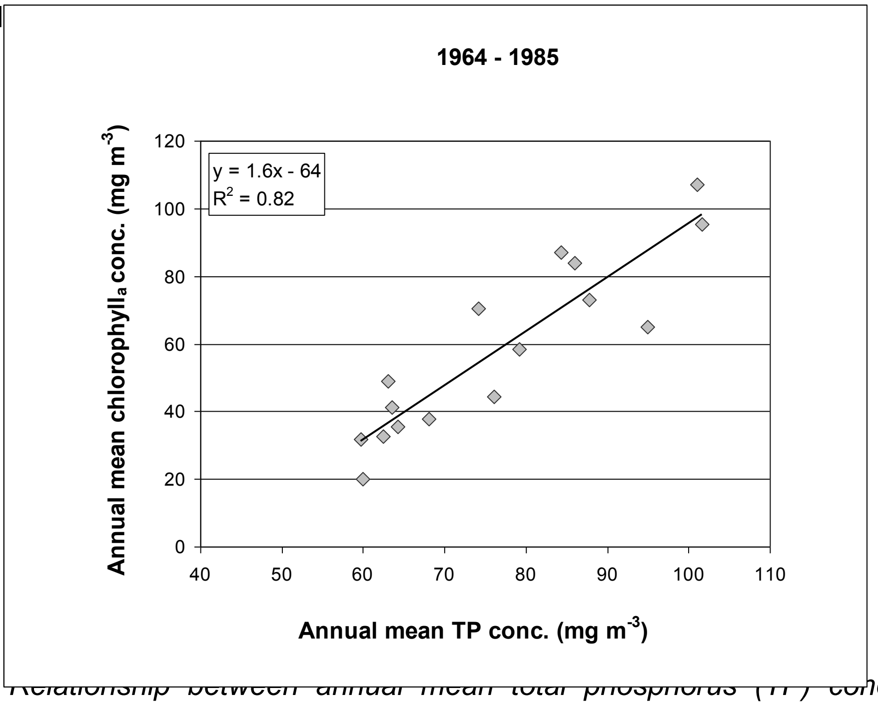


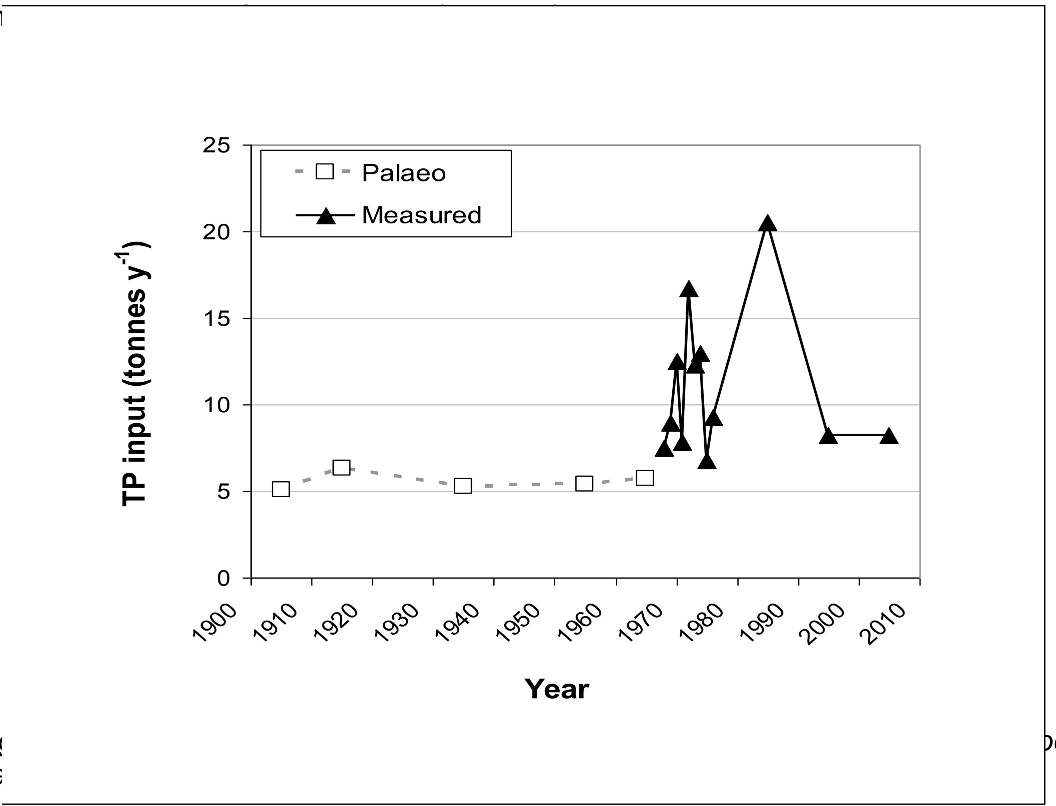
Figure 1: Relationship between annual mean total phosphorus (TP) concentration and annual mean algal abundance (expressed as chlorophyll$_a$ concentration) 1964 to 1985 (after May & Spears, 2012c).

Between 1967 and 1976, the TP input to the loch ranged between between 7 t y^{-1} and 17 t y^{-1} (Figure 2), with up to 70% of the input emanating from a single industrial source (Holden & Caines, 1974; Holden et al., 1975; Caines & Harriman, 1976). By 1985, the TP input had risen to about 20 t y^{-1} (Figure 2), with about 57% attributable to discharges from waste water treatment works (WWTWs) and industrial sources (Bailey-Watts & Kirika, 1987). Between 1985 and 1995, controls were imposed on industrial discharges (D'Arcy, 1991) and local sewage treatment facilities were upgraded (Bailey-Watts & Kirika, 1996; 1999). These improvements reduced the TP input to the loch from point sources by about 70% (Bailey-Watts & Kirika, 1996; 1999). A 53% reduction in diffuse inputs of TP was also recorded over the same period (Bailey-Watts & Kirika, 1999). By 1995, the external TP load to the lake had fallen to about 8 t y^{-1} (Figure 2), although some of this reduction may have been caused by variation in rainfall rather than changes in catchment management activities (Bailey-Watts & Kirika, 1996; 1999). This is because the level of precipitation was about 30% lower in 1995 than in 1985.

Management activities aimed at reducing the TP input to the lake continued beyond 1995, albeit at a much lower level than before and mainly aimed at reducing inputs from diffuse sources. A particular focus of these activities was the Pow Burn and its catchment. Although this accounted for only about 10% of the lake's catchment, had been shown to deliver about 30% of the total river-borne TP entering the lake during 1985 (Bailey-Watts & Kirika, 1987).

As soils in the Pow Burn area were identified as being at high risk of erosion (Frost, 1994), improved in-field management practices were introduced to reduce this risk. These included a series of grass or grass/tree buffer strips up to 20m wide that were installed along the banks of the streams to prevent eroded soils from entering the watercourses and being transported to the lake (Castle et al., 1999). In addition, farmers across the whole lake catchment were encouraged to control livestock grazing, fertiliser usage, and slurry/manure spreading (Castle et al., 1999).

The most recent survey of the TP input to the lake was undertaken in 2005 (Defew, 2008). This concluded that 8.2 t P y^{-1} was entering the loch from its catchment (Figure 2), a value that was only about 2 t P y^{-1} greater than the estimated annual average load for the relatively un



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Annual P balances for the lake were estimated at decadal intervals between 1975 and 2005 (Figure 3). These showed that varying proportions of the TP entering the lake were retained each year. In 1975, 44% of the external TP input (3 t y^{-1}) was retained. By 1985, this had risen to 61% (12.5 t y^{-1}). Once external inputs from the catchment had been reduced by about 60%, TP retention rates fell dramatically to 12% (1 t y^{-1}) in 1995 and 15% (1.25 t y^{-1}) in 2005. In contrast, however, the actual amount of TP discharged via the outflow was relatively constant throughout, i.e. 1975: 4 t y^{-1}; 1985: 8 t y^{-1}; 1995: 7 t y^{-1}; 2005: 7 t y^{-1}.

In spite of the significant reduction in TP input to the loch, annual phosphorus (P) retention values remained positive. So, P continued to accumulate in the lake sediments. These accumulations are important in terms of lake recovery because, when released into the water column under anoxic conditions, they can delay recovery in the short term (Sas, 1989; Spears et al., 2012). However, in most lakes, P release from the sediments promotes recovery in the longer term, because it increases the rate at which P is exported from the system, especially during the summer months (Sondergaard et al., 2001). This is in contrast to the situation at Loch Leven, where management of the sluice gates may restrict P discharge from the outflow in summer (May & Spears, 2012c), potentially slowing down the recovery process.

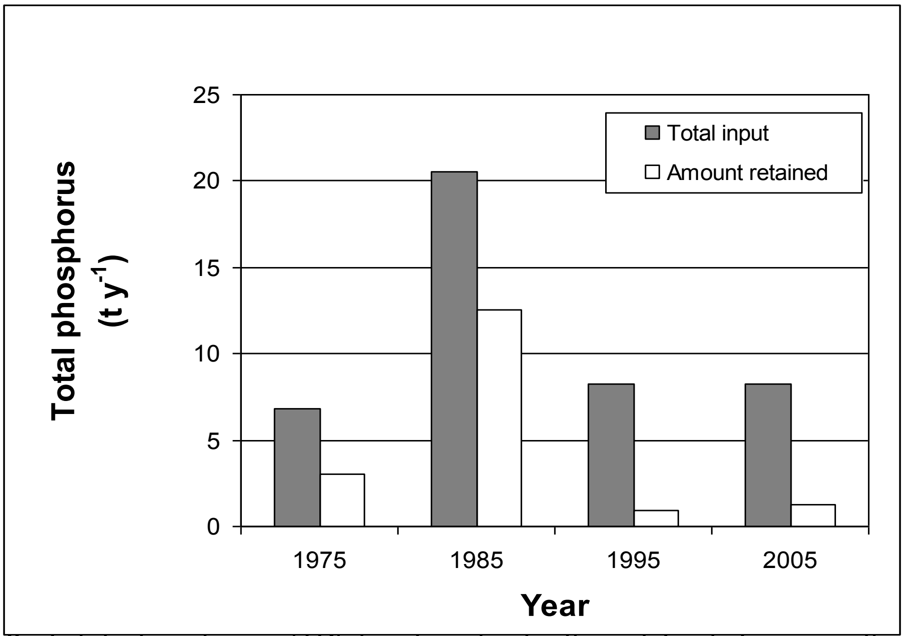


Figure 3: Total phosphorus (TP) input and retention at Loch Leven estimated at decadal intervals from 1975 to 2005 (after May et al., 2012).

1.3 Changes in internal phosphorus loading

When external inputs of P to the loch were reduced (Bailey Watts & Kirika, 1999; May et al., 2012), the lake responded slowly and significant improvements in water quality have been observed only recently (D'Arcy et al., 2006; Figure 4). Such prolonged periods of recovery are common in shallow lakes, such as Loch Leven, and are widely believed to result from a re-equilibration process ("internal loading"), whereby P that has accumulated in the sediments over many years of high inputs is slowly released to the overlying water column and flushed from the lake via its outflow (Sas, 1989).

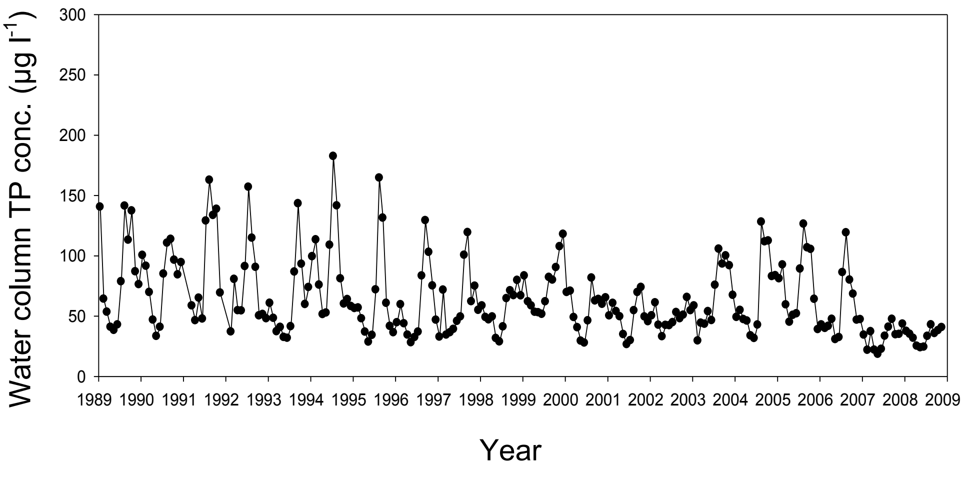


Figure 4: Monthly average total phosphorus (TP) concentrations in the water column of Loch Leven, 1989-2008 (after Spears et al, 2012).

The importance of sediment P sources and internal P cycling in determining the rate of recovery of Loch Leven was investigated by Spears et al. (2012). The results confirmed that annual mean TP concentrations within the loch were driven, mainly, by internal releases from the sediments once external inputs had been reduced (Figure 5). Variability in the magnitude and timing of internal loading during the recovery period seemed to be driven, primarily, by meteorological conditions (i.e. wind and temperature) and, to a lesser extent, water clarity (i.e. Secchi depth), rather than by TP concentrations in the sediment (Spears et al, 2012).

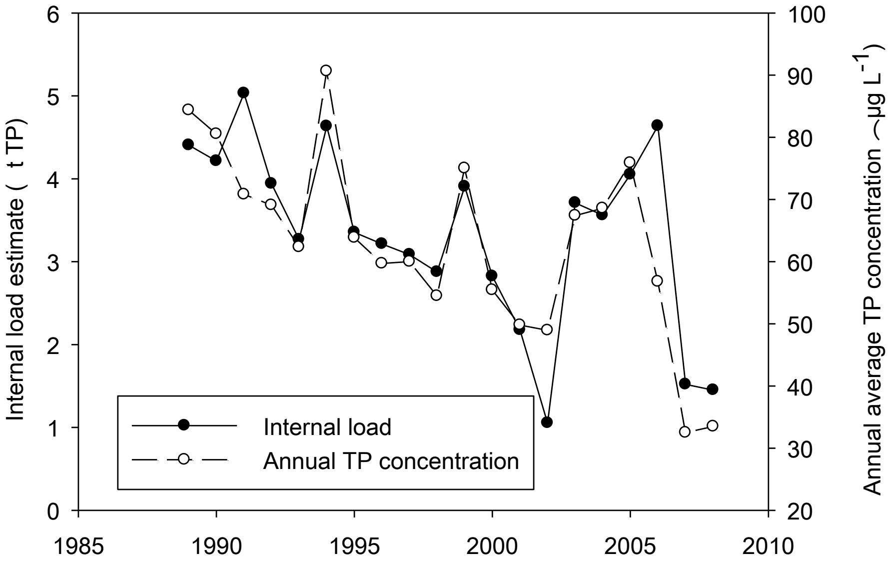


Figure 5: Internal load estimates and annual average TP concentrations for Loch Leven, 1989 to 2008 (after Spears et al., 2012).

1.4 Changes in chemical and ecological water quality, and in drivers of water quality

1.4.1 Water quality targets

Water quality targets for the restoration of Loch Leven were originally set by the Loch Leven Area Management Advisory Group (LLAMAG, 1993), based on the estimated water clarity required to support aquatic plant growing to a depth of 4.5 m (D'Arcy et al., 2006). Those initial targets were 40 µg L^{-1} annual mean TP concentration, 15 µg L^{-1} annual mean chlorophyll$_a$ concentration, and 2.5 m annual mean Secchi depth. The TP target was consistent with that inferred from diatom remains in the sediments for the pre-enrichment period of the loch (Bennion et al., 2001).

In 2006, the TP target was revised to 32 µg P L^{-1} (Carvalho et al., 2006), on the basis of data from all shallow lakes in Great Britain (not including Northern Ireland). This target was type-specific, meaning it applied to all low altitude, high alkalinity, shallow lakes in GB. More recently, new water quality targets for the loch have been proposed under the EU Water Framework Directive (WFD; European Parliament, 2000; UKTAG, 2008), which are more stringent than any earlier targets. The good/moderate (G/M) boundary concentration for a shallow, high alkalinity loch in Scotland, such as Loch Leven, is 23 µg P L^{-1} and the moderate/poor (M/P) boundary is 46 µg P L^{-1} (UKTAG, 2008). The G/M class boundary for chlorophyll$_a$ for a shallow, high alkalinity lake, such as Loch Leven, is 7.5 µg L^{-1} (annual arithmetic mean; Carvalho et al., 2006). However, as Loch Leven is near the 3m mean depth boundary that separates WFD 'shallow' and 'very shallow' lake types, the chlorophyll$_a$ targets for this site have been further revised, on the basis of site-specific depth and alkalinity values, to 11 µg L^{-1} for the G/M boundary and 22 µg L^{-1} for the M/P boundary (Carvalho et al., 2009). Site-specific TP targets based on the Morpho-Edaphic Index (MEI) approach are also recommended if sufficient data on depth and alkalinity are known (UKTAG, 2008). These use regression equations, which are different for Central and Northern European lakes, to identify a lake-specific reference TP concentration (Cardoso et al., 2008) and, from this reference value, TP concentrations for status class boundaries are set based on Ecological Quality Ratios (EQRs) formally agreed in the WFD CIS intercalibration process (EC 2008). SEPA have chosen to intercalibrate all its lakes in the Northern European IC group (Northern GIG) and for this reason site-specific TP targets for Loch Leven, using the Northern European regression, should be 16, 22 and 45 µg L^{-1} for the H/G (high/good), G/M and M/P boundaries.

If Loch Leven had site-specific TP targets based on the Central European regression slope, these would be 25, 35 and 69 µg L^{-1} for the H/G, G/M and M/P boundaries. Central European standards are applied to all lowland lakes in England other than the south-west, whereas Northern European standards are applied to all lakes in Scotland, irrespective of location and altitude (UKTAG, 2008). It is our opinion that the Northern site-specific target of 22 µg L^{-1} for Good status is too stringent for Loch Leven and more or less unachievable. In our expert opinion, the current ecological structure and functioning of Loch Leven relates more to good ecological status than moderate status in terms of our historical understanding of the lake's conidtion (diatom-inferred TP and macrophyte composition; Bennion et al., 2001; Salgado et al. 2009; Dudley et al., 2012). Also, Loch Leven's location in lowland Eastern Scotland corresponds better to Central European lake types than to the Nordic lake types. For these reasons, we believe that a G/M TP boundary of 35 µg L^{-1} is more appropriate than 22 µg L^{-1}.

1.4.2 Water quality indicators

Annual and seasonal trends in the main water quality target variables (i.e. TP and chlorophyll$_a$ concentrations, and Secchi depth transparency) were examined for the period 1968 to 2008 (Carvalho et al., 2012). Annual mean TP concentrations were found to have declined from over 100 µg L^{-1} in the early 1970s to below the LLAMAG target of 40 µg L^{-1} in recent years (Figure 6a). The most noticeable declines were in the early 1970s, when *Daphnia* re-appeared after a long absence, and in 2007 and 2008 (to 32 µg L^{-1} and 33 µg L^{-1}, respectively), when there was a marked reduction in P recycling from the sediments. The latter values are at or near the WFD G/M status boundary of 32 µg L^{-1}.

Chlorophyll$_a$ concentrations also declined rapidly in the first half of the 1970s, with annual means falling from over 90 µg L^{-1} to about 40 µg L^{-1}. Since then, annual mean concentrations have fluctuated, mainly between 30 µg L^{-1} and 50 µg L^{-1} (Figure 6b). Although chlorophyll$_a$ levels were relatively low in 2007 (26 µg L^{-1}) and 2008 (25 µg L^{-1}), they were still well above the LLAMAG target of 15 µg L^{-1} and WFD M/P and G/M class boundaries of 22 µg L^{-1} and 11 µg L^{-1}, respectively.

Annual mean Secchi depth transparency showed a rapid improvement in the early 1970s, increasing from around 1.0m to about 1.5m in about 5 years. Since then, water clarity has been relatively stable with annual mean values generally ranging between 1.2m and 1.7m (Figure 6c) (Carvalho et al., 2012). Although Secchi depths were slightly above average (1.63m and 1.61m) in 2007 and 2008, they were still well below the LLAMAG target of 2.5m. However, a recent investigation into the relationship between macrophyte growing depth and water clarity at Loch Leven (May & Carvalho, 2010) suggests that an annual mean target value may not reflect improvements in the loch. This is because the growing depth of submerged macrophytes is more sensitive to spring water clarity than annual mean water clarity. As such, the water clarity target set by LLAMAG, which was based on annual mean values, may be too stringent. It is recommended that this is reviewed.

1.4.3 Biologically available nutrients

Changes in the seasonality of biologically available nutrients (i.e. soluble reactive phosphorus (SRP), nitrogen as nitrate and soluble reactive silicon (SRSi)) affect ecological responses such as chlorophyll$_a$ concentrations. Spring SRP concentrations were found to have declined slightly in the 1980s and increased slightly from the 1990s, onwards. In contrast, winter concentrations showed a more consistent declining trend (Carvalho et al., 2012). A comparison of the seasonality in SRP concentrations for 1968-1977 and 1998-2007 (Figure 7a), clearly shows that SRP concentrations have declined in most months of the year, apart from spring, over the monitoring period.

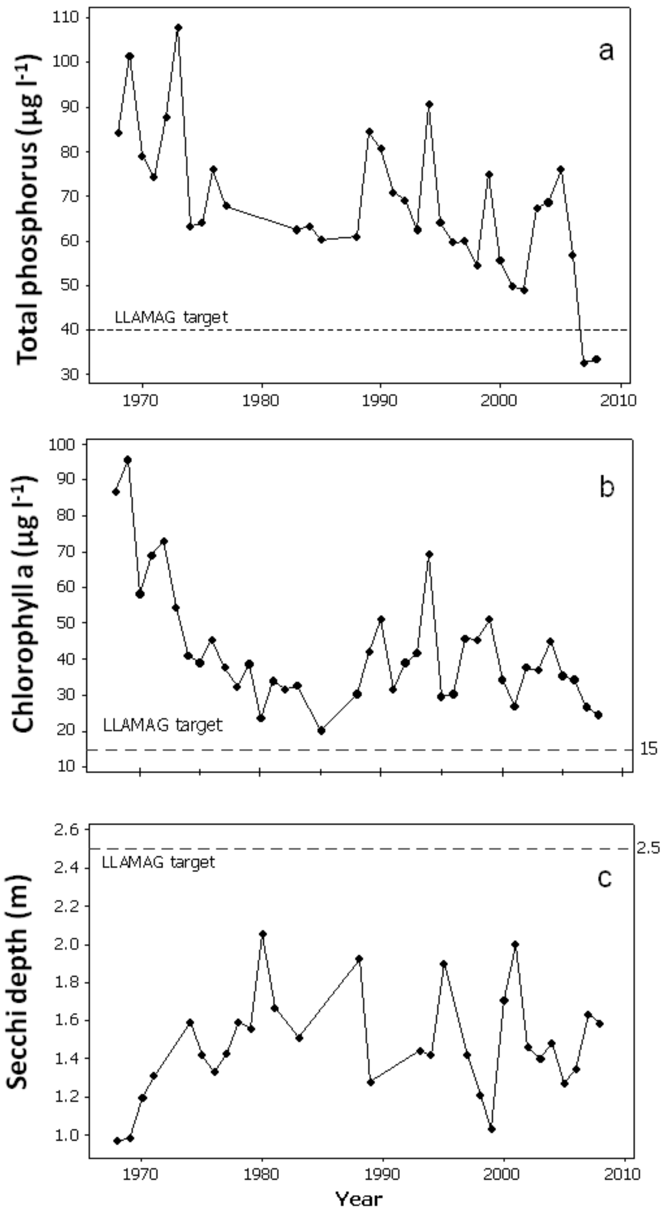


Figure 6: Annual mean total phosphorus (a) and chlorophyll$_a$ concentrations (b), and Secchi depth transparency (c) in Loch Leven, 1968-2008. LLAMAG water quality targets are shown (after Carvalho et al., 2012).

Winter nitrate concentrations, apart from those in 1991/92 which were excluded from the analyses due to data quality issues, showed an increasing trend. In terms of seasonality, nitrate concentrations for 1968-1977 and 1998-2007 were much higher in the first six months of the year in the last decade than in the earlier decade (Figure 7b). Concentrations in August, September and October were at or below the analytical detection limit (0.01 mg L^{-1}) in both decades.

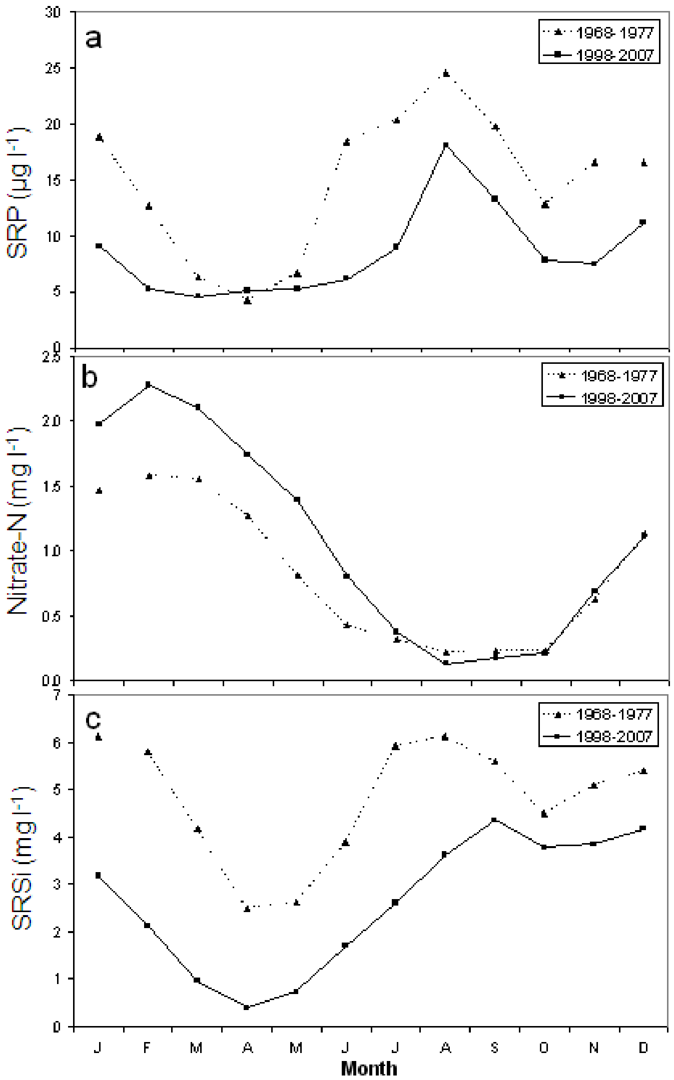


Figure 7: Comparison of seasonality in monthly mean concentrations of (a) soluble reactive phosphorus (SRP), (b) nitrate-nitrogen (nitrate-N) and (c) soluble reactive silicon (SRSi) in Loch Leven for two decades: 1968-1977 and 1998-2007 (after Carvalho et al., 2012).

Comparing the seasonality in SRSi concentrations for 1968-1977 and 1998-2007 showed that concentrations have declined in every month of the year (Figure 7c) over the monitoring period. Since 2000, concentrations in March and April have frequently remained below 0.5 mg L^{-1}.

1.4.4 Air temperature and rainfall

Air temperature has increased in spring, autumn and winter over the period 1968 to 2008 (Figure 8), but not in summer (Carvalho et al., 2008). That said, the summers of 2003-2006 were warmer than average, with 2003 being the warmest (i.e. 1.6 degrees warmer than the 30 year average summer mean between 1971 and 2000). The most anomalous air temperature was recorded in the winter of 1989, when the mean air temperature was 5.8 °C, almost three degrees above the 30 year average.

In general, winter rainfall increased over the monitoring period (Figure 9). However, rainfall was very variable in all seasons. The years 1990, 1995, 2000 and 2007 had particularly wet winters, with about 50 percent more rainfall than the 30 year average of 293 mm (Carvalho et al., 2012). In contrast, 1976, 1996 and 2006 had particularly dry winters, with about

40 percent less rainfall than the 30 year average. The wettest summers were 1985, 1988, 2007 and 2008, with more than 300mm of rain (compared to the 30 year average of 196mm) and the driest summer, by far, was 1995 with only 64 mm of rain.

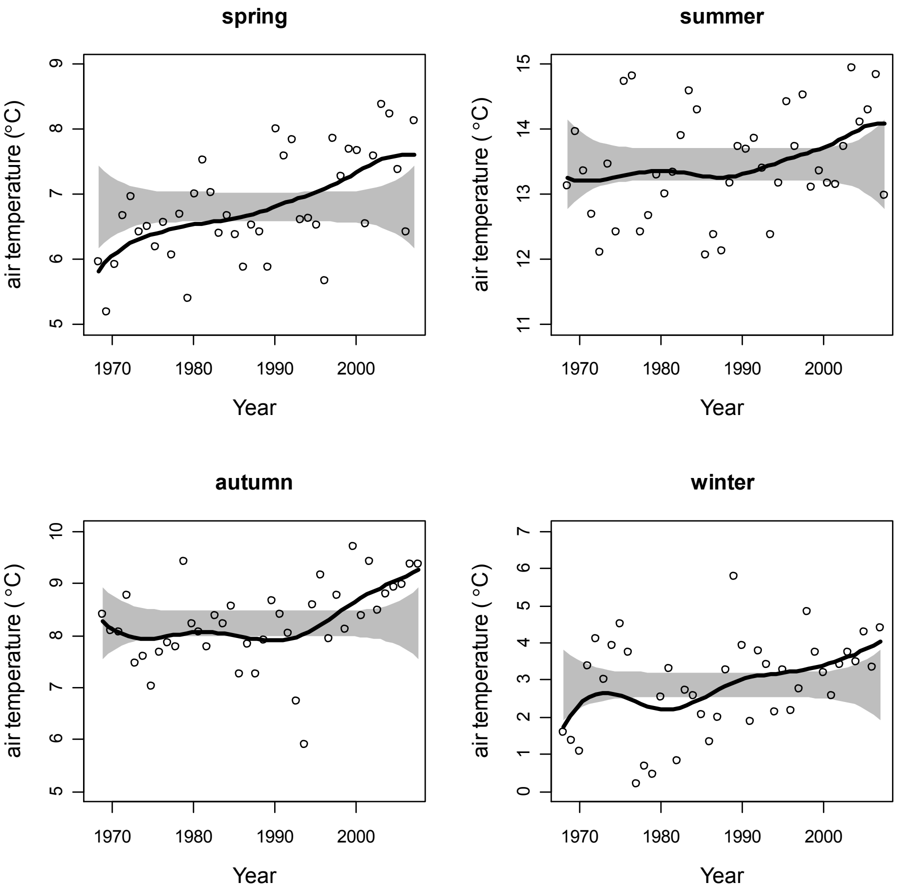


Figure 8: Trends in seasonal mean air temperature at Loch Leven, 1968-2007; shaded reference band indicates 'no effect' and shows where the curve is expected to lie if there has been no change over time; NB each plot has a different scale on the y-axis (after Carvalho et al., 2012).

1.4.5 Daphnia population densities

Daphnia are the main grazers of algae in Loch Leven. Since 1975, *Daphnia* densities have shown no significant trends in terms of seasonal means. However, when the last two decades are compared (Figure 10), it is clear that densities have increased markedly in spring (May), and declined in summer (July to October) (Carvalho et al., 2012).

1.4.6 Chlorophyll$_a$ concentrations

Chlorophyll$_a$ concentrations have declined in spring, summer and winter over the 40 year period, although reductions in the latter two seasons were largely confined to the first decade of monitoring (Figure 11) (Carvalho et al., 2012). Over the last 20 years (1988-2007), decreasing concentrations were only recorded in spring, with particularly low chlorophyll$_a$ concentrations, often < 10 µg L^{-1}, being recorded in May from 2000, onwards. These concentrations contrast with those of more than 100 µg L^{-1} that were often recorded in this month between 1968 and 1973.

Low summer chlorophyll$_a$ concentrations in recent years (1985, 2004, 2007, 2008) were associated with very wet summers. Conversely, some of the highest chlorophyll$_a$ concentrations were recorded in the driest summers (1994, 1995, 2006). Regression analysis was carried out to investigate the possible impact of other potential drivers (e.g. nutrients, air temperature and *Daphnia* abundance) on summer chlorophyll$_a$ concentrations, but only rainfall showed a significant relationship.

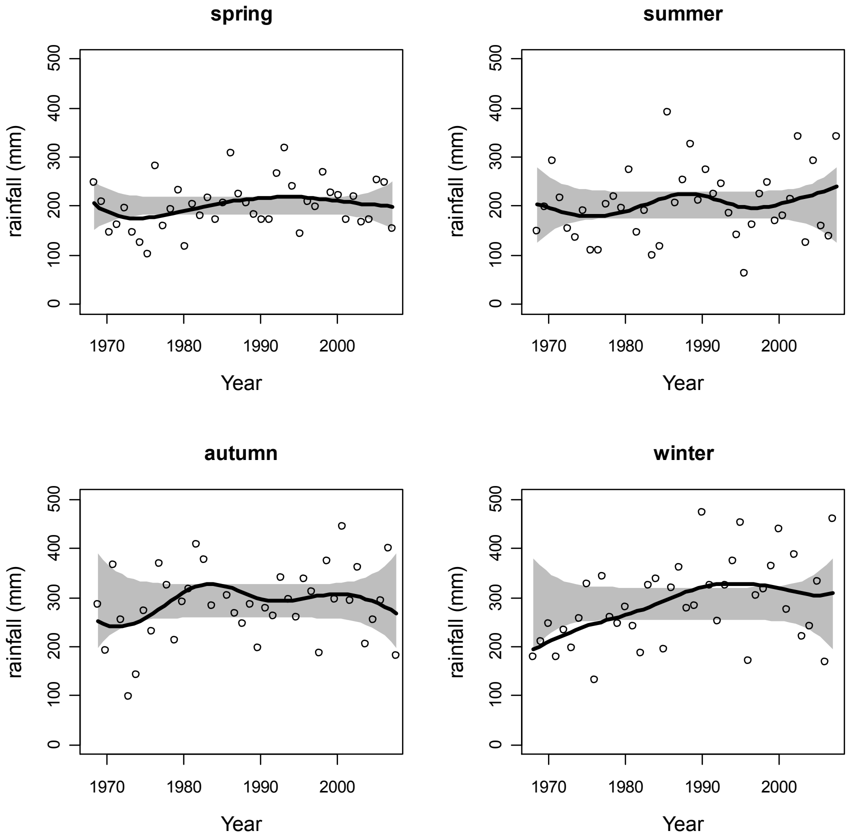


Figure 9: Trends in seasonal rainfall at Loch Leven, 1968-2007; shaded reference band indicates 'no effect' and shows where the curve is expected to lie if there has been no change over time (after Carvalho et al., 2012).

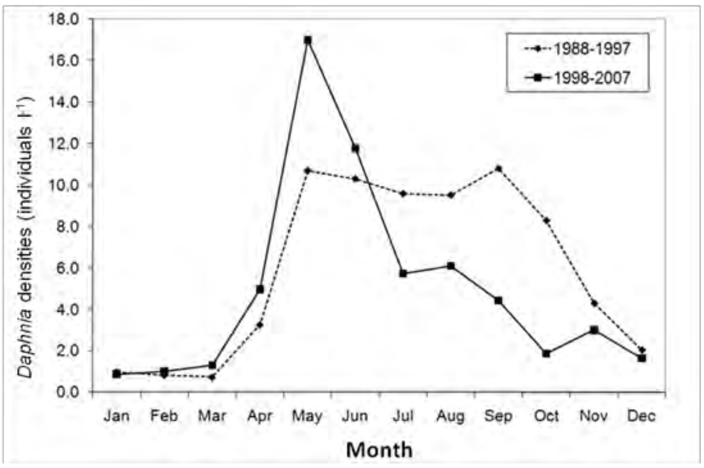


Figure 10: Average monthly <u>*Daphnia*</u> *densities in Loch Leven, 1988-1997 and 1998-2007 (after Carvalho et al., 2012).*

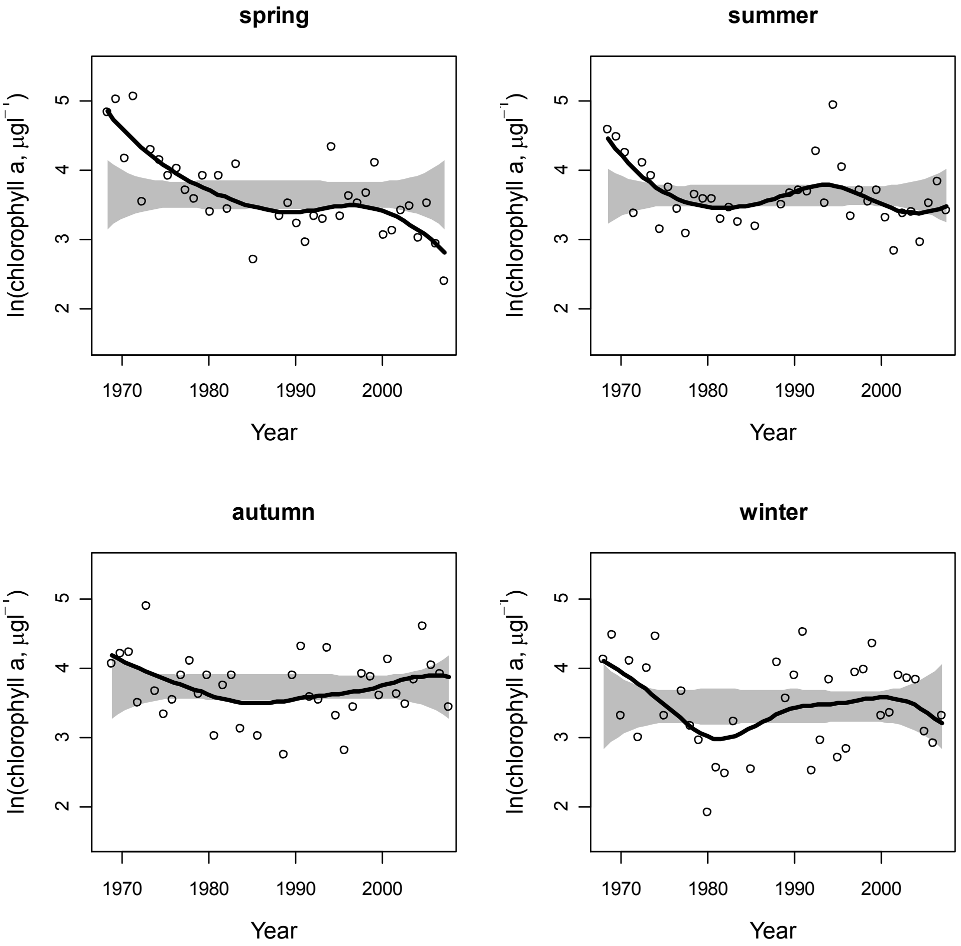


Figure 11: Trends in seasonal mean Ln chlorophyll$_a$ concentrations in Loch Leven, 1968-2007; shaded reference band indicates 'no effect' and shows where the curve is expected to lie if there has been no change over time (after Carvalho et al., 2012).

1.4.7 Water clarity

Analysis of Secchi disc transparency revealed an increasing trend in spring, largely driven by the marked increases that were recorded in the early 1970s (Figure 12; Carvalho et al., 2012). However, transparency has also increased markedly in May and June in recent years, with Secchi depths of ≥ 3 m being recorded for these months since 2000. These values contrast with those of < 1 m that were commonly recorded during the first three years of the monitoring record.

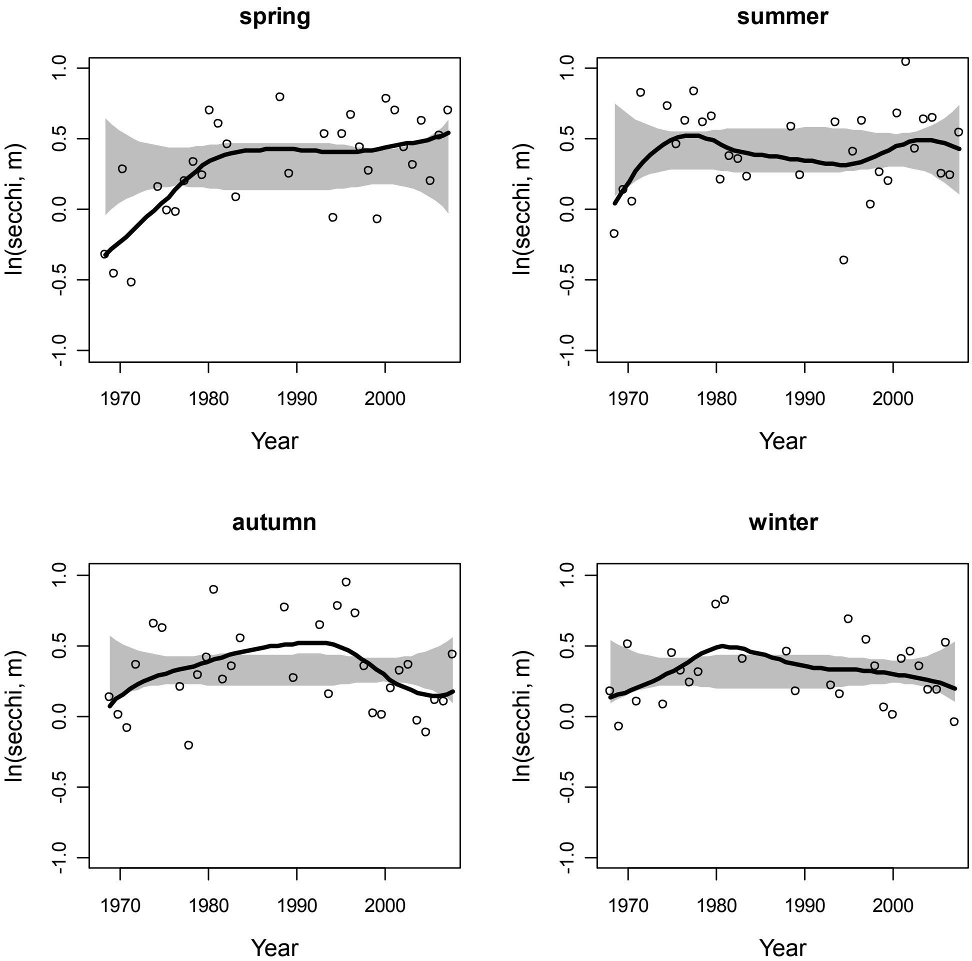


Figure 12: Trends in seasonal mean Ln Secchi depth in Loch Leven, 1968-2007; shaded reference band indicates 'no effect' and shows where the curve is expected to lie if there has been no change over time (after Carvalho et al., 2012).

1.5 Other ecological responses

1.5.1 Macrophytes

Dudley et al. (2012) assessed changes in the macrophyte community of Loch Leven over a 100 year period by comparing survey data from 1905, 1966, 1972, 1975, 1986, 1993, 1999 and 2008 (Table 2). Eutrophication and recovery, once nutrient inputs to the loch had been reduced, were associated with changes in community composition.

In each of the four most recent surveys, the loch was divided into 19 sectors. Each of these contained at least one transect ranging from the shallowest to the deepest occurrence of macrophytes. A range of indicators of recovery were applied to these data: relative abundance of taxa, taxon richness and evenness. All of these metrics indicated that an improvement had taken place since 1972, and that species richness had increased in recent years. These measures, together with ordination of presence/absence composition data from all survey years, indicated that the macrophyte community in the loch is now recovering towards the state that was recorded in 1905.

Both the taxa present, and their relative abundances, varied considerably over time (Table 3). All of the metrics presented suggested that there had been a decline in the macrophyte community between the early 1900s and the 1970s/1980s, followed by a recovery that appeared to be continuing into 2008. This is consistent with similar patterns observed in water quality and other biota, as reported by Carvalho et al. (2012), Gunn et al. (2012), May et al. (2012) and Spears et al (2012). Changes in species richness over time showed this pattern most clearly. Of the 18 taxa recorded over the entire period of study, the highest number (16) occurred in 1905, while eight or less were found in the 1966, 1986 and 1993 surveys, and 13 were recorded in the most recent survey (2008).

Table 2 - Sources of macrophyte data showing year of survey, total number of samples collected, and literature source (after Dudley et al., 2012).

Year	**Samples**	**Source**
1905	unknown	West (1910)
1966	853	Jupp et al. (1974)
1972	744	Jupp et al. (1974)
1975	556	Britton (1975)
1986	190	Robson (1986)
1993	233	Murphy & Milligan (1993)
1999	233	Griffin & Milligan (1999)
2008	255	CEH unpublished data

These improvements in community measures are supported by recent increases in the maximum growing depth of macrophyes (May & Carvalho, 2010). This measure was used as a target water quality indicator by the Loch Leven Catchment Management Plan (LLCMP, 1999), and was set to 4.5 m to reflect the value observed in 1905 (West, 1910). This target appears to have been almost met, now that the loch is recovering (Figure 13).

Growing depths of only 1.5 m in the early 1970s would have restricted plant growth to the more wave-disturbed and sandier sediments of the loch (Jupp & Spence, 1977; Spears & Jones, 2010), while growing depths of up to 4.5 m enable additional species usually associated with more stable, finer sediments, to flourish. In addition, reductions in open water nutrient concentrations (Carvalho et al., 2012), especially soluble reactive phosphorus (SRP) concentrations, and enhanced nitrogen limitation in summer, can reduce epiphyte burdens on macrophyte leaves. This explains why slower growing macrophytes, such as *Potamogeton praelongus*, are beginning to compete more successfully with faster growing species, such as *Potamogeton berchtoldii* / *pusillus* and *Chara globularis*.

These results clearly show that the Loch Leven macrophyte community is becoming more diverse in terms of species richness and evenness, and increasingly similar to the species recorded in 1905. However, palaeobotanical studies of aquatic plant macrofossils at Loch Leven (Salgado et al., 2009) suggest that even 1905 was not an undisturbed baseline. In the 17th and 18th centuries, even less competitive species, such as *Isoetes lacustris* and *Lobelia dortmanna*, were abundant in the loch. By 1905, these had already been lost, probably due to a combination of increasing nutrient pressures (Salgado et al., 2009) and the hydrological modifications in the mid 1800s that are described in Section 1.1.

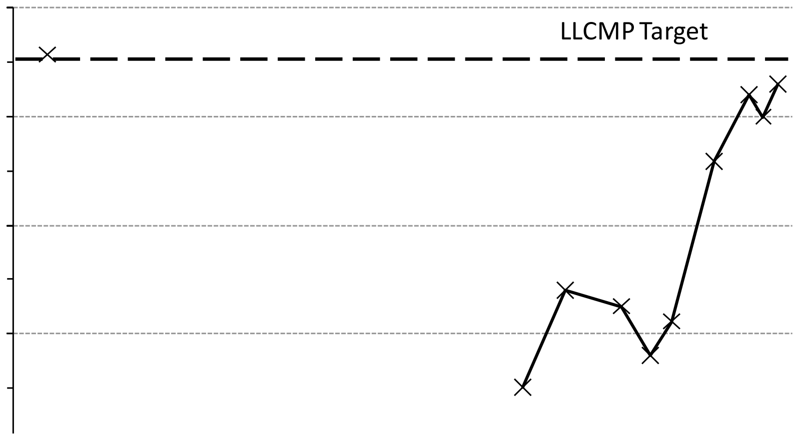


Figure 13: Long term change in maximum growing depth of aquatic plants in Loch Leven. The dashed line indicates the target for recovery set by the Loch Leven Catchment Management Plan (LLCMP) (after Dudley et al., 2012).

Table 3 - Taxa included in the analyses of macrophyte data from Loch Leven, including year of survey and relative abundance, expressed as a percentage. NB 'x' indicates no abundance information available (after Dudley et al., 2012).

Taxa	**1905**	**1966**	**1972**	**1975**	**1986**	**1993**	**1999**	**2008**
Chara spp.	x	97	30	37	56	56	52	22
Potamogeton berchtoldii / pusillus	x			1		1	8	21
Nitella / Tolypella	x	1	13	28				19
Callitriche hermaphroditica	x		3	1	3	6	10	14
Potamogeton perfoliatus	x		1	1	6	7	7	7
Potamogeton filiformis/pectinatus	x	1	25	28	21	5	19	7
Elodea canadensis	x	2	1	2	2	6	1	6
Zannichellia palustris		1	30	5	13	21	2	5
Eleocharis acicularis	x			2		1	2	1
Myriophyllum spp.	x	1	1					1
Potamogeton praelongus	x							1
Ranunculus spp.	x							1
Littorella uniflora	x		1	1	2	1	2	1
Potamogeton crispus		2	1	1	1	2		
Potamogeton gramineus	x							
Potamogeton lucens	x							
Potamogeton obtusifolius	x		1					
Potamogeton x *zizii*	x							

1.5.2 Benthic macroinvertebrates and zooplankton

Gunn et al. (2102) assessed the responses of benthic macroinvertebrates and zooplankton to the eutrophication and subsequent recovery of Loch Leven. They found that, in recent years, the number of macroinvertebrate and zooplankton taxa had increased (including taxa considered to be sensitive to nutrient enrichment) and that abundances had declined. These changes appear to reflect improvements in water quality and habitat conditions that have occurred as a result of the recent reduction in nutrient loads (Table 4), although a substantial delay in ecological response was detected. This time lag in recovery has important implications for the use of invertebrate data to assess improvements in the ecological status of lake systems, as is required by the EU Water Framework Directive.

Table 4 - Summary, by invertebrate group, of biotic signals indicative of ecological recovery in Loch Leven since the IBP baseline studies of 1966-1973 (after Gunn et al., 2012).

Invertebrate Group	Species Composition	Abundance
Crustacean zooplankton	Increased species diversity	Increased *Daphnia* densities in May; lower mean summer monthly maxima
Rotifer zooplankton	Increased species diversity	Total rotifer abundance significantly declined in 1990s
Littoral benthos	Increased species diversity	Unclear - insufficient data to draw conclusions
Sub-littoral benthos	Unclear - no data since 1994	Unclear - no data since 1994

1.5.2.1 Zooplankton

The crustacean zooplankton in Loch Leven has been co-dominated by *Daphnia* and *Cyclops* populations since the early 1990s, with population maxima occurring in late spring/early summer (e.g. Gunn et al., 1999). Over the last two decades, *Daphnia* densities have increased markedly in May, probably in response to warmer spring temperatures, and have generally decreased later in the year (July to October). This is reflected in the decline in the mean summer (June to August) monthly maxima shown in Figure 14 (upper panel). The taxon richness of the crustacean zooplankton community between 1975 and 2007 was greater than recorded in the IBP study (1968-1973), when the crustacean zooplankton community was almost entirely composed of the copepod *Cyclops abyssorum*, although small numbers of *Bythotrephes longimanus*, *Leptodora kindti* and *Eudiaptomus gracilis* were also found (Johnson & Walker, 1974). In 1970, the *Daphnia hyalina* species complex was detected in the loch for the first time after an absence of 15-20 years (Morgan, 1970; Johnson & Walker, 1974). From the mid 1970s onwards, the species composition of the crustacean zooplankton remained fairly stable with *Daphnia* and *C. abyssorum* co-dominating the community, the copepod *E. gracilis* occurring in smaller numbers, and the predatory cladocerans *L. kindti* and *B. longimanus* occurring occasionally and in very low numbers in summer (May et al., 1993; Gunn et al., 1994; Gunn & May, 1995, 1996, 1997, 1998, 1999). The community composition changed again in the 2000s, with *Cyclops vicinus* returning after a long period of absence (Carvalho et al., 2002, 2007, 2008) and *Bosmina longirostris* being recorded in large numbers for the first time in 2007 (Carvalho et al., 2008). It should be noted that the most significant change in the crustacean zooplankton community, the reappearance of *Daphnia* in 1970, occurred before the most recent restoration programme began and is believed to have been a response to reductions in the discharge of dieldrin by local industry (D'Arcy et al., 2006; May & Spears, 2012c) rather than to any change in nutrient input.

Total rotifer zooplankton abundance was relatively low in the 1990s compared with 1977-82 (Figure 14). This suggested that the reduction in nutrient input to the loch, which began in the late 1980s, had strongly influenced rotifer abundance, probably through its impact on food availability. Fourteen rotifer species were recorded from samples collected between 1977 and 1982 (May, 1980), and those collected between 1991 and 1998 contained 21 species. All of the species present in the earlier period were also present in the later period. However, seven additional species appeared in the later period: *Collotheca mutabilis*, *Conochilus hippocrepis*, *Kellicottia longispina*, *Polyarthra vulgaris*, *Synchaeta oblonga*, *Synchaeta pectinata* and *Synchaeta tremula*. Although most of these species can be found in many types of lake across the trophic gradient, *C. hippocrepis* and *K. longispina* are usually associated with more oligotrophic systems (Ruttner-Kolisko, 1974) and their appearance may be associated with the reduction in nutrient inputs that occurred in the early 1990s. In contrast, the appearance of *S. tremula* in the 1990s probably reflects the increase in macrophyte coverage (Dudley et al., 2012; May & Carvalho, 2010), as this species tends to live in the littoral zone (Ruttner-Kolisko, 1974).

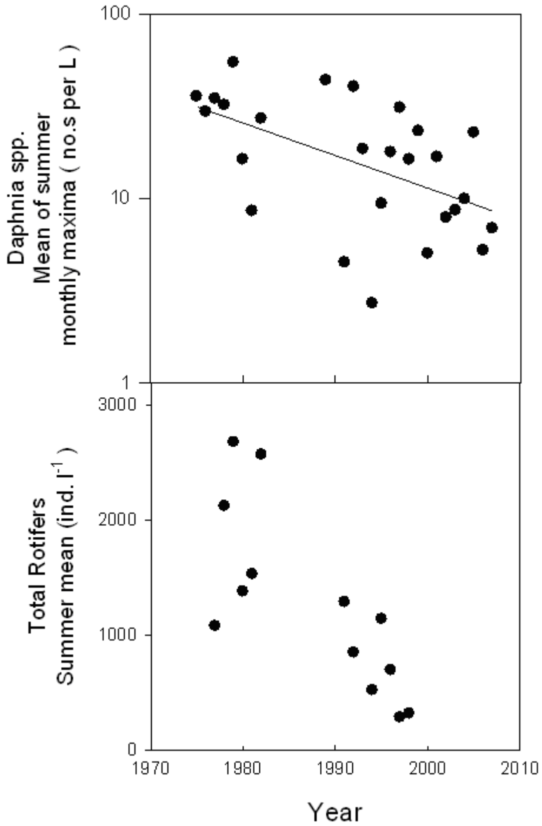


Figure 14: Decreasing trends in abundance of zooplankton in Loch Leven over the study period for Daphnia (upper panel) and rotifers (lower panel) (after Gunn et al., 2012).

1.5.2.2 Littoral benthos

Ninety-six taxa were recorded from the littoral benthos before 1966, 41 taxa between 1966 and 1973 (IBP studies), and 153 taxa between 1993 and 2006 (Figure 15). Of these, 24 of those recorded before 1966 were not recorded again, and 48 of those recorded between 1993 and 2006 were recorded for the first time during that period. The littoral community is now dominated, numerically, by Crustacea (*Asellus aquaticus, Gammarus pulex*), Oligochaeta (Tubificidae, Lumbriculidae), Chironomidae, Trichoptera (*Tinodes waeneri*, *Agraylea multipunctata*) and Ephemeroptera (*Caenis luctuosa*). The numbers of Plecoptera, Ephemeroptera, Coleoptera and Trichoptera species recorded between 1998 and 2006 were much higher than those recorded in earlier surveys and, in the later surveys, the species recorded included pollution intolerant species such as *Diura bicaudata* and *Ecdyonurus dispar*.

1.5.2.3 Sub-littoral benthos

The species composition, mean densities and spatial distribution of the sub-littoral benthos recorded in 1994 were similar to those recorded in the IBP studies (Maitland & Hudspith, 1974; Charles et al., 1974a). The most abundant macroinvertebrate groups were Nematoda, Oligochaeta (mainly Tubificidae), Diptera (mainly Chironomidae) and Mollusca (mainly *Pisidium* spp. and *Valvata piscinalis*). However, new records included a chironomid genus (*Paracladopelma*) that is considered to be relatively tolerant of organic pollution (Wilson & Ruse, 2005) and three oligochaete species (*Uncinais ucinata*, *Psammoryctides barbatus*, *Limnodrilus claparedeianus*), that have also been recorded in other eutrophic lakes within the UK (Carter & Murphy, 1993; Potter & Learner, 1974). The chironomid *Endochironomus*, a taxon considered tolerant of organic pollution (Wilson & Ruse, 2005), last recorded in 1969 (Charles et al., 1974), re-appeared in 1994. Several taxa found during the IBP studies were not recorded in the May 1994 survey, including a number of chironomid genera (e.g. *Diamesa*, *Microtendipes*, *Harnischia*, *Micropsectra*, *Pentaneura*, *Psilotanypus*) and several naid species (i.e. *Stylaria lacustris*, *Nais pardalis* and *Nais variabilis)*.

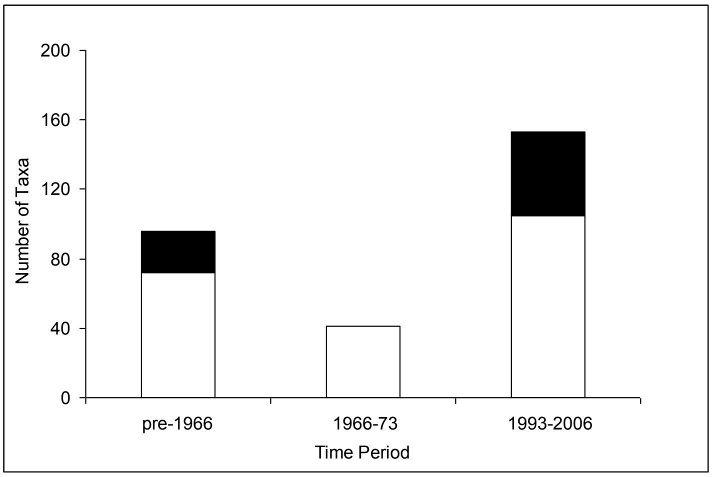


Figure 15: Total number of macroinvertebrate taxa recorded in studies of the littoral area of Loch Leven over time; the black shaded areas show the number of taxa that were unique to one time period, only (after Gunn et al., 2012).

1.5.3 Fish

Winfield et al. (2012) combined historical, published and unpublished information with contemporary gill net and hydroacoustic surveys to examine long-term changes in the fish community in Loch Leven. Observed changes were then interpreted within the context of historical and current anthropogenic pressures. The authors found that only 12 resident and migratory fish species had been recorded in Loch Leven and that the present day population consisted of brown trout (*Salmo trutta*), eel (*Anguilla anguilla*), minnow (*Phoxinus phoxinus*), perch (*Perca fluviatilis*), pike (*Esox lucius*) and three-spined stickleback (*Gasterosteus aculeatus*) (Figure 16), with brook lamprey (*Lampetra planeri*) and stone loach (*Barbatula barbatula*) also present in its tributaries. Arctic charr (*Salvelinus alpinus*), Atlantic salmon (*Salmo salar*) and flounder (*Platichthys flesus*) used to be found in the loch, but are now extinct. It is likely that the obstruction of migratory routes, when the sluices were installed on the outflow in the mid 1800s, caused the loss of Atlantic salmon and flounder, while the lowering of the water level at that time likely caused the extinction of Arctic charr.

The fish in the loch have been affected by anthropogenic influences beyond the pressures induced by increases and decreases in nutrient loading. Management activities in support of angling activity (Montgomery, 1994) have included fish stocking, which began as early 1882. In more recent years, various anthropogenic pressures (e.g. Duncan, 1994), the socio-economic vagaries of the angling community (e.g. Montgomery, 1994) and predation by cormorants (*Phalacrocorax carbo*) (e.g. Stewart et al., 2005) have also affected the fish community. Many of these influential factors have a long local history (May & Spears, 2012c), so an understanding of the contemporary fish community requires a correspondingly long-term perspective.

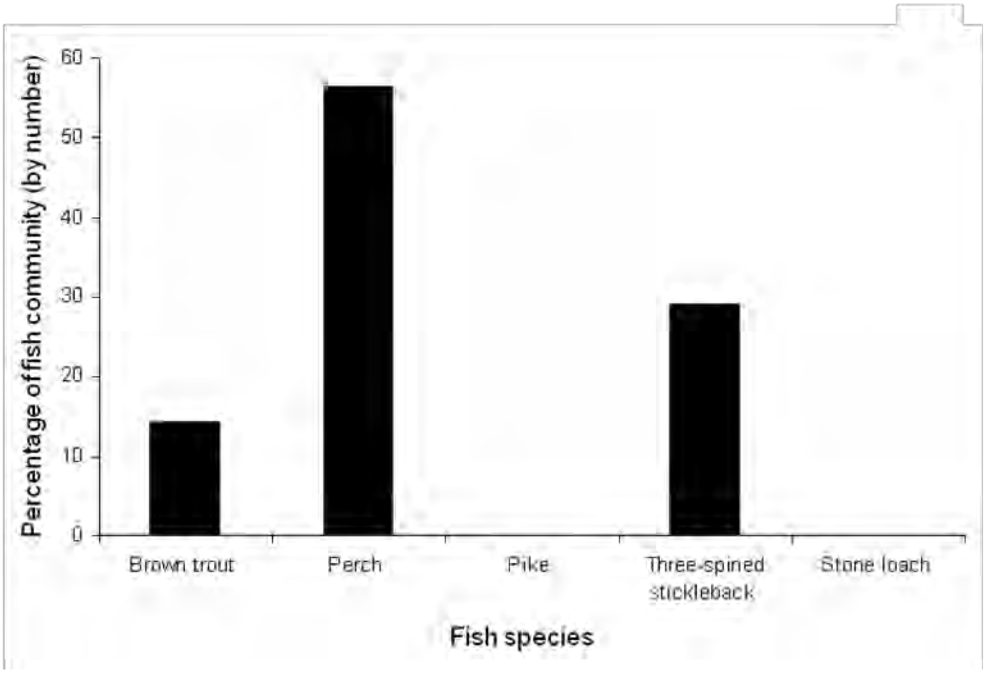


Figure 16: Composition of the contemporary fish community of Loch Leven based on sampling in March, August and October 2008 (n = 1,139). Although single pike and stone loach were also collected, they are not visible at this scale (after Winfield et al., 2012).

The long term data show that perch abundance has fluctuated markedly, being influenced by disease and eutrophication. Even allowing for the substantial natural variation in abundance of perch that is observed in relatively undisturbed habitats, such as that of the north basin of Windermere (Paxton et al., 2004), the decline at Loch Leven has been particularly marked. However, some recovery might now be expected as the loch's level of eutrophication reduces (Carvalho et al., 2012) and its macrophytes increase in abundance (Dudley et al., 2012; May & Carvalho, 2010). As adult perch consume the equivalent of over 6% of their body weight per day in summer (Thorpe, 1977), and young perch are probably a major planktivore within the loch, such population changes are likely to have important effects on water quality indicators in the loch. This is especially true of chlorophyll$_a$ concentrations, through impacts on zooplankton grazing community.

Although the brown trout population has undoubtedly shown a long-term decline, individuals are currently in excellent condition. Extensive sampling in 2008 recorded only five species in the loch, i.e. brown trout, perch, pike, three-spined stickleback and stone loach. In numerical terms, the community is now dominated by just three species, i.e. brown trout, perch and three-spined stickleback (Figure 16), with a total abundance of about 72.7 fish ha^{-1}.

The present Loch Leven fish community composition is also noteworthy in the context of absent species. Most, or all, of the 370,000 rainbow trout stocked into the loch between 1993 and 2004 have now either died or been removed and no local reproduction has been recorded. Also, although there is potential for nuisance fish species such as roach (*Rutilus rutilus*) and ruffe (*Gymnocephalus cernuus*) to be introduced into Loch Leven, with implications for the native fish community, the extensive sampling of 2008 did not record any such new arrivals.

1.5.4 Aquatic birds

Loch Leven has been designated as a UK Ramsar Site (1976), a Site of Special Scientific Interest (1985), and a Special Protection Area (2000) because of its importance as a site for overwintering waterfowl. However, until recently (Carss et al., 2012), no comprehensive assessment of trends in waterfowl populations, comparing local and national trends, had been conducted at the site.

Carss et al., (2012) assessed the effects of environmental change on waterfowl species at Loch Leven by identifying (1) trends in particular species at local and national scales, (2) discontinuities between local and national trends, and (3) drivers of change at the local site, using available water quality and climate change data (Favero & Becker, 2006). Coherence between trends in 5-year mean species abundance at Loch Leven and the Underhill Indexing Method values for Scotland (or GB in the case of geese) were assessed for ten study species between 1968 and 2006 (Table 5 -).

Of the ten species examined (Table 5 -), five showed trends at Loch Leven that were coherent, or broadly so, with those at the Scottish scale (Figure 17). These were two surface feeders (Eurasian Teal [*Anas crecca*] and Mute Swan [*Cygnus olor*]), one diver (the Great Cormorant [*Phalacrocorax carbo*]) and both species of goose (pink-footed [*Anser brachyrhynchus*] and greylag [*Anser anser*]). The other five species showed distinct differences between the local Loch Leven trends and those at the Scottish scale. These comprised a surface feeder (Mallard [*Anas platyrhynchos*]) and four of the five diving species (Coot [*Fulica atra*], Great Crested Grebe [*Podiceps cristatus*], Tufted Duck [*Aythya fuligula*], Pochard [*Aythya farina*]).

When local and national trends were similar, and Loch Leven was assumed to be merely ‘following’ a more geographically widespread trend, the likely causes of population changes were found to vary across local, national and international scales, and among types of bird. For example, changes in numbers were thought to be due to widespread climatic factors for Tufted Duck and Mute Swan, reduced local shooting pressure for Tufted Duck and Cormorant, changes in prey abundance or availability Cormorant, pink-footed and greylag geese, and more widespread changes in the availability of food due to changes in agricultural practices for pink-footed and greylag geese. It is likely that many of these factors are linked (e.g. changes in food availability and climate), so teasing apart key drivers is not a trivial task.

One of the aims of the Carss et al. (2012) study was to identify lack of coherence between local and national trends in waterfowl abundance, as a precursor to identifying the main drivers of numerical changes at Loch Leven and maximising the effectiveness of management intervention, there. Each of the five cases that showed a dissimilarity between Scottish (or GB) and local trends at Loch Leven were declining at the national level while remaining stable (Mallard, Great Crested Grebe) or increasing (Coot, Tufted Duck, Pochard) at Loch Leven. Furthermore, for each of these five species, pan-European breeding populations were showing declining trends (Burfield & van Bommel, 2004).

The Loch Leven wintering populations of both Tufted Duck and Pochard have continued to increase since the 1990s, suggesting that their relative conservation value at this site is increasing. This probably reflects the improvement in water quality and habitat availability that has occurred now that nutrient inputs have been reduced. However, it is not clear how local changes in the numbers of these birds relate to changes in local feeding conditions or reflect changes in the abundance of migrating birds. A study of their feeding ecology is needed to determine how changes in local water quality, habitat and food resources interact with macro-scale population dynamics to influence local and regional patterns of abundance.

Table 5 - Details of the ten waterfowl species included in the study of Carss et al. (2012). Information from Forrester & Andrews (2007) unless otherwise stated (e.g. [1]Bruun & Singer, 1978) (after Carss et al., 2012).

Species	**Predominant feeding habit**	**Body size[1]**	**Predoninant diet**	**Present all year?**	**Mainly winter visitor?**	**Loch Leven a main site?**
Eurasian Teal	Surface	Small	Omnivorous		Yes	
Mallard	Surface	Medium	Onmivorous	Yes		
Mute Swan	Surface	Large	Herbivorous	Yes		
Coot	Shallow diver	Small	Herbivorous	Yes		Yes
Great crested grebe	Diver	Medium	Fish-eater	Yes		Yes
Great cormorant	Diver	Large	Fish-eater		Yes	
Tufted duck	Benthic diver	Medium	Invertebrates	Yes		Yes
Common Pochard	Benthic diver	Medium	Onmivorous	Yes		
Pink-footed goose	Land	Large	Herbivorous		Yes	Yes
Greylag goose	Land	Large	Herbivorous		Yes	Yes

1.6 Future impacts on phytoplankton and water quality

Elliott et al. (2012) modelled the phytoplankton community of Loch Leven in 2005 and simulated the response of the algal community to a combination of different flushing rates and water temperatures to assess sensitivity to future climate change. Whilst the annual mean total chlorophyll$_a$ concentrations proved relatively insensitive to these changes, there were marked changes at the community composition level. Some responded positively to increased temperature (e.g. *Aulacoseira*) and some negatively (e.g. *Asterionella*), others were enhanced by increased flow (e.g. *Stephanodiscus*) while others were negatively affected (e.g. *Aphanocapsa*). However, the relationship with flow was seasonally dependent. For example, a simulated increase in drier seasons benefitted some species by increasing the supply of nutrients, while an equivalent increase in flow in wetter seasons affected some species negatively by increasing flushing losses. Overall, the simulations showed that the range of species within the community was sufficient for at least one to benefit from the changing niches created by the combinations of climatic drivers applied in this study. The level of exploitation by such a species was only constrained by the nutrient carrying capacity of the system and it is this that led to the dampened response in total chlorophyll$_a$ at the annual and seasonal scale. Thus, whilst overall biomass showed relatively little reaction to the climatic drivers tested, the phytoplankton community composition showed a marked response.

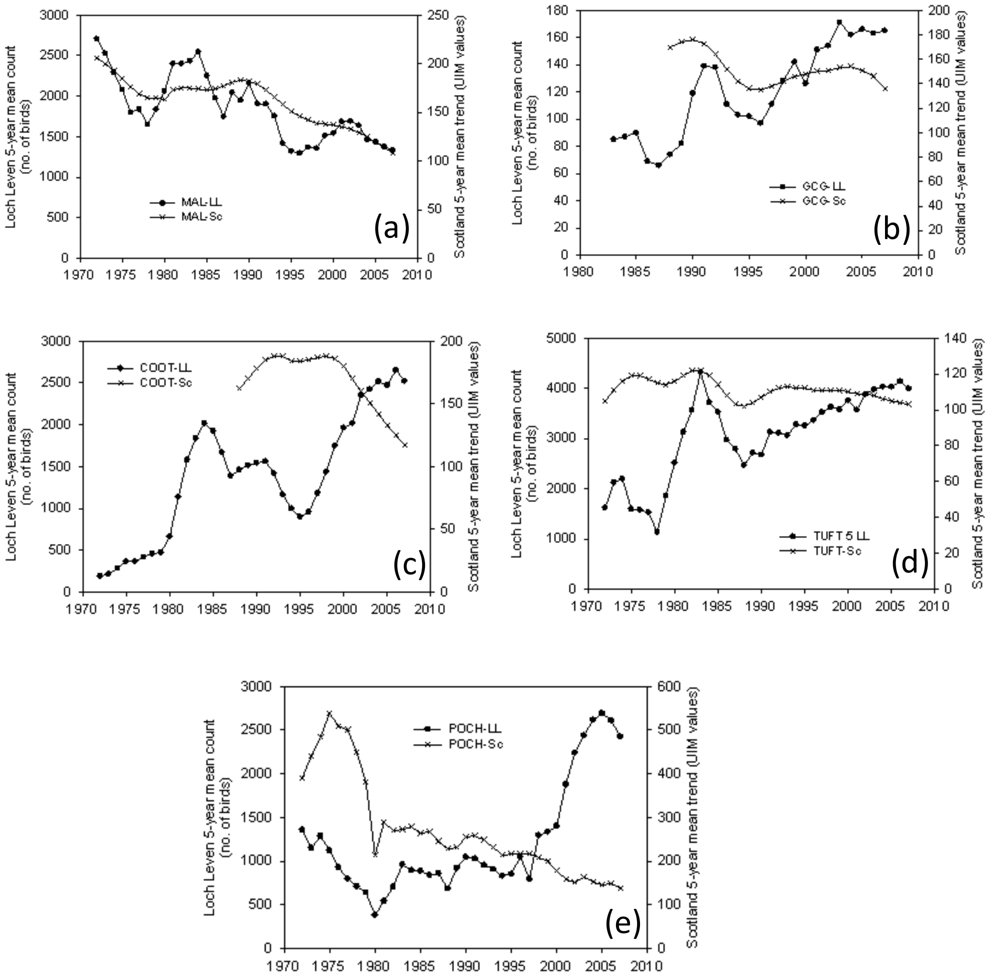


Figure 17: Comparison of 5-year mean winter trends for (a) Mallard (MAL), (b) Great Crested Grebe (GCG), (c) Coot (COOT), (d) Tufted Duck (TUFT), (e) Pochard (POCH), at Loch Leven and across Scotland. UIM = Underhill Indexing Method (after Carss et al., 2012).

2 METHODS

This section describes methods used for collection of samples in the field and subsequent laboratory analyses for the period 2008-2010. All sampling and analytical methods were consistent with the ongoing Loch Leven long-term monitoring project and are, therefore, compatible with previous reports from this project (Carvalho & Kirika, 2005; Carvalho et al., 2007; 2008).

2.1 Field surveys

Field surveys were conducted by boat at approximately fortnightly intervals, with some exceptions during the winter months. On each survey, water samples were collected at two sites, Reed Bower and the Sluices (Figure 18). When particularly bad weather, or ice cover, prevented safe use of a boat, samples were taken from the shore close to the outflow of the lake, very close to the Sluices open water site. Occasionally, all sampling was prevented by bad weather, when roads were not safe for travel. Table 6 shows the number of surveys undertaken in the period 2008-2010.

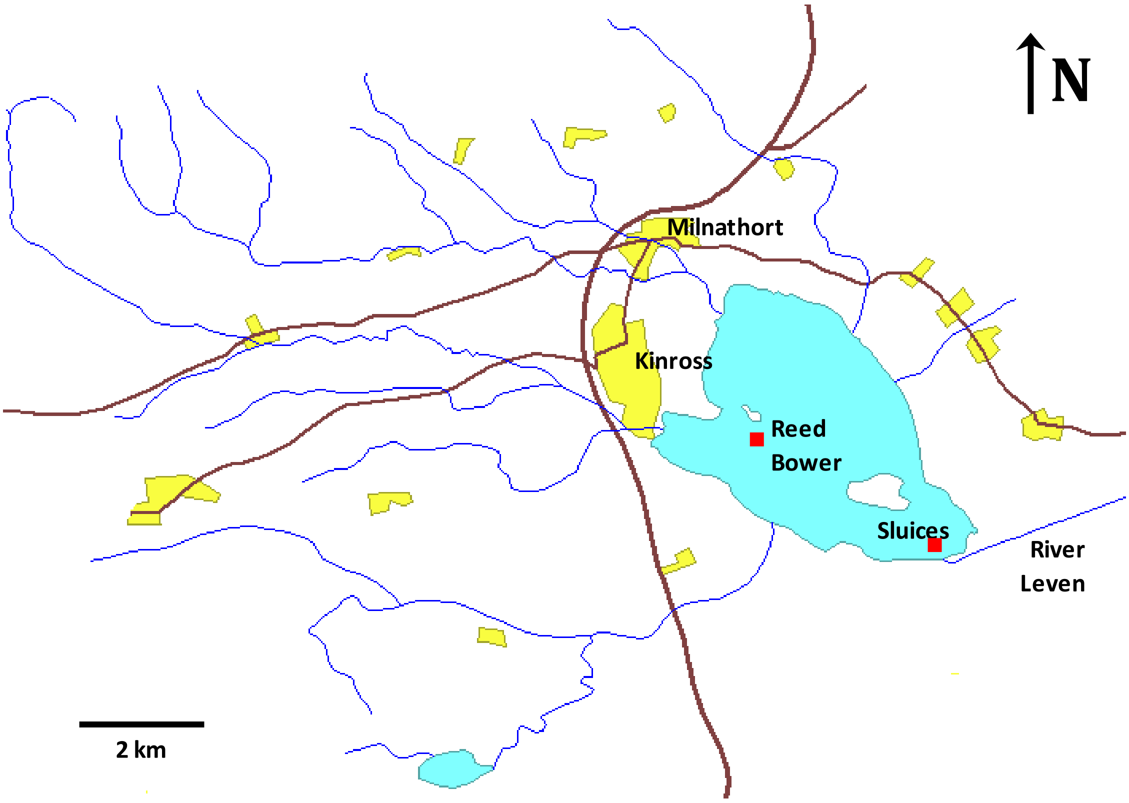


Figure 18: Map of Loch Leven, showing positions of routine sampling sites (red squares), inflow and outflow (River Leven) streams (blue), major roads (brown) and urban areas (yellow), including Kinross and Milnathort. Map data from Ordnance Survey Strategi dataset. Contains Ordnance Survey data © Crown copyright and database right 2012.

Table 6 - Sampling undertaken at Loch Leven in 2008, 2009 and 2010.

Year	Boat sampling	Shore sampling
2008	22	1
2009	23	3
2010	20	5

On each sampling occasion, duplicate 2 L water samples were collected from each site. At the Reed Bower site, an integrated sample was collected using a PVC tube with an internal diameter of 25 mm, which was weighted at one end. This tube was lowered into the water until the weighted end was within 0.5 m of the lake bottom, filling as it was lowered. The weighted end was then lifted out of the lake while ensuring that the other end of the tube remained stationary. This ensured that the sample was collected from the entire water column. At the Sluices site, water was collected using a bottle that was held 10 cm below the surface.

Water clarity was measured as Secchi depth, i.e. the depth to which a Secchi disc (a disc of 25 cm diameter, divided into quadrants, which are alternately black and white) can be lowered before it is no longer visible. This was measured at the Reed Bower site. Water level was measured against a reference point within the harbour.

Water temperature was measured using electronic probes (Hach-Lange) at both sites. Averages of all measurements on a particular day were calculated for presentation in this report.

On return to the laboratory, samples were shaken and sub-samples of filtered (Whatman® GF/C, within 6 hours of collection) or unfiltered water were taken from each duplicate site sample for soluble reactive phosphorus (SRP), total soluble phosphorus (TSP), nitrate (filtered and stored frozen), soluble reactive silicate (filtered and stored in fridge), total diatom silica (unfiltered and stored in fridge) and total phosphorus (TP; unfiltered and frozen) analyses. Two sub-samples of unfiltered water were taken for phytoplankton analysis and stored in either 4% formaldehyde or Lugol's iodine solution. Samples for chlorophyll$_a$ analysis were prepared for each site by filtering 400 ml of lake water through a GF/C grade filter. The filter was stored (frozen) in a 15 ml centrifuge tube until analysis.

2.2 Rainfall data

UK Meteorological Office rainfall data were obtained from the British Atmospheric Data Centre website. As there was no single rain gauge for which data were available for the entire period of interest, data from all gauges in the Fife area were used to create an average daily rainfall. These daily rainfall values were then summed over each month to create total monthly rainfall values.

2.3 Chemical analyses

2.3.1 Phosphorus determination

Tubes of filtered water were defrosted, sub-sampled and analysed for SRP and TSP concentrations, and those containing unfiltered water were analysed for TP.

SRP concentrations were determined following the method of Murphy & Riley (1962). This method uses a reagent of ammonium molybdate, potassium antimony tartrate, and L-ascorbic acid in 1 M sulphuric acid, which reacts with the phosphate ion to form a phospho-molybdenum blue complex. Concentration was determined by measuring the absorption of red light (882 nm) and comparing this against known standards.

TP concentrations were determined on unfiltered samples, which were digested using a solution of sulphuric acid and potassium persulphate that converted all forms of phosphorus to SRP. This was then measured in a similar way to that described above. The method used was as described for TP by Wetzel and Likens (2000), with an added acidification step (0.1 ml of 30% H_2SO_4 was added to the samples before addition of persulfate). TSP was determined in the same way as described for TP, but using a filtered sample.

2.3.2 Nitrogen determination

Filtered water samples were defrosted and analysed for nitrate (nitrate-N) content on a SEAL AQ2 analyser (SEAL Analytical Limited, Burgess Hill, West Sussex, UK). Nitrate was determined by the sulphanilamide/NEDD (N-1-naphthylethyene diamine dihydrochloride) reaction, which produces a reddish-purple dye. This was measured spectrophotometrically at 546 nm.

2.3.3 Silica determination

Filtered water samples were analysed for soluble reactive silicate ($SRSiO_2$) according to the method of Golterman et al. (1978). The optical absorbance of the sample solutions at a wavelength of 810 nm was measured directly against that obtained for known standard solutions of SiO_2.

2.4 Chlorophyll$_a$ analysis

Frozen filters were submersed in 90% methanol overnight in a dark fridge. The following day, the tubes were centrifuged for 10 minutes at 2500 r.p.m. Chlorophyll$_a$ in the extract was measured spectrophotometrically at 665 nm with a turbidity correction conducted at 750 nm. The concentration of chlorophyll$_a$ was determined using equation 1 (APHA, 1992).

$$[\text{chlorophyll}_a]\ \mu g\ L^{-1} = (O.D._{665} - O.D._{750}) \times \{(13.9 \times v)/(V \times L_p)\} \qquad \text{(eq. 1)}$$

where:

$O.D._{665}$ = optical density (absorbance) at 665 nm, a distinctive peak for chlorophyll$_a$
$O.D._{750}$ = optical density at 750 nm, a correction for any background turbidity
v = the volume of the extract in millilitres (e.g. 11.5 ml)
V = the volume of water filtered in litres
L_p = the path-length of the cuvettes used in cm (e.g. 4 cm)
13.9 = an absorption coefficient (a constant) for chlorophyll$_a$

2.5 Biological analyses

2.5.1 Phytoplankton

Phytoplankton was sub-sampled from the integrated water sample collected from Reed Bower. Phytoplankton counting procedures followed UK standard guidance (Brierley & Carvalho, 2007) with taxonomy following John et al. (2003). Separate sub-samples were stored in formaldehyde and Lugol's iodine solution. Samples were concentrated by settling before enumeration.

2.5.2 Crustacean zooplankton

Open water crustacean zooplankton samples were collected, and concentrated, at the Reed Bower site, by drawing a plankton net (mesh size 120 µm, net mouth diameter 20 cm) slowly to the water's surface along a 4 m angled net tow. At the Sluices site, 30 L sub-surface samples were collected with a bucket and concentrated by passing the sample through the plankton net. All samples were preserved in 4% formaldehyde solution.

In the laboratory, each sample was placed in a glass vessel and made up to a final volume of 250 ml with distilled water. The sample was thoroughly mixed, to distribute the animals randomly, and then sub-sampled with a Stempel pipette (volume 5 ml). The animals present in each subsample were identified (Dussart and Defaye 1995; Einsle 1996; Flößner and Kraus, 1986; Harding and Smith 1974; Scourfield and Harding 1966) and counted under a low power binocular microscope. In all cases, three sub-samples were examined. The preserved freshwater crustacean zooplankton were identified to species level wherever possible. No specimen was identified beyond the level justified by its condition of preservation or stage of maturity (as recommended in the appropriate key). The sub-sample counts were converted to numbers of individuals per litre (ind. L^{-1}) using appropriate multiplication factors. After counting, the samples were reconstituted and archived for potential future analysis.

3 RESULTS

3.1 Physical factors

3.1.1 Water temperature

As surface water temperatures were similar at the two sites (Reed Bower and Sluices) throughout the study period, averages of both sites are presented here for simplicity. Surface water temperature varied from near freezing in the winter to just below 20 °C in the summer (Figure 19). There were some minor differences apparent among the three years of data, particularly the evidence of a longer winter in 2010.

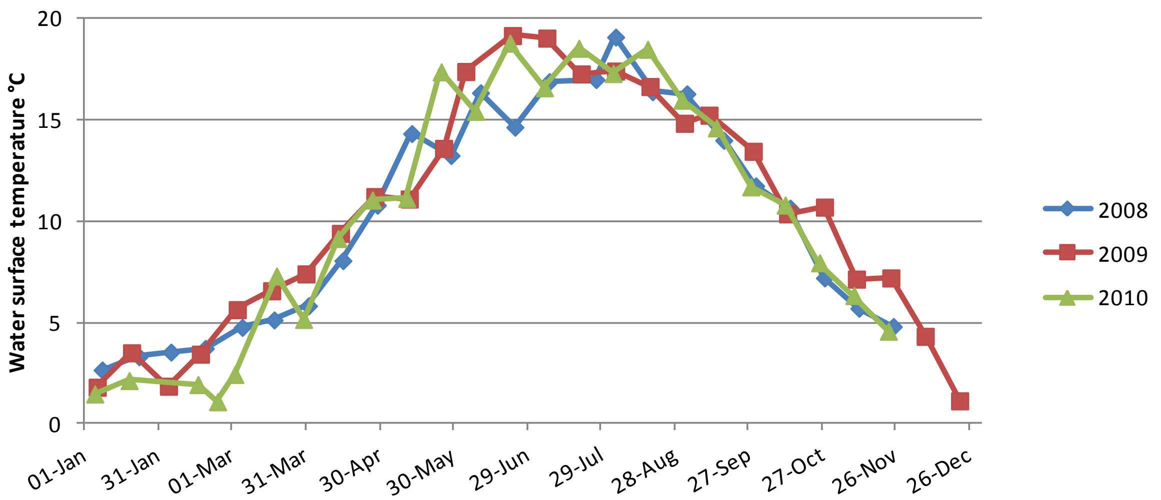


Figure 19: Surface water temperature in Loch Leven from 2008-2010. Values shown are the average of temperatures taken at the Reed Bower and Outflow sites.

3.1.2 Water clarity (Secchi depth)

Secchi depth at the Reed Bower site was generally between 1 and 2 m, except during May and June when the water cleared in all three years. Between 2008 to 2010 the spring peak in Secchi depth both increased and advanced, from 3.1 m on 8th July 2008 to 3.9 m on 4th June 2009, then 4.8 m on 25th May 2010 (Figure 20). In 2010, as well as the Spring clear-water phase, the water cleared again in September (3.4 m) and late November (3.7 m). The annual average (average of monthly averages) improved in each successive year, from 1.6 m in 2008, to 2.3 m in 2010 (Table 7).

Table 7 - Annual means of Secchi depth, total phosphorus (TP), soluble reactive phosphorus (SRP), nitrate, soluble reactive silica (SRSi) and chlorophyll$_a$ (Chl$_a$) concentrations, as averages of monthly averages, as measured in Loch Leven, from 2008-2010.

Year	Secchi (m)	TP (µg L^{-1})	SRP (µg L^{-1})	Nitrate (mg L^{-1})	SRSi (mg L^{-1})	Chl$_a$ (µg L^{-1})
2008	1.6	32	5.4	0.92	2.2	24
2009	1.8	36	8.5	0.68	4.2	15
2010	2.3	31	5.4	0.63	2.0	24

3.1.3 Water level

As has been the observed pattern previously (Carvalho & Kirika, 2005; Carvalho et al., 2007; 2008), in all years water level increased during the winter months, then declined in Spring and Summer (Figure 21). This mainly reflects management practices associated with the opening and closing of the sluice gates to provide water to downstream users (May & Carvalho, 2010).

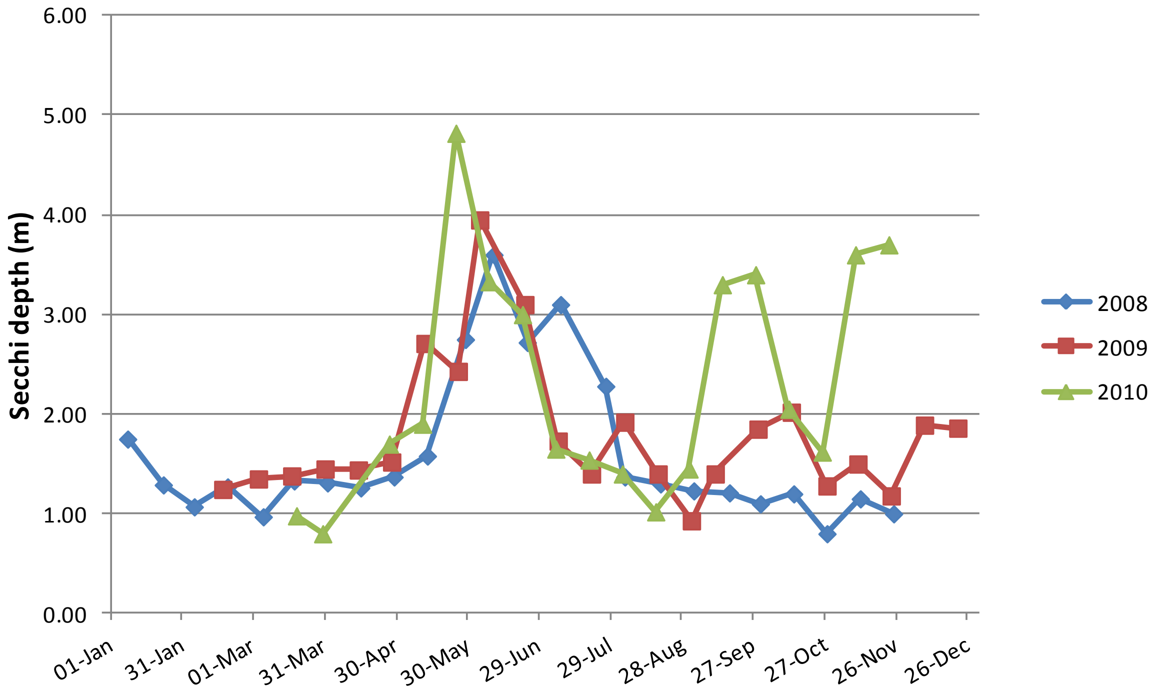


Figure 20: Water clarity, measured as Secchi depth (m) at the Reed Bower site of Loch Leven from 2008-2010.

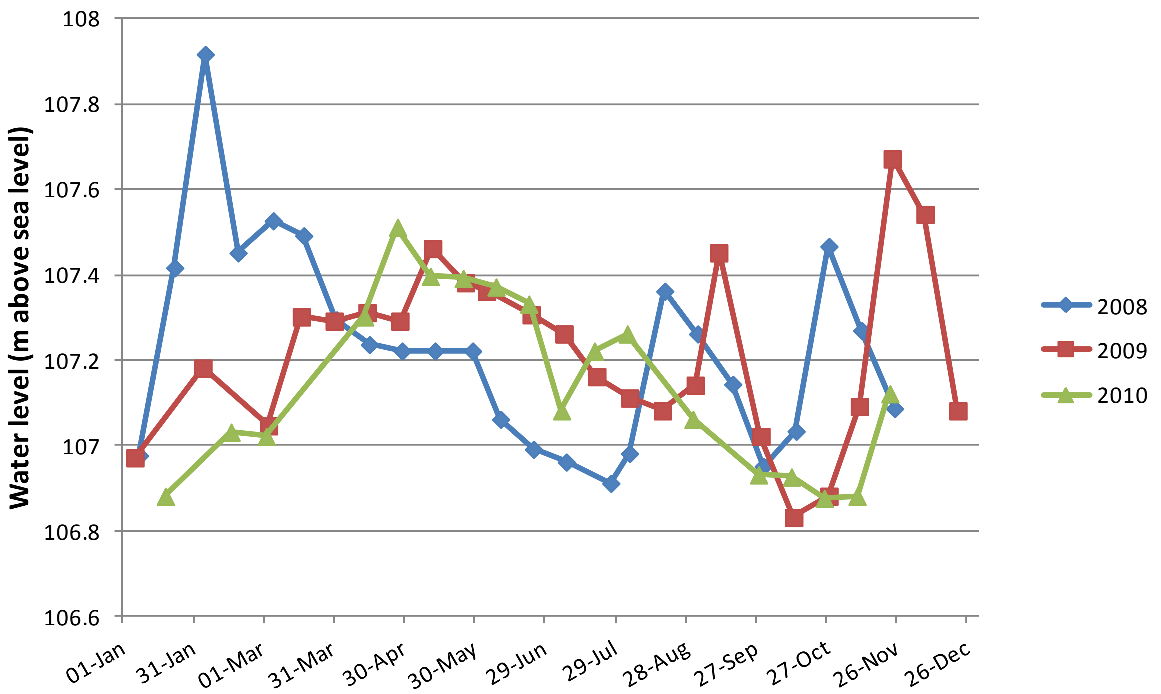


Figure 21: Water level, in metres above sea level, as measured in Loch Leven Harbour.

3.1.4 Rainfall

Monthly total rainfall ranged from 29 mm (May 2008) to 213 mm (August 2008). Monthly rainfall patterns were highly variable among years, though roughly consistent with the 11-year average for each month (Figure 22).

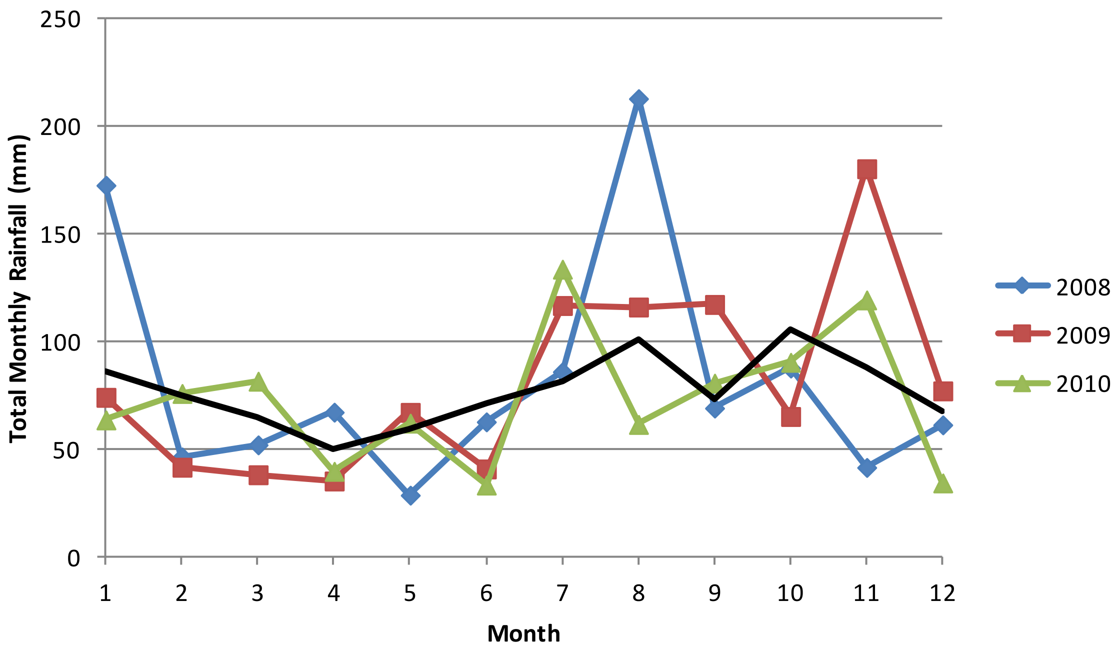


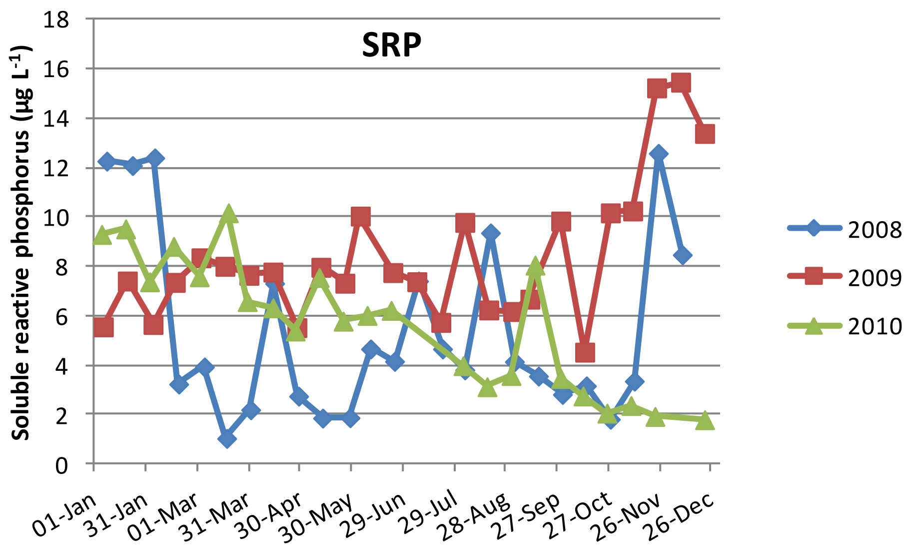
Figure 22: Total monthly rainfall during 2008-2010, averaged across all rain gauge data available from Fife (see Methods). The black line shows the total monthly rainfall for the same stations averaged across 11 years (2000-2011).

3.2 Chemical factors

3.2.1 Phosphorus concentrations

Monthly averages of soluble reactive phosphorus (SRP) concentrations were between 1.8 and 14.5 μg L^{-1} (Figure 23). The annual average (i.e. the average of all monthly averages, for each year) was slightly higher in 2009 (8.5 μg L^{-1}) than in 2008 or 2010 (both 5.4 μg L^{-1}). There were no clear seasonal patterns in SRP over the three year period.

Monthly average total phosphorus (TP) concentrations were between 20 and 63 μg L^{-1} (Figure 23). In each year from 2008 to 2010, TP was generally at a minimum in Spring, corresponding to the peak in water clarity, and a peak in concentration was evident in Summer. Concentrations were generally similar at the two sites, except on 30/03/2010 when the average TP at the Reed Bower site (106 μg L^{-1}) was more than double that at the Outflow (42 μg L^{-1}).



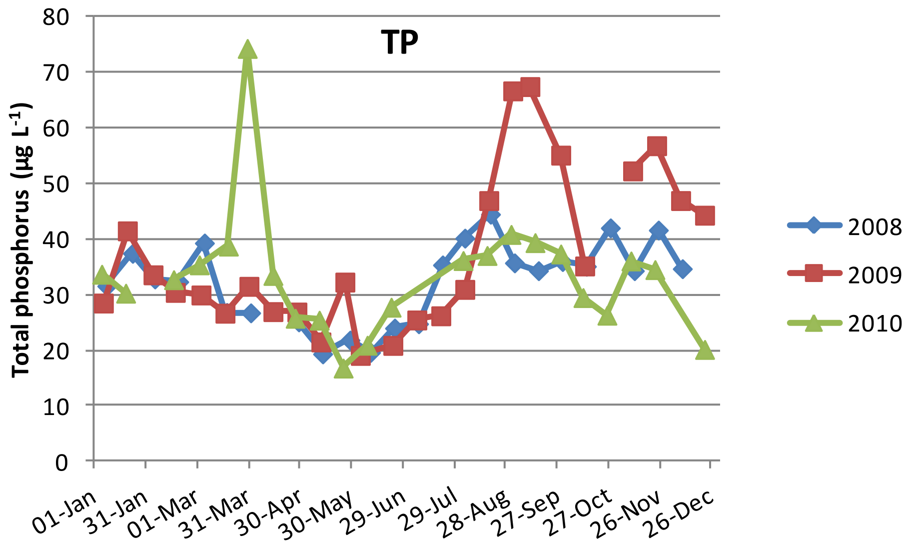


Figure 23: Concentrations of Soluble Reactive Phosphorus (SRP, top) and Total Phosphorus (TP, bottom) in Loch Leven from 2008-2010. Values are averages of both Outflow (L) and Reed Bower (RB) sites.

3.2.2 Nitrate concentration

During 2008-2010, nitrate concentration was very low in summer (many samples had less than the detection limit of 0.01 mgN L^{-1}), and relatively high in Winter (2.0 mgN L^{-1} in 2008 and 1.5 mgN L^{-1} in both 2009 and 2010) (Figure 24).

3.2.3 Silica concentrations

Concentrations of Soluble Reactive Silica varied between 8.7 and 0.08 mg L^{-1} at the Reed Bower site (Figure 25). Highest concentrations were recorded in the winters of 2007-2008 and 2009-2010. Lower concentrations were generally measured during summer, though in 2009, concentrations were substantially higher than in the other two years.

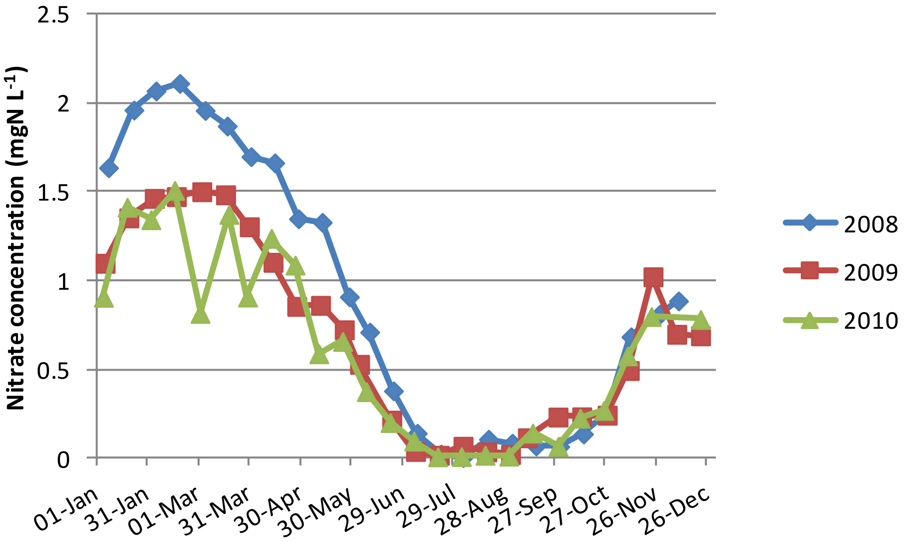


Figure 24: Concentrations of Nitrate in Loch Leven from 2008-2010. Values are averages from the lake Outflow and Reed Bower sites.

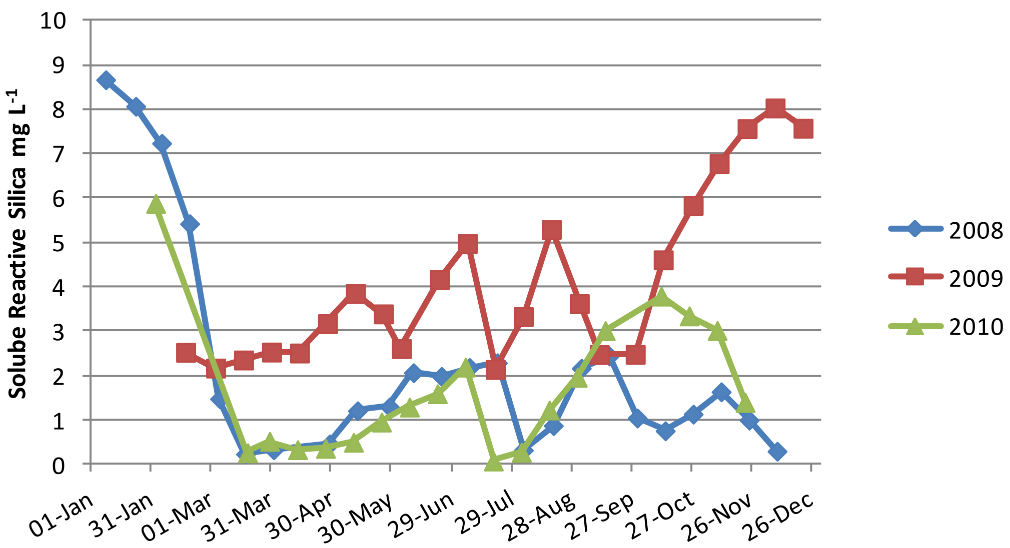


Figure 25: Concentrations (mgSi L^{-1}) of Soluble Reactive Silica at the Reed Bower (RB) site of Loch Leven from 2008-2010.

3.3 Chlorophyll$_a$ concentration

With the exception of May and June each year, chlorophyll$_a$ concentrations frequently exceeded 20 µg L^{-1} (Figure 26). Annual mean concentrations for 2008 and 2010 of 24 and 25 µg L^{-1}, respectively, were higher than the target mean annual chlorophyll concentration of 15 µg L^{-1} set by the Loch Leven Area Management Advisory Group (LLAMAG 1993). This target was, however, just met in 2009 when the annual mean was 15 µg L^{-1}. This value would also place Loch Leven in the moderate ecological status class, using site-specific WFD G/M and M/P targets of 11 µg L^{-1} and 22 µg L^{-1} respectively (Carvalho et al., 2009).

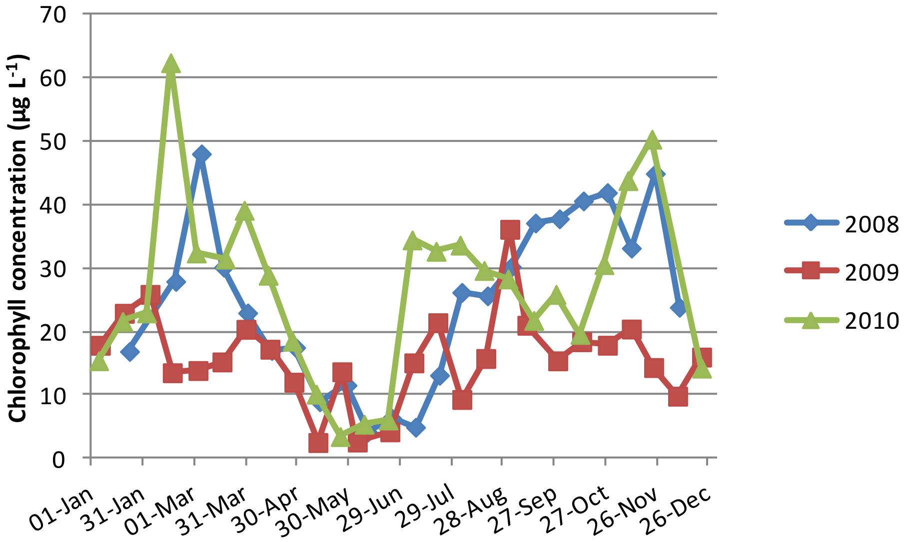


Figure 26: Average monthly chlorophyll$_a$ concentrations in Loch Leven from 2008-2010. Values are averages from the Outflow and Reed Bower sites.

3.4 Biological factors

3.4.1 Phytoplankton

In all three years, the phytoplankton community of Loch Leven was largely dominated by diatoms, although cyanobacteria dominated in August 2009 (Figure 27). There was a very large spring bloom of diatoms in March 2008. This resulted in a sharp decline in SRSi concentrations (Figure 25), followed by a crash in diatoms by April 2008. Low total phytoplankton biovolume was sustained in May and June, the typical clear-water period in Loch Leven, when *Daphnia* densities reach their peak (Figure 30). In 2008, the late summer community was unusually dominated by diatoms. Cyanobacteria were never dominant during the summer and autumn of 2008, their numbers remaining below the WHO (1999) low risk threshold for recreational waters of 2 mm^3 L^{-1} (Figure 29). Diatom populations remained dominant for the rest of 2008 and reached a peak in January 2009 despite low $SRSiO_2$ concentrations in Dec 2008. Total phytoplankton biovolume declined through the spring of 2009, possibly due to low availability of SRP and SRSi (Figure 23 and Figure 25). A short period of dominance by a fine filamentous cyanobacterium, *Oscillatoria limnetica*, occurred in April 2009; this may have resulted from wind resuspension of benthic populations of this species. This was followed by a clear-water phase in June with low phytoplankton biovolume that was, again, correlated with the annual peak in *Daphnia* densities. Diatoms returned to dominance in July 2009 but, by August, the typical cyanobacteria peak occurred, with densities remaining above WHO (1999) low risk thresholds throughout September, too (Figure 28 and Figure 29). In summer 2008 and 2009, and Feb 2010, *Anabaena* species were observed to have specialised cells (heterocysts) that enable them to fix nitrogen and continue to grow when other species of algae may have become nitrogen-limited (Figure 24). Diatoms returned to dominance in September 2009, but their numbers had crashed by October and remained very low for the rest of the year, despite increasing concentrations of all nutrients, including SRSi. By January 2010, diatom populations had started to increase again (Figure 27), peaking in March then declining to very low levels in May & June, associated with the *Daphnia* peak in 2010 (Figure 37). The late summer and autumn were dominated by diatoms again, but more unusually, there was very little development of cyanobacteria (Figure 27); these remained well below the WHO (1999) low risk threshold throughout (Figure 28). The complete dominance by diatoms in 2010 may be an indicator of improving quality; rainfall was not particularly high, so the lack of cyanobacteria did not appear to be a response to a particularly high flushing rate.

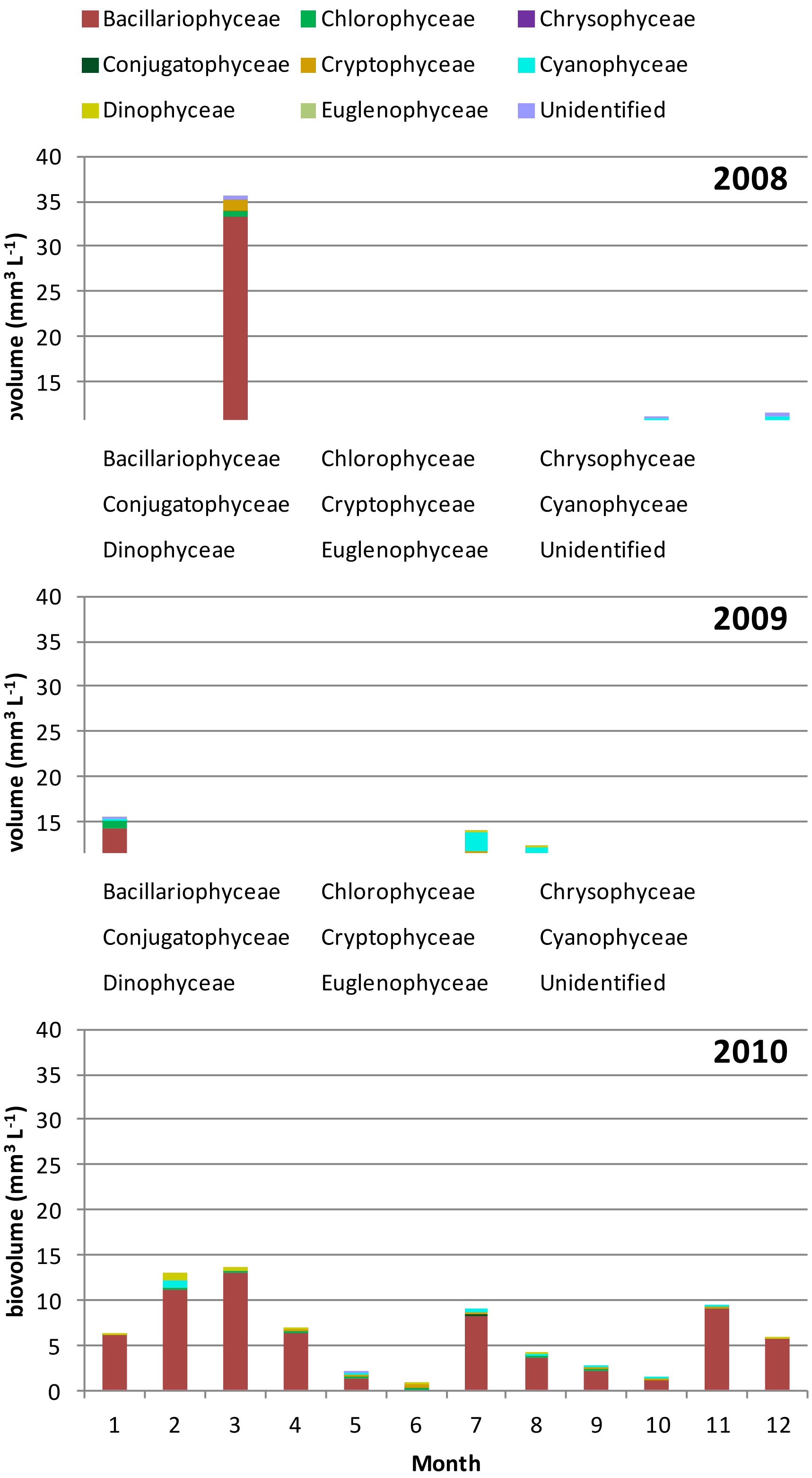


Figure 27: Biovolumes of phytoplankton classes during the period 2008-2010; data are from samples taken from the Loch Leven Reed Bower site, except for January and February 2010, when samples from the Sluices site were used.

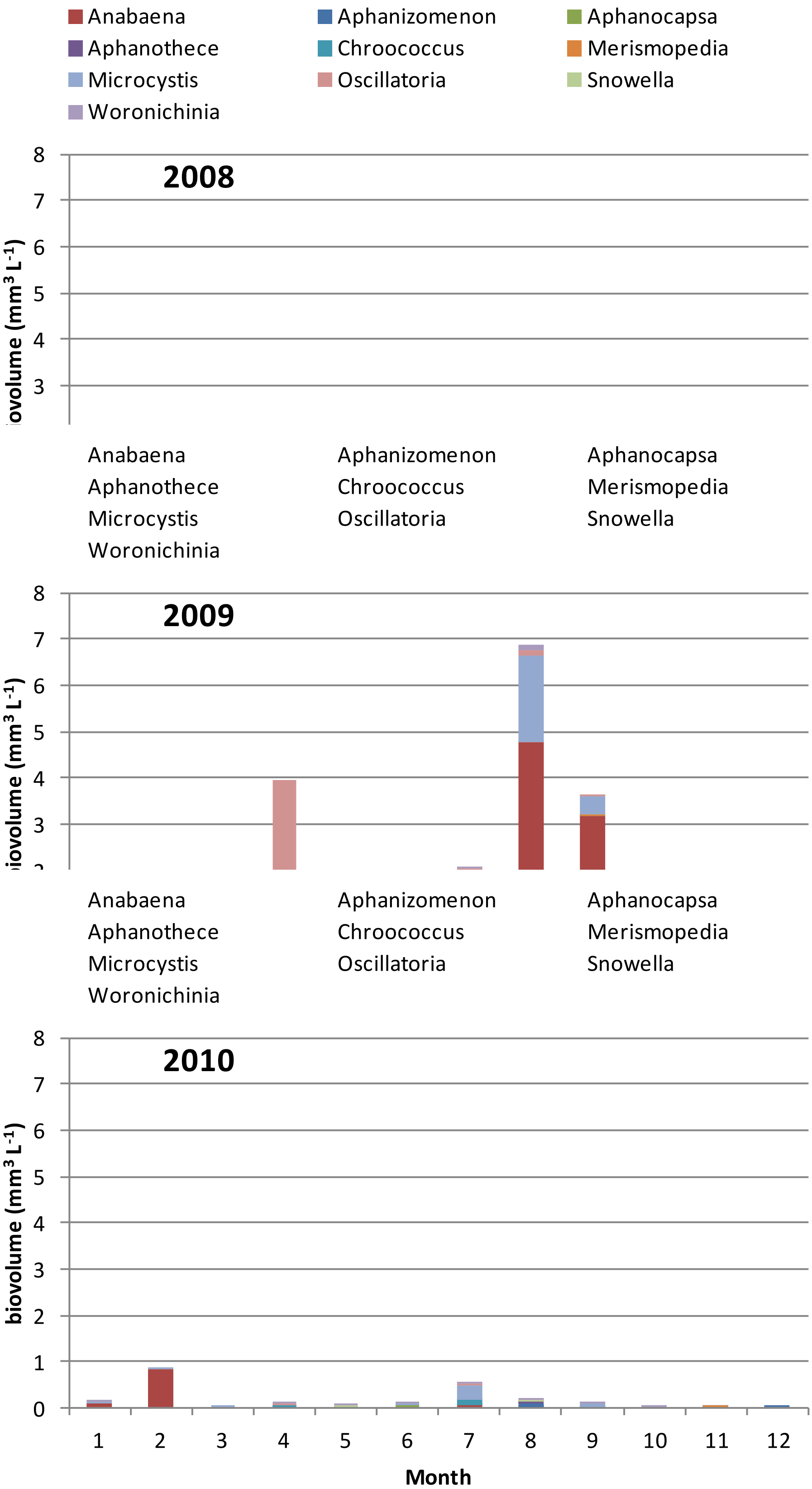


Figure 28: Biovolumes of cyanobacteria genera during the period 2008-2010; data are from samples taken from the Loch Leven Reed Bower site, except for January and February 2010, when samples from the Sluices site were used.

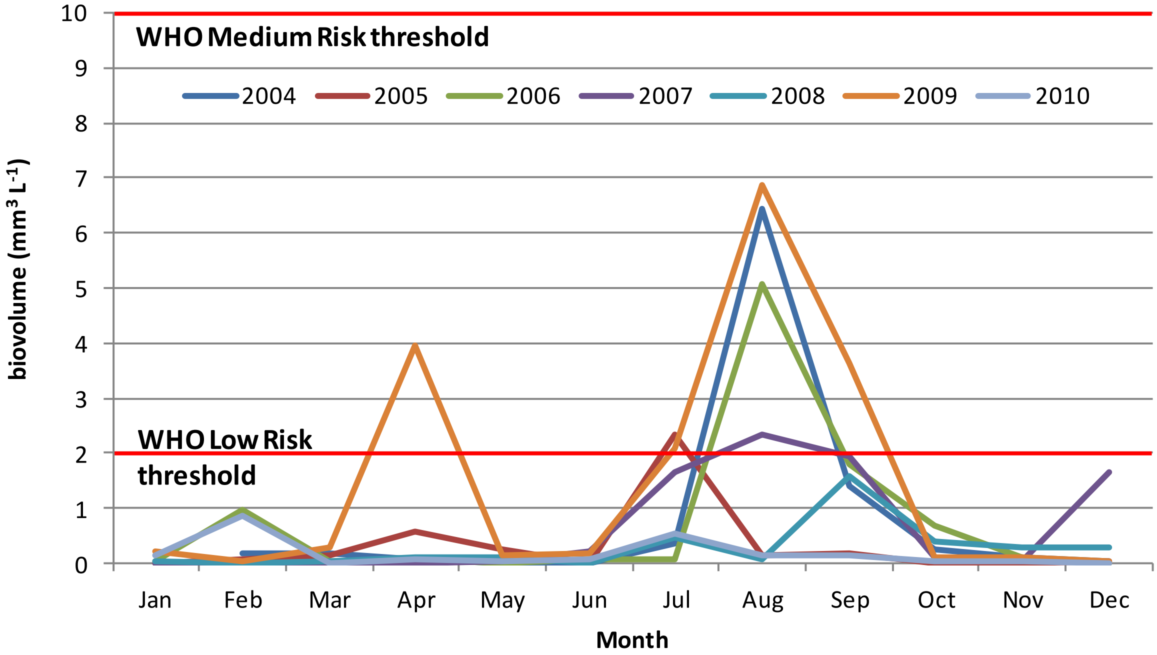


Figure 29: Monthly total cyanobacterial biovolume during the period 2008-2010; data are from samples taken from the Loch Leven Reed Bower site, except for January and February 2010, when samples from the Sluices site were used.

3.4.2 Crustacean zooplankton

Seven crustacean zooplankton species and occasional miscellaneous specimens of Chydoridae were recorded from the Reed Bower and Sluices samples collected during 2008-2010 (Table 8). Temporal variation in the main crustacean zooplankton taxa is indicated by their monthly mean densities (Figure 30). The principal taxa were the cladoceran referred to as the *Daphnia hyalina* species-complex (cf. May et al. 1993; Gunn et al. 1994), the cyclopoid copepod *Cyclops* spp. (both *abyssorum* & *vicinus* species), the calanoid copepod *Eudiaptomus gracilis* and the cladoceran *Bosmina longirostris*.

The general seasonal features of the population dynamics of each of the main crustacean zooplankton taxa over the period 2008-2010 can be summarised as follows. In 2008, the principal crustacean zooplankton taxa recorded were *Daphnia*, *Cyclops* and *Eudiaptomus*. Mean monthly densities of *Daphnia* were very low (<2 ind. L^{-1}) during the first four months of the year, followed by a rapid increase in numbers to a peak of 59.95 ind. L^{-1} in May. As is common at Loch Leven, *Daphnia* numbers then declined to relatively low densities over the summer months. From the end of October 2008 onwards, *Daphnia* numbers remained at over-wintering levels of less than 1 ind. L^{-1}. In 2008, unlike 2007, the Cyclopod copepod population was dominated by *Cyclops abyssorum*, although *Cyclops vicinus* was recorded in low numbers in the early and latter parts of the year. The concentrations of copepodites and adults of *Cyclops* were low (<6 ind. L^{-1}) in the first three months of the year before increasing to a peak of 49.3 ind. L^{-1} in May. From July onwards *Cyclops* monthly mean densities remained consistently low (<5 ind. L^{-1}). *Eudiaptomus* was consistently present in the plankton but mean population densities remained low (<5 ind. L^{-1}) throughout the year. Maximum annual population densities of *Daphnia* (111.3 ind. L^{-1}), *Cyclops* 68 ind.L^{-1}) and *Eudiaptomus* (7.6 ind.L^{-1}) were all recorded at the Reed Bower sample site on the 29 May 2008. *Bosmina longirostris* was only recorded on three sampling visits, always in very low numbers (<1 ind. L^{-1}) during 2008. As is usual, the predatory cladocerans *Leptodora kindti* and *Bythotrophes longimanus* occurred in extremely low numbers (<0.2 ind. L^{-1}) over the summer months during 2008. Chydoridae were also occasionally recorded in the zooplankton samples in 2008.

Table 8 - Crustacean zooplankton species recorded in Loch Leven during 2008-2010.

Cladocera (Branchiopoda)

Anompoda
Bosmina longirostris (O.F. Müller)
Daphnia hyalina species-complex (formerly *D. hyalina var.lacustris* Sars)
Chydoridae
Halopoda
Leptodora kindti (Focke)
Onychopoda
Bythotrophes longimanus Leydig

Copepoda

Calanoida
Eudiaptomus gracilis (Sars) (formerly *Diaptomus gracilis* Sars)
Cyclopoida
Cyclops abyssorum Sars (formerly *Cyclops strenuous abyssorum* Sars)
Cyclops vicinus Uljanin

The crustacean zooplankton recorded at the Sluices sampling site in 2008 followed a generally similar pattern, in terms of species composition and temporal variation, to that of the Reed Bower sampling site apart from *Daphnia* numbers peaking approximately two months later. *Cyclops* dominated for the first four months before *Daphnia* numbers increased in May reaching a mean monthly density of about 101 ind. L^{-1} in August, mainly due to 200 ind. L^{-1} being recorded in the Sluices sample on the 4 August 2008. From September onwards, crustacean zooplankton numbers were very low at the Sluices, with *Cyclops* and *Eudiaptomus* predominating.

In 2009, the principal crustacean zooplankton taxa recorded were *Daphnia, Cyclops, Bosmina* and *Eudiaptomus.* Mean monthly densities of *Daphnia* were very low (<2 ind. L^{-1}) during the first three and half months of the year, followed by an increase to 35.6 ind. L^{-1} in May. Unusually, *Daphnia* numbers then remained high throughout June before increasing to an exceptionally high mean monthly density of 97 ind. L^{-1} in July. A maximum annual population density of *Daphnia* (124 ind. L^{-1}) was recorded at the Reed Bower sampling site on the 21 July 2009. *Daphnia* numbers then crashed in August and remained at over-wintering levels of less than 1 ind. L^{-1} from October onwards. In 2009, unlike 2008, the Cyclopoid copepod population was dominated by *Cyclops vicinus*, although *Cyclops abyssorum* was recorded in low numbers in the summer months and in December. The concentrations of copepodites and adults of *Cyclops* gradually increased to an annual peak of 27.4 ind. L^{-1} in April. A maximum annual population density of *Cyclops* (31.4 ind. L^{-1}) was recorded at the Reed Bower sampling site on the 14 April 2009. Throughout the summer months, *Cyclops* monthly mean densities remained consistently low (<2 ind. L^{-1}) before increasing again in October and November (<10 ind. L^{-1}). *Eudiaptomus* was again consistently present in the plankton, but with mean population densities remaining low (<5 ind. L^{-1}) throughout the year. The maximum annual population density of *Eudiaptomus* recorded (4.8 ind. L^{-1}) was at Reed Bower on the 13 October 2009. In contrast to 2008, *Bosmina longirostris* was relatively common in the plankton throughout 2009 with a mean monthly density of 26.5 ind. L^{-1} in September. A maximum annual population density of *Bosmina* (72.3 ind. L^{-1}) was recorded at Reed Bower on 29 October 2009. The predatory cladocerans, *Leptodora kindti* and *Bythotrophes longimanus*, occurred in extremely low numbers (<0.2 ind. L^{-1}) over the summer months during 2009. Chydoridae were also occasionally recorded in the zooplankton samples in 2009.

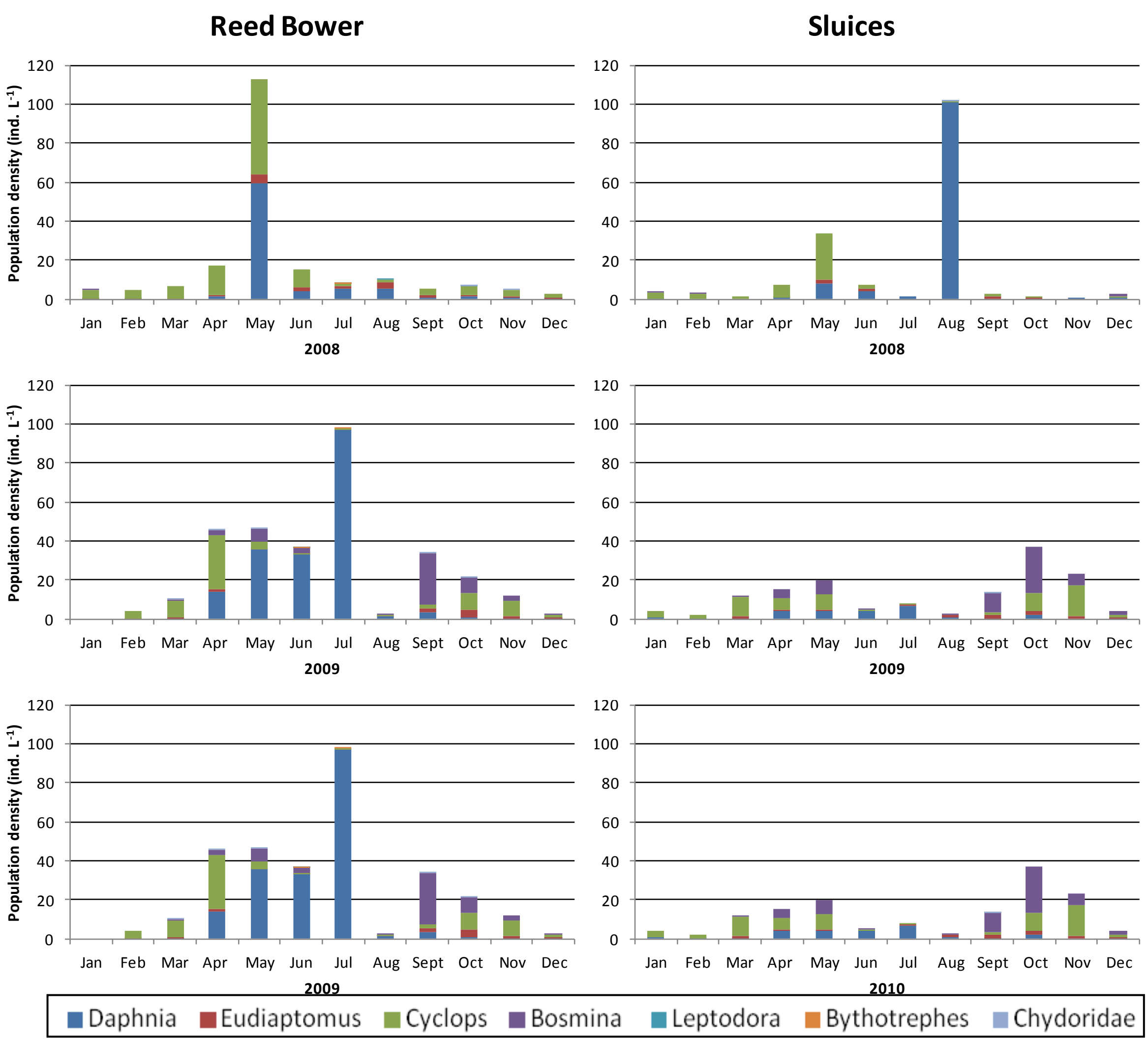


Figure 30: Temporal variation in crustacean zooplankton at the Reed Bower and Sluices sampling sites, Loch Leven, 2008-2010.

The crustacean zooplankton recorded at the Sluices in 2009 were similar in terms of species composition and temporal variation to those found at Reed Bower. However, *Daphnia* numbers at the Sluices were much reduced, in comparison with Reed Bower, over its peak growth period from April – July. From September onwards, the crustacean zooplankton were mainly composed of *Cyclops* and *Bosmina*.

In 2010, the principal crustacean zooplankton taxa were *Daphnia, Cyclops, Bosmina* and *Eudiaptomus.* Compared with 2009, *Daphnia* was less dominant in the plankton reaching a mean monthly population density of only 22.2 ind. L^{-1} in May. As is common at Loch Leven (but unlike the situation in 2009), *Daphnia* numbers then declined to relatively low densities over the summer months before increasing to a secondary peak of 14.1 ind. L^{-1} in September before again dropping to overwintering levels, i.e. less than 1 ind. L^{-1}. A maximum annual population density of *Daphnia* (37.4 ind. L^{-1}) was recorded at the Reed Bower sample site on the 25 May 2010. In 2010, the Cyclopoid copepod population was dominated by *Cyclops vicinus* for the first five months of the year, before the species disappeared from the plankton until the end of September. *Cyclops abyssorum*, in contrast, was recorded in low numbers from the summer months onwards. Copepodites and adults of *Cyclops* reached a mean annual population density of 28.2 ind. L^{-1} in April. The maximum annual population density of *Cyclops* (48.4 ind. L^{-1}) recorded was at the Reed Bower sample site on the 27 April 2010. *Eudiaptomus* was, again, consistently present in the plankton but with mean population densities generally remaining very low (<2 ind. L^{-1}) throughout the year. The maximum annual population density of *Eudiaptomus* recorded (3.3 ind.L^{-1}) was at Reed Bower on 6 July 2010. *Bosmina longirostris* was, again, relatively common in the plankton throughout 2010, with a mean monthly density of 28 ind. L^{-1} recorded in May. A maximum annual population density of *Bosmina* (50.8 ind. L^{-1}) was recorded at Reed Bower on 11 May 2010. The predatory cladocerans *Leptodora kindti* and *Bythotrophes longimanus* again occurred in extremely low numbers (<0.2 ind. L^{-1}) over the summer months during 2010. Chydoridae were also recorded in the zooplankton samples occasionally in 2010.

The crustacean zooplankton recorded at the Sluices sample site in 2010 were very similar in terms of species composition and temporal variation to that found at the Reed Bower sample site, i.e. a May peak in *Cyclops*, *Bosmina* and *Daphnia* numbers followed by low numbers over the summer months before a secondary peak of growth in September with very low crustacean zooplankton numbers thereafter.

4 DISCUSSION

4.1 Recent trends in physical and chemical variables

Recovery from eutrophication, as evidenced by annual mean TP (Figure 31) and chlorophyll$_a$ (Figure 32) concentrations, and by Secchi depth transparency (Figure 33), is apparent from the late 1980s to date. In general, all three of these variables show improving trends over this period that are consistent with expectations, given the major reductions in catchment P loadings to the lake that occurred between the mid-1980s to mid-1990s (May et al., 2012). In general, the values reported between 2007 and 2010 for TP, chlorophyll$_a$ and (to a lesser extent) Secchi depth, indicate that this has been one of the best periods on record for water quality in Loch Leven. In relation to WFD targets, TP concentrations between 2007 and 2010 were at, or slightly above, the (2006) WFD good/moderate boundary target of 32 µg L^{-1} (Figure 31 and Figure 35; Carvalho et al., 2006). Also, annual mean chlorophyll$_a$ concentrations were some of the lowest on record even though, in terms of WFD targets, they still indicated poor ecological status in 2008 and 2010 and moderate ecological status in 2009.

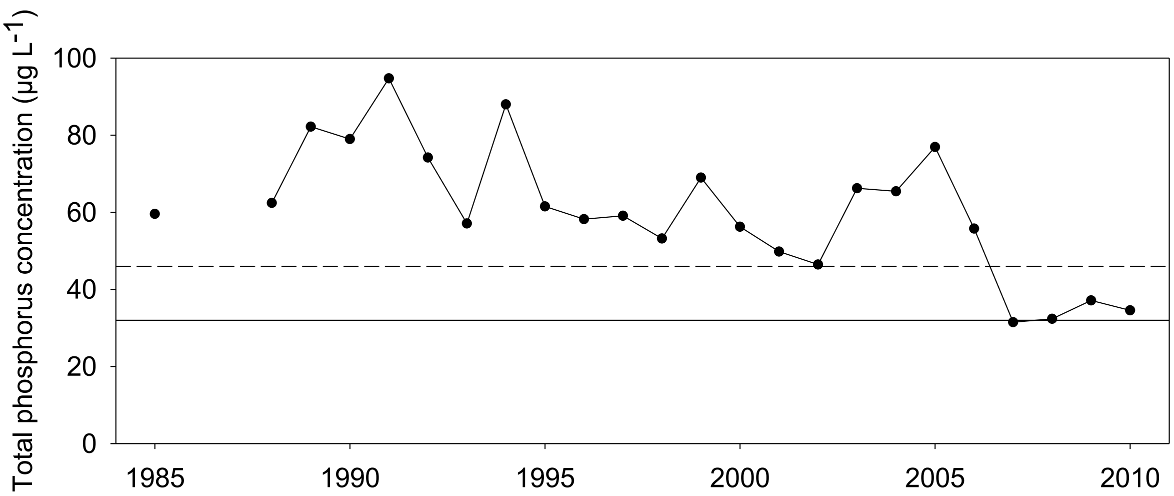


Figure 31: Long term trends in annual mean total phosphorus (TP) concentrations. Full black line represents the (2006) WFD good/moderate boundary and the dotted black line represents the WFD moderate/poor boundary (Carvalho et al., 2006).

A general deepening of Secchi depth is reported between 2005 and 2010, and the most recent annual mean value (2010: 2.3 m) is very close to the Loch Leven Area Management Group (LLAMAG, 1993) target of 2.5 m (Figure 33). However, wind induced wave mixing may also be an important factor in limiting the response of water clarity to the reduction in nutrient loading as this may results in an increase in turbidity through disturbance of bed sediments. Spears and Jones (2010) concluded that about 20-35% of the loch bed was sensitive to wave disturbance. As such, further consideration should be given to the effects of wind induced sediment disturbance on the ability of water managers to achieve the LLAMAG target for Secchi depth.

Although the data are incomplete, a general decrease in annual mean nitrate-N concentrations was observed at Loch Leven between 1985 and 2010 (Figure 34). This may indicate that N loading from the catchment has been reduced as a result of catchment management practices aimed at reducing P inputs. However, the reduction in nitrate-N concentrations may also be related to in-lake processes, such as increased rates of denitrification. At present, no nitrogen targets are available for Loch Leven. Further consideration of N targets in Loch Leven is important for future water quality management at the site.

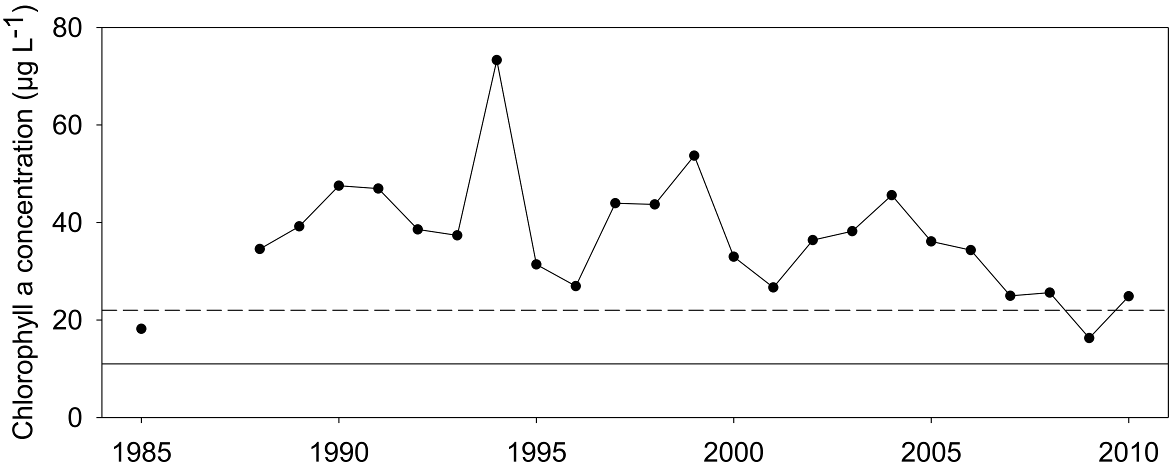


Figure 32: Long term trends in annual mean chlorophyll$_a$ concentrations. Full black line represents the WFD good/moderate boundary and the dotted black line represents the WFD moderate/poor boundary (UKTAG, 2008).

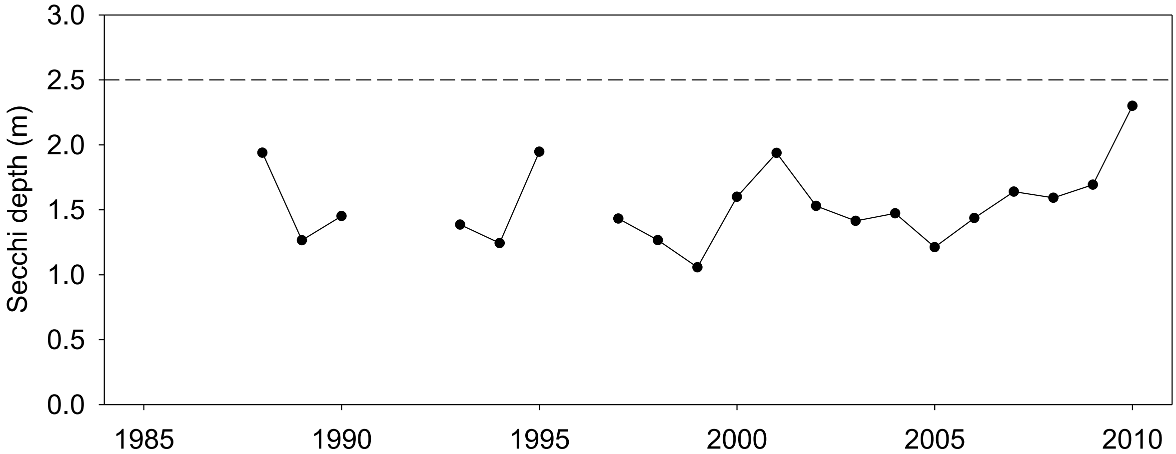


Figure 33: Long term trends in annual mean Secchi depths. Dotted black line represents the LLAMAG target of 2.5 m.

Water column nutrient concentrations in Loch Leven (Figure 34, Figure 35, Figure 36) show strongly seasonality. However, it is important to recognise that both in-lake processes and catchment nutrient loads may regulate this seasonality. In terms of N, for example, monthly nitrate-N concentrations fall over the period between winter and late summer/early autumn. This is probably due to high catchment supply in winter and spring and a combination of denitrification and biological uptake, both of which tend to increase with rising temperature, in summer and autumn. The N removal processes appear to be capable of reducing nitrate-N concentrations in the loch in summer sufficiently to maintain a degree of N-limitation. In winter/spring, however, it is likely that the reduction in nitrate-N concentrations is driven by changes in runoff from agricultural practices and point sources inputs, rather than within lake denitrification.

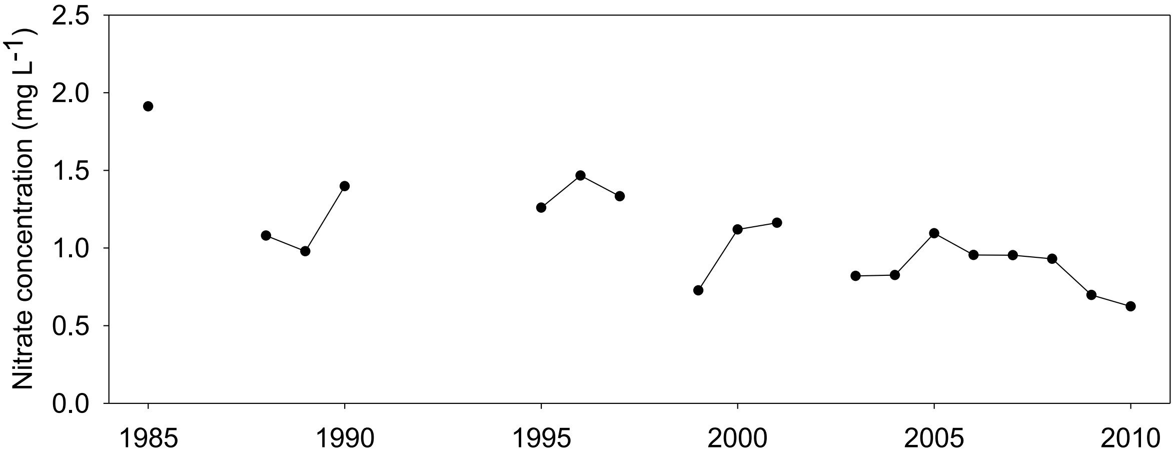


Figure 34: Long term trends in annual mean nitrate concentrations.

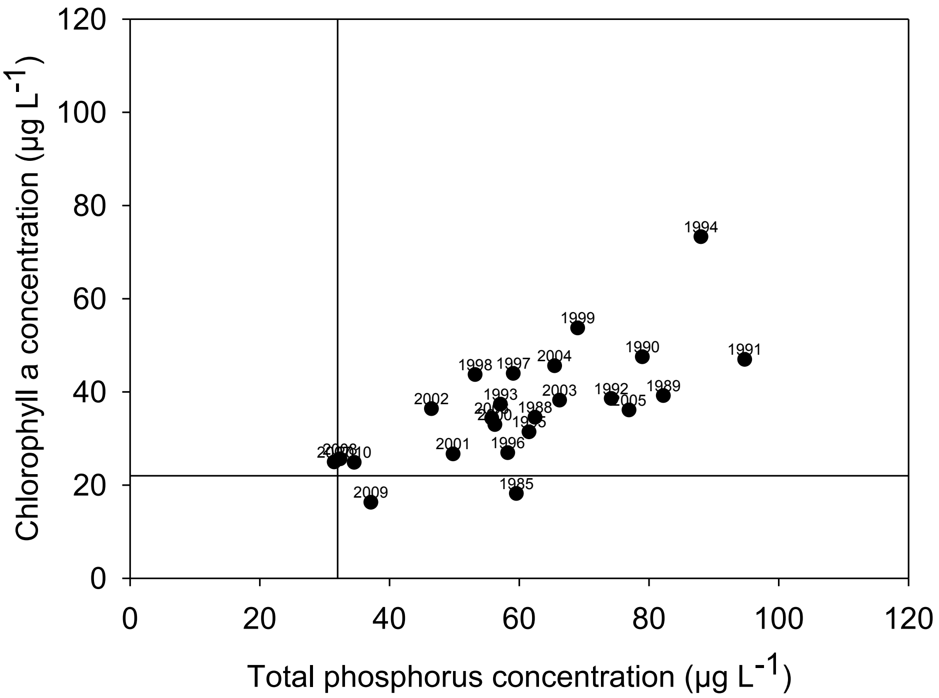


Figure 35: Plot of annual mean total phosphorus and chlorophyll$_a$ concentration (1985-2010). Horizontal full black line represents the WFD good/moderate chlorophyll$_a$ concentration target and the vertical full black line represents the WFD good/moderate TP concentration target.

The TP and SRP concentrations observed in Loch Leven during the high internal loading years (i.e. especially prior to 2007) showed a seasonal pattern that is common to most eutrophic shallow lakes, i.e. with winter and summer peaks reflecting external and internal P sources, respectively. The summer peak in P concentration in Loch Leven was similar to the summer/autumn peaks observed in many other eutrophic shallow lakes and, between 1985 and 2006, reflected the dominant P source to the water column in terms of the annual mean level (Spears et al., 2012).

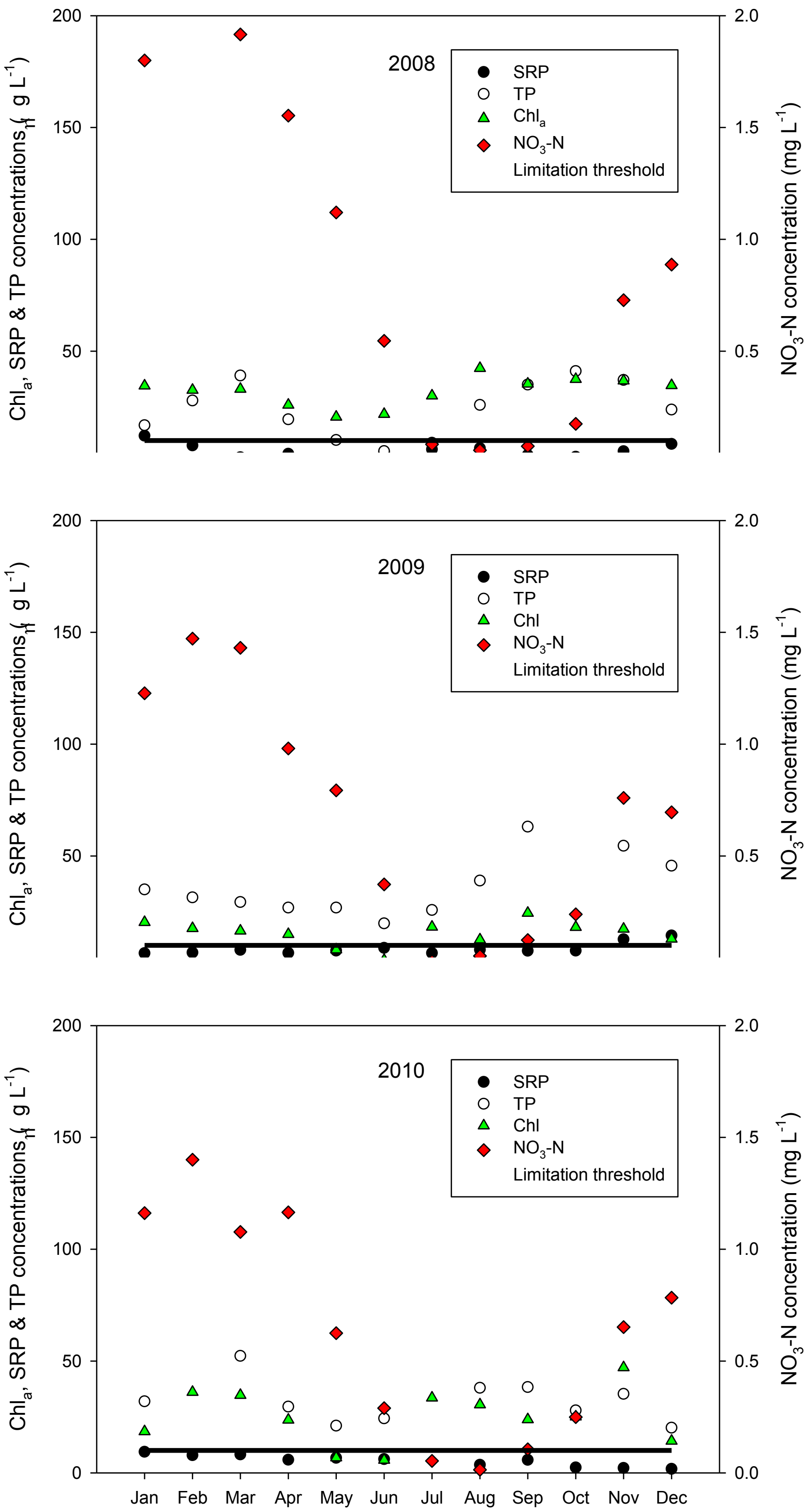


Figure 36: Monthly mean concentrations of chlorophyll$_a$ (Chl$_a$), total phosphorus (TP), soluble reactive phosphorus (SRP), and nitrate nitrogen (nitrate-N) in Loch Leven for 2008, 2009 and 2010. The full black line represents an estimate of nutrient limitation of phytoplankton for both for SRP (10 µg L^{-1}) and nitrate-N (0.1 mg L^{-1}).

4.2 Recent trends in biological communities

4.2.1 Phytoplankton and chlorophyll$_a$

Low chlorophyll$_a$ concentrations in May and June are now an established feature at Loch Leven, with a typical *spring clear water phase* being observed every year. Analysis of Loch Leven's long-term datasets suggest that this is, in part, a response to enhanced *Daphnia* grazer abundance associated with warmer spring temperatures in recent years (Ferguson et al., 2007a; 2007b). However, when compared across 2008-2010 (Figure 36), it is clear that nutrient limitation of phytoplankton by N and P may also play an important role in limiting chlorophyll$_a$ concentrations, now. In general, SRP concentrations are low enough to potentially limit algal crops through much of the year, with co-limitation of N and P evident during July and August, and intermittently in September.

The increased likelihood of N-limitation of phytoplankton during the summer-autumn period in recent years appears to be leading to a compositional shift to N-fixing cyanobacteria, such as *Anabaena*, in late summer, instead of the colonial cyanobacterium *Microcystis*. In terms of total cyanobacterial abundance, however, the apparent increase in the strength of N & P limitation in late summer appears to be resulting in a general decline, with particularly low densities in 2008 and 2010 that were well below the WHO (1999) low risk threshold (Figure 28).

Autumn 2009 stands out in the most recent three years of data as having chlorophyll$_a$ concentrations and phytoplankton biovolumes that were particularly low when compared to TP concentrations. There are two possible explanations for this. Firstly, more abundant zooplankton in autumn 2009 may have resulted in higher grazing pressures (Figure 30) and, secondly, wind storms may have disturbed the inorganic sediments and increased the water column TP concentration without, necessarily, increasing the chlorophyll$_a$ concentration.

The dominance of the Loch Leven phytoplankton community by diatoms appears to be becoming a more typical pattern. This dominance was particularly marked in 2010 and may indicate an improvement in water quality. Also, rainfall was quite high in July 2010, which may have helped by preventing slower-growing cyanobacteria from establishing effectively and developing into large populations in August and September. Recent studies have shown that flushing rate is as important to the development of a cyanobacterial bloom as nutrient availability (Carvalho et al., 2011; Elliott, 2010)

4.2.2 Zooplankton

In terms of species composition and the relative abundance of the principal taxa, the 2008-2010 results for the Loch Leven crustacean zooplankton community were broadly similar to previous years. However, in terms of seasonality, there has been a general increase in the abundance of *Daphnia* in spring, in recent years. As *Daphnia* are the main grazers of algae in Loch Leven, this may have led to corresponding reductions in chlorophyll$_a$ concentrations and improvements in water clarity (Carvalho et al., 2012).

Between 2008 and 2010, the patterns of *Daphnia* occurrence were broadly similar in terms of mean monthly densities, although there were some anomalies (Figure 37). For example, in 2008, an exceptionally high concentration of *Daphnia* (200 ind L^{-1}) was recorded at the Sluices at the beginning of August. In 2009, *Daphnia* did not decline in summer after the spring peak in May, as in most other years. Instead, numbers remained high throughout June before increasing to 124 ind. L^{-1} at Reed Bower at the end of July.

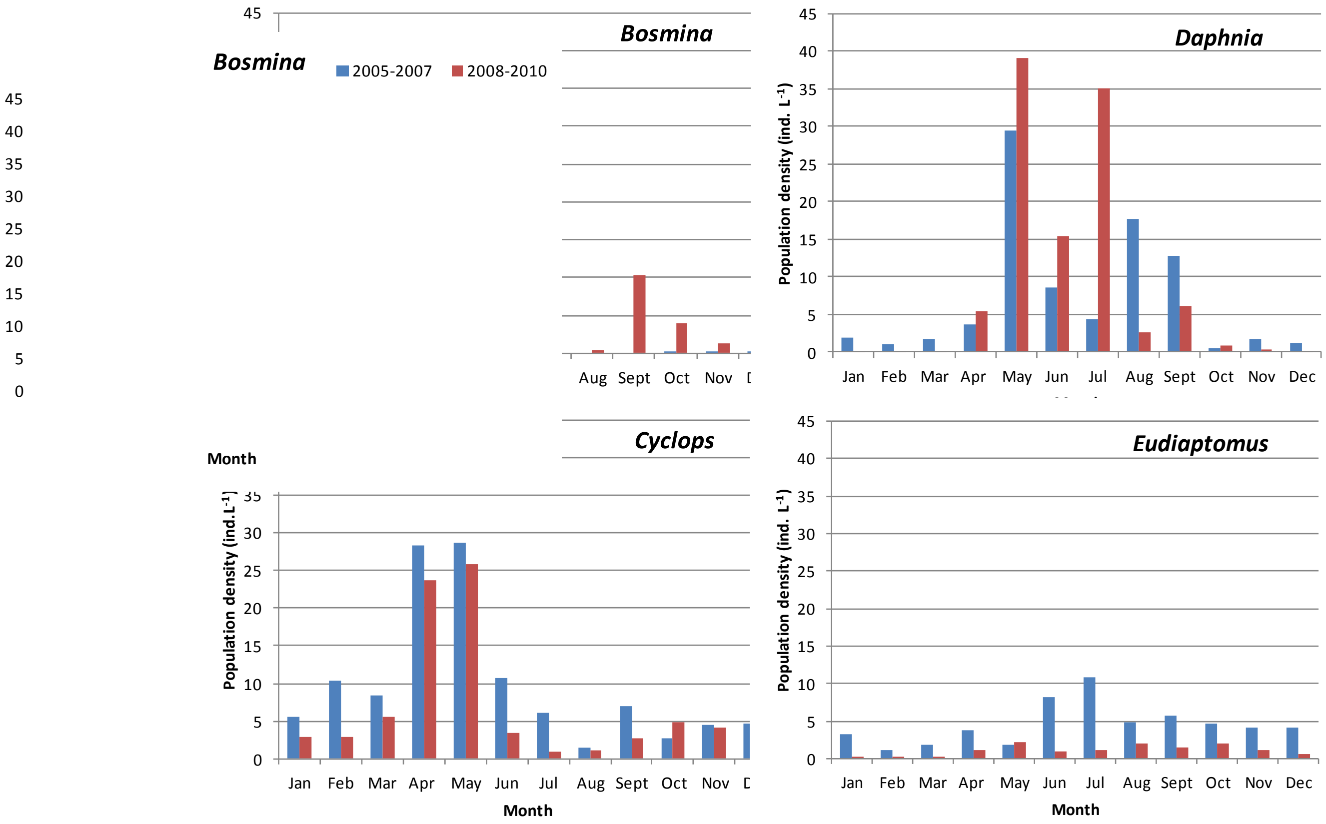


Figure 37: Mean densities of <u>Bosmina</u>, <u>Cyclops</u>, <u>Daphnia</u> and <u>Eudiaptomus</u> in Loch Leven 2005-2007 (blue bars) & 2008-2010 (red bars).

In 2007, *Cyclops*, which normally co-dominated the Loch Leven crustacean zooplankton community with *Daphnia*, occurred in very low numbers in comparison with earlier years (Carvalho et al., 2008; Figure 37). The cause of these low densities was unclear. However, between 2008 and 2010, overall numbers appeared to recover to levels comparable to earlier years with peak densities occurring during the spring months. However, the dominant species alternated between two species, i.e. *C. abyssorum* and *C. vicinus*, the latter having returned to the loch in the 2000s after a long period of absence from the plankton samples (Gunn et al., 2012). *C. abyssorrum* had co-dominated the crustacean zooplankton community since the early 1970s but, since then, copepodites and adults of *C. vicinus* have become increasingly prevalent in the early and latter part of the year with *C. abyssorum* still being recorded, particularly in the summer months. *Cyclops* species are typically omnivorous in Loch Leven, feeding on small crustacea and rotifers (Rutkowski, 1980).

The herbivorous species *Eudiaptomus gracilis* was the dominant copepod species during most of 2007, when *Cyclops* was relatively absent. However, in 2008-2010, it returned to the relatively low numbers characteristic of earlier years (Figure 37). This was probably due to changes in predation pressure by *Cyclops,* which have been observed feeding on young *Eudiaptomus* and *Daphnia* in Loch Leven by Johnson and Walker (1974).

The appearance of *Bosmina* in large numbers in Loch Leven in May 2007 was very surprising as only a few individual specimens had been recorded in the plankton in recent years (Gunn et al., 2012). Although barely recorded in 2008, *Bosmina* was commonly recorded in both spring and autumn in 2009 and 2010, suggesting that this species has now established itself as a regular feature of the crustacean zooplankton community (Figure 37). *Bosmina longirostris* is a grazer of phytoplankton that is characteristic of large eutrophic water waterbodies (Fryer, 1993).

4.3 Implications for management

4.3.1 Catchment considerations

Phosphorus inputs to Loch Leven were reduced from about 20 t y^{-1} in 1985 to about 8 t y^{-1} by 1995, mostly as a result of reductions in discharges from point sources (May et al., 2012). Although the loch now seems to be in recovery, it is important to ensure that water quality continues to improve by keeping point source discharges at a low level and addressing inputs from diffuse sources, such as farm runoff and on-site sewage treatment systems. In relation to the latter, it is important that the '125% rule' (LLCMP, 1999) continues to form part of the local planning process associated with building development in rural areas of the catchment. It is also important that local farmers continue to reduce nutrient runoff and soil erosion from agricultural land through best management practices, and that water treatment facilities control nutrient laden discharges, especially during heavy rainfall. Although work so far has concentrated on reducing P inputs to the lake, it is important to consider N inputs too, now that the lake appears to be becoming increasingly N-limited in summer. More importantly, the series of decadal assessments of nutrient inputs to the loch should be continued to monitor changes in the levels of nutrient inputs associated with changes in catchment management activities. The next survey should be undertaken in 2015.

4.3.2 In lake processes

Spears et al. (2012) indicated that the drop in annual mean TP concentration recorded in 2007 was the result of a significant reduction in the magnitude of the summer internal loading of P. Reduced levels of internal P load have continued through 2008-2010. However, in summer/autumn 2009, concentrations were higher than in 2007, 2008 or 2010, suggesting a temporary rise in internal P loading in that year. Spears et al. (2012) identified a range of drivers that control the magnitude of internal P loading in Loch Leven. These included wind speed (positive effect), summer temperature (positive effect), and spring water clarity (negative effect). They hypothesised that variations in the weather could shape the relationships between lake ecology and biogeochemical cycling, so that internal loading would be 'switched off' in calm, cool and clear water years and 'switched on' in windy, hot and turbid-water years. Similar observations were made by Carvalho et al. (2012) who showed that TP and chlorophyll$_a$ concentrations were significantly lower in wetter years than drier years. Both of these papers suggest that water quality may now be driven strongly by in-lake biogeochemical feedbacks, the magnitude of these being highly sensitive to changes in weather conditions.

The importance of internal loading in shaping water quality is illustrated in Figure 38, where measured TP concentrations closely mirror estimated internal loading figures. Internal loading estimates for 2008 and 2010 indicate that these were the two lowest years on record and that even 2009, which had more than double the level of internal P loading recorded in 2008 and 2010, was still much lower than 2004-2006.

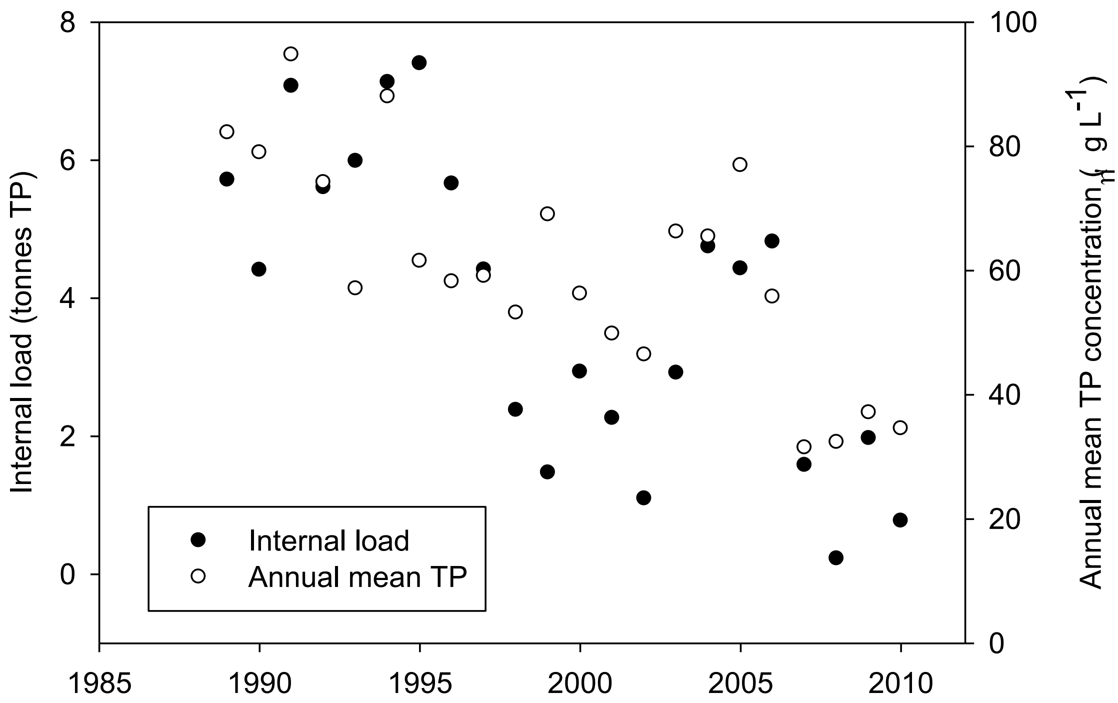


Figure 38: Estimates of internal phosphorus load based on the difference between trough (spring) and peak (summer-autumn) total phosphorus (TP) concentrations expressed as mass increase of TP in the lake water column. Annual mean TP concentrations are also shown.

A statistical comparison of 'low' internal loading (I_{load}) years (2008-2010; average I_{load} = 0.99 tonnes TP) and 'high' internal loading years (2004-2006; average I_{load} = 4.66 tonnes TP) was conducted (Table 9). Low internal loading years were found to be characterised by lower *Daphnia* (and total zooplankton) abundances and lower spring TP concentrations. This suggests a generally reduced level of ecosystem production in 'low' internal loading years, although a more functional assessment of linkages between pressure and response variables is required to substantiate these inferences. Summer mean TP and SRP concentrations are significantly lower, and water clarity significantly greater, in 'low' internal loading years. In terms of annual means, TP, chlorophyll$_a$ and SRP concentrations are all significantly lower, and water clarity is significantly greater, in 'low' internal loading years.

As discussed in Section 4.1, denitrification in surface sediments is expected to occur mainly during summer months, under high temperatures and associated anoxic conditions. Where denitrification is an important driver of water quality, a reduction in nitrate-N concentration and an increase in NH_4-N concentration (relative to nitrate-N concentration) is commonly observed. These patterns are clearly evident in the nitrate-N data for Loch Leven (Figure 34) and the decrease in nitrate-N concentrations during warmer weather suggests a moderate possibility of N-limitation occurring between 8°C and 19°C (Figure 39).

Table 9 - Results of analysis of variance (1-way ANOVA; season: DF, 5; α = 0.05; annual: DF, 23; α = 0.05) between high internal loading years (2004-2006) and low internal loading years (2008-2010). I_{load} - internal P load estimate (tonnes total phosphorus); TP – total phosphorus concentration (μg L^{-1}); Chl_a – chlorophyll$_a$ concentration (μg L^{-1}); nitrate-N (mg L^{-1}); Silica – Soluble Reactive Silica (mg L^{-1}); SRP – soluble reactive phosphorus (μg L^{-1}); Clarity – Secchi disk reading (m); Daphnia – Daphnia *abundance (L^{-1}); Zoop tot – average total zooplankton abundance (ind. L^{-1}); Diatoms, Green, Cyano., Crypto. – average total biovolumes (mm^3 L^{-1}) of diatoms (Bacillariophyceae), green algae (Chlorophyceae), cyanobacteria (Cyanophyceae) and cryptophytes (Cryptophyceae). Values in bold indicate significant results (p < 0.05).*

	Winter			***Spring***			***Summer***			***Autumn***			***Annual***		
Variable	High	Low	p	High	Low	p	High	Low	p	High	Low	p	High	Low	p
I_{load}							**4.66**	**0.99**	**0.02**						
TP	59.5	35.8	NS	**41.5**	**29.5**	**0.03**	**78.3**	**30.3**	**<0.01**	84.9	43.1	NS	**66.0**	**34.68**	**<0.01**
Chl_a	28.4	23.8	NS	23.9	19.3	NS	35.6	16.0	NS	67.0	29.9	NS	**38.7**	**22.2**	**0.03**
nitrate-N	1.7	1.3	NS	1.5	1.2	NS	0.3	0.2	NS	0.3	0.4	NS	1.0	0.8	NS
Silica	2.6	5.5	NS	0.5	1.4	NS	3.2	2.2	NS	3.0	3.1	NS	2.3	3.0	NS
SRP	12.1	10.0	NS	6.3	5.7	NS	**19.1**	**6.1**	**<0.01**	11.6	5.5	NS	**12.3**	**6.8**	**0.02**
Clarity	1.4	1.5	NS	1.6	1.8	NS	**1.5**	**2.4**	**0.03**	1.0	1.8	NS	**1.4**	**1.9**	**0.02**
Daphnia	**1.5**	**0.2**	**0.02**	6.2	14.9	NS	8.3	17.7	NS	1.9	2.5	NS	4.4	8.8	NS
Zoop tot	**17.2**	**4.2**	**<0.01**	30.5	37.5	NS	19.5	21.5	NS	15.0	13.8	NS	20.6	19.2	NS
Diatoms	8.1	7.7	NS	7.0	7.0	NS	2.1	3.7	NS	20.0	4.7	NS	9.3	5.7	NS
Green	0.2	0.4	NS	0.4	0.5	NS	0.3	0.5	NS	0.2	0.2	NS	0.3	0.4	NS
Cyano.	0.2	0.4	NS	0.1	0.5	NS	1.6	1.2	NS	0.5	0.7	NS	0.6	0.7	NS
Crypto.	0.1	0.4	NS	0.3	0.5	NS	1.6	0.9	NS	0.2	0.4	NS	0.6	0.6	NS

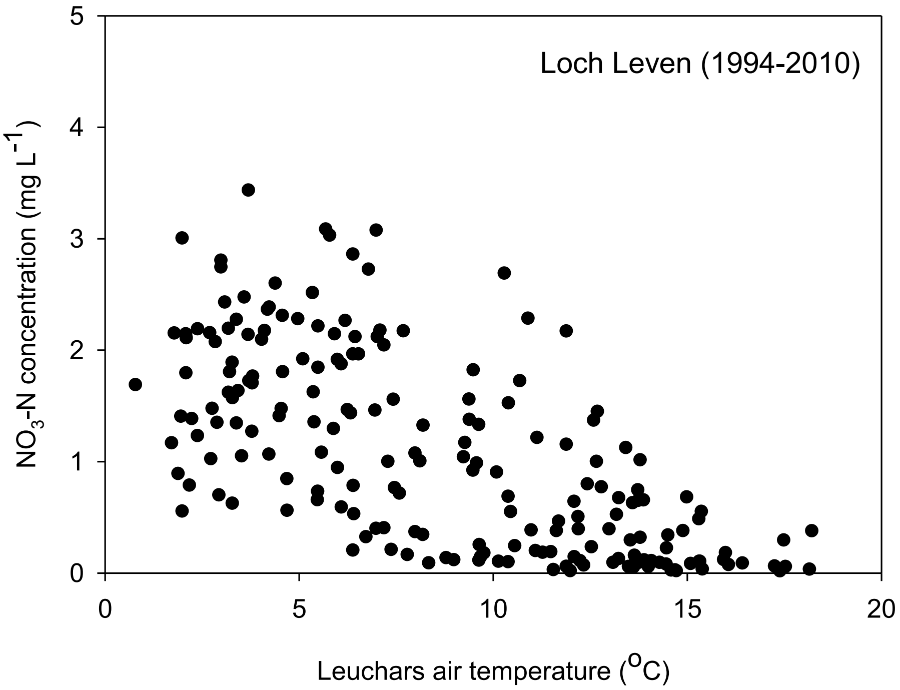


Figure 39: Plot of air temperature and nitrate-N concentration in Loch Leven. Values are monthly means between 1994 and 2010.

4.3.3 Climate drivers

Monitoring at Loch Leven has spanned periods of increasing and decreasing nutrient loads and of changing climate. Our research is improving understanding of how annual and seasonal trends in key water quality parameters respond directly and indirectly to changes in air temperature, rainfall and wind, and how climatic changes impact on the way that shallow lakes respond to restoration measures (Carvalho et al., 2012; Spears et al., 2012). It is often assumed that climate change will have negative impacts on water quality, especially by increasing the incidence of cyanobacterial blooms (Paerl &Huisman, 2008). However, our research shows that climate change is not a single stressor. Some seasonal changes in temperature and rainfall can have both positive (more *Daphnia* in spring and increased summer flushing) and negative (higher internal P loading in late summer) impacts.

The most obvious climatic impact on water quality at Loch Leven is the significant negative relationship between summer rainfall and chlorophyll$_a$ concentrations from 1989 to 2008 (Carvalho et al., 2012). This is especially clear in extreme weather years, with the wettest summers (2004, 2007 and 2008) having very low summer chlorophyll$_a$ concentrations and the driest summers (1994, 1995 and 2006) having high chlorophyll$_a$ concentrations. This pattern is probably caused by the direct causal relationship between flushing rate and phytoplankton loss processes. Negative impacts of climate change include increased P release from the sediments in years with warmer summers and those with more wind (Spears et al., 2012). Warmer summers may also now have positive effects on water quality, because they appear to enhance denitrification. This may help limit algal blooms by increasing nitrogen limitation.

What is clear is that lake managers need to acknowledge that climate changes have an impact on lake functioning and that this will lead to shifting water quality baselines (or reference conditions) against which current status and the success of restoration measures can be assessed.

4.4 Recommendations for future monitoring and management

The results from monitoring Loch Leven between 2008 and 2010 appear to show a sustained improvement in water quality in comparison with earlier years. Metrics such as TP and chlorophyll$_a$ concentrations, although variable, are beginning to converge on their targets, and an increase in zooplankton diversity is reassuring because it increases grazing pressure and, consequently, reduces algal concentrations and increases water clarity.

The measures undertaken to reduce loads of phosphorus during the 1980s and 1990s have had a significant impact on water quality in the lake, but it is unclear whether these measures are sufficient to achieve the targets set by LLAMAG (1993) or under the WFD. The most recent data suggest that, while targets for TP concentration are probably achievable, the WFD 'Good' status target for chlorophyll$_a$ concentrations may still be very challenging. It is possible that the current WFD targets for the site, which were established on the basis of chlorophyll$_a$ and TP relationships derived from a large population of shallow lakes from across Northern Europe, are not appropriate for an individual lake in the central lowlands of Scotland that has a mean to maximum depth ratio that is relatively unique. More refined site-specific targets that take individual lake characteristics into account, such as depth, fetch and natural flushing regimes, might be more appropriate.

The main sources that contribute to the overall P content of the water in Loch Leven continue to vary significantly, resulting in contrasting water quality conditions (Table 9). However, even the best water quality conditions are still characterised by occasional summer/autumn cyanobacterial blooms and failure to meet WFD and/or LLAMAG water quality targets. At present, the dominant source of P to the water column is legacy P in the lake bed sediments, although winter and spring in-lake TP concentrations suggest that catchment derived P sources are still important. Although catchment management has, undoubtedly, been successful in improving the ecology of the loch, recovery has taken about 20 years with relative stability of the improved conditions only being evident since 2007.

The magnitude of internal P loading in the loch is driven, predominantly, by interactions between weather conditions and antecedent water quality (Spears et al., 2012), with significant variation reported in internal loading over the last 20 years. A comparison of years with high or low internal loading, provides strong evidence that water quality is significantly better in years with low internal loading than those with high internal loading. Inter-annual changes in the magnitude of internal P loading can occur very suddenly (e.g. cf. 2006-2007 & 2008-2009). This suggests that water quality at this site is highly sensitivite to sudden shifts in internal P loading, and that the loch has low capacity to buffer the effects of such changes. However, a leading theory in shallow lake restoration is that this resistance to change ('resilience') can be enhanced by further reducing nutrient pressures and, where possible, controlling secondary pressures.

There is a need for a greater understanding of how flushing rate affects algal blooms in general, especially at Loch Leven, where it may be possible to use the long-term data to examine different options for managing the sluice gates to help mitigate the effects of climate change and further support the achievement of water quality targets. Other climate impacts requiring further investigation are the impacts of warmer and wetter summers on nutrient cycling processes, such as denitrification, and the release of P and ammonium from the lake sediments. Both of these studies could be carried out at Loch Leven, but it may also be beneficial to assess sensitivity to climate change in a number of Scottish waterbodies, especially those prone to nuisance algal blooms.

Spears et al. (2011) summarised all of the known pressure-response relationships in Loch Leven. These are shown in Figure 40. A number of the secondary pressures shown have been manipulated at this site, in recent years. Such activities have included:

- Re-introduction of white-tailed sea eagles (RSPB, 2012)
- Cessation of fish stocking (Winfield et al., 2012)
- Water level management (May & Spears, 2012c)

In addition, increases in temperature have resulted from climate change impacts. The combined effects of these changes on water quality and resilience at the site are not well understood.

Given the current level of knowledge about Loch Leven and its recovery in relation to previous and current restoration targets, we recommend that the following issues are taken into account when developing future management plans and updating water quality targets:

- Water quality targets need to take shifting climatic baselines into account
- Nitrogen concentrations may be an important factor in determining the response of chlorophyll$_a$ concentrations to changes in phosphorus availability
- Spring water clarity may be a more appropriate indicator of better growing conditions for macrophytes than annual average water clarity
- Pressures that affect the abundance of *Daphnia* need to be examined because these zooplankton grazers are one of the key drivers of spring water clarity; they also provide an important food resource for fish.
- Consideration should be given to the development of management targets based on ecosystem service provision (clean water, fish, amenity value) in addition to water quality indicators.
- Interactions between macrophytes, fish and water fowl may confound recovery from eutrophication at the site.

Further research is required to address these issues.

In addition, the impact that existing catchment management activities have had on nutrient inputs to the loch needs to be evaluated by undertaking a new nutrient loading survey that incorporates a detailed source apportionment study.

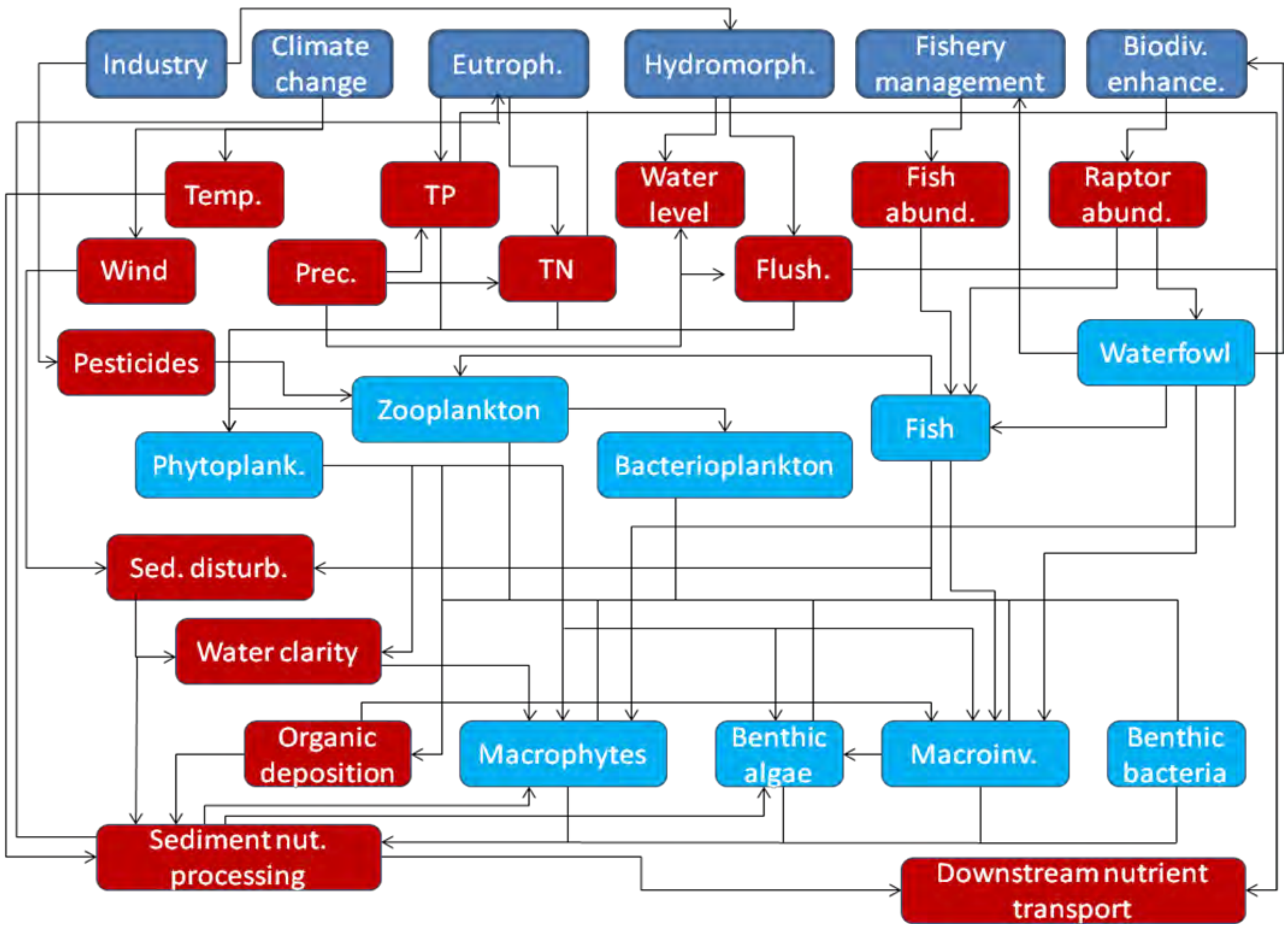


Figure 40: Links between primary and secondary pressures [dark blue], environmental state [red] and biological response[light blue] at Loch Leven, 1968-2010.

5 CONCLUSIONS

This report summarises the results of long-term research into water quality at Loch Leven and places the monitoring data from 2008-2010 into the longer-term perspective. The main conclusions are summarised below:

- Water quality improvements, which we attribute to catchment load reductions since the 1980s, have been sustained in 2008-2010.

- Cyanobacterial blooms continue to occur in late-summer and autumn in Loch Leven. During these blooms, cyanobacterial abundance often exceeds the WHO ‘Low Risk’ threshold, but has not exceeded the ‘Medium Risk’ threshold since 2004.

- A spring clear-water phase continues to be a characteristic feature of Loch Leven. This is encouraging for the further colonisation of macrophytes. The clear-water phase may be related to an increase in zooplankton diversity that may, in turn, be associated with increased grazing pressure on phytoplankton.

- In contrast to the 1970s and 1980s, when catchment sources dominated, the main source of phosphorus to the water column during the growing season has been the sediment of the lake (internal loading) since at least 2000.

- The magnitude of internal loading is related to multiple factors, including catchment pressures, weather and water quality. Climate change is likely to affect many of these factors, but the net effect of climate change on the lake’s water quality is unknown.

- The intensity of internal loading in summer determines total phosphorus and chlorophyll concentrations, and water clarity (through a feedback loop). Since 2005, it appears that the magnitude of internal loading has reduced.

- Nitrogen limitation appears to be playing a larger role in restricting phytoplankton abundance in summer. The management of nitrogen in the catchment should be considered.

- Further research is required to determine the impact that existing catchment management activities have had on nutrient inputs to the loch. This would be achieved by a nutrient loading survey that incorporates detailed source apportionment.

- Further research is required to understand the interacting roles of biology, climate and nutrient supply on water quality in Loch Leven.

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ISBN: 978-1-85397-863-0

Policy and Advice Directorate, Great Glen House,
Leachkin Road, Inverness IV3 8NW
T: 01463 725000

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