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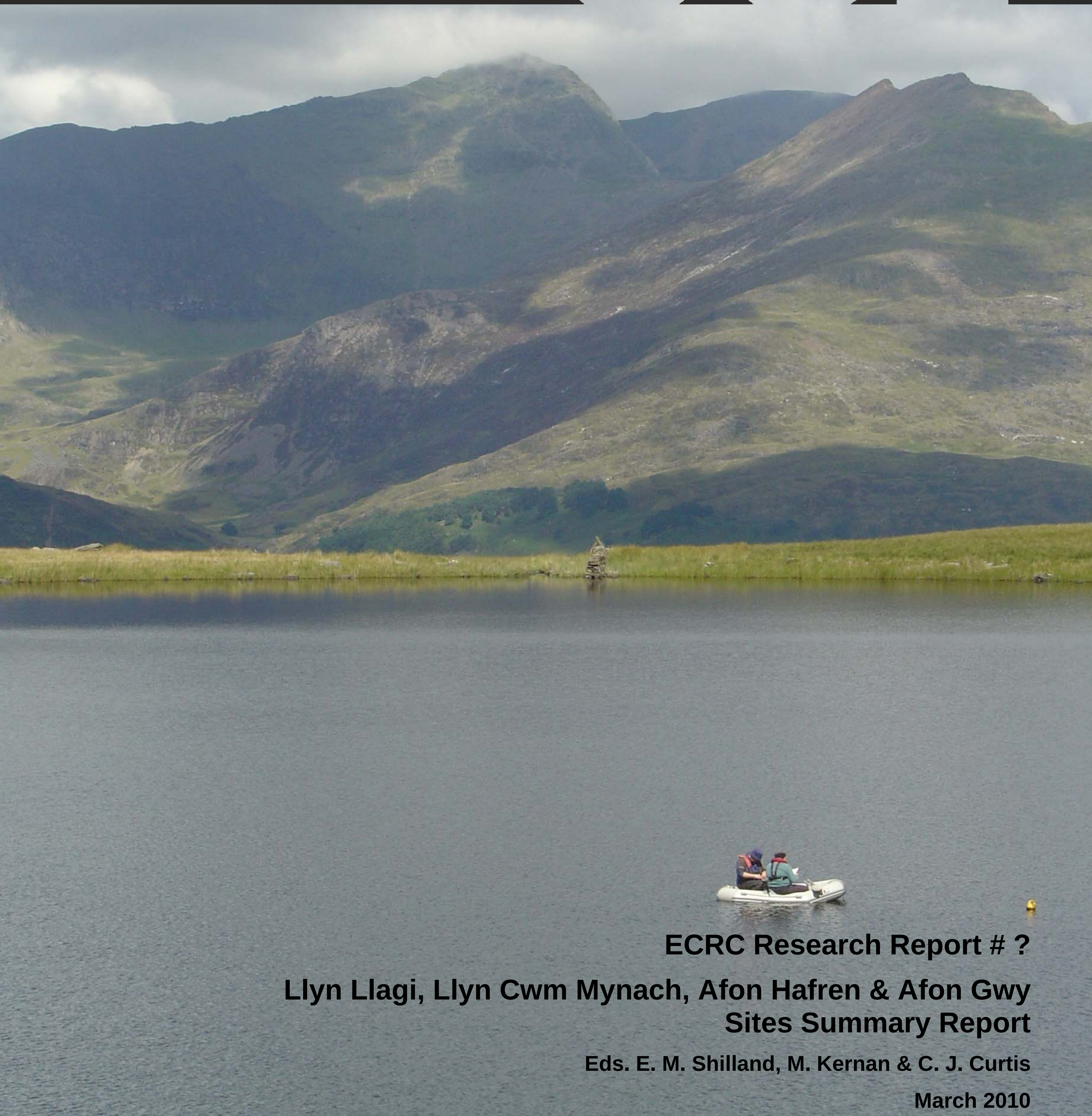

ECRC Research Report # ?

Llyn Llagi, Llyn Cwm Mynach, Afon Hafren & Afon Gwy Sites Summary Report

Eds. E. M. Shilland, M. Kernan & C. J. Curtis

March 2010

UK ACID WATERS MONITORING NETWORK (AWMN)

LLYN LLAGI, LLYN CWM MYNACH, AFON HAFREN AND AFON GWY. SITES SUMMARY REPORT

REPORT TO THE COUNTRYSIDE COUNCIL FOR WALES

Eds. E. M. Shilland, M. Kernan & C. J. Curtis

March 2010

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5 Llyn Llagi

Figure 5.1. Llyn Llagi.

5.1 Site and Catchment Characteristics

Falling within the Snowdonia National Park, the Llyn Llagi catchment (Figure x and x, Table x) covers 152 ha and reaches a maximum altitude of 672 m. The geology consists primarily of Ordovician slates and shales of the Glanarfon series. The backwall is composed of a large doleritic intrusion with small intrusions of fine microgranites and volcanic tuff. Away from the precipitous bare rock of the backwall the catchment soils are chiefly stagnopodsols interspersed with blanket peats.

The lake and its catchment receive an annual rainfall of c. 3000 mm, with mean non-marine sulphate deposition of 0.53 kiloequivalents ha^{-1} yr^{-1} and mean oxidised nitrogen deposition of 0.48 kiloequivalents ha^{-1} yr^{-1} (Table x).

The catchment is unafforested and characterised by acid moorland species, notably *Calluna*, *Festuca, Nardus, Molinia* and *Eriophorum*. Palynological studies by Patrick *et al.* (1987) showed that during the recent history of the catchment *Calluna* declined whilst *Gramineae* species increased. The vegetation is grazed at a low intensity by sheep but there is no other contemporary land-management.

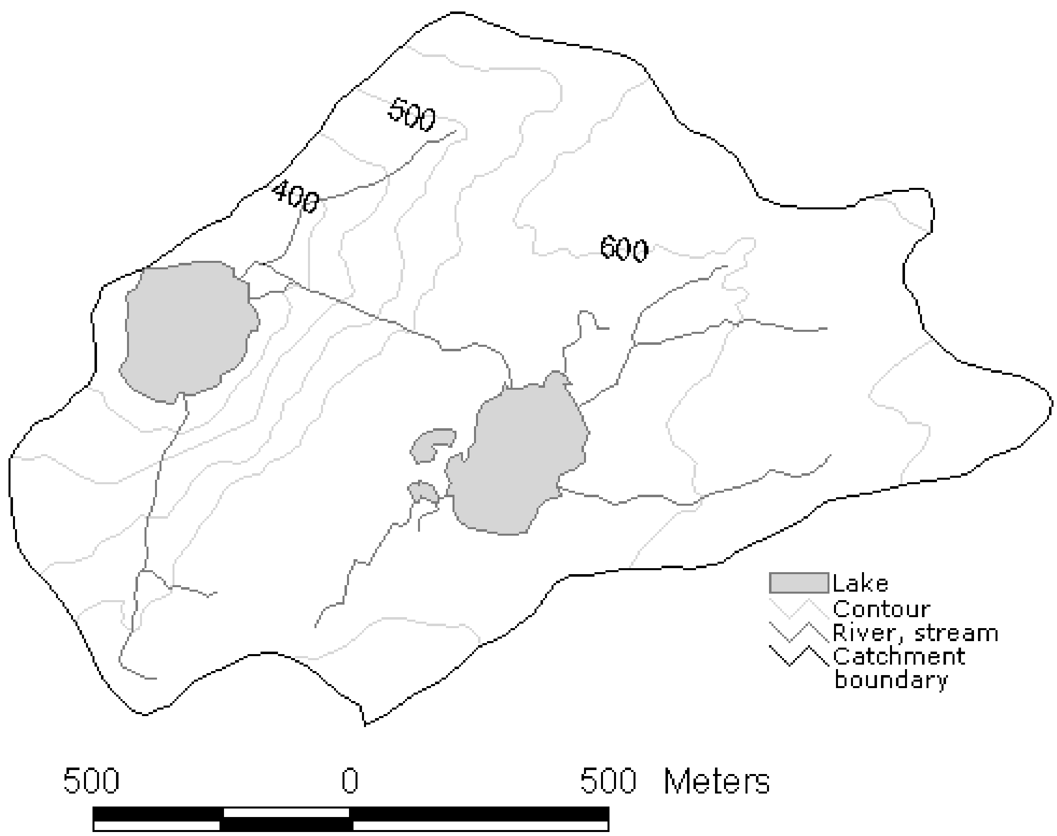
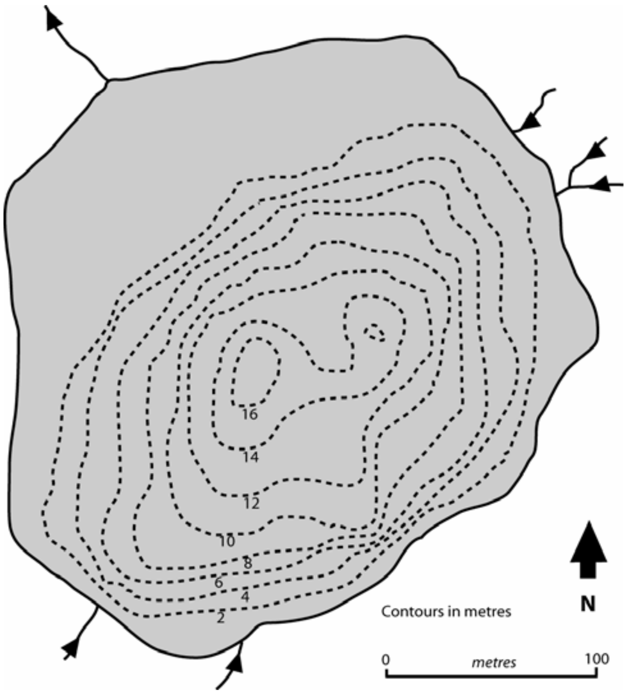


Figure 5.2. Llyn Llagi catchment



Llyn Llagi occupies a north-facing corrie in the central area of the Snowdonia region of north Wales. The lake lies at 375 m beneath a steep backwall and comprises a deep, almost circular, basin (maximum depth 16.5 m) bordered to the north by an extensive shallow (1 m deep) rim (Figure x). The lake covers an area of 5.1 ha and receives discrete drainage from a series of small streams draining the steeper part of the catchment. The primary inflow constitutes the outflow stream from Llyn yr Adar which lies above the backwall. The lake drains to the north-west to the Afon Nanmor, a tributary of the Afon Glaslyn.

Figure 5.3. Bathymetry, Llyn Llagi

Table 5.1 Site characteristics, Llyn llagi

Grid Reference	SH 649483
Lake Altitude	375 m
Maximum depth	16.5 m
Mean Depth	5.8 m
Volume	295000 m^3
Lake Area	5.1 ha
Catchment area	152 ha (excluding lake)
Catchment;lake ratio	30.84
Maximum catchment altitude	672 m
Catchment geology	Ordovician slates and shales. Doleritic and volcanic intrusions
Catchment soils	56% Stagnodposols
	44% Raw Peat Soils
Catchment landcover	7% Water (inland)
	4% Dense dwarf shrub heath
	6% Open dwarf shrub heath
	<1% Broad-leaved / mixed woodland
	<1% Coniferous woodland
	2% Improved grassland
	<1% Bracken
	75% Acid grassland
	<1% Suburban / rural development
	6% Inland bare ground

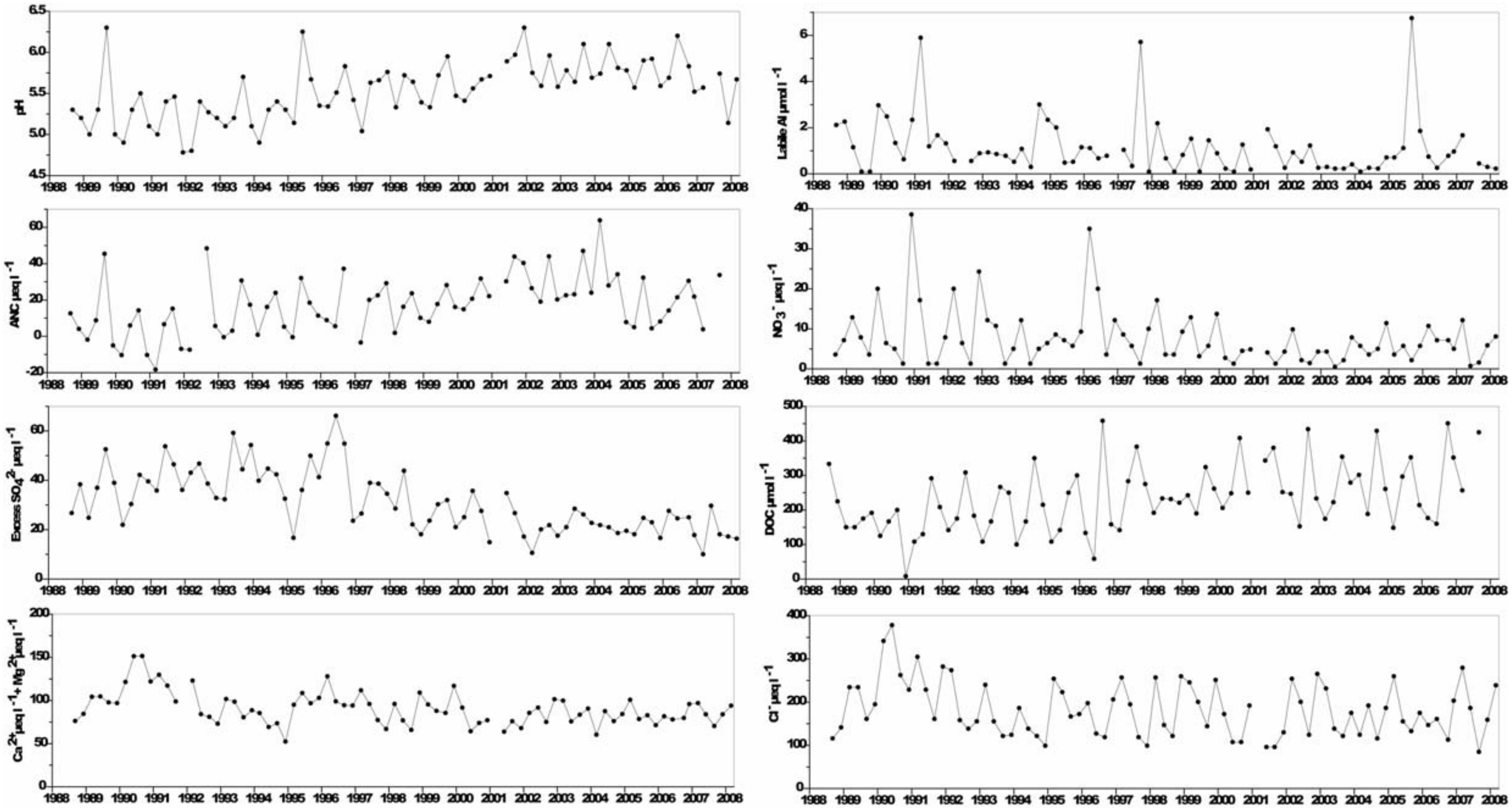
Mean annual rainfall	2941 mm

5.2 Deposition

Table 5.2. Deposition, Llyn Llagi

Mean total sulphate deposition*	0.83 kiloequivalents ha^{-1} yr^{-1}
Mean non-marine sulphate deposition*	0.53 kiloequivalents ha^{-1} yr^{-1}
Mean oxidised nitrogen deposition*	0.48 kiloequivalents ha^{-1} yr^{-1}
Mean reduced nitrogen deposition*	0.57 kiloequivalents ha^{-1} yr^{-1}

5.3 Hydrochemistry



µeq L^{-1} except * (µmol L^{-1})	pH	*ANC	Ca^{2+}	Mg^{2+}	Na^{+}	K^{+}	Si	*Soluble Al	*labile Al	Cl^{-}	SO_4^{2-}	Excess SO_4^{2-}	NO_3^{-}	DOC
mean 1st 5 yrs	5.23	5.86	57.06	50.53	187.40	3.60	34.96	2.81	1.58	222.40	62.17	38.82	10.42	177.90
07-08 mean	5.52	33.68	42.69	40.32	150.40	4.12	28.60	1.07	0.32	167.07	38.38	20.87	4.05	424.61
07-08 std.dev	0.33		2.45	7.89	34.14	1.73	5.00	0.23	0.11	64.17	9.70	6.43	3.50	

Figure 5.4. Spot sampled chemistry data, Llyn Llagi

5.4 Thermistors

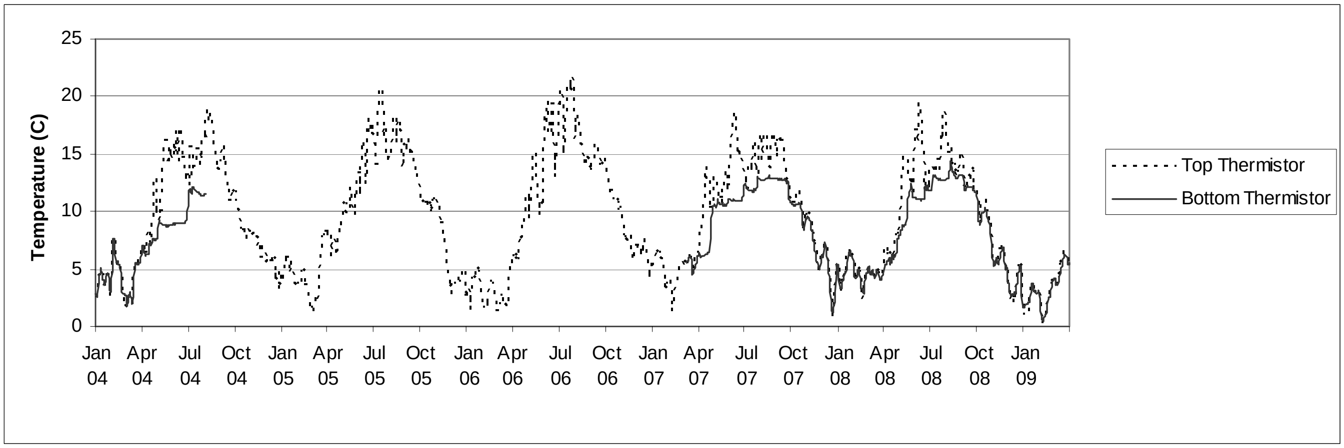


Figure 5.5. Thermistor data, Llyn Llagi

5.5 Diatoms

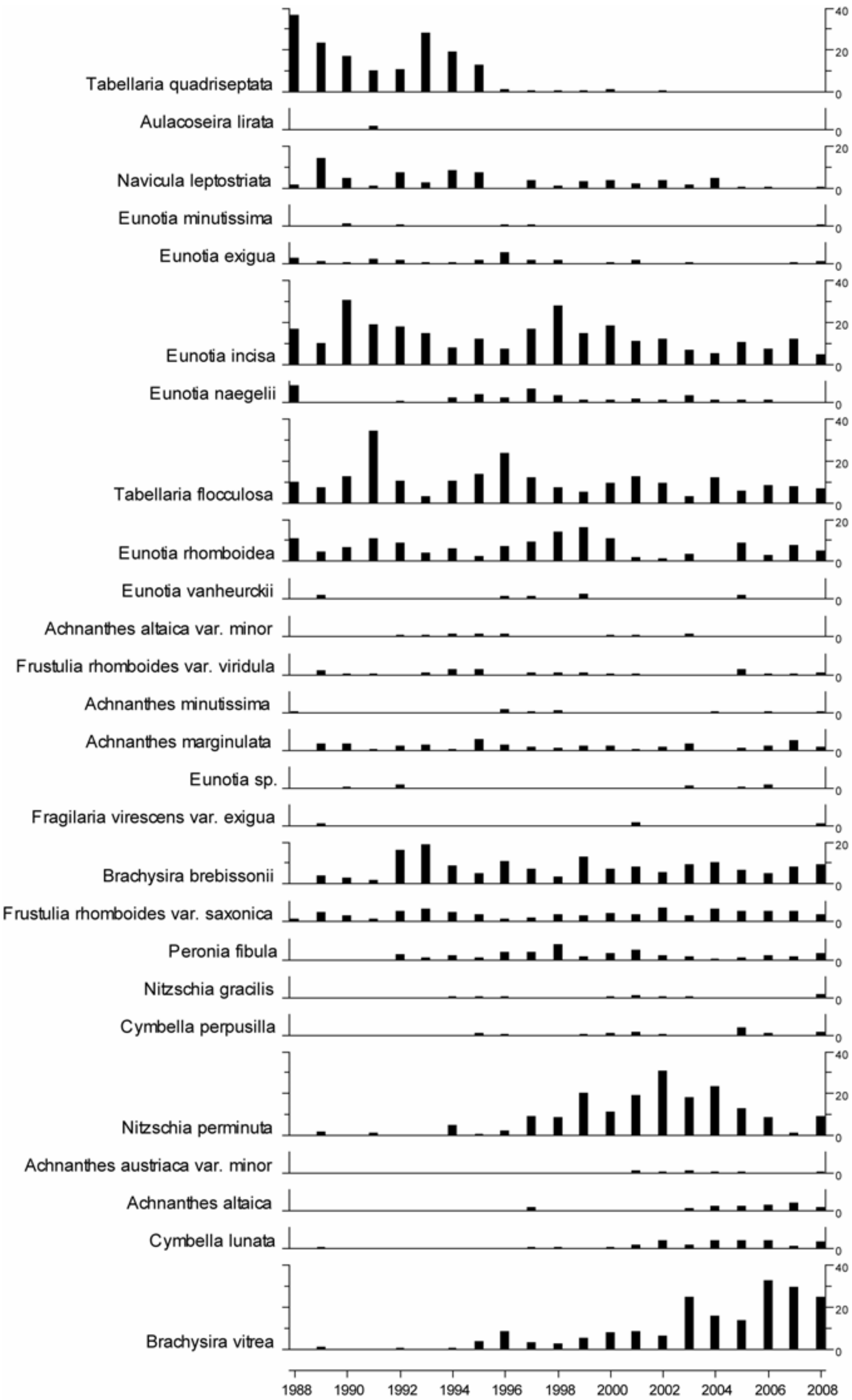


Figure 5.6. Epilithic diatom percentage abundance summary, Llyn Llagi

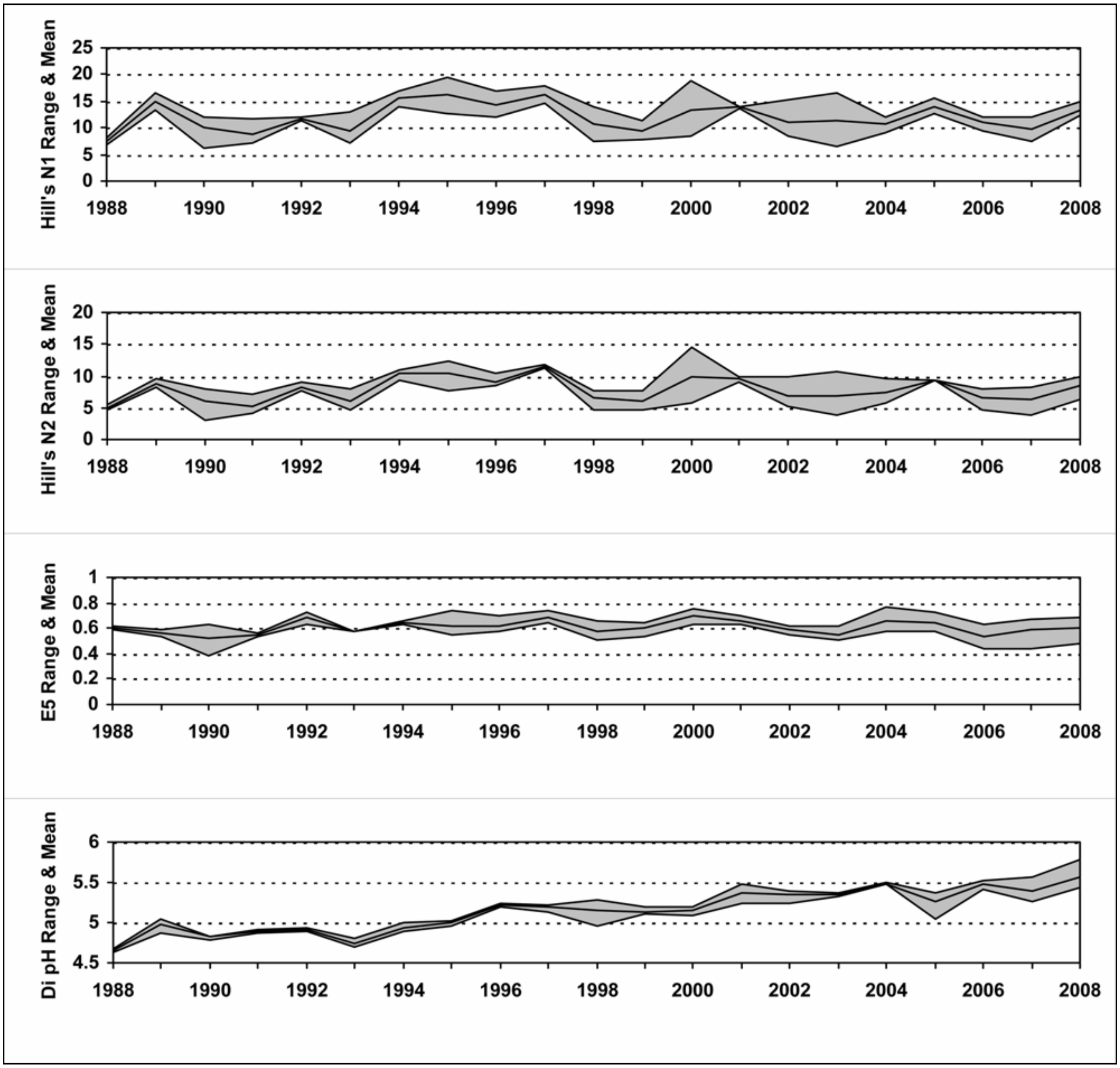


Figure 5.7. Epilithic diatom summary statistics, Llyn Llagi

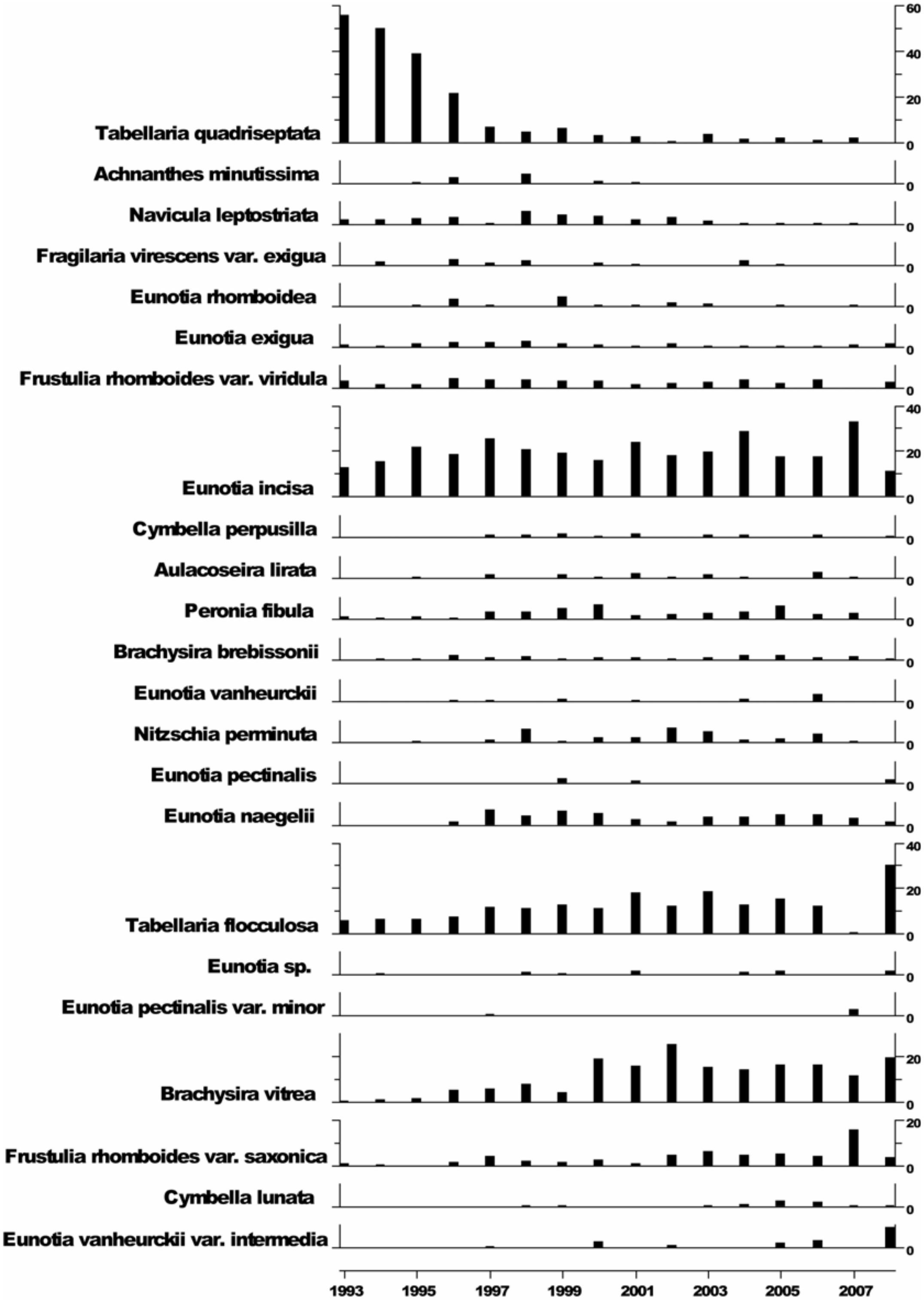


Figure 5.8. Relative percentage frequency of sediment trap diatom taxa, Llyn Llagi

5.6 Macroinvertebrates

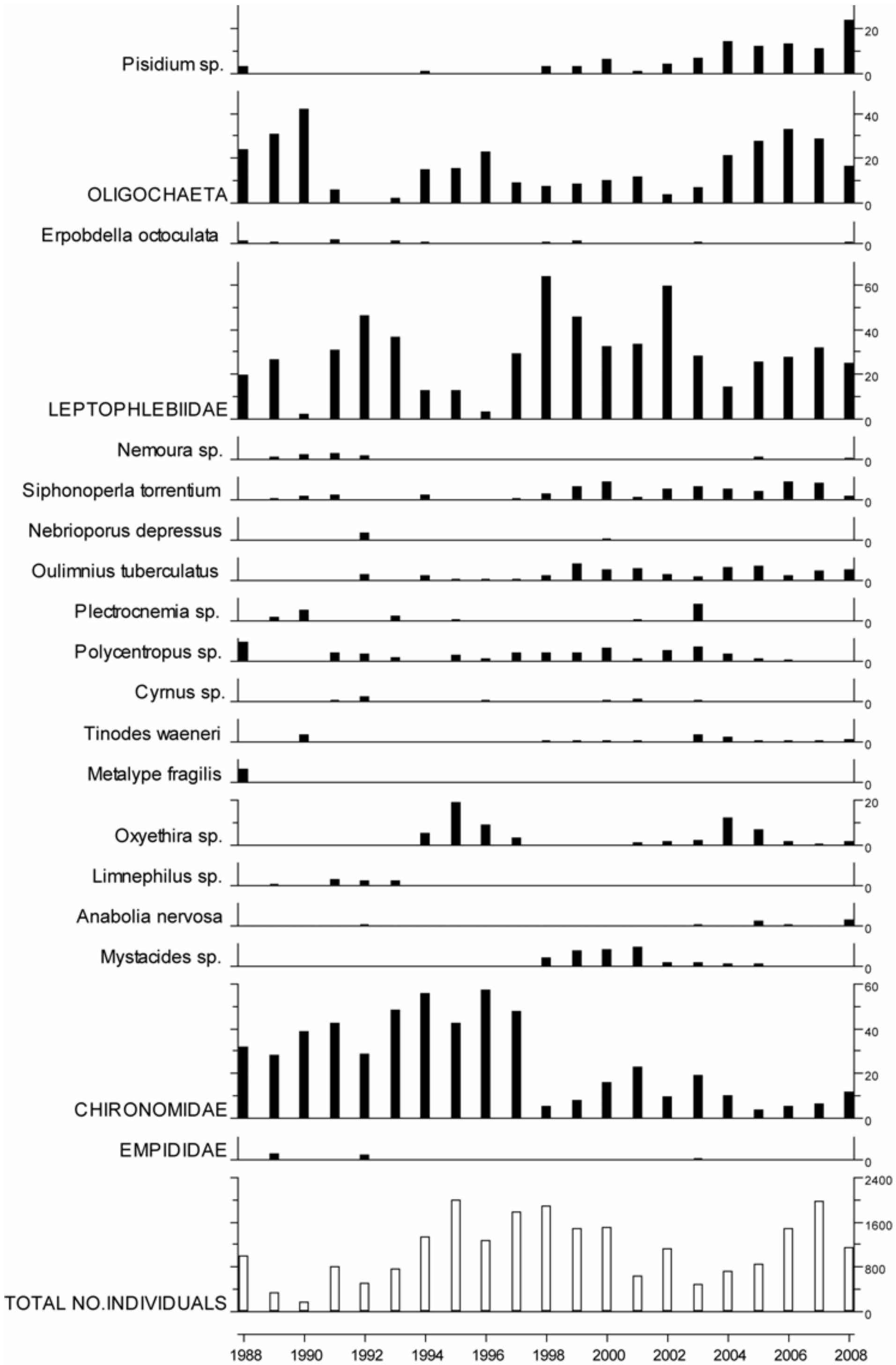


Figure 5.9. Macroinvertebrate percentage abundance summary, Llyn Llagi

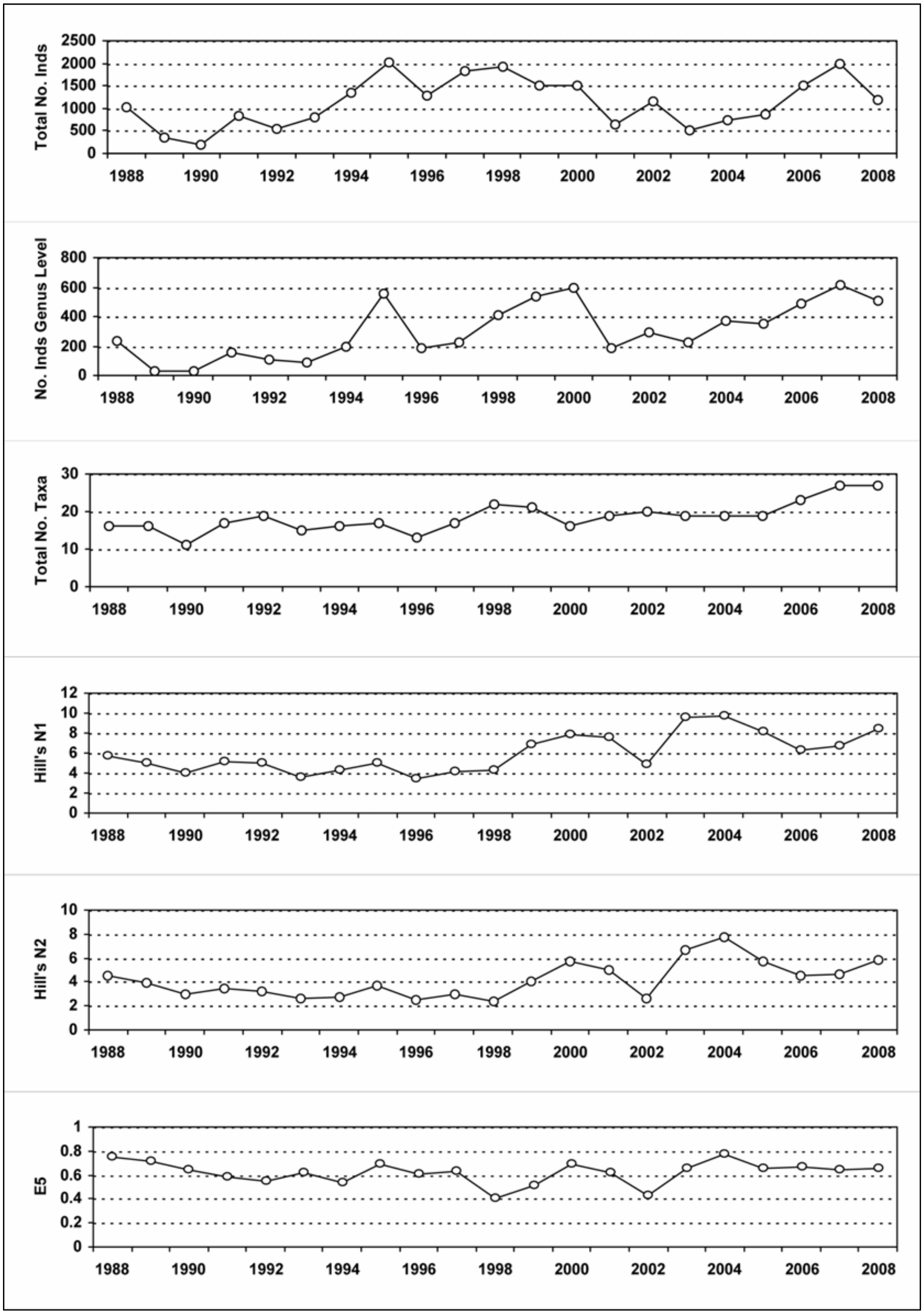
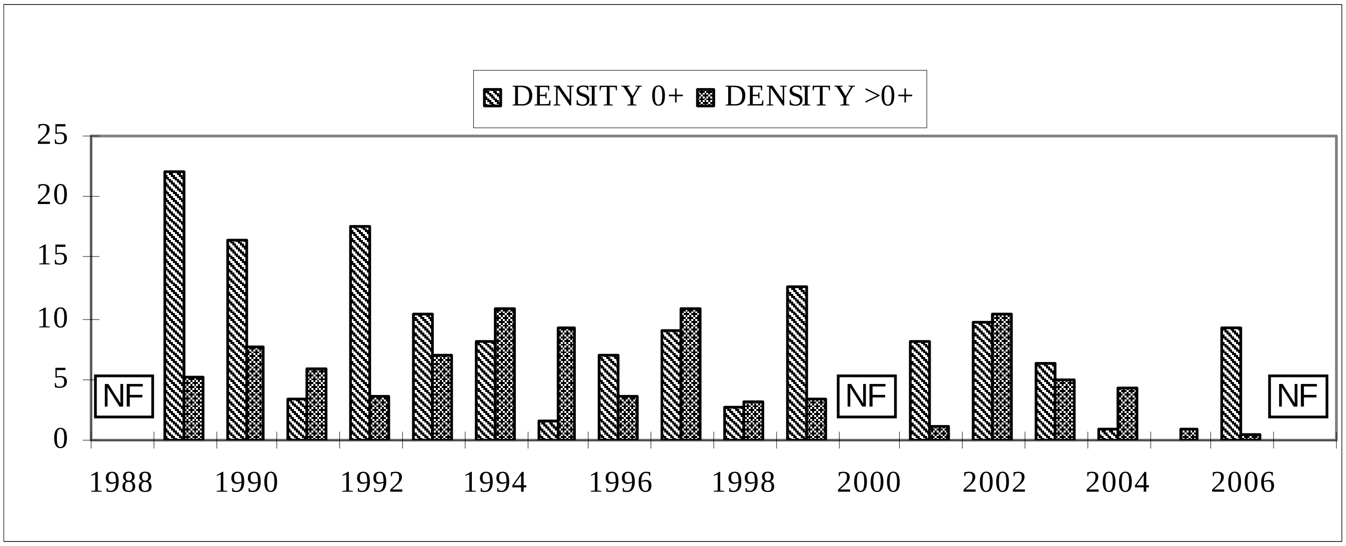


Figure 5.10. Macroinvertebrate summary statistics, Llyn Llagi

5.7 Fish



NF = Not fished

Figure 5.11. Summary of mean Trout density for outflow stream (numbers 100m^{-2}), Llyn Llagi

5.8 Aquatic Macrophytes

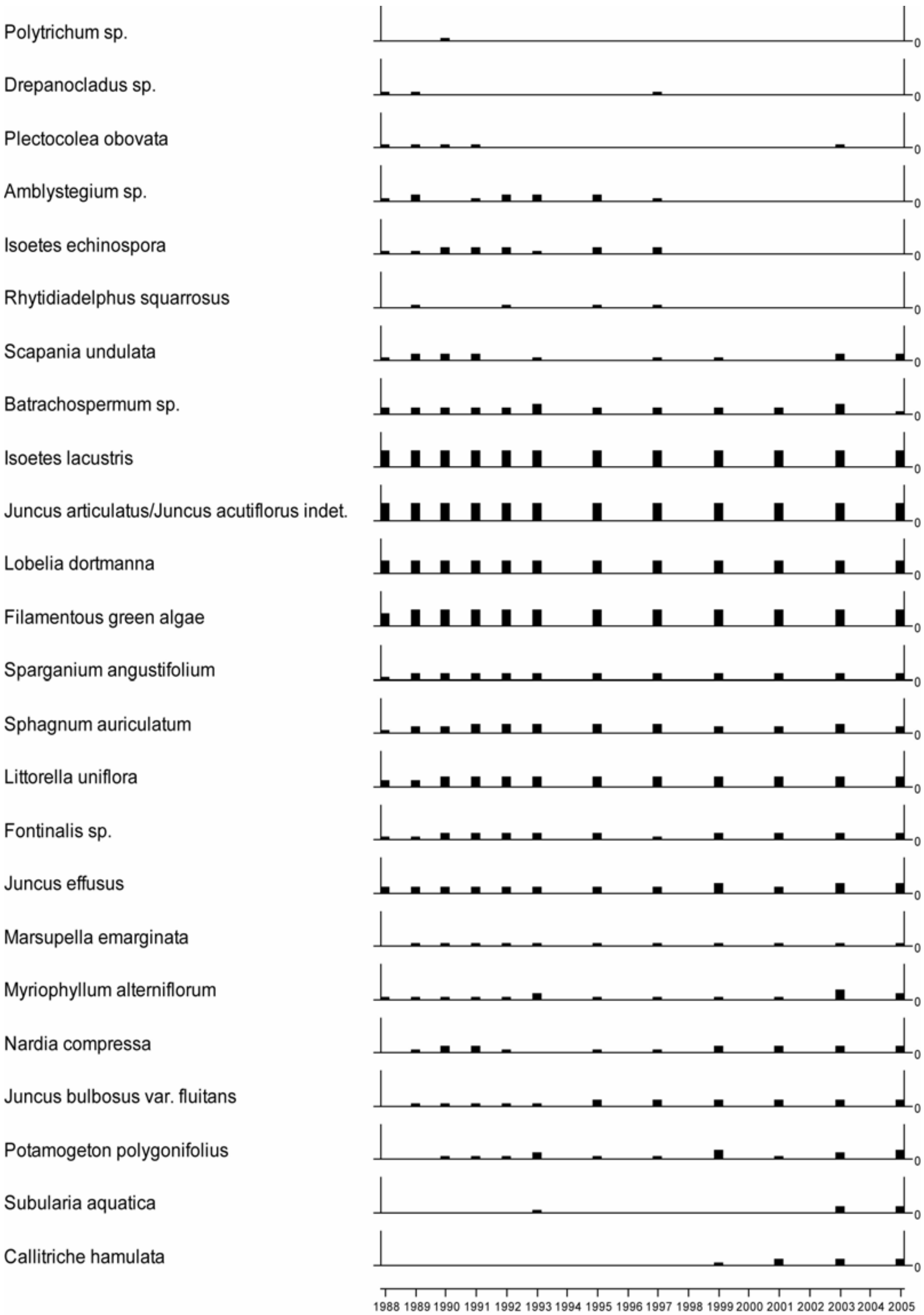


Figure 5.12. Aquatic macrophyte species scores (1-5), Llyn Llagi

At the outset of monitoring the aquatic macrophyte community of Llyn Llagi was dominated by the acid-tolerant isoetids, *Isoetes lacustris*, *Lobelia dortmanna* and *Littorella uniflora*. The more sensitive *Callitriche hamulata* was first recorded in 1999 and is now well established. Another sensitive species, *Subularia aquatica*, was first recorded in 1993, and has been found again in surveys from 2003 onwards. Further increase in the numbers of more sensitive species was observed in 2009, with the discovery of *Nitella flexilis* var. *flexilis* agg. and *Elatine hexandra*. The spatial distribution of the acid-tolerant *Juncus bulbosus* var. *fluitans* appears to have expanded slightly over the full monitoring period.

Llyn Llagi shows the greatest signs of macrophyte recovery of any site in the Network and has gained several new submerged plant species. Accordingly the site has moved from the C1 to the C2 category in Duigan *et al*'s. (2007) lake classification scheme.

5.9 Summary

5.9.1 Chemistry

5.9.2 Biology

5.10 Llyn Llagi Recent Publications Using AWMN Data

Battarbee, R. W., Kernan, M., Monteith, D. T.& Curtis, C. J. (2010) Summary and Recommendations. In: *UK Acid Waters Monitoring Network 20 Year Interpretative Report*, 279-293, ENSIS Ltd, Environmental Change Research Centre, University College London, London.

Curtis, C. J., Simpson, G. L. (2010) Acid Deposition Trends at AWMN Sites. In: *UK Acid Waters Monitoring Network 20 Year Interpretative Report*, 31-52, ENSIS Ltd, Environmental Change Research Centre, University College London, London.

Curtis, C. J., Battarbee, R. W., Heliwell, R., Flower, R. J., Simpson, G. L., Monteith, D. T., Shilland, E. M., Aherne, J.& MacDougall, G. (2010) Recovery Progress: Reference Conditions and Restoration Targets. In: *UK Acid Waters Monitoring Network 20 Year Interpretative Report*, 206-237, ENSIS Ltd, Environmental Change Research Centre, University College London, London.

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Battarbee, R. W., Monteith, D. T., Juggins, S., Simpson, G. L., Shilland, E. M., Flower, R. J.& Kreiser, A. M. (2008) Assessing the accuracy of diatom-based transfer functions in defining reference pH conditions for acidified lakes in the United Kingdom. *The Holocene*, **18**, 57-67.
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6 Llyn Cwm Mynach

Figure 6.1. Llyn Cwm Mynach.

6.1 Site and Catchment Characteristics

Situated at the southern edge of the Rhinog Mountains in north Wales, the catchment of Llyn Cwm Mynach (Figures x and x, Table x) covers an area of 119 ha and reaches a maximum altitude of 680 m. The solid geology of the catchment consists of Cambrian siltstones to the north and grits and greywackes of the Cambrian Rhinog formation to the south. The north and east of the catchment is dominated by amorphous blanket peats, whereas acid ranker soils characterise the south and west. There are areas of exposed rock on the steeper slopes to the west.

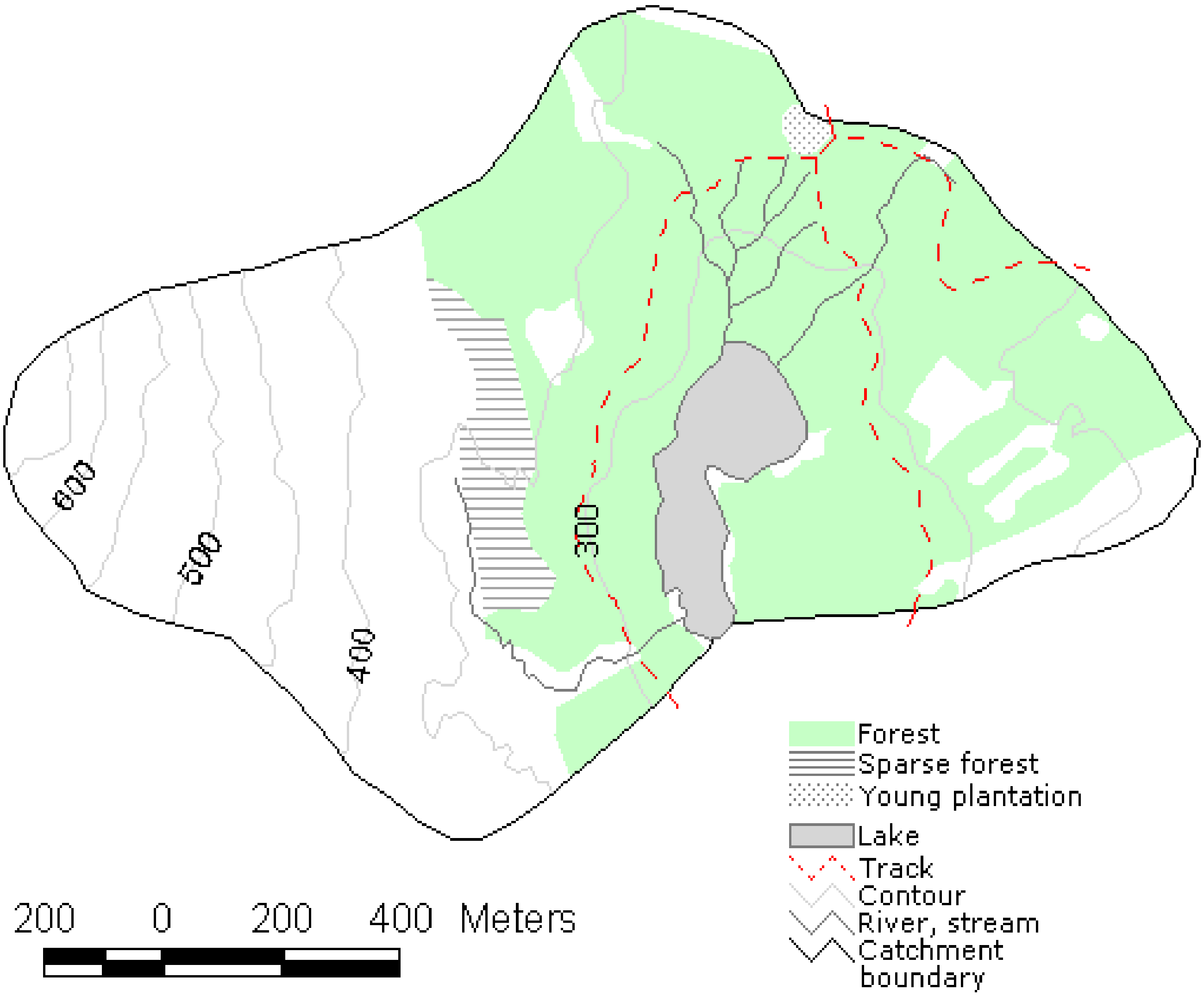


Figure 6.2. Llyn Cwm Mynach catchment

The lake and its catchment receive an annual rainfall of c. 2200 mm, with mean non-marine sulphate deposition of 0.47 kiloequivalents ha^{-1} yr^{-1} and mean oxidised nitrogen deposition of 0.63 kiloequivalents ha^{-1} yr^{-1} (Table x).

Approximately half of the catchment is forested with Japanese larch, Lodgepole pine and Sitka spruce planted between 1967 and 1973. The remainder is acid moorland characterised by *Calluna* and *Vaccinium* and utilised as rough grazing for sheep. The remains of short-lived small manganese mines are to be found the south-west of the catchment but there is no evidence to suggest these abandoned excavations have had any effect on water chemistry during monitoring. (Table x). Both the lake and the catchment lie within the Snowdonia National Park.

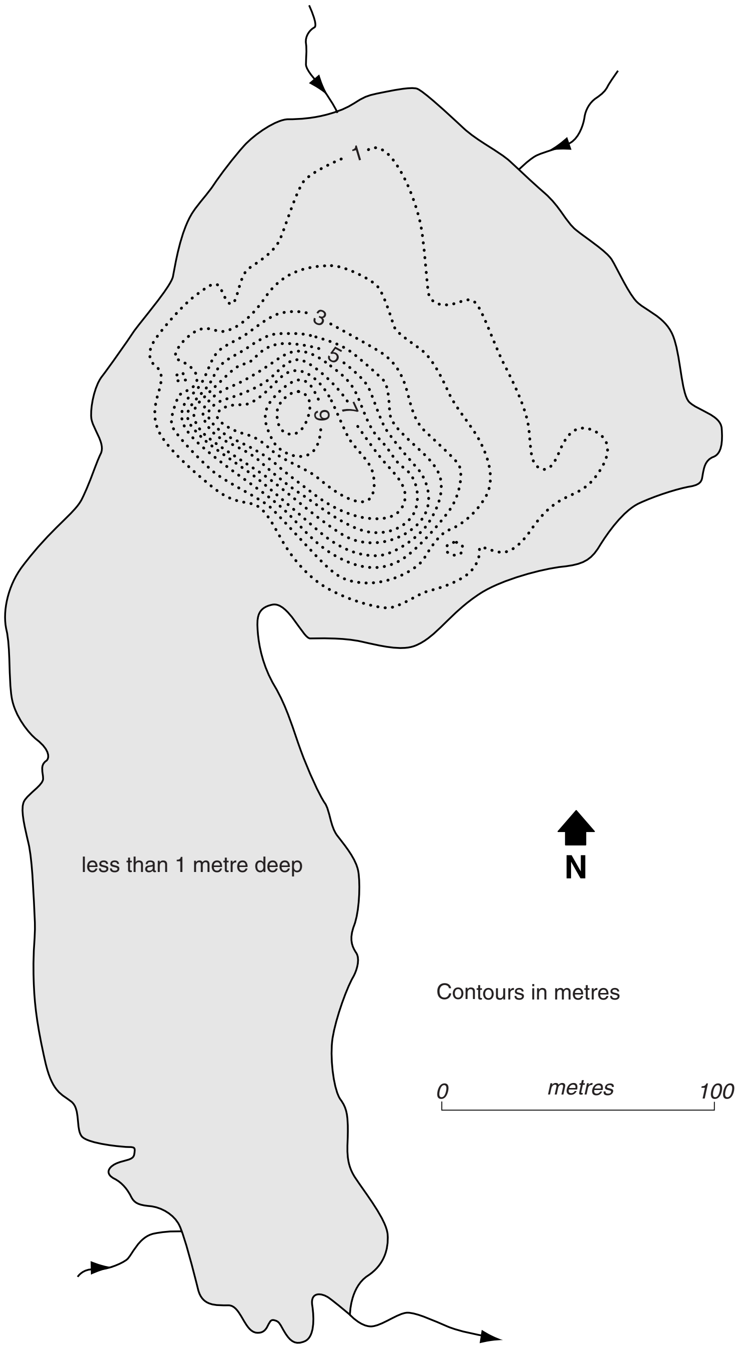


Figure 6.3. Bathymetry, Llyn Cwm Mynach

Llyn Cwm Mynach lies at 287 m altitude. The lake consists of two distinct basins; the southern section comprises a shallow limb separated from the deeper northern section by an old and broken stone causeway. The maximum depth of 11 m occurs in the northern basin in a localised hollow (Figure x). Discrete drainage to the lake is by three small inflows and it is drained to the south-east by a small stream which flows to the Mawddach estuary. At some stage in the past a wooden weir (now derelict) was constructed at the outflow, presumably to raise the level of the southern section of the lake.

Table 6.1 Site characteristics, Llyn Cwm Mynach

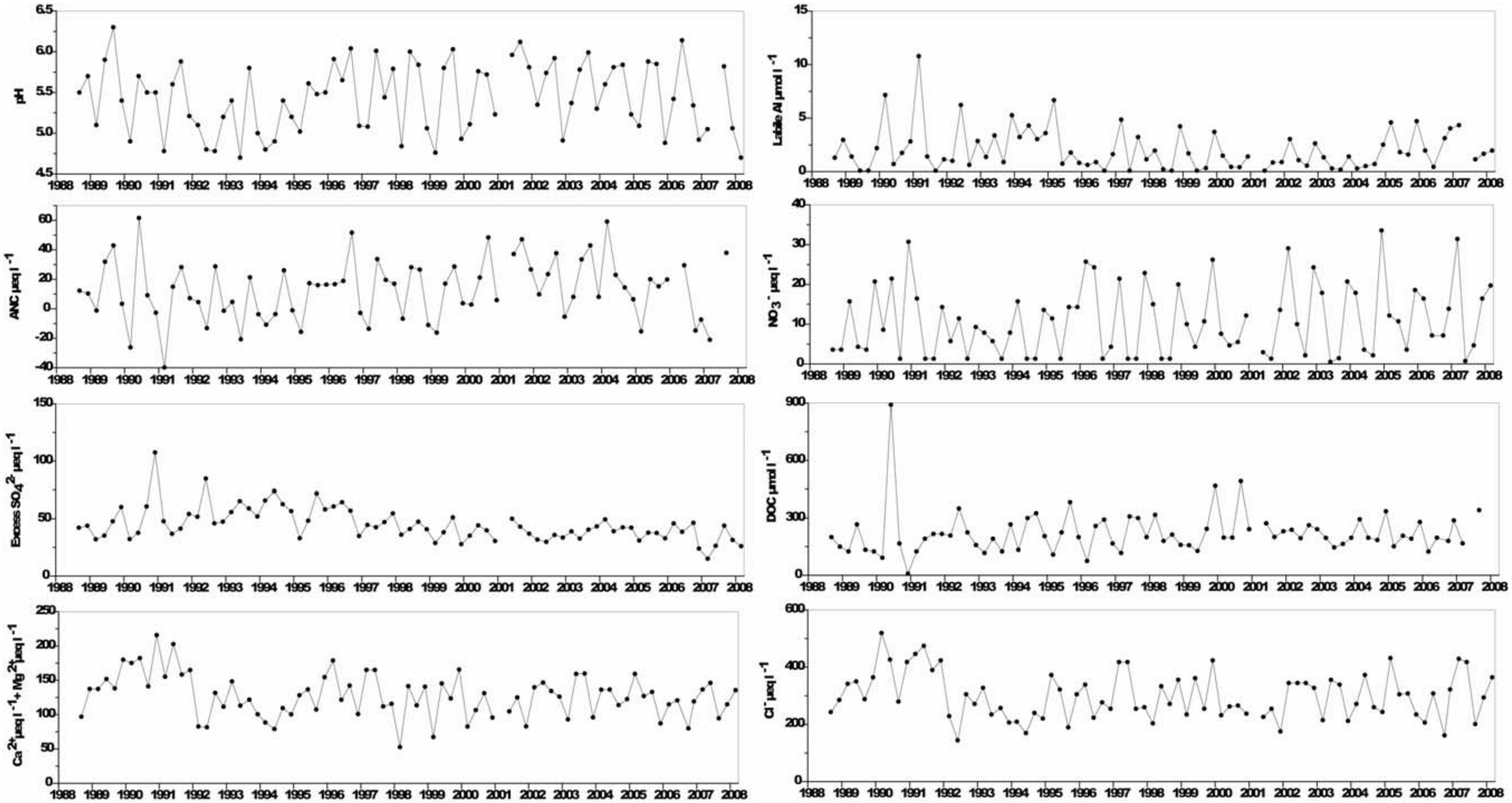
Grid Reference	SH 678238
Lake Altitude	287 m
Maximum depth	11 m
Mean Depth	0.9 m
Volume	51000 m^3
Lake Area	5.7 ha
Catchment area	119 ha (excluding lake)
Catchment;lake ratio	22.04
Maximum catchment altitude	680 m
Catchment geology	Cambrian sedimentary
Catchment soils	76% Rankers 24% Raw Peat Soils
Catchment landcover	3% Water (inland) 13% Dense dwarf shrub heath 23% Open dwarf shrub heath <1% Broad-leaved / mixed woodland 31% Coniferous woodland 2% Neutral grass 27% Acid grassland
Mean annual rainfall	2197 mm

6.2 Deposition

Table 6.2. Deposition, Llyn Cwm Mynach

Mean total sulphate deposition	0.75 kiloequivalents ha^{-1} yr^{-1}
Mean non-marine sulphate deposition	0.47 kiloequivalents ha^{-1} yr^{-1}
Mean oxidised nitrogen deposition	0.63 kiloequivalents ha^{-1} yr^{-1}
Mean reduced nitrogen deposition	0.61 kiloequivalents ha^{-1} yr^{-1}

6.3 Hydrochemistry



µeq L^{-1} except * (µmol L^{-1})	pH	*ANC	Ca^{2+}	Mg^{2+}	Na^+	K^+	Si	*Soluble Al	*labile Al	Cl^-	SO_4^{2-}	Excess SO_4^{2-}	NO_3^-	DOC
mean 1st 5 yrs	5.38	9.18	79.13	68.86	294.50	3.41	56.77	3.94	2.42	342.60	88.16	52.17	9.59	208.80
07-08 mean	5.19	37.90	60.25	62.51	260.67	5.03	46.46	3.09	1.59	318.77	66.19	32.79	10.37	341.35
07-08 std.dev	0.57		11.10	13.22	55.94	1.26	16.10	0.53	0.41	93.93	3.32	8.61	9.12	

Figure 6.4. Spot sampled chemistry data, Llyn Cwm Mynach

Over the first 5 years of monitoring (1988-1993) the mean xSO_4 concentration of 52.4 µeq l^{-1} was relatively high for an AWMN site and is likely to have been enhanced by the interception of sulphate aerosol by the forest canopy. Chloride concentrations were also relatively high and climbed above 500 µeq l^{-1} as a result of very high inputs of seasalt in 1990. Mean NO_3 concentrations were low (9.6 µeq l^{-1}). The lake was moderately acidic with a mean pH of 5.38, mean alkalinity and ANC just above 0 µeq l^{-1} and a mean labile aluminium concentration of 65.3 µg l^{-1} – within the range where aluminium toxicity effects on salmonids are likely (Rosseland et al., 1990).

Over the monitoring period, Llyn Cwm Mynach has undergone a relatively small decline in xSO_4 concentration, with most of change occurring in the late 1990s. Mean concentration for 2003-2006 was 40.6 µeq l^{-1} – approximately 25% below the average for the initial period. Nitrate and chloride concentrations have shown substantial inter-annual variability but no long-term trend.

There has been little indication of any change in pH or ion balance ANC. The mean labile aluminium concentration in 2003-2006 was 28% lower than for the first five years but the trend in this variable was not significant according to the Seasonal Kendall Test.

Unusually for the AWMN there is little indication of any long-term trend in DOC. Llyn Cwm Mynach is an exceptionally transparent lake – secchi disc depths are often greater than the maximum depth of the lake. However, the moderate DOC concentrations suggest that the molecular composition of DOC in Llyn Cwm Mynach differs from most other sites on the Network where DOC is predominantly in the form of coloured humic substances. The dominant DOC forms here may not be as responsive to changing deposition chemistry as most AWMN sites.

6.4 Thermistors

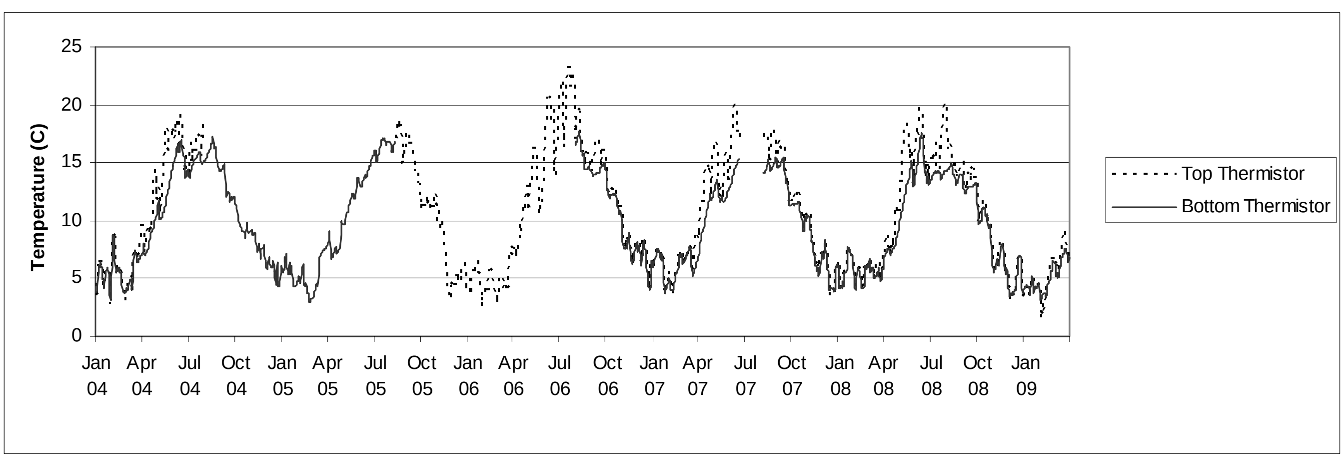


Figure 6.5. Thermistor data, Llyn Cwm Mynach

6.5 Diatoms

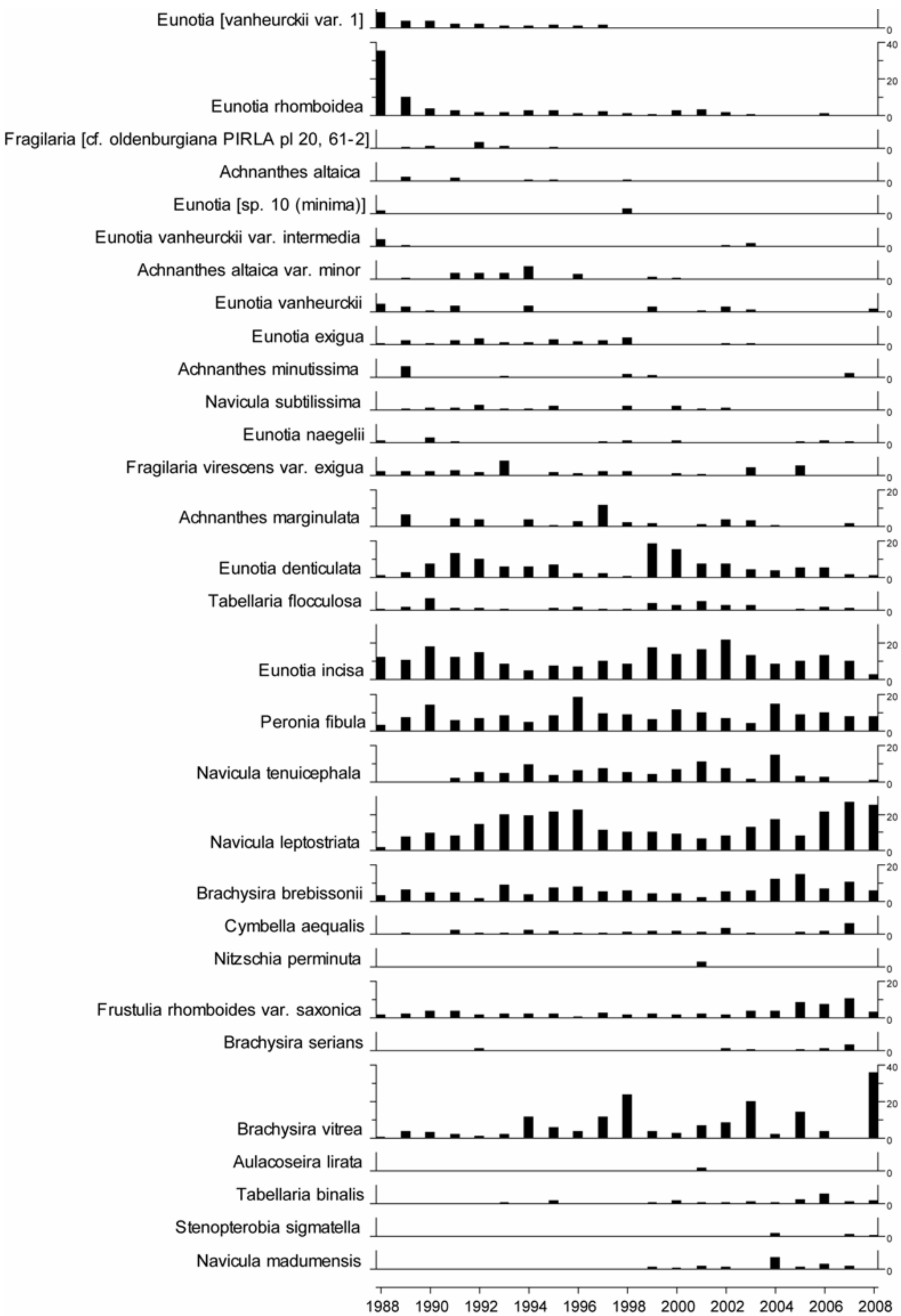


Figure 6.6. Epilithic diatom percentage abundance summary, Llyn Cwm Mynach

The epilithic diatom flora of this lake has shown considerable and sustained change over the monitoring period. Most noticeably there has been a gradual replacement of *Eunotia rhomboidea* (SWAP pH optimum = 5.1) and *Eunotia vanheurkii* var. 1 (SWAP pH optimum = 5.1) by a range of generally slightly less acid indicating species. Initially these inlcuded *Brachysira brebissonii* (SWAP pH optimum = 5.3), *Eunotia denticulata* (SWAP pH optimum = 5.2) and *Navicula tenuicephala* (SWAP pH optimum = 5.3). After 1993 a further increase in less acid taxa was observed, including *Brachysira vitrea* (SWAP pH optimum = 5.9) and *N. leptostriata*. These two taxa dominate the assemblage in 2008. The sediment trap data are however very different from the epilithic data and suggests that habitats other than the epilithon may be more important at this site. The trends in trap species composition are also partly contradictory to those in the epilithon (with respect to acidity indicators), with reductions in the deep water benthic species *Fragilaria virescens* var. *exigua* (SWAP pH optimum = 5.7) and a small increase in the acidobiontic species *Tabellaria binalis* (SWAP pH optimum = 4.7). The former showed a sharp increase in abundance in a sediment core from the lake in levels dated to the latter half of the 20th century and it remained abundant in the epilithon until around 1995. After the mid 1990s other species including *Nitzschia perminuta* (SWAP pH optimum = 5.7), *Brachysira vitrea* (SWAP pH optimum = 5.9) and *Cymbella minuta* (SWAP pH optimum = 6.1) became more abundant with *B vitrea* dominating the assemblages after 2005. The sediment trap data also show very similar species changes and indicate the importance of the epilithic community in this lake.

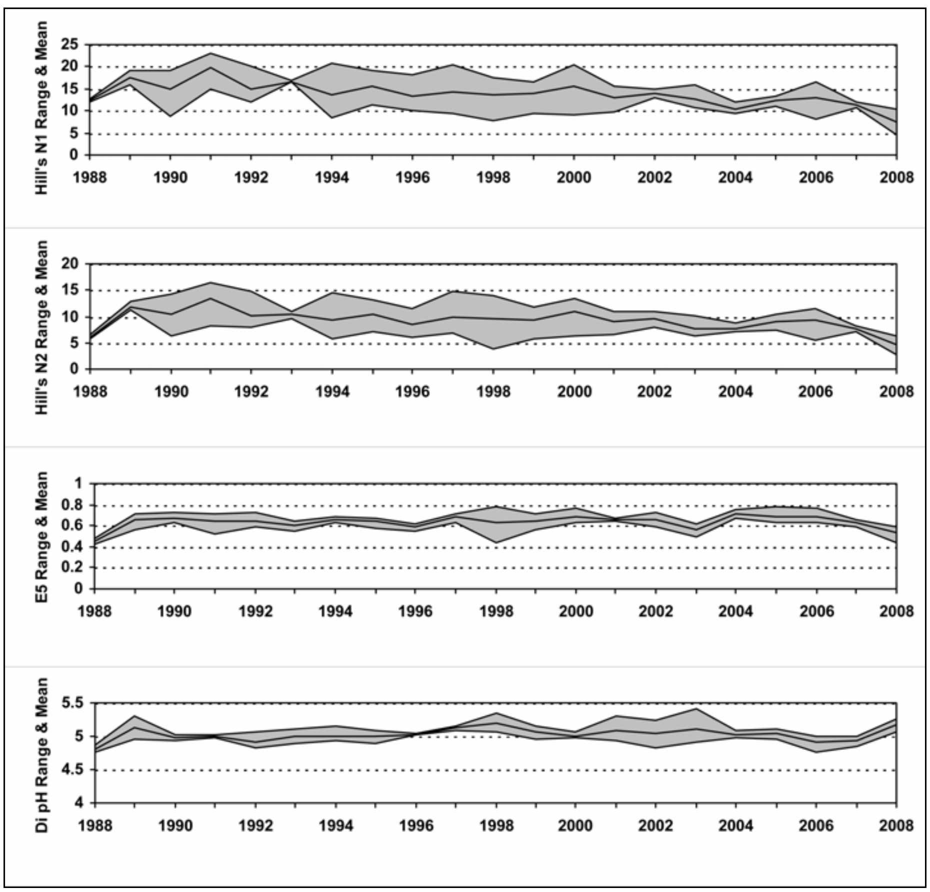


Figure 6.7. Epilithic diatom summary statistics, Llyn Cwm Mynach

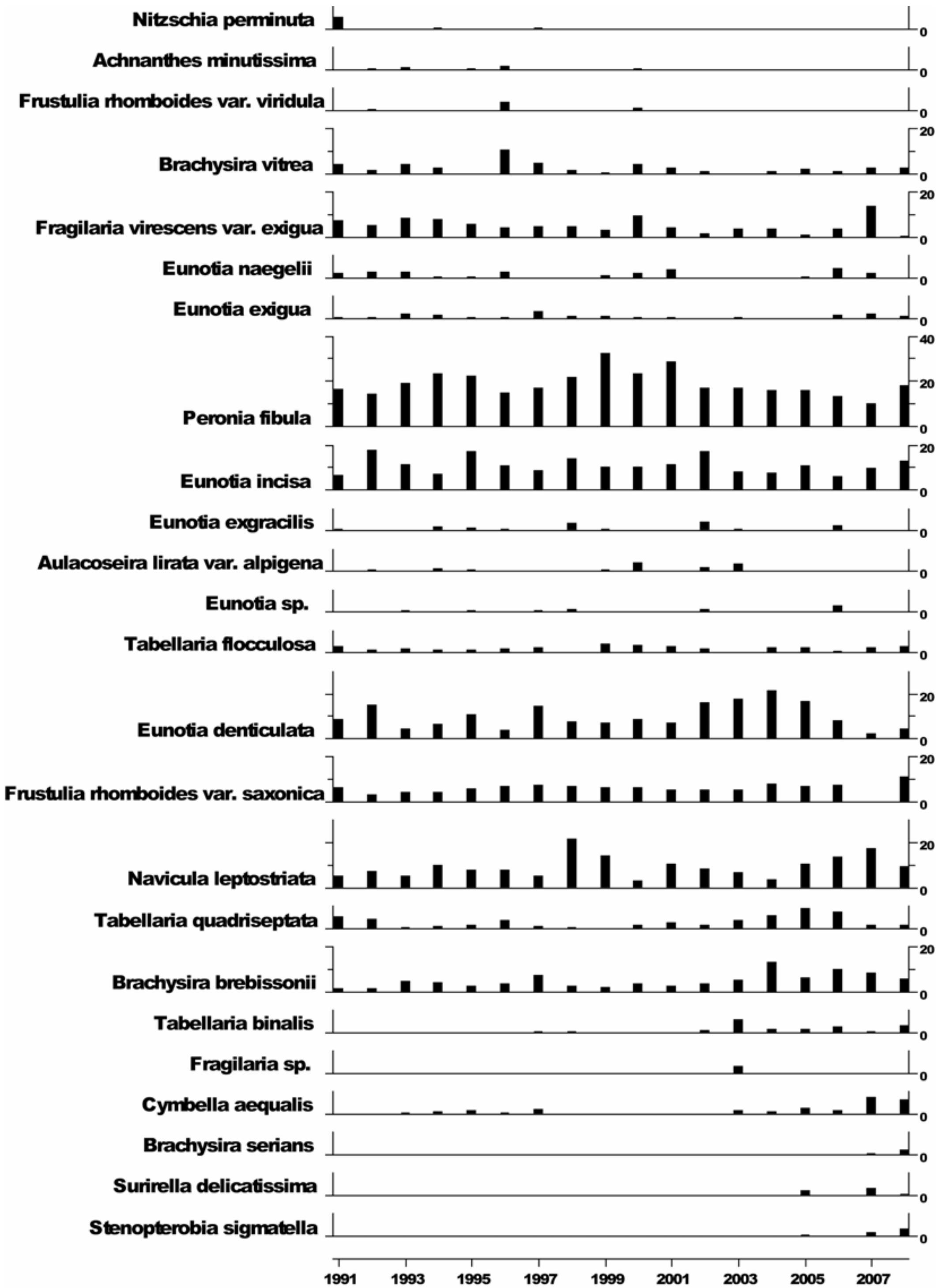


Figure 6.8. Relative percentage frequency of sediment trap diatom taxa, Llyn Cwm Mynach

6.6 Macroinvertebrates

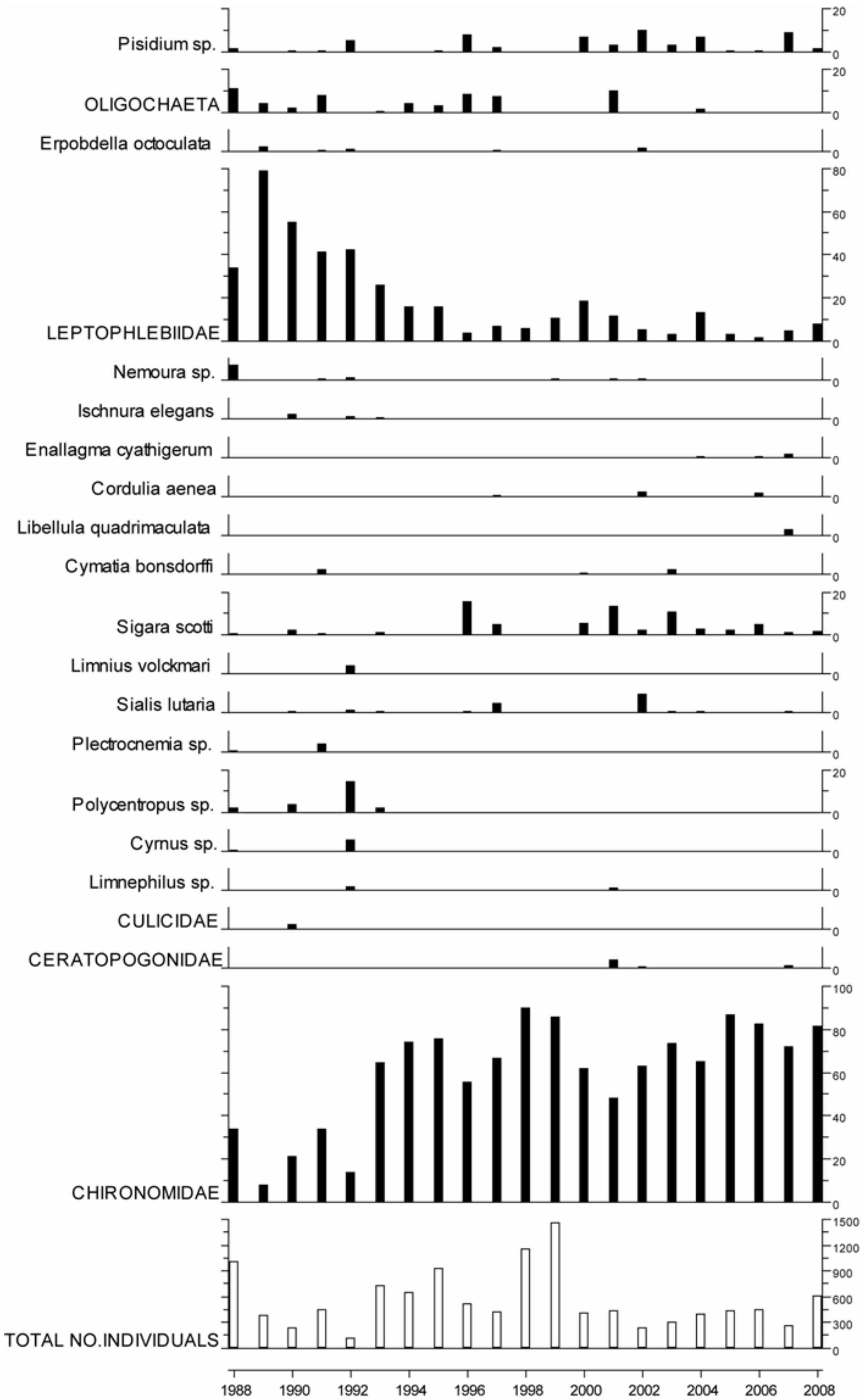


Figure 6.9. Macroinvertebrate percentage abundance summary, Llyn Cwm Mynach

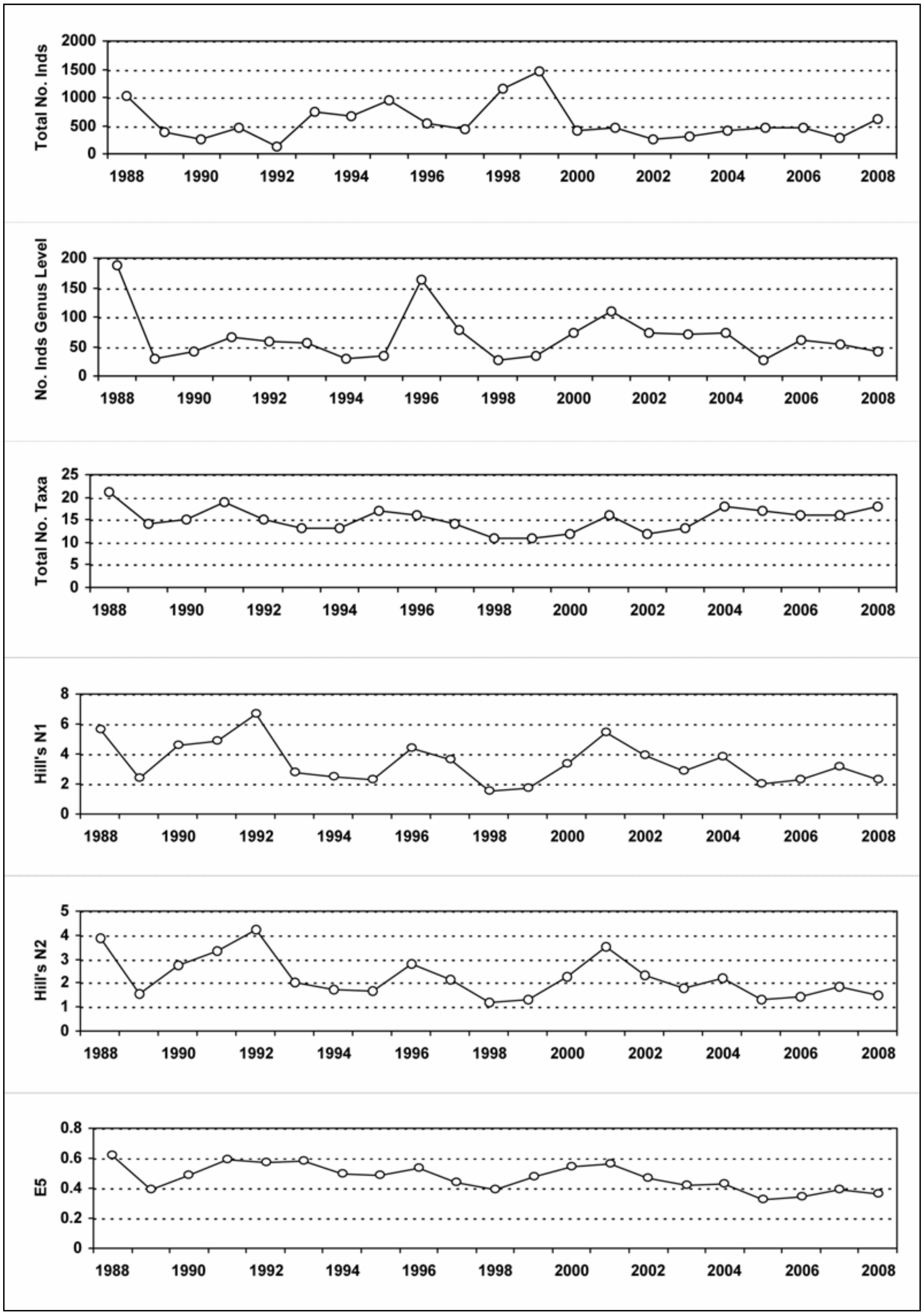
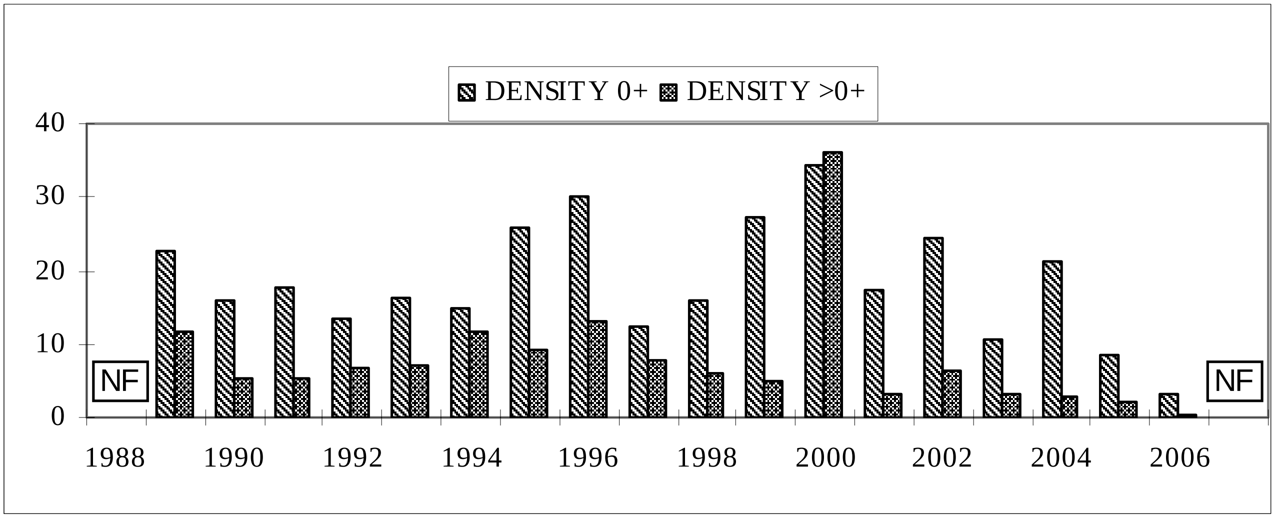


Figure 6.10. Macroinvertebrate summary statistics, Llyn Cwm Mynach

6.7 Fish



NF = Not fished

Figure 6.11. Summary of mean Trout density for outflow stream (numbers 100m^{-2}), Llyn Cwm Mynach

6.8 Aquatic Macrophytes

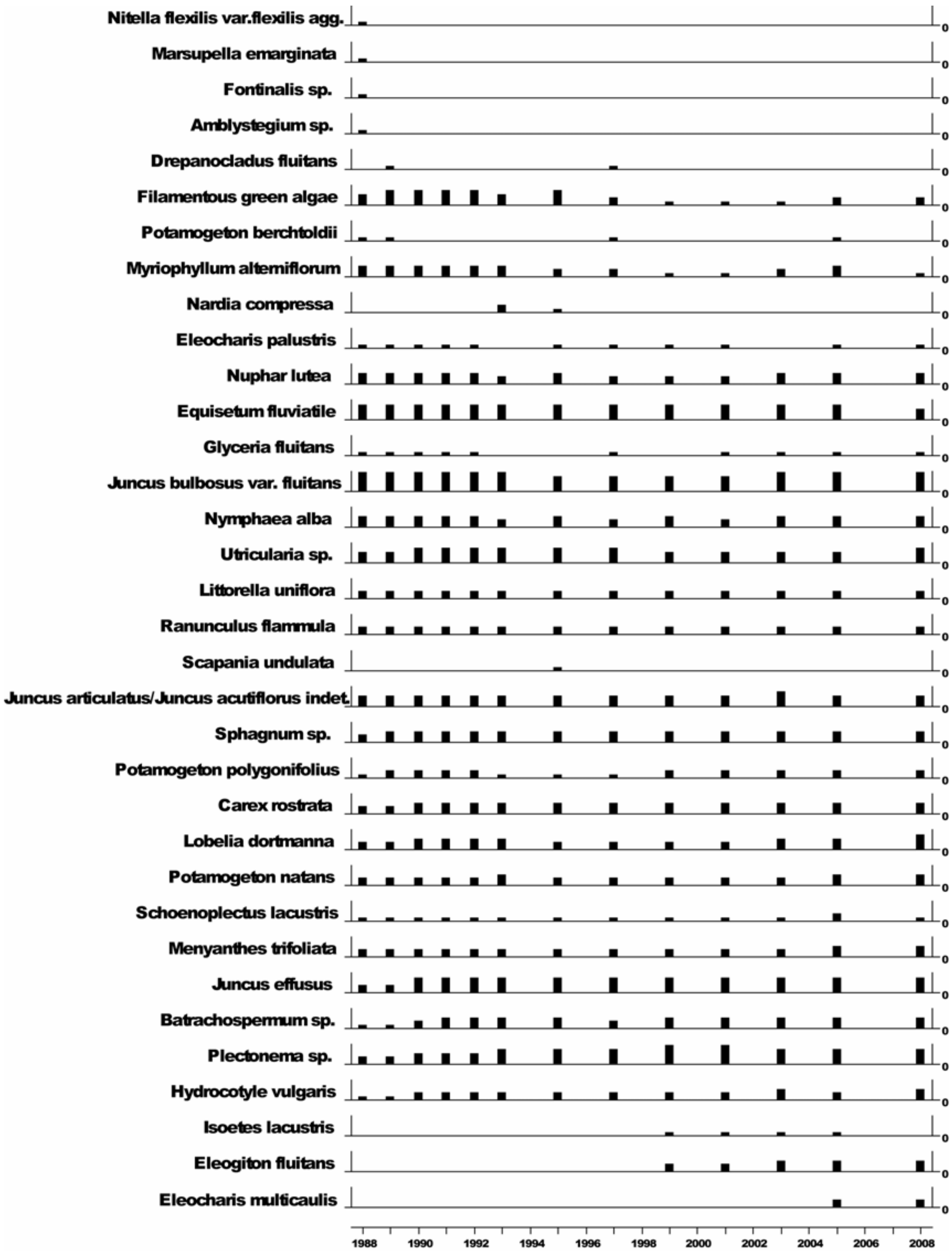


Figure 6.12. Aquatic macrophyte species scores (1-5), Llyn Cwm Mynach

The aquatic macrophyte flora in Llyn Cwm Mynach is relatively diverse for an AWMN lake, partly as a result of the range of habitats at this site. There are two distinct basins with a shallow limb separated from the deeper main basin by a stone ciaeway. The main basin is dominated by a range of oligotrophic species with varying sensitivity to acidity. The clearest change in the main basin has been a major expansion over the first few years of a blue green alga, *Plectonema* sp. This genus has a preference for slightly elevated levels of metals (John *et al*., 2002) and may indicate the influence of some small abandoned Manganese mine workings within the catchment on the water chemistry. *Plectonema* has formed a thick blanket over other submerged vegetation to the extent that *Juncus bulbosus* var. *fluitans* plants in parts of the lake were unable to survive and now form substantial rafts of decaying debris around the shore of the lake. Despite these apparently deleterious changes for the ecology of the lake, one acid-sensitive species, *Eleogiton fluitans*, was first detected in 1999 as has been found on each survey since. Small amounts of *Isoetes lacustris* were first recorded in 1999 and it has been found again in all but the most recent survey.

6.9 Summary

6.9.1 Chemistry

6.9.2 Biology

6.10 Llyn Cwm Mynach Recent Publications Using AWMN Data

Battarbee, R. W., Kernan, M., Monteith, D. T. & Curtis, C. J. (2010) Summary and Recommendations. In: *UK Acid Waters Monitoring Network 20 Year Interpretative Report*, 279-293, ENSIS Ltd, Environmental Change Research Centre, University College London, London.

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7 Afon Hafren

Figure 7.1. Afon Hafren.

7.1 Site and Catchment Characteristics

The Afon Hafren (Figure x, Table x) lies in the Cambrian Mountains of mid-Wales and from its confluence with the Afon Hore forms the headwaters of the River Severn. The catchment area is 358 ha and rises from 355 m at the sampling station to 690 m at Blaenhafren (Figure x). Stagnoodsols cover approximately 40% of the catchment and organic peaty soils comprise the remaining area. The underlying geology consists of Ordovician grits and Silurian mudstones and shales.

Fifty percent of the catchment is planted with conifers, primarily Sitka and Norway spruce, and forms part of the larger Hafren Forest; planting took place primarily between 1948-1950 and 1963-1964. At the start of monitoring about 5% of the catchment consisted of recently felled forest. Since that time there has been steady harvesting and replanting of large areas continuing right up to the present day though this appears to have had little impact on stream water chemistry (Neal *et al.* 2004). Moorland grasses and *Calluna* occupy the remainder of the catchment (all of which is a SSSI) and are utilised for rough grazing. Prior to afforestation the catchment was exploited as upland sheepwalk.

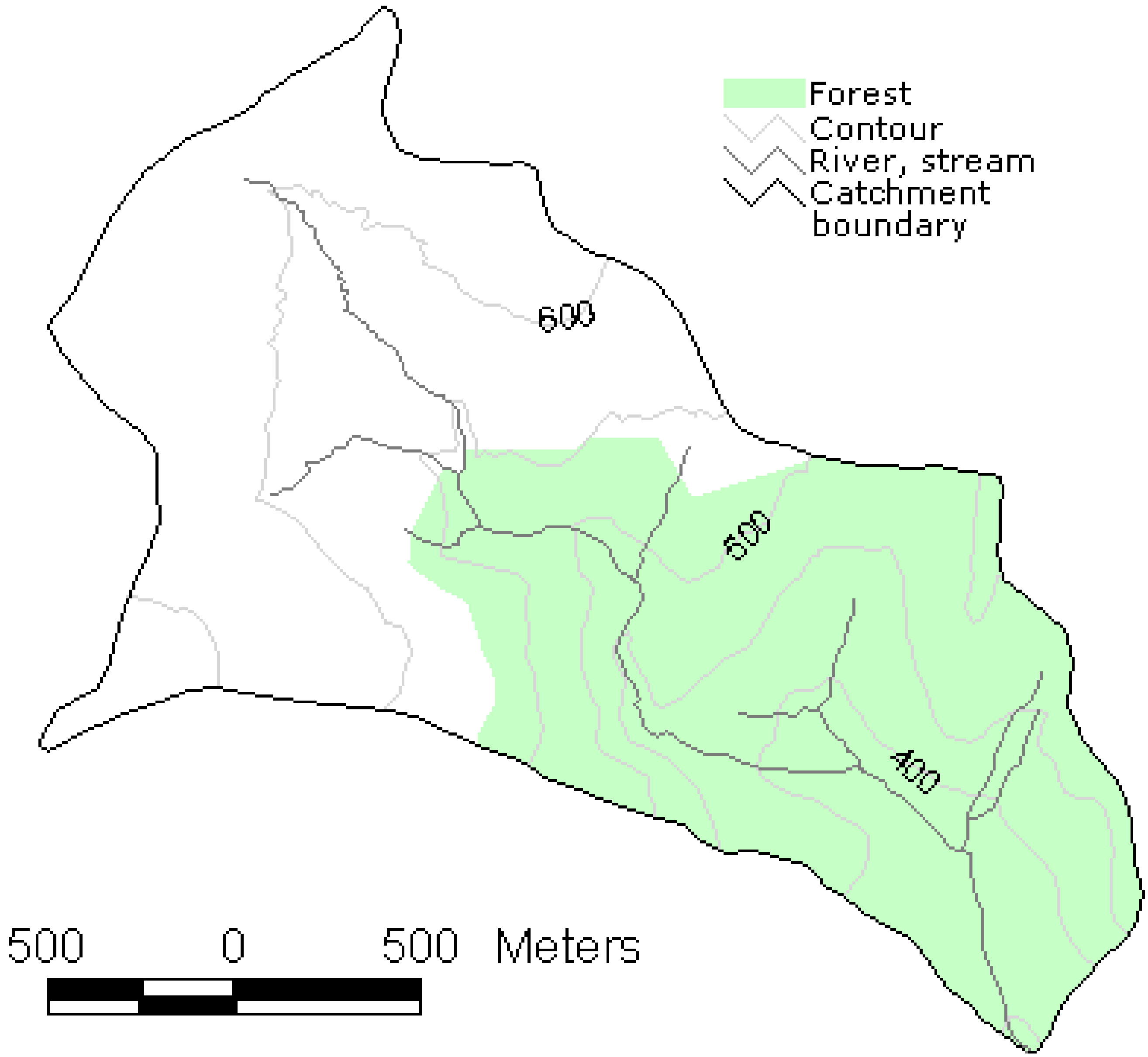


Figure 7.2. Afon Hafren catchment

The altitude range is 335 m from the sampling station to the headwaters. The channel section utilised for biological sampling lies in a forest clearing and is circa 3.5 m wide comprising shallow rapids and a deep pool. The stream bed substrate consists largely of boulders, cobbles and pebbles.

Table 7.1 Site characteristics, Afon Hafren

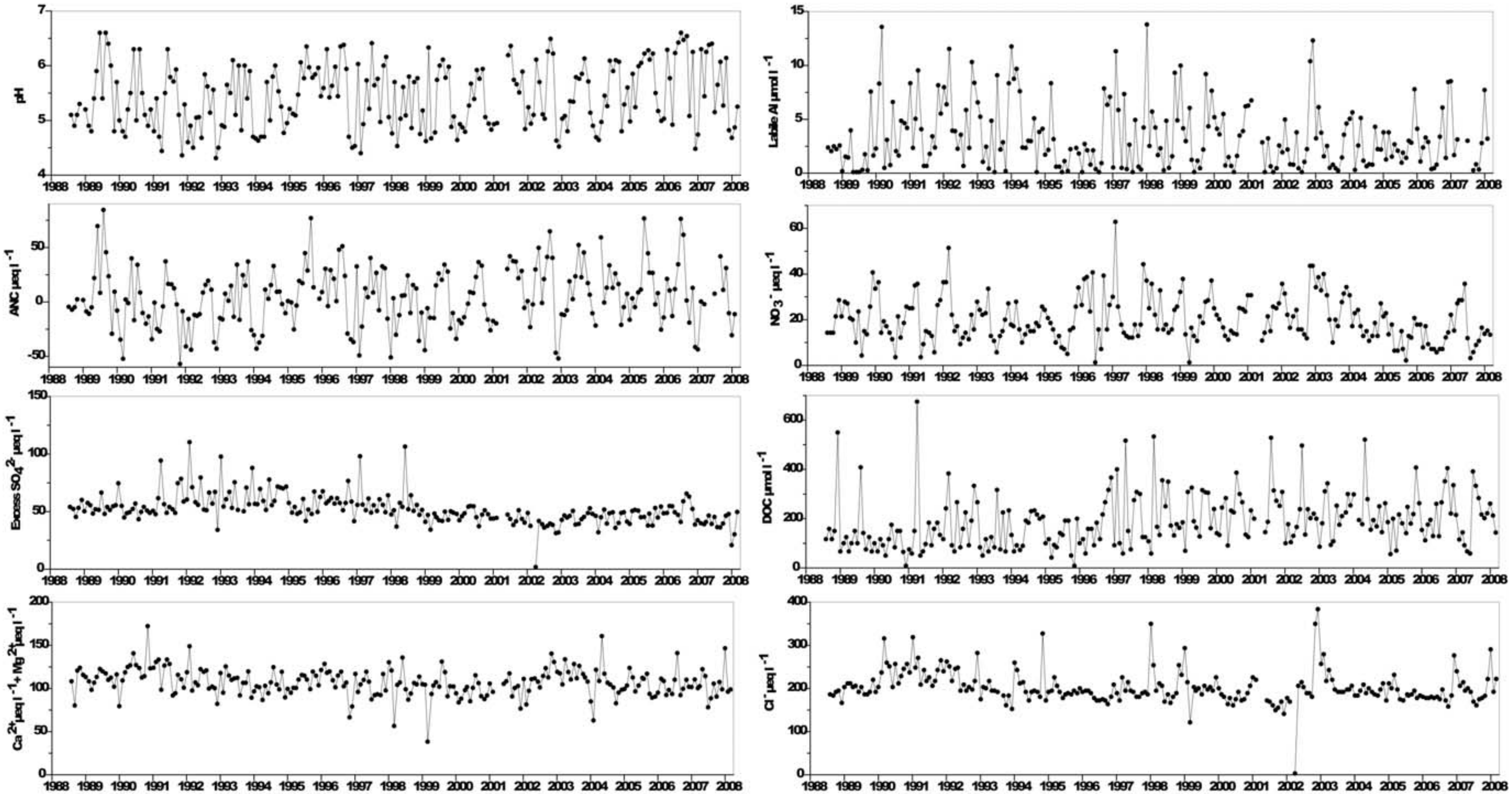
Grid Reference	SN 844 876
Catchment area	373 ha
Minimum catchment altitude	355 m
Maximum catchment altitude	690 m
Catchment geology	Ordovician and Silurian sedimentary
Catchment soils	40% Stagnodposols 60% Raw Peat Soils
Catchment landcover	<1% Water (inland) 5% Dense dwarf shrub heath 17% Open dwarf shrub heath 42% Coniferous woodland 32% Acid grassland <1% Suburban / rural development <1% Continuous urban 3% Inland bare ground
Mean annual rainfall	2468 mm

7.2 Deposition

Table 7.2. Deposition, Afon Hafren

Mean total sulphate deposition	1.01 kiloequivalents ha^{-1} yr^{-1}
Mean non-marine sulphate deposition	0.56 kiloequivalents ha^{-1} yr^{-1}
Mean oxidised nitrogen deposition	0.83 kiloequivalents ha^{-1} yr^{-1}
Mean reduced nitrogen deposition	0.84 kiloequivalents ha^{-1} yr^{-1}

7.3 Hydrochemistry



$\mu eq\ L^{-1}$ except * ($\mu mol\ L^{-1}$)	pH	*ANC	Ca^{2+}	Mg^{2+}	Na^{+}	K^{+}	Si	*Soluble Al	*labile Al	Cl^{-}	SO_4^{2-}	Excess SO_4^{2-}	NO_3^{-}	DOC
mean 1st 5 yrs	5.27	-2.57	48.08	67.32	201.00	3.20	113.80	6.38	3.83	221.80	82.70	59.41	20.45	149.30
07-08 mean	5.58	5.53	40.66	63.45	184.48	5.31	100.30	4.35	2.58	198.22	61.76	40.98	16.01	206.62
07-08 std.dev	0.65	25.35	7.10	11.99	26.27	4.16	32.25	3.53	2.60	34.62	7.76	8.53	9.91	103.61

Figure 7.3. Spot sampled chemistry data, Afon Hafren

7.4 Diatoms

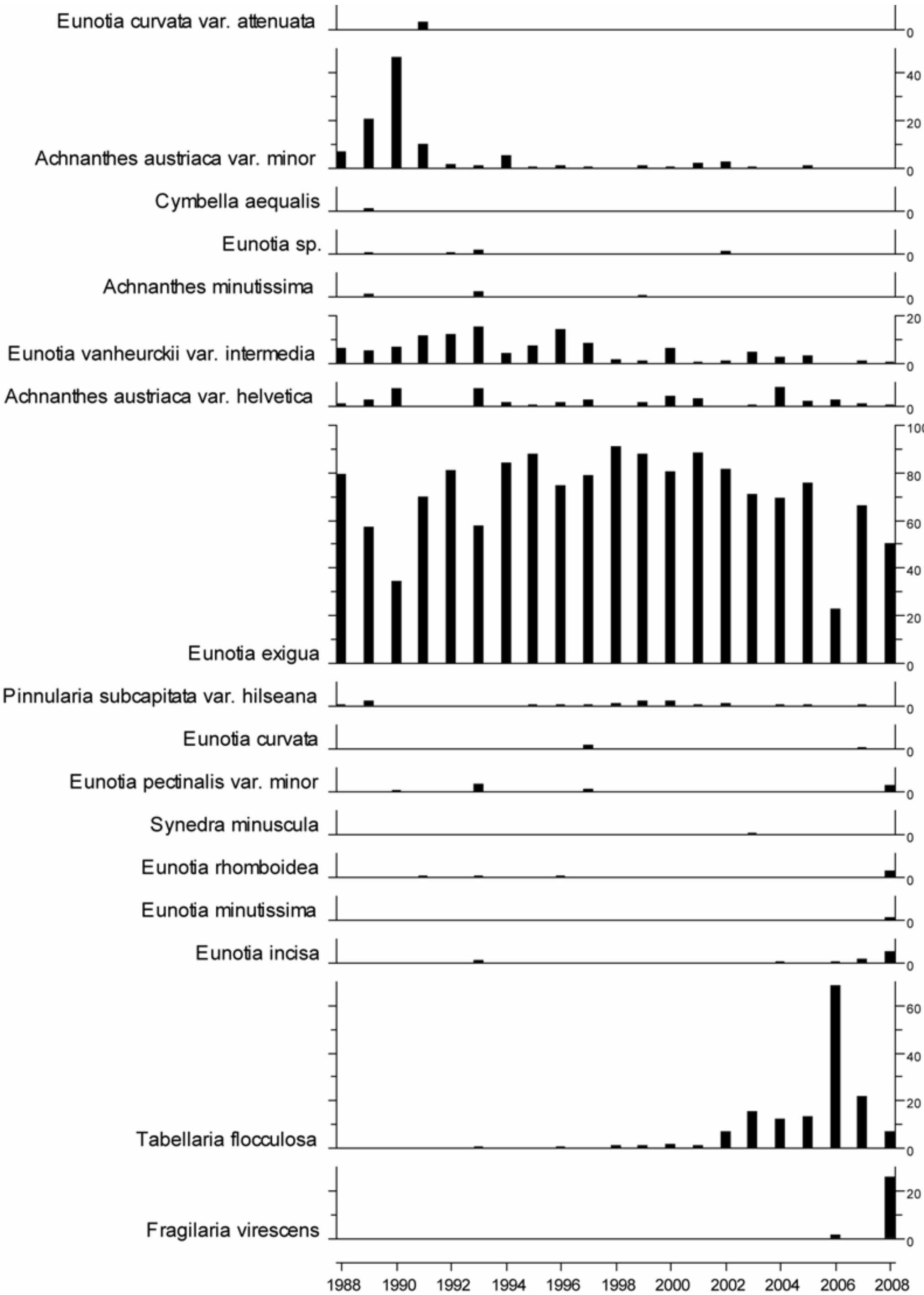


Figure 7.4. Epilithic diatom percentage abundance summary, Afon Hafren

The epilithic diatom community in this stream has been dominated by one acidophilous/acidobiontic species (Shilland *et al.*, 2009), *Eunotia exigua* (SWAP pH optimum = 5.1). Only in years 1990 and 2005 did frequencies of this diatom fall below 40% and in most samples, in most years, *E. exigua* formed >70% of the assemblage. In 1990 its abundance was suppressed by *Achnanthes austriaca* var. *minor* (SWAP pH optimum = 5.1) and in 2005 by *Tabellaria flocculosa* (SWAP pH optimum = 5.4). Since 1997, abundances of this latter species have progressively increased from trace amounts to becoming the dominant species in 2006. In 2008 another marked change occurred with the arrival of *Fragilaria virescens* at the site and by 2008 its abundance was second only to *E. exigua.* In general, these epilithic diatom changes are consistent with a gradual recent improvement in pH. However, the strikingly high abundances of *T. flocculosa* in 2006 may also reflect the influence of other factors affecting pH, especially the low sea-salt inputs during the winter and spring of that year. The unusual abundance of *Fragilaria virescens* in 2008 could indicate the onset of other significant water quality changes but further monitoring would be needed to verify this.

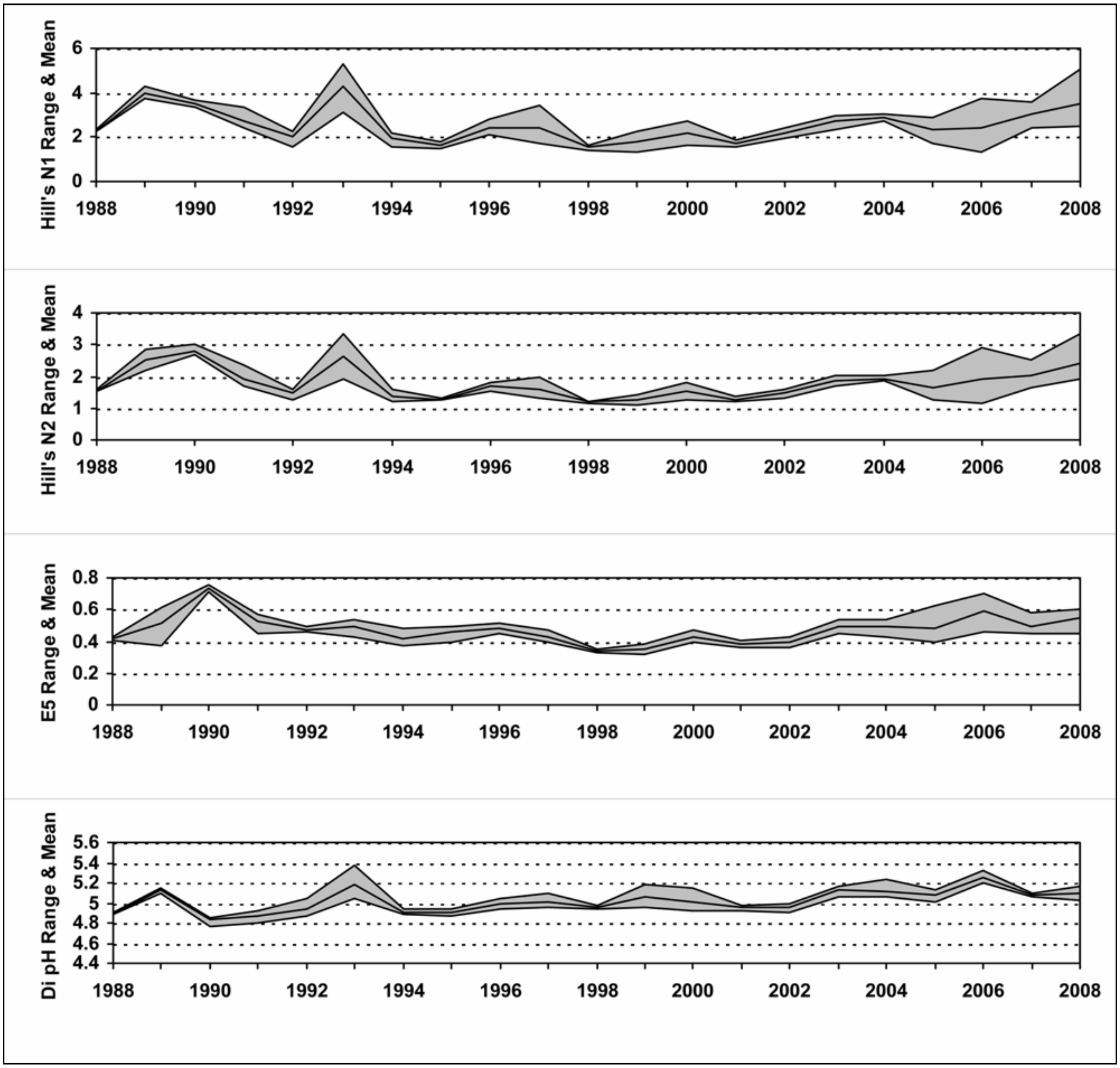
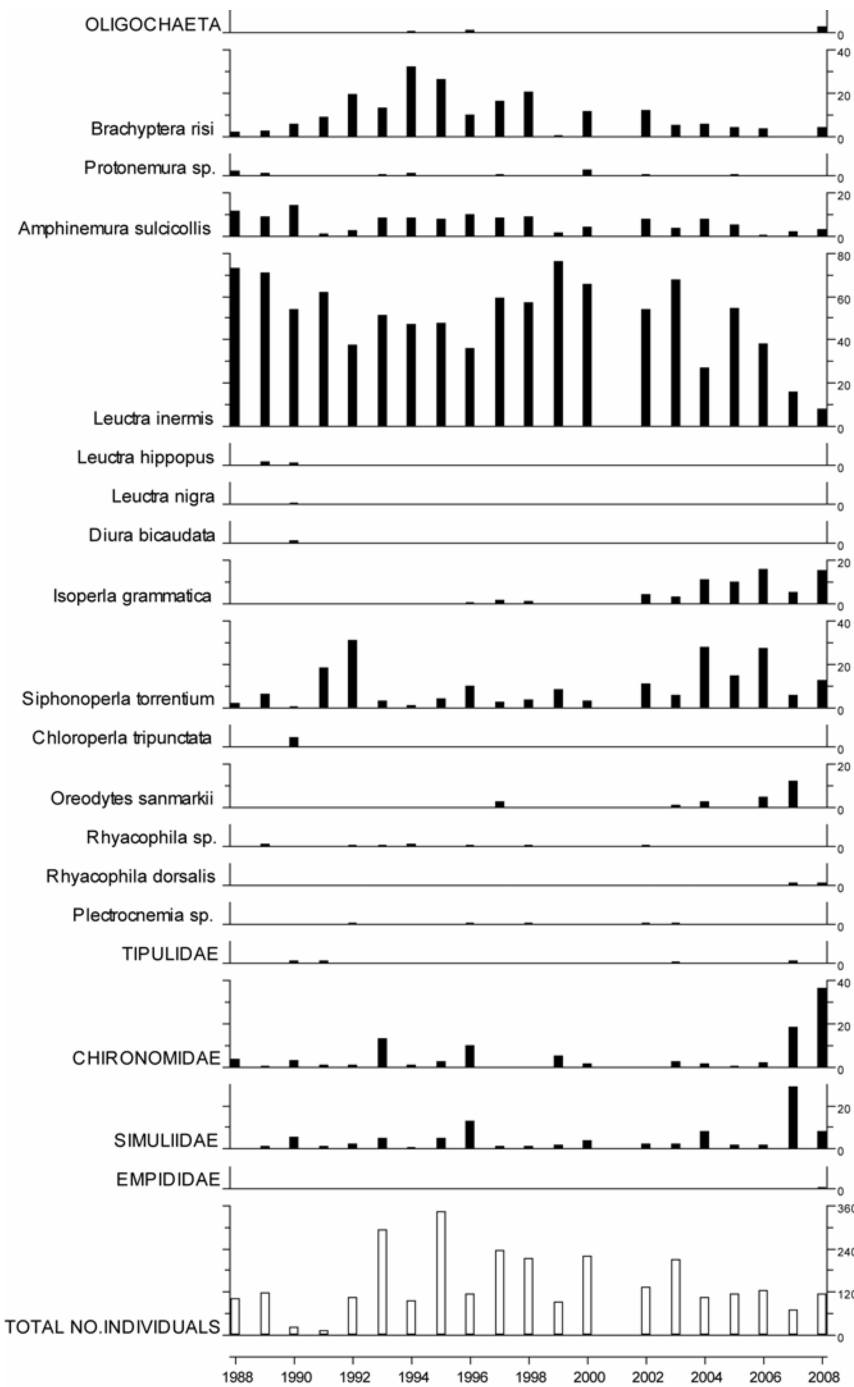


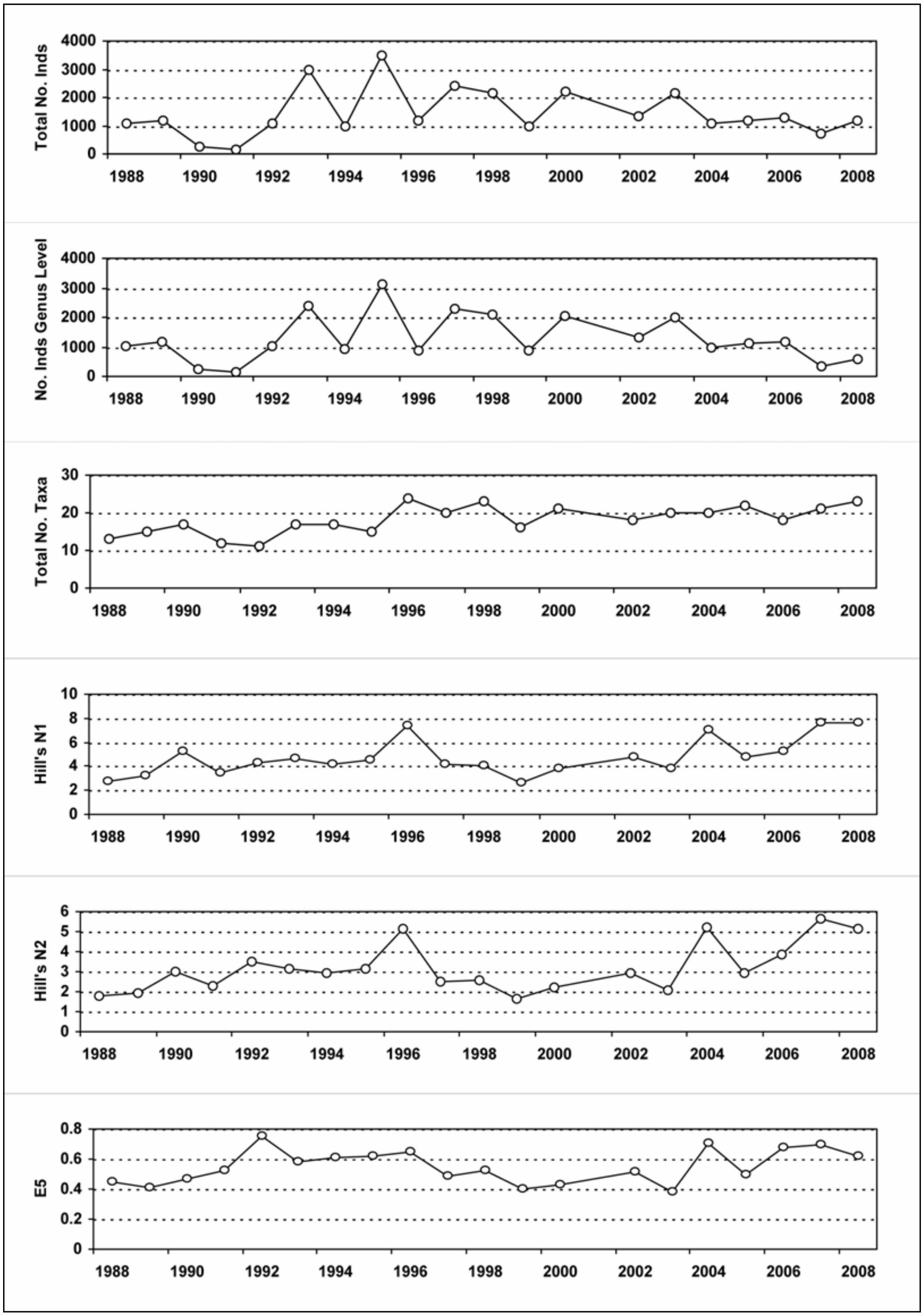
Figure 7.5. Epilithic diatom summary statistics, Afon Hafren

7.5 Macroinvertebrates



No sampling in 2001 due to Foot and Mouth restrictions.

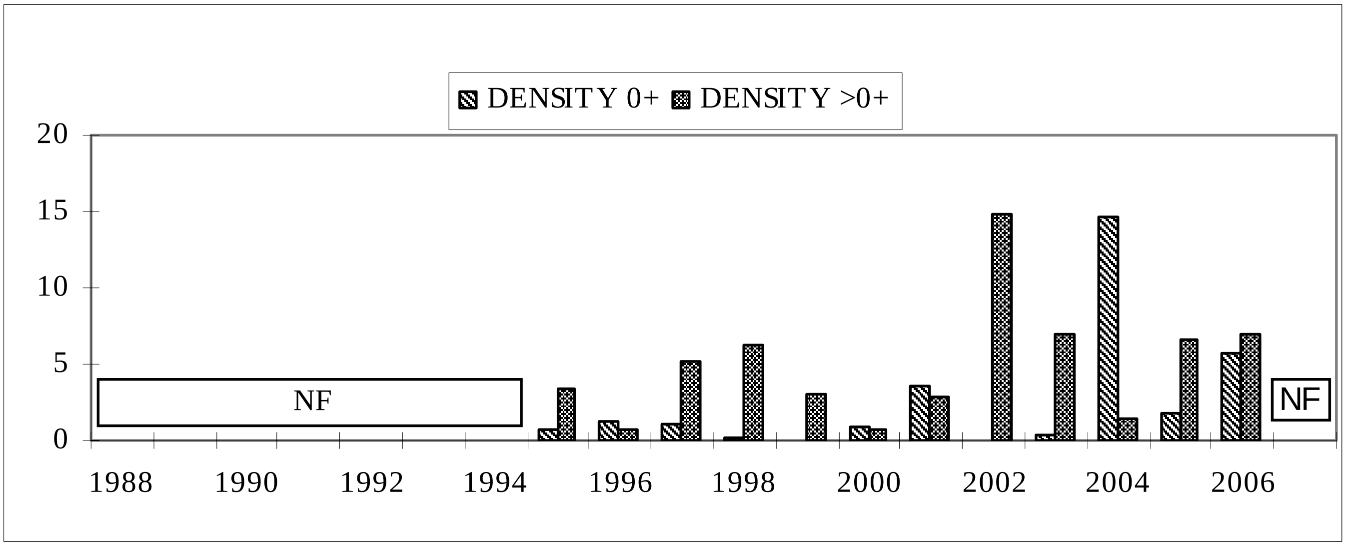
Figure 7.6. Macroinvertebrate percentage abundance summary, Afon Hafren



No sampling in 2001 due to Foot and Mouth restrictions.

Figure 7.7. Macroinvertebrate summary statistics, Afon Hafren

7.6 Fish



NF = Not fished

Figure 7.8. Summary of mean Trout density (numbers $100m^{-2}$), Afon Hafren

Electrofishing of the Afon Hafren, a stream with a partially afforested catchment, only commenced in 1995. Fry recruitment is sporadic and densities variable. There is some indication that trout parr densities are increasing. However there is considerable inter-annual and inter-reach variability.

7.7 Aquatic Macrophytes

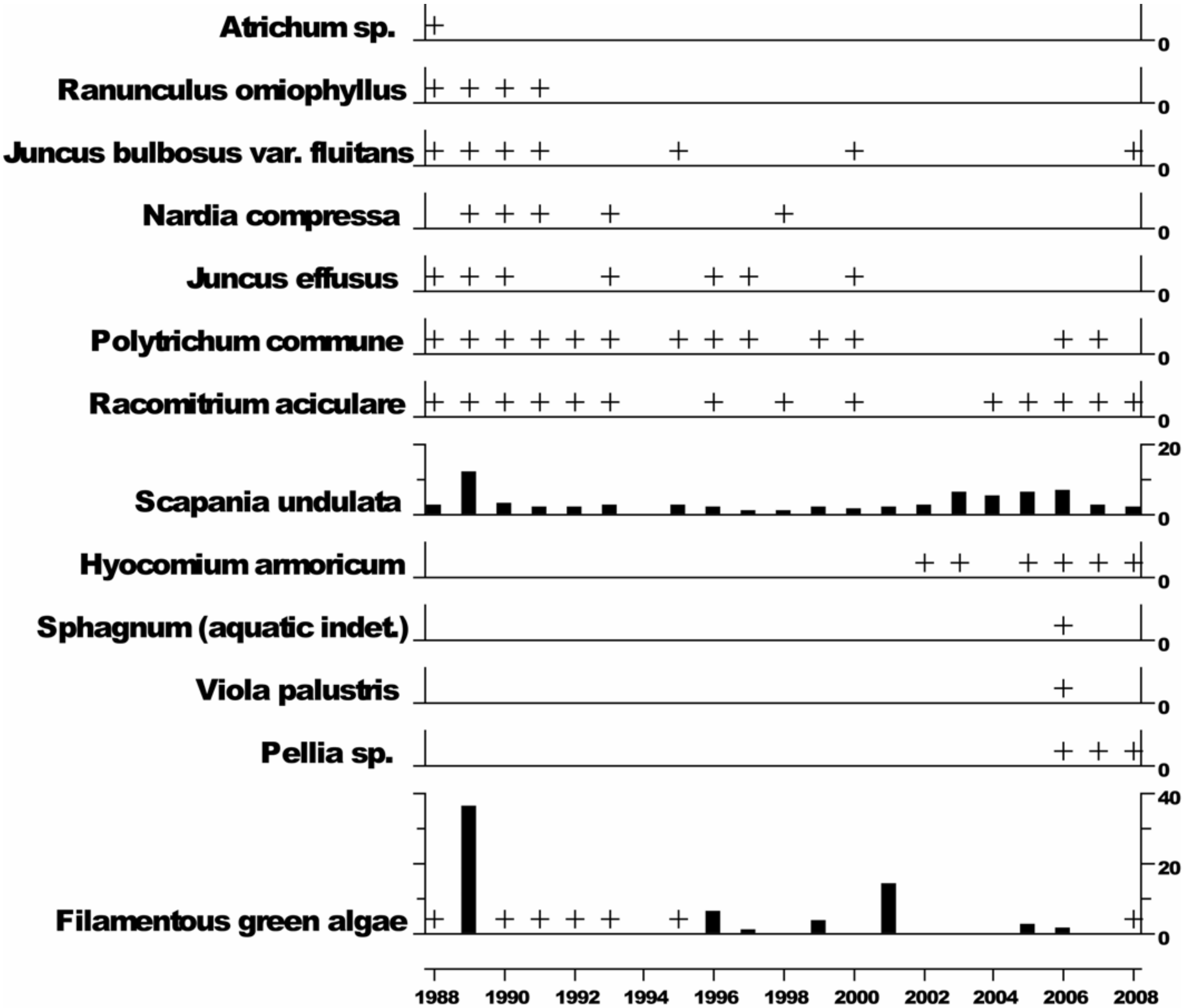


Figure 7.9. Aquatic macrophyte percentage species cover (+ represents >1% abundance), Afon Hafren

The aquatic macrophytes in the survey stretch at Afon Hafren have been restricted largely to one acid tolerant and ubiquitous liverwort species, *Scapania undulata,* which after many years of limited coverage has expanded slightly since 2003. The liverwort *Nardia compressa*, which is most common in some of the most acidic streams on the AWMN, was found in trace amounts over most of the early years of monitoring but has not been recorded since 1998. The aquatic moss *Hyocomium armoricum* was found in one location in 2002 and has been recorded there every year since.

7.8 Summary

7.8.1 Chemistry

7.8.2 Biology

7.9 Afon Hafren Recent Publications Using AWMN Data

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8 Afon Gwy

Figure 8.1. Afon Gwy.

8.1 Site and Catchment Characteris

The Afon Gwy was introduced into the Monitoring Network in 1991 as a replacement site for the Nant y Gronwen, for which access permission had been withdrawn by the landowner.

The Afon Gwy (Figure x, Table x) lies to the east of Plynlimon in central Wales and forms part of the headwater system of the River Wye. The catchment area is 389 ha and rises from 380 m at the sampling station to a maximum altitude of 730 m (Figure x). Catchment soils are dominated by peats and stagnopodsols. The underlying geology is Lower Palaeozoic mudstones, shales and grits of the Gwestyn and Van formations, overlain in places by locally-derived glacial drift. The catchment consists of moorland grasses, notably *Molinia* and *Nardus*, with *Eriophorum* and the moss *Racomitrium* on areas of wetter land and *Pteridium* on the lower slopes. The majority of this grassland supports rough grazing for sheep and in summer, on the lower land, cattle. The local area is crossed with tracks used for rally testing, one of which passes through the southern edge of the catchment. More than half of the western side of the catchment falls within a SSSI and the stream itself forms the headwater part of the River Wye/Afon Gwy SSSI.

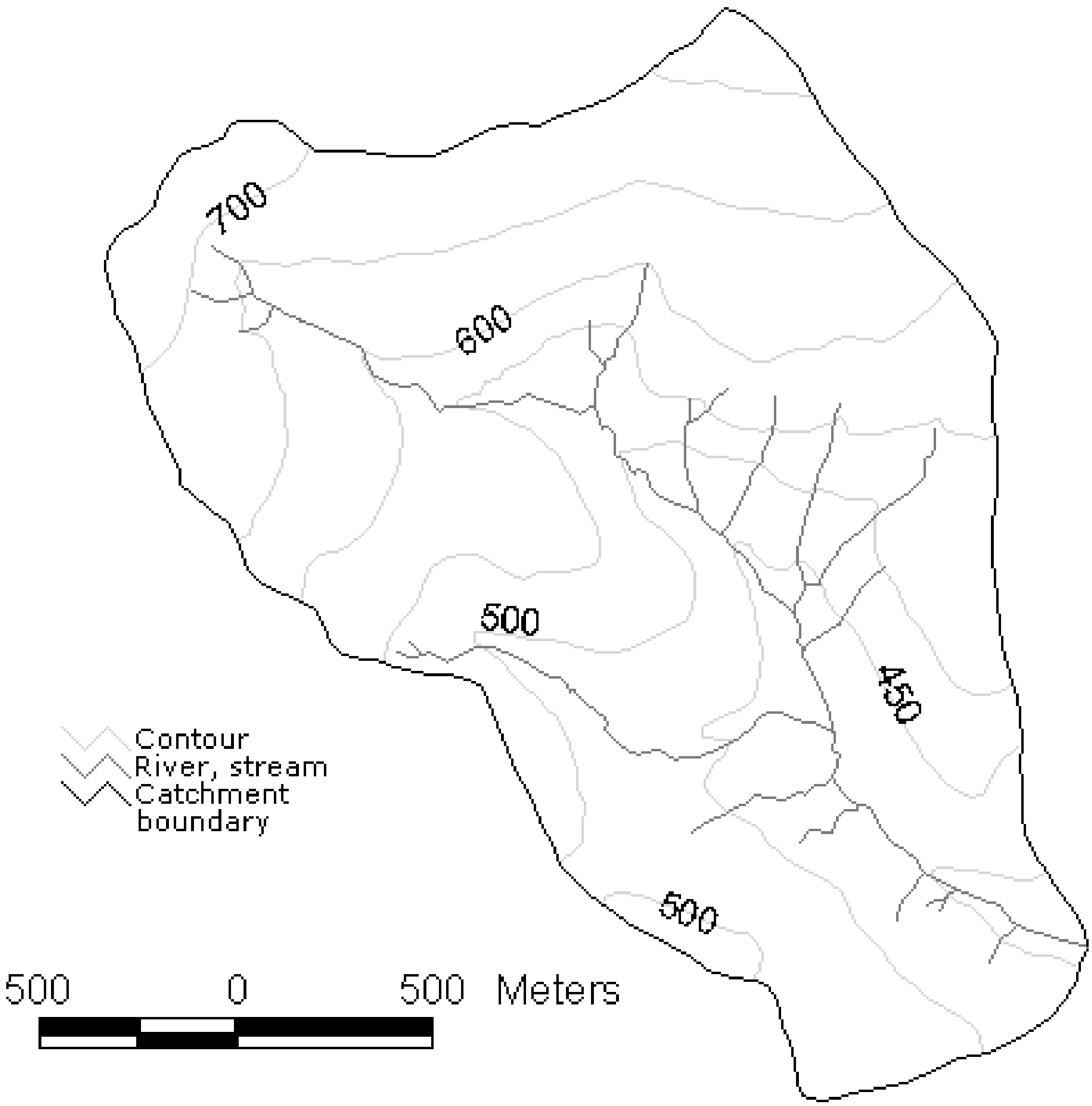


Figure 8.2. Afon Gwy catchment

The altitude range is from 440 m to 700 m in the headwaters. The channel section utilised for biological sampling is 3-4 m wide and comprises a series of bedrock ledges, falls and boulder strewn riffles.

Table 8.1 Site characteristics, Afon Gwy

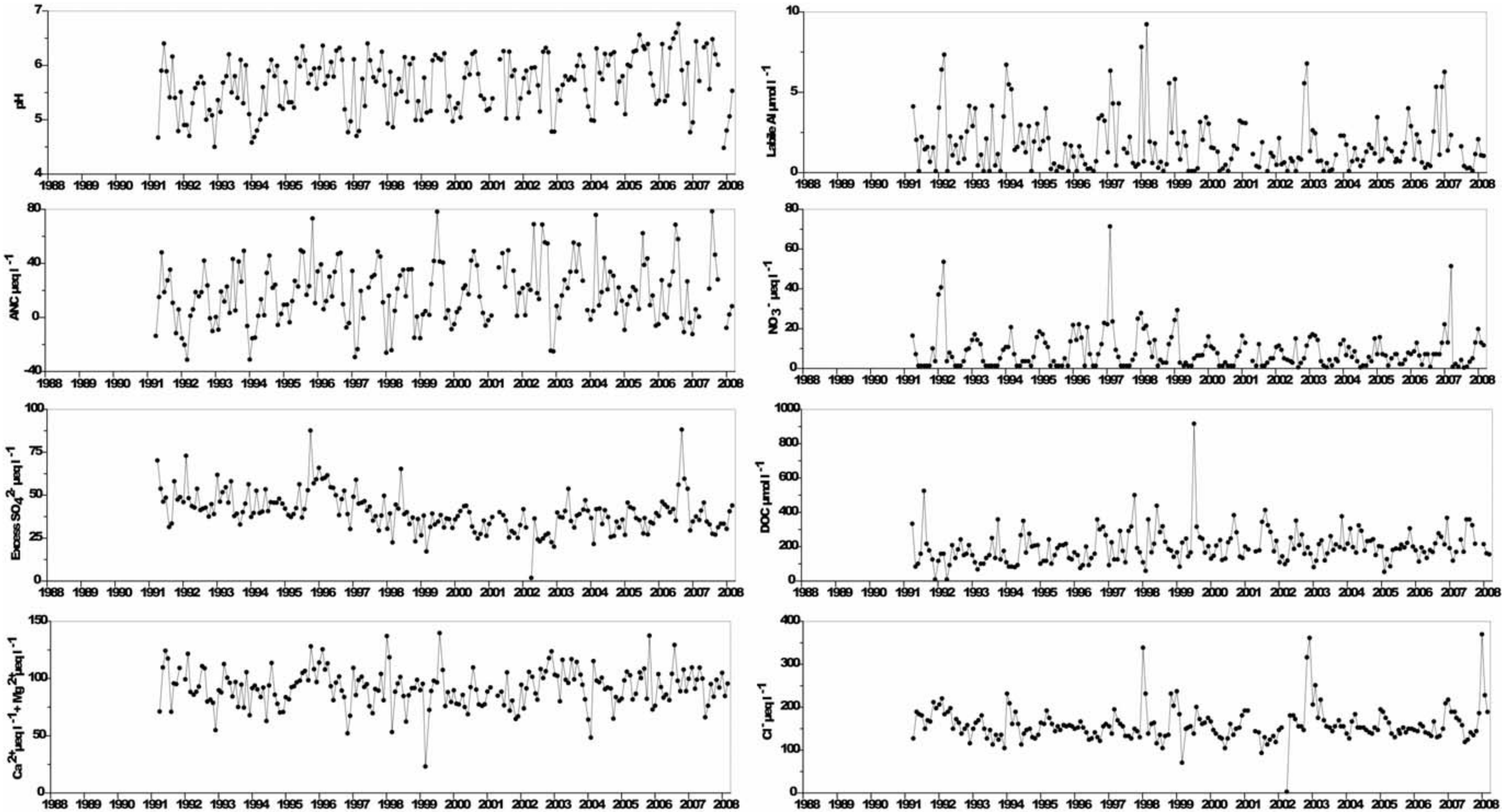
Grid Reference	SN 824 854
Catchment area	389 ha
Minimum catchment altitude	390 m
Maximum catchment altitude	730 m
Catchment geology	Lower Palaeozoic sedimentary
Catchment soils	60% Stagnopodsols 40% Raw Peat Soils
Catchment landcover	<1% Bog (deep peat) <1% Open dwarf shrub heath 2% Coniferous woodland 2% Bracken 95% Acid grassland <1% Suburban / rural development <1% Inland bare ground
Mean annual rainfall	2599 mm

8.2 Deposition

Table 8.2. Deposition, Afon Gwy

Mean total sulphate deposition	0.84 kiloequivalents ha^{-1} yr^{-1}
Mean non-marine sulphate deposition	0.47 kiloequivalents ha^{-1} yr^{-1}
Mean oxidised nitrogen deposition	0.53 kiloequivalents ha^{-1} yr^{-1}
Mean reduced nitrogen deposition	0.69 kiloequivalents ha^{-1} yr^{-1}

8.3 Hydrochemistry



µeq L^{-1} except * (µmol L^{-1})	pH	*ANC	Ca^{2+}	Mg^{2+}	Na^{+}	K^{+}	Si	*Soluble Al	*labile Al	Cl^{-}	SO_4^{2-}	Excess SO_4^{2-}	NO_3^{-}	DOC
mean 1st 5 yrs	5.51	14.00	40.57	54.08	147.50	3.24	69.88	3.93	1.98	159.70	65.90	49.13	8.87	163.50
07-08 mean	5.69	25.18	38.63	53.54	149.74	4.26	67.40	1.90	0.89	178.03	54.67	36.01	6.22	244.13
07-08 std.dev	0.71	29.48	6.96	7.96	26.66	4.47	22.72	1.15	0.68	67.87	9.98	6.40	6.45	82.59

Figure 8.3. Spot sampled chemistry data, Afon Gwy

8.4 Diatoms

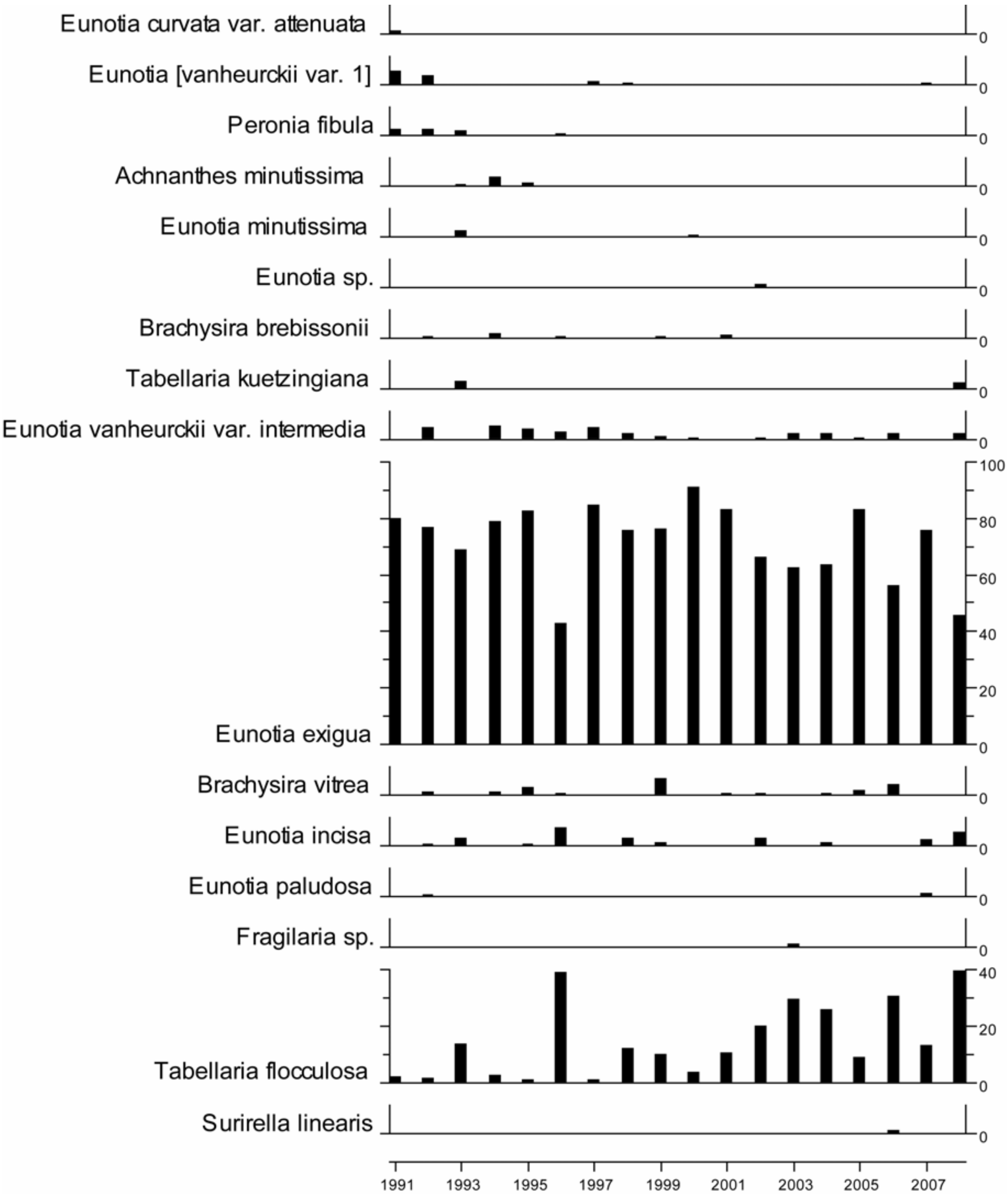


Figure 8.4. Epilithic diatom percentage abundance summary, Afon Gwy

Similar to the Afon Hafren, in the early years of monitoring the acidophilous / acidobiontic species *Eunotia exigua* (SWAP pH optimum = 5.1) dominated the epilithon diatom community of this stream. However, abundances have declined from peak levels in the late-1990s, as *Tabellaria flocculosa* (SWAP pH optimum = 5.4), again in common with the Afon Hafren, has increased from very low abundances in the early 1990s. The epilithon changes at this site are consistent with a gradual improvement to elevated water pH after about 1995. There has been a general decline in representation of less common taxa at this site, so that diversity has declined.

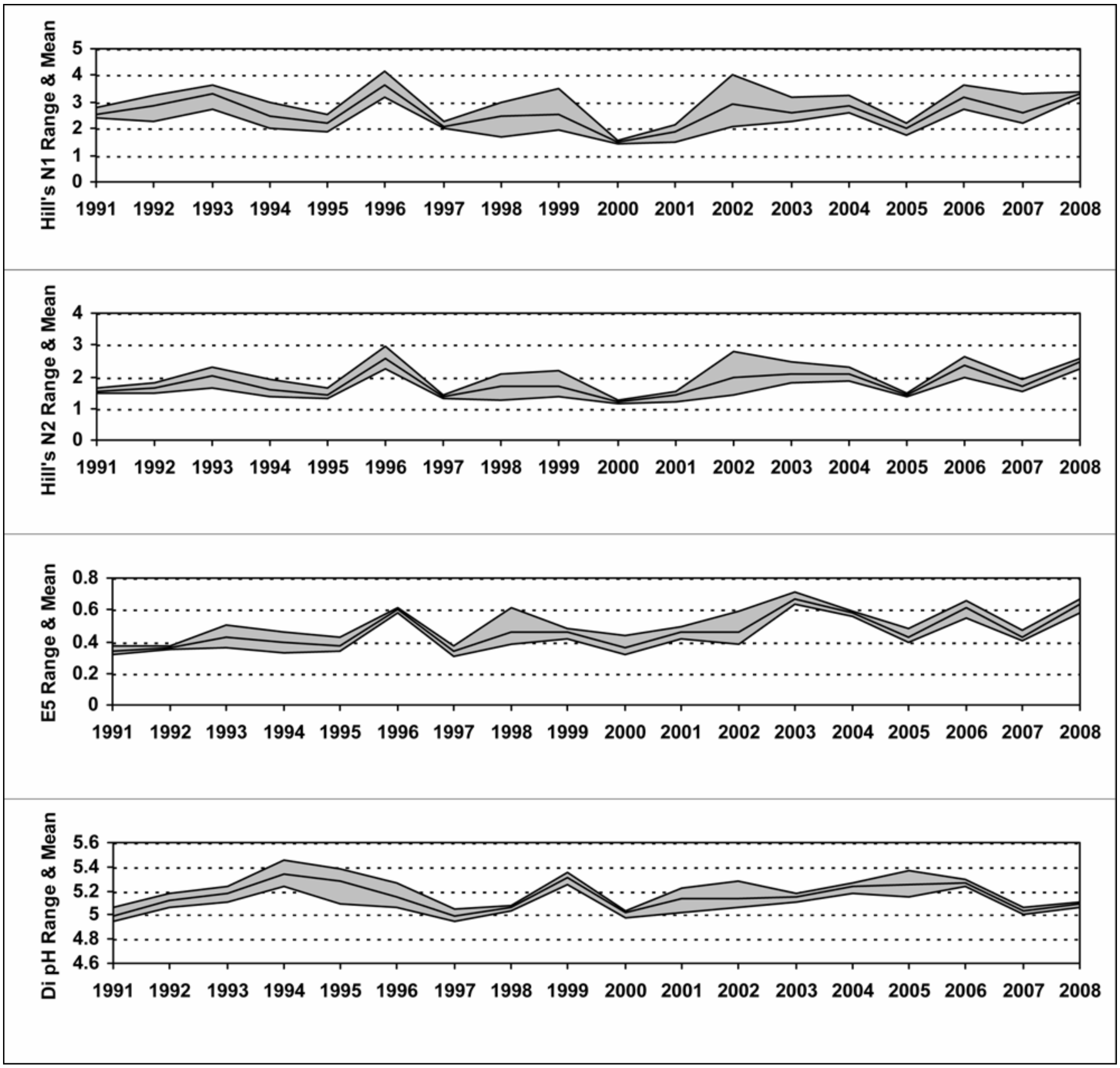
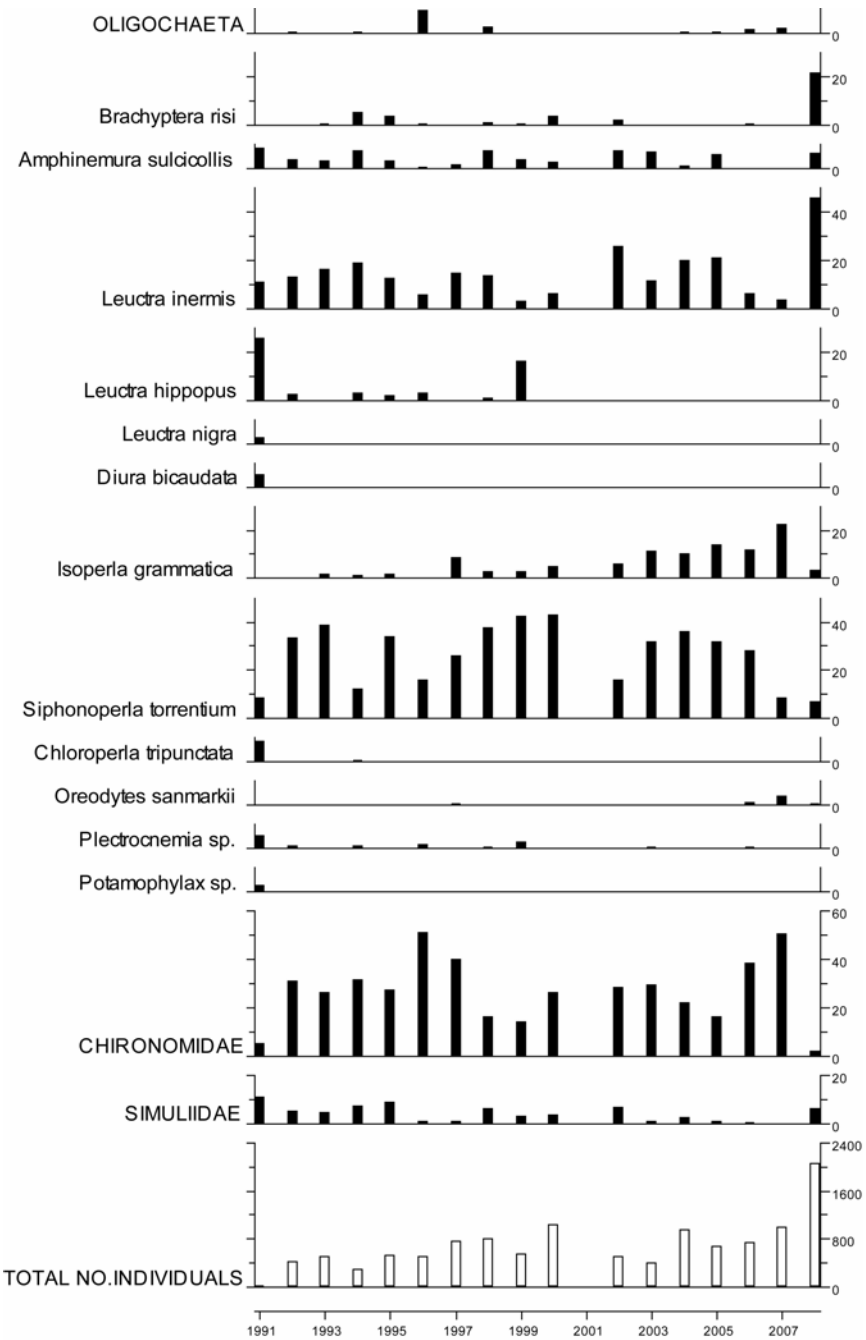


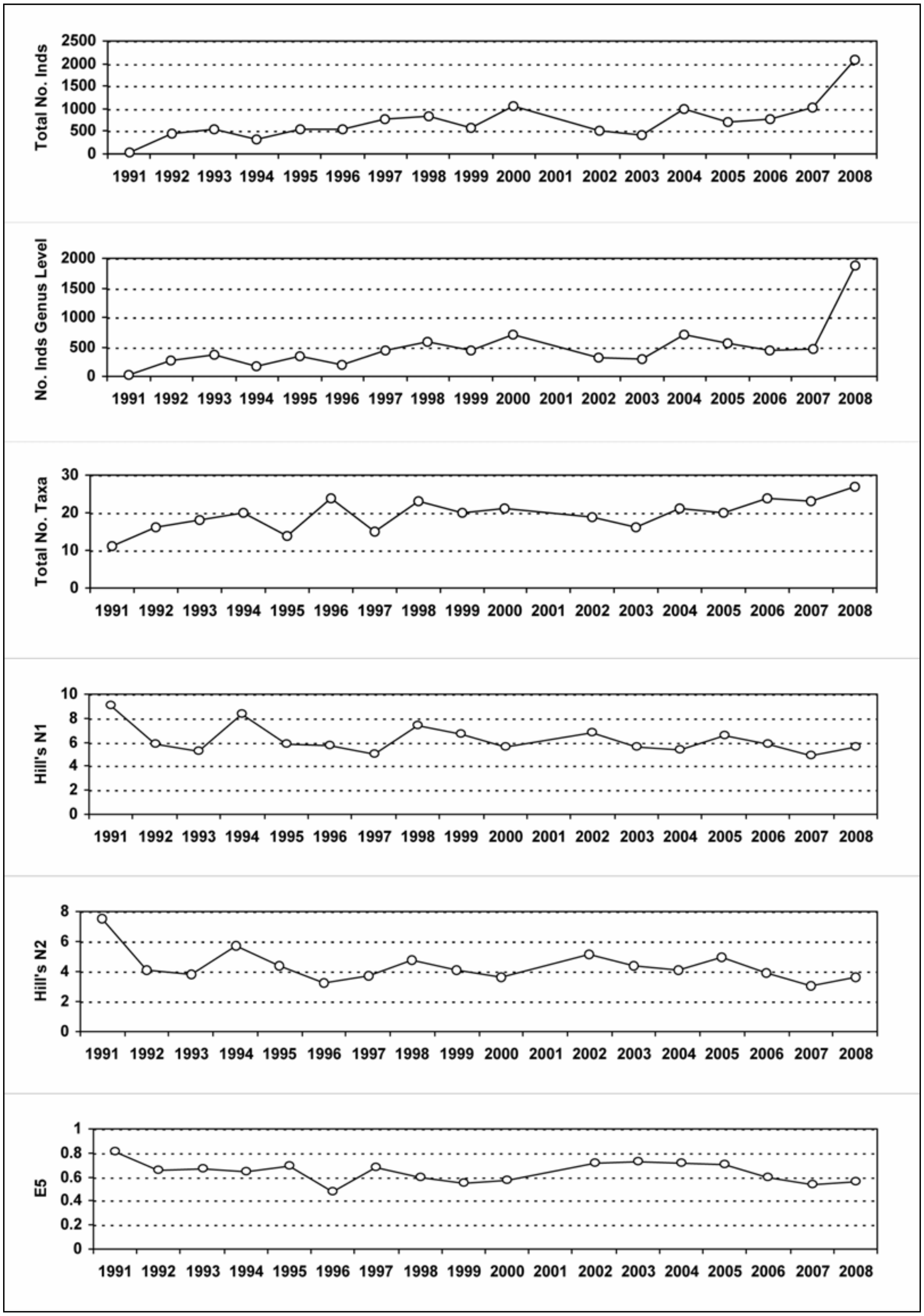
Figure 8.5. Epilithic diatom summary statistics, Afon Gwy

8.5 Macroinvertebrates



No sampling in 2001 due to Foot and Mouth restrictions.

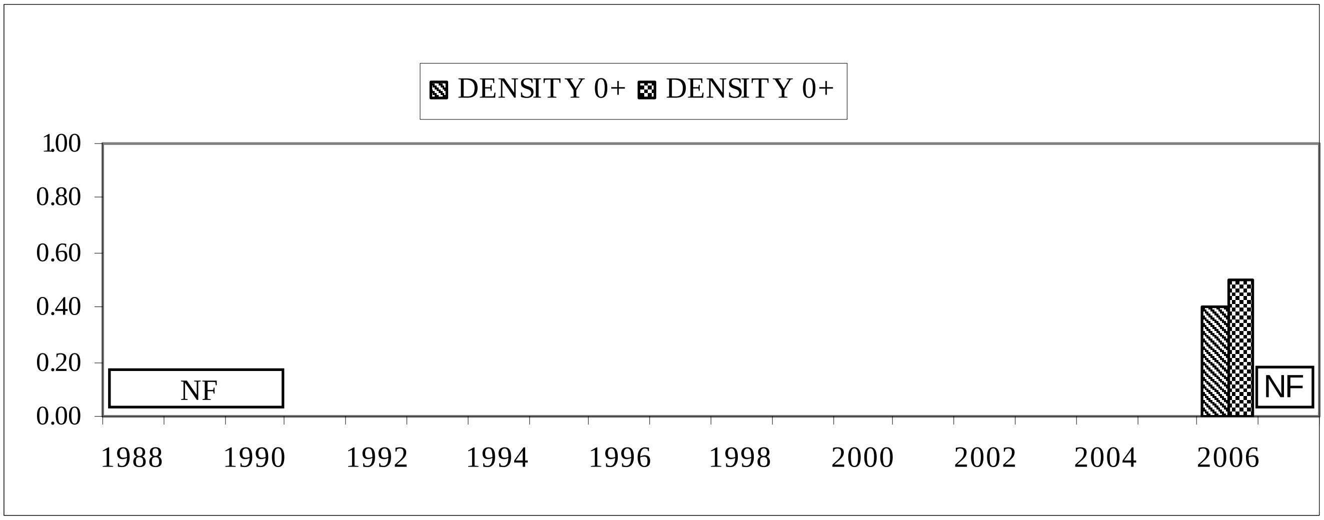
Figure 8.6. Macroinvertebrate percentage abundance summary, Afon Gwy



No sampling in 2001 due to Foot and Mouth restrictions.

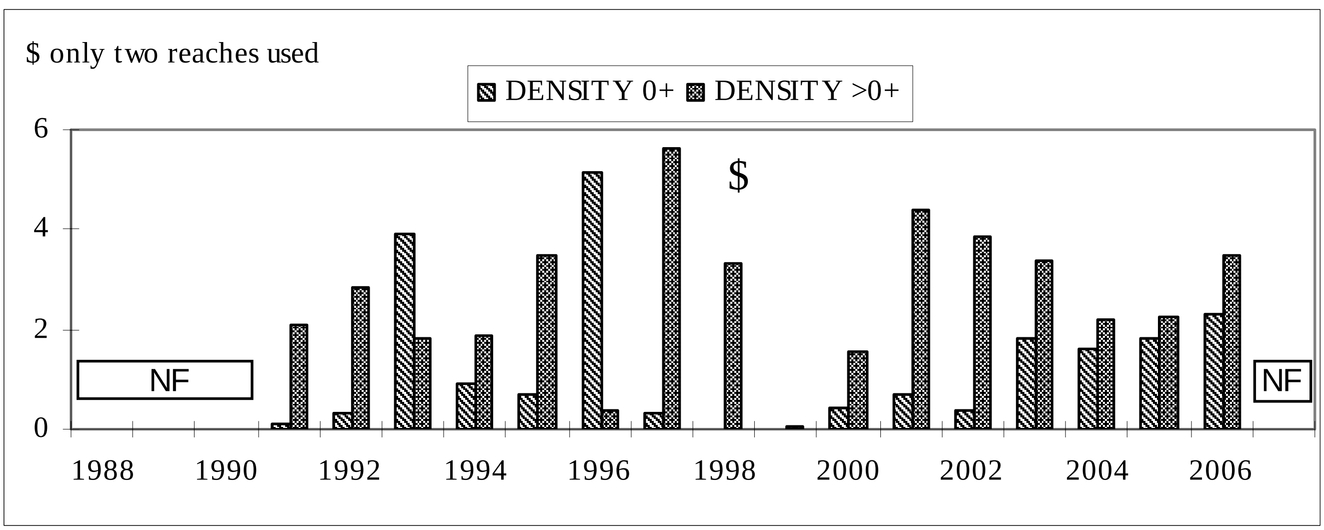
Figure 8.7. Macroinvertebrate summary statistics, Afon Gwy

8.6 Fish



NF = Not fished. Salmon recorded in 2006 for the first time.

Figure 8.8. Summary of mean Salmon density (numbers 100m^{-2}), Afon Gwy



NF = Not fished

Figure 8.9. Summary of mean Trout density (numbers 100m^{-2}), Afon Gwy

The Afon Gwy is a stream site with a moorland catchment. Brown trout recruitment is sporadic, although regular recruitment has occurred in recent years. Trout parr densities do not appear to show any long-term trend. A few juvenile Atlantic salmon (a highly acid-sensitive species) were recorded in 2006. However, it is important to determine whether the appearance of salmon could be linked to recent changes in stocking practice immediately downstream before any conclusions can be drawn with respect to possible recovery.

8.7 Aquatic Macrophytes

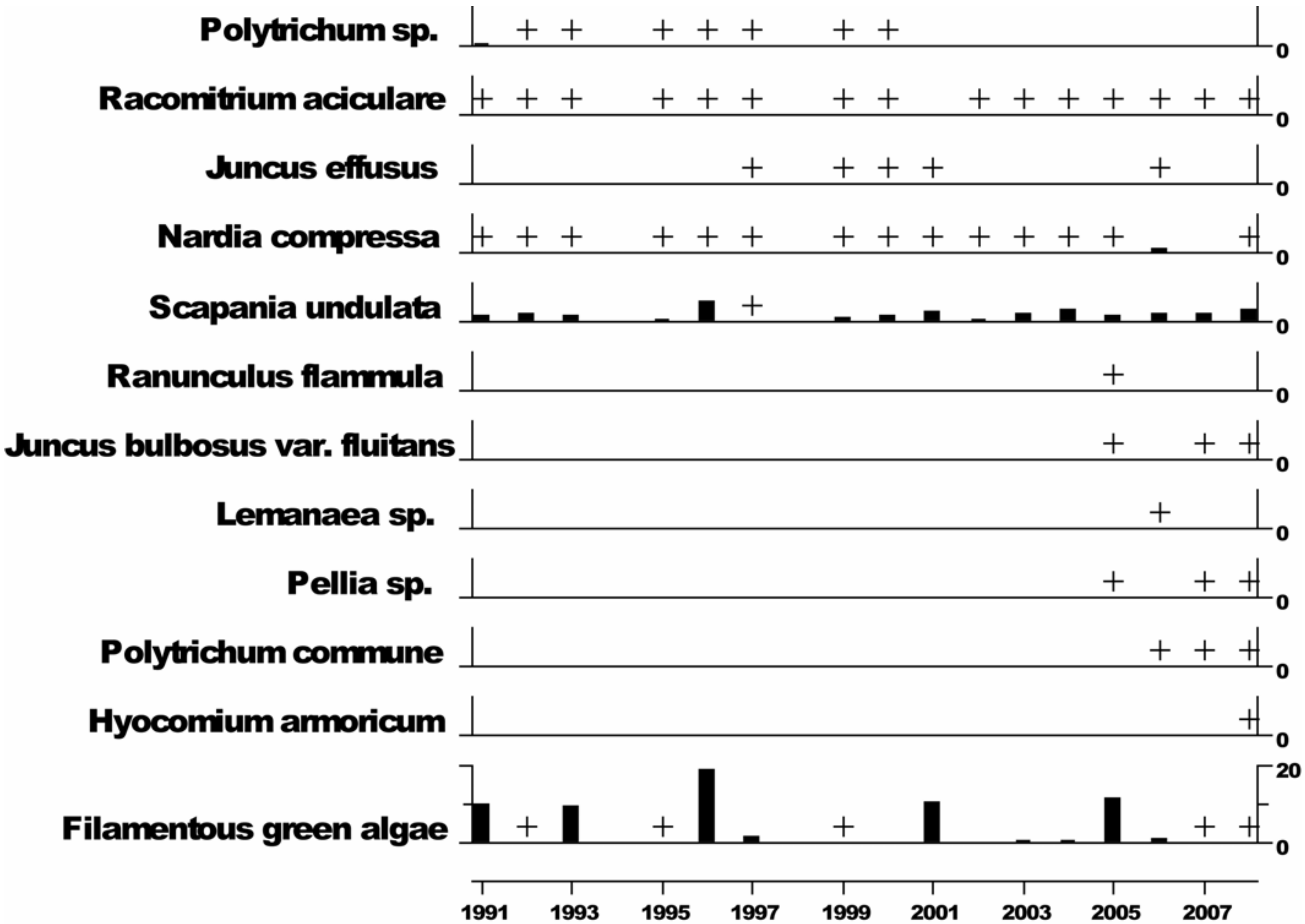


Figure 8.10. Aquatic macrophyte percentage species cover (+ represents >1% abundance), Afon Gwy

In common with the Afon Hafren, the aquatic macrophyte flora in the survey stretch of the Afon Gwy is restricted largely to one acid tolerant liverwort species, *Scapania undulata,* which has shown no indication of long-term changes in coverage. No acid-sensitive mosses have been recorded in the survey stretch. However the acid-sensitive red alga, *Lemanea* sp., which is common more circumneutral streams, was recorded for the first time in several locations in 2006 and found again in 2009.

8.8 Summary

8.8.1 Chemistry

8.8.2 Biology

8.9 Afon Gwy Recent Publications Using AWMN Data

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9 Acidification and Recovery in the Welsh AWMN Sites.

The monitoring data presented show clearly that there has been significant change in both the chemistry and biology of Welsh sites in the Acid Waters Monitoring Network (AWMN) consistent with the reduction in acid deposition that has occurred over the last 20 years. A key issue now is the extent to which improvement that has taken place meets the targets for recovery required by current legislation governing acid and acidified surface waters in the UK. Here we present the results of three different methods for the assessment of actual or potential recovery from acidification, based on;

1. critical load models, showing where achievement of critical loads provides the necessary reductions in acid deposition to facilitate chemical recovery over the longer term (years to decades);
2. dynamic acidification models, calibrated to observed water chemistry data to determine how chemistry has changed from a pre-industrial baseline (here assumed to be 1860); and
3. palaeolimnological studies (lakes only) linking changes in preserved diatom communities in lake sediments to changes in lakewater pH though the use of diatom-pH transfer functions.

The critical loads approach is currently used under international conventions including the Gothenburg Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (1999) under the UNECE Convention on Long Range Transboundary Air Pollution and the EU National Emission Ceiling Directive of 1999. The approach for surface waters employs simple mass-balance hydrochemical models to determine the deposition load of acidity that will depress acid neutralizing capacity (ANC) below a pre-selected critical value.

9.1 Critical loads and exceedance at Welsh AWMN sites

The critical load is defined as: "*a quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge*" (Nilsson & Grennfelt, 1988). For critical load applications in the UK, a critical chemical value for ANC of 20 µeq l^{-1} is used for most waters, corresponding to a 10% probability of damage to brown trout populations according to a wide-ranging study of Norwegian lakes (Lien *et al.*, 1996). For the four Welsh AWMN sites, critical load exceedance implies that deposition will depress ANC below 20 µeq l^{-1} at long-term steady-state and conversely, reduction of deposition below the critical load is required to allow long-term recovery of ANC to the critical value of 20 µeq l^{-1}.

Two complementary static modelling approaches have been used here, the Steady-State Water Chemistry (SSWC) model (Henriksen *et al.*, 1992) and the First-order Acidity Balance (FAB) model (Posch *et al.*, 1997; Henriksen & Posch, 2001). In the SSWC, the critical load exceedance uses measured water chemistry to calculate the critical load and measured NO_3^- concentration to provide an estimate of the leaching flux contributing to critical load exceedance. For N deposition this may be considered a "best-case" scenario where the possibility of N saturation and increased NO_3^- leaching under constant deposition is ignored. To calculate exceedance with the SSWC model, contemporary NO_3^- leaching data are required for a given deposition period because measured NO_3^- is used in lieu of N deposition data in the calculation of exceedance. For FAB the situation differs in that once the model has been applied to provide various critical load parameters then in theory an exceedance for any specified period of deposition data may be calculated without the requirement for contemporary NO_3^- data.

Best case critical load exceedances using the SSWC model with contemporary water chemistry for four deposition periods, including the future 2020 scenario employed in the recent Review of Transboundary Air Pollution (RoTAP), are shown in Figure X.1.

All four sites showed exceedance of critical loads in the 1980s and 1990s. Although contemporary water chemistry data are missing for the period 1986-88 for the Afon Gwy, deposition loads were greater across the country in the mid 1980s relative to the 1990s so it is reasonable to assume that critical loads would have been exceeded in the 1980s. For the most recent period of deposition data (2004-06) only Llyn Cwm Mynach still exceeds its SSWC critical load (exceedance = 0.01 keq ha^{-1} yr^{-1}); negative values of exceedance at the other sites indicate that deposition levels are lower than the critical value. By 2020, all four sites will no longer exceed SSWC critical loads. Hence under the best-case modelling scenario, three sites were already undergoing chemical recovery by 2004-06 and all four will be recovering towards the critical ANC of 20 μeq l^{-1} by 2020, although static models do not indicate how long this recovery will take. Note that increasing ANC may still occur under declining deposition loads which are still above the critical load, but can only reach the critical ANC value when the critical load is no longer exceeded.

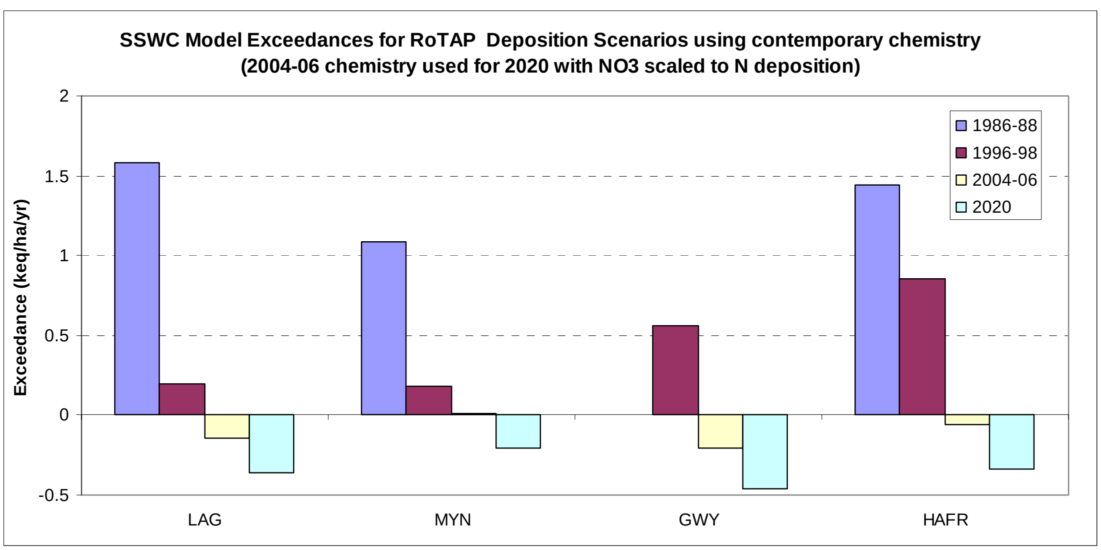

Figure 9.1: **SSWC Model critical load exceedances (best case) under four deposition scenarios (NB no contemporary chemistry available for Afon Gwy in 1986-88)**

Using the same four deposition scenarios with the worst-case FAB model, a very different picture is apparent. If NO_3^- levels reach those predicted by FAB, then all four sites are exceeded under all scenarios including 2020 when further reductions in emissions from the present are expected (Fig. X.2). Again, with a static model, timescales of change cannot be calculated, but the FAB model suggests that planned emissions reductions to 2020 will be insufficient to allow recovery of ANC to 20 μeq l^{-1} at any site over the long term.

Since the FAB model is used for UK national critical loads data submissions under the Gothenburg Protocol, these “worst-case” critical load exceedances contribute to the official dataset.

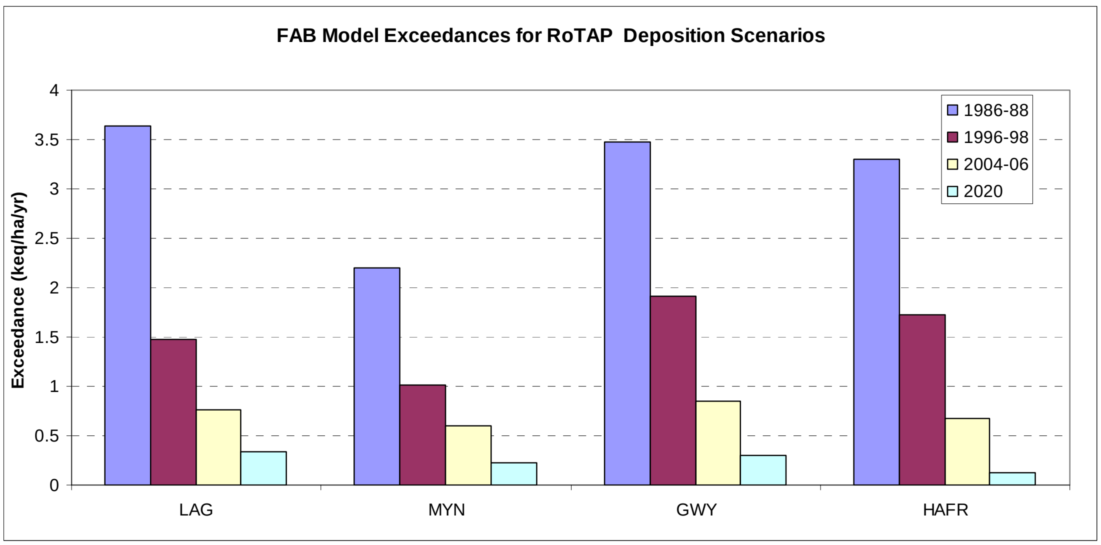

Figure 9.2: **FAB Model critical load exceedances (worst case) under four deposition scenarios**

9.2 Dynamic Modelling of Past and Future Change in Acidity at Welsh AWMN sites

Whilst the critical loads concept is a powerful tool for assessing acidification status for deposition loads for specific years, critical loads and exceedances provide no indication of the time over which damage or geochemical reversibility and biological recovery will be achieved in response to reduced emissions. The critical loads concept neglects the time components of acidification, namely soil buffering, sulphate desorption, nitrogen saturation, and organic matter dynamics. Dynamic models offer the only opportunity to determine the level of deposition reduction required to achieve a given chemical restoration target, and hence a biological response, within a given time scale.

The dynamic model MAGIC (Model of Acidification of Groundwater In Catchments) has been applied to the AWMN to assess the evidence for chemical recovery in long-term water chemistry data sets. MAGIC is a process-orientated model, developed to provide long-term reconstructions and predictions of soil and stream water chemistry in response to scenarios of acid deposition and land use (Cosby *et al.*, 1985). The model simulates soil solution and surface water chemistry to predict average concentrations of the major ions for annual time steps.

Deposition chemistry from the bulk deposition collector at Pumlumon (part of the UK Acid Deposition Monitoring Network) was used to drive the model from 1988 to 2007. Because the deposition collector is not directly co-located at the monitoring sites (except for Afon Gwy) and since bulk deposition does not accurately reflect dry and occult deposition inputs, the annual bulk deposition concentrations at each site were corrected to match the observed Cl^- and SO_4^{2-} in surface water. The historical (pre-1988) deposition sequences for SO_4^{2-}, NO_3^- and NH_4^+ were estimated by scaling currently observed deposition to reconstructions of S emissions (Warren Spring Laboratory, 1983) and N emissions (Wright *et al.*, 1998) from Concentration Based Estimated Deposition (CBED), based on a 5x5 km grid for 2004-06. All other ions in deposition were assumed to remain constant throughout the simulation.

MAGIC modelled trends in pH and ANC are shown in Figs. 9.3-4. Clear acidification is shown by declining pH from 1860 reference conditions at all sites, with the most rapid change occurring from about the 1940s. Recovery in pH is apparent from the 1970s onwards, with the greatest recovery towards reference pH values seen at Llyn Llagi. For Llyn Cwm Mynach the model calibrates very poorly to observed data and therefore the reference pH of c. 6.8 must be considered unreliable. Both the Afon Hafren and Afon Gwy show only partial recovery towards reference pH values of around 6.5.

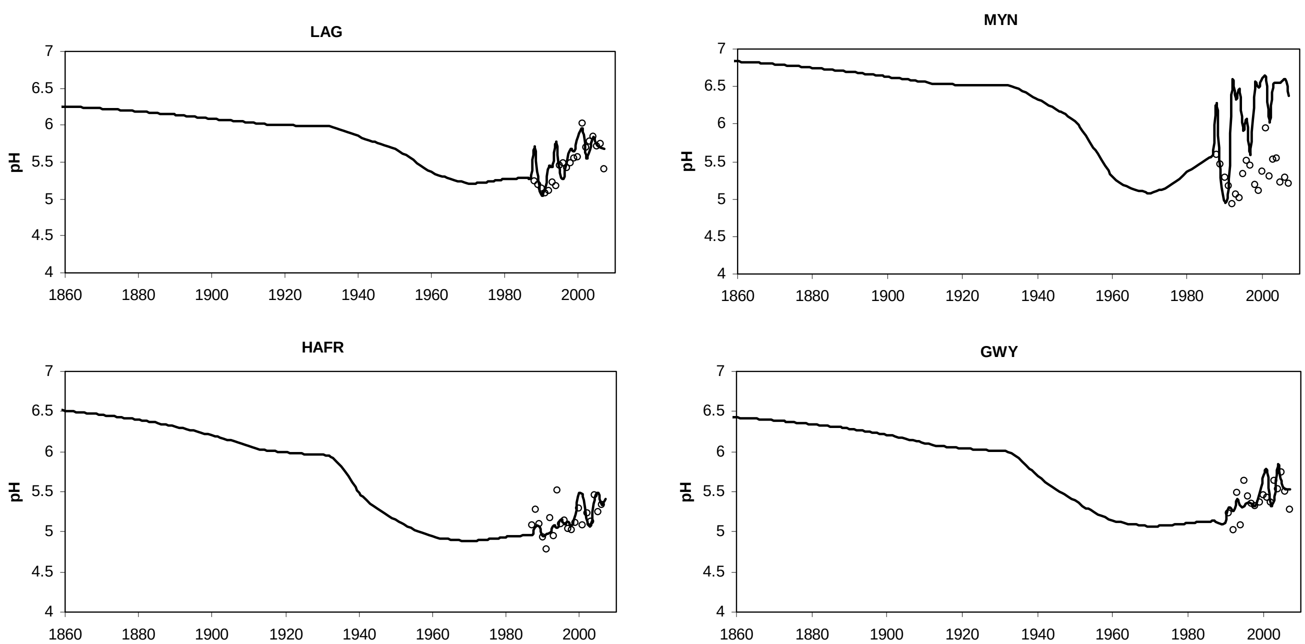

Figure 9.3.MAGIC modelled changes in pH from 1860 baseline to present (calibration); dots represent annual mean monitoring data (measured)

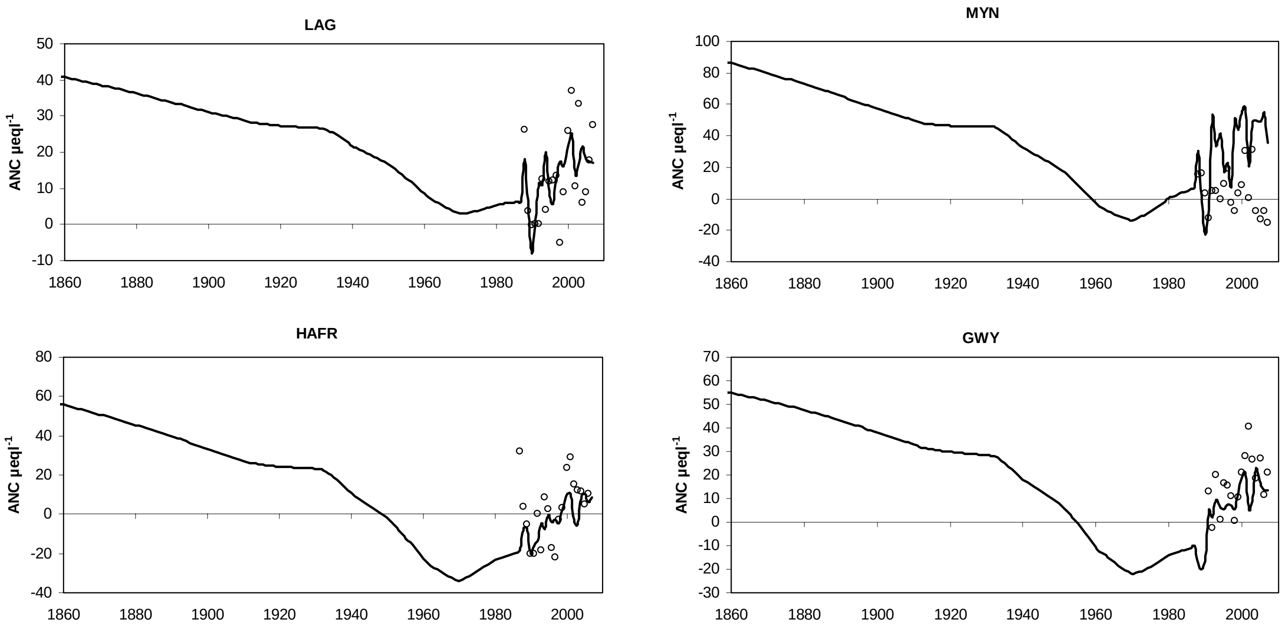

Figure 9.4: MAGIC modelled changes in ANC from 1860 baseline to present (calibration): dots represent annual mean monitoring data (measured)

For ANC MAGIC tracks measured data well at all sites except Llyn Cwm Mynach. Temporal trends in ANC closely match pH as would be expected. According to MAGIC, ANC declined below the critical value of 20 µeq l^{-1} by 1940 at Afon Hafren and Afon Gwy and around 1950 at Llyn Llagi and Llyn Cwm Mynach. At the Plynlimon stream sites, severe acidification is shown by the decline of ANC below zero in about 1950 at the Afon Hafren and a few years later at the Afon Gwy. However, both stream sites show a return to positive annual mean ANC values since about 2000 according to both monitoring data and MAGIC. Furthermore, ANC values occasionally exceed 20 µeq l^{-1} since around 2000 at all sites except Llyn Cwm Mynach, supporting the best-case SSWC critical loads showing non-exceedance of critical loads at present. The future role of NO_3^- leaching as predicted by the worst-case FAB model is therefore critical in determining whether chemical recovery will be sustained in these sites.

9.3 Acidification and recovery assessed using palaeolimnological studies of diatom communities in lake sediment cores and current monitoring data

Palaeolimnological studies at the two Welsh lake sites Llyn Llagi and Llyn Cwm Mynach have provided both historical changes in diatom communities and inferred changes in lakewater acidity status back to around 1800. The most direct method of assessing the degree of recovery at the AWMN lake sites is a comparison between the diatom assemblages of sediment core reference samples (representing pre-acidification conditions in approximately 1800 AD), the record of diatom assemblage change through to the top of sediment cores taken in the late 1980s (representing the conditions during or slightly after the period of maximum acid deposition in the UK) and the record of diatom change from samples through to the present day (2008) taken from the sediment traps that were installed in each AWMN lake in 1991 and emptied annually since then.

9.3.1 Chemical Recovery

The chemical reference conditions suggested by the diatom assemblage at around 1800 in the sediment core show pH values of 5.77 at Llyn Llagi and 5.95 at Llyn Cwm Mynach, though these are somewhat lower than MAGIC modelled pH values, especially at Mynach where there is clearly a calibration problem with MAGIC (Table Y.1). The extent of acidification at the start and end of the monitoring period can be assessed by comparing measured pH values with the reference; here we use 3-year mean values (for complete years of data only) to account for year-to-year variability in pH. At Llyn Llagi the mean pH of 5.13 at the end of the 1980s shows the site to be strongly acidified from the reference of 5.77, but by the end of the monitoring period the pH has recovered to 5.62. In fact this site shows one of the clearest chemical recovery signals in the entire Acid Waters Monitoring network (Kernan et al., 2010). Llyn Cwm Mynach, on the other hand, shows severe acidification relative to the diatom-inferred reference at both the start and end of the monitoring period, with no indication of recovery in pH. In fact Mynach shows the least evidence of chemical recovery of all sites in the AWMN. Reasons for the opposing recovery responses in the two Welsh lakes are not known, but it is possible that forestry activities at Llyn Cwm Mynach may be having an effect, though there is no direct evidence for this at present.

9.3.2 Biological Recovery

By combining the data from the traps with the diatom data from sediment cores as describe above, it is possible to track changes in the diatom flora of each site continuously from 1800-1850 AD through to the present day using, with the exception of the switch from core sediment to trap sediment, an identical standard methodology.

To follow the changes in the diatom assemblages at each site we have entered the core and trap data passively into a Principal Components Analysis (PCA) of diatom assemblage data from a large data-set of 121 low alkalinity lakes from across the UK. The sites represent the full range of low alkalinity lake types found in the UK (Battarbee et al. in press) varying in degree of acidification, base cation status, DOC, altitude and distance from the coast. Each of the 121 sites is represented by two samples, one from the ca 1800 AD level in a core from that site and one from the surface sediment sample (= present day at the time of sampling). The diatom data from each AWMN site are thereby constrained by the range of variability in the overall data-set, and the time trajectory for each site shows the post-1850 changes in diatom assemblages caused by acidification and the post-1985-1990 changes that represent the response to emission reduction and the degree of recovery.

The sediment trap monitoring data at the start of monitoring map nicely onto the top of the sediment core for Llyn Llagi (Fig. X.5) and there are strong signs of recovery in the diatom community towards the reference assemblage. For Llyn Cwm Mynach, there is a gap between the top of the sediment core and the first trap sample, and no consistent overall trend in diatom assemblages towards the reference community. These biological indicators are therefore broadly consistent with recovery in pH (or lack of, in the case of Mynach) albeit more muted than chemical trends, suggesting a hysteresis in biological trends.

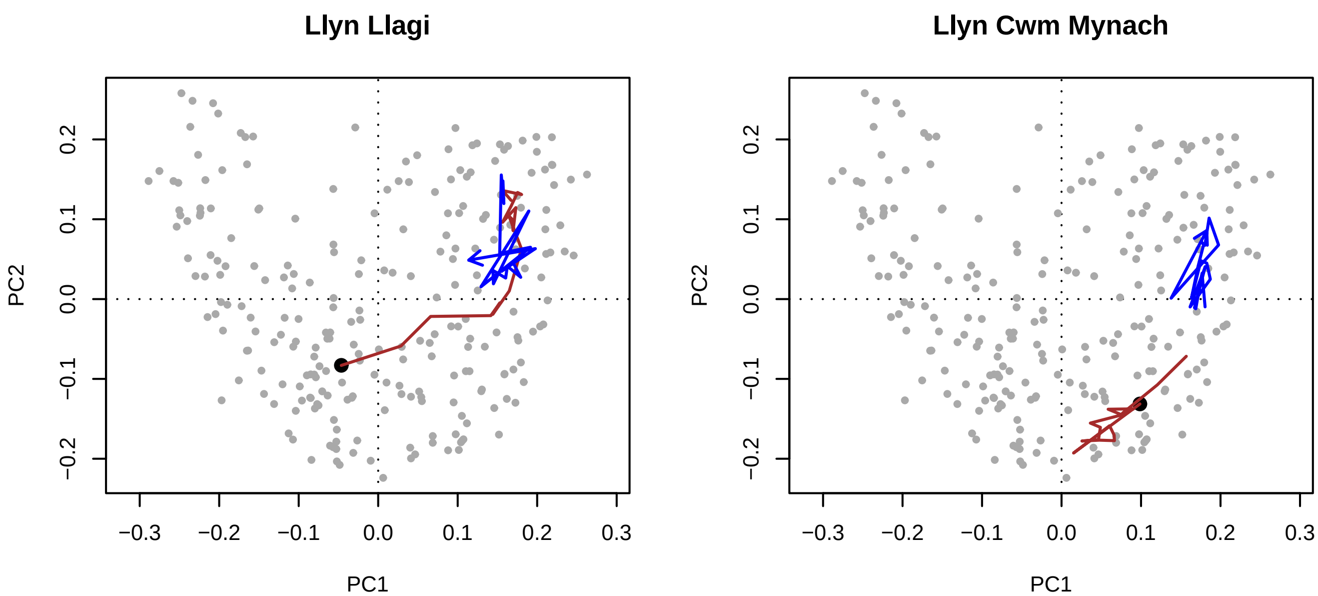

Figure 9.5: "Recovery" trajectories for diatom assemblages at two Welsh AWMN lakes based on differences between species composition diatom floras of the reference period through to the present day. Core data (ca. 1800AD to ca. 1990) are plotted in red, sediment trap data are plotted in blue. The black spot indicates the reference sample; the arrow head indicates the most recent (2008) sample.

Table 9.1 Reference (diatom-inferred, DI-pH, for lakes only, and MAGIC modelled) and measured pH (3-year mean) at start and end of monitoring period

Site	DI-pH (1800)	MAGIC pH (1860)	Mean pH (1989-91)	Mean pH (2005-07)
Llyn Llagi	5.77	6.26	5.13	5.62
Llyn Cwm Mynach	5.95	6.83	5.29	5.23
Afon Hafren	n/a	6.58	5.08	5.34
Afon Gwy	n/a	6.42	5.12*	5.48

*Mean for later start period April 1991-March 1994

For the two stream sites there are no comparable diatom data, but the MAGIC reference and measured data from the start and end of the monitoring period are provided in Table Y.1 for comparison. MAGIC suggests a comparable decline in pH to the lake sites from reference to the late 1980s. Both streams do however show recovery in pH by the end of the study period, although not as pronounced as at Llyn Llagi.

9.4 Comparison of CLRTAP and WFD Approaches to Assessing Recovery

The critical loads approach employed under the Gothenburg Protocol of the UNECE-CLRTAP uses a fixed value of ANC against which exceedance or non-exceedance is calculated. While based on sound empirical data, the critical ANC value used here takes no account of natural spatial variability in sensitivity to acidification or in the pre-industrial reference conditions found at a site. Hence a very acid sensitive site may have a reference ANC value close to 20 µeq l^{-1}, meaning that the critical load will be very small and very little change in ANC is allowed by the approach. Conversely, a more circumneutral site may have a reference ANC of, say, 100 µeq l^{-1} and still be acid sensitive; in this case the critical loads approach allows a major change in the chemical status of the lake before the critical load is exceeded.

In contrast to many earlier water directives, and those discussed above, where chemical drivers and targets are of key importance, the EU Water Framework Directive (WFD) places emphasis on the ecological structure and function of aquatic ecosystems with biological elements (fish, invertebrates, macrophytes, phytobenthos and phytoplankton) at the centre of the status assessments, and hydromorphology and physico-chemistry as supporting elements. Ecological quality is judged by the degree to which present-day conditions deviate from those prevailing in the absence of anthropogenic influence, termed reference conditions. Sites in which the various elements correspond totally or almost totally to undisturbed (reference) conditions are classed as High status. Four further categories of Good, Moderate, Poor and Bad status refer to the degree of deviation from the reference state.

In the context of the WFD, the key measure of damage and/or recovery is therefore change relative to reference conditions. Both the dynamic hydrochemical model MAGIC and the palaeolimnological diatom-pH transfer function can provide estimates of certain chemical reference conditions (e.g. pH). For comparing CLRTAP and WFD approaches here though, changes in ANC from the MAGIC modelled reference are assessed for the beginning and end of the study period (Table Y.2).

Table 9.2. Reference (MAGIC modelled) and measured ANC (3-year mean) at start (1989-91) and end (2005-07) of monitoring period, with relative change towards targets of 20 µeq l^{-1} or reference value through the period of monitoring

Site	MAGIC ANC (1860)	Initial ANC	Final ANC	% recovery to ANC_{crit}	% recovery to Reference
Llyn Llagi	30.4	3.6	17.5	85%	52%
Llyn Cwm Mynach	33.4	10.7	7.1	0%	0%
Afon Hafren	42.0	1.7	10.4	48%	22%
Afon Gwy*	35.9	9.1	19.8	98%	40%

*Mean for later start period April 1991-March 1994

According to MAGIC, all four sites are naturally very acid sensitive with low reference ANC values, with the highest value being found at the Afon Hafren and the lowest at Llyn Llagi. The critical load approach therefore allows a much greater change in ANC at the Hafren before the critical load is exceeded (ANC threshold of 20 µeq l^{-1} is crossed), while a much smaller critical load would be calculated for Llyn Llagi. At the start of monitoring, all four sites had ANC values much lower than the critical value of 20 µeq l^{-1} and showed critical load exceedance according to the best-case SSWC model (Fig. X.1). At the Afon Hafren, the ANC decline from the reference value was >40 µeq l^{-1} while the smallest decline was found at Llyn Cwm Mynach (decline of c. 23 µeq l^{-1}). By the end of the monitoring period, both Llyn Llagi and Afon Gwy had almost recovered to the critical ANC of 20 µeq l^{-1} while the Afon Hafren had only achieved a 48% closure of the ANC gap measured at the start of monitoring. Since ANC at Llyn Cwm Mynach declined slightly through the period of monitoring there was no recovery towards any target.

Under the more stringent demands of the WFD, only partial recovery was found at any site, with Llyn Llagi only showing a 52% recovery back towards the reference ANC over the period of monitoring. At the Afon Gwy, a 98% chemical recovery according to the terms of the Gothenburg Protocol equates to only a 40% chemical recovery under the requirements of the WFD.

10 References