Strand 4 Water Quality Technical Report

Lough Melvin Nutrient Reduction Programme

Water Quality and Limnology of Lough Melvin 1990-2007

C. D. Barry, School of Biological Sciences, Queens University Belfast.
Newforge Lane, Belfast BT9 5PX, Northern Ireland.
c.barry@qub.ac.uk

R. H. Foy, Agri-Food & Biosciences Institute, Newforge Lane, Belfast BT9 5PX
Northern Ireland.
Bob.Foy@afbini.gov.uk

Preface

Lough Melvin has been designated as a Special Area of Conservation on the basis of its low to moderate nutrient concentrations, high quality aquatic vascular plant community and unique complement of salmonid fishes. In 1990, 2001/02 and 2006/07 water quality monitoring programmes were carried out on Lough Melvin to examine the health and sustainability of its ecological status. This report details the finding of each of these surveys and examines the trends which have taken place over time.

The results show that Lough Melvin is highly sensitive to changes in the pattern of land use within the catchment and that a gradual process of nutrient enrichment in response to increasing exports of phosphorus has occurred. While phosphorus concentrations are currently within mesotrophic limits an upward trend is clearly apparent, presenting a danger that future increases will raise concentrations to eutrophic levels. It is recommended that strategies be implemented not only to curb phosphorus loadings, but to reduce loadings sufficiently to revert to the status quo observed in the early 1990s'.

Acknowledgements

This project was possible due to the financial support provided to the Lough Melvin Nutrient Reduction Programme by Interreg IIIA. We are also happy to express our thanks for the kind help and assistance given by the following:

The Lough Melvin Nutrient Reduction Programme team for logistical and technical support

The Northern Regional Fisheries Board for regular use of their boat.

The Fisheries Officers of the Northern Regional Fisheries Board for their extensive help during sampling, which is greatly appreciated.

Special thanks to Neil Armour and Sharon O'Rourke of Queen's University Belfast for assistance in the field and laboratory and to Louise Davis, Phil Dinsmore, Peter Irvine, Emma Larson, Colm McKenna, Liam O'Flaherty, and Elaine Sayers of AFBI for their assistance with laboratory analyses.

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Executive Summary

In 2001/2 a water quality monitoring survey was carried out in recognition of Lough Melvin's candidate designation as a special area of conservation (SAC). The results of this work when compared with a previous survey in 1990 showed a significant deterioration in habitat quality by phosphorus enrichment and reductions in water clarity. Phosphorus concentrations were shown to have increased 54% from 19.1 to 29.5 µg L^{-1}. On the basis of these findings it was recommended that management strategies be urgently implemented to prevent further deterioration. In support of the Lough Melvin Nutrient Reduction Programme an additional year-long monitoring survey was commissioned in 2006 to provide up to date information on water quality. This report presents the results of each monitoring programme and examines the changes that have taken place since monitoring began in 1990.

During each monitoring survey the lake and its network of major inflowing rivers were monitored for physico-chemistry and biological limnology at fortnightly intervals for one year, allowing comparisons of the seasonal cycles in the lake and patterns of nutrient export from the catchment. In 2001/02 it was found that rates of phosphorus export from the catchment had increased, but not sufficiently to account for the observed rise in lake phosphorus concentration. This suggested that an unusual perturbation had occurred in the catchment resulting in a pulse of phosphorus to the lake in years preceding the 2001/02 survey. Clear-felling of coniferous forestry has been shown to substantially increase exports of phosphorus and organic matter, which reduces light penetration. Elevated lake phosphorus concentrations and reductions in water clarity observed in 2001/02 were therefore consistent with a period of widespread accelerated clear felling that occurred in the catchment following storm damage to coniferous forestry in late 1998.

In 2006/07 water clarity improved significantly suggesting a degree of catchment recovery, however phosphorus export rates again increased and lake concentrations remained at high levels. In contrast to the 2001/02 monitoring period, lake phosphorus concentrations in 2006/07 closely matched the pattern of loadings from the catchment. Thus while reductions in phosphorus concentration and increases in water clarity associated with catchment recovery have occurred, they have been offset by sustained increases of phosphorus export intensity in the catchment.

In freshwaters eutrophication is closely linked with an increased availability of phosphorus as it is the nutrient that usually limits primary production. Despite considerable phosphorus enrichment algal abundances in Lough Melvin have remained at low levels indicative of oligo-mesotrophic status. Clearly factors other than phosphorus availability limit algal production. Sufficient light for photosynthesis and growth by algae only reaches to five metres depth in Lough Melvin due to rapid attenuation by the peat stained water. As a result algae spend the majority of time in darkness and receive insufficient light to exploit the abundance of

phosphorus. Peat staining has therefore exerted a stabilising effect in Lough Melvin by counteracting the algal response to phosphorus enrichment.

Nevertheless phosphorus-enhanced algal growth still presents a significant threat. The frequency and severity of blue-green algal blooms, that are unsightly and potentially toxic to humans, pets and livestock, is far greater under conditions of high phosphorus availability. Sheltered bays and backwaters which posses a high recreational and aesthetic value are particularly prone to prolonged blue-green algal blooms. The littoral zones of the lake may also be subject to increasing pressure on the basis that algae attached to the substrate and aquatic plants are not limited by light and will exploit increases in phosphorus availability. Fast growing filamentous species of algae tend to become dominant under conditions of nutrient enrichment and these have the potential to displace natural floras and alter community structure. This is of particular significance in Lough Melvin where the littoral macrophyte community is a primary reason for its designation as a Special Area of Conservation (SAC) under the Habitats Directive.

Widespread increases in phosphorus export intensity have occurred in the catchment. The south-east area of the catchment, largely devoted to agriculture, has consistently shown the greatest increases. Accumulation of soil phosphorus, poor septic tank functioning and a number of agricultural practices are highlighted as potential causes. Forested areas show among the highest phosphorus export rates but some of the lowest nitrate export rates, while agricultural areas displayed both high phosphorus and nitrate export rates. Discharges of effluent from the three waste water treatment works within the catchment presently play a negligible role in the enrichment of the lake. However there are numerous new developments and the *per capita* nutrient burden upon the lake is expected to rise considerably in the future. Phosphorus loading from diffuse sources is the most significant cause of the recent enrichment of Lough Melvin. However the rapid increases of phosphorus export following clear felling, although relatively short lived, have highlighted the need for a measured approach and suggest that an annual limit should be set.

The annual pattern of temperatures and dissolved oxygen concentrations has been relatively consistent since 1990 and remains favourable for aquatic biota. Periods of stratification were observed in 2007 during which dissolved oxygen became depleted in the deeper waters, however these were of short duration and concentrations remained sufficiently high not to warrant concern. The principal components of the zooplankton and phytoplankton communities have not displayed any considerable shifts over time. However the abundances of species with the capacity to utilise terrestrial organic matter within have significantly increased, suggesting that such exports have increased. This hypothesis is reinforced by water clarity that still falls short of that observed in 1990. Although characteristic spring and summer phytoplankton peaks were observed in 2006/07, abundances were considerably lower than that observed in both 1990 and 2001/02. As no limiting factors were apparent compared to the earlier surveys it is impossible to give any definite causes. The low abundances may simply reflect a natural fluctuation however in view of evidence that suggests greater organic matter loading, attenuation of light by dissolved terrestrially derived compounds may in fact have increased the degree of light limitation and consequently primary production.

Lough Melvin currently fulfils the criteria required to justify designation as a mesotrophic lake. Nevertheless the clear upward trend in phosphorus loading and lake concentration demonstrates that action must be taken to avoid further deterioration of the habitat and eutrophic designation.

Introduction

Lough Melvin and its inflowing river network have been monitored for physico-chemistry and biological limnology at fortnightly intervals for three one year periods in 1990, 2001/02 and 2006/07. Serious deterioration of water quality occurred between 1990 and 2001/02, most notably by phosphorus enrichment where concentrations rose by 54% to 29.5 µg L^{-1}. At this rate of increase concentrations in the lake would reach the lower limit designated for eutrophic status by 2007 (OECD, 1982). Following the recommendations of the 2001/02 monitoring programme a further years monitoring was commissioned under the Lough Melvin Nutrient Reduction Programme (LMNRP) in order to provide up to date information on water quality.

<u>Phosphorus increases: 1990 - 2004</u>

Phosphorus is the nutrient that usually limits growth and production by algae in freshwaters and is used as a key parameter for defining the trophic status or productivity of water bodies (OECD, 1982). The mean total phosphorus (TP) concentration in Lough Melvin was relatively stable between 1990 and 1995/96 at 19.1 and 18.4 µg L^{-1} respectively (Girvan & Foy, 2006). Between 1995/96 and 2001/02 concentrations increased markedly to 29.5µg L^{-1} approaching the lower limit of 35µg L^{-1} defined for eutrophic status.

Significantly higher TP concentrations were observed in inflowing stream waters in 2001/02 compared to 1990 indicating increases in the intensity of catchment export. However on the basis of TP loadings to the lake in 2001/02 the predicted TP concentration, while greater than that predicted for 1990, was considerably lower than that observed. Thus increases in phosphorus export intensity alone could not account for the observed rise in concentration. This suggested that an unusual perturbation in the catchment had caused a pulse of phosphorus to the lake. In fact severe storms in December 1998 left extensive areas of fallen coniferous trees in the catchment and in the following 3 years timber recovery and clear felling operated at greater intensity. Clear felling of conifers on peat soils in Ireland has been found to increase phosphorus concentrations in drainage water (Cummins & Farrell, 2003a). It is therefore likely that forestry activity was largely responsible for the rapid increase in lake TP above the pattern indicated by catchment export rates.

The mean TP concentrations between August 2002 and 2004 showed a decline to 24.5 µg L^{-1} with lower values observed in the range (18 - 41 µg L^{-1} in 2001-02; 17 - 35 µg L^{-1} in 2002-04), raising the question of whether phosphorus concentrations will continue to decline as the catchment recovers or if a gradual rise of catchment phosphorus exports is occurring, irrespective of the 2001/2 peak.

Effects of elevated phosphorus concentrations: 1990-2004

As phosphorus usually limits algal production in freshwaters increases in TP are expected to result in greater primary productivity. However chlorophyll concentrations, a proxy for algal abundance, remained similar throughout the period 1990–2004. Since nitrogen concentrations in Lough Melvin are considered sufficiently high that they are not limiting (Girvan & Foy, 2006) algal productivity was limited by factors other than the availability of nutrients.

Light limitation of primary production is a relatively common occurrence in Ireland where many lakes receive high inputs of coloured humic material (Jewson & Taylor, 1978). These catchment-derived compounds significantly reduce the depth to which photosynthetically available radiation (PAR) penetrates (Fig. 1).

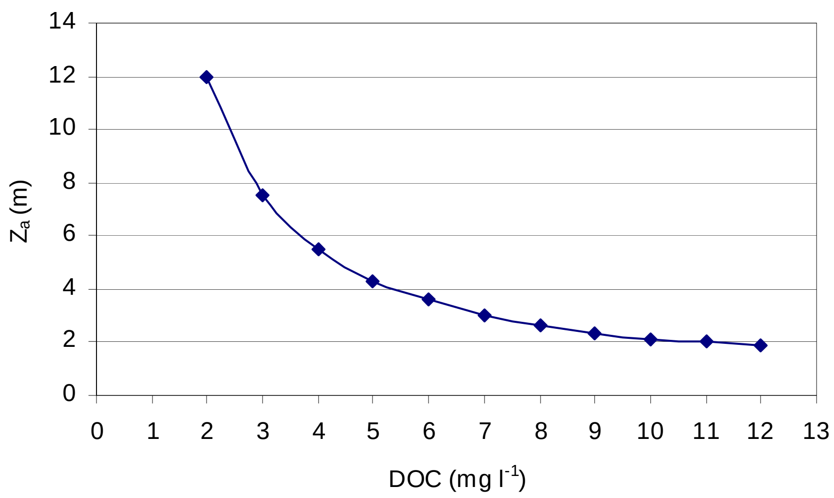


Figure 1 Relationship between the 1% attenuation depth (Z_a = 1% of surface irradiance) of photosynthetically available radiation (400-700nm) and dissolved organic carbon concentration (DOC) based on a survey of 65 glacial lakes in North and South America. (Adapted from Wetzel (2001) after data of Morris *et al.* (1995) and Williamson *et al.* (1996)).

In relatively deep, polymictic, humic stained lakes such as Lough Melvin the optical depth and the mixing depth are particularly strong determinants of primary productivity as circulating algal cells spend the majority of time in darkness. In neighbouring Lough Erne for example, where TP concentrations are indicative of eutrophic conditions, light reduction by humic compounds limit the algal response to levels that merit mesotrophic classification (Foy *et al.,* 1993).

Terrestrially derived organic compounds can therefore exert an optical control upon productivity, donating a degree of 'trophic stability' to the system (Girvan & Foy, 2006).

Possible effects of continued nutrient enrichment

In alkaline, humic, polymictic lakes such as Lough Erne and Lough Melvin the phytoplankton community is dominated by cyanobacteria. Many species of which posses gas vacuoles that confer positive buoyancy, allowing them to float into the illuminated surface waters during calm conditions and increase their light dose relative to other species. An unfortunate consequence is that unsightly and potentially hazardous cyanobacterial surface scums can form rapidly. Often these cyanobacterial surface scums are incorrectly described as algal blooms where in actuality calm conditions have simply concentrated cells or colonies at the surface that were previously distributed within a large volume of water. They have probably been a natural characteristic of these lakes since before the onset of anthropogenic eutrophication, nevertheless the frequency and severity of these scums and actual blooms can be expected to be greater under elevated nutrient concentrations.

In contrast to the pelagic system, where lack of light has effectively limited responses to phosphorus enrichment, the littoral zone of the lake may be more sensitive. Fast growing filamentous algae often become dominant in areas subject to nutrient enrichment, displacing naturally occurring algae and vascular aquatic plants, species of which are a primary reason for Lough Melvin's designation as an SAC under the Habitats Directive. Filamentous algae can also impede water movement and cause the accumulation of silt and organic matter which are factors linked with the loss of high quality macrophyte communities. Littoral invertebrate communities can also suffer through loss of suitable habitat with subsequent effects upon the consumer species that depend upon them. Additionally pebble and gravel beds are used for spawning by various salmonid species and these may become unsuitable with excessive growths of algae. There is already an example of a littoral response to nutrient enrichment in Lough Melvin where the Garrison waste water treatment works discharges directly into the lake. Here dense epilithic carpets of the filamentous green alga, *Cladophora glomerata* grow on the surfaces of rocks along a significant stretch of the littoral zone. At present this species appears confined to this area of the lake.

Summary

Despite a period of significant nutrient enrichment the pelagic productivity of Lough Melvin has remained largely unchanged between 1990 and 2004. If nutrient concentrations continue to rise the effects may be more varied and subtle than the typical phytoplankton response but no less serious in the potential for degradation of the ecosystem as a whole.

As part of the LMNRP, monitoring of the lake and inflowing catchment rivers for water quality commenced in March 2006. Higher resolution sampling at fortnightly intervals was conducted for one year between September 2006 and 2007. The results of this work and data from the previous monitoring programmes of 1990 and 2001/2 are presented and discussed.

Purpose

● Present physico-chemical and biological data based on the monitoring programmes of 1990, 2001/02 and 2006/07

● Construct an up to date nutrient budget for the catchment with emphasis upon determining the principal nutrient sources

● Examine what changes have taken place since monitoring began in 1990

Methods

Lake sampling

In the previous monitoring surveys of 1990 and 2001/02 three sites on the lake were sampled in order to cover spatially distinct areas over a range of depth. As biological and physicochemical variables showed little variation between sites, only samples from the deepest point (Site 1) were monitored in 2006/07.

Lough Melvin was sampled monthly from March to August 2006 and fortnightly from September 2006 to September 2007. A mooring buoy was deployed over the deepest point to maintain a constant position between sampling visits and during sampling. Discrete water samples were taken from the surface, 20, 30 and 40 metres depth using a Kemmerer water sampler (Wildco, USA). A composite sample to 10m was taken using an epilimnetic tube sampler (Lund & Talling, 1957). Zooplankton were sampled by vertical hauls over the depth of the water column using a 53µm mesh plankton net. Water transparency was measured using a Secchi disk. Depth profiles of dissolved oxygen and temperature were taken at 5m intervals using a YSI model 59 dissolved oxygen and temperature meter. A YSI model 6600 sonde was deployed unattended over the deepest point of the lake at 38m depth between the 2nd May 2007 and 16th August 2007. The sonde recorded dissolved oxygen, temperature, conductivity and pH at ten-minute intervals for the duration of deployment.

Laboratory analyses

Analyses were carried out at the Agriculture and Food-Biosciences Institute (AFBI) according to well-established protocols that are subject to internal quality control and formal quality assurance schemes (ISO9001;2000). The dissolved fraction for analysis was determined by filtration at 0.45µm. Total phosphorus (TP), total soluble phosphorus (TSP), soluble reactive phosphorus (SRP), total oxidised nitrogen (TON), nitrite (NO_2), ammonium (NH_4) and soluble silica (SiO_2) were determined using standard methods according to Gibson et al. (1980) and Jordan (1997). The soluble organic phosphorus (SOP) fraction was determined by subtraction of SRP from TSP concentrations and the particulate phosphorus (PP) fraction was determined by subtraction of TSP from TP concentrations. Nitrate was determined as Nitrite (NO_2) subtracted from total oxidised nitrogen (TON). Samples for TON, NO_2, NH_4 and major ions were frozen prior to analysis. pH and conductivity were measured on unfiltered samples. Alkalinity was measured on filtered samples by Gran titration. Filtered samples were analysed for calcium, magnesium, potassium and sodium using a Perkin-Elmer atomic absorption spectrophotometer. Soluble chloride and sulphate were measured by ion chromatography. Chlorophyll a was determined by extraction into hot methanol (Talling & Driver, 1963). Phytoplankton samples were preserved with acidified Lugol's iodine and enumerated from composite samples using an inverted microscope according to the Utermohl-Lund sedimentation technique (Lund et al., 1958). Zooplankton were preserved using 4% formaldehyde or 90% ethanol and enumerated using a Sedgewick-Rafter slide according to

Wetzel & Likens (1979). The following keys were used for identification: John, Whitton & Brook (2003), Scourfield & Harding (1966), Harding & Smith (1974), Pontin (1978) and Huber-Pestalozzi (1975).

River Sampling

The five largest rivers accounting for 77% of the catchment were monitored in all surveys with additional streams increasing coverage to >80% in each year (Table 1.). Rivers were sampled on the same day that lake monitoring took place. In 2001/02 and 2006/07 a stream at the village of Kinlough that receives effluent from a sewage treatment works was monitored in addition to the lake outflow (river Drowse). During 2001/02 and 2006/07 inflowing streams and rivers were sampled from the most downstream bridge or ford; these were all within a kilometre of the lake. Road closures around the lake in 1990 made vehicle access problematic and the rivers lying wholly within the Republic of Ireland were sampled by boat at the point at which they entered the lake.

Table 1. Grid references for river sampling points and year(s) in which they were sampled.

River	Irish Grid reference	Catchment area (km^2)	Years sampled
Muckenagh	G 918 543	10.4	1990, 2001/02, 2006/07
Roogagh	G 939 520	59.2	1990, 2001/02, 2006/07
County	G 937 507	56.1	1990, 2001/02, 2006/07
Ballagh	G 925 495	14.1	1990, 2001/02, 2006/07
Glenaniff	G 921 497	27.4	1990, 2001/02, 2006/07
Derrynaseer	G 889 559	2.4	2001/02
Breffni	G 876 531	1.95	2001/02
Kinlough	G 815 558	3.4	2001/02, 2006/07
Clancys	G 860 543	7.5	1990, 2006/07
Glen	G 938 515	2.1	1990
Drowse	G 832 567	240.2*	2001/02
Drumgormly[a]	H 020 485	7.37	2006/07
Glen East[a]	G 994 517	10.60	2006/07
Glen Bridge[a]	G 993 521	5.66	2006/07
Lattone[b]	G 947 469	9.25	2006/07
Deans[b]	G 978 454	2.17	2006/07
Sraduffy[b]	G 973 453	10.99	2006/07

[a] denote tributaries of the Roogagh river, [b] denote tributaries of the County river. * includes lake area of 22.7 km^2

Roogagh and County sub-catchment tributaries

In order to examine the effects of land use in greater detail three tributaries of the Roogagh and County rivers were monitored in 2006/07. These rivers were selected as they account for the greatest proportion of the catchment and together account for approximately half of the phosphorus and nitrogen entering the lake. Additionally they drain the most extensive areas of coniferous forestry in the catchment.

Tributary sampling points were located in upland areas and were selected in order to reflect as homogenous a land use type as possible. In this respect a compromise was achieved between desired land use characteristics and logistic constraints imposed by their accessibility (Fig. 2). On the Roogagh river the sampling point at Drumgormly principally drains the Big Dog and Tullyloughan Forests; that located at Glen East drains Dog Little and Conagher Upper, and the Glen Bridge sampling point mostly drains unimproved natural grassland and tracts of forest and peatland around Meenacloyabane and Tullyloughdaugh. On the County river the Lattone tributary drains areas of forestry, peatland and steep scrub below Saddle hill (this stream should not be confused with another that drains into Lattone Lough); the Sraduffy tributary, sampled at the village of Kiltyclogher, drains a variety of land uses from Dough Mountain down to the village, and Dean's tributary drains from Dean's Lough, an 8 hectare lake that drains areas of agricultural pasture and unimproved grassland from Scribbagh to Aghavanny. River water samples were analysed for pH, conductivity, alkalinity, major ions, phosphorus, nitrogen and soluble silica following the same protocols as for lake samples.

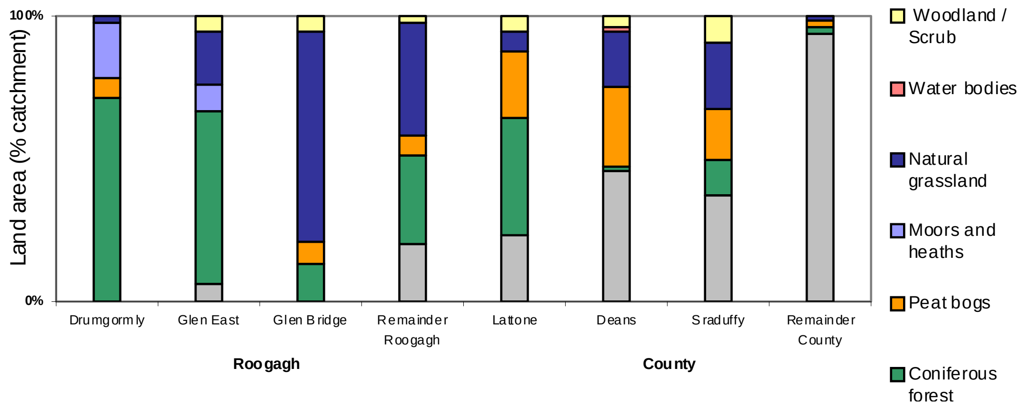


Figure 2. Percentage land use cover for tributary catchments of the Roogagh and County river sub-catchments (CORINE, 2000)

The office of public works (OPW) operates a hydrological station at Lareen bay on the river drowse just downstream of the lough. They maintain a level recorder on the lough and a flow recorder on the river Drowse (G832 567). In July 2001 the Rivers Agency of the Department of Agriculture and Rural Development installed a level recorder on the County river close to where it enters Lough Melvin, they also operate a level recorder on the Roogagh River. In November

2006 the Environmental Protection Agency (ROI) installed data loggers on the Glenaniff, Ballagh, Clancys, Breffni, and Kinlough rivers.

Nutrient Loading and Export Estimation: Comparison between monitoring periods

In order to examine changes in nutrient loading and nutrient export between monitoring periods a similar methodology was employed in 2006/07 to that used previously. In 1990 and 2001/02 sub-catchment monthly flows were estimated from the monthly total runoff from the OPW flow recorder on the outflow and scaled by each sub-catchment area assuming uniform runoff across the catchment. The contribution of small streams that were not monitored was estimated by calculating mean catchment nutrient export rates and scaling these up by the unmonitored area of the catchment.

At the time of completion of this report flow data for 2006/07 from the outflow were unavailable. 5 major rivers were gauged for flows for the duration of the monitoring period from which mean monthly catchment runoff rates ($m^3\ ha^{-1}$ or mm) were calculated. These were applied to data using the same methodology as that employed in 1990 and 2001/02.

Annual sub-catchment loadings were estimated as follows:

$$\mathbf{L^{SC} = \sum (Q^{n}_{W}\, C^{n}_{M}) / 10^6}$$

where Q^{n}_{W} = inflow to L. Melvin during month *n* (m^3) for river

C^{n}_{M} = mean concentration during month *n* for river ($g\ m^{-3}$)

L^{SC} = annual sub-catchment nutrient load (tonnes)

Sub-catchment export rates were calculated as:

$$\mathbf{M^{SC}_{EXPORT} = L^{SC} / A^{SC} * 10^3}$$

where M^{SC}_{EXPORT} = sub-catchment export rate ($kg\ ha^{-1}\ yr^{-1}$)

A^{SC} = sub-catchment area (ha)

To estimate the contribution of small streams a mean catchment nutrient export rate was combined with the unmonitored catchment area:

$$\mathbf{M^{C}_{EXPORT} = \sum L^{SC} / \sum A^{SC} * 10^3}$$

$$\mathbf{L^{UM} = M^{C}_{EXPORT} * A^{UM} / 10^3}$$

Where M^{C}_{EXPORT} = Catchment export rate ($kg\ ha^{-1}\ yr^{-1}$)

A^{UM} = unmonitored catchment area (ha)

L^{UM} = loading from un-monitored catchment area (tonnes)

To account point sources of nutrients to the lake from sewage treatment works and to distinguish between nutrient loadings from rural and urban sources (where WWTW's discharge into a catchment river) an annual *per capita* nutrient loading estimate is combined with the most recent population census data. This assumes that each person produces 0.766 kg total phosphorus, 0.64

kg total soluble phosphorus, 0.56 kg soluble reactive phosphorus and 2.4 kg of Nitrogen (nitrate + ammonia) each year.

The total nutrient load to the lake is then calculated as:

$$\mathbf{L^{T} = (\sum L^{SC}) + L^{UM} + L^{PS}}$$

where: $\mathbf{L^{T}}$ **=** total nutrient load (tonnes)
$\mathbf{L^{PS}}$ **=** loading from point source direct discharges (tonnes)

Nutrient Loading and Export Estimation 2006/07

The addition of flow gauges to 5 routinely monitored rivers covering 73% of the catchment area in 2006/07 allowed greater accuracy in estimating the nutrient loading. For the Roogagh and the County rivers (> 50% catchment area) that lie within Northern Ireland flow data was available at 15 minute intervals. For the Glenaniff, Ballagh and Clancy's rivers that lie within the Republic of Ireland mean daily flows were available. GIS delineation of sub-catchment boundaries was revised in 2006/07 to better reflect true boundaries. Given the greater resolution available, revised nutrient budgets were calculated by according to 2 methods.

<u>Method 1: Loading and export rate calculation by mean monthly concentration</u>

Sub-catchment loadings were calculated as:

$$\mathbf{L^{SC}} = \sum (\mathbf{Q}^{n}{}_{\mathbf{W}}\, \mathbf{C}^{n}{}_{\mathbf{M}}) \ \mathbf{* 10^6}$$

where $\mathbf{Q}^{n}{}_{\mathbf{W}}$ = inflow to L. Melvin during month n (m^3) for river,
$\mathbf{Q}^{n}{}_{\mathbf{W}} = \sum \mathbf{(Q^{n}_{XD} * K)}$
$\mathbf{Q_{XD}}$ = mean daily flow for month n
$\mathbf{K}$ = conversion factor for period, 24*60*60 (hours, minutes, seconds)
$\mathbf{C}^{n}{}_{\mathbf{M}}$ = mean concentration during month n for river ($g\ m^{-3}$)
$\mathbf{L^{SC}}$ = annual sub-catchment nutrient load (tonnes)

Sub-catchment export rates were calculated as:

$$\mathbf{M^{SC}_{EXPORT} = L^{SC} / A^{SC} * 10^3}$$

where $\mathbf{M^{SC}_{EXPORT}}$ = sub-catchment export rate (kg $ha^{-1}\ yr^{-1}$)
$\mathbf{A^{SC}}$ **=** sub-catchment area (ha)

Loadings from direct point source discharges ($\mathbf{L^{PS}}$) were calculated by the methodology described previously. Loadings from un-monitored areas of the catchment were estimated using nutrient export rates for specific sub-catchments considered to be most representative. On this basis the un-monitored areas of the catchment were divided into 7 areas with export rates calculated from gauged sub-catchments assigned as follows:

- North-west Muckenagh sub-catchment export rate

- North-east Roogagh sub-catchment export rate
- South-east County sub-catchment export rate
- South-west Clancy's sub-catchment export rate
- Islands Clancy's sub-catchment export rate
- Breffni sub-catchment Clancy's sub-catchment export rate
- Derrynaseer sub-catchment Muckenagh sub-catchment export rate

The total loading to the lake is calculated by summing each respective component:

$$\mathbf{L^{T} = (\sum L^{SC}) + L^{UM} + L^{PS}}$$

where: $\mathbf{L^{T}}$ = total nutrient load (tonnes)
$\mathbf{L^{PS}}$ = loading from point source direct discharges (tonnes)
$\mathbf{L^{UM}}$ = loading from un-monitored catchment areas (tonnes)

Method 2: Loading and export rate calculation by nutrient concentration and mean flow on the day of sampling

Sub-catchment loadings were estimated using a flow-weighted mean concentration:

$$\mathbf{FWMC = \sum (C_i\, Q_i) / \sum Q_i}$$

where $\mathbf{FWMC}$ = flow weighted mean concentration (g m^{-3})
$\mathbf{C_i}$ = concentration on day of sampling (g m^{-3})
$\mathbf{Q_i}$ = flow on day of sampling, (m^3 sec^{-1}, mean daily flow)

The annual riverine / sub-catchment nutrient load is then the product of the FWMC and the total annual flow:

$$\mathbf{L^{SC} = FWMC * Q_r / 10^9}$$

where $\mathbf{L^{SC}}$ = annual sub-catchment nutrient load (tonnes)
$\mathbf{Q_r}$ = Annual flow (m^{-3})

Sub-catchment export rates are then the annual sub-catchment load divided by the sub-catchment area:

$$\mathbf{M^{SC}_{EXPORT} = L^{SC} / A^{SC} *10^3}$$

where $\mathbf{A^{SC}}$ = sub-catchment area (ha)
$\mathbf{M^{SC}_{EXPORT}}$ = sub-catchment export rate (kg ha^{-1} yr^{-1})

Loadings from direct point source discharges ($\mathbf{L^{PS}}$) and loadings from un-monitored areas of the catchment ($\mathbf{L^{UM}}$) were estimated using the same methodology used in method 1.

Loading and Export within the Roogagh and County sub-catchments

Sampling three tributaries within the Roogagh and County Rivers combined with the samples at their point of inflow to Lough Melvin divided these catchments into four discrete areas. Flow data for these rivers was recorded at the inflow at 15 minute intervals allowing the use of instantaneous flow data. The methodology used for the rivers was employed with the mean daily flow on the day of sampling substituted for the flow to the closest 15 minutes. Flows within each tributary catchment were then estimated as the proportion of the flow at inflow by area assuming uniform precipitation over each catchment. To calculate loadings and exports for the downstream area of the catchment between the inflow and the tributary sampling points, the loading and flows from the tributary catchments were summed and subtracted from the total loading and flow for the entire river at the downstream sample point.

Results

For an overview of the limnology, location, climate, morphology, geology and soils please refer to Girvan & Foy (2003, 2006).

Lake and catchment hydrology

Between 1975 and 2002 runoff from the lake and catchment averaged 1100 mm $year^{-1}$ but during the 1990, 2001/02 and 2006/07 monitoring periods runoff was above average at 1402 mm, 1277 mm and 1314 mm respectively so that water retention times were below average at 0.81, 0.89 and 0.87 years for each of these period respectively. Based on flows measured In 2006/07 the distribution of effective precipitation (rainfall less evaporation) across the catchment shows an altitudinal effect with those rivers draining land on the high southern shore receiving a greater proportion of the rainfall (Table 2).

Table 2. Annual precipitation in 2006/07 for 5 major sub-catchments.

Sub-catchment	Annual Precipitation (mm)
Glenaniff	1580
Ballagh	1349
Clancys	1331
County	1112
Roogagh	1112

Temperature

During each monitoring period the water column failed to develop any well defined, prolonged thermal stratification (Fig. 3). During the summer small but significant differences in temperature were observed between the surface and bottom (Table 3). These are calm periods of surface water warming with subsequent water column mixing that reflect the polymictic character of Lough Melvin. Maximum temperature differences between the surface and bottom waters were 6ºC, 3.7ºC and 5ºC in 1990, 2002 and 2007 respectively and are small in terms of the relative thermal resistance generated.

Table 3. Mean surface and bottom temperatures (°C) for all years monitored. The range is in parentheses. ∆t May-August is the mean difference in temperature between the surface and bottom for May through August. Significant difference from zero is indicated as: *p < 0.1, ** p < 0.05, *** p < 0.01

	1990	**2001/02**	**2006/07**
Mean surface temperature	10.7 (4.4 - 17.3)	12.2 (5.4 - 18.5)	11.8 (6.1 - 19.1)
Mean bottom temperature	9.7 (4.4 - 15.3)	10.49 (5.2 - 16.1)	10.99 (6.1 - 16.6)
∆t May-August	2.4***	1.4*	1.8**

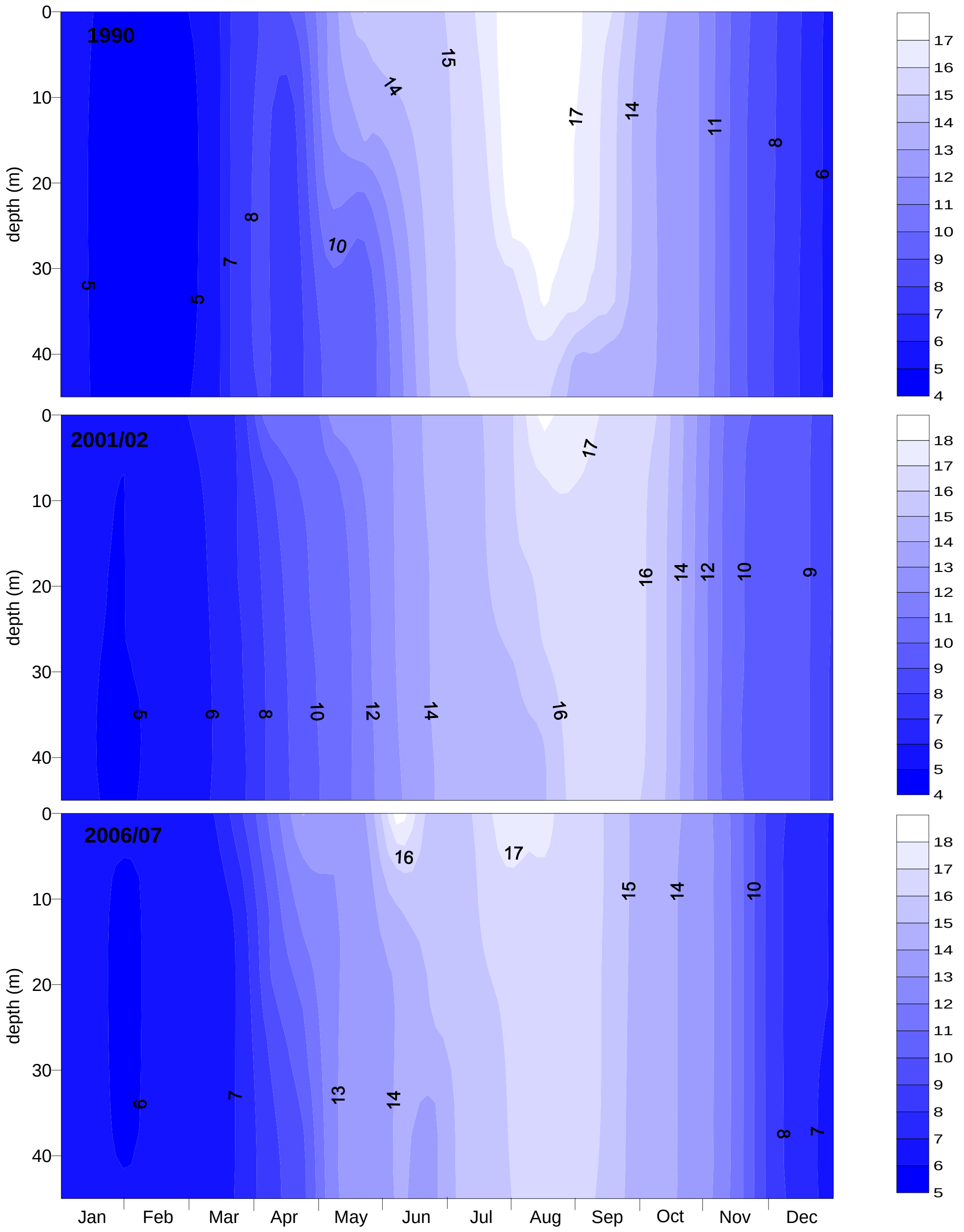


Figure 3. Annual depth-time isomers (ºC) at site 1 in 1990, 2001/02 and 2006/07.

<u>Summer temperatures in the deep water basin 2007</u>

In June and July 2007 the weather consisted largely of calm, clear weather. Between early May and August surface water temperatures rose 6 ºC to an annual maximum of 19.1ºC (Fig.3). At 38 metres depth a 1ºC increase over the same period demonstrates the development of a definite thermal structure (Fig. 4). A short period of poor weather and high winds in early July mixed the water column resulting in a uniform temperature profile at 15.5ºC and re-oxygenated the deeper waters. Calm weather throughout the remainder of July and into early August again caused surface water warming with a 2 and 0.5 ºC increase at the surface and 38m respectively. Higher winds and deteriorating weather from the 6th August onwards again mixed the water column giving a uniform temperature of 16.8 ºC throughout (Fig. 4).

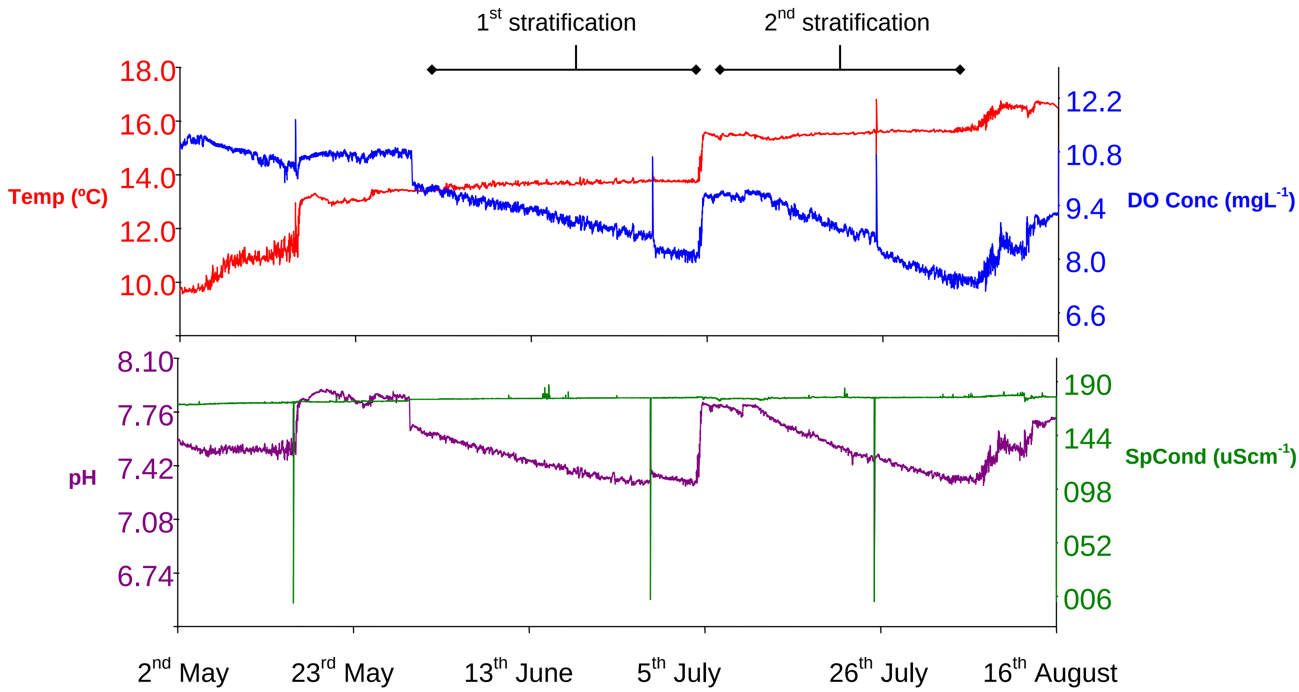


Figure 4. Temperature, dissolved oxygen, pH and specific conductivity at 38m depth (Site 1; Z_{max} = 45m) recorded between 2nd May '07 and 16th August '07. Measurements were taken at 10-minute intervals. (Sharp peaks recorded for specific conductance, dissolved oxygen and temperature correspond to recalibration events where the sonde was raised to the surface)

Dissolved oxygen

Dissolved oxygen (DO) levels were high during each monitoring survey, typically > 90% saturation (Table 4) with frequent mixing of the entire water column preventing serious depletion of DO in the deeper waters. The record from the sonde in 2007 revealed two distinct periods of appreciable DO depletion when thermal stratification effectively isolated the deeper waters from the mixed, oxygenated surface waters. The first was between late May and early July and the second from early July to early August. Over these periods respiratory consumption of DO in the deep waters was relatively constant and resulted in steady decreases in DO. The first period of stratification lasted 39 days and the second 26 days during which respiratory oxygen consumption occurred at a rate of 0.058 and 0.094 mg O_2 L^{-1} day^{-1} respectively. The reasons for the higher respiration rate observed during the second stratification period are unknown but may have simply reflected the increase in temperature that followed lake mixing in July combined with the high inputs of organic carbon from the catchment that resulted from the usually high summer rain and flows that accompanied the weather systems that led to the lake mixing. If the lake had not mixed in July 2007, when DO increased by approximately 1.5 mg O_2 L^{-1}, then an uninterrupted stratification to early August would have resulted in a minimum DO approaching the 6 mg O_2 L^{-1} limit for salmonids. Alternatively the first stratification period would have had to have continued for 83 days respectively for DO concentrations to fall to 6 mg O_2 L^{-1}.

Table 4. Mean surface and bottom dissolved oxygen saturation (%) for all years monitored. The range is given in parentheses**.**

	1990	2001/02	2006/07
Mean dissolved oxygen at surface	103.0 (94 – 108)	95.8 (89.2 – 101.5)	96.7 (88.6 – 104.0)
Mean Dissolved Oxygen at bottom	97.0 (92.8 – 105)	91.6 (89.5 – 107.3)	90.2 (75.4 – 98.3)

Secchi Depth

Mean Secchi disk depths lie within the OECD eutrophic category in all years due to light attenuation by humic substances rather than phytoplankton (Table 5). Minimum values have been observed in early spring in each survey following higher runoff during the winter months that deliver greater amounts of dissolved and particulate material to the lake (Fig. 5). The minimum Secchi depth observed in 2006/07 was 0.45m deeper than in observed in 1990 and 2001/02.

Table 5. Mean summer, winter and annual Secchi disk depths in metres for all years monitored.

	1990	2001/02	2006/07	Difference
Mean winter	1.99	1.74	1.71	NS
Mean summer	2.60	1.78	2.11	$p < 0.001$, $p < 0.025$, $p < 0.01$
Mean annual	2.27	1.71	1.91	$p < 0.01$, $p < 0.05$, $p < 0.05$

Significant differences are denoted: in red for between 1990-2001/02; in blue for between 1990-2006/07, in black for between 2001/02-2006/07.

The OECD trophic classification of lakes by Secchi depth is generally accurate as phytoplankton tend to dominate the particulate pool of clear water lakes. In these situations Secchi depths are negatively correlated with measurements of chlorophyll a. However in dystrophic lakes rapid light attenuation by humic compounds destroys the relationship, particularly in deeper polymictic lakes. Correlations between chlorophyll concentration and Secchi depth have never been observed in Lough Melvin, indicating that algae do not contribute significantly to light attenuation.

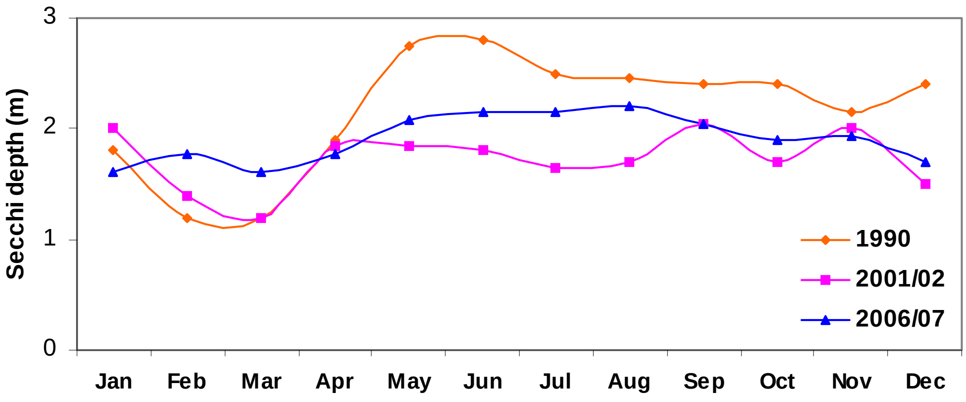


Figure 5. Annual Secchi depth cycles at site 1 for 1990, 2001/02 and 2006/07.

Major Ion Chemistry

No appreciable change in the major ion chemistry of Lough Melvin has been observed during the course of the three monitoring surveys (Table 6). Humic or dystrophic lakes generally tend to have acidic waters due to the influence of drainage water originating from peatlands. In contrast Lough Melvin and other humic stained lakes in the region are alkaline due to a high proportion of carboniferous rock in the catchment. Interestingly pH was significantly higher in 2006/07 than in 1990 and 2001/02 (paired t-test; $p < 0.01$ & < 0.05 respectively).

Table 6. Major ions (mg L^{-1}), pH (-log [H^+]) and conductivity (µS cm^{-1}) for all years

	1990	2001/02	2006/07
Ca^{2+}	22.0	23.6	23.2
Mg^{2+}	3.2	3.1	3.0
Na^{+}	10.7	8.3	9.0
K^{+}	1.0	1.2	1.0
SO_4^{2-}	Na	14.8	15.6
Cl^{-}	Na	5.3	6.9
pH	7.72	7.85	7.99
Conductivity	197	169	165

Note: Data for 2001/02 and 2006/07 are annual means. Dat for 1990 based on an individual sample taken in February 1991.

Nutrients and trophic status

Chlorophyll a

Mean chlorophyll a concentrations were not significantly different between 1990 and 2001/02 but significantly lower in 2006/07 compared with both previous surveys (paired t-test; $p < 0.005$) (Table 7). Peaks corresponding to the spring and summer blooms are apparent in each year but concentrations were considerably lower in 2006/07 (Figs. 6 & 7).

Table 7. Mean and maximum chlorophyll a concentrations (µg L^{-1}) for all years monitored.

	1990	2001/02	2006/07
Mean chlorophyll a	4.79	4.83	2.68
Maximum chlorophyll a	13.5	10.63	6.84

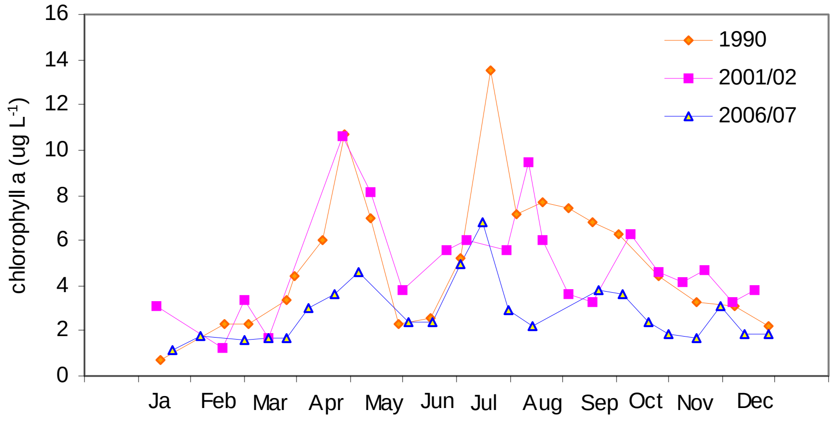


Figure 6. Annual cycle of chlorophyll a for 1990, 2001/02 and 2006/07 (based on composite samples).

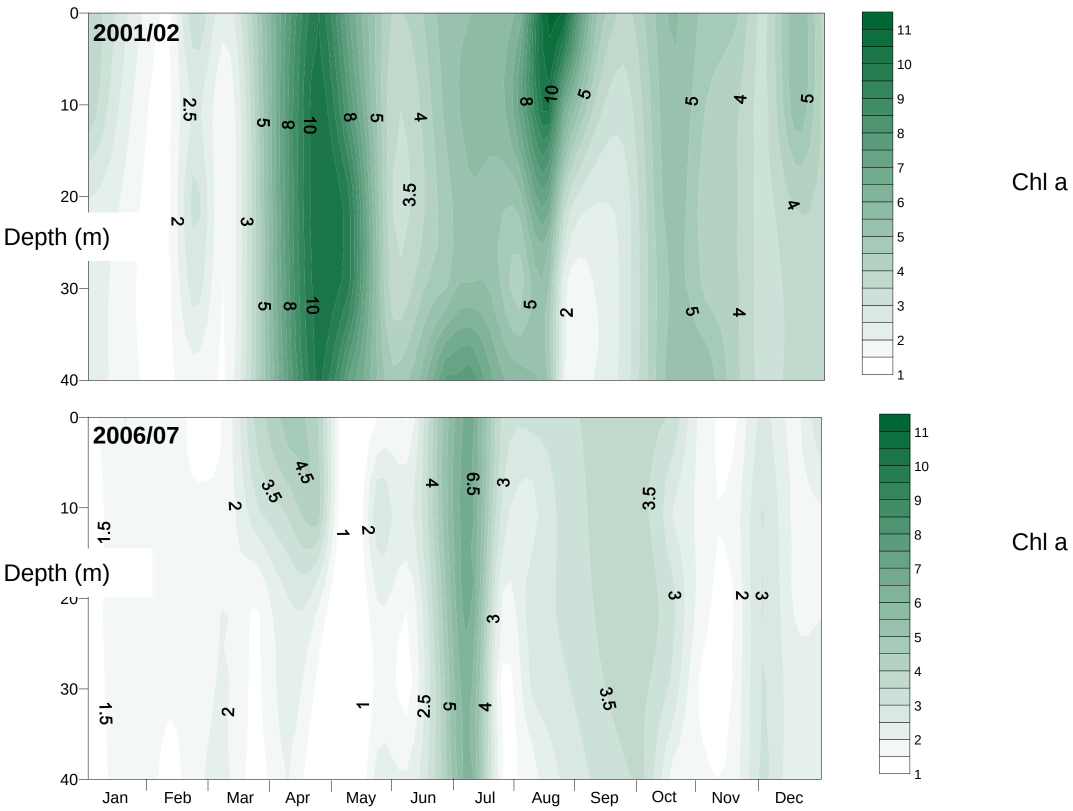


Figure 7. Depth-time distribution of chlorophyll a (µg L^{-1}) during 2001/02 and 2006/07

Soluble silica

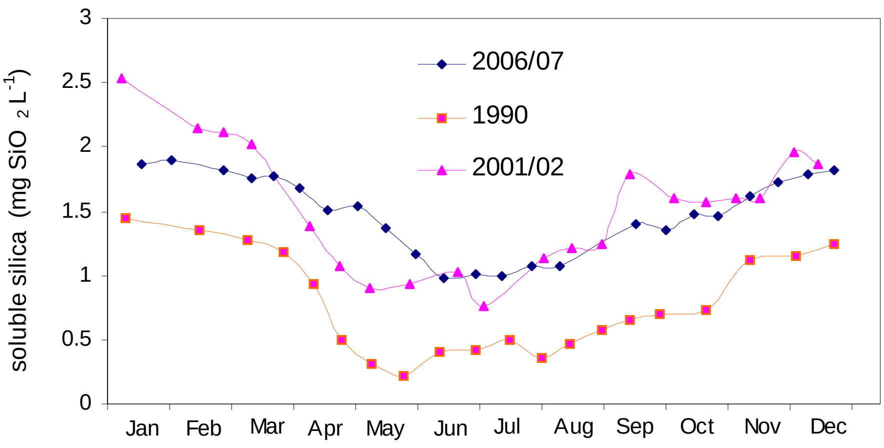


Figure 8. Annual soluble silica cycle for 1990, 2001/02 and 2006/07

Silica concentrations were significantly higher in 2006/07 and 2001/02 than in 1990. On an annual basis concentrations between 2001/02 and 2006/07 were not significantly different. From January to March, a period characterised by peak runoff, concentrations were significantly higher than in 2001/02 than 2006/07 (paired t-test; $p < 0.05$).

Annual Nutrient Cycles

Concentrations of nitrate and ammonia have remained consistent between surveys whereas phosphorus concentrations in 2001/02 and 2006/07 were significantly higher than in 1990 (Figs 9 & 10). Whilst mean concentrations of each phosphorus fraction decreased between 2001/02 and 2006/07 the lack of a significant difference between the two periods is a cause for concern (Table 7).

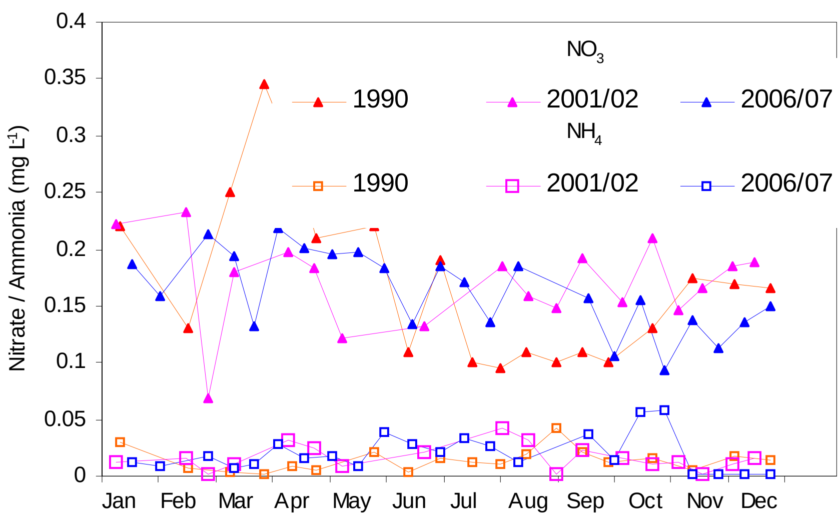
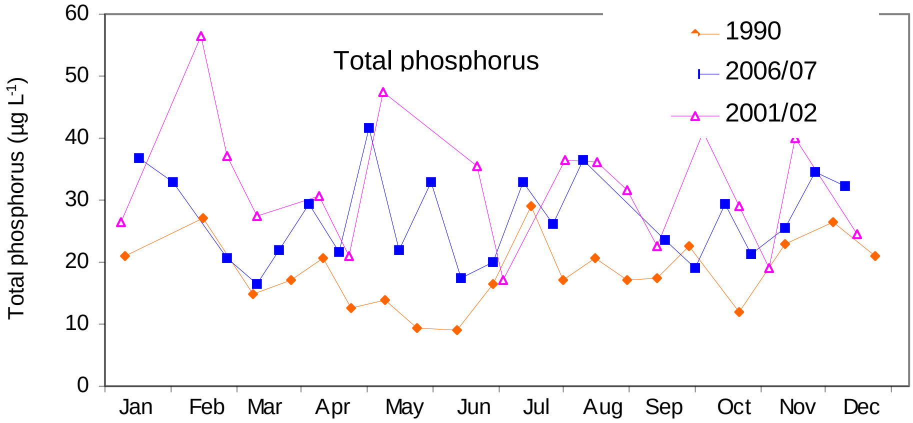


Figure 9. Annual cycles of Nitrate (NO_3) and Ammonia (NH_4) for all years



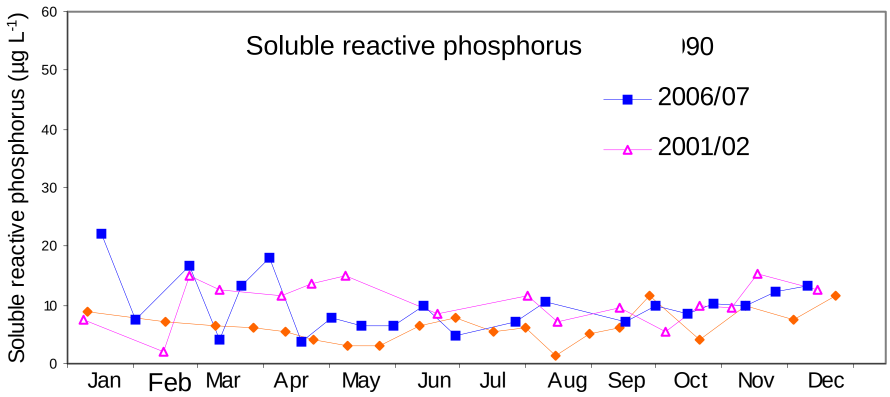


Figure 10. Annual cycles of total phosphorus and soluble reactive phosphorus**.**

Trophic status

Mean total phosphorus concentrations in Lough Melvin are currently within the limits for mesotrophic lakes (Table 8 and Fig 11). Mean and maximum chlorophyll values have consistently indicated mesotrophic status except in 2006/07 when concentrations were in the oligotrophic class for lakes. In Table 4 statistically significant differences between years are denoted in red for the comparison between 1990 vs 2001/02, blue for 1990 vs 2006/07, and black for 2001/02 vs 2006/07. Concentrations of TP, total soluble phosphorus and soluble reactive phosphorus were significantly lower in 1990 compared to either 2001/02 or 2006/07. No difference was observed for the phosphorus fractions measured in 2001/02 and 2006/07. Neither nitrate nor ammonium showed any difference between years.

Table 8. Mean annual phosphorus, nitrogen and silica concentration for 1990, 2001/02 and 2006/07.

	1990	2001/02	2006/07	Significance level
Total phosphorus (µg P L^{-1})	19	30	27	$p<0.005$, $p<0.05$, NS
Total soluble phosphorus (µg P L^{-1})	13	21	18	$p<0.005$, $p<0.005$, NS
Soluble reactive phosphorus (µg P L^{-1})	7	11	10	$p<0.01$, $p<0.025$, NS
Nitrate (mg N L^{-1})	0.17	0.19	0.16	NS, NS, NS
Ammonia (mg N L^{-1})	15	20	20	NS, NS, NS
Soluble silica (mg SiO_2 L^{-1})	0.84	1.70	1.48	$p<0.001$, $p<0.001$, NS

Note: Significant differences between years calculated by paired t-test are denoted in red for between 1990-2001/02; blue for between 1990 and 2006/07, black for between 2001/02 and 2006/07, NS = no significant difference.

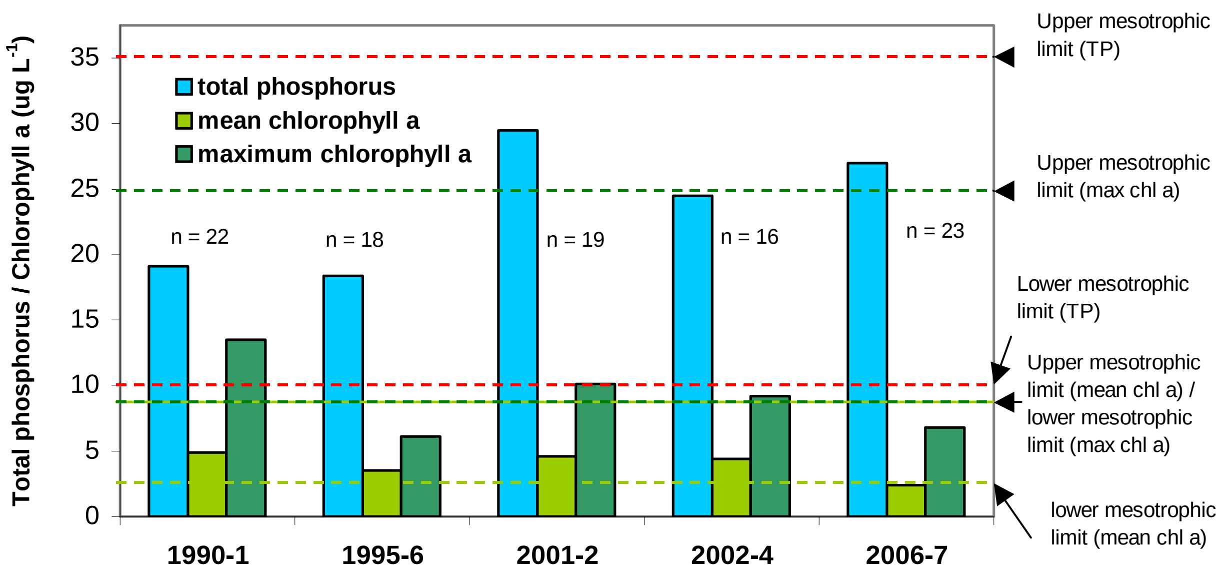


Figure 11. Mean annual total phosphorus, mean anual chlorophyll a and maximum chlorophyll a concentrations recorded since 1990. Upper and lower limits for mesotrophic classification based upon the OECD scheme (1982) are shown.

Nutrient Budget

Nutrient Loadings: Comparisons between monitoring periods

Table 9 summarises annual nutrient loadings from the monitored catchments. The County River receives nutrients from the waste water treatment works (WWTW) at Kiltyclogher and these are included in the values presented. The WWTW at Garrison discharges directly into the lake but the discharge from the WWTWs that serves the village of Kinlough is included in the loading of the Kinlough stream. At the whole catchment scale SOP loss decreased between

2001/02 and 2006/07 while PP and SRP loss increased, raising phosphorus loss (TP) overall (Figure 12). Loss intensities of all phosphorus fractions from each catchment area were greater in 2006/07 than observed in 1990. Although loadings are dependent upon flows, the values presented are comparable due to the similar amounts of precipitation observed during each period. In contrast nitrate plus ammonium inputs to the lake declined.

Figures 12-15 show the annual loading of each phosphorus fraction and Figure 16 shows the loadings of nitrate and ammonia from monitored sub-catchments in 1990, 2001/02 and 2006/07.

Table 9. Annual loading (tonnes) of soluble reactive phosphorous (SRP), total soluble phosphorus (TSP), total phosphorus (TP), particulate phosphorus (PP), soluble organic phosphorus (SOP), nitrate (NO_3-N) and Ammonia (NH_4-N) to Lough Melvin from sub-catchments in 1990, 2001/02 and 2006/07. * denotes NO_3-N + NH_4-N.

River	Year	SRP	TSP	TP	PP	SOP	NH_4-N	NO_3-N
Roogagh	1990	1.24	2.19	2.67	0.48	0.95	2.13	10.37
	2001-02	1.35	2.34	2.98	0.64	0.99	5.03	13.86
	2006-07	1.39	2.44	3.13	0.69	1.05	2.84	10.08
County	1990	1.35	2.24	3.20	0.96	0.89	3.42	22.14
	2001-02	1.13	2.24	2.95	0.71	1.11	1.40	18.07
	2006-07	1.49	2.39	3.63	1.24	0.90	1.65	12.54
Glenaniff	1990	0.38	0.47	0.71	0.24	0.09	0.68	9.82
	2001-02	0.41	0.74	1.06	0.32	0.33	0.68	9.78
	2006-07	0.81	0.96	1.27	0.31	0.15	0.80	7.70
Ballagh	1990	0.18	0.28	0.42	0.14	0.10	0.38	6.43
	2001-02	0.22	0.41	0.52	0.11	0.19	0.49	5.44
	2006-07	0.35	0.53	0.70	0.17	0.19	5.38	0.40
Clancys	1990	0.02	0.03	0.04	0.01	0.01	0.03	0.61
	2001-02							
	2006-07	0.10	0.16	0.24	0.09	0.06	0.16	1.48
Muckenagh	1990	0.26	0.45	0.60	0.15	0.19	0.16	3.90
	2001-02	0.26	0.48	0.62	0.14	0.21	0.33	2.71
	2006-07	0.25	0.45	0.63	0.18	0.20	0.28	1.62
Glen	1990	0.32	0.49	0.64	0.15	0.17	0.43	2.28
Brefni	2001	0.02	0.03	0.08	0.05	0.01	0.03	0.42
Derrynaseer	2001/02	0.06	0.12	0.15	0.03	0.06	0.07	0.81
Kinlough	2001/02	0.18	0.31	0.42	0.11	0.13		1.92*
	2006/07	0.20	0.27	0.40	0.13	0.07		1.46*

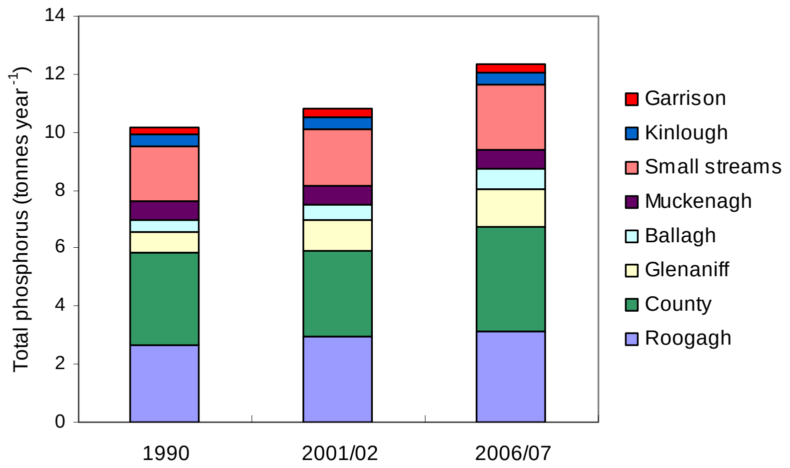


Figure 12. Annual Total phosphorous loading to Lough Melvin from sub-catchments.

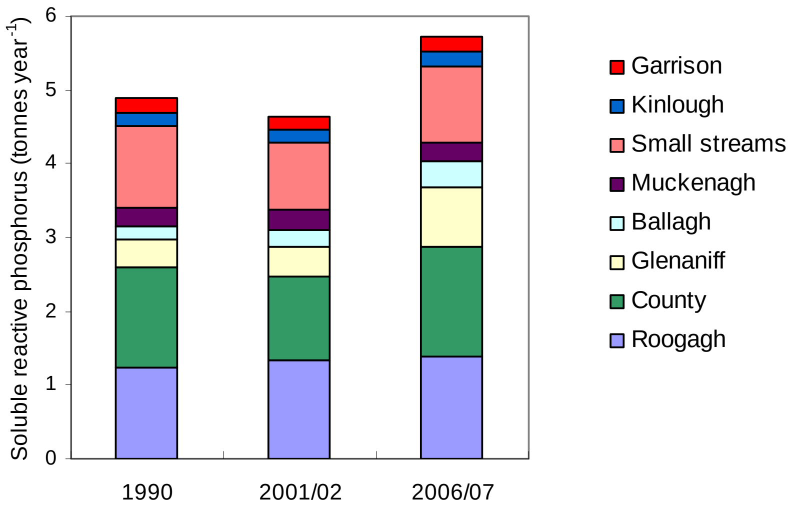


Figure 13. Annual soluble reactive phosphorous loading to Lough Melvin from sub-catchments.

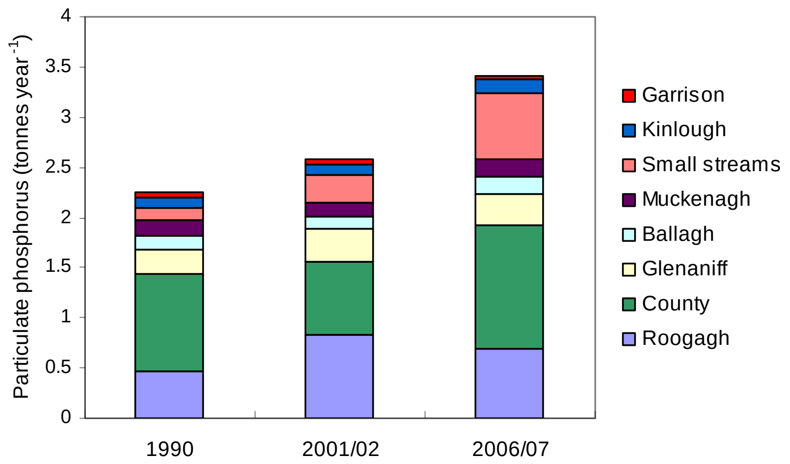


Figure 14. Annual particulate phosphorous loading to Lough Melvin from sub-catchments.

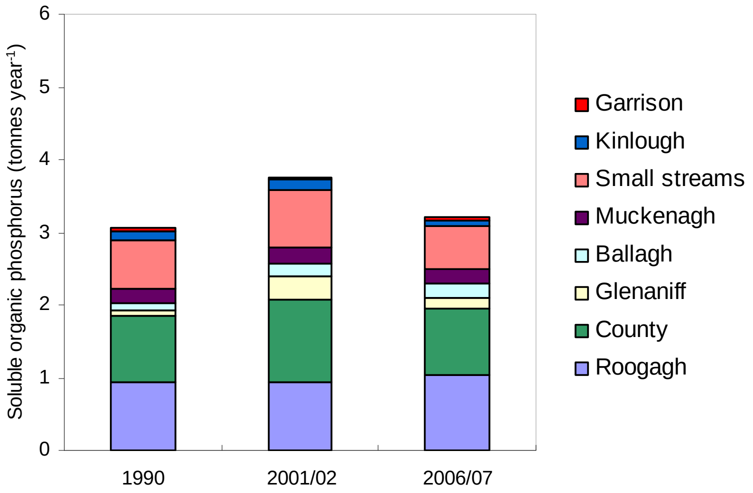


Figure 15. Annual soluble organic phosphorous loading to Lough Melvin from sub- catchments

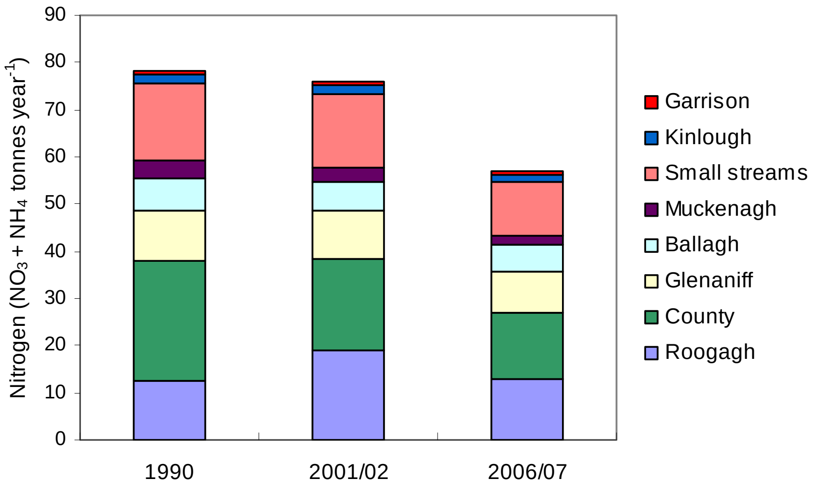


Figure 16. Annual nitrogen (NO_3 + NH_4) loading to Lough Melvin from sub- catchments

Catchment flow-weighted mean concentration

A flow-weighted mean concentration accounts for the effect of differences in precipitation upon nutrient loading between periods and is calculated by dividing the total loading by the total runoff. Decreases in flow-weighted soluble organic phosphorus (SOP) export observed between 2001/02 and 2006/07 were overshadowed by increases in the flow-weighted exports of soluble reactive phosphorus (SRP) and particulate phosphorus (PP), resulting in greater flow-weighted exports of total phosphorus (TP) overall (Fig 17). This indicates an increase in the intensity of SRP and PP loss over time within the catchment.

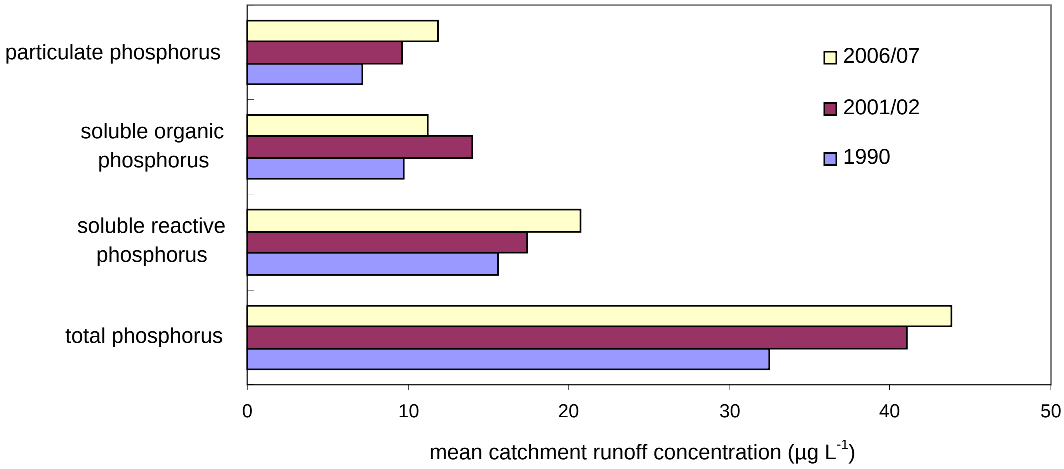


Figure 17. Mean runoff concentration of phosphorus fractions (µg L^{-1}) from the Lough Melvin catchment in 1990, 2001/02 and 2006/07

Nutrient export rates: Phosphorus

At the whole catchment scale (Fig. 17) SOP loss decreased while PP and SRP loss increased, raising phosphorus loss (TP) overall. At higher resolution within the sub-catchments it is apparent that certain areas contribute more than others to the overall trend than others. Loss intensity of all phosphorus fractions from each sub-catchment remains higher in 2006/07 than that observed in 1990 (Fig. 20).

Summary of changes in the intensity of phosphorus loss: 2001/02 & 2006/07

● No change in the intensity of SRP, SOP or PP loss from the Roogagh sub-catchment was observed.

● The intensity of SOP loss decreased in all other monitored sub-catchments.

● The intensity of PP loss increased in all sub-catchments except that of the Ballagh which did not change.

● Startling increases in the loss intensity of SRP from the Glenaniff and Ballagh sub-catchments were observed. The flow-weighted mean SRP concentration in these catchments increased by 88% and 47% respectively between 2001/02 and 2006/07 and 28% and 40% between 1990 and 2001/02. Within the Glenaniff catchment the increase of TP loss intensity was solely due to the increase in SRP loss. Within the Ballagh sub-catchment increases in the intensity of SRP and PP loss accounted for 71% and 31% of the increase in TP loss respectively.

● The intensity of SRP loss from the Muckenagh catchment decreased to a similar level to that observed in 1990. Coupled with the decrease SOP loss intensity and a slight increase in PP loss intensity this was the only sub-catchment to show a reduction in the intensity of TP loss since 2001/02.

● Loss intensities of SRP and PP from the County sub-catchment, that were similar in 1990 and 2001/02, increased markedly with 25% and 65% greater export intensities respectively.

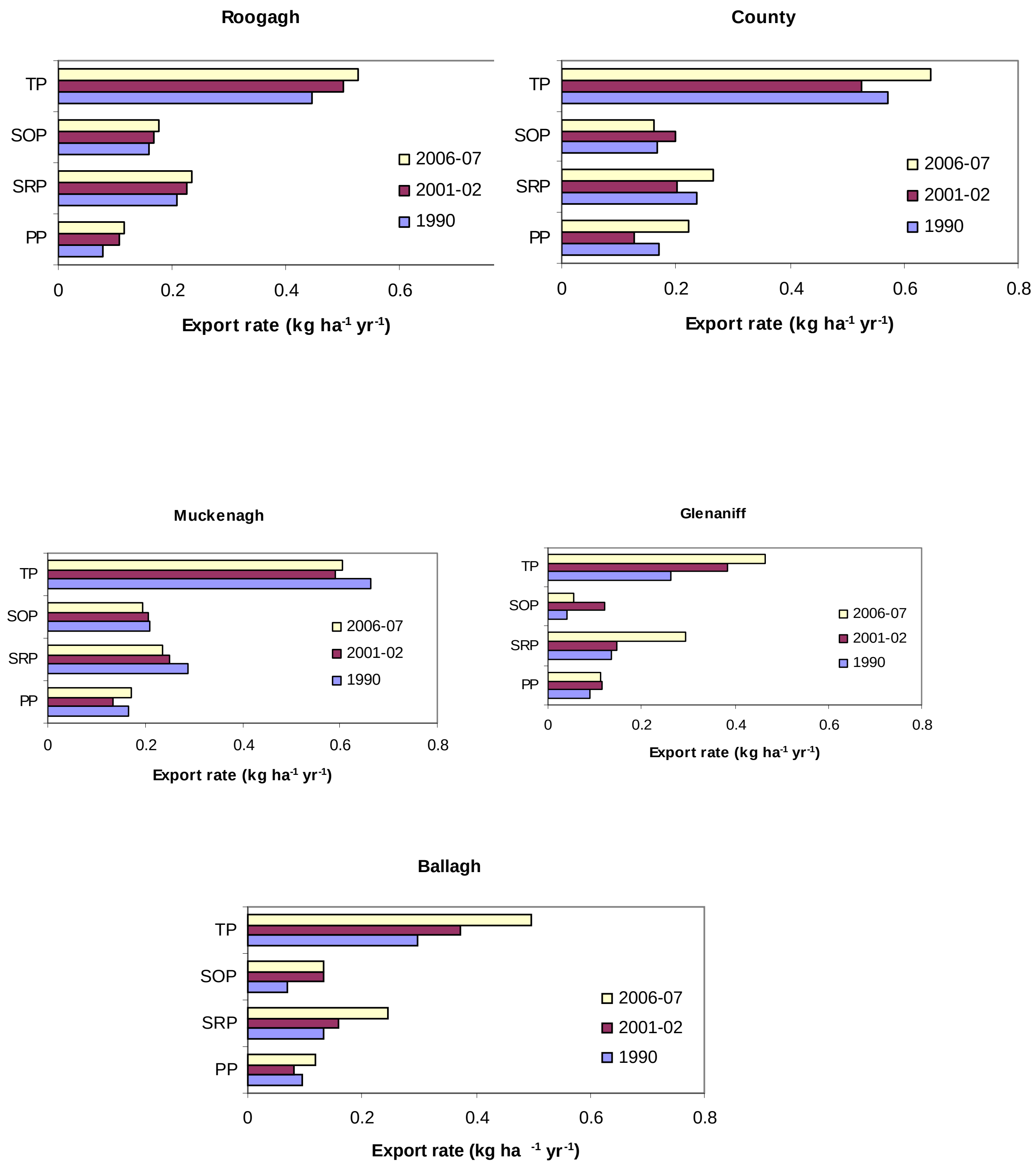


Figure 18. Sub-catchment export rates of total phosphorus (TP), soluble organic phosphorus (SOP), soluble reactive phosphorus (SRP) and particulate phosphorus (PP) for 1990, 2001/02 and 2006/07.

Nutrient export rates: Nitrate and Ammonia

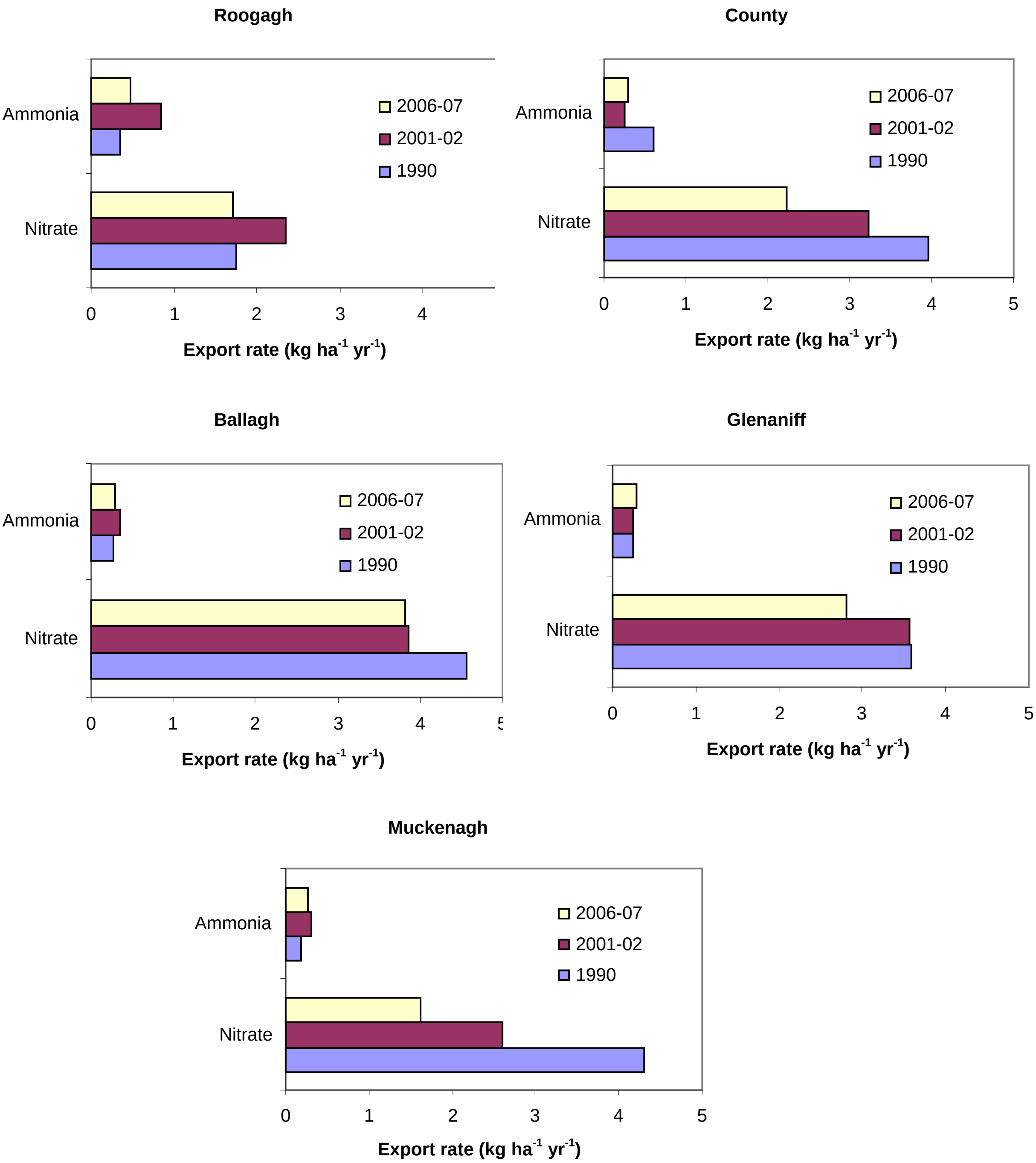


Figure 19. Sub-catchment export rates of Nitrate and Ammonia for 1990, 2001/02 and 2006/07

Sub-catchment flow-weighted mean concentration

Export rates correct for differences in loading due to variations in catchment size however they do not account for the influence of flow on nutrient loss. Flow weighted mean concentrations, which are the annual load divided by the annual flow, standardise export rates by correcting for different levels of precipitation between monitoring periods (Fig. 20).

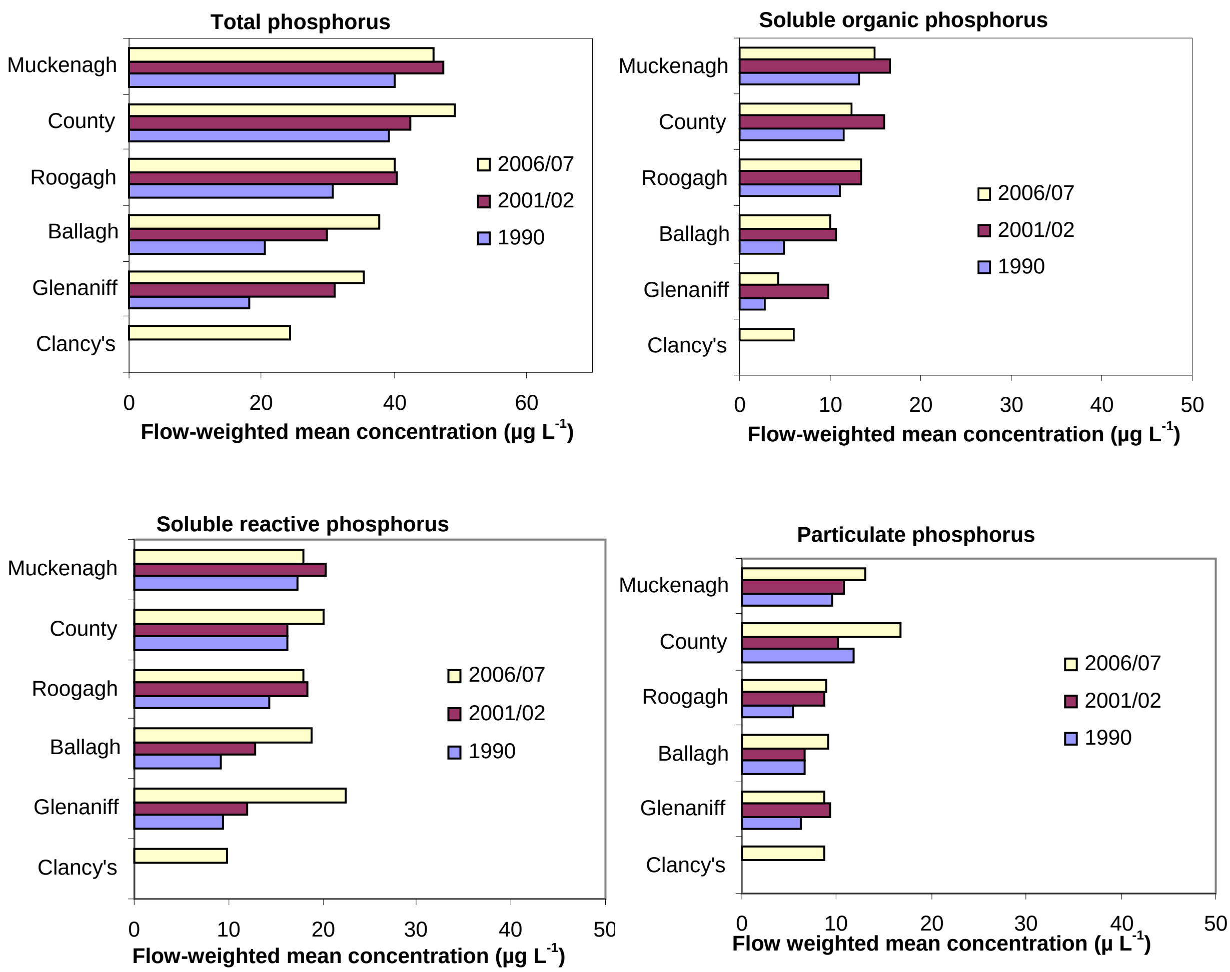


Figure 20. Flow weighted mean concentration of phosphorus fractions by sub-catchment

Kinlough WWTW 2001/02 & 2006/07

The small stream into which the Kinlough WWTW discharges effluent was monitored in 2001/02 and 2006/07 (Table 10). The census of 2001/02 recorded a population in the village of 300 persons, but by 2006/07 this had more than doubled to 626 persons. Loadings from the WWTW can be estimated by multiplying the population by the per capita loadings for each of the phosphorus fractions. These values can then be compared with exports of phosphorus from the stream that are based on the water samples taken. Given that the phosphorus export from the stream would include phosphorus from, for example, agricultural land and septic tanks, it would be expected that the measured stream export of phosphorus should be larger than the estimated export of phosphorus from the WWTW discharge. Such a difference was observed in

2001/2 when the WWTW contributed only 55% of the measured TP export from the stream. However, despite the increased population of Kinlough by 2006, the measured annual TP export from the stream declined slightly from 0.42 to 0.40 tonnes in 2006/07. The most probable explanation for the failure of the stream loading to respond the increased population of Kinlough was the implementation by Leitrim County Council of phosphorus removal at the Kinlough WWTW in 2006. That the reduction in stream loading was not larger as a result of phosphorus removal is likely to be a reflection of the fact that the WWTW remains overloaded.

Table 10: Loading (tonnes year^{-1}) of phosphorus fractions to Lough Melvin from the Kinlough stream in 2001/02 and 2006/07 according to observed concentrations and calculations by capita contribution.

Year	2001/02		2006/07	
	per capita	Stream load observed	*per capita*	Stream load observed
SRP	0.168	0.18	0.35	0.2
TSP	0.192	0.31	0.4	0.23
TP	0.23	0.42	0.48	0.4

Principal Sources of Phosphorus: 1990 – 2006/07

The percentage increase in nutrient loading can be used to examine the sources principally responsible for the overall increase in TP loading, highlighting areas of the catchment where phosphorus export intensity has risen. Figure 21 shows that sub-catchment contributions to the TP load are approximately the same in each period, illustrating the dominant role of sub-catchment area in determining the magnitude of P export. Despite the similarity certain changes are apparent. The County and Ballagh sub-catchments account for a greater proportion of the TP load in 2006/07 and the Roogagh sub-catchment accounts for a lower proportion.

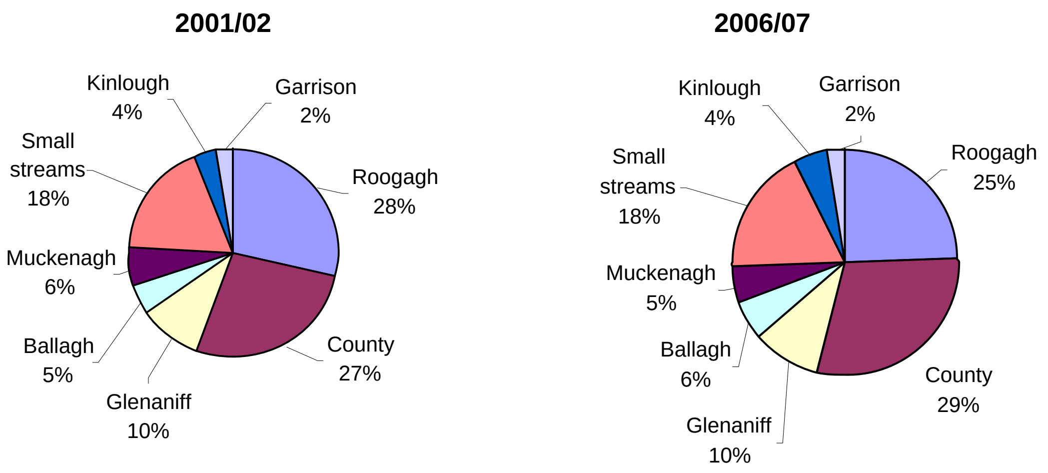


Figure 21. Pie charts showing sub-catchment contributions to the total phosphorus load in 2001/02 and 2006/07.

Figure 22 shows that increases in SRP and PP export between 2001/02 and 2006/07 did not occur uniformly across the catchment. Whilst increases in SRP export were observed across the catchment the most substantial increases were observed within southern area of the catchment in the adjacent Glenaniff, Ballagh and County sub-catchments. The increase in particulate P loading was largely restricted to the County sub-catchment. The increase indicated for the small streams component is likely an overestimate as this component is based upon average catchment export rates and thus reflects a strong contribution from the County sub-catchment which accounts for approximately a quarter of the catchment. For comparative purposes the area of the small streams component represents approximately 18% of the catchment.

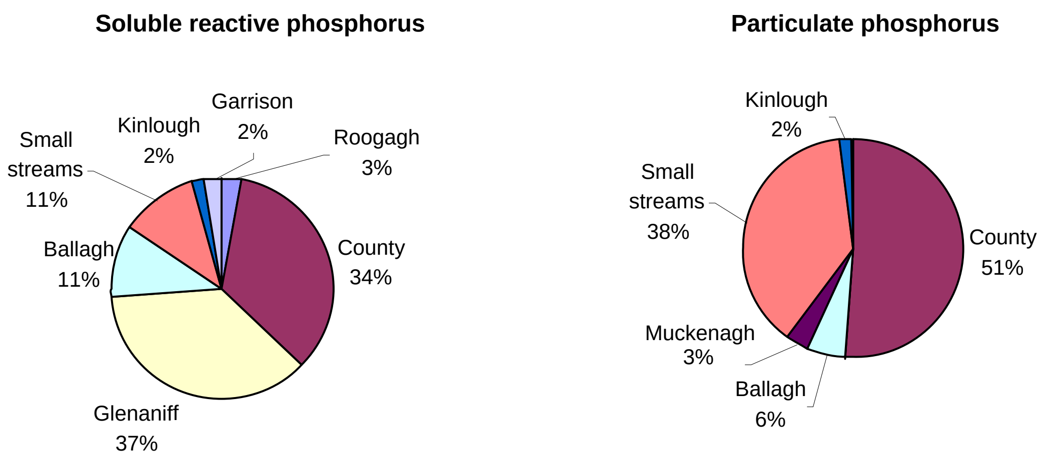


Figure 22. Pie charts showing the sub-catchment contributions to the increase in SRP and PP loading observed between 2001/02 and 2006/07

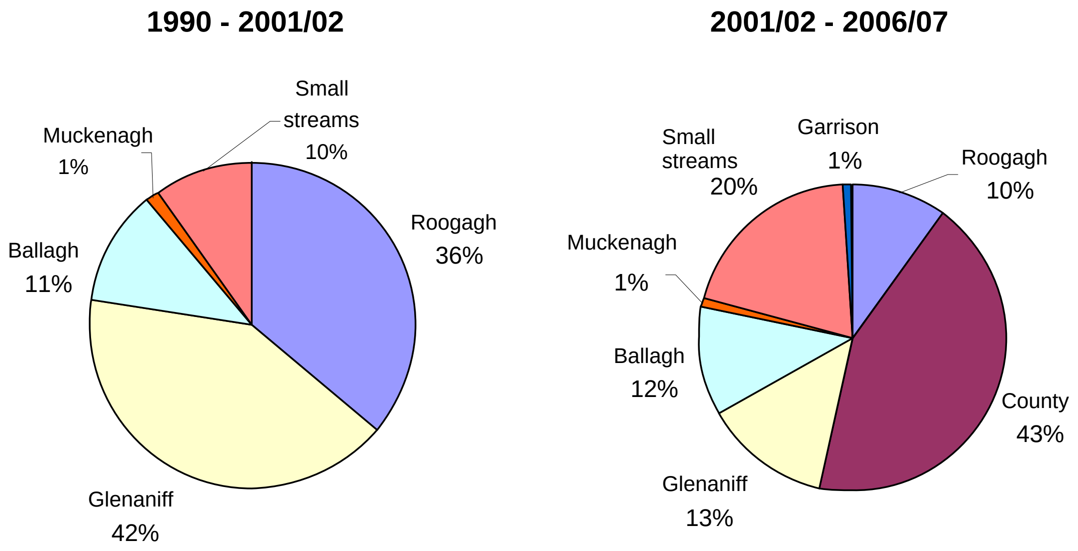


Figure 23. Pie chart of sub-catchment contributions to the increase in total phosphorus loading between 1990 - 2001/02 and 2001/02 - 2006/07

Since 2001/02 the intensity of phosphorus loss from the Roogagh sub-catchment has remained static with a 10% contribution to the increase in TP loading due to greater flows during this period. Between 1990 and 2001 TP export intensity rose within this sub-catchment. 43% of the land area of this sub-catchment is occupied by coniferous forestry providing circumstantial support for the hypothesis that accelerated forestry practices following storm damage in 1999 resulted in a pulse of phosphorus to the lake. The absence of a discernible change in phosphorous loss intensity between 2001/02 and 2006/07 could be due to several synergistic diffuse factors, such as Soil P accumulation, continued loss of P following forestry clearance of storm damage, recent forestry operations, higher stocking rates and poorly functioning septic tank systems.

Loadings of total phosphorus from the Ballagh and Glenaniff sub-catchments increased between each monitoring period suggesting more intensive land use or an accumulation of soil P within these areas. The substantial contribution to increased P loading from the County sub-catchment is a serious cause for concern. Future measures aimed at curbing and reducing P inputs to the lake could therefore be most effective by targeting this sub-catchment together with the Ballagh and Glenaniff sub-catchments.

Phosphorus loading models

The Vollenweider (1976) loading relationship, from which the global OECD (1982) general equation is derived, correlates lake TP concentration with the mean inflow TP concentration weighted by the water retention time, expressed as (1+ √ water retention time) under steady state conditions. OECD General Equation:

$$P_{\underline{l}} = 1.55 * (P_{\underline{in}} / 1+ \sqrt{t_w})^{0.82}$$

where $P_{\underline{l}}$ = mean annual lake phosphorus concentration (µg l^{-1})

$P_{\underline{in}}$ = mean annual phosphorus concentration / catchment flow weighted mean concentration (µg l^{-1})

t_w = water residence time (years)

Foy (1992) in a study of 10 Northern Irish lakes found that the OECD (1982) general equation underestimated lake TP concentrations. The difference is thought to be a consequence of the different characteristics of the lakes involved. For example the majority of lakes in Foy (1992) had water retention times of less than one year which are subject to marked seasonal fluctuations in TP loading, whereas the Vollenweider relationship was derived from a study that did not include such lakes. Vollenweider's (1976) assumption of steady state conditions is more representative of lakes with longer water residence which consequently exhibit greater P retention and lower predicted lake TP concentration.

The OECD (1982) general equation underestimates lake TP for Lough Melvin and Foy's (1992) equation below, although it did not include Lough Melvin, provides a much closer fit:

$$P_{\underline{l}} = 1.234 * (P_{\underline{in}})^{0.991} / (1+ \sqrt{t_w})^{1.13}$$

Figure 24 shows that the TP loading model of Foy (1992) agrees well with the observed mean lake concentrations except for 2001/02 where there is an appreciable difference. A high lake TP concentration that does not fit with the observed pattern of phosphorus exports from the catchment suggests that a pulse of TP has been delivered to the lake in the years preceding the survey. The decline in lake TP observed between 2002 and 2004 (represented as 2003 in Figure 24) supports this hypothesis as temporary lake TP concentrations should be flushed from the system within 2 years. The increase in lake TP observed in 2006/07 matches the pattern of exports for that period, demonstrating that phosphorus export intensity in the catchment has gradually increased. This suggests that high concentrations observed in 2001/02 were the combined result of a catchment perturbation and increasing phosphorus export from the catchment.

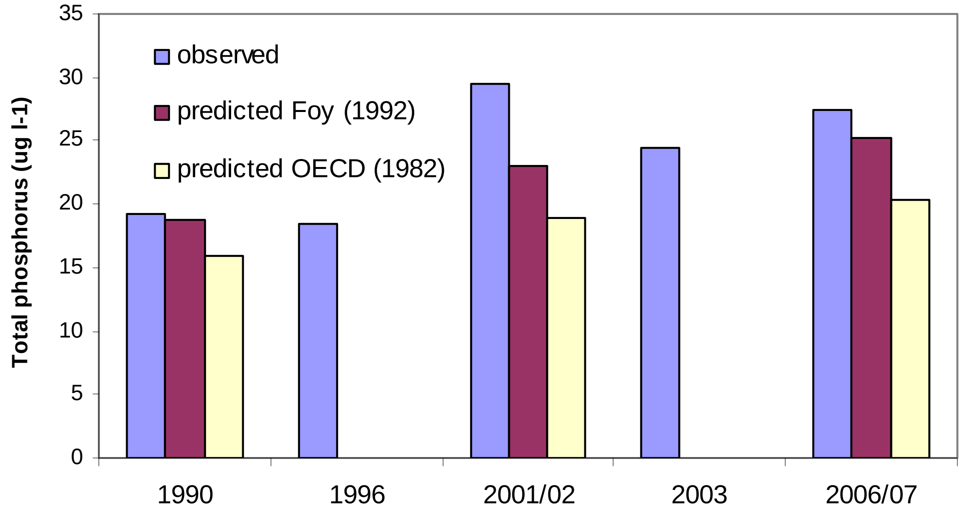


Figure 24. Chart showing mean lake TP concentration and predicted lake TP concentrations according to Foy (1992) and OECD (1982).

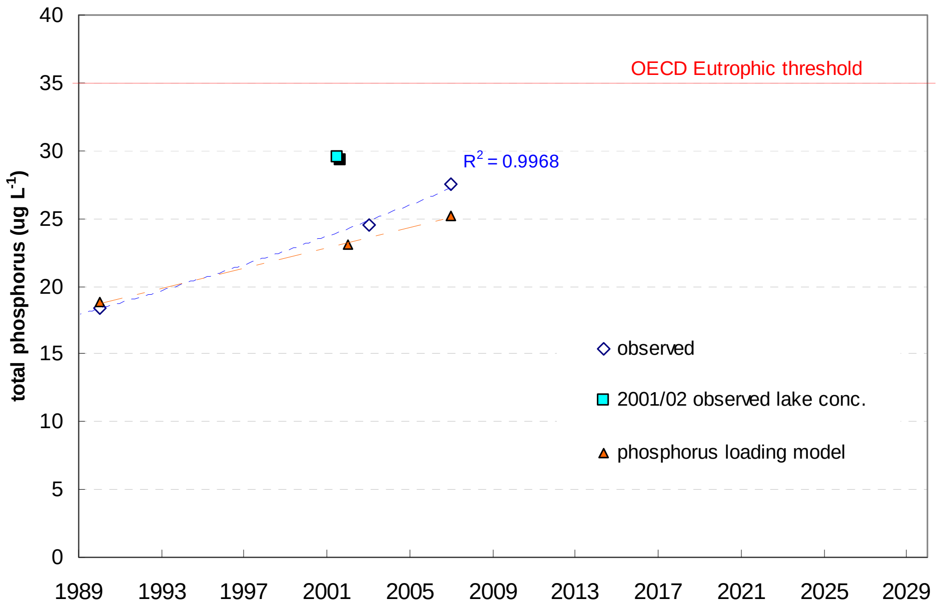


Figure 25. Graph showing the trend of observed lake TP and that of the loading model (Foy, 1992) over time, excluding the observed 2001/02 outlier. The red line marks the lower eutrophic limit (OECD, 1992)

Lake TP concentration were consistent between 1990 and 1995/96 suggesting that increases of catchment export began to occur at some point after this. Figure 25 shows the rise in lake TP from this point to the most recent data. The 2001/02 data are excluded from this relationship as this year did not fit with the observed pattern of loadings to the lake from the catchment. It is likely that the pattern of exports from the catchment was very similar in 1995/96 to that observed in 1990. Consequently the predicted lake TP for 1990 according to Foy (1992) has been presented for 1995/96.

Nutrient Budget 2006/07

Comparison of estimation methods

The data presented in the previous section was calculated to allow the most accurate comparison of data between years. With better flow gauge coverage of the catchment in 2006/07 more accurate approaches were available for calculating the nutrient budget. Figure 26 below shows the results obtained for phosphorus fractions by applying each method.

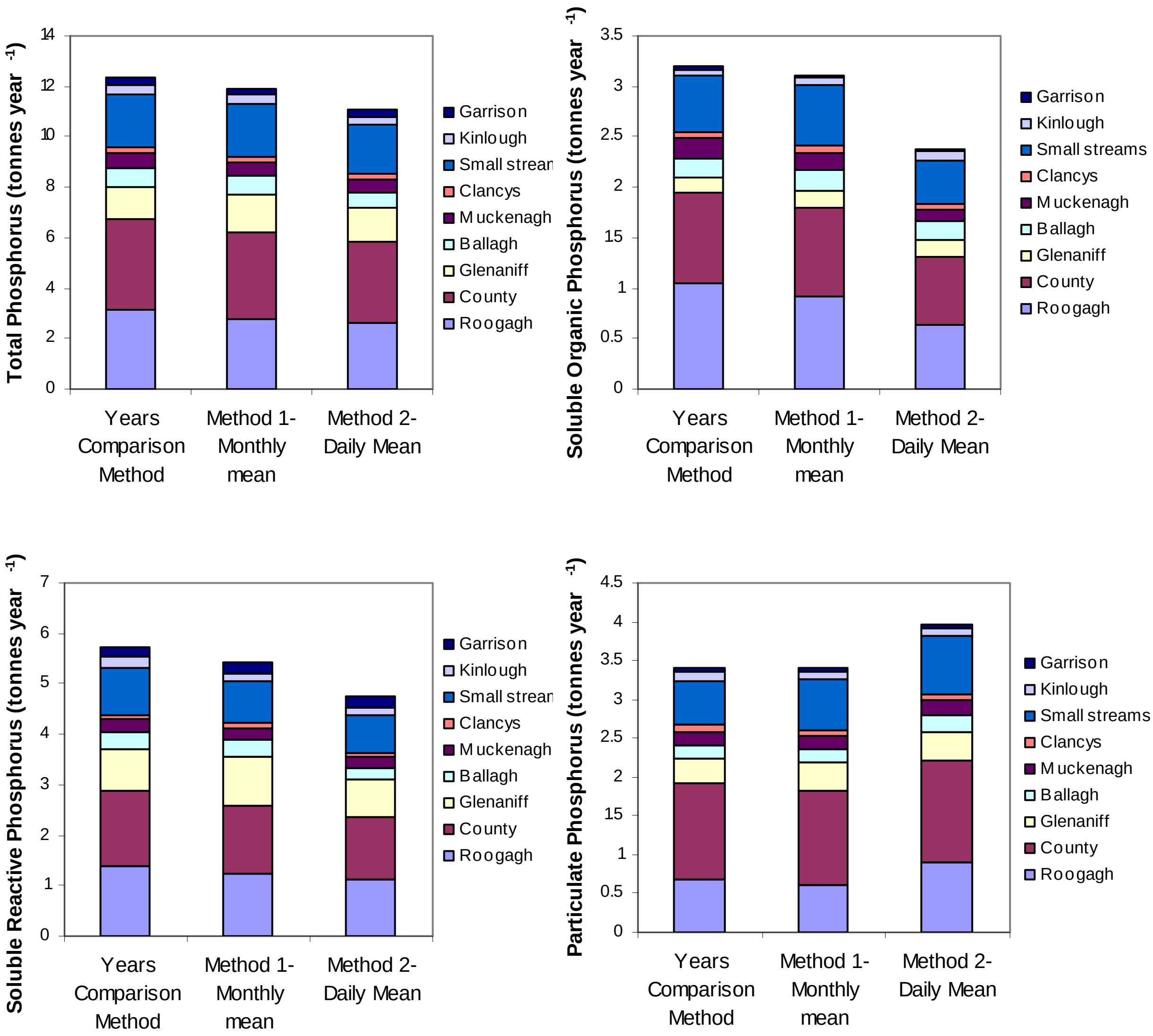


Figure 26. Annual loading of phosphorus fractions to Lough Melvin in 2006/07, with contributions by sub-catchment according to 3 methods: Years Comparison Method, Method 1-monthly mean concentration, Method 2-Mean daily flow & instantaneous sample concentration.

Method 2 is the most appropriate approach as it suffers from less bias than method 1 by accounting for a degree of flow-dependent variation in concentration. However it can be biased by a low sampling frequency that leads to a strong tendency for sampling during base flow conditions and falling limbs of the hydrograph. This is a potentially strong source of error within the Melvin catchment where rivers are flashy and storm events typically contribute in excess of 50% of the total flow.

Rivers were sampled during peak flows on only one occasion during the monitoring period (6th December 2006) however this measurement had a significant effect on the budget calculation. Among the highest PP concentrations observed during the monitoring period were recorded during this storm event without a concomitant increase of SOP and SRP concentrations. This resulted in a marked increase in the calculated PP loading.

For example if the peak mean daily flows observed on the 6th December 2006/07 are substituted for the mean daily flow for the month of December for the Roogagh river, the flow-weighted mean concentration of PP is reduced from 12.98 µg L^{-1} to 9.61 µg L^{-1} reducing the calculated annual PP load by 26%. Riverine transport of particulate matter is greatest during hydrograph rising limbs and peak flows but water clarity improves rapidly during the falling limb. For this reason the incorporation of data from one peak flow sampling event has likely lead to an underestimation of the annual PP loading.

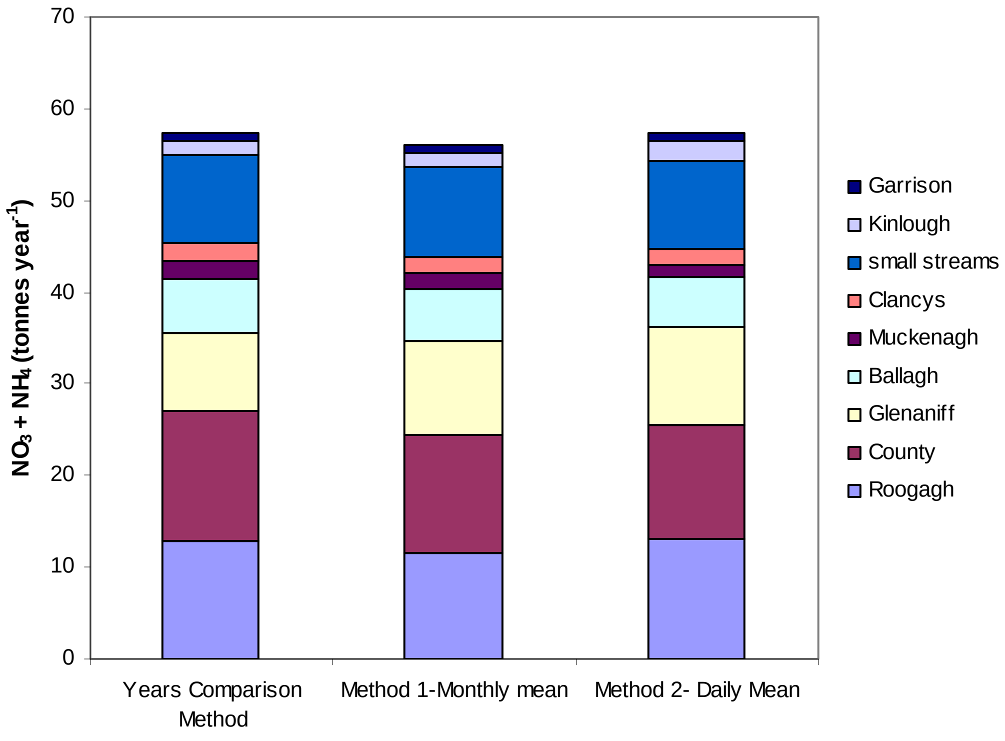
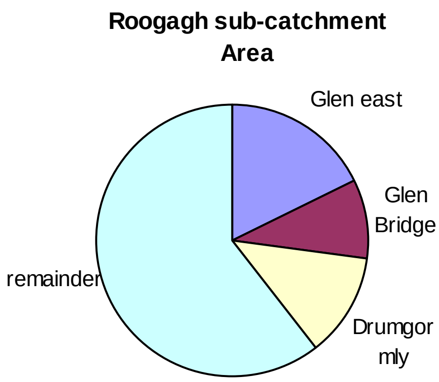


Figure 27. Annual loading of Nitrogen (NO_3 + NH_4) to Lough Melvin in 2006/07, with contributions by sub-catchment according to 3 methods: Years Comparison Method, Method 1-monthly mean concentration, Method 2-Mean daily flow & instantaneous sample concentration.

Nutrient sources within the Roogagh and County Sub-catchments 2006/07

The Roogagh and County sub-catchments together account for more than half of the Lough Melvin catchment and historically have shown among the highest sub-catchment export rates. Figure 28 shows that the monitored tributaries and the remainder of the catchment exported phosphorus approximately in accordance with their respective areas. Soluble reactive phosphorus export was relatively uniform across the sub-catchment. The Glen Bridge tributary had the lowest nutrient exports rates of all phosphorus and nitrogen fractions. The downstream 'remainder' section of the catchment had significantly higher export rates of SOP and nitrate than the tributaries. The Drumgormly and Glen East catchments had the highest TP export rate; this was largely due to greater exports of particulate phosphorus. The remainder of the catchment had the lowest PP export rate observed within the catchment.



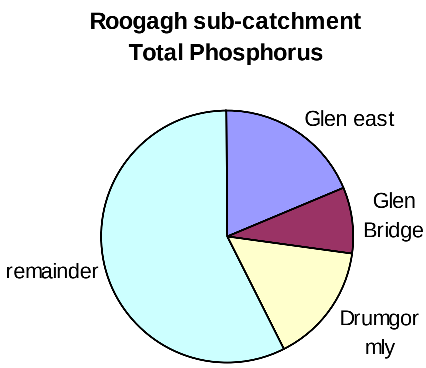
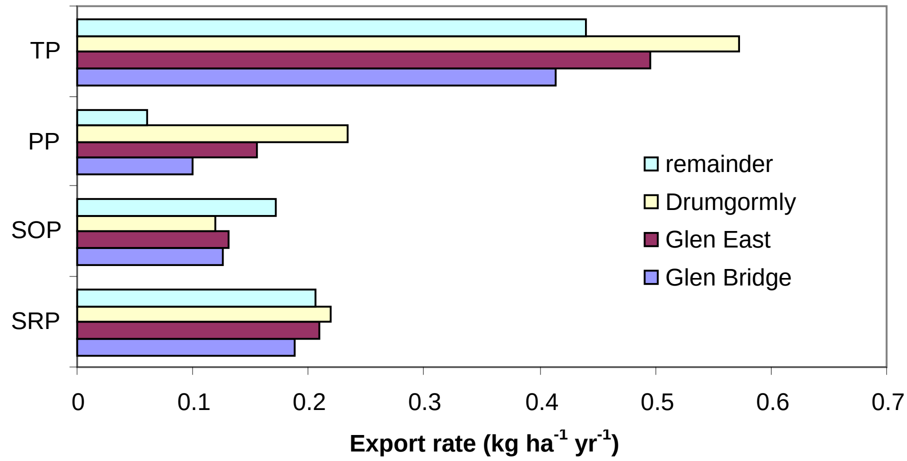


Figure 28. Pie charts showing a) the proportions of the Roogagh sub-catchment area occupied by each monitored tributary and the remainder of the catchment b) the contribution of each monitored tributary and the remainder to the total phosphorus load.

The Drumgormly and Glen East tributaries drain land consisting of 71% and 60% coniferous forestry had the greatest TP export rates. The downstream remainder of the catchment consisting of 20% pasture, 30% coniferous forestry and 40% grassland had the highest nitrate export rates. The Glen Bridge sub-catchment, consisting of 74% natural grassland and 12% coniferous forestry, had the lowest export rates of both TP and nitrate. From this data it appears that coniferous forestry contributes the most P (particularly particulate P) per unit area and that pastures or agricultural land contributes more nitrate per unit area.



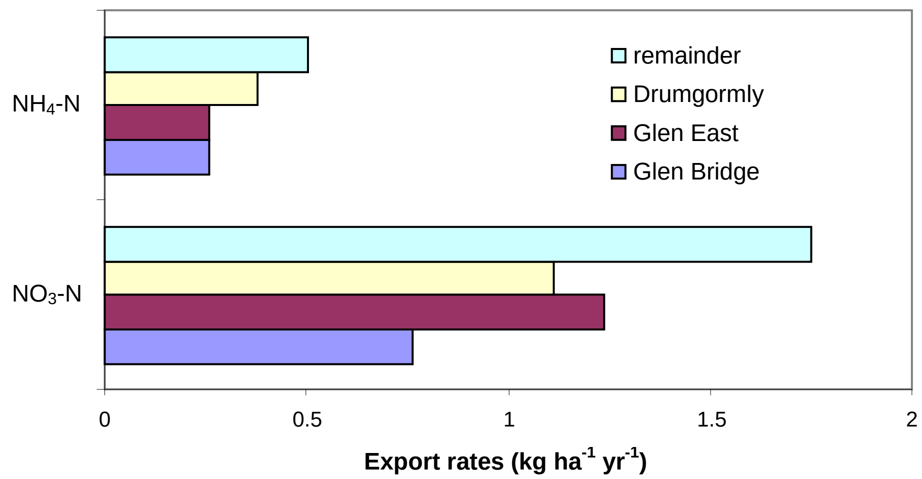


Figure 29. Export rates for total phosphorus (TP), particulate phosphorus (PP), soluble organic phosphorus (SOP), soluble reactive phosphorus (SRP) and Ammonia (NH_4) and Nitrate (NO_3) for sections of the Roogagh sub-catchment.

County

The County River was responsible for 28% of the increase in SRP loading, 51% of the increase in PP loading and 40% of the increase in TP loading to Lough Melvin observed between 2001/02 and 2006/07 (Figs. 22 & 23), highlighting the need for greater resolution regarding nutrient sources in this sub-catchment. The river receives additional nutrients from a waste water treatment works at the village of Kiltyclogher. This discharges into the stretch of river downstream from the village and would be included in the ‘remainder’ section of the sub-catchment. The data presented in Figures 30 and 31 have been corrected for this input and therefore only represent rural sources of nutrients.

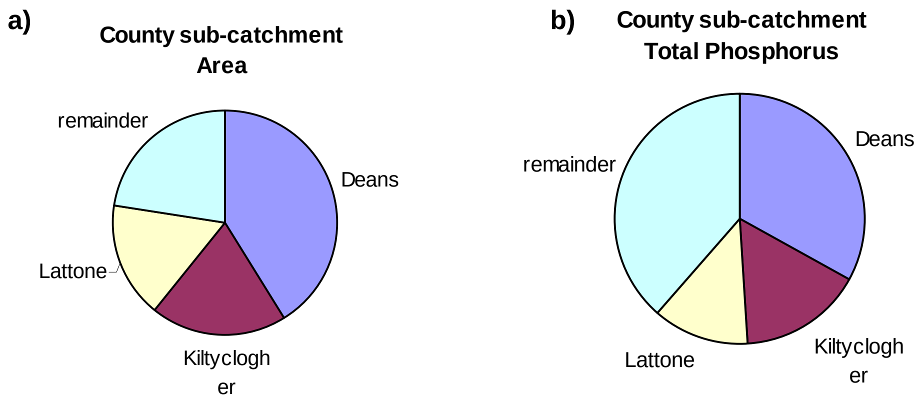


Figure 30. Pie charts showing a) the proportions of the County sub-catchment area occupied by each monitored tributary and the remainder of the catchment and b) the contribution of each monitored tributary and the remainder to the total phosphorus load of the County sub-catchment

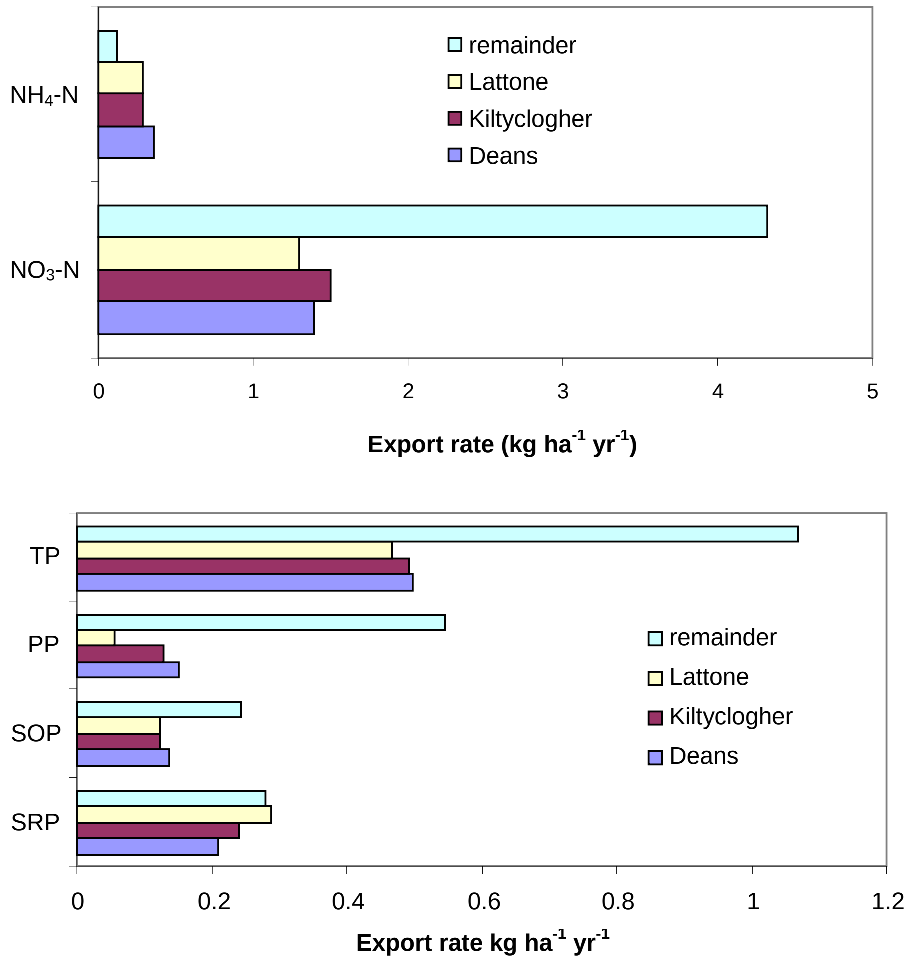


Figure 31. Export rates for total phosphorus (TP), particulate phosphorus (PP), soluble organic phosphorus (SOP), soluble reactive phosphorus (SRP) and Ammonia (NH_4) and Nitrate (NO_3) for sections of the Roogagh sub-catchment.

The urban contribution of nutrients from the Kiltyclogher sewage treatment works accounted for a small proportion of the nutrient load at 7% of the SRP load, 2% of the PP load, 2% of the SOP load, 4% of the TP load and 4% of the Nitrogen load (NO_3+NH_4). The estimated population increase of 16 people between 2001/02 and 2006/07 is therefore a negligible fraction of the increased exports observed for this sub-catchment.

Figure 30 shows a discernable difference in the total phosphorus loading expected from the areas of each section of the sub-catchment, with the higher order, 'remaining' area exporting more phosphorus and nitrate per unit area. The total phosphorus export from this area consisted principally of the particulate fraction. The intensity of nitrate loss was three times than that of the upstream tributary catchments. In contrast the SRP export rate was markedly lower than the tributaries. Within the low order tributaries exports of phosphorus and nitrogen were relatively uniform.

Nutrient export intensity 2006/07

The small stream that receives effluent from the Kinlough WWTW shows the highest nutrient loss intensities. The up-stream flows are very small and consequently effluent entering the stream is diluted very little. The observed changes in nutrient loading to the lake from the WWTW are discussed in the nutrient loading comparison on page xx. In terms of phosphorus, sub-catchments with large proportions of agricultural land and/or coniferous forestry displayed the greatest loss intensities (Fig. 32). Overall SOP loss intensities tended to be slightly higher for land used for agriculture compared to sub-catchments dominated by coniferous forestry. In contrast nitrate loss intensities, with the exception of the Muckenagh sub-catchment, were higher in catchments dominated by agricultural land use.

Although the Ballagh and Glenaniff sub-catchments have been highlighted for the increases observed in phosphorus loss intensity over time they still display among the lowest export rates.

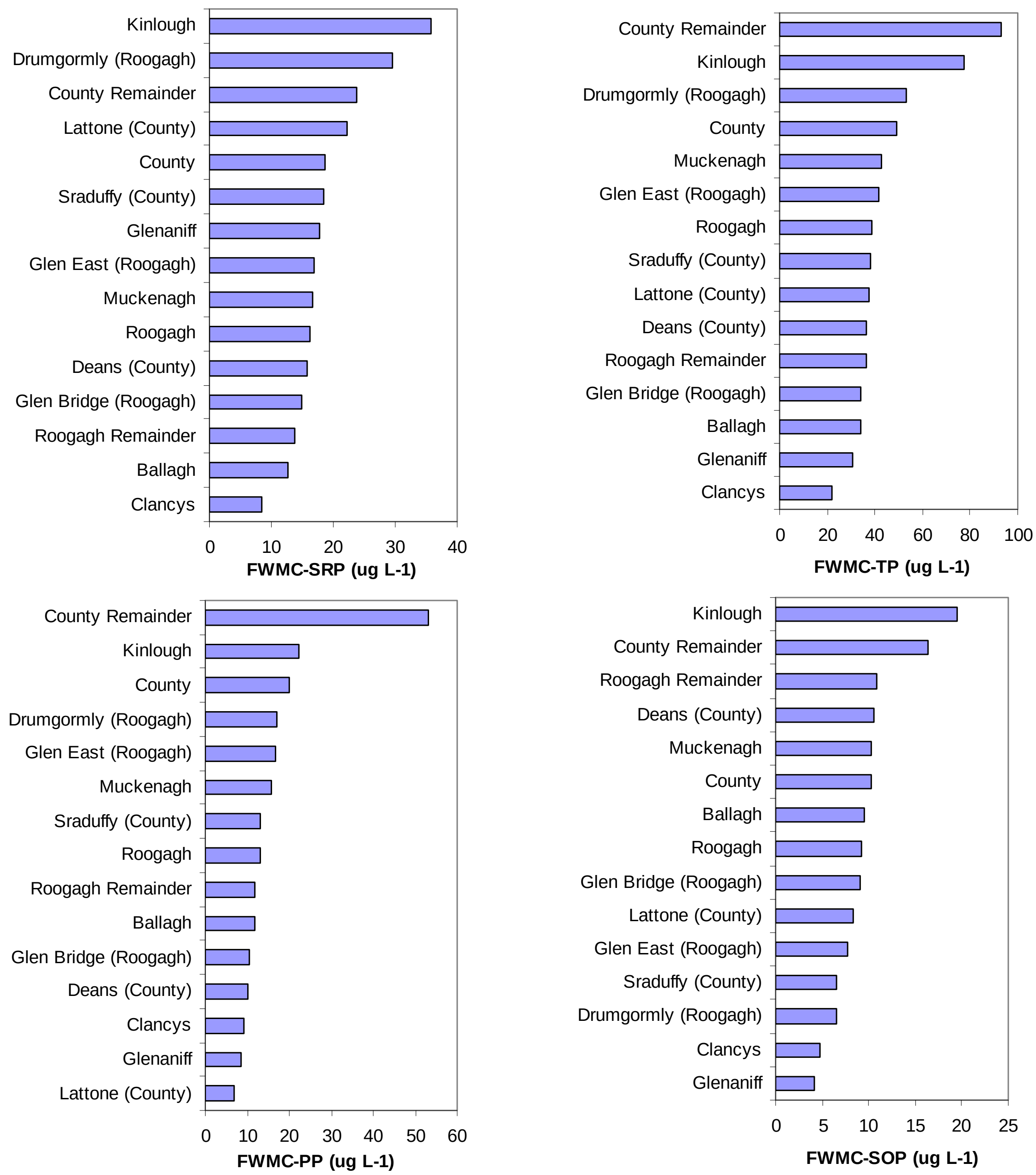


Figure 32. Ranked flow-weighted mean concentrations of phosphorus fractions for each monitored sub-catchment in 2006/07.

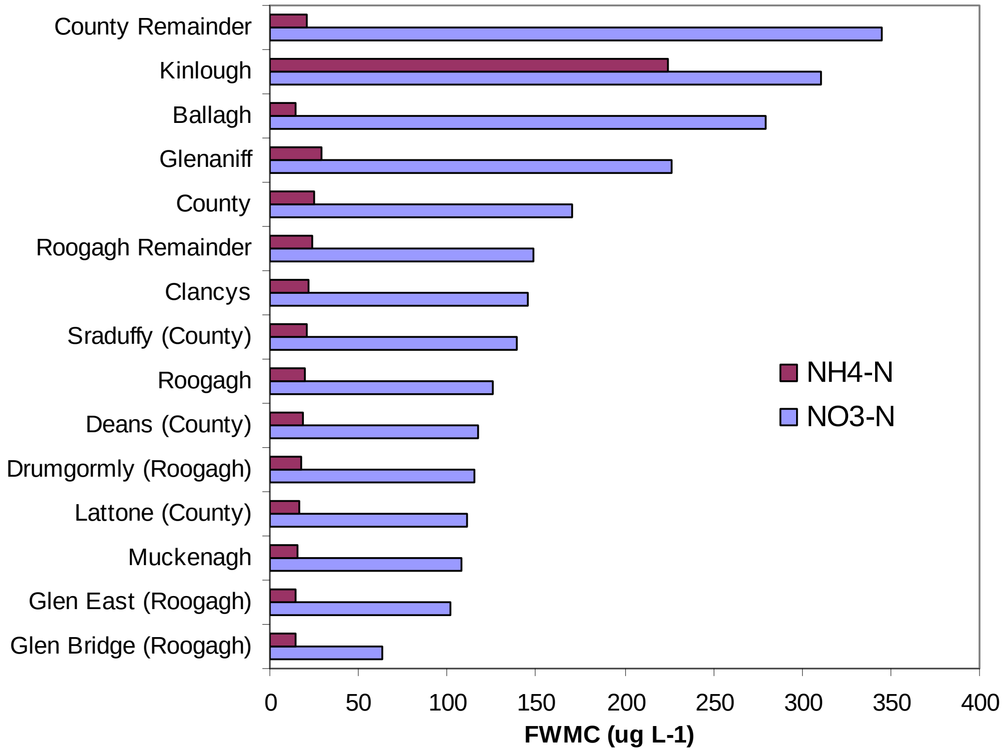


Figure 33. Flow-weighted mean concentrations of nitrate nitrogen (NO_3-N) and ammonia nitrogen (NH_4-N) for each monitored sub-catchment in 2006/07.

Nutrient export intensity and CORINE 2000

Regression models were used to examine the effect of land use on nutrient export intensities. The proportions of different land use (as CORINE, 2000 categories), expressed as percentages of the sub-catchment area, were employed as predictors of sub-catchment nutrient loss intensity expressed as the sub-catchment flow-weighted mean concentration. A multiple regression model of SOP loss intensity with the CORINE categories ‘Land principally occupied by agriculture’ + ‘Pastures’ and ‘transitional wood land / scrub’ + ‘coniferous forestry’ and ‘peat bogs’ as predictors explained 85% of the variation of SOP loss intensity in the catchment (R^2 = 0.849, $p < 0.001$). SOP loss intensity was strongly positively related to the proportion of pastures and agricultural land and negatively related to the proportion of peat bogs in the catchment.

Nitrate loss intensity with the CORINE categories ‘Land principally occupied by agriculture’ + ‘Pastures’ as a predictor showed a positive relationship (R^2 = 0.37, $p < 0.05$). Nitrate loss intensity showed a strong negative relationship with the CORINE category ‘Coniferous forestry’ + ‘Transitional wood land / scrub’ and ‘Natural grassland’ (R^2 = 0.61, $p < 0.001$). These categories could not be used as separate predictors in a multiple regression model as they were closely correlated.

There was no clear relationship between SRP and PP loss intensities and land use as delineated by CORINE 2000.

Biological Limnology: Zooplankton

While the species composing the zooplankton community have been largely consistent since 1990 (Table 12, Appendix I) significant changes in abundance have been observed (Table 11). Figures 34-40 show the annual cycles of the dominant zooplankton species present in each year monitored.

Table 11. Differences in abundance between surveys for dominant components of the zooplankton community

	Significant difference		
Species	**1990 – 2001/02**	**1990 – 2006/07**	**2001/02 – 2006/07**
Daphnia hyalina var. *galeata* Sars	NS	NS	NS
Cyclops strenuus abyssorum Sars	$p < 0.05$	$p < 0.001$	$p < 0.05$
Eudiaptomus gracilis (Sars)	$p < 0.05$	NS	$p < 0.05$
Arctodiaptomus laticeps (Sars)	NS	NS	NS
Total dominant rotifers: *Keratella cochlearis* (Gosse) *Kellicottia longispina* (Kelicott) *Polyarthra dolichoptera* Idelson	$p < 0.005$	$p < 0.005$	NS

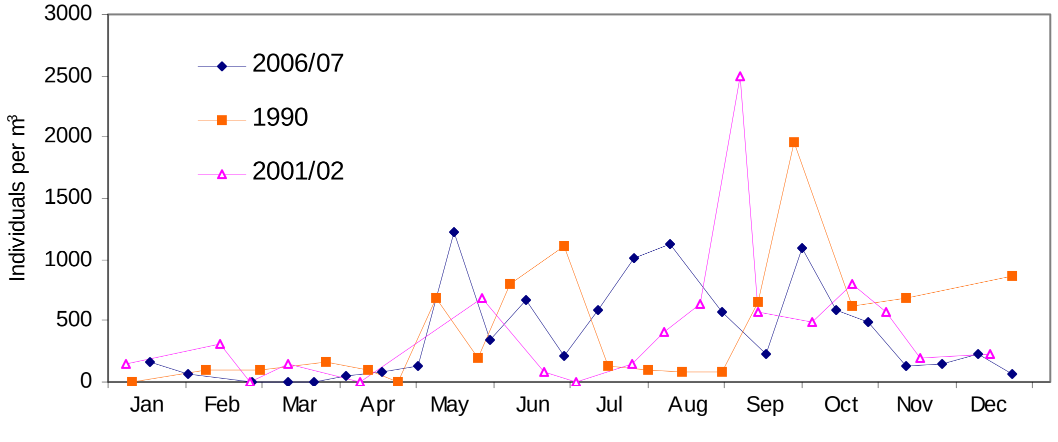


Figure 34. Annual cycles of *Daphnia hyalina* var. *galeata*

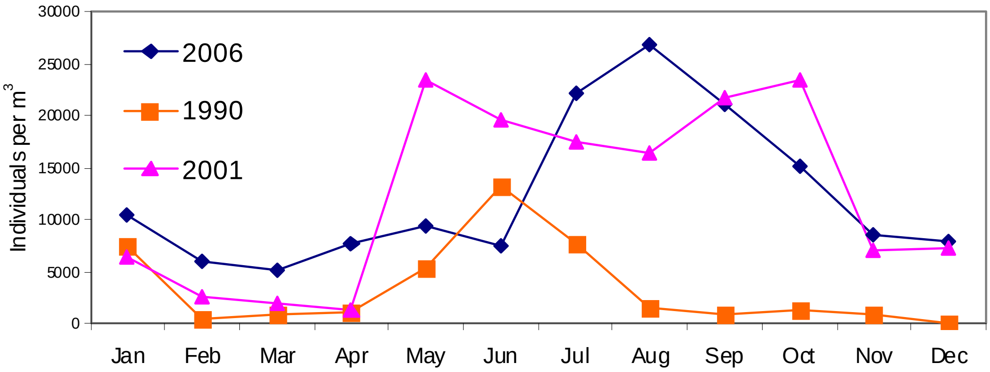


Figure 35. Annual cycles of dominant rotifer species: *Keratella cochlearis, Kellicottia longispina* and *Polyarthra dolichoptera*

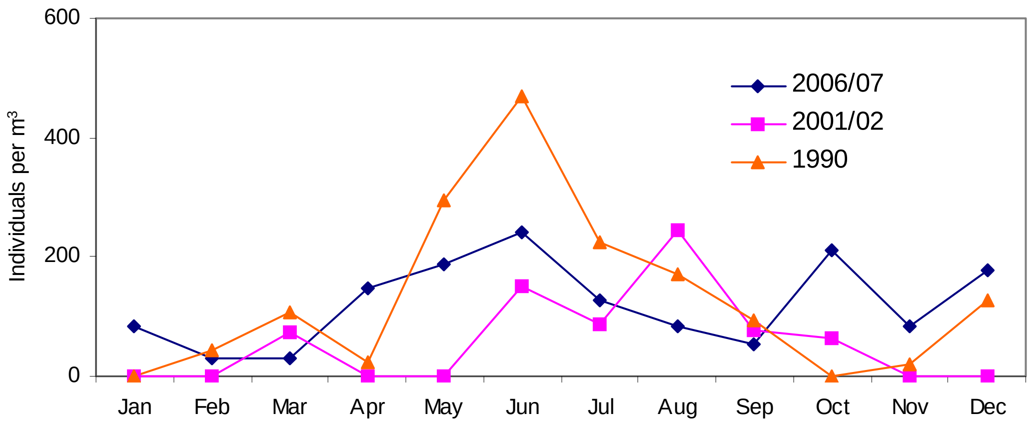


Figure 36. Annual cycles of adult *Arctodiaptomus laticeps*.

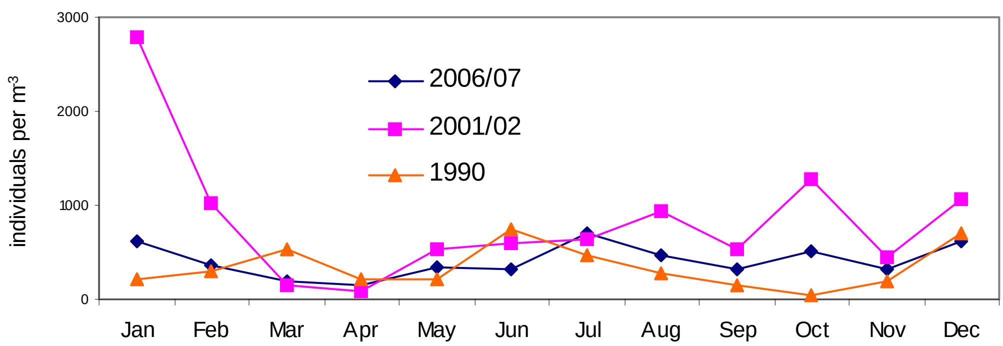


Figure 37. Annual cycles of adult *Eudiaptomus gracilis*.

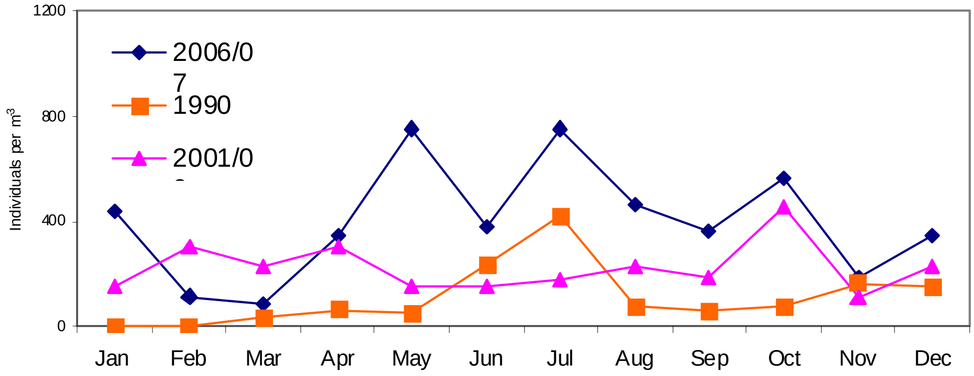


Figure 38. Annual cycles of adult *Cyclops strenuus abyssorum*.

A distinct change was observed in the abundance of two colonial species of rotifer that belong to the same genus: *Conochilus hippocrepis* (Schrank) and *Conochilus unicornis* (Rousellet). *C. hippocrepis* was recorded during each monitoring period. In 1990 two abundance peaks were observed, both exceeding 10,000 individuals m^{-3} whereas in 2001-2 and 2006-7 the annual maximums totalled only 301 and 4,086 individuals m^{-3} respectively (Fig. 39).

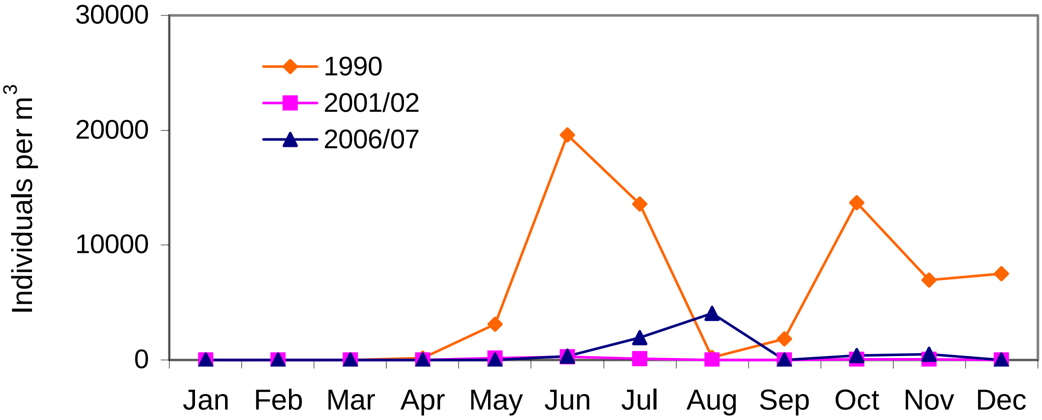


Figure 39. Annual cycle of abundance for *Conochilus hippocrepis* for each year monitored

Conochilus unicornis was first recorded in 2001-2 with the population increasing rapidly from early September, rising to peak abundance in November and December. The abundance and timing was very similar in 2006-7 (Fig. 40)

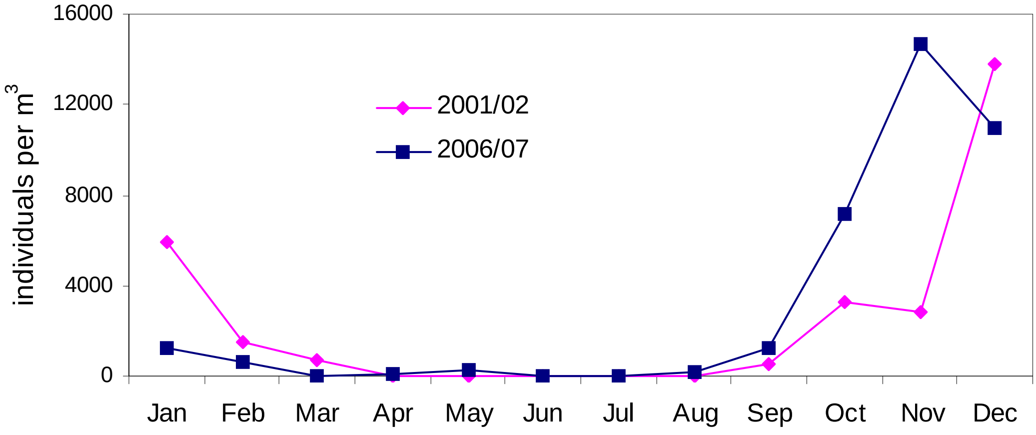


Figure 40. Annual cycle of abundance for *Conochilus unicornis* in 2001/2 and 2006/07. This species was unrecorded before 2001.

Biological Limnology: Phytoplankton

Phytoplankton species observed during each monitoring survey and during additional sampling in the 1950's and 1980's are listed in Table 13 Appendix I. In agreement with observed chlorophyll concentrations, algal biovolumes were significantly lower in 2006/07 compared with 1990, particularly during the spring (Fig. 41). Mean growing season biovolumes (May to October) were 0.92, 0.46 and 0.59 $mm^3\ L^{-1}$. The annual mean biovolumes were 0.74, 0.44 and 0.41 $mm^3\ L^{-1}$ in 1990, 2001/02 and 2006/07 respectively. These values fall at the low end of the range commonly reported for mesotrophic lakes.

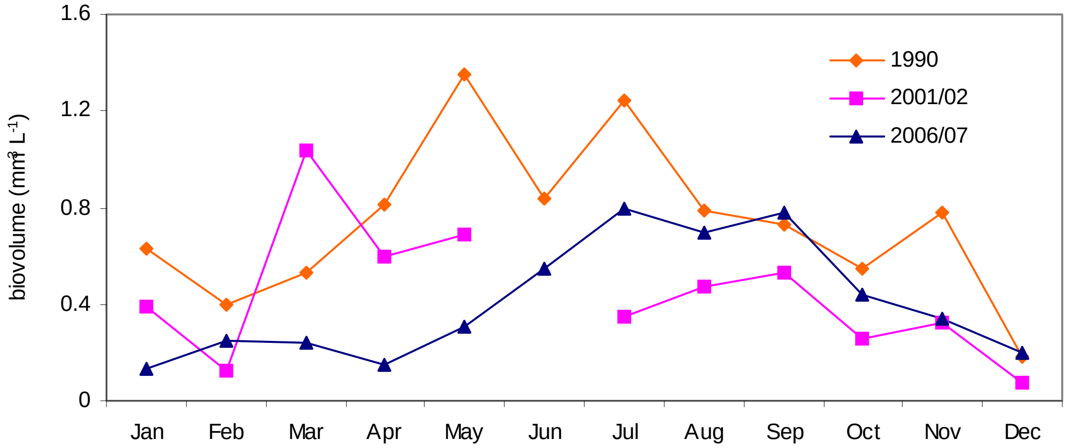


Figure 41. Annual cycles of phytoplankton biovolume.

Annual diatom abundance usually peaks during the spring however this characteristic bloom failed to occur during 2007 (Figs 41 & 42). Nevertheless diatoms were sufficiently abundant to dominate the algal biovolume through much of the spring implying overall limitation of pelagic primary productivity. Figure 42 shows the annual cycle of the two main groups of algae (cyanobacteria and diatoms) and all other species (total) for each monitoring survey. The dominant cyanobacterial species have been consistently recorded over time. These are *Woronichinia naegliana*, *Anabaena flos aquae*, *Microcystis aeruginosa, Oscillatoria agardhii and Oscillatoria redecki*. Species dominating diatom biovolumes have also remained very similar over time. These are *Aulacoseira italica* var. *subarctica* and *Aulacoseira islandica*, *Asterionella formosa, Stephanodiscus neoastrae, Cyclotella sp.* and *Synedra ulna* var. *danica.*

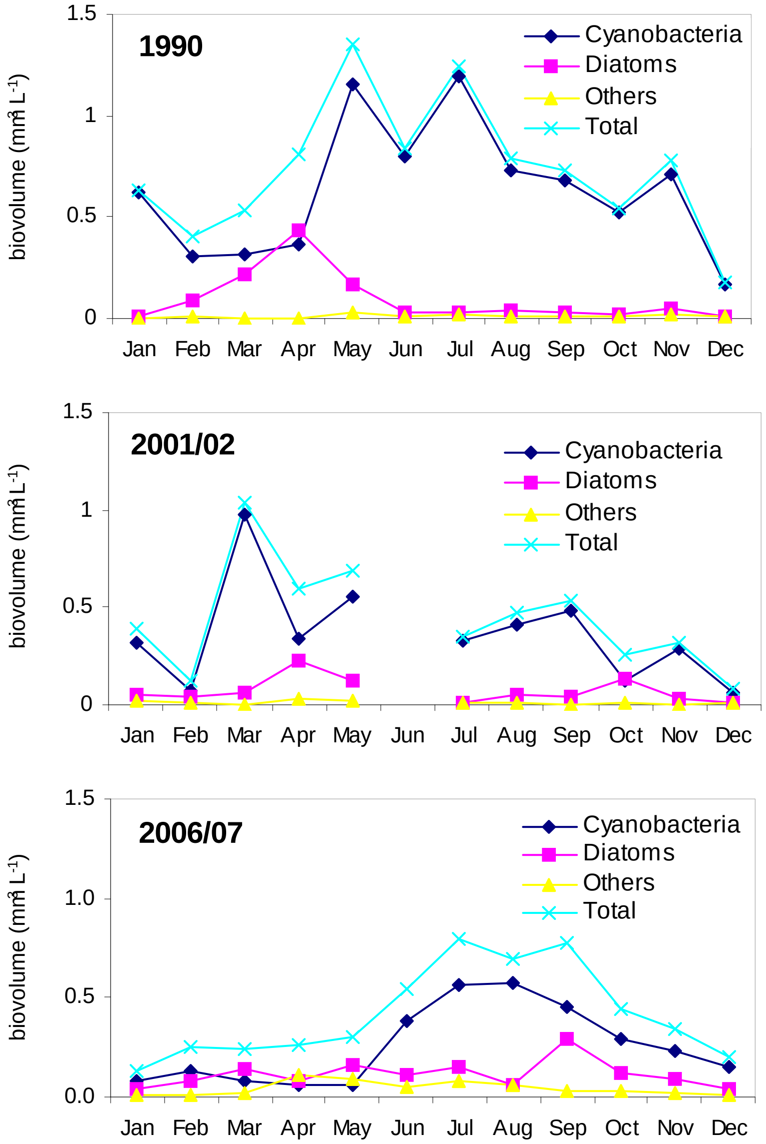


Figure 42. Algal volumes of the dominant taxa in each sampling period (monthly means).

Significant increases in the abundance of Cryptomonads and dinoflagellates ($p < 0.01$, paired t-test) were observed in 2006/07 compared with previous years (Figs. 43 & 44). The cryptophyte, *Cryptomonas erosa* Ehrenberg and the Dinoflagellate, *Gymnodinium discoidale* Harris were recorded for the first time in 2006/07. The cryptophyte *Rhodomonas sp.* was only recorded during one month in 2002 whereas it was present throughout the year in 2006/07.

Similarly *Mallomonas sp.* (Chrysophyta) was only recorded on one occasion in 2002 compared to 3 consecutive fortnightly observations in 2007. Interestingly dinoflagellates of the genus *Gymnodinium* and cryptophytes were not recorded until September 1983 (DARD, unpublished data).

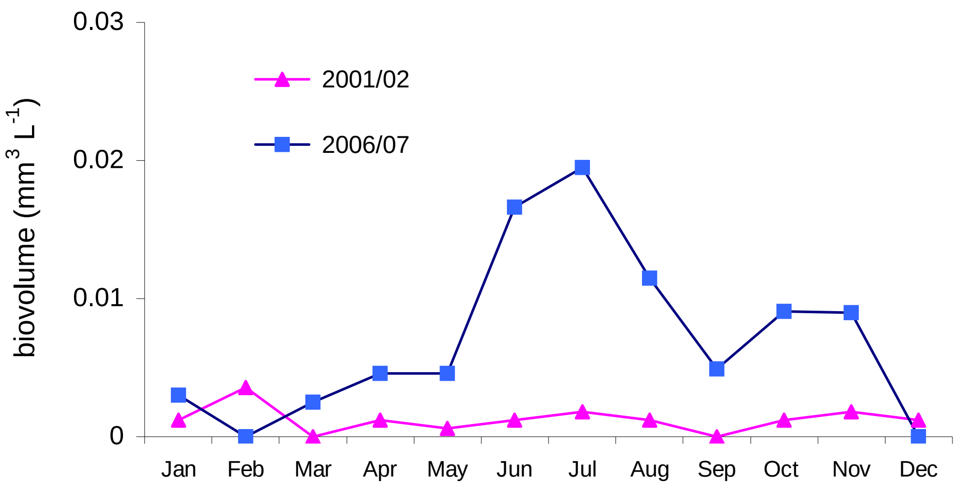


Figure 43. Annual cycles of Dinoflagellate abundance for 2001/02 and 2006/07. No dinoflagellates were recorded during 1990.

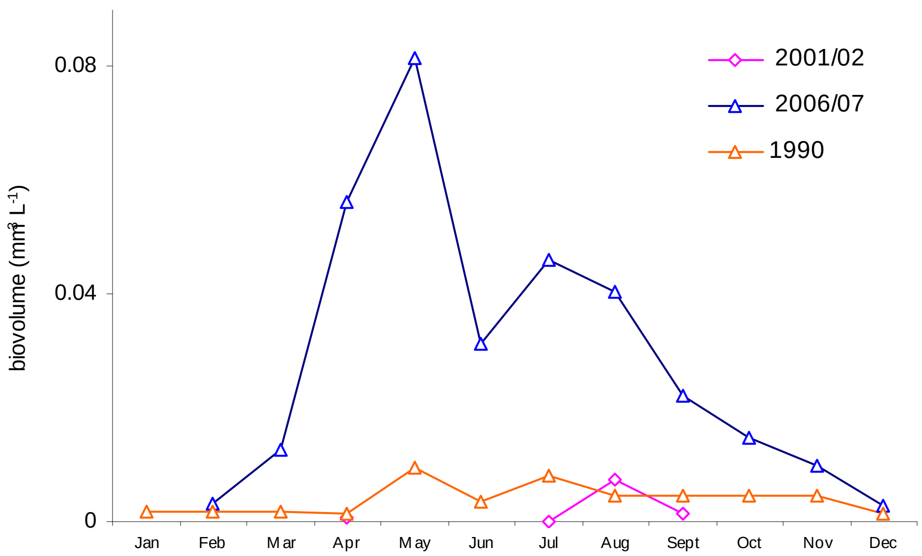


Figure 44. Annual cycle of Cryptophyte abundance for 1990, 2001/02 & 2006/07.

Discussion

Temperature and dissolved oxygen status

Temperature and dissolved oxygen (DO) are key factors driving the distribution of aquatic species. They are inherently linked in lentic freshwaters by a negative solubility relationship and the potential for thermal stratification leading to hypolimnetic deoxygenation. Lough Melvin's designation as a Special Area of Conservation stems, in part, from its unique salmonid community. Salmonids require cool, well-oxygenated water and their long-term survival in the lough relies upon the persistence of these conditions throughout the year.

Arctic char are the most temperature sensitive salmonids in the lake and are intolerant of high temperatures. A cool, well-oxygenated deep-water zone that persists throughout the year is of paramount importance to their survival in Lough Melvin. Lethal temperatures for Arctic Charr have been experimentally measured at between 18.7 – 26.6ºC depending on the life history stage (alevins, fry or parr) and temperature acclimation history (Baroudy & Elliott, 1996). Maximum bottom temperatures in Lough Melvin are usually achieved by a relatively gradual warming and have yet to exceed 17ºC providing a strong indication that the habitat is likely to remain favourable for salmonids in this respect.

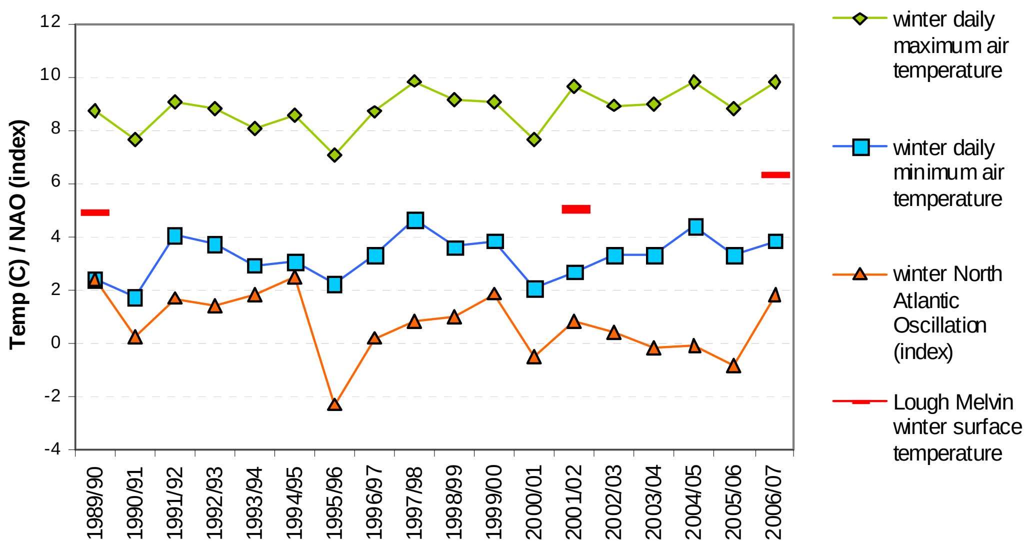


Figure 45. Winter maximum and minimum mean air temperatures and winter means of the North Atlantic Oscillation (dimensionless) between 1989/90 – 2006/07. Mean winter surface water temperatures of Lough Melvin are shown for 1990, 2001/02 and 2006/07. (Air temperatures recorded at Ballyshannon, Co. Donegal -7km north of Lough Melvin). NAO data was obtained from the Climate Research Unit, University of East Anglia (**http://www.cru.uea.ac.uk/cru/data/**). (Air temperatures provided courtesy of Teagasc, ROI).

Interestingly mean annual bottom temperatures have risen in each successive monitoring period, (Table 3) significantly so between 1990 and 2006-7 (paired t-test, $t = 4.08$, $p < 0.005$). However global warming is unlikely to have affected the Lough as the observed 1.29 ºC rise (0.59 – 1.98; 95% CI) is inconsistent with global increases of approximately 0.15ºC over the same period (Brohan et al., 2006). Local weather differences in response to changes in the North Atlantic oscillation appear to exert the greatest effect on Lough Melvin's temperature regime (Fig. 45).

Despite two distinct periods of deep-water deoxygenation caused by weak stratification events (Fig. 4), DO concentrations did not reach the threshold value of 6mg O_2 L^{-1} specified for salmonid waters under the EU Fish Directive. Had the two periods of stratification not been separated by a short period of poor weather, DO concentrations by the second turnover event would only have decreased to an estimated 7.01 mg O_2 L^{-1}. The first and second stratification periods would have had to continue for an estimated 83 and 40 days respectively for DO concentrations to fall to 6mg O_2 L^{-1}. The higher respiration rate observed during the second stratification period was likely a consequence of greater pelagic primary productivity before stratification. Peak chlorophyll concentrations (water column average: 5.6 µg L^{-1}) between the 28th June and 11th July preceded the 2nd stratification event whereas low concentrations (~2.0 µg L^{-1}) preceded the 1st period of stratification. Furthermore the mixing event separating the two periods would have redistributed phytoplankton from the mixed surface waters throughout the water column. These cells would have experienced a more favourable light history and would consequently have a greater respiratory demand due to the accumulation and storage of respirable carbohydrate (Gibson, 1975).

The likelihood of continuous calm weather for a period in excess of 2 months on Lough Melvin is small. Lake morphometry, local catchment topography and a coastally exposed location ensure that winds are sufficient to prevent strong persistent stratification from developing. Stratification observed to date in Lough Melvin can be considered very weak compared to lakes that permanently stratify. Without significant increases in productivity and/or external organic matter loading, DO concentrations are likely to remain favourable for salmonids.

Trophic Status

Well-defined chlorophyll peaks in April 1990 and 2002 show the occurrence of a spring bloom in Lough Melvin that is characteristic of temperate lakes. Biovolumes of phytoplankton taxa present at these times demonstrate the occurrence of an annual diatom bloom. In 1990 and 2002 cyano-bacterial biovolumes exceeded diatom biovolumes during each spring. In 2007 however, despite dominating the phytoplankton biovolume throughout the spring, diatoms failed to achieve a similar peak. This implies that all phytoplankton taxa were limited in early 2007 since biovolumes of all taxa were atypically low (Fig. 39). Sharp declines in soluble silica were observed between March and April 1990 and 2002 coinciding with the spring diatom bloom. This well studied occurrence is observed when appreciable amounts of silica are incorporated into diatoms to form their cells walls or frustules (Lund, 1964). In the absence of a pronounced diatom peak in spring 2007 no such declines in soluble silica were evident. Only in 1990 did soluble silica concentrations fall below the level of 0.5 mg L^{-1} considered limiting to the growth of most diatoms.

The ability of diatoms to monopolise increasing solar irradiance, rising temperatures and well-mixed conditions during the spring are key factors driving their annual bloom. Unusual intervals of clear, calm weather during spring 2007 may have disrupted the usual pattern of phytoplankton succession by loss of a significant proportion of diatoms through sedimentation. Higher grazing pressure in April & May 2006-7 could also have resulted in lower algal numbers. However the abundance of the principal grazing components of the zooplankton community was similar in all surveys, but top down control of zooplankton by higher consumers may have reduced grazer numbers. Stable isotope analysis of zooplankton over the same time period indicates they fed almost exclusively upon phytoplankton during this period, nevertheless without data to show that zooplanktivores were more abundant in 2006-07, top down control of algal numbers remains a tenuous explanation.

Lower algal abundances recorded during the remainder of the year are equally difficult to explain in view of similar nutrient concentrations (N, P, SiO_2) and similar Secchi depths indicating a similar light climate to those observed during previous monitoring surveys where algal abundances were significantly higher. A potential explanation is that greater dissolved organic matter loading from the catchment has reduced the light climate and limited algal production. The peat staining of Lough Melvin’s waters is caused by dissolved organic matter (DOM) that is commonly referred to as humic substances. These absorb light strongly in the ultra-violet and low end of the visible spectrum (200-500nm). Dissolved organic carbon concentrations, used as a proxy for DOM concentration, strongly correlate with absorbances at 430nm. The photosynthetic pigments chlorophyll a & b also absorb strongly in this region providing circumstantial evidence for competition for light with catchment derived DOM.

Lough Melvin continues to fall within the OECD (1982) mesotrophic category based upon mean annual total phosphorus concentration. There is evidence of some decline in concentrations since the peak observed in 2001-2. Lower silica concentrations and deeper Secchi disk depths observed in 2006-7 are consistent with the idea that the catchment may be recovering from a perturbation. Nevertheless there was no significant difference between monthly total phosphorus concentrations in 2001-2 and 2006-7 and high phosphorus concentrations remain a serious cause for concern. While the impacts of a widespread catchment perturbation may be exerting less nutrient pressure upon the lake, increases in loading from more diffuse sources may be contributing to rising levels.

Both mean and maximum chlorophyll values recorded since the 1990’s have consistently shown oligo-mesotrophic characteristics. In concert with the annual wind-induced mixing regime a threshold level of nutrients exists in humic stained lakes beyond which light limits phytoplankton productivity rather than nutrients. Under such circumstances the use of chlorophyll concentrations for trophic classification becomes inadequate in much the same way as Secchi disk depths fail to reflect algal abundance. The development of additional parameters for the trophic classification of humic stained lakes would be beneficial for both for individual lake assessment and for examining lake responses over longer timescales (White & Irvine, 2003; McCarthy *et al.*, 1999).

The results of three annual monitoring surveys have shown the absence of a phytoplanktonic response to phosphorus enrichment. On the basis of the 1990 results phosphorus limitation could be expected to occur at some point below a mean annual concentration of 19µg L^{-1}. At the whole lake scale these results are unlikely to provoke calls for urgent action. However the higher phosphorus concentrations observed in recent years relative to 1990 can be expected to result in a greater frequency and severity of algal blooms. Many areas used for recreational purposes are situated in relatively quiet backwaters where unsightly and in some cases hazardous cyanobacterial surface scums form most rapidly, posing a serious threat to the amenity value of the lough.

Biological Limnology

Marked changes in the zooplankton community and more subtle changes within the phytoplankton community observed between 1990 and 2007 suggest that inputs of terrestrially derived carbon may have increased, stimulating components of the plankton either directly or via the microbial loop.

Phytoplankton

Within the phytoplankton community the abundance of Dinophytes and Cryptophytes was significantly higher in 2007 compared to previous years (figures 40 & 41) and two additional species were recorded for the first time. Many species of Dinophyta and Cryptophyta have the capacity for mixotrophic nutrition. Mixotrophic algae are able to supplement photoautotrophy in the light by assimilation of particulate and/or dissolved organic compounds in the dark (via phagotrophy and diffusion).

Humic stained lakes such as Lough Melvin receive high concentrations of terrestrial organic matter, microaggregates of which are always abundant in the plankton. These microaggregates can form foci for bacteria and can serve as significant food supplements for mixotrophic algae & ciliates (Laybourn-Parry *et al.*, 1992, 1994). Dissolved organic compounds can also be metabolised by bacteria (Tranvik, 1988, 1992; Moran & Hodson, 1990) providing another pathway for utilisation. Increases in the abundance of mixotrophic algae, particularly in view of Lough Melvin’s light limited environment, suggest that the supply of terrestrial organic compounds may have increased.

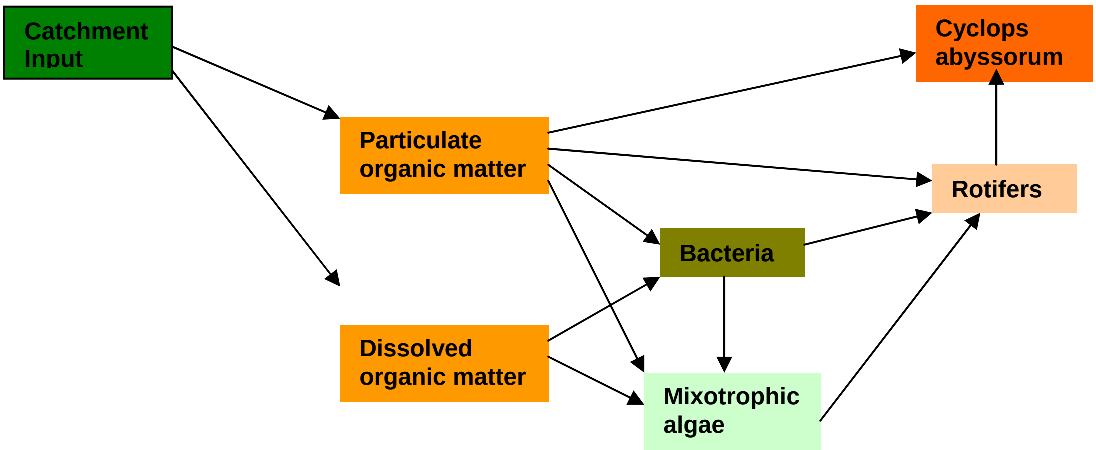


Figure 46. Simplified diagram showing potential pathways of terrestrial support of components of a pelagic food web

Zooplankton

Higher abundances of the dominant rotifer species present in the lough were observed in 2001/02 and 2006/07 compared to 1990, and the rotifer, *Conochilus unicornis* was recorded for the first time in 2001/02. Rotifers depend to a large extent upon bacterial grazing to gain nutrition and can therefore be expected to benefit from increased bacterial production resulting from conditions of greater organic matter loading, providing a greater abundance of bacterial substrates.

The temporal occurrence of *Conochilus unicornis* appeared unrelated to algal abundance (fig. 18) and suggests that this species utilises other dietary sources. In fact colonies were sufficiently abundant in 2006 to allow carbon and nitrogen stable isotope analysis on three consecutive fortnightly occasions. The results of which were consistent with a diet composed almost exclusively of matter derived from catchment terrestrial vegetation. The cyclopoid copepod, *Cyclops strenuus abyssorum* has attained significantly higher abundances in each successive monitoring period. This species is omnivorous and feeds raptorially with taxa such as rotifers and ciliates constituting a large part of its diet. Greater production of these prey items in response to increased organic matter loading may therefore have stimulated production by *Cyclops*.

Bacteria, ciliates, protozoa and mixotrophic algae that can utilise terrestrially derived organic carbon represent potential trophic pathways to zooplankton and higher consumers. Although consumers should select the most energetically optimal food source, in this case phytoplankton, during periods of low phytoplankton productivity terrestrial carbon subsidies may play an important role in supporting the growth and production of microbes and zooplankton.

Appendix I

Table 12. Species of Crustacean and Rotiferan zooplankton observed in Lough Melvin to date. Species observed only in one particular year are marked [1990, 2001/02, 2006/07] respectively.

	1990	2001/02	2006/07
Rotifera			
Keratella cochlearis (Gosse 1851)	OO	OOO	OOO
Kellicottia longispina (Kelicott 1879)	OO	OO	OO
Keratella testudo (Ehrenberg 1832) [2001/02]			
Keratella quadrata (Müller 1786)			
Polyarthra dolichoptera Idelson 1925	OOO	OOO	OOO
Euchlanis dilatata Ehrenberg 1832			
Conochilus unicornis (Rousselet 1892)		OOO	OOO
Conochilus hippocrepis (Schrank 1803)	OOO	O	OO
Filinia terminalis (Plate 1886)	OO	O	O
Trichocerca cylindrica (Imhof 1891)			
Brachionus urceolaris Müller 1773			
Asplancha multiceps (Schrank 1793)			
Asplancha priodonta Gosse 1850			
Ploesoma hudsoni (Imhof 1891)			
Polyarthra vulgaris Carlin 1943[2001/02]			
Testudinella patina (Hermann 1783) [2001/02]			
Crustacea			
Copepoda: Calanoida			
Eudiaptomus gracilis (Sars, 1863)	O	OO	O
Arctodiaptomus laticeps (Sars 1863)	O	O	O
Copepoda : Cyclopoida			
Cyclops strenuus abyssorum Sars	O	O	OO
Cladocera			
Daphnia hyalina var. *galeata* Sars	OO	OO	OO
Daphnia hyalina var. *lacustris* Sars			
Daphnia cucullata var. *kahlbergensis* Schödler			
Bythotrephes longimanus Leydig			
Leptodora kindti (Focke)			
Chydorus ovalis Kurz[2001/02]			
Diaphanosoma brachyurum Liéven[1990]			
Peak Abundance	**> 10000 m^{-3}**	**OOO**	
	1000 – 9999 m^{-3}	**OO**	
	< 1000 m^{-3}	**O**	

Table 13. Phytoplankton species observed from Lough Melvin to date. August 1951 (Macan & Lund, 1954), September 1953 (Brook, 1958; Round, 1959; Round & Brook, 1959) and September 1983 (DARD, unpublished data)

Species	**Aug 1951** net haul	**Sept 1953** net haul	**Sept 1953** sediment	**Sept 1983** net haul	**1990**	**2001/02**	**2006/07**
Phylum Cyanophyta Blue-Green Algae							
Microcystis aeruginosa (Kützing)	■	■			■	■	■
Woronichinia naegliana (Unger) Elenkin [*Coelospaerium naeglianum* Unger]						■	■
Coelospaerium küetzingianum Nägeli		■				■	■
Merismopedia punctata Meyen		■					
Merismopedia glauca (Ehrenberg) Nägeli		■					■
Chroococcus limneticus Lemmerman		■		■			■
Anabaena flos aquae [(Lyngbye) Brébisson]	■	■		■	■	■	■
Anabaena circinalis [Rabenhorst]						■	■
Chaemaesiphon sp.						■	
Aphanizomenon flos aqauae [(Linnaeus) Ralfs]				■		■	■
Oscillatoria agardhii Gomont	■	■			■	■	■
Oscillatoria agardhii var. *isothrix* Skuja							■
Oscillatoria bourellyi Lund						■	
Oscillatoria redekei Goor					■	■	■
Phormidium mucicola Huber-Pestalozzi & Naumann	■						
Aphanothece clathrata West & West	■						■
Aphanothece nidulans Richter	■						
Snowella lacustris (Chodat) Komárek & Kindák							■
Phylum Bacillariophyta Diatoms							
Aulacoseira varians (Agardh) Simonsen		■		■			■
Aulacoseira granulata (Ehrenberg) Simonsen		■			■		■
Aulacoseira italica var. *subarctica* (Müller) Simonsen		■		■	■	■	■
Aulacoseira ambigua (Grunow) Simonsen				■		■	■
Aulacoseira islandica (Müller) Simonsen				■		■	■
Tabellaria fenestrata var. *asterionelloides* Grunow			■			■	■
Tabellaria flocculosa (Roth) Kützing		■	■			■	■
Tabellaria flocculosa var. *pelagica* Holmboe		■					

Species	Aug 1951 net haul	Sept 1953 net haul	Sept 1953 sediment	Sept 1983 net haul	1990	2001/02	2006/07
Tabellaria flocculosa var. *teilingii* Knudson		■					
Tabellaria fenestrata (Lyngbye) Kützing		■					
Fragilaria crotonensis Kitton		■		■			■
Fragilaria capucina Desmazières		■					
Fragilaria intermedia Grunow			■				
Fragiliaria harissonii			■				
Asterionella formosa Hassall		■		■	■	■	■
Asterionella formosa var. acaroides Lemmerman		■					
Synedra ulna var. *danica* (Kützing) Grunow		■		■	■	■	■
Stephanodiscus neoastrae Grunow			■	■	■	■	■
Gyrosigma sp. Hassall			■			■	■
Surirella sp. Turpin						■	■
Opephora martyi Héribaud			■				
Eunotia pectinalis var. *minor* of. *Impressa* (Her)			■				
Cocconeis placentula (Her)			■				
Diatoma sp. Bory de St-Vincent					■		■
Skeletonema sp. Greville					■		
Phylum Chlorophyta Green algae							
Eudorina elegans Ehrenberg		■				■	■
Pseudospaerocystis neglecta (Teiling emend. Skuja) Bourelly		■					
Pediastrum duplex Meyen		■				■	■
Pediastrum boryanum (Turpin) Meneghini		■				■	■
Closteriopsis longissima Lemmerman		■				■	■
Botryococcus braunii Kützing		■		■		■	■
Stylosphaeridium stipitatum (Bachman) Geitler & Gemesi		■					
Kirchneriella obesa (West) Schimle							■
Scenedesmus communis Hegewald				■	■	■	■
Dictyosphaerium pulchellum (Wood)				■			■
Ankistrodesmus falcatus (Corda) Ralfs	■					■	■

Species	Aug 1951 net haul	Sept 1953 net haul	Sept 1953 sediment	Sept 1983 net haul	1990	2001/02	2006/07
Chlorella vulgaris						■	■
Mougeotia sp.(Agardh) Beijerinck						■	
Chlamydomonas sp. Ehrenberg							■
Monoraphidium contortum (Thuret) Komárková-Legnerová							■
Monoraphidium arcuatum (Korshikov) Kindák							■
Coenococcus planktonicus Korshikov							■
Closterium acutum (Lemmerman) Kreiger						■	
Closterium aciculare West		■		■	■		■
Closterium ceratium (*Closterium acutum* var. *ceratium* (Perty) Kreiger)					■		
Closterium setaceum Ehrenberg ex. Ralfs						■	
Closterium dianae Ehrenberg ex Ralfs						■	■
Cosmarium depressum (Nägeli) P.Lundell		■		■		■	■
Staurodesmus dejectus		■				■	
Staurodesmus dejectus var. *inflatus*		■					
Staurastrum furcigerum		■		■		■	■
Staurastrum lunatum Ralfs	■					■	■
Staurastrum lunatum var. *planktonicum* (West & West)		■					
Staurastrum longipes fac. Quadrata		■		■		■	
Ulothrix sp. Kützing						■	
Phylum Cryptophyta							
Cryptomonas sp. Ehrenberg					■	■	■
Cryptomonas erosa Ehrenberg							■
Rhodomonas lacustris var. *nannoplanktonica* (Skuja) Javornický				■		■	■
Phylum Pyrrophyta Dinoflagellates							
Ceratium hirundinella (Müller) Dujardin		■				■	■
Ceratium furcoides (Levander) Langhans							■
Gymnodinium helveticum (Penard)				■		■	■
Gymnodinium discoidale Harris							■

Species	Aug 1951 net haul	Sept 1953 net haul	Sept 1953 sediment	Sept 1983 net haul	1990	2001/02	Specie s
Phylum Chrysophyta Golden brown algae							
Mallomonas sp.						■	■
Mallomonas caudata Ivanov emend. Wili Kreiger							■

Appendix II

Proposal I. Grounds for the establishment of a long term monitoring strategy for Lough Melvin

Three year long monitoring programmes in addition to specific lake monitoring between each has demonstrated that Lough Melvin is under increasing pressure from phosphorus enrichment, and that lake concentrations are expected to rise in the future. While stakeholders have demonstrated a strong willingness to remedy the situation through a number of measures and agri-environmental schemes there is no framework for assessing how effective these measures will be or if they will be sufficient to curb the recent rises in phosphorus export intensity. The LMNRP has shown that cross border cooperation is crucial for the successful management of Lough Melvin as a sustainable resource, but has highlighted the fact that there is no consensus as to who is willing or responsible for monitoring the lake in the future.

There are strong grounds for monitoring Lough Melvin, not only to asses its changing status and the efficacy of management strategies, but also as a means to examine lake-catchment dynamics at the regional scale. The basis for this argument is that while we may be able to mitigate against the direct pressures exerted upon freshwaters through agri-environmental and socio-economic management practices, changes in climate related biogeochemical cycling are largely beyond the reach of management at the catchment scale.

The Global Lake Ecological Observatory Network (GLEON) is an organisation that links limnological data from lakes distributed across the world. Monitoring is carried out largely by the use of remote sensing instruments and the database forms a powerful tool for examining climate related impacts upon freshwaters. Currently there is one GLEON site in the Republic of Ireland, however this is of a very different lake type to Lough Melvin, which possesses characteristics that are representative of the unique lake types found in this region.

We propose the establishment of a long term monitoring strategy for Lough Melvin, involving the use of remote sensing technology that is also linked to a global database of lake observatories such as GLEON.

Proposal II: Assessing the impact of changes in terrestrial organic matter loading upon primary production

The effects of terrestrially derived organic carbon upon lake ecosystems are numerous. In the context of Lough Melvin, which receives a considerable input, they exert a considerable influence by causing the rapid attenuation of light. Despite sustained phosphorus enrichment in recent years the typical phytoplankton response has been absent because light attenuation caused by terrestrial organic matter has limited primary productivity rather than nutrient availability.

On one hand terrestrial organic matter loading can therefore be considered as beneficial in regard to elevated nutrient concentrations, but on the other greater terrestrial carbon loads will cause a concomitant increase in light attenuation, which can be expected to lead to reductions in pelagic (Jansson *et al.,* 2003), benthic-littoral (Qin *et al.,* 2007) and subsequent higher consumer productivity (Vander Zanden & Vadeboncoeur, 2002). In Lough Melvin, where several salmonid species are of high conservation status and posses a significant recreational and socio-economic importance, lower ecosystem productivity can be expected to result in lower salmonid production.

Observations of rising fluxes of organic carbon to watercourses from a variety of different geographical regions in the last 40 years (Forsberg, 1992; Freeman et al., 2001; Tranvik & Jansson, 2002) in addition to circumstantial evidence that Lough Melvin may be subject to greater loads reinforce the need for additional studies in this area.

We propose a study that will:

i. Utilise existing data generated under the LMNRP to develop an organic carbon loading model for the Lough Melvin

ii. Quantify how increasing exports of terrestrially derived organic matter will affect primary productivity within the lake

iii. Develop a model of biomass production within Lough Melvin based upon changes in land use, nutrient status and allochthonous carbon loading.

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