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Evaluating climatic effects on aquatic invertebrates in southern English rivers

Science Report – SC070046

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Steve Killeen

Head of Science

Executive summary

Long-term ecological data are critical for assessing the effects of climate change, but are generally scarce. For British rivers, the Environment Agency's routine survey data on river invertebrates might be usable, but few investigations have assessed whether they reveal climatic trends.

For 52 sites on 50 surface- and chalk-fed rivers in southern England (Dorset and Wiltshire Avon, Bristol Avon, Hampshire and Devon), trends in invertebrate composition and crude abundance were assessed for 10 to18 years over the period 1989-2006. Five sites had data spanning 31-35 years, but their use was affected by a change in sampling method in 1989. Invertebrate trends were examined in relation to temperature, discharge and major water-quality variables that might affect climatic signals. Emphasis was placed on average pattern across sites, since these were more likely to reflect region-wide climatic trends.

River temperature data were scarce in the study region and only one site (Itchen at Southampton) had long-term records (1978-2006). A strong relationship between these records and air temperature at Southampton airport ($r^2 = 0.92$), allowed the reconstruction of a full record from 1980 to 2006. Over this period, average winter temperatures increased by 2.7 °C (± 0.7 SE), average summer temperatures by 1.3 °C (± 0.8 SE) and summer maxima by 2.0 °C (± 1.2 SE), with trends more pronounced in winter ($r^2 = 0.36$) than summer ($r^2 = 0.10$). Winter temperatures tracked the North Atlantic Oscillation only weakly ($r^2 = 0.06$), but long-term winter trends (1982-2006) in the Itchen were related significantly to more distant streams at Llyn Brianne in upland Wales ($r^2 = 0.53$).

Other chalk-fed sites on the Avon (Amesbury) and Bere stream had continuous temperature runs of three to four years. Monthly means correlated strongly with the Itchen ($r^2 > 0.97$), with intercepts differing by less than 2.9 °C and slopes by less than one. Using the period 2002-2006 for calibration, and long-term trends in the Itchen as an index, the chalk-fed Bere and Avon have probably warmed respectively by 1.8°C (± 0.9 SE) and 2.4°C (± 1.3 SE) in winter, or 0.9°C (± 1.1 SE) and 1.2°C (± 1.4 SE) in summer, over the last 26 years. Although characterised by uncertainty, these patterns imply that i) long-term temperature records like the Itchen can represent index sites against which to calibrate other southern rivers and ii) river warming has occurred widely in southern Britain over the last 25 years, mostly in winter.

While dry and wet years occurred throughout the data run, there were no marked trends in discharge that were consistent across all monitored sites.

Large variations in invertebrate assemblages were found across the region, and sites were classified prior to trend analysis into four groups: chalk-fed rivers in Dorset and Wiltshire Avon, and the Bristol Avon (Group 1); chalk-fed streams in Hampshire (Group 2); surface-fed streams in Hampshire (Group 3) and surface-rivers in Devon and the Bristol Avon (Group 4).

Over the period 1989-2007, there were small but significant shifts in the composition of invertebrate assemblages in chalk-fed Groups 1 and 2, on average by around five per cent. After accounting for altered sampling methods from 1989 onwards, rates of change were near identical at four long-term chalk-fed sites, where composition changed by around 14 per cent over the period 1977-2007. Trends indicated a shift towards taxa typical of faster flow and/or well oxygenated conditions. No unidirectional shifts were apparent in composition across sites in Groups 3 and 4.

Among 26 widespread invertebrate taxa (those occurring at more than half the sites), 14 changed through time (1989-2007) in average rank-abundance in spring or autumn at either Group 1 or Group 2 sites. Those with at least one positive trend included Planaridae, Heptageniidae, Caendiae, Ephemeridae, Baetidae, Limnephilidae, Sericostomatidae and Lepidostomatidae, while Erpobdellidae, Simuliidae and Rhyacophilidae declined. Rank-abundances of twelve taxa also had significant trends at Group 3 or Group 4 sites.

Winter river temperature (indexed to the Itchen) explained between 24 and 40 per cent of the inter-annual variance in average assemblage composition in Group 1 and 2 sites during 1989-2006, and 45 per cent of the variance at four long-term sites in Group 2 (1980-2006). Long-term variations in composition at one Group 3 site (Beaulieu) also tracked temperature (r^2 = 0.22). However, these potential temperature effects were confounded by marked, region-wide trends in water quality at Group 1 and 2 sites from at least 1989 onwards. Concentrations of ammonia, orthophosphate and biochemical oxygen demand (BOD) all declined significantly across the sites in each group (r^2 = 0.25-0.86), while total oxidised nitrogen increased (r^2 = 0.52-0.69). These effects were tracked by increases in biological monitoring working party (BMWP), average score per taxon (ASPT), taxon richness and family LIFE (Lotic Index for Flow Evaluation) scores (r^2 = 0.35-0.70). Water quality trends, and invertebrate responses, were less clear in Groups 3 and 4 but still apparent. We conclude that, despite increased river temperatures in southern Britain over recent decades that are consistent with climate change, region-wide ecological effects cannot be detected from these data. Three possible explanations are that: i) only limited climatic effects on southern river organisms have occurred; ii) effects have been masked by data resolution, particularly family-level identification; iii) effects have been negated or subsumed by stronger trends in water quality over the period evaluated. Detection of the third possibility reveals the value of long-term chemical and biological monitoring by the Environment Agency.

We recommend:

i) Further investigation of all or some of these data using multiple regression, factorial analysis and further temperature modelling should be carried out, to separate the ecological effects of water quality, discharge and climate.

ii) The Environment Agency should consider whether its current strategy for biological sampling is adequate to detect and appraise climate change effects, particularly with respect to taxonomy and sample-site distribution. Investigations at British sites with species-level data (such as Llyn Brianne; various locations in Eastern England) should be done to assess whether family-level identification gives sufficient resolution to detect climatic effects.

iii) The Environment Agency should consider further the organisms and river types at risk of climate change, and consider also consequences for river ecological function and conservation.

iv) The Environment Agency should bolster its current programme for monitoring river temperature.

v) The Environment Agency and water companies should consider how best to appraise and manage interactions between water quality, abstraction, land use and climate change risks as part of an adaptive strategy. The evidence from this project is that positive management to diminish other pressures might offset some climate change impacts.

Acknowledgements

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1 Introduction

There is increasing concern that streams and rivers might be among the most sensitive of all ecosystems to climate change for five principal reasons:

i) Temperatures in running waters track air temperature closely, particularly in headwaters, and many streams have already warmed in response to recent climate change (Caissie 2006; Durance and Ormerod 2007).

ii) Rivers are characterised by many ectothermic organisms, where variations in temperature have direct effects on the metabolism, growth, development, phenology and distribution of many river species (Hogg and Williams 1996; Daufresne *et al*. 2004). Temperature also affects organisms indirectly, for example through dissolved oxygen, primary production, organic decomposition and litter processing (Richardson 1992).

iii) Climate change effects on precipitation have direct corollaries for hydraulics, discharge, floods and droughts in rivers, which are all of central ecological importance (Lake 2000; Lytle and Poff 2004). Discharge pattern also affects organisms indirectly through interactions with water quality, fluvial geomorphology, habitat structure and connectivity between water bodies. As with temperature, the data show how discharge variations can regulate taxonomic composition and abundance (Lake 2000; Beche *et al*. 2006; Dewson *et al*. 2007; Bonada *et al*. 2007).

iv) Climate change interacts with other anthropogenic pressures on river systems such as eutrophication, organic pollution, acidification and abstraction (Root *et al*. 2003; Evans 2005; Daufresne *et al*. 2007). Interactions here are complex, but include a blend of altered metabolic activity, the dilution of pollutants and changes in the frequency of flood-mediated effects such as acidic episodes or discharge of combined sewer overflows (Wilby *et al*. 2006).

v) Climate change interacts with quasi-natural climatic variations such as the North Atlantic Oscillation (NAO) (Blenckner and Hillebrand 2002; Hallett *et al.* 2004; Beche and Resh 2007). By affecting temperature, precipitation and water quality, the NAO affects the growth, phenology and persistence of stream organisms in Western Europe (Elliott *et al*. 2000; Bradley and Ormerod 2001; Briers *et al*. 2004). Although the strength of the NAO is probably linked to greenhouse-gas forcing (Osborn 2004; Shindell 2006), distinction between the ecological effects of the NAO and directional climate change is important in climate change studies (Straile and Adrian 2000; Bradley and Ormerod 2001). Moreover, the ecological effects of such large-scale factors as the NAO are interesting as potential analogues of future climate change.

Detecting climate change effects on rivers is now an important requirement in river management. The practical benefits include weighing the effects of abstraction, assessing possible interactions with other pressures, understanding the effects of climate on monitoring data, identifying resources at risk, and developing adaptive management responses (Wilby *et al* 2006). A prime need at present is the evaluation of long runs of ecological data, but these are scarce globally (Jackson and Fureder 2006).

Recent work by the Catchment Research Group (Cardiff University) under the Environment Agency-funded PRINCE project (*Preparing for the impacts of climate change on ecosystems*) illustrated, for the first time, substantial climate change effects on British rivers and river invertebrates (Durance and Ormerod 2007). PRINCE also demonstrated the feasibility of models to simulate future consequences. So far, however, such trends have been appraised only in Wales and Northern England. There has been no similar work on southern English rivers, and in particular chalk streams, despite their national importance to fisheries and conservation.

This report used data from 52 sites on 50 surface- and chalk-fed rivers in southern England (Dorset and Wiltshire Avon, Bristol Avon, Hampshire and Devon) to assess trends in invertebrate composition and crude abundance over the period 1989-2006 in relation to temperature, discharge and major water-quality variables that might confound or mask climatic signals. Particular emphasis was placed on average pattern across sites, since these were more likely to reflect region-wide climatic effects. However, specific attention was given to trends at five sites with longer-term data spanning 31-35 years.

This study tested the three general hypotheses that:

> i) Unidirectional changes in hydrology and/or thermal regime are now in progress in southern English rivers.
>
> ii) The character of any such changes varies between surface and chalk-fed rivers.
>
> iii) Trends in temperature or discharge in chalk- or surface-fed rivers have been sufficient over recent decades to cause measurable change among aquatic macroinvertebrates against background variability.

Our specific objectives were to:

1) Establish whether any unidirectional trends in temperature or discharge have been apparent over the study period that might reflect climate change.
2) Measure any variations in invertebrate assemblages (composition, family-specific abundance and biotic indices) in relation to temperature and discharge.
3) Assess spatial patterns in any climatic response.
4) Establish whether any other factors might confound or mask climatic effects on invertebrates, in particular water quality.

Further analysis of these data will be reported in a second phase of work, and also in a paper prepared for publication in the peer-reviewed literature (Durance and Ormerod *submitted*).

2 Methods

2.1 Site selection and data sources

Sites for this study were chosen to reflect a blend of surface- and chalk-fed streams in southern England (Dorset and Wiltshire Avon, Bristol Avon, Hampshire and Devon) where other pressures were minimal or constant through time. Sites or rivers were ideally required to have data on invertebrates, water quality, temperature and discharge, with data runs that were as long as possible. In practice, however, discharge data, and in particular temperature data, were collected from a far smaller array of locations than invertebrates and water quality. Moreover, sites with invertebrate records prior to 1989 were scarce.

2.1.1 Temperature data

Only one long-term run of river temperature data was available (1978-2006), collected by the Environment Agency using daily measurements at 09:00 by mercury thermometer on the River Itchen 38 km from the source (SU 454 157; Figure 2.1). Within one km of this site, dry-bulb air temperatures were continuously recorded (every 15 minutes) between 1980 and 2006 by the UK Meteorological Office at Eastleigh (SU 449 168; 11 m above sea level). Shorter data runs, using 15-minute continuous electronic recording, were available from the Avon at Amesbury (SU151413; 2004-2007) and from 10 sites in the Frome and Piddle catchments, Dorset, sampled by the Centre for Ecology and Hydrology (CEH) during the LOCAR project (2002/03-2006).   Temperature availability was explored for the Bristol Avon, Devon and Avon areas, but data were in private hands, or were too irregular (such as Cannington Brook, Bristol Avon) or too short for use (Wylye and Ebble data sets, Avon).

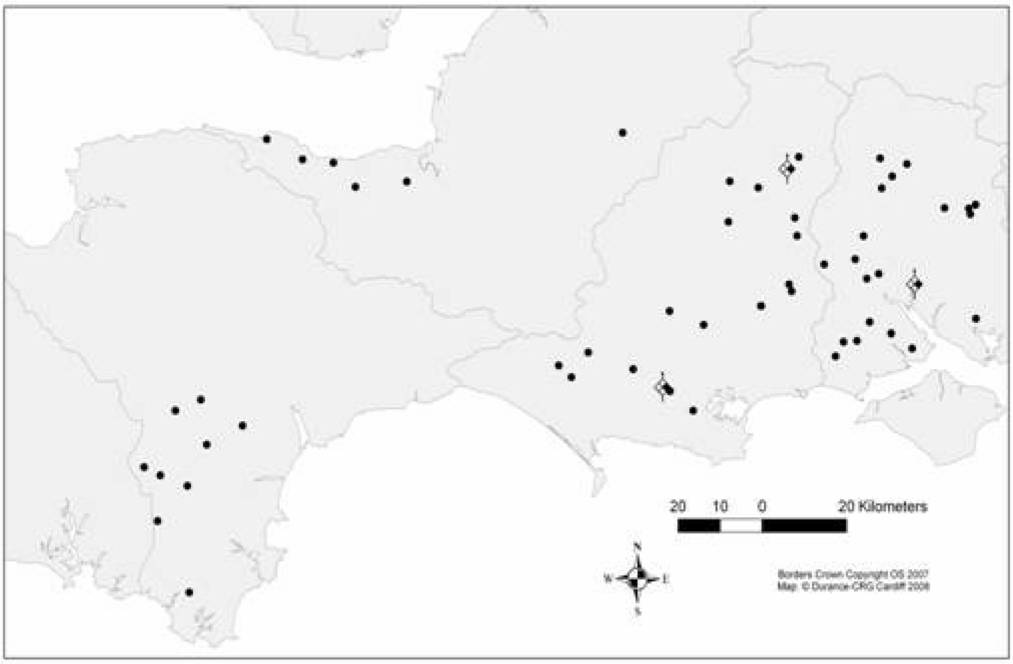
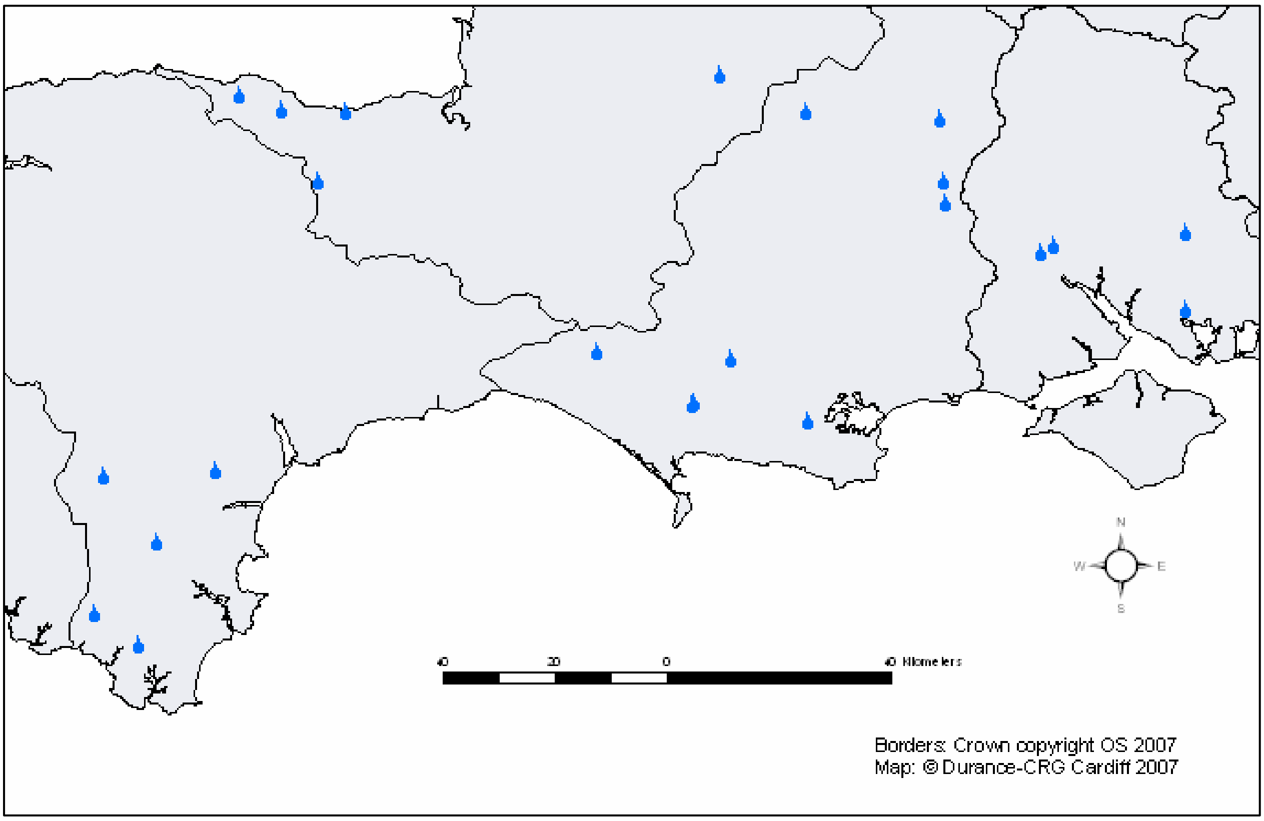


Figure 2.1 Locations with river temperature data in southern England. Special symbols indicate sites with data runs of sufficient length for use in this study (see Section 2.1.1)

2.1.2 Discharge data

Discharge data were available from 22 locations gauged either by the Environment Agency (National Riverflow Archive) or CEH, for example during the LOCAR project (Figure 2.2). All data were collated as annual average daily flows expressed in m^3 per second (m^3/s). Some of the sites, such as the Bristol Avon, only had data runs from 1995 onwards.

Figure 2.2 Locations with discharge data used in this project



2.1.3 Other climatic indices: North Atlantic Oscillation

In addition to data on discharge and temperature, the North Atlantic Oscillation (NAO) was used in explorations of temperature trends. The NAO was parameterised using the winter index (December-March inclusive; provided by the Climate Analysis Section, NCAR, Boulder, USA; Hurrell 1995), calculated from the difference is sea-surface pressure between the Azores and Iceland (Hurrell *et al.* 2003). Positive values are associated with mild, wet winters in North West Europe and negative values with cold, dry winters (Hurrell 1995).

2.1.4 Chemical data

Most invertebrate sites had Environment Agency chemical data available at the same sampling locations or within 500 metres. Despite small numbers of missing data before 1995, total oxidised nitrogen, ammonia, biochemical oxygen demand (BOD) and orthophosphate were generally characterised by good data availability, with 84 per cent of sites having mean values from monthly samples for 15-17 years. These variables were used as indicators of nutrient or organic loading from common pollution sources.

2.1.5 Macroinvertebrate data

Macroinvertebrate data for each site (Figure 2.3) were obtained from the Environment Agency BIOSYS data base. The 52 stream locations were chosen by the Environment Agency on the criteria that they should be from the upper section of rivers (called the “trout and grayling” zones by the Environment Agency); as free from confounding effects such as pollution and abstraction as possible; as long-running and complete as possible in sample collection, and reflecting a mix of surface-fed and chalk-fed streams.

The final data set for region-wide analysis comprised 1,116 invertebrate kick-samples, identified mostly to family level. They were collected at an average frequency of 21 samples per site (± 11.8 SD) using standard Environment Agency protocols, mostly from spring (March, April, May) and/or autumn (September, October, November). Average coverage spanned 15.6 years (± 6.3 SD) per site from 1989-2007 (Figure 2.4). Invertebrate taxonomy, coverage by taxa and systematics were harmonised so that 79 families were retained, all recorded on the Environment Agency log abundance scale (1, 2, 3, 4…). Because of changes in some aspects of taxonomic resolution through time, some families were pooled (for example, Limonidae, Pedicidae and Scertidae were grouped under Tipulidae; Lumbricilidae, Naididae, Tubifidae, Haplotaxidae and Lumbricidae were pooled as Oligochaetes; Dugesidae were included in Planaridae; Bithynidae were included in Hydrobiidae; and families typical of the terrestrial margins were deleted).

Five locations in Hampshire (see Figure 2.3) had longer-term data from 1978-2006, and these were subject to specific analyses (see below). However, the data run at each site was affected by a change in sampling method from 1989 onwards, when Surber sampling (five replicate samples per occasion) was replaced with kick-sampling across all habitats. The assessment of long-term trends in invertebrate composition at these sites is dependent on a correction applied to adjust the slopes and intercepts of trends before and after this methodological change (see Results section).

2.1.6 Indices of ecological quality (BMWP, LIFE scores...)

In addition to family-level identifications and abundance values for invertebrates, biological monitoring working party (BMWP), average score per taxon (ASPT), number of taxa and family LIFE scores were also extracted.

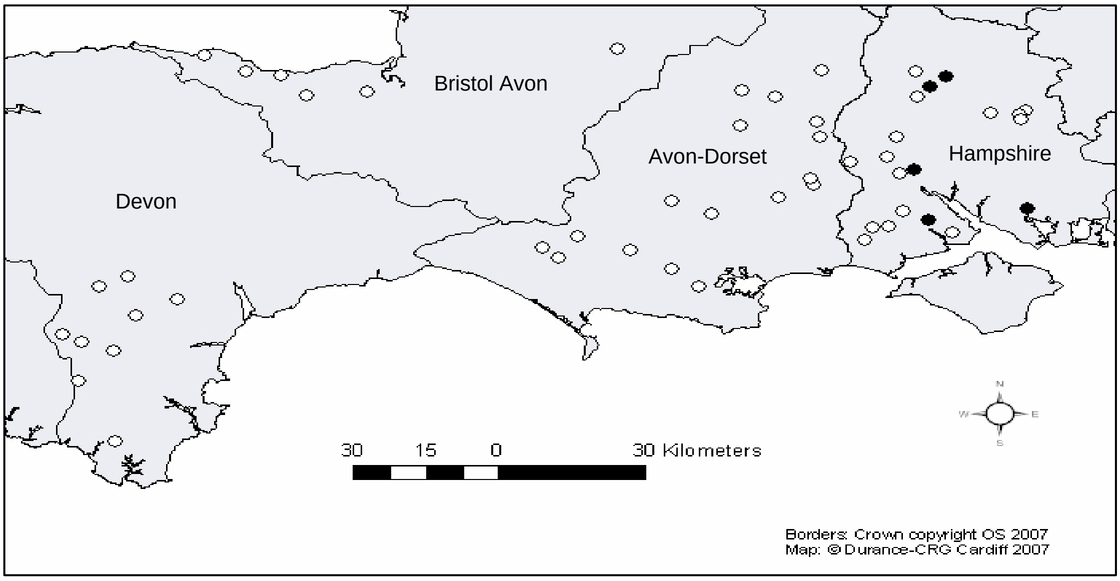
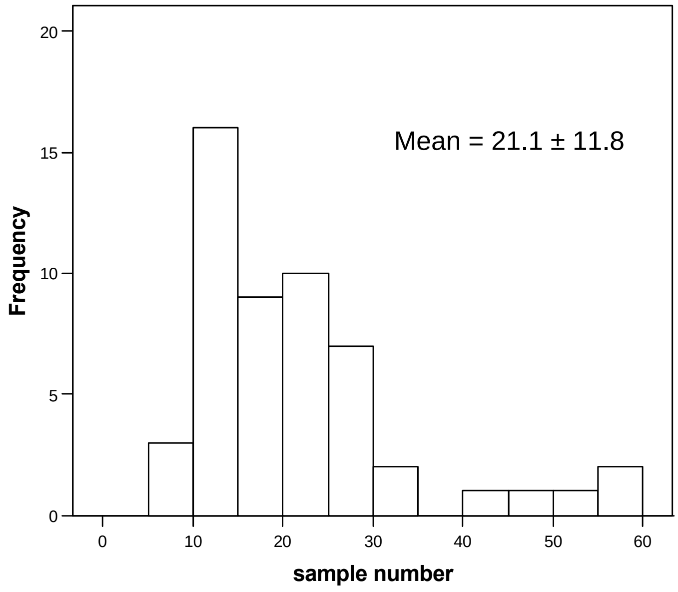


Figure 2.3 Locations of macroinvertebrate sampling sites used in this project. Long-term sites are indicated by solid dots.

a



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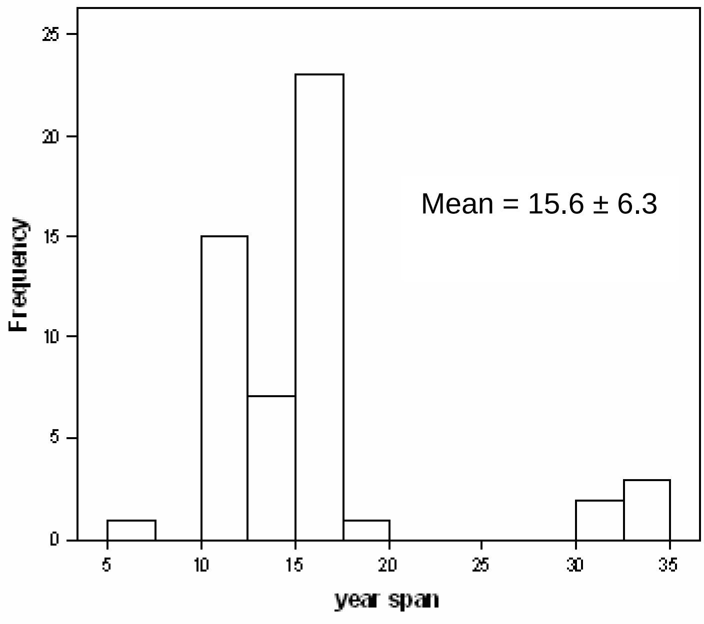


Figure 2.4 Frequency-distributions of number of invertebrate samples (a) and year span (b) for the macroinvertebrate sampling-sites used in this project

2.2 Data analysis

2.2.1 Trends in temperature and discharge

Despite the length of the River Itchen temperature record, measured temperatures were missing for winter 1996 and the entire year of 2000. Prior to any further analysis, we filled these gaps by reconstructing monthly river temperatures from the monthly air temperature at Eastleigh using a highly significant regression (Table 2.1). Trends through time in daily temperature records from the Itchen were assessed using linear regression of monthly mean values, winter means (December-February), summer means (June-August), summer 75 percentiles and summer maxima against time. Winter mean temperatures for the Itchen were also regressed against winter index values of the NAO, and against winter mean values for long-term data available from Llyn Brianne (central Wales; Durance and Ormerod 2007), 217 km to the north west.

To assess how closely temperatures in the Itchen reflected values at other sites in southern England, we regressed its monthly mean values against corresponding monthly means from the Avon at Amesbury (41 km to the north west) and the Bere stream in Dorset, 67 km to the south west for the period where data from both were available (2002/3-2006). The resulting regressions allowed the calculation of longer term trends in temperature at these other sites, assuming that the 2002-2006 inter-calibration period could be taken to represent longer-term behaviour.

The assessment of trends in discharge was complicated since sites with available data differed in absolute average daily flow. Before trend analysis, therefore, annual mean discharge for winter and summer (defined as above) was standardised within sites by subtraction of the mean and division by the standard deviation. Standardised discharges in each year relative to the overall mean for each site could now be pooled for region-wide assessments of trends. Trends through time were assessed using regression in each of four site groups from our classification of the invertebrate assemblages (see below).

2.2.2 Trends in composition and abundance of macroinvertebrates

Prior to any trend analysis, overall variations in invertebrate assemblages across all sites and time were appraised using ordination by detrended correspondence analysis (DCA), a simple and flexible method of unconstrained ordination (Van Der Maarel 1969). DCA uses reciprocal averaging to order samples objectively according to the frequency of co-occurrence of their constituent taxa. In other words, it orders sites or samples such that those with similar composition are placed together. Sample scores reflect turnover in taxonomic composition along orthogonal axes such that four standard deviations are equivalent to 100 per cent change on any one axis. Scores can be related quantitatively to sample attributes or conditions, in this case climatic variables or water quality. The resulting relationships allow the prediction of ordination score under new conditions. For this study, DCA was performed for each stream type using CANOCO 1.4, with rare species downweighted and abundances log transformed.

Once ordinations were complete, DCA scores on the first four axes were classified objectively using Ward's method so that sites with the most similar taxonomic composition were grouped together. This classification was intended to control some natural sources of variation, allow a more reliable detection of any trends through time, and an investigation of trends in different river types as required by the aims of this report. The resulting sites were: chalk-fed rivers in Dorset and Wiltshire Avon, and the Bristol Avon (Group 1); chalk-fed streams in Hampshire (Group 2); surface-fed streams in Hampshire (Group 3) and surface-rivers in Devon and the Bristol Avon (Group 4).

Once sites were classified, mean DCA scores were determined across all the sites in each group in each year to avoid pseudo-replication, and so that average behaviour among sites within stream groups could be assessed through time using regression. Such analyses were performed for each group by regression mean DCA score against year, temperature and other potentially important variables such as water quality.

The assessment of trends in abundance using Environment Agency invertebrate data presented a particular problem because of the recording of family-specific abundance values as logarithmic categories in which 1 = one organism; 2 = two to nine organisms; 3 = 10-99 organisms; 4 = 100-999 and so on. We considered these values as rank-abundances values along an ordinal scale, and determined mean ranks across all the sites in each stream group for each year for all families occurring at more than half of all the sites. Next, trends in mean rank-abundance were regressed against year for each stream group individually.

2.2.3 Trends at long-term invertebrate data sites

The five long-term sites were also treated as part of the overall ordination by DCA. While trends through time were apparent at similar rates before and after 1989, the effect of the change from Surber to kick-sampling was evident (Figure 2.5). Any long-term analysis of trends in these data therefore required a correction to account for this change. There are risks of errors resulting from any such correction because the array of taxa collected using the two methods might not respond equally to any effects through time.

We first attempted to calibrate kick-samples (n = 43) against Surber samples (n = 129 aggregated at three per site) taken simultaneously at the same locations at 43 sites along eight rivers using data provided by Wessex Water. These sites were entirely independent from the main analysis, but data were harmonised to match the list of families in the main data sets, and abundances were transformed into logarithmic categories. The new samples were added as supplementary variables for scoring using the main DCA described in Section 2.2.2. Using this procedure, differences in DCA scores between kick and Surber samples were barely detectable on either Axis 1 (= 0.04 ± 0.01 SD) or Axis 2 (0.12 ± 0.08 SD), with values far less than those indicated between methods in Figure 2.5. The most likely explanation is that these additional samples came from rivers with simplified margins where differences due to sampling methods would be less pronounced. We therefore considered these data unsuitable for inter-calibrating the two sampling methods.

In the absence of any other suitable data, ordination scores from the long-term sites were corrected by adjusting pre-1989 samples by the difference in mean ordination scores (= 0.51) before and after 1989.

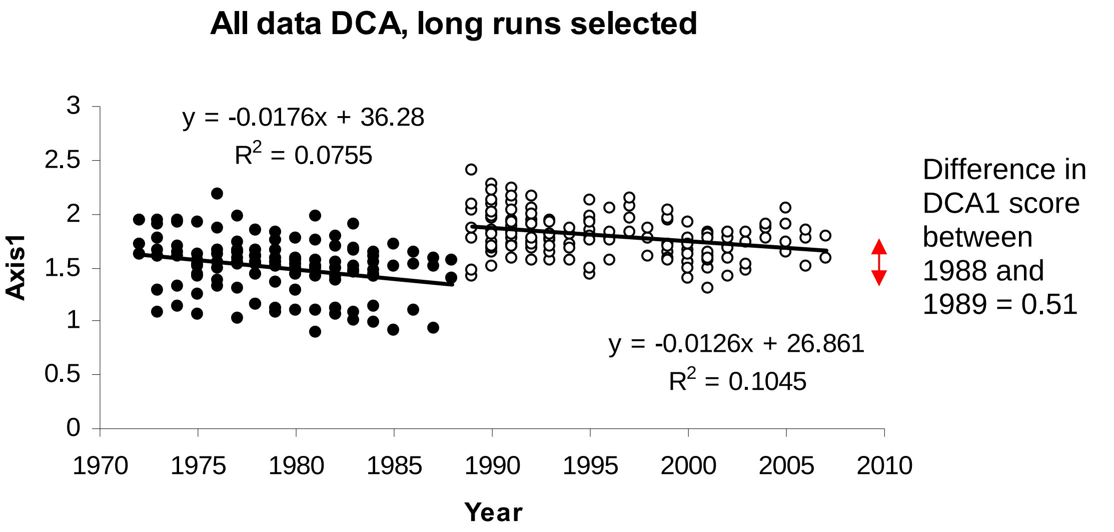


Figure 2.5 Preliminary ordination by DCA of all the invertebrate data (52 samples from 1972 to 2007) showing the positions of the five Hampshire sites with long-term data.

Four were chalk-fed streams (Wallington, Test at Chibolton, Test at Broadlands, Dever) and one was surface-fed stream. Pre-1989 data are shown as solid symbols, while post-1989 data are open symbols. Regression lines are indicated for each data-series.

2.2.4 Trends in water quality

Any systematic variations in water quality could be important in confounding, masking or exacerbating climate change effects on invertebrates. For each year, we therefore calculated average total oxidised nitrogen, ammonia, biochemical oxygen demand (BOD) and orthophosphate for all the sites in each of the four invertebrate groups (see Section 2.2.2). Trends through time were then appraised using regression. This procedure revealed aggregate trends among sites rather than site-specific patterns.

Using a similar procedure for averaging across sites, trends in water quality indicators, or other biotic indices, derived from invertebrates (BMWP, ASPT, LIFE scores and so on) were also assessed for each site group.

Table 2.1 Regression relationships ($y = a + bx$) between i) actual monthly river temperature (y) in the Itchen and monthly air temperature at Eastleigh (x) over the period 1980-2006; ii) monthly temperature in the Bere stream (y) and the River Itchen (x), 2003-2006; and iii) the River Avon at Amesbury (y) and the River Itchen (x) over the period 2004-2007; * $p < 0.05$; ** $p < 0.01$, *** $p < 0.001$. See maps for each location.

Dependent variable	Independent variables	a (±SE)	b (±SE)	100.r² (%)	F	d.f.
River temperature, Itchen	Air temperature, Eastleigh	2.90 (± 0.18)	0.76 (± 0.01)	92.0***	2741	1, 242
Winter temperature, Itchen	Year	-196.70 (± 55.11)	0.1023 (± 0.0276)	35.4***	13.7	1, 26
Summer temperature, Itchen	Year	-81 (± 60)	0.05 (± 0.03)	$9.5^{0.11}$	2.6	1, 26
Maximum temperature, Itchen	Year	-129 (± 93)	0.07 (± 0.05)	$11.8^{0.12}$	2.6	1, 25
75^{th} percentile Itchen	Year	-88 (± 72)	0.05 (± 0.04)	$8.4^{0.12}$	2.2	1, 25
Stream temperature, Bere	River temperature, Itchen	-0.0536 (± 0.315)	0.891 (± 0.023)	97.8***	1440	1, 33
Stream temperature, Avon at Amesbury	River temperature, Itchen	2.92 (± 0.23)	0.672 (± 0.017)	97.4***	1484	1, 41

3 Results

3.1. Trends in temperature and discharge

Over the period 1980-2006, reconstructed records of temperature from the Itchen revealed marked warming (Figure 3.1; Table 2.1). Despite some inter-annual variability, average winter temperatures increased by 2.7 °C (± 0.7 SE), average summer temperatures by 1.3 °C (± 0.8 SE), 75th percentile by 1.4 °C (± 0.9 SE), and summer maxima by 2.0 °C (± 1.2 SE). All these trends were significant, but more pronounced in winter ($r^2 = 0.36$) than summer ($r^2 = 0.10$) (Figures 3.7, 3.8). Temperature trends in the Itchen were affected by the NAO at rates similar to streams in NW Britain, but only weakly, with the NAO explaining far less variation than trends through time ($r^2 = 0.06$) (Figure 3.4).

A key consideration was how temperature in other southern rivers might track the Itchen, particularly given potential differences in the effects of groundwater damping in chalk streams rather than surface-fed streams. For the three to four years in which data were available, monthly mean temperatures in the chalk-fed Avon (Amesbury), and Bere (Dorset) correlated highly significantly with the Itchen, with monthly data form the latter explaining over 97 per cent of the variance in the other rivers (Figure 3.5; Table 2.1). Temperatures in the Bere and Avon were highly inter-correlated (Figure 3.10c). Long-term winter trends (1980-2006) in the Itchen were also related significantly to more distant streams at Llyn Brianne in upland Wales ($r^2 = 0.53$; Figure 3.6).

In all these cases of inter-comparison, the intercepts suggested that other rivers differed by up to 2.9°C on average temperature from the Itchen as would be expected from their contrasting locations. More importantly, the slopes of the regressions relating temperature in the Avon and Bere to the Itchen were both below one, indicating that temperature increase in the two chalk-fed sites was slower than in the lower Itchen. As a result, reconstructed increases in temperature through time are likely to have been moderately slower in the Bere and Avon than the Itchen, particularly in summer (Table 3.1). In all these cases, standard errors are large reflecting uncertainty in the temperature trend for the Itchen rather than in the inter-calibration between rivers.

	Winter temperature °C	**Summer temperature °C**
Itchen	2.7 ± 0.7	1.3 ± 0.8
Avon	2.4 ± 1.3	1.2 ± 1.4
Bere	1.8 ± 0.9	0.9 ± 1.1

Table 3.1 Real (Itchen) and modelled (Bere, Avon) increases in average winter and summer temperature (with SE) between 1980 and 2006 in three rivers in southern Britain.

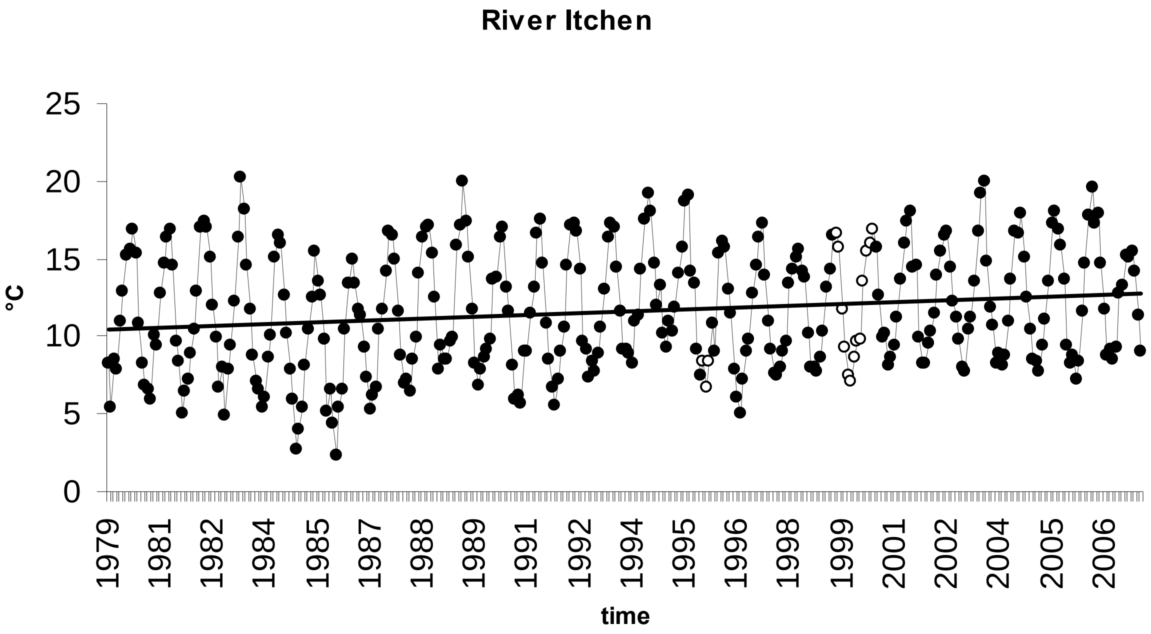


Figure 3.1 River Itchen average monthly water temperatures from 1978 to 2006 sampled at Southampton (SU 457 157) on a daily basis. The trend line is the line of best fit through all months. Open symbols correspond to reconstructed data.

Figure 3.2a

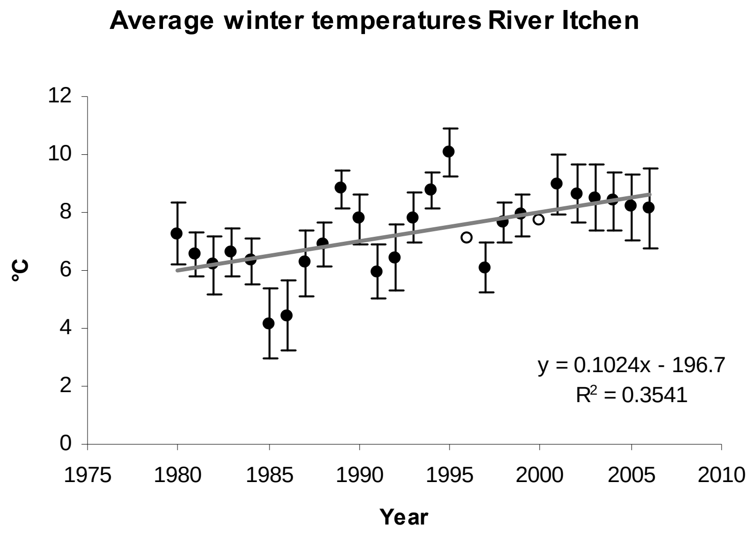


Figure 3.2b

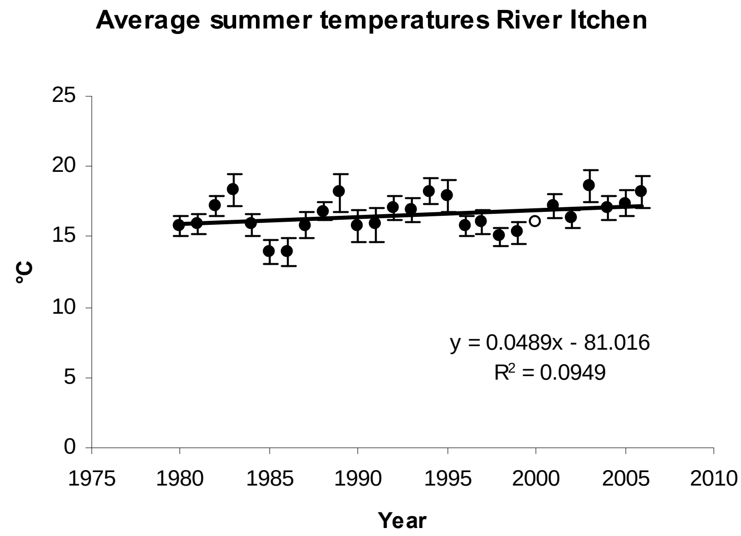


Figure 3.2 Average winter (a; December-February) and summer (b; June-August) temperatures in the lower River Itchen between 1980-2006. Error bars correspond to standard deviation in daily data. The trend lines were fitted by regression. Open symbols correspond to reconstructed data.

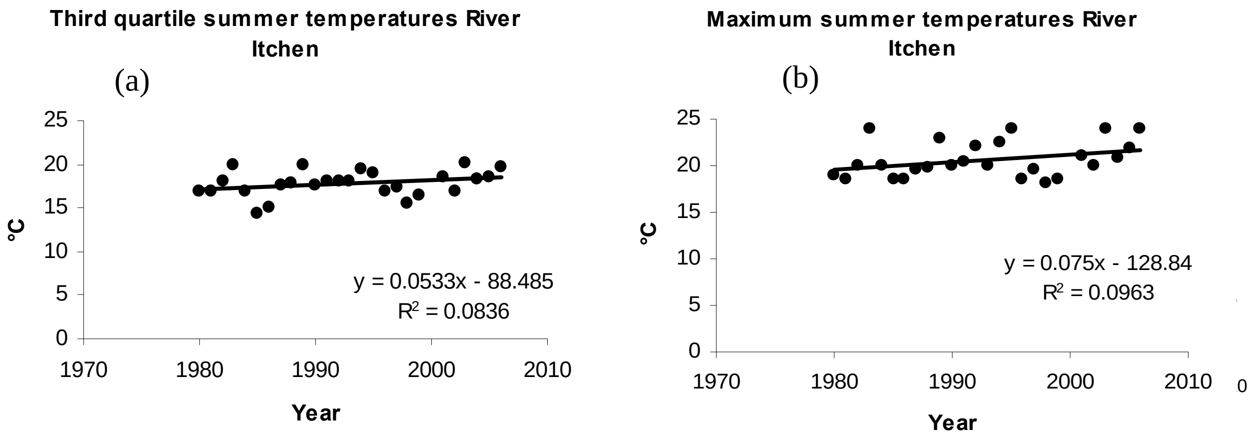


Figure 3.3 Trends in summer high temperatures lower River Itchen between 1980-2006 as revealed by 75th percentiles (a) and maxima (b) for each year. The trend lines were fitted by regression.

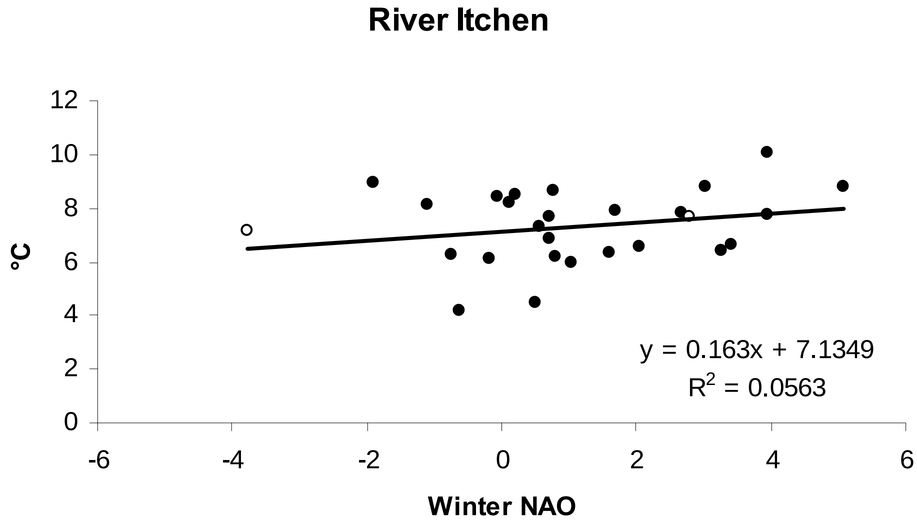


Figure 3.4 Trends in winter mean temperatures for the lower River Itchen between 1980-2006 and the winter index of the North Atlantic Oscillation. The trend lines were fitted by regression. Open symbols correspond to reconstructed data.

Figure 3.5a

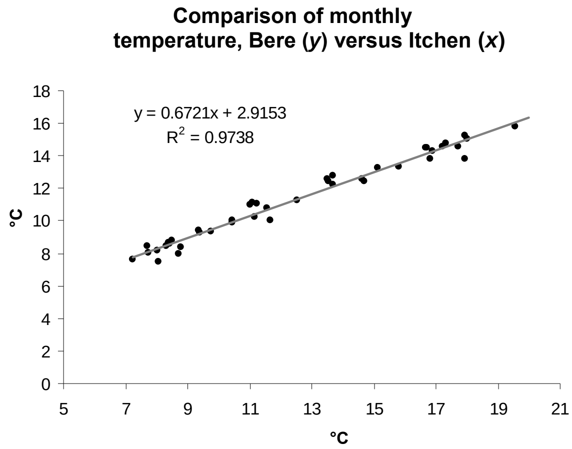


Figure 3.5b

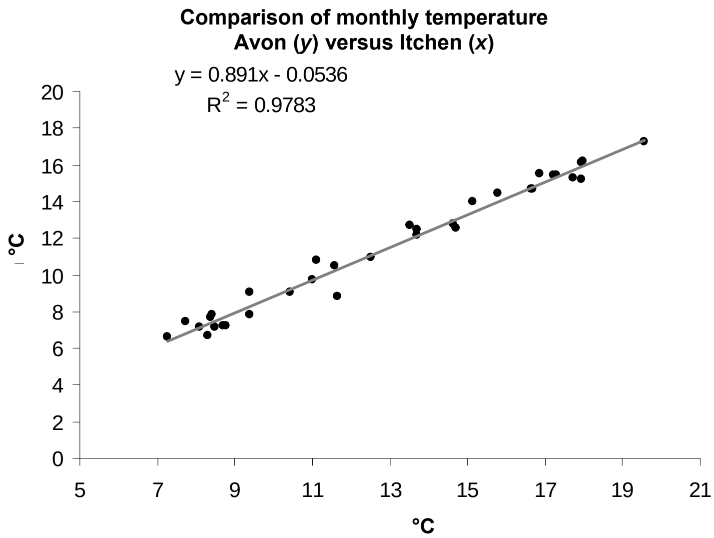


Figure 3.5c

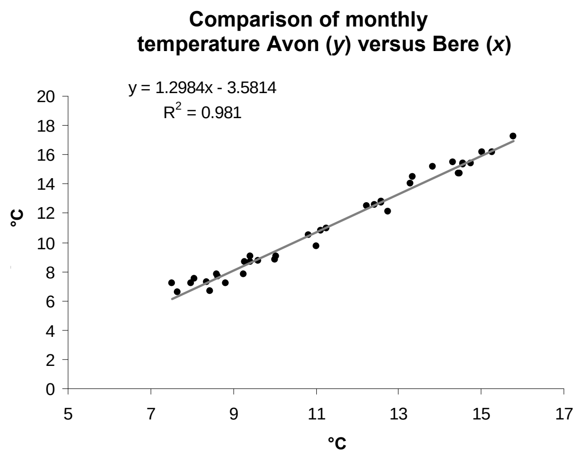


Figure 3.5 Relationships between average monthly temperatures in the Bere (a; Dorset area) and the Avon (b; Avon area) and the lower Itchen over the period November 2002-2006, and between the Avon and Bere (c).

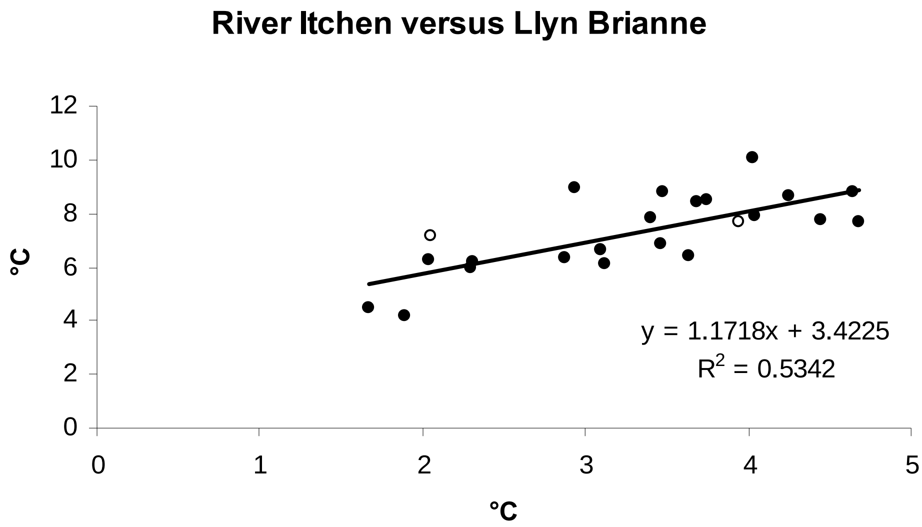


Figure 3.6 Relationships between average winter temperatures in the lower Itchen (*y*) and Llyn Brianne streams (Ll6; *x*) over the period 1982-2006. The line was fitted by regression. Open symbols correspond to reconstructed data.

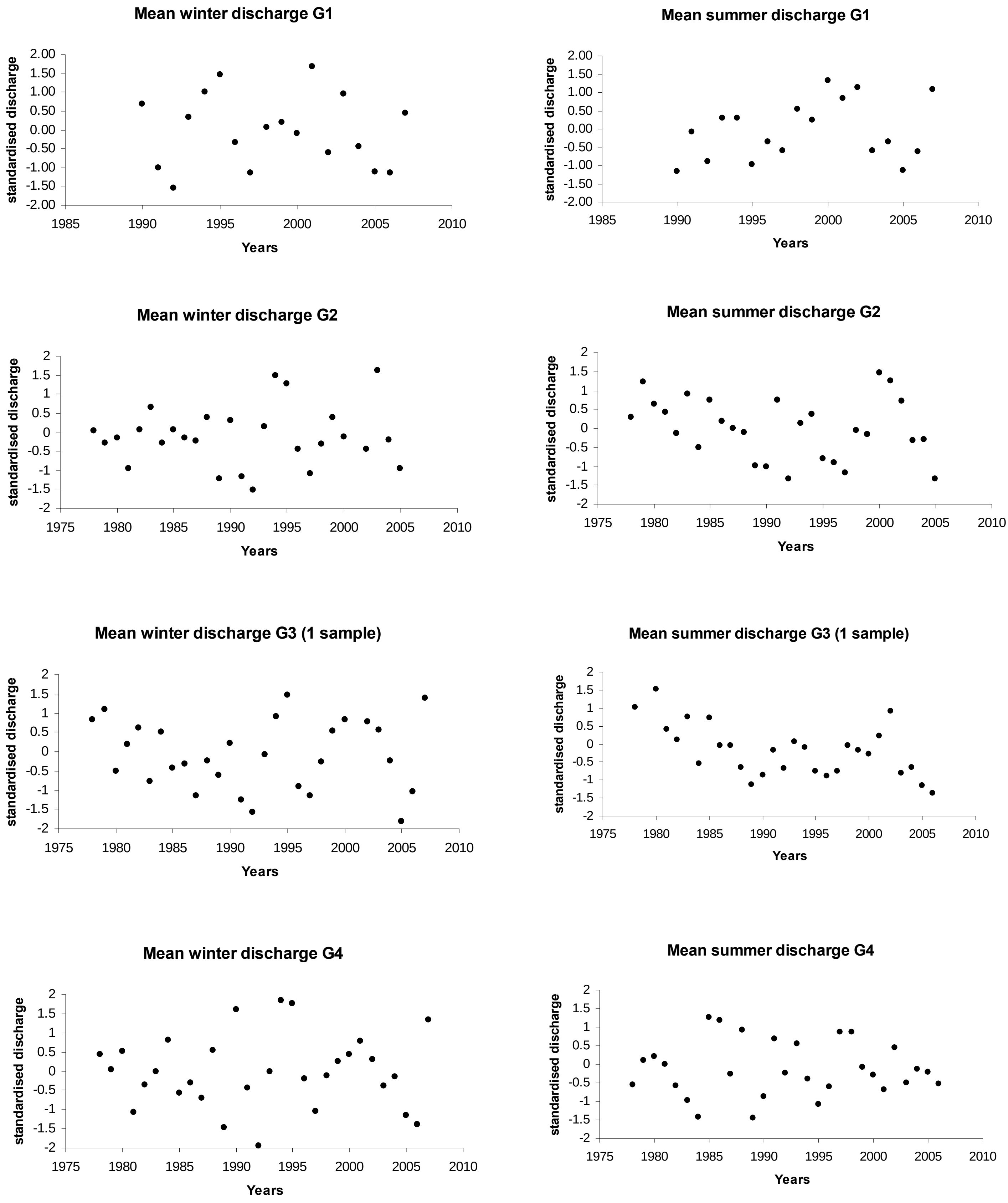


Figure 3.7 Variations in standardised discharge for sites in each of the four invertebrate groups.

Although variations between wet and dry years were clear in the standardised discharge data from all site groups, no statistically significant trends through time were apparent. Summer discharges at Group 3 sites showed some tendency towards lower flows through time.

3.2 Trends among macroinvertebrates

3.2.1 General patterns

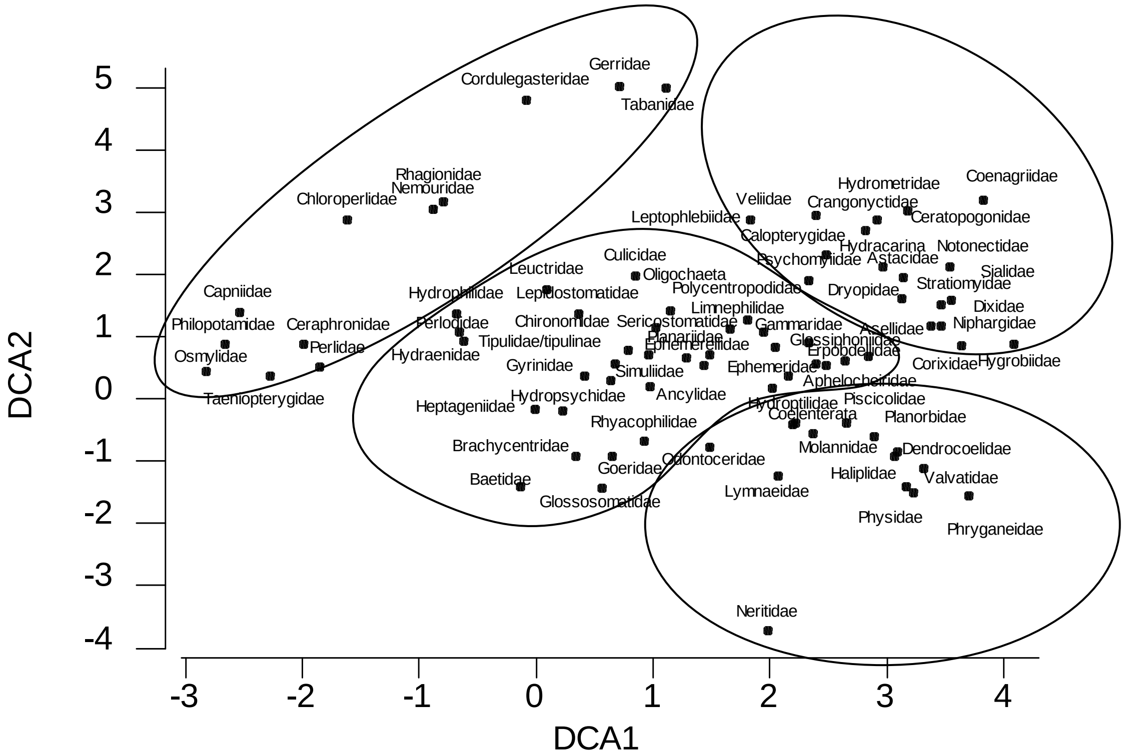
Detrended correspondence analysis revealed large variations across sites in invertebrate taxonomic composition, with a large proportion of explained variance (14 per cent) arising on Axis 1 and six per cent on Axis 2 (Figure 3.8a). Within site-year combinations, there were strong similarities in assemblages from samples in spring and autumn, although there was apparently greater seasonality in chalk streams ($r = 0.78$, $P < 0.001$, $n = 240$) than in surface-fed streams ($r = 0.93$, $P < 0.001$, $n = 210$). We therefore made no attempt at a separate analysis of trends through time on data separated by season.

The position of each family in ordination spaces suggested several influences. For example, families with low scores on DCA 1 had significantly greater LIFE scores ($r = 0.58$, $n = 68$, $P < 0.001$) and significantly greater BMWP scores ($r = -0.36$, $n = 70$, $P < 0.003$), suggesting preference for faster flows and greater oxygen levels. BMWP and LIFE scores are inter-correlated for the array of families represented in the data ($r = -0.62$, $n = 68$, $P < 0.001$).

In the 998 samples collected from 52 sample locations after 1989, there were large variations in invertebrate assemblages across the region. Classification using Ward's method on DCA axes scores 1-4 suggested four clear groups that were respectively chalk-fed rivers in Dorset and Avon, and the Bristol Avon (Group 1); chalk-fed streams in Hampshire (Group 2); surface-fed streams in Hampshire (Group 3) and surface-rivers in Devon and the Bristol Avon (Group 4) (Figure 3.8b). As well as differing in composition, the groups differed in average scores on some biological indices (Table 3.2). BMWP and family richness were particularly elevated at the chalk-fed sites in Groups 1 and 2, while life scores and ASPT tended to be greatest at the surface-fed sites in Group 4 (Table 3.2).

Table 3.2 Mean (with SD) values of four biotic indices for each site group post-1989.

	BMWP	ASPT	No of taxa	Family LIFE
Group 1	172.2 ± 10.2	5.76 ± 0.16	29.8 ± 1.7	7.18 ± 0.14
Group 2	187.9 ± 16.5	5.72 ± 0.16	32.7 ± 2.3	6.96 ± 0.14
Group 3	130.2 ± 21.6	5.80 ± 0.35	22.2 ± 2.8	6.97 ± 0.21
Group 4	147.2 ± 20.3	6.56 ± 0.12	22.3 ± 2.9	7.96 ± 0.24



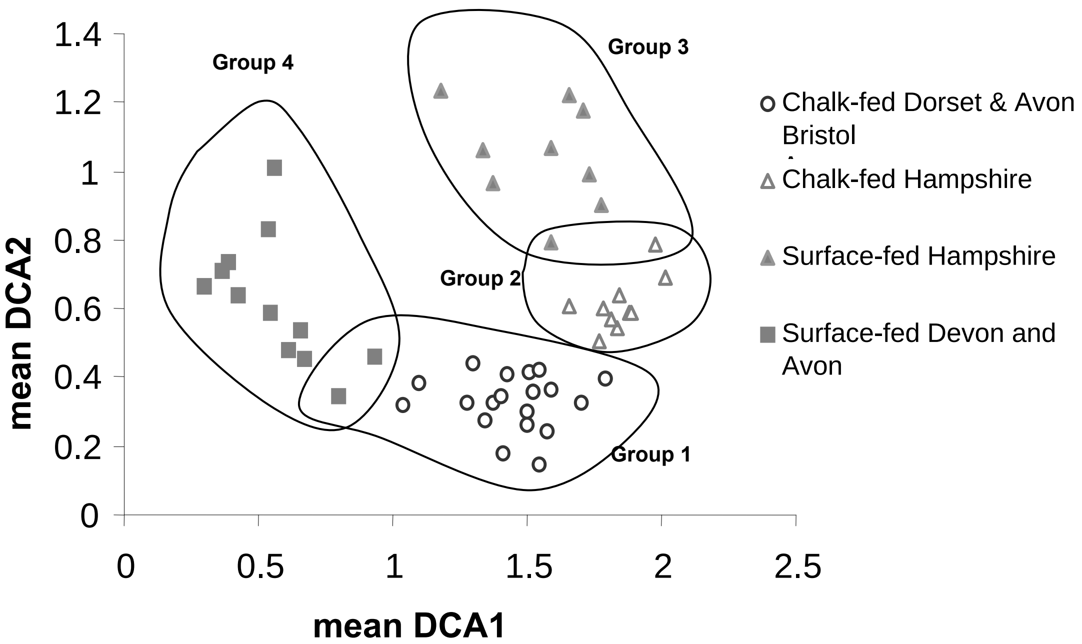


Figure 3.8 Ordination plot showing positions of each invertebrate family in all sites and samples (a) and mean positions of each of the 52 sites (b) during the period 1989-2006 on DCA Axes 1 and 2. Site groups detected using Ward's classification are illustrated.

3.2.3 Trends in assemblage composition within groups

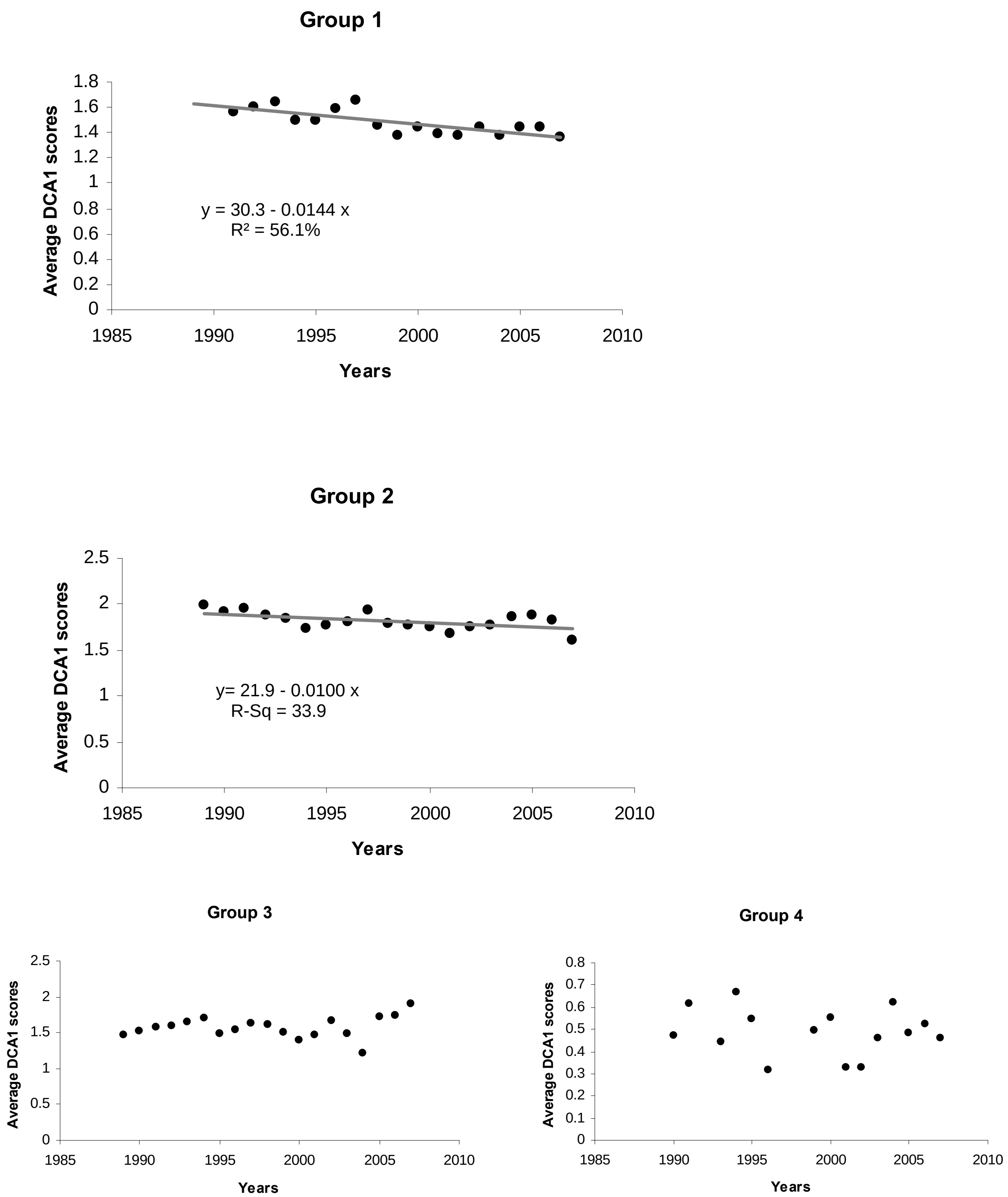


Figure 3.9 Trends in DCA scores through time for the four site groups classified on the basis of invertebrate composition (see Figure 3.8b). Each point represents the average, group-wide ordination score for all the sites sampled in each year.

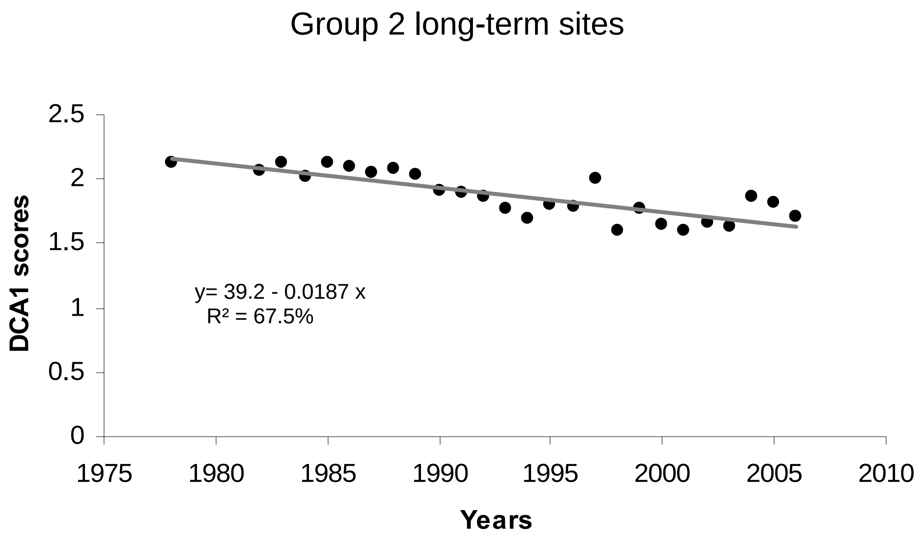


Figure 3.10 Trends in mean ordination scores of long-term sites from Group 2 after correction for change in sampling procedure in 1989 (see Section 2.2.3 and Figure 2.5). Sites were Wallington, the Test at Chibolton, the Test at Broadlands and the Dever.

Over the period 1989-2007, there were small but significant shifts in the average composition of invertebrate assemblages in each of chalk-fed Groups 1 and 2, but not Groups 3 and 4 (Figure 3.9). Based on the average magnitude of change in DCA scores, these changes represented a turnover in typical composition of around five per cent of families (Figure 3.9). Interestingly, after accounting for altered sampling methods from 1989 onwards, annual rates of change were near identical at the four long-term chalk-fed sites, where turnover in composition was around 14 per cent over the period 1977-2007 (Figure 3.10). In all these cases, DCA scores declined through time at Group 1 and 2 sites, implying a shift towards taxa typical of faster-flowing conditions, of more well-oxygenated conditions, or both.

3.2.3 Trends in family rank-abundances

Among the 26 most widespread invertebrate taxa (those occurring at more than half of sites), there were 35 instances of families changing significantly in average rank-abundances across the sites over the period 1989-2006 in any group in either spring or autumn (Table 3.3). This is around three times greater than would be expected by chance given that 208 tests were undertaken.

Fourteen families changed in average rank-abundance at either Group 1 or Group 2 sites, although in two of these cases trends conflicted between groups or seasons (Chironomidae and Elidae). Otherwise, taxa with at least one positive trend included Planaridae, Heptageniidae, Caendiae, Ephemeridae, Baetidae, Limnephilidae, Sericostomatidae and Lepidostomatidae, while Erpobdellidae, Simuliidae and Rhyacophilidae declined. The rhyacophilid trend was particular strong and consistent across groups.

The rank-abundances of twelve taxa had significant trends at Group 3 or Group 4 sites.

Table 3.3 Trends in mean rank-abundance through time of the most widespread families (occurring in more than half of samples) in each site group from 1989-2006.

* P < 0.05; ** P < 0.01; *** P < 0.001.

	Season	Occurrence (%)	Group1	Group 2	Group 3	Group 4
Chironomidae	*spring*	0.97		neg**	pos**	
	autumn	0.95	pos**			
Elmidae	*spring*	0.95	pos**			
	autumn	0.96	neg**		pos**	
Gammaridae	*spring*	0.86				
	autumn	0.87				pos*
Limnephilidae	*spring*	0.61				
	autumn	0.75		pos**		
Sericostomatidae	*spring*	0.81	pos*			
	autumn	0.84				
Simuliidae	*spring*	0.81		neg**		
	autumn	0.81				
Ephemerellidae	*spring*	0.79			pos**	
	autumn	0.56		pos*		
Leptoceridae	*spring*	0.76				
	autumn	0.75			pos*	pos**
Hydropsychidae	*spring*	0.75			neg**	
	autumn	0.79			pos**	
Lepidostomatidae	*spring*	0.73		pos**		
	autumn	0.57	pos***		pos*	
Planariidae	*spring*	0.72	pos*		neg*	
	autumn	0.72	pos**			
Tipulidae/Tipulinae	*spring*	0.72				
	autumn	0.69				
Heptageniidae	*spring*	0.72	pos*		neg*	
	autumn	0.59				pos*
Hydrobiidae	*spring*	0.71				
	autumn	0.69				
Sphaeriidae	*spring*	0.69				
	autumn	0.70				
Glossiphoniidae	*spring*	0.68				
	autumn	0.79				
Caenidae	*spring*	0.66				
	autumn	0.56		pos*	pos*	
Ancylidae	*spring*	0.64				
	autumn	0.76				
Asellidae	*spring*	0.63				
	autumn	0.65				
Erpobdellidae	*spring*	0.62	neg*			
	autumn	0.69	neg*			
Rhyacophilidae	*spring*	0.58	neg**		neg***	
	autumn	0.58			neg*	
Ephemeridae	*spring*	0.57	pos**			
	autumn	0.57				
Baetidae	*spring*	0.55	pos*			
	autumn	0.60				
Goeridae	*spring*	0.55				
	autumn	0.71				
Planorbidae	*spring*	0.54				
	autumn	0.62				

3.3 Potential response of macroinvertebrates to climate

3.3.1 Response of assemblages to temperature

Increasing river temperatures in winter (as indexed to the Itchen) explained 23-40 per cent of the inter-annual variance in average assemblage composition in Group 1 and 2 sites, as revealed by DCA scores, over the period 1989-2006 (Figure 3.11). Trends at the four longer-term sites in Group 2 were near identical over the period 1977-2006 after accounting for changes in sampling method, with winter temperature (indexed to the Itchen) now explaining 45 per cent of the inter-annual variance (Figure 3.12). In all these cases, however, changes most likely reflected a modest shift towards taxa typical of faster-flowing and well-oxygenated waters, meaning that confounding effects of other factors cannot be excluded (see below).

Long-term variations in composition at one Group 3 site (Beaulieu) also tracked temperature ($r^2 = 0.22$).

There were no systematic long-term trends in discharge, and no relationships with invertebrate assemblage composition had been explored at the time of this report, though they are reported elsewhere following incorporation into phase II of this project (see Durance and Ormerod submitted).

Figure 3.11a

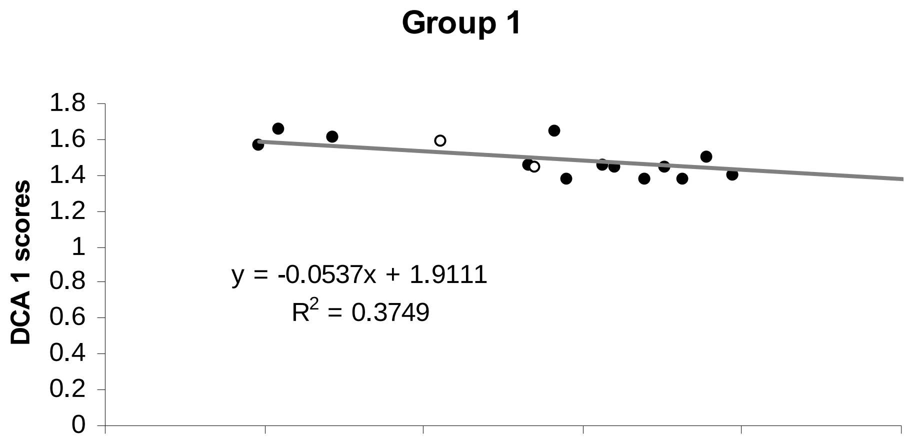


Fig 3.11b

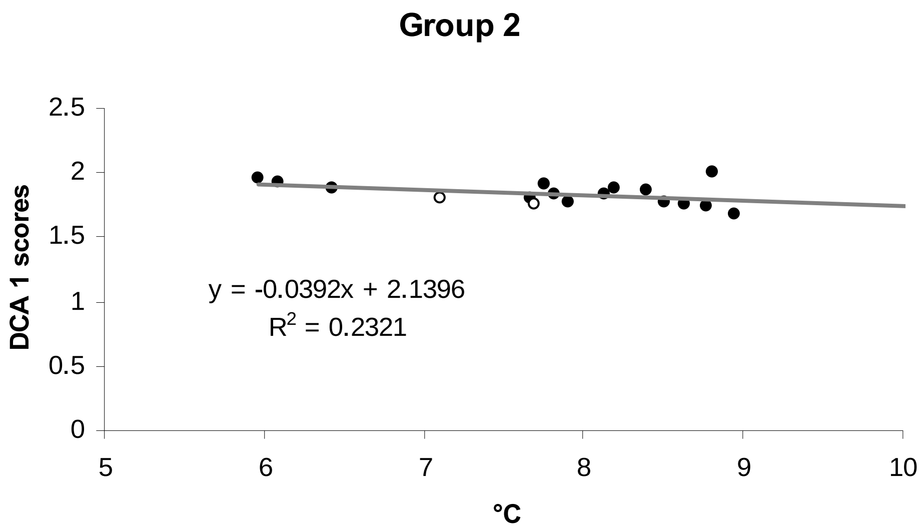


Figure 3.11 Trends in average DCA scores across Group 1 and 2 sites with increasing winter temperature (as indexed to the Itchen) over the period 1989-2006. Open symbols correspond to reconstructed data.

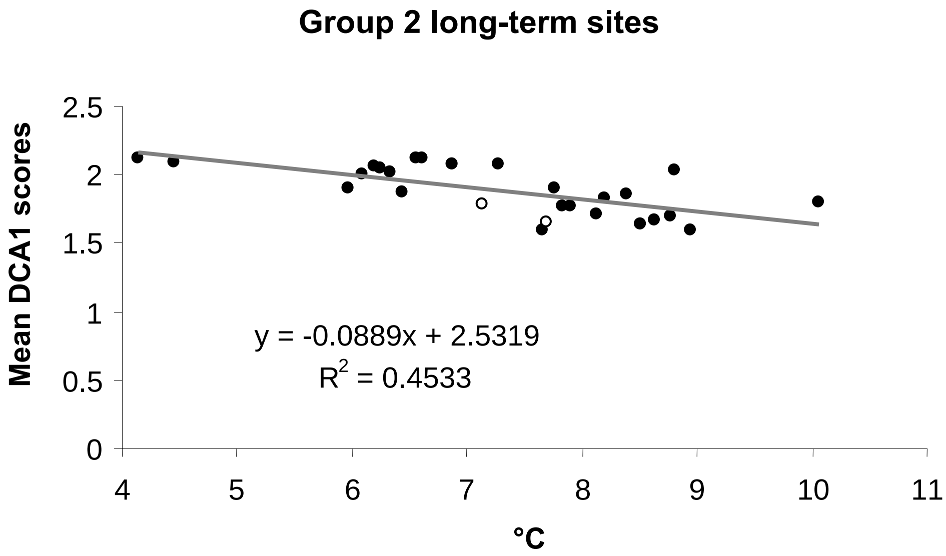


Figure 3.12 Trends in average DCA scores with increasing winter temperature (as indexed to the Itchen) over the period 1977-2006 for long-term sites in Group 2 (Wallington, Test at Chibolton, Test at Broadlands, Dever). Note that these data required a correction to account for a change in sampling method in 1989. Open symbols correspond to reconstructed temperature data.

3.3.2 Water quality trends and possible effects

Although sites were selected to be as free as possible from other stressors, past water quality problems at many of the sites were apparent. There were marked, group-wide trends in water quality at Group 1 and 2 sites from at least 1989 onwards, with average concentrations of ammonia, orthophosphate and BOD all declining highly significantly (r^2 = 0.25-0.86) (Figure 3.13). In contrast, total oxidised nitrogen increased (r^2 = 0.52-0.69). Similar trends were apparent at sites in Group 3 (Figure 3.14). Although trends in average concentrations were minimal across site groups, they reflected marked reductions at certain sites, as well as a drop in the frequency of occasional large values of BOD and ammonia.

Water quality trends could evidently confound the effects of temperature on organisms: trends in water quality were tracked by increases in BMWP, ASPT, taxon richness and family LIFE scores at sites in Groups 1 and 2 (r^2 = 0.35-0.70), with these patterns sufficient to explain declining DCA scores through time (Table 3.4).

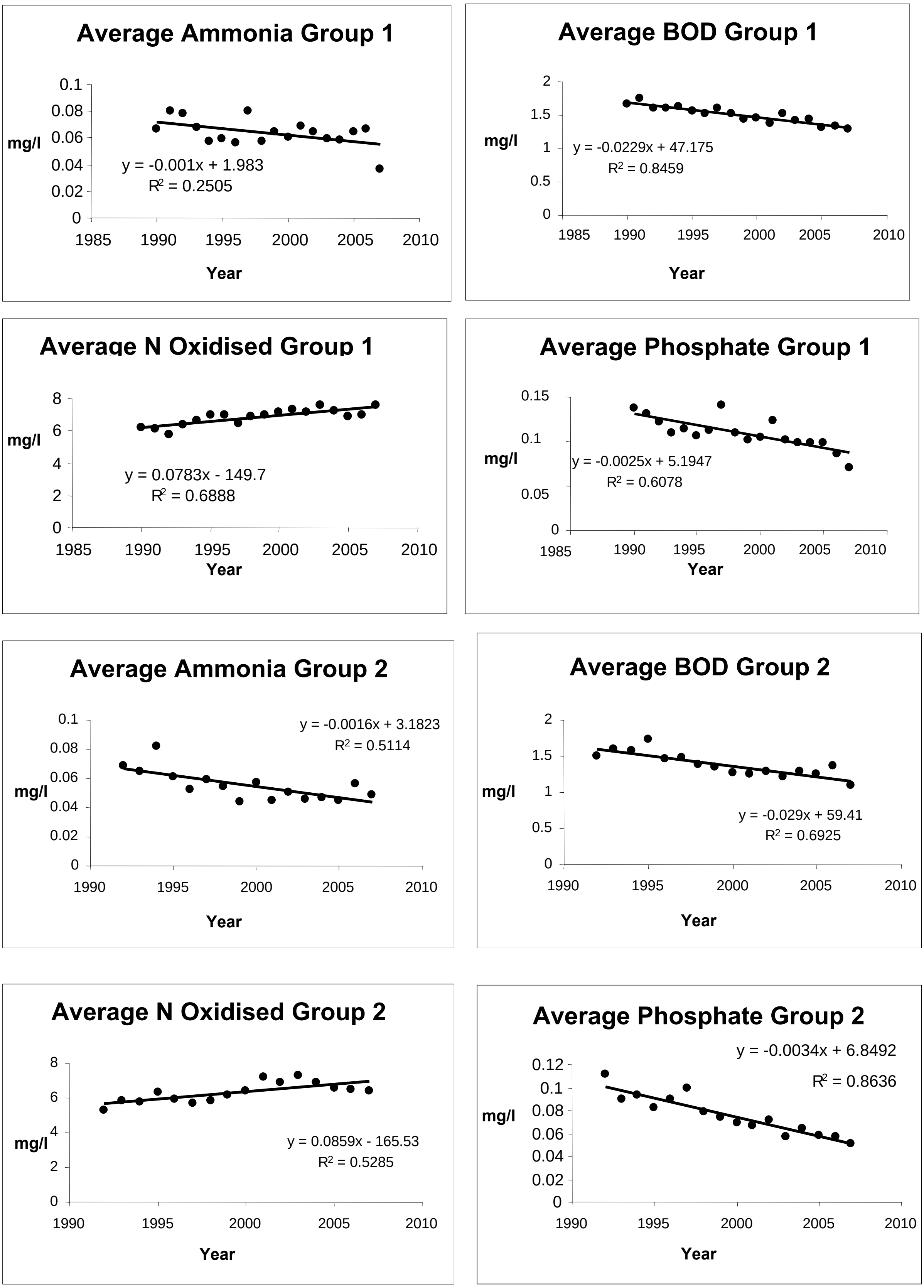


Figure 3.13 Trends in average water quality across sites in Groups 1 and 2 through time, with lines fitted by regression.

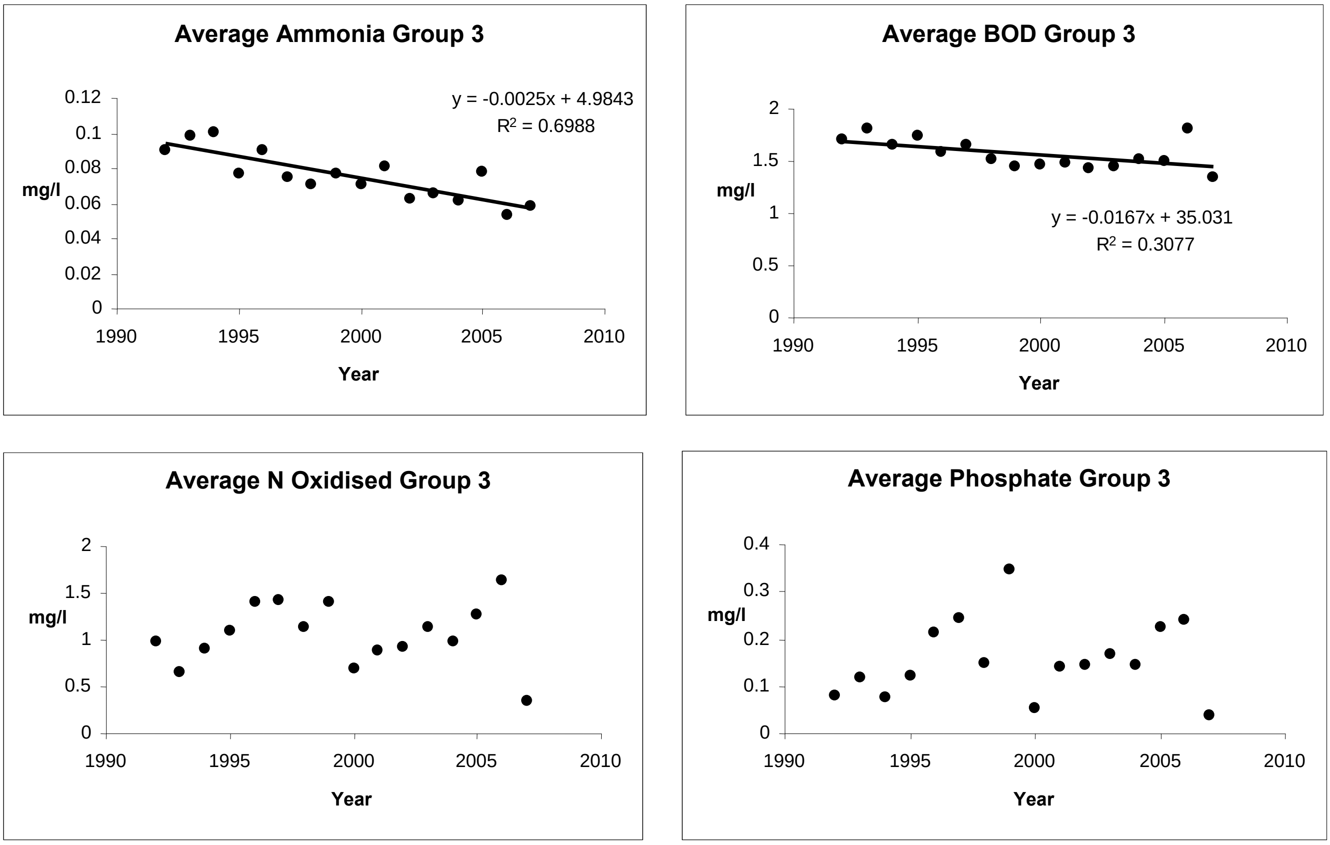


Figure 3.14 Trends in average water quality across sites in Group 3 through time, with lines fitted by regression.

Table 3.4 Regression parameters reflecting significant trends in average BMWP, ASPT, taxon richness and LIFE scores through time for each group. No significant trends were apparent for Group 3. * - significant at p<0.05; **- p<0.01***; p<0.001

		BMWP	ASPT	No of taxa	Family LIFE
Group 1:					
DF (1, 16)	R^2		67.80%		36.50%
	F		31.64***		8.61**
	Slope		0.02		0.01
Group 2					
DF (1, 18)	R^2	35.40%	70.30%	19.10%	52.30%
	F	9.3**	40.17***	4.01*	18.63***
	Slope	1.74	0.03	0.18	0.02
Group 4					
DF (1, 14)	R^2		26.00%		
	F		4.56*		
	Slope		0.01		

4 Discussion

This report aimed to test the three hypotheses that:

i) Unidirectional changes in hydrology and/or thermal regime are now in progress in southern English rivers.

ii) The character of any such changes varies between surface- and chalk-fed rivers.

iii) Trends in temperature or discharge in chalk- or surface-fed rivers have been sufficient over recent decades to cause measurable change among aquatic macroinvertebrates against background variability.

In the case of discharge regime, there was little support for the first hypothesis. Although discharge has varied between recent wet and dry years, there is little evidence of a consistent trend through time across the geographical area covered by this report; this accords with the prediction that changes in rainfall-runoff pattern due to climate change are not yet separable from background variation (Wilby 2006). However, inter-annual variations in discharge in southern English chalk streams have been large, and possible effects on invertebrates are explored further in phase II of this work. Considerable evidence on the effects of groundwater level and associated variations in discharge on chalk-stream ecology has accumulated in Britain over the last 30 years. Marked droughts have included those in 1976, 1988-92, 1995-96 and 2003-2005, although the lengths of dry period have varied, and in some years (such as 1995) winter recharge of groundwater levels has moderated effects on discharge. As a result, impacts on organisms are more pronounced in some years than others through mechanisms that might include variations in velocity, encroachment by marginal vegetation, variation in channel macrophyte cover (particularly *Ranunculus*) and the removal of lotic taxa during drought by increased sedimentation.

In contrast to discharge, unidirectional trends in temperature in southern rivers now seem clear. Although long-term temperature data are extremely scarce in southern Britain, available data from the lower Itchen show strong evidence of long-term increase. Trends are clearer in average winter than average summer values, but are also reflected in increasing maxima and 75th percentiles so that potential upper limits could be reached for some sensitive species. Increasing trends accord not only with more distant sites in upland Wales, as shown here, but also with other locations across Britain (such as Exmoor, Scotland) and Europe (such as Austria, the upper Rhone, Switzerland), implying they are part of a wider phenomenon (Durance and Ormerod 2007). As in Wales, temperature increase in the Itchen was apparent in spite of inter-annual variations in other large-scale climatic phenomena such as the NAO. Indeed, in the Itchen, NAO effects on temperature were small compared with trends through time, with the slope and strength of the NAO effect here (b = 0.163x; r^2 = 0.06; n = 26 years) less than that detected in Wales (b = 0.20-0.24; r^2 = 0.35; n = 25 years; Plynlimon; b = 0.32-0.34x; r^2 = 0.51-0.57) or Cumbria (Black Brows Beck b = 0.285 x; n = 29 years; r^2 = 0.42) (Elliott *et al*. 2000; Briers *et al*. 2004; Durance and Ormerod 2007).

Although river temperatures can vary over small spatial-temporal scales for a range of reasons, over longer average periods they track air temperature closely providing there are no large effects from evaporative cooling, warm-water additions or groundwater damping (Caissie 2006). Chalk-fed streams are an interesting case where groundwater effects can be large. However, extremely strong relationships between temperatures in the Itchen, Avon at Amesbury and Bere detected here over a four-year period show how all these rivers reflect wider regional variations that are ultimately driven by air temperature. Importantly, however, the slopes relating the Avon and Bere to the Itchen are below one, illustrating slower warming in the two small chalk streams with increasing temperature. Assuming that these relationships between rivers can be extrapolated over longer time periods, temperature increases in the Bere and Avon appear to have been smaller in both summer and winter than in the Itchen (Table 3.1). This supports Hypothesis 2. However, the scale of temperature increase in these smaller streams is in keeping with winter increases at Llyn Brianne (around 1.7 °C) and on Exmoor, and might reflect their headwater character as well as their groundwater-fed nature.

With respect to Hypothesis 3, there is some limited evidence in these data of a relationship between temperature and shifts through time in the assemblage composition of chalk-fed sites in Groups 1 and 2. Compositional change appeared to reach around five to 15 per cent over a 20-30 year period, and correlated significantly with temperature increase. On this basis, Hypothesis 3 cannot yet be rejected entirely, and some declining families, such as the Rhyacophilidae, are cold-water stenotherms whose distribution could be limited by temperature. Moreover, effects or changes among individual temperature-sensitive species could conceivably be masked by the family-level data available. This is a real risk with the Environment Agency's current family-level approach to invertebrate monitoring, where different species within families can have markedly different thermal ranges (such as the Hydropsychidae).

At the same time, support for Hypothesis 3 is limited because significant trends in water quality have occurred at sites in Groups 1 and 2 over the same period as temperature has apparently increased. In particular, indicators of episodic or organic pollution have declined (BOD, ammonia, orthophosphate) while total oxidised nitrogen has increased. The latter could reflect eutrophication, but is also consistent with reduced phosphorus flux and hence decreased plant use of available nitrogen. Some lines of reasoning suggest that water quality may well have had stronger effects on macroinvertebrates than temperature because:

i) More invertebrates have increased in abundance at Group 1 and 2 sites than decreased, often high BMWP-scoring taxa that might be expected to increase at sites recovering from pollution incidents.

ii) BMWP, ASPT, and LIFE scores increased at Group 1 and 2 sites. A rise in temperature might be expected to trigger reductions in these indices, given increased metabolic costs and reduced oxygen concentrations at higher temperatures. The increase in LIFE scores is interesting because of possible links with flow pattern, but LIFE scores for invertebrate families are also strongly related to oxygen tolerance and BOD.

iii) Temperatures have increased more in the winter than in summer, which might be expected to lead to changes in winter biological phenomena such as growth, development and phenology rather than loss or species exclusion.

Aside from headwaters which continue to be affected by acidification or other diffuse pollutants such as nitrogen, water quality has recovered widely in British rivers over the last 15-20 years thanks to new regulation such as the EC Freshwater Fish Directive (78/659/EEC) and the Urban Wastewater Treatment Directive (91/271/EEC). These directives have enforced standards in water courses while controlling polluting discharges, for example from sewage treatment works and some farms. Several of the streams and rivers considered here are designated sensitive waters under the second directive (such as Test, Itchen, Wylye, Hampshire and Bristol Avon), and will have been affected directly. Privatisation of the water industry in 1989 also improved quality in a range of wastewater treatment facilities. The Environment Agency's own data shows that the number of pollution incidents from sewage treatment and farm works has declined in southern Britain, and the length of river classed as unpolluted has increased (Figures 4.1 and 4.2). While the changes in water quality in mildly polluted chalk-streams have been relatively small, even trends of this magnitude can be sufficient to affect invertebrate assemblage composition. More interesting in the context of this study is that the resulting effects on invertebrates, particularly at Group 1 sites, were apparently stronger than those due to temperature. This concords with results from the Rhone system, where long-term temperature effects only became apparent in some rivers after water quality problems declined (Daufresne *et al.* 2007). In

Wales, invertebrate assemblages responded to temperature change only in clean, unpolluted streams unaffected by acidification (Durance and Ormerod 2007).

The suggestion that recent water quality trends might have had larger effects on invertebrates, at least at some sites, than temperature or discharge has important ramifications for management in that water quality enhancement can negate some potentially adverse climatic effects. For example, there are potential benefits in avoiding even small increases in organic loading and de-oxygenation in rivers at risk of rising temperature. There are clear implications for the adaptive management of river systems, for example in implementing the European Water Framework Directive (2000/60/EC), or equivalent regulatory instruments, with full recognition of the climate change context (Wilby *et al.* 2006). Similar regulatory requirements are being enforced elsewhere. A converse perspective is that the successful management of water quality problems over the past 15-20 years might now lead to climatic effects becoming increasingly important as a management challenge. Further work in Phase II of this project should focus on sites where confounding effects due to water quality can be excluded.

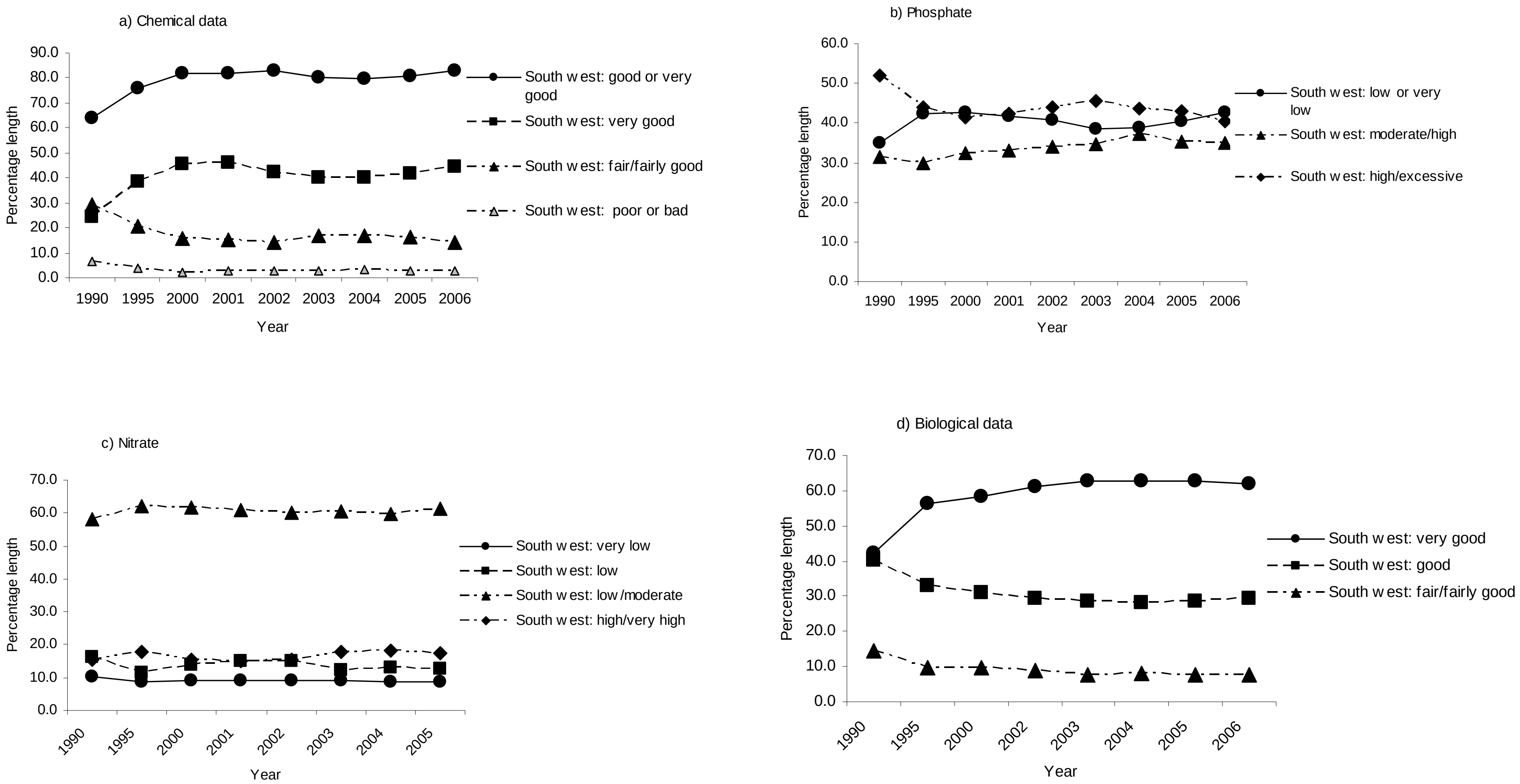


Figure 4.1 General Quality Assessment (GQA) trends in quality categories for overall chemical quality (a), phosphate (b), nitrate (c) and biological quality (d), Environment Agency, South West region (on 6,000 km classified)

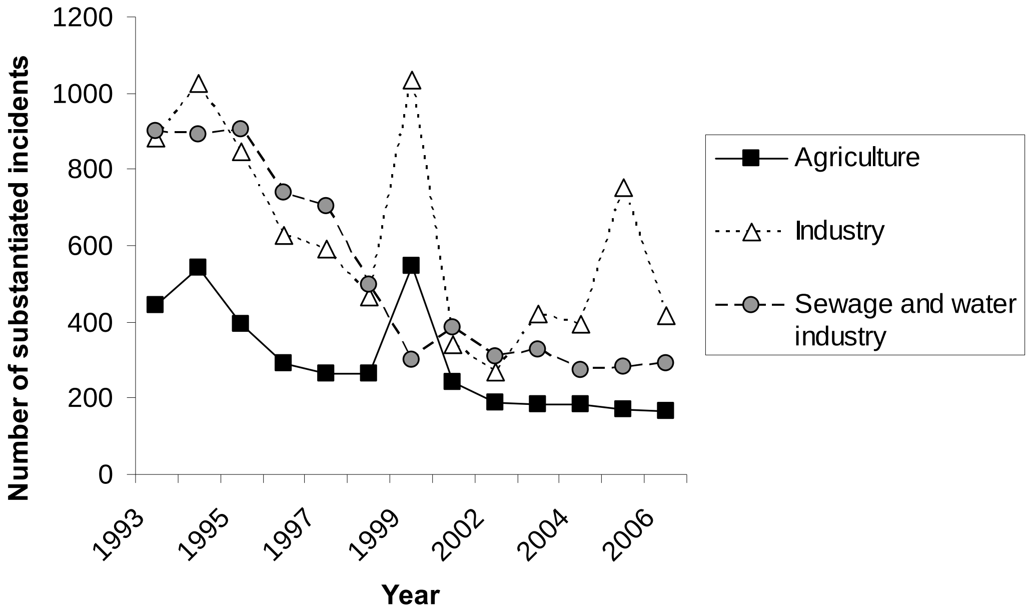


Figure 4.2 Recent trends in annual frequency of substantiated pollution incidents (Categories 1-3) in Environment Agency South West region from three major sources (taken from Environment Agency)

5 Recommendations

Following the work on this project, we make the following recommendations

Further investigation should be carried out on these data using multiple regression, factorial analysis and temperature modelling to separate the effects of water quality, discharge variation and climate on biological trends through time – including community composition, biotic indices and family-level abundances. This should be undertaken during the second project phase at sites where there are no water quality trends.

The Environment Agency should consider whether its current strategy for biological sampling is adequate to detect and appraise climate change effects, particularly with respect to taxonomy and sample-site distribution. Where possible, long-term data sets with species-level data should be appraised at both species and family level to establish whether specific identification is required to detect climatic effects.

The Environment Agency should consider further the organisms and river types at risk of climate change, and any consequences for river ecological function and conservation.

The Environment Agency should bolster its current programme for monitoring river temperature.

The Environment Agency and water companies should consider how best to appraise and manage interactions between water quality, abstraction catchment land use and climate change risks. The evidence from this study is that positive management to diminish other pressures might offset some climate change impacts.

References

Beche LA, McElravy EP, Resh VH (2006) Long-term seasonal variation in the biological traits of benthic-macroinvertebrates in two Mediterranean-climate streams in California, USA. *Freshwater Biology,* 51, 56-75

Beche LA and Resh VH (2007) Short-term climatic trends affect the temporal variability of macroinvertebrates in California 'Mediterranean' streams. *Freshwater Biology*, 52, 2317-2339.

Blenckner T and Hillebrand H (2002) North Atlantic Oscillation signatures in aquatic and terrestrial ecosystems - a meta-analysis. *Global Change Biology,* 8, 203-212.

Bonada N, Doledec S and Statzner B (2007) Taxonomic and biological trait differences of stream macroinvertebrate communities between Mediterranean and temperate regions: implications for future climatic scenarios. *Global Change Biology*, 13 (8), 1658-1671.

Bradley DC and Ormerod SJ (2001) Community persistence among upland stream invertebrates tracks the North Atlantic Oscillation. *Journal of Animal Ecology,* 70, 987-996.

Briers RA, Gee JHR, Geoghegan R (2004) Effects of the North Atlantic Oscillation on growth and phenology of stream insects. *Ecography,* 27, 811-817.

Caissie D (2006) The thermal regime of rivers: a review. *Freshwater Biology,* 51, 1389-1406.

Daufresne M, Roger MC, Capra H, Lamouroux N (2004) Long-term changes within the invertebrate and fish communities of the Upper Rhone River: effects of climatic factors. *Global Change Biology,* 10, 124-140.

Daufresne M, Bady P and Fruget JF (2007) Impacts of global changes and extreme hydroclimatic events on macroinvertebrate community structures in the French Rhone River. *Oecologia*, 151 (3), 544-559.

Dewson ZS, James ABW and Death RG (2007) Invertebrate community responses to experimentally reduced discharge in small streams of different water quality. *Journal of the North American Benthological Society*, 26 (4), 754-766.

Durance I and Ormerod SJ (2007) Climate change effects on upland stream invertebrates over a 25-year period. *Global Change Biology*, 13, 942-957.

Durance I and Ormerod SJ (In review) Trends in water quality and discharge offset long-term warming effects on river macroinvertebrates.

Elliott JM, Hurley MA, Maberly SC (2000) The emergence period of sea trout fry in a Lake District stream correlates with the North Atlantic Oscillation. *Journal of Fish Biology,* 56, 208-210.

Evans CD (2005) Modelling the effects of climate change on an acidic upland stream. *Biogeochemistry*, 74, 21-46.

Hallett TB, Coulson T, Pilkington JG, Clutton-Brock TH, Pemberton JM, Grenfell BT (2004) Why large-scale climate indices seem to predict ecological processes better than local weather. *Nature,* 430, 71-75.

Hogg ID, Williams DD (1996) Response of stream invertebrates to a global-warming thermal regime: An ecosystem-level manipulation. *Ecology,* 77, 395-407.

Hurrell JW (1995) Decadal trends in the North Atlantic Oscillation: Regional temperatures and precipitation. *Science,* 269, 676-679.

Hurrell JW, Kushnir Y, Visbeck M, Ottersen G (2003) *The North Atlantic Oscillation: Climate Significance and Environmental Impact*. The American Geophysical Union, Washington. 279 pp.

Jackson JK, Fureder L (2006) Long-term studies of freshwater macroinvertebrates: a review of the frequency, duration and ecological significance. *Freshwater Biology,* 51, 591-603.

Lake PS (2000) Disturbance, patchiness, and diversity in streams. *Journal of the North American Benthological Society,* 19, 573-592.

Lytle DA, Poff NL (2004) Adaptation to natural flow regimes. *Trends in Ecology and Evolution,* 19, 94-100.

Osborn TJ (2004) Simulating the winter North Atlantic Oscillation: the roles of internal variability and greenhouse gas forcing. *Climate Dynamics*, 22, 605-623.

Richardson JS (1992) Coarse particulate detritus dynamics in small montane streams of the southwestern British Columbia. *Canadian Journal of Fisheries and Aquatic Sciences*, 49, 337-346.

Root TL, Price JT, Hall KR, Schneider SH, Rosenzweig C, Pounds JA (2003) Fingerprints of global warming on wild animals and plants. *Nature,* 421, 57-60.

Shindell DT (2006) Decadal-scale modulation of the NAO/AO by external forcing: Current state of understanding. *Nuovo Cimento della Societa Italiana di Fisica C-Geophysics and Space Physics,* 29, 137-145.

Straile D, Adrian R (2000) The North Atlantic Oscillation and plankton dynamics in two European lakes - two variations on a general theme. *Global Change Biology,* 6, 663-670.

Van Der Maarel, E (1969) On the use of ordination models in phytosociology. *Vegetatio,* 19**,** 21-46.

Wilby RL (2006) When and where might climate change be detectable in UK river flows? *Geophysical Research Letters*, 33, Article Number L19407

Wilby RL, Orr HG, Hedger M, Forrow D and Blackmore M (2006) Risks posed by climate change to the delivery of Water Framework Directive objectives in the UK. *Environment International*, 32, 1043-1055.

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