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Recent trends among aquatic biota in the catchment of the River Wye (Wales) and the effects of riparian management

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Summary

Rod catches of Atlantic salmon (*Salmo salar*) from the River Wye were previously the greatest in England and Wales. However, a 30-year decline in catches of salmon and brown trout (*S. trutta*) prompted management action. Since 1996, the Wye and Usk Foundation have excluded livestock, managed riparian trees, protected banks, cleared migratory barriers and limed selected tributaries. The aim was to enhance salmon habitat and extend spawning opportunities. The outcomes of such activities in Britain are still poorly understood.

This thesis i) identified variations in the water quality, aquatic invertebrates and salmonids of the Wye catchment; ii) evaluated the impact of recent management on habitats and aquatic organisms; iii) assessed whether any larger-scale factors could explain management effects. Because no suitable project-specific data were collected, routine monitoring data and surveys were applied in the most applicable *post-hoc* experimental designs.

Ecological quality varied widely among the Wye's tributaries. Combined biotic indices supported the need to mitigate acidification in some upland streams and reduce diffuse nutrients in the lower catchment.

Riparian management appeared to reduce bank poaching and increase algae by comparison with reference streams. Post-treatment invertebrate communities were richer in recently managed streams than in controls. However, there was no evidence that management reversed the decline in salmonid populations.

The typical life-cycle of salmonids in the Wye might delay response to management, but this effect cannot be evaluated with only six years' post-treatment data. Alternatively, local effects could be masked by larger-scale trends. In particular, salmonid abundance in the Wye declines significantly with increasing summer temperatures, decreasing summer rainfall and discharge.

I conclude that riparian management has had some of the desired outcomes at the reach or tributary scale. However, salmonid numbers in the Wye potentially reflect climatic effects, implying a need to consider climate-change in future management action.

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Chapter 1

General Introduction:
Recent trends among aquatic biota in the catchment of the River Wye (Wales) and the effects of riparian management

[1.1] Background to the project

Despite their ecological importance, the fragmentation and loss of riparian corridors is continuing on a global scale (Naiman *et al.,* 1993; Kentula 1997; Pusey and Arthington 2003). Agricultural practices, such as intensification, are amongst the major causes of riparian habitat decline (Kentula 1997; Hendry *et al.,* 2003). Recognition of such changes has generated management activities to maintain this diverse ecotonal habitat and to restore damaged riparian zones (Malanson 1993; Naiman *et al.,* 1993; White 1996; Kentula 1997; Naiman and Décamps 1997; Hendry *et al.,* 2003). Management of riparian habitat is a step towards a whole ecosystem approach to maintaining river habitat and diversity (Imhof *et al.,* 1996; Kauffman *et al.,* 1997). While much of this work was previously undertaken in Northern America, there is increasing emphasis on riparian management also along British rivers. Here, however, scientific evaluations of ecological outcomes are still scarce.

Riparian management has frequently targeted individual key species on the assumption that their protection will benefit characteristics of the wider ecosystem (Landers 1997; Roper *et al.,* 1997; Mather *et al.,* 1998; Muotka *et al.,* 2002; Hendry *et al.,* 2003). For example, salmonids are economically important and charismatic species that are sensitive to environmental change, and are subjects of a large body of literature (Mather *et al.,* 1998). Consequently, much riparian management is undertaken to promote in-stream habitat that is preferable to salmonids and intended to increase their populations (Krog and Hermansen 1985; Hemphill and Bramley 1989; Salmon Advisory Committee 1991; O'Grady 1993; Imhof *et al.,* 1996; Kauffman *et al.,* 1997; Hendry *et al.,* 2003; Pusey and Arthington 2003). Studies assessing the consequences for other organisms, and for overall habitat structure, are scarcer (e.g. Liljaniemi *et al.*, 2002; Kiffney *et al.*, 2003; Wright *et al.*, 2003; Lepori *et al.*, 2005). This gap in understanding increasingly needs to be filled as European Directives and UK legislation emphasise the importance of freshwaters for a range of organisms in addition to fishes (e.g. Habitats Directive 92/43/EEC).

The management of riparian habitat to promote salmonids and associated evaluations have often been reach-based. However, factors that affect the riparian and aquatic environment, such as livestock overgrazing or other changes in catchment land-use, often impact rivers at large scales (Hendry *et al.,* 2003). Long-term, catchment-wide experiments are therefore potentially the most appropriate means to assess the efficacy of riparian management but their scale also creates substantial challenge (Imhof *et al.*, 1996; White, 1996; Kauffman *et al.*, 1997; Kershner, 1997; Naiman and Décamps, 1997; Downes *et al.*, 2002; Milner *et al.*, 2003). For example, to effectively isolate the effects of management from confounding factors, pre-treatment data would ideally be collected at managed and control sites and compared against data recorded after management (Stewart-Oaten *et al.*, 1986; Kondolf 1995; Downes *et al.*, 2002) using 'Before-After-Control-Intervention' experiments (Stewart-Oaten *et al.*, 1986; Downes *et al.*, 2002). Although BACI designs are often advocated in the evaluation of in-stream and riparian management, they have rarely been applied to such studies (Minns *et al.*, 1996; Sarr 2002; Pretty *et al.*, 2003). In part, this reflects the spatial extent and duration over which landscape-scale effects on rivers develop (Manel *et al.* 2000). Yet, despite the challenge, reliable evaluations of riparian and catchment-scale effects on rivers will be increasingly required in Europe as the Water Framework Directive begins to promote large-scale 'programmes-of-measures' intended to deliver 'good ecological status' over most water bodies (2000/60/EC).

This thesis explores some of these generic and international themes in a specific case-study in the catchment of the Welsh river Wye.

[1.2] The project.

Of all English and Welsh rivers, the Wye has long been the most noted for its rod catch of Atlantic salmon *Salmo salar,* and in particular its population of spring-run fish characterised by multiple winters at sea (Environment Agency, 2003b). Although catches once contributed over 25% of all declared catches from England and Wales, a major decline over the last 30 years has caused widespread concern for conservation agencies, the Environment Agency and for those concerned with the Wye's salmon fishery (Environment Agency, 2003a, b).

While government bodies (Countryside Council for Wales and the Environment Agency) have remained centrally involved with the assessment and management of salmon in the Wye, some of the major practical steps have been undertaken by a rivers' trust, the Wye and Usk Foundation (WUF). Aiming to restore wild populations of salmon and trout for conservation and economic benefit, the WUF have, since 1996, undertaken extensive management work on tributaries considered as traditionally 'good' salmon rivers (Slater 1988; Luxton, 2002; Wye and Usk Foundation, 2006a): the middle and upper Wye tributaries, initially under the Wye Habitat Improvement Project (WHIP) and later as part of the Powys Habitat Improvement Scheme (pHISH) (Wye and Usk Foundation (2006a, b, c, d). In addition to the use of calcium carbonate to buffer acidification (Lewis, 2006), management has involved a range of livestock exclusion measures, riparian tree-management, bank protection, spawning gravel restoration, the clearance of barriers to fish movement and the installation of fish passes (Wye and Usk Foundation 2006b; see Chapter 2). The overall aim has been to benefit salmonid populations, enhance habitat character, open more of the river catchment to fish spawning and locally increase the production of prey organisms.

In the early stages of the Wye Habitat Improvement Project (WHIP), the Clywedog Brook was used as an experimental river on which riparian management techniques were compared between reaches. However, this pseudoreplicated design limited conclusions about treatment effects on stretches within the same river (Hurlbert, 1984; Luxton, 2002). Moreover, by 2004, the WUF had expanded riparian management activity to cover 13 different tributaries (stream orders 2-4), with barriers removed from an additional 29 (Table 1.1). This meant that a catchment-scale assessment of the effects of recent management, with whole tributaries acting as true replicate units, might be appropriate to examine management effects on stream habitats and organisms. However, there were also major constraints in that treatments were applied at different times, treatments were not randomised across rivers, and no organised data on habitats or river organisms were collected before riparian management occurred (Stewart-Oaten *et al.,* 1986; Kondolf and Larson 1995; Manel *et al.*, 2000; Downes *et al.,* 2002). The only options for evaluating treatment effects were therefore:

i) To use surveys following treatments to compare conditions among tributaries grouped by treatment type. Although the lack of time-series data in this approach means that effects could not be ascribed unequivocally to treatment, this was the only pragmatic possibility where no pre-treatment data existed. This is a common problem with large-scale and long-term effects on ecosystems (Manel *et al.* 2000).

ii) To attempt to evaluate treatment effects using data available from other sources, for example long-term monitoring data from the Environment Agency. This approach clearly required that monitoring data be available, and that the frequency and extent of collection from treatment and control rivers should allow some form of organised analysis. While experimental design in such cases is seldom ideal, the standardised approach to data collection is an advantage and a range of qusasi-experimental methods are applicable (e.g. 'Before-after-control-intervention' type designs; Stewart-Oaten *et al.,* 1986; Downes *et al.,* 2002).

Both of these approaches would benefit from an understanding of general trends in the ecology of the Wye that might affect interpretation. Thus, in addition to evaluating differences in ecological conditions among sites with and without recent management, some of the following chapters also assess wider spatial and temporal patterns in ecological conditions in the Wye.

[1.3] Aims and thesis structure

This thesis aimed to identify any spatio-temporal trends in the water quality, aquatic invertebrates and salmonids of the River Wye catchment. Second, and against this background, it aimed to evaluate the impact on riparian habitats, aquatic invertebrates and salmonids within the Wye system of recent riparian management intended to promote salmonid populations. Specific research questions addressed in each chapter were:

i) From current literature, what is the current understanding of riparian influences on the aquatic environment? (Chapter 2)

ii) What are the current ecological conditions across the Wye catchment, and how might these inform management priority? (Chapter 3)

iii) Do riparian and in-stream habitats in the Wye differ under different forms of riparian land-use? (Chapter 4)

iv) Does riparian management affect aquatic invertebrate assemblages and salmonids? (Chapters 5 and 6)

v) Are there any larger trends among salmonid populations that might affect the outcomes from more local habitat management? (Chapters 6 and 7)

vi) Has recent riparian management locally within the Wye system reversed declining trends in salmonid populations? (Chapter 6)

In common with modern thesis styles, the work has been prepared so that each individual chapter is self-contained with its own contents list and references. This is intended to facilitate the development of subsequent publications.

[1.4] Management of the Wye Catchment, Wales.

The Wye is a Welsh, upland river system (Edwards and Brooker, 1982; Jarvie *et al.*, 2003). The rural catchment (4136km^2) is dominated by rough pastoral agriculture and conifer forestry in the upland north west, and by arable and dairy faming on the eastern lowlands (Ormerod, 1987, 1988; Ormerod and Edwards, 1987; Edwards *et al.*, 1990; Ormerod *et al.*, 1993; Jarvie *et al.*, 2003). A comprehensive description of the physico-chemical and biotic character of the catchment is detailed in Chapter 3.

Techniques to create optimum habitat for salmonids by the management of riparian habitat are designed to promote spawning and nursery habitat for salmonids and increase autochthonous production (Hemphill and Bramley, 1989; Salmon Advisory Committee, 1991; Giles and Summers, 1996; Holmes, 2002; Chapter 2). Management usually involves fencing of a riparian buffer and coppicing or thinning of bank-side trees (Hemphill and Bramley, 1989; Hendry and Cragg-Hine, 1997). It aims to reduce livestock poaching (trampling) and shading of channels that limits bank vegetation and contributes to bank destabilisation (García de Jalón, 1995; Hendry *et al.*, 2003). By encouraging riparian vegetation, management also aims to reduce silt input to the river from the banks as well as from runoff from the surrounding landscape. Trees are also coppiced to promote bank and channel vegetation. The idea is to create more complex in-stream habitats that provide more refuges for fry and juvenile fish and more prey organisms thus increasing the carrying capacity of the river (O'Grady, 1993; García de Jalón, 1995; Hendry *et al.*, 2003).

Management work on the River Wye, Mid-Wales, between 1996 and 2003 aimed to change the in-stream environment to increase the quality and quantity of habitat available to salmonid fish, primarily Atlantic salmon (*Salmo salar*) and brown trout (*S. trutta*) (The Wye and Usk Foundation, 2006 c, d). Alder (*Alnus sp.)* and willow (*Salix sp.)* trees on the river banks were coppiced or thinned and approx 1-3m of the riparian zone was fenced to exclude grazing (Luxton, 2002). In instances where there was no grazing access, or adjacent land was not stocked, only coppicing of riparian trees was undertaken (Table 1.1).

Sites selected for restoration by the WUF were those traditionally considered to be good salmon rivers. Riparian management primarily took place on the lower reaches of tributaries. The Clywedog and Edw were the most extensively managed tributaries and were coppiced, fenced or both along much of their length (Table 1.1). Management was least (<1.5km) on the Felindre, Cniffiad, Tregoyd and Sgithwen. On the Clettwr, Duhonw, Bach Howey, Marteg and Triffrwd less than 33% of the river length was fenced, coppiced or both. On the Hafrena and Llynfi Dulas between 33 and 66% of the bank length was altered (Table 1.1; Chapters 4 and 6; the exact locations and extents are confidential due to information regarded grant-in-aid).

In 1996, obstructions which may have impeded salmonid migration to spawning grounds were removed from the tributaries of the upper and mid Wye (The Wye and Usk Foundation, 2006b; Chapter 6). The programme included the removal of debris dams and installation of fish passes where practical.

Table 1.1 Tributaries of the Wye system, Wales, on which the riparian zone was managed by fence and/or coppice between 1997 and 2004. Sub-tributaries are given in parenthesis. Fish passes were installed or barriers to migration removed in 1996 as well as from a further 29 tributary streams (see Chapter 6 for locations of fish passes). Precise locations and extents of riparian management are confidential (locations of tributaries are given in Chapters 4 and 6).

Wye tributary	**Tributary length** (km)	**Riparian Management**			**Fish pass or clearance**
		Year	**Type**	**Extent** (% of tributary length)	
Tregoyd	6.9	2002	Coppice	<33%	√
			Fence and Coppice		
Felindre	7.4	2002	Fence and Coppice	<33%	√
Triffrwd	8.0	2003	Fence and Coppice	<33%	
Llynfi Dulas	11.1	2003	Fence and Coppice	33% - 66%	
Bach Howey	18.2	2002	Coppice	<33%	√
Clettwr	11.0	2002	Coppice	<33%	√
Sgithwen	9.2	2000	Coppice	<33%	√
		1997	Coppice		
Edw	24.5	1999	Fence and Coppice	>66%	
Duhonw (Nant Gwyn)	18.1	2001	Coppice	<33%	√
		2002	Fence		
Hafrena	4.5	2003	Coppice	33% - 66%	√
Cniffiad	7.5	1997	Coppice	<33%	√
Clywedog (Bachell Brook, Cwm Hir)	23.3	1998	Fence and Coppice	>66%	√
Marteg	18.4	1998	Fence	<33%	
		2001	Coppice		

[1.5] References

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Chapter 2

The ecology and management of river riparian zones: a review.

[2.0] Abstract

Riparian zones are key elements of riverine environments. Their ecological functions are many, including the mediation of two-way energy transfer across the land-water boundary; hydrological and hydrochemical controls on catchment exports; influences on channel habitat conditions; and the provision of diverse habitat types in the wider riverine landscape. In addition to their basic geomorphology, important features in the riparian zone include the nature and extent of riparian vegetation; soils and associated wetlands.

Recognition of the importance of riparian zones and concerns about their fragmentation and degradation is increasingly leading to attempts at riparian management, protection and restoration. Although undertaken for a wide range of purposes, such management in temperate areas is often undertaken to favour populations of economically important salmonid fishes. In these cases, consequences for other organisms are likely to be large, but are less often evaluated. This review evaluates i) the ecological functions of the riparian zone, ii) the types of riparian management undertaken for salmonids and iii) the potential effects on habitats and other riverine organisms and iv) the methods used to evaluate the effects of riparian management .

In the UK, riparian management for salmonids is primarily undertaken on streams that have been affected by intensively grazed pasture, plantation forest and flood defence. Primary aims include the reduction of siltation (particularly of spawning gravels); increased habitat suitability for all fish requirements (feeding, resting, movement etc.); enhanced prey availability; increased production; and the creation of favorable flow and/or thermal conditions. Satisfactory management in all these cases requires a sound understanding of natural river ecology and this may differ between different locations.

The effects of riparian management on salmonid fish and macroinvertebrates, when reported, have been mixed. Increases in the biomass, density, abundance, and size of salmonids have all been reported, but not in all cases. This could be attributed to variations in outcome between individual catchments. Responses to riparian management might also lag behind action depending on the features measured or the life-cycles of organisms. There are many cases of riparian management where assessment has been poor, badly designed and under-reported.

Currently, there is a need for improved evaluation of riparian restoration and management. There are also opportunities to develop multi-benefit strategies that will augment conditions for a wide range of organisms in addition to salmonids as well key ecological processes (e.g. production; nutrient processing). The need to adapt to climate change, and the need to see reach-based actions as part of wider catchment management, will also change management emphasis. Systematic, scientific design of management projects (that include pre-treatment data) and the dissemination of results are now necessary to further scientific understanding and inform evidence-based conservation and management.

[2.1] Introduction

Riparian ecosystems are terrestrial or wetland habitats that border rivers and interact strongly with the wetted channel to support many important ecological functions (Gregory *et al.*, 1991; Malanson, 1993; Goodwin *et al.*, 1997; Naiman and Décamps, 1997). Recognition of the importance of the riparian zone and concerns raised about the fragmentation and degradation of riparian landscapes, largely attributed to agriculture, commercial forestry, urbanization and the needs of flood defence, have generated management activities to maintain this ecotone habitat as a key part of river conservation (Malanson, 1993; White, 1996; Naiman and Décamps, 1997; Hendry *et al.*, 2003; Reeves *et al.*, 2006; Lovell and Sullivan, 2006).

Some of the current themes in riparian management can be gleaned by assessing the types and scope of scientific papers published over recent years. For example, riparian research is currently increasing as a contribution to river and wetland science (Figure 2.1; Goodwin *et al.*, 1997). Of papers published between 1981-2004 and listed on the ISI® database, using the term 'riparian' somewhere in the title or abstract, 25%, 10% and 2%, respectively, also used the terms 'management', 'restoration' or 'rehabilitation'. Increasing use of the terms 'riparian' and 'management' demonstrates increasing recognition of the importance of the riparian zone in river restoration (Figure 2.1; Naiman *et al.*, 1993; White, 1996; Goodwin *et al.*, 1997; Naiman and Décamps, 1997). A large proportion (84%) of these made recommendations or suggested considerations for riparian management. However, not all such recommendations have been adequately researched. Only a quarter of papers published between 1981 and 2004 on riparian management/restoration/rehabilitation (n=30) directly evaluated the impact of management, which supports the assertion that many riparian restoration projects were conducted without adequate monitoring and/or experimental design (O'Grady, 1998; Rinne, 1999).

Twenty five percent of papers on riparian management with specific reference to the aquatic environment concerned fish populations, often focusing primarily on salmonid species (Table 2.1; Landers, 1997, Muotka *et al.*, 2002a, Hendry *et al.*, 2003; Pusey and Arthington, 2003). Some authors (e.g. Kauffman *et al.*, 1997; Roper *et al.*, 1997; White, 2002) have advocated a move towards an ecological perspective on salmonid restoration programmes. This is inline with the relatively recent concept of management to maintain ecological integrity; the terms 'riparian management' and 'ecological integrity' were first used together in the 1990's (e.g. Ward, 1998).

Another key finding from an overview of published papers is geographical bias. The majority of research (66% of papers) has been conducted in North America (predominantly the USA), which represents just 12% of global surface waters (Figure 2.2; World Resources Institute; 2003). Europe contributed only 11% of the papers demonstrating a need for further exploration of riparian and in-stream dynamics within the European region (Minshall *et al.*, 1983). As a consequence, increased research activity here seems warranted.

This review focuses on studies relevant to temperate ecoregions of the world but draws on studies from elsewhere, when appropriate. This review evaluates: i) the ecological functions of the riparian zone, ii) the types of riparian management undertaken for salmonids, iii) the potential effects on habitats and other riverine organisms and iv) the methods used to evaluate the effects of riparian management. More specifically, it investigates the effects of the riparian environment, primarily riparian habitat, on aquatic habitats and organisms - notably benthic macroinvertebrates and salmonid fish.

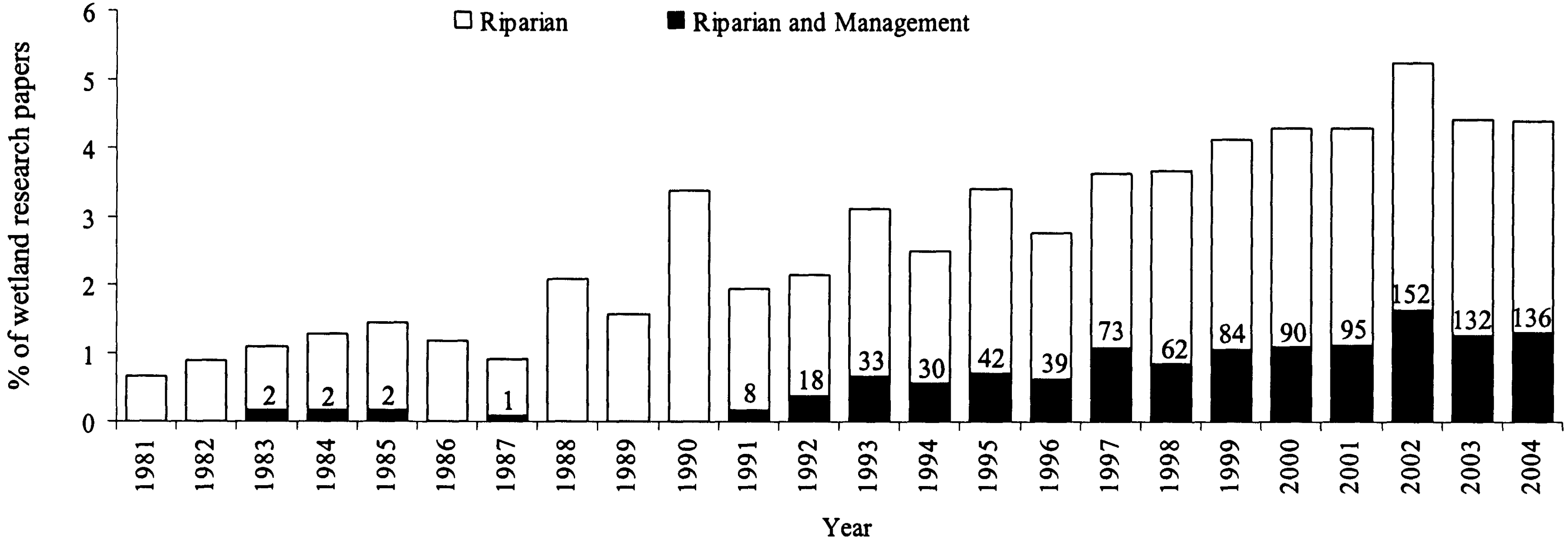

Figure 2.1. The proportion of papers using the search terms 'riparian' or 'riparian and management' as a percentage of wetland research (defined as those papers with 'stream', 'river', 'riparian' or 'wetland' in their title or abstract) listed on the ISI® database (1981-2004). Numbers of papers with 'riparian and management' as search terms are shown within bars.

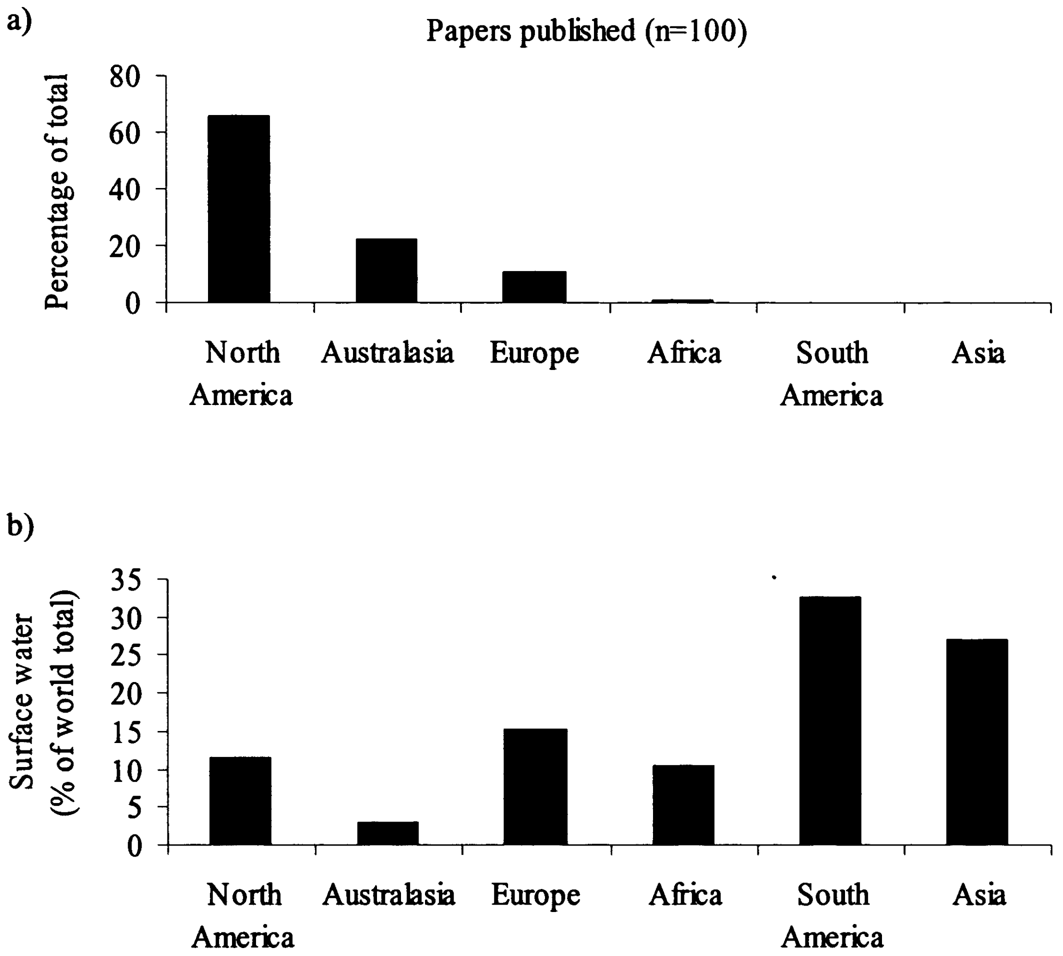

Figure 2.2.

a) The percentage of papers published by region with 'riparian management', 'riparian restoration' and 'riparian rehabilitation' in their title, abstract or as keywords, (excluding reviews, theoretical models and laboratory experiments), listed by the ISI® database, 1981 -2004.

b) The annual extent of the contribution of surface water to Internal Renewable Water Resources (IRWR) (km^3) by region, expressed as a percentage of global surface IRWR (Global surface IRWR =40,594 km^3) (World Resources Institute 2003).

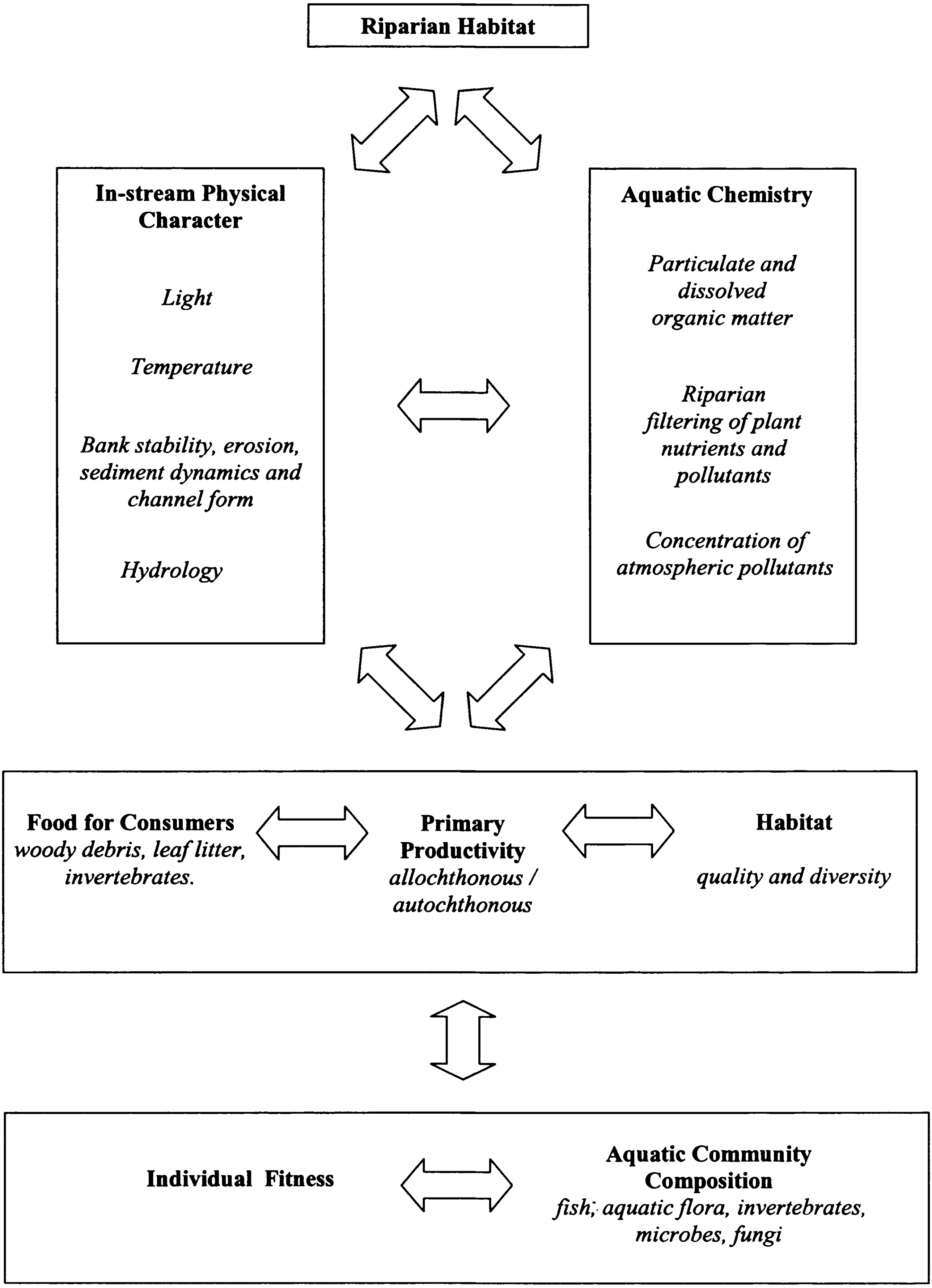

Figure 2.3. The influence of riparian habitat on the aquatic environment (adapted from Sweeney 1992 and Pusey and Arthington 2003). Arrows indicate interactions.

[2.2] Riparian influence on the aquatic environment.

Riparian vegetation affects the physical and chemical environment within a river or stream (Figure 2.3). By shaping the quality and quantity of habitat available to stream biota, the character of the riparian zone influences the floral and faunal communities and trophic nature of the stream (Figure 2.3; Sweeney, 1992; Pusey and Arthington, 2003). The physical environment is primarily altered by shading of the stream channel, modification of water cycling, the input of matter and structural bank changes (Vought *et al.*, 1994).

Light is intercepted by the riparian canopy (Platts *et al.*, 1989). In a study of streams with widths of 3-5m in New Zealand, willow trees reduced incident stream illumination by as much as 80% (Lester *et al.*, 1994). Riparian land use and vegetation type affect the degree of stream shading and direct input of solar energy to the stream (Lester *et al.*, 1994; Knapp and Mathews, 1996; Friberg and Winterbourn, 1997). Friberg and Winterbourn (1997) found that wooded buffers in Wells Creek, a tributary to the Mississippi River, Minnesota, had a significantly higher percentage shade compared to grazed or successional buffers. Seasonal fluxes in riparian cover also vary the light available to streams (Sweeney, 1992).

The importance of canopy shading on the stream environment varies along the river continuum (Vannote *et al.*, 1980). Small, narrow, headwater streams can be completely shaded by bank-side vegetation. Friberg and Winterbourn's model simulations indicated that grasses and forbs provided as much shade as wooded buffers in streams with a width less than 2.5m. Further downstream, on far wider, lowland rivers, the riparian trees shade a much smaller proportion of the stream area.

The extent of riparian influence on water temperature depends on local and regional conditions (Gregory *et al.*, 1991). Water temperature is affected by the density and upstream extent of streamside canopy, the size of the canopy opening above the stream relative to the size of the stream, the season, geographic location and climatic conditions (Barton *et al.*, 1985; Sweeney, 1992; Li *et al.*, 1994; Collier and Smith, 2000). Understanding of these interactions is good enough to enable development of models that relate riparian forest and stream temperature (Theurer *et al.*, 1985; Chen *et al.*, 1998; Blann, 2002; Davies-Colley and Rutherford, 2005; Gaffield *et al.*, 2005). Changes in the productivity and composition of riparian vegetation can alter stream temperature (Beschta and Platts, 1986). Comparatively cooler temperatures are associated with shaded, forested streams and increased temperatures following the loss of riparian vegetation, through logging or grazing have most frequently been reported (Barton *et al.*, 1985; Martin *et al.*, 1986; Platts *et al.*, 1989; Davies and Nelson, 1994; Li *et al.*, 1994; Tait *et al.*, 1994; Blann, 2002; Baillie *et al.*, 2005; Welsh *et al.*, 2005). For example, Barton *et al.* (1985) found that 58 % of variation in maximum water temperatures was explained by the fraction of bank forested within 2.5 km upstream.

Impacts of riparian vegetation on water cycling can be divided into physical and physiological impacts (Bren, 1993; Tabacchi *et al.*, 2000). Tabacchi *et al.* (2000) summarised the main physical impacts of riparian vegetation on water cycling as: 1) interaction with over-bank flow by stems, branches and leaves (turbulence), 2) flow diversion by log jams, 3) change in infiltration rate of flood waters and rainfall by litter, 4) increase of turbulence as a consequence of root exposure, 5) increase of substrate macroporisity by roots, 6) increase of the capillary fringe by fine roots, 7) stemflow (the concentration of rainfall by leaves, branches and stems), and 8) condensation of atmospheric water and interception of dew by leaves. In addition, riparian poaching (trampling) by livestock can reduce the hydraulic conductivity, macropore volume and bulk density of soil (see review by Drewry, 2006). Intensive livestock grazing can also directly destabilize banks and reduce the growth of vegetation which might normally give additional support to the banks (Easson and Yarbrough, 2002; Hook, 2003; Evans *et al.*, 2006). Deep, silted, homogenous channels can result (Whol and Carline, 1996; Hook, 2003; Kondolf, 1993).

The main physiological impacts of riparian vegetation on water cycling were identified as: 1) hydraulic lift, 2) hydraulic redistribution, 3) water storage in large roots, 4) water storage in the stem, 5) water storages in branches and leaves, and 6) evapotranspiration. Surface runoff and subsurface flows are the main pathways of exchanges between the stream and its surroundings (Vought *et al.*, 1994). Water cycling variation has important implications for the physical processes, such as erosion and sediment dynamics, and the chemical properties of stream water. The storage of water in the vascular systems of plants, retention of water in soils, and additional friction imposed by riparian vegetation can lower the flood hydrograph peak, lessen the force of water flowing over-land and reduce its erosive power (Hook, 2003; Dabney *et al.*, 2006). For example, under simulated rainfall events, Butler *et al.* (2006) found that mean runoff volume from bare ground was generally twice that observed from plots with 45-95% cover of grassland vegetation (*Festuca arundinacea* Schreb. and *Paspalum dilatatum* Poir.). Models suggested that bare areas of manured riparian pasture could contribute substantial sediment (> 215 kg ha^{-1}) and P (0.7 kg P ha^{-1}) to surface waters during heavy rainfall and that export may be reduced equally well by low (45%) vegetation cover.

Riparian vegetation, especially that with relatively high stem densities and root mass development, helps to stabilise channel banks and reduce bank erosion (Beschta and Platts, 1986; Hupp, 1992; Sweeney, 1992; Abernethy and Rutherfurd, 1998; Easson and Yarbrough, 2002; McKergow *et al.*, 2003). Lyons *et al.* (2000) found significant effects of riparian land use on erosion and in-stream habitat, with riparian land use accounting for 51% of variation in percentage bank erosion. Williamson *et al.* (1992) suggested that the dominant erosion mechanism was the undercutting of banks. In Bear Creek in a rural Iowa, U.S.A., Zaimes *et al.* (2004) discovered that bank erosion rates were greatest from streams draining row-crop fields, followed by continuously grazed pastures with the lowest erosion rates from stream-banks with riparian forest buffers. Zaimes *et al.* (2006) estimated that the establishment of riparian forest buffers along all non-buffered sub-reaches Bear Creek would have reduced stream-bank soil loss by 77-97 %. On the same water course, grass and woody buffers of approx .16m removed 95-97%, 80-94% of the total-nitrogen, 62-85% of the nitrate-nitrogen, 78-80% of the total-phosphorus (P), and 58-80% of the phosphate-phosphorus (PO4-P) from runoff (Lee *et al.*, 2003).

Along the length of a river different processes may be dominant in the transfer of sediment to flow. In a study of the Latrobe River, Abernethy and Rutherfurd (1998) identified the second quarter of the river's length as the critical zone in which revegetation would be most effective in reducing bank erosion. Riparian vegetation not only influences the amount of sediment that reaches the stream from the adjacent banks but coupled with accumulated leaf litter, vegetation increases hydraulic roughness and traps sediment from overland flow (Herron and Harsine, 1988; Rabeni and Smale, 1995). Once in the stream, sediment alters the form of the channel itself. If hydraulic conditions in the channel promote sediment retention, sediment accumulates in gravel interstices, modifying the composition of the riverbed and reducing the heterogeneity of the stream channel (Naiman and Décamps, 1997; Dale Jones III *et al.*, 1999).

Riparian land use can have significant effects on width/depth channel dimensions (Sweeney, 1992; Lyons and Weigel, 2000; Anderson *et al.*, 2004). Streams with grassy riparian vegetation tend to have smaller stream widths and greater depths compared to those which have either been intensively grazed or shaded by riparian forest. (e.g. Sweeney, 1992; Knapp and Mathews, 1996). Vegetation overhang into the stream is a feature of grassy vegetation. Sediment deposits into the vegetation and gradually builds up the bank, narrowing the channel (Williamson *et al.*, 1992; Kondolf, 1993). Anderson *et al.* (2004) found that the effect of vegetative controls on channel width depended on the size of the river. In channels with small watersheds <10km^2, streams with wooded bank vegetation were wider than those with un-forested banks, but the converse was true for larger rivers with catchment areas of 10 to 100km^2. In eastern Pennsylvanian streams with drainage basin areas from 0.4 to 13 km^2 forested reaches were wider than non forested reaches with rates of deposition and lateral migration higher in non-forested than forested reaches (Allmendinger *et al.*, 2005).

The provision of woody debris to streams has received much attention in the literature in recent years (e.g. Inoue and Nakano, 1998; Keim *et al.*, 2002; Tabacchi and Planty-Tabacchi, 2003, Haschenburger and Rice, 2004; Chen *et al.*, 2006). The amount of woody debris in streams increases with forest cover (Liljaniemi *et al.*, 2002; Heartsill-Scalley and Aide, 2003). Accumulations of woody debris from the riparian zone impact on the hydrological, hydraulic, morphological and biological characteristics of river channels (Gurnell *et al.*, 1995). Woody debris increases pool volume and cover in stream channels and provides more varied and complex habitats (Montgomery, 1997; Keim *et al.*, 2002; Rosenfeld and Huato, 2003; Haschenburger and Rice, 2004; Opperman, 2005). The retention of organic matter is also important to stream productivity and will be discussed later (Vannote *et al.*, 1980; Cummins *et al.*, 1989; Sweeney, 1992). Riparian vegetation is a major determinant of efficiency with which matter is retained within streams, which is a function of stream hydrology and channel structure (Speaker *et al.*, 1984).

Riparian vegetation controls the quantity and type of terrestrially derived organic matter delivered to streams (Gregory *et al.*, 1991; Molinero and Pozo, 2004). The role of riparian vegetation in the provision of particulate organic matter to streams not only contributes to the physical properties of aquatic habitat, but also to chemical and energetic qualities of streams. The amount of coarse organic matter provided by the vegetation varies with the type of vegetation and size of the stream (Tabacchi *et al.*, 2000). Stream bank vegetation is a source of organic matter and large woody debris also controls the transport of organic matter by modifying the retentiveness of the channel (Speaker *et al.*, 1984; Maridet *et al.*, 1997; Pretty and Dobson, 2004; Lepori *et al.*, 2005). Lester *et al.* (1994) found that stream reaches with riparian willow trees had significantly higher amounts of coarse particulate organic (CPOM; >5mm) compared to reaches without trees. Similarly, Pretty and Dobson (2004) found that woody debris increased retentiveness but also that deciduous leaves tended to be more retentive than conifer needles.

The type of riparian vegetation also affects the quality of organic matter that enters the stream. Herbs and shrubs provide a high quality (low C:N ratio, e.g. alder 15:1) source of energy compared with forested streams that receive larger woody debris of low energetic content (Gregory *et al.*, 1991; Giller and Malmqvist, 1998). Deciduous and non-woody plants with higher C:N ratios (typically 200->1000:1) are more rapidly broken down by microbes, fungi and invertebrates (Cummins *et al.*, 1989; Giller and Malmqvist, 1998; Hicks and Laboyrie, 1999; Quinn *et al.*, 2000). Similarly, in British Columbia, decomposition rates of deciduous alder (*Alnus rubra*) were about 50% faster than the two conifer litter types (cedar (*Thuja plicata*) and western hemlock (*Tsuga heterophylla)* (Richardson *et al.*, 2004). Elliott *et al.* (2004) found that C:N ratios at coniferous sites were significantly higher than those at alder-dominated deciduous sites or at sites without riparian cover. Litter fall in deciduous stands exceeds that of evergreen stands and exhibits more pronounced seasonal patterns of input (Gregory *et al.*, 1991). The timing of leaf–fall influences the seasonal dynamics of organic matter, with maximum input during the autumn months (Kochi *et al.*, 2004).

Breakdown of coarse and fine particulate organic matter will also contribute to dissolved organic matter through breakdown in the stream channel, but the riparian zone is also a direct source of dissolved organic matter to the stream (Fiebig *et al.*, 1990; Sweeney, 1992; Naiman and Décamps, 1997; Tabacchi *et al.*, 2000).

Streamside forests also alter many other chemical qualities of stream water, dependent on the species composition, age distribution and density of streamside trees as well as the rate and spatial pattern of water movement through the riparian zone (Sweeney, 1992). The main impacts of riparian vegetation on water quality have been comprehensively reviewed by Tabacchi *et al.*, (2000) and were summarised as: direct nutrient uptake, root excretion, storage and concentration of mineral and organic components, fast decomposing organic matter release from litter, slow decomposing organic matter release from woody debris, indirect uptake through symbiotic associations (bacteria and fungi) and leaching of pollutants and natural compounds at the surface of the plant .

The addition of a riparian buffer to a landscape can decrease the discharge of material to streams, slow down the transport of chemicals to the stream, and change the chemical composition of runoff that enters a stream (Gregory *et al.*, 1991; Giles and Summers, 1996; Weller *et al.*, 1998; McKergow *et al.*, 2003; Fisher and Acreman, 2004; Nieminen *et al.*, 2005). Riparian zones are increasingly used to control diffuse sources of pollution, such as fertilisers, pesticides and sediment (Norris *et al.*, 1991; Waters, 1995; Naiman and Décamps, 1997; Gippel and Collier, 2000; Lowrance *et al.*, 2000; Felsot *et al.*, 2003; McKergow *et al.*, 2003; Anbumozhi *et al.*, 2005).

The use of riparian buffer zones in mitigating against the export of plant nutrients (primarily nitrogen and phosphorus) from agriculture has received most attention in the literature (see recent review by Lovell and Sullivan, 2006). Riparian zones may significantly modify the amount, form, and timing of nutrient export from watersheds (Gregory *et al.*, 1991). Roots intercept soil-water and uptake nutrients directly. They can also increase the time that soil-water stays in contact time with sites of biogeochemical processing (Sweeney, 1992; Vought *et al.*, 1994).

Riparian zones can therefore act as sinks for nitrogen and phosphate (Phillips, 1989; Mulholland, 1992). However, buffer effectiveness is variable and limited by spatial and temporal conditions, such as season, soil type and redox-potential, and the magnitude of pollutant loads (Phillips, 1989; Tabacchi *et al.*, 1998; McKergow *et al.*, 2003). When the riparian system becomes saturated, it can also act as a source of organic and inorganic matter to streams. At the end of the growing season in temperate regions, terrestrial litter provides seasonal pulses of dissolved leachates to streams (Gregory *et al.*, 1991). Chemicals that adsorb to sediment, such as phosphorus, can be removed from overland flows when sediment is deposited in the riparian zone (Vought *et al.*, 1994; Naiman and Décamps, 1997; Lockaby *et al.*, 2005). Conversely, erosion may also provide a source of such chemicals to streams, particularly under anaerobic conditions in the case of phosphate (Mulholand, 1992).

Nutrients in a stream do not cycle between water, particulates and consumers in one place, but are transported downstream (Vannote *et al.*, 1980). This 'nutrient spiraling' concept dictates that riparian influence on nutrient dynamics in the headwaters of streams will also have downstream effects (Newbold *et al.*, 1981).

Dry atmospheric deposits can provide additional sources of nitrogen, as well as sulphates (Tabacchi *et al.*, 2000). Also, the leaf canopy provides cations such as potassium, calcium and magnesium (Tabacchi *et al.*, 2000). Atmospheric pollutants, such as sulphates, 'scavenged' by the forest canopy are transported to the soil and groundwater (Tabacchi *et al.*, 2000). Plantations of exotic conifers in particular can accentuate the acidification of freshwaters in catchments with base-poor rocks and soils (Ormerod *et al.*, 1989; Ormerod *et al.*, 1993). Acidity promotes the mobilisation of metals in the soil. For any given pH in Welsh upland streams, aluminium concentrations were, on average $46 \mu g l^{-1}$, higher in streams draining conifer catchments than in streams draining catchments of moorland or deciduous woodland. This effect occurred irrespective of buffer strips in conifer catchments and the taxon richness of macroinvertebrates aquatic declined with increasing acidity and aluminium concentration (Ormerod *et al.*, 1989; Ormerod *et al.*, 1993).

[2.3] Riparian impact on aquatic biology.

Riparian influence on channel form, substrate characteristics and the provision of overhead cover shapes the quality and quantity of in-stream habitat for aquatic flora and fauna (Wesche *et al.*, 1987). Through its impact on the aquatic habitat, productivity and food web, the riparian zone ultimately affects the fitness of individuals and composition of aquatic communities.

Riparian vegetation provides allochthonous energy inputs into stream ecosystems whilst shading the stream channel and limiting autochthonous productivity. Reduced riparian shading and higher irradiance is associated with increased algal abundance and biomass (Murphy *et al.*, 1981; Davies and Nelson, 1994; Li *et al.*, 1994; Friberg and Winterbourn, 1997; Quinn *et al.*, 1997; Sabater *et al.*, 1998). Seasonal patterns of light and riparian cover therefore affect algal productivity (Sweeney, 1992). On investigating small, headwater streams (1st - 3rd order) on the Olympic Peninsula of Washington, U.S.A., Elliott *et al.* (2004) found higher concentrations of diatoms in sites that lacked canopy cover. Riparian forests also affect the structure and productivity of the microbial (algae and bacteria) food web by modifying the levels of dissolved organic carbon and nutrients (Sweeney, 1993).

Stream width relative to vegetation overhang determines the degree of riparian influence on the production of aquatic flora. In streams that are comparatively narrow, a higher proportion the channel bed can potentially be shaded by riparian vegetation (Vannote *et al.*, 1980; Gregory *et al.*, 1991). In these smaller, headwater streams, allochthonous and autochthonous food resources for invertebrate consumers, detritivorous and herbivorous fish are more strongly controlled by riparian zones (Gregory *et al.*, 1991; England and Rosemond, 2004). Downstream, as stream size increases, the importance of terrestrial organic input is less, increasing the significance of autochthonous primary production and organic transport from upstream sources (Vannote *et al.*, 1980).

[2.3.1] Riparian influence on benthic invertebrates.

Benthic invertebrates are the primary consumers within river systems and have a diverse range of feeding mechanisms. They include filter-feeders, detritivores, scrapers and grazers and others, such as Odonata, a number of Heteroptera, Diptera and Trichoptera and some leeches that are predatory (Moog, 1995). Modification of the riparian character influences the relative composition of functional feeding groups and trophic structure of invertebrate assemblages (Gregory *et al.*, 1991; Rundle *et al.*, 1992). By controlling the sources of energy reaching the stream, riparian zones determine the quality and quantity of food available to aquatic consumers. So, the abundance of shredders decreases with increasing stream width, as the relative influence of bank-side trees is less (Vannote *et al.*, 1980; Statzner *et al.*, 2004).

Following alteration of the riparian wetlands, changes in the structure of invertebrate communities are frequently reported (Ormerod *et al.*, 1993; Davies and Nelson, 1994; Sabater *et al.*, 1998; Benstead *et al.*, 2003; Fuchs *et al.*, 2003; Parkyn *et al.*, 2003; Wright *et al.*, 2003; Sweeney *et al.*, 2004; Hernandez *et al.*, 2005; Kreutzweiser *et al.*, 2005). Riparian induced changes in community assemblage are often attributed to an increase in herbivore abundance in open sites and higher numbers of shredders in forested sites (Hawkins *et al.*, 1982; Minshall *et al.*, 1983; Gregory *et al.*, 1991; Sweeney, 1993; Tait *et al.*, 1994; Sabater *et al.*, 1998; Wright *et al.*, 2003). For example, in first-order streams in northern New England (U.S.A.), proportional representation of shredders was positively correlated with canopy cover ($r = 0.584$) which increased post-logging. The proportion of grazing invertebrates was correlated with recently logged, high-density stands with low mean tree diameter ($r = 0.604$). Total macroinvertebrate and chironomid abundance were correlated with canopy cover ($r = 0.586$ and 0.80, respectively). In contrast, Sweeney (1993) observed a decline in the abundance of grazing species and an increase in filter feeders in response to reduced light levels (hence algal availability). Sweeney suggested that under reduced algal conditions, the grazing species expanded their territory in order to meet their individual nutrient requirements, which resulted in lower densities of grazers, and hence more room for filter feeding species.

Riparian vegetation primarily affects macroinvertebrate diversity through effects on benthic habitat (Sweeney, 1993; Naiman and Décamps, 1997). Complex in-stream habitat is created by living and dead components of riparian vegetation, and diversifies the macroinvertebrate community (Sweeney, 1992; Ormerod *et al.*, 1993). Conversely, the loss of riparian vegetation can cause sedimentation of gravel substrates and decrease the abundance macroinvertebrates in riffles (Davies and Nelson, 1994; Waters, 1995). For example, the density and abundance of macroinvertebrates, biomass of shredders and amount of particulate organic matter are often lower in streams without riparian grazing than in streams with livestock (Davies and Nelson, 1994; Whol and Carline, 1996; Robertson and Rowling, 2000; Scrimgeour and Kendall, 2003). The surface area of organic and inorganic substrate that is available for colonisation by aquatic species is modified through riparian effects on the structure and retentive capacity of the stream (Speaker *et al.*, 1984; Sweeney, 1992, 1993). In support of this idea, Sweeney (1992) found that woody roots were also particularly important habitat for invertebrates, as was large woody debris (>0.3m long) in a small forested stream.

The provision of particulate organic matter to streams promotes detritivore assemblages but their abundance and composition varies with the structure and chemical make-up of riparian plant communities (Vannote *et al.*, 1980; Speaker *et al.*, 1984; Cummins *et al.*, 1989; Gregory *et al.*, 1991; Dobson *et al.*, 1995; Linklater, 1995; Haapala *et al.*, 2003). For example, in British Columbia during the autumn, invertebrate assemblages in streams draining alder (*Alnus rubra*) and cedar (*Thuja plicata*) forests were distinguished from those with western hemlock (*Tsuga heterophylla)* by the high densities numbers of detritivores (*Lepidostoma*, *Zapada*, and *Paraleptophlebia*) on the former litter types (Richardson *et al.*, 2004).

Overhanging tree shade, total tree cover and abundance of *Fraxinus angustifolia* and *Alnus glutinosa* explained 18% of total invertebrate variation in the lowland Sado basin, Portugal (Aguiar *et al.*, 2002). The type of tree, native or exotic, coniferous or deciduous will also shape the invertebrates that inhabit the stream. In the British uplands, grazer abundances were lowest at sites with riparian zones of conifer compared with riparian moorland and grassland and broadleaved trees (Rundle *et al.*, 1992). Similarly, in New Zealand and Australia, exotic European willow trees, *Salix sp.*, were widely planted in an attempt to stabilize river banks but may support fewer terrestrial and aquatic species of invertebrates than in rivers with native trees (Lester *et al.*, 1994; Wilkinson, 1999; Greenwood *et al.*, 2004).

The width of the riparian zone has often been implicated in the control of invertebrate density, and the presence of individual species, e.g. the dragonfly *Chlorolestes tessellatus* which prefers a buffer width of > 30 m (Samways and Steytler, 1996; Fitzpatrick *et al.*, 2001). Wider buffers might be able to satisfy a greater range of ecological functions (Davies and Nelson, 1994; Nakamura and Yamada, 2005).

Riparian effects on macroinvertebrates result from complex interactions between energy inputs, stream physical structure and chemistry (Ormerod *et al.*, 1993). General trends in invertebrate abundance and diversity in relation to riparian vegetation are therefore difficult to define and vary with local environmental conditions. This is illustrated by two studies undertaken by Lester *et al.* (1994) and Glova and Sagar (1994). They compared invertebrate communities in small streams in New Zealand, with and without (exotic) riparian willow (*Salix sp.*). Glova and Sagar (1994) found higher species richness and diversity of benthic invertebrates in willowed sections. Conversely, Lester *et al.* (1994) observed lower invertebrate densities and biomass in willowed sections. Environmental variables other than riparian vegetation, such as acidity, may have overriding effects of on aquatic communities, such effects may be moderated by riparian vegetation (Ormerod *et al.*, 1993; Tierney *et al.*, 1998; Liljaniemi *et al.*, 2002).

[2.3.2] Riparian influence on fish.

The availability of habitat and food to fish is intricately linked to the riparian zone (Gregory *et al.*, 1991). It controls the food resource of herbivorous and detritivorous fishes by provision of organic matter (leaf litter, woody debris, etc.) and by the regulation of primary production through shading (Gregory *et al.*, 1991; Glova and Sagar, 1994; Whol and Carline, 1996; Abe *et al.*, 2003). The biomass of predatory fishes is limited by the provision of terrestrial invertebrates as well as the production of aquatic prey within the stream (Lobón-Cerviá, 2000; Allan *et al.*, 2003; Baxter *et al.*, 2005; Bojsen, 2005; Nakamura and Yamada, 2005). Brook trout (*Salvelinus fontinalis*) density (R^2 = 0.431) and biomass (R^2 = 0.26) were directly and positively related to total benthic invertebrate abundance in a study investigating the effects of logging and riparian forest characteristics on biota in headwater streams (Nislow and Lowe 2006). Brown trout (*Salmo trutta*) take approximately equal proportions of terrestrial and aquatic prey, and may alter their primary food source to compensate for losses of prey due to acidification or riparian fragmentation (Allan *et al.*, 2003; Ormerod *et al.*, 2004; Baxter *et al.*, 2005; Bojsen, 2005).

Riparian effects on fish are site-specific and vary along the river continuum (Vannote *et al.*, 1980). In headwater streams, inputs of allochthonous material may increase the density of invertebrates and hence fish biomass (Wipfli and Musslewhite, 2004).

Further downstream, shading of in-stream macrophytes by riparian trees may limit aquatic production and reduce the abundance and diversity of fishes adapted to lowland rivers (Growns *et al.*, 2003).

Modification of riparian forest by deforestation, afforestation, livestock grazing or exclusion can induce changes in channel habitat, such as the amount of overhead cover or pool habitat which may then induce shifts in the structure of fish assemblages (Chapman and Knudsen, 1980; Maridet and Souchon, 1995; Dale Jones III *et al.*, 1999; Bojsen, 2005; Feyrer *et al.*, 2006). For example, salmonids spawn in redds created in gravel substrate and eggs can be suffocated by sediment entering the stream unimpeded by riparian vegetation (Kondolf and Wolman, 1995; Chapman, 1988; Platts *et al.*, 1989; Waters, 1995; Whol and Carline, 1996; Moir *et al.*, 1998; Malcolm *et al.*, 2003). In reaches with fragmented riparian vegetation and high sediment loads, fish species that require swift, shallow water and well aerated substrates or do not guard eggs may decrease; whereas fishes that guard nests or live in slower, deeper water may increase in abundance (Dale Jones III *et al.*, 1999; Brazner *et al.*, 2005). Dale Jones III *et al.* (1999) found that longer deforested patches were associated with decreased abundance of benthic-dependent species, which were replaced by sediment tolerant and sometimes invasive species. Fine sediment also adversely impacts the productivity of periphyton and aquatic invertebrates and hence fish feeding, growth and survival (Suttle *et al.*, 2004; Nakamura and Yamada, 2005).

The relationship between riparian cover and fish populations is unclear, probably because effects are specific to species and to the reach environment (Armstrong *et al.*, 2003). Without riparian vegetation, temperatures in some streams may exceed the tolerance ranges of fish species (Theurer *et al.*, 1985; Tait *et al.*, 1994; Nakamura and Yamada, 2005). Whilst reduced temperatures in some shaded, temperate streams may slow the growth of trout (Eklov *et al.*, 1999; Lobón-Cerviá, 2000). Lobón-Cerviá (2000) found that in forested, shaded sites in northern Spain, adult trout grew more slowly and spawned fewer larger eggs that resulted in larger alevins, compared to open sites. In Irish rivers, 'tunnelling' by dense vegetation over the stream was correlated with reduced salmonid numbers during summer days (O'Grady, 1993). However, the availability of cover to juvenile salmonids is a major factor that promotes the abundance of salmonids (Armstrong *et al.*, 2003). Cover protects against predators and delimits feeding territories (Keeley and Grant, 1995, Steingrimsson and

Grant, 1999; Imre *et al.*, 2002; Mossop and Bradford, 2004). It is provided by surface water turbulence, submerged structures, undercut banks, overhanging vegetation, the last three of which are promoted by the presence of riparian vegetation (Wesche *et al.*, 1987; Armstrong *et al.*, 2003; Spina, 2003). Woody debris and riparian cover offer protection for fishes, especially during the winter (Wesche *et al.*, 1987; Cunjak *et al.*, 1998; Inoue and Nakano, 1998; Young *et al.*, 1999; Mossop and Bradford, 2004). Brooks *et al.* (2004) reintroduced woody debris to the Williams River, Australia. Areas of pools and riffles, as well as pool depth increased, as did the species richness and abundance of fish assemblages. A greater availability of habitat area can increase fish biomass; whilst complex habitats promote species diversity (Iwata *et al.*, 2003; Brooks *et al.*, 2004; Coutant, 2004).

The large body of literature exploring the habitat preferences of salmonids was recently reviewed by Armstrong *et al.* (2003). Salmonids require different water velocity, depth, substrate, cover and oxygen requirements in spawning, nursery (used during the first summer after hatching) and rearing areas, so need complex habitats to complete their lifecycle. Well aerated gravels are necessary for spawning and egg survival (Armstrong *et al.*, 2003). Fry establish feeding territories and refuges in riffles thus avoiding competition or predation from older salmon which prefer deeper water (Armstrong *et al.*, 2003).

Density dependent mechanisms are important in limiting the capacity of streams to accommodate salmonid populations, especially in the juvenile life stages (Alabaster and Lloyd, 1980; Kondolf and Wolman, 1995; Armstrong *et al.*, 2003; Milner *et al.*, 2003). Population bottlenecks that occur when the standing crop approaches the carrying capacity of the environment and restrict the growth of salmonid populations have been reported to occur at times of spawning (redd construction) and fry emergence (Cunjak *et al.*, 1998; Bardonnet and Baglinière, 2000; Armstrong *et al.*, 2003; Milner *et al.*, 2003). Self-thinning is the term used to define the progressive decline in density caused by competitively induced losses in a cohort of growing individuals (Lonsdale, 1990; Steingrimsson and Grant, 1999; Milner *et al.*, 2003). Examination of the density-dependent mechanisms responsible for self-thinning of populations has generated two hypotheses: the energy equivalence and the space hypothesis (Steingrimsson and Grant, 1999). The energy equivalence hypothesis suggests that food availability limits the production of juveniles, whilst the space hypothesis proposes competition for space as the driving mechanism. The two hypotheses are not mutually exclusive. Both mechanisms have been reported and one or the other has rarely been entirely rejected (Elliott, 1993; Grant *et al.*, 1998; Steingrimsson and Grant, 1999; Milner *et al.*, 2003). The distribution and abundance of salmonids are therefore strongly influenced by their habitat, especially during population bottlenecks (Armstrong *et al.*, 2003).

[2.4] Management of riparian habitat for salmonids.

Modification of the riparian zone, namely through livestock grazing and forestry practices in rural catchments, changes the composition and structure riparian vegetation and consequently the stream habitat (Platts, 1991; Putman *et al.*, 1991; Lewis and Williams, 1994; Liljaniemi *et al.*, 2002; Kauffman *et al.*, 2004). Plantations of exotic tree species, clear cutting of plantations and livestock grazing all contribute to a loss of riparian habitat diversity and potentially reduce the benefits of the riparian zone to the aquatic environment (Putman *et al.*, 1991; Davies and Nelson, 1994; Lewis and Williams, 1994; Knapp and Mathews, 1996; Tierney *et al.*, 1998; Robertson and Rowling, 2000; Hendry and Cragg-Hine, 2002; Kauffman *et al.*, 2004; Wissmar, 2004).

In response to widespread salmonid declines, fisheries managers now advocate the restoration of natural processes and functions of aquatic ecosystems in order to sustain the production of wild salmon populations (Salmon Advisory Committee, 1991; Schramm and Hubert, 1996; Kauffman *et al.*, 1997; Roper *et al.*, 1997; Williams *et al.*, 1999; White, 2002). They recognise the importance of riparian vegetation in the functioning of the aquatic ecosystem and modification of the riparian zone is a central theme of salmonid restoration (Platts and Wagstaff, 1984; Barton *et al.*, 1985; Theurer *et al.*, 1985; Martin *et al.*, 1986; O'Grady, 1993; Roper *et al.*, 1997; Hendry *et al.*, 2003). Traditionally, only the stream channel itself was altered by the provision of in-stream structures such as groynes, brush bars, large woody debris or provision of gravels (House and Boehne, 1985; Armantrout, 1991; Konynenbelt, 1993; Karle and Densmore, 1994; Jungwirth *et al.*, 1995; House, 1996; O'Grady, 1998; Clarke and Scruton, 2002; Lehane *et al.*, 2002; Roni, 2002; White, 2002; Merz *et al.*, 2004). Direct alteration of in-stream habitat is now considered a short-term solution to stream enhancement, in advance of development of the functional attributes of riparian vegetation (Beschta, 1991; Karle and Densmore, 1994; Imhof *et al.*, 1996; Lehane *et al.*, 2002; White, 2002; Erskine and Webb, 2003; Opperman, 2005). For example, the artificial provision of woody debris to a channel might not be necessary once trees planted in a riparian zone started to contribute their trunks and branches to the river.

Technical guidance on the management of riparian vegetation has been provided alongside recommendations for structural modification of the channel by research organizations, consultancies, government agencies, river and conservation trusts

(White and Brynildson, 1967; Hemphill and Bramley, 1989; Salmon Advisory Committee, 1991; Giles and Summers, 1996; Hendry and Cragg-Hine, 1997; Rickard, 2002; Urbani, 2002; Correll, 2005). Such management has been implemented under the auspices of projects such as the US Fish and Wildlife Service's Partners for Fish and Wildlife Programme, Columbia River Basin Fish and Wildlife Program, Trout Unlimited in Virginia and the Aquatic Conservation Strategy, in the Pacific Northwest, USA, Australia's Landcare Programme, Trout 2010 in Germany and the River Restoration Project in the UK (Campbell *et al.*, 1997; Butler *et al.*, 1999; River Restoration Centre, 1999; Williams *et al.*, 1999; Gippel and Collier, 2000; McGurrin and Duff, 2002; Tent, 2002).

Management of riparian habitat to promote conditions within the stream that are preferable to salmonids is primarily undertaken on streams that drain intensively grazed pastures and commercially managed forests (Welsh, 1993; Davies and Nelson, 1994; Lyons and Weigel, 2000; Hendry and Cragg-Hine, 2002; Rowe *et al.*, 2002; Hendry *et al.*, 2003; Northcote and Hartman, 2004). It includes the reduction of siltation of spawning gravels and promotion of cover (Krog and Hermansen, 1985; Hemphill and Bramley, 1989; Salmon Advisory Committee, 1991; Lewis and Williams, 1994; Giles and Summers, 1996; Naiman and Décamps, 1997; Glen, 2002) Attempts have also been made to increase supplies of allochthonous or autochthonous energy to streams, to enhance biotic production and hence salmonid density (Krog and Hermansen, 1985; O'Grady, 1993; Lewis and Williams, 1994; Giles and Summers, 1996; Clarke and Scruton, 2002). Although some controversy exists as to the value of in-stream plant production against the loss of leaf litter as a food source for invertebrates in headwaters, the maintenance of structural diversity within riparian areas is generally considered to be beneficial to the aquatic environment (Lewis and Williams, 1994; Maltby, 1994; García de Jalón, 1995; Hendry *et al.*, 2003).

Riparian buffer strips can be effective in maintaining native fish communities and macroinvertebrates (Webster *et al.*, 1992; Davies and Nelson, 1994; Dale Jones III *et al.*, 1999; Rowe *et al.*, 2002). Davies and Nelson (1994) recommended a minimum width of >30m for riparian buffers within commercial forestry plantations, to increase salmonid production in Tasmanian streams. Petersen *et al.* (1992) suggested buffer strips at least 10m wide on each side of the stream as a restoration goal, based on nutrient reduction and habitat considerations. Buffer widths of 10m tend to protect the physical and chemical characteristics of a stream, while the maintenance of ecological integrity requires widths closer to 30 meters (Broadmeadow and Nisbet, 2004). Conversely, Lakel *et al.*, (2006) suggested that a relatively small (7.62 meters per side) forested streamside management zone is just as effective at protecting against post-harvest erosion losses of sediment as much larger (15.24 and 30.48 meters) buffers, probably due to an increase in understorey vegetation following harvest. Alternative widths of buffers are likely to be appropriate for different management targets, such as nutrient reduction, protection of fish or bird communities, and biodiversity conservation (Petersen *et al.*, 1992).

Suitable widths of riparian buffers to ameliorate forestry practices have received much attention, but very little guidance exists on the extent of fencing for grazed catchments (Keller and Burnham, 1982; Platts and Wagstaff, 1984; Anderson *et al.*, 1993; Corner and Bassman, 1993; O'Grady, 1998). In catchments of multiple landownership the width of riparian buffers often varies because the extent of grazing elimination is dependent on the support of riparian owners (Konynenbelt, 1993; Gippel and Collier, 2000; Rhodes *et al.*, 2002; Opperman, 2005). The dimensions of riparian zones that are required for effective buffering also vary with land use and hydrogeology of the catchment and riparian zone (Davies and Nelson, 1994; Lowrance *et al.*, 2000; Sabater *et al.*, 2003; Hefting *et al.*, 2004; Burt, 2005).

Dale Jones III *et al.* (1999) advised that the length of a riparian buffer be considered, as well as adjacent riparian width, in stream protection and restoration plans. Parkyn *et al.* (2003) also noted that improvement in invertebrate communities was linked to decreases in water temperature and suggested that the restoration of in-stream communities could only be achieved after canopy closure along long buffer lengths. Small, headwater streams are important sources of invertebrates, detritus and sediment to downstream reaches as well as being important rearing habitat for trout (Rosenfeld *et al.*, 2002; Moerke and Lamberti, 2003; Correll, 2005; Wipfli, 2005). However, they are often excluded from restoration initiatives, a failing which could counteract any positive effects of restoration (Rosenfeld *et al.*, 2002; Moerke and Lamberti, 2003).

Fencing is frequently erected to exclude livestock grazing, and to encourage the development of riparian vegetation; in addition, native trees may be planted and / or exotic species removed (Keller and Burnham, 1982; Platts and Wagstaff, 1984;

Williamson *et al.*, 1992; Welsh, 1993; Butler *et al.*, 1999; McKergow *et al.*, 2003). Thinning of riparian trees is advocated to promote grassy vegetation and to prevent excessive shading of channels (O'Grady, 1993; García de Jalón, 1995; Hendry and Cragg-Hine, 1997; Lyons and Weigel, 2000; Hendry *et al.*, 2003). Long-term programmes of fencing, alongside planting and pruning of trees have been proffered to maintain the diversity and complexity of bank-side habitat and to optimise the in-stream environment for salmonids (García de Jalón, 1995; Giles and Summers, 1996; Hendry *et al.*, 2003).

Once fenced, riparian habitat often changed rapidly but gradual changes in channel morphology and bed characteristics were not always sufficient to alter fish communities (Platts and Wagstaff, 1984; Kondolf, 1993; Lyons and Weigel, 2000; Nerbonne and Vondracek, 2001; Agouridis *et al.*, 2005). Conversely, on rivers in catchments that were previously overgrazed in Ireland, bioengineering techniques in combination with fencing programmes demonstrated a relatively quick ecological recovery once physical stability was restored (O'Grady, 1998; Gargan *et al.*, 2002; O'Grady *et al.*, 2002b). For example, the numbers of trout parr and macroinvertebrate taxa increased after 4 years, as did the variety of invertebrate feeding guilds in the Glenglosh River (O'Grady *et al.*, 2002a). In other cases, the management of riparian habitat has successfully promoted the production of fish. Increases in the biomass, density, abundance, and size of fishes have all been reported in response to exclusion or reduction of livestock grazing (Chapman and Knudsen, 1980; Keller and Burnham, 1982; Li *et al.*, 1994; Knapp and Mathews, 1996; Whol and Carline, 1996). Most of those studies were concerned with cattle grazing, but there were some examples of sheep or mixed grazing (e.g. Platts, 1991; Williamson *et al.*, 1992; Hendry and Cragg-Hine, 2002). Riparian buffers within logged catchments have also been associated with increased salmonid abundance, fish numbers, and enhanced native fish communities (Davies and Nelson, 1994; Rowe *et al.*, 2002).

The impacts of livestock grazing and plantation forestry on salmonids do vary from place to place. For example, elevation of temperature to levels at the extreme of salmonid tolerance can be induced by livestock grazing in desert streams, a factor unlikely to be of similar significance in temperate streams (Li *et al.*, 1994; Tait *et al.*, 1994). Similarly, the impact of plantations of exotic conifers on stream acidity, and potentially on fish production, may be exacerbated by base poor soils (Edwards *et al.*, 1990; Lacroix and Korman, 1996; Tierney *et al.*, 1998). It is hardly surprising then that the outcomes of riparian restoration are also variable. The vast majority of studies on riparian buffers were conducted in the U.S.A.'s Pacific north-west (primarily in Oregon), with contributions from the American mid-west, Canada and New Zealand (Platts, 1991; Simpkins *et al.*, 2002; Compton *et al.*, 2003; Donnelly, 2003; Meleason *et al.*, 2003; Parkyn *et al.*, 2003; Reeves *et al.*, 2003; Boothroyd *et al.*, 2004; Kauffman *et al.*, 2004; Moerke and Lamberti, 2004; Mossop and Bradford, 2004). Whilst general trends can be gleaned from these studies, restoration efforts elsewhere should tailor management to suit river-basin characteristics (Blinn and Kilgore, 2001; Brazner *et al.*, 2005; Watanabe *et al.*, 2005). Characterization of river catchments is therefore important to identify priorities for management (Imhof *et al.*, 1996; Boon *et al.*, 2002; Logan and Furse, 2002; Dudgeon, 2003; Sear and Newson, 2003). This is now recognised in European legislation. Under the Water Framework Directive (Directive 2000/60/EC) 'River Basin Management Plans' will establish the ecological status of surface waters and detail programmes of measures to prevent deterioration and to 'protect, enhance and restore all bodies of surface water'.

Methods employed to restore rivers should be based on fundamentals of ecological and physical science and appropriate to the stream character and management objectives, but not loose sight of conditions at the landscape scale (Platts and Rinne, 1985; White, 1991; Muhar *et al.*, 1995; Fiest *et al.*, 2003; Martin *et al.*, 2006). Simple, reach scale, solutions are unlikely to be effective on their own. Often the cause of change has occurred over a larger scale than the remedial solutions, and restoration treats the symptoms rather than the cause of the problem (O'Grady *et al.*, 2002a). Large-scale, watershed characteristics and climatic conditions may have greater control over fish populations than riparian factors alone (Platts and Wagstaff, 1984; Ormerod *et al.*, 1993; Imhof *et al.*, 1996; Whol and Carline, 1996; Kauffman *et al.*, 1997; Stefansson *et al.*, 2003). Management of riparian habitat is a step towards a whole ecosystem approach to maintaining river habitat and diversity, restoration might best be administered at the catchment scale (Harper *et al.*, 1999; Giller and O'Halloran, 2004).

[2.5] Evaluation of riparian management.

Evaluation of management has two primary functions. First, to establish the effects of management, second, to inform management so that practices can be improved. Each restoration project, or management action, constitutes an experiment, and assessed as such (Kondolf, 1995; Minns *et al.*, 1996; White, 1996; Landers, 1997). Objective and robust project evaluation enables scientists and managers to learn from management experiences and improve approaches to management (Kondolf, 1995; Minns *et al.*, 1996; White, 1996).

For effective evaluation, scientific design, systematic post-project evaluation, and dissemination of results (both positive and negative) are important elements in development of management strategy (Kondolf, 1995; Minns *et al.*, 1996; White, 1996; Bash and Ryan, 2002). In the past, all or some of these elements were lacking in riparian research, and within the field of river restoration (Kondolf, 1995; Muhar *et al.*, 1995; O'Grady, 1998; Rinne, 1999; Sarr, 2002). The key strategy for inference of impacts is to find some evidence for impact that cannot easily be explained away by various other processes, such as natural variation in the system (Downes *et al.*, 2002). The principles of restoration ecology provide a unifying model framework for riparian management (Sarr, 2002). Kondolf (1995) set out five elements for effective evaluation of river and stream restoration projects; clear objectives, baseline data, good study design, commitment to the long term and a willingness to acknowledge failures.

The need for clear objectives to enable effective study design and evaluation has been repeatedly expressed in the literature (White, 1991; Frissel and Nawa, 1992; Muhar *et al.*, 1995; Jones *et al.*, 1996). Muhar *et al.* (1995) stated that the main objective should be a comprehensive ecological improvement, taking the entire catchment into consideration. Ideally, this would be achieved through a series of broad objectives leading to more specific, quantifiable objectives (White, 1991; Kondolf, 1995). In this way, monitoring designs can be developed from project goals (Bash and Ryan, 2002; Ryder and Miller, 2005).

Monitoring variables are dictated by project objectives. Species-specific approaches to riparian management may result in limited data retrieval. Bash and Ryan (2002) suggest that biological measures, such as salmonid population data, are appropriate measures of project goals, as population conditions are subject to a multitude of variables. However, due to the large number of variables to which fish respond, it is also appropriate to monitor physical, chemical and water quality parameters (Bash and Ryan, 2002). White (1996) noted that the literature, pre 1988, contained little about the effects of habitat management at ecosystem or community levels and that some early papers covered only the physical durability of structural work. Positive responses of fish to management have been attributed to increased habitat availability, and / or productivity, but such assumptions have rarely been tested in the evaluation of management (Chapman and Knudsen, 1980; Whol and Carline, 1996).

The majority of recent papers that have evaluated riparian management focused on physical and fish based parameters (e.g. Chapman and Knudsen, 1980; Kondolf, 1993; Knapp and Mathews, 1996; Lyons and Weigel, 2000; Brooks *et al.*, 2004; Opperman, 2005). There are few examples of studies that evaluate how 'single-goal' restorations aimed at salmonid fisheries affect other stream biota (Muotka and Laasonen, 2002). Some assessed benthic invertebrate or algal responses to management (e.g. Sabater *et al.*, 1998; Liljaniemi *et al.*, 2002; Kiffney *et al.*, 2003; Wright *et al.*, 2003; Harrison *et al.*, 2004; Kiffney *et al.*, 2004). Very few explored any other aspect of the aquatic environment, such as microbial responses and ecosystem function (e.g. Murphy *et al.*, 1981; Lepori *et al.*, 2005). Cognizance of natural stream features and processes has increased, as reflected by more recent papers that have explored the response of water chemistry, invertebrates and the aquatic community to management (e.g. White, 1996; Whol and Carline, 1996; Scrimgeour and Kendall, 2003).

Baseline data are collated to establish the pre-treatment condition of study sites. These data should be consistent with project objectives as an objective basis for evaluating change caused by the project, to determine whether objectives were met (Kondolf, 1995). Baseline data can also be used in the planning of habitat improvements and to confirm whether the project objectives were appropriate (Diamond *et al.*, 2002; Stanfield and Kilgour, 2002). For example, if objectives call for the addition of woody debris to streams, baseline data may confirm a paucity of woody debris in treatment streams.

Pre-treatment data are necessary to identify change and is particularly important when management activities cannot be replicated or controls are absent, which is often the case in studies on livestock exclusion (Sarr, 2002). Data should therefore be collected for as long as possible during the pre-project period (Kondolf, 1995). Whilst fisheries and ecologically based management have better track records in collection of pre-project data than projects focusing on 'engineering' goals, this is the area of project evaluation most frequently absent from stream and riparian management projects (Bash and Ryan, 2002; Sarr, 2002). A recent review of stream restoration and enhancement projects in Washington State, USA, found only 53% of had baseline data, and a survey of stream restorations in Indiana found that fewer than half conducted pre- or post-project monitoring (Bash and Ryan, 2002; Moerke and Lamberti, 2004). In some cases, when pre-treatment was available, results were expressed in the literature as percentage change (e.g. Platts and Nelson, 1985). Statistical evaluation would have established the significance of those results, and further aided subsequent projects.

Project success can only be evaluated objectively in the context of quantifiable change (Kondolf, 1995; Rinne, 1999). Any effects of management on the complex river environment must be distinguished from many other simultaneous changes in the aquatic environment (Kondolf, 1995). Studies therefore need to be carefully designed prior to treatment and a number of authors have called for guidelines on study design (Kondolf, 1995; Minns *et al.*, 1996; Rinne, 1999; Bash and Ryan, 2002; Sarr, 2002)

Such guidelines would probably constitute an experimental design involving a framework of pre- and post-treatment data with control sites and replication, where possible, suitable for statistical evaluation (Table 2.1; Kondolf, 1995; Downes *et al.*, 2002). Such guidelines must be flexible in their approach, to account for environmental, anthropogenic and financial constraints on ideal study designs (Platts and Rinne, 1985). For example, lack of pristine (ungrazed) reference streams is a constraint common to many studies evaluating effects of management of riparian grazing (Platts and Rinne, 1985; Sarr, 2002).

Control sites are extremely useful in highlighting changes that may have resulted from influences other than effects of the project (Stewart-Oaten *et al.*, 1986; Kondolf, 1995; Landers, 1997; Downes *et al.*, 2002). Spatial replication of treatment and control sites help to overcome confounding factors which may be specific to a individual sites and can draw out factors associated with management and increase statistical power (Hair *et al.*, 1995; Downes *et al.*, 2002; Sarr, 2002).

Replicate reaches or streams within experimental design have been common, and many (e.g. paired designs) have replicate controls (Table 2.1). In the evaluation of riparian and stream management for salmonids control sites have most frequently been located on the same stream as treatment sites, often as paired reaches on the same stream (Table 2.1; Chapman and Knudsen, 1980; Murphy *et al.*, 1981; Davies and Nelson, 1994; Robertson and Rowling, 2000; Pretty *et al.*, 2003; Opperman and Merenlender, 2004). Sampling of independent control streams is an objective method to avoid the pseudoreplication of data within streams due to downstream effects (Hurlbert, 1984). It is preferable to have a large number of control streams to overcome confounding factors introduced by stream variability (Green, 1993; Hair *et al.*, 1995). However, in practice this is not always possible due to the availability of suitable control or replicate streams (Platts and Rinne, 1985; Sarr, 2002; Opperman and Merenlender, 2004).

A few studies evaluating management for salmonids have made significant inferences from experimental designs with only one control (e.g. Scrimgeour and Kendall, 2003) and some studies without controls and / or replicates have been reported (Table 2.1; House *et al.*, 1991; Frissel and Nawa, 1992; Karle and Densmore, 1994; Penczak, 1995; Blann, 2002; McKergow *et al.*, 2003). Subjectivity is introduced in determining whether observed changes were a response to management or wider environmental conditions, especially in single reach studies (Karle and Densmore, 1994; Penczak, 1995; McKergow *et al.*, 2003). However, the majority of study designs had some replicate and control reaches or streams (85%) and 79% presented a degree of statistical assessment of the significance of results (Table 2.1). Analysis of variance designs have been most commonly applied to determine significant differences between control and treatment site variables in the absence of pre-treatment data (e.g. Keller and Burnham, 1982; Williamson *et al.*, 1992; Sabater *et al.*, 1998; Robertson and Rowling, 2000; Muotka and Laasonen, 2002; Rowe *et al.*, 2002; Kiffney *et al.*, 2003; Harrison *et al.*, 2004; Opperman and Merenlender, 2004). Occasionally correlation and regression statistics have been applied to data, and potential relationships and variance explained by different physical variables presented (e.g. Frissel and Nawa, 1992; Linløkken, 1997; Kiffney *et al.*, 2003; Pretty *et al.*, 2003).

Before-After–Control-Impact (BACI) designs involve sampling control and impact locations, both before and after putative impact, together with proper replication of each of these four elements, where possible (Stewart-Oaten *et al.*, 1986; Green, 1993; Downes *et al.*, 2002). Replicated BACI-type designs allow us to separate, with relatively high confidence, human-caused effects from natural processes (Downes *et al.*, 2002). Despite recommendation for use in the design of experiments involving the modification of riparian and in-stream habitat, BACI designs have rarely been applied to such studies (Table 2.1; Minns *et al.*, 1996; Sarr, 2002; Pretty *et al.*, 2003). Only 15% of studies identified in this review that evaluated habitat manipulation that aimed to promote salmonids incorporated pre-treatment data, control sites and replication as well as statistical analysis (Table 2.1).

Emphasis has been frequently placed on the need for long-term commitment to the post-evaluation period of projects, in order to identify management impacts, and the durability of impacts (Reeves *et al.*, 1991; Kondolf, 1995; Trexler, 1995; Bash and Ryan, 2002; Sarr, 2002). The duration of post-project monitoring necessary to determine efficacy depends on project goals and on trends in the natural variability of the riparian and aquatic environment (Reeves *et al.*, 1991; Trexler, 1995; Landers, 1997; Bash and Ryan, 2002; Parkyn *et al.*, 2003). For example, if the endpoint defined for a project is a rise in fish population size, large natural fluctuations in fish populations may confound the difficulty in detecting changes (Reeves *et al.*, 1991; Trexler, 1995). The time for required changes to take effect may be substantial, for example, between fifty and one hundred years may be required for riparian forest regeneration. Five to ten years has been suggested as a suitable period for identification of changes due to river restoration projects, such as woody debris addition to streams (Kondolf, 1995; Trexler, 1995; Bash and Ryan, 2002).

In its most simplistic form, evaluation identifies whether or not management objectives were achieved, with a view to implement those practices with the desired outcome. This approach was frequently adopted by riparian projects aimed at single species, such as salmon (Muotka *et al.*, 2002). However, this 'black box' approach to evaluation fails to identify causative mechanisms involved in the pathway from management to effect, and is of limited value to adaptive management (Platts and Rinne, 1985; Jones *et al.*, 1996; Rinne, 1999). Studies of fish responses to management have yielded very different results but the mechanisms underlying responses have rarely been explored (Platts and Rinne, 1985; Jones *et al.*, 1996; Rinne, 1999). Scientific studies can help to evaluate the ecological success of restoration projects, beyond specific management objectives (Muhar *et al.*, 1995). Spatial and temporal scales of ecosystem responses are often greater than the scales of human intervention and assessment but understanding of the scales and mechanisms of ecosystem recovery remains limited (Imhof *et al.*, 1996; Minns *et al.*, 1996; Poff, 1997; Sarr, 2002; Allan, 2004).

The scale of observation is an important consideration in designing a strategy for the evaluation of management. There is a nested hierarchy of spatial and temporal scales at which ecosystems operate, human impacts occur and assessments are carried out, in which larger scale variables exert control on finer scale variables (Figure 2.4; Imhof *et al.*, 1996; Minns *et al.*, 1996). However, linkages between processes operating at different scales are not necessarily linear or straightforward (Imhof *et al.*, 1996, Poff, 1997; Royer and Minshall, 2003; Allan, 2004). Action, or evaluation, at one scale cannot be assumed to affect, or represent, processes operating at another scale (Imhof *et al.*, 1996). Small-scale investigations may yield contrasting results to larger scale investigations of the same management event (Fiest *et al.*, 2003). The scale and method of evaluation should therefore be appropriate for the assessment of project objectives (Rabeni and Smale, 1995; Minns *et al.*, 1996).

Spatial units	Approximate linear dimensions (m)	Biotic assemblage scale
Ecoregion	$>10^4$	Community / Population
Stream-system	10^3 to 10^4	Community / Population
Reach	10^1 to 10^2	Population
Site	10^0 to 10^1	Individual
Habitat element	10^0 to 10^{-1}	Individual

Figure 2.4. A nested hierarchy of spatial scales of ecosystem function and evaluation, approximate linear dimensions of spatial units and associated scales of biotic assemblages (adapted from Rabeni and Smale 1995; Imhof *et al.*, 1996 and Royer and Minshall 2003).

Small scale-investigations can provide valuable information on the effects of habitat variables, such as depths, velocities, cover factors on habitat preferences and of distribution individuals (Figure 2.4; Rabeni and Smale, 1995). Intermediate investigations at the stream-system level can relate individual fish preferences to the availability of habitat to populations, as determined by geomorphic and fluvial dynamic forces. Meanwhile, evaluation at the ecoregion level can demonstrate the overriding influence of physiographic variables on both the composition of communities and abundance of populations. Perception of which habitat variables are important in the restoration of salmon habitat is therefore a function of the scale of observation (Fiest *et al.*, 2003). The design of restoration activities at all spatial scales, will enable the examination of functional relationships among them and help to identify the scales at which management is most appropriate (Imhof *et al.*, 1996; Landers, 1997).

Table 2.1. Elements of study design applied by work published (between 1981 and 2003, n=33) on the evaluation of management undertaken in riparian areas and streams to benefit salmonids.

		Elements of study design				
Management focus	**Reference**	**Pre-treatment data**	**Control**	**Replication**	**Statistical evaluation**	**All 4 elements**
Grazing	Lyons and Weigel 2000		√	√	√	
	Scrimgeour and Kendall 2003		√	√	√	
	Kondolf 1993		√	√	√	
	Chapman and Knudsen 1980		√	√	√	
	Li *et al.*, 1994			√	√	
	Keller and Burnham 1982		√	√	√	
	Platts and Nelson 1985	√	√	√		
	Robertson and Rowling 2000		√	√	√	
	Knapp and Mathews 1996		√	√	√	
	Williamson *et al.*, 1992		√	√	√	
	Whol and Carline 1996		√	√	√	
In-stream modification	Muotka and Laasonen 2002	√	√	√	√	√
	Brittain *et al.*, 1993	√	√	√	√	√
	Pretty *et al.*, 2003		√	√	√	
	Jungworth *et al.*, 1995		√	√		
	Muotka *et al.*, 2002	√	√	√	√	√
	Armantrout 1991	√		√		
	House *et al.*, 1991	√		√		
	House and Boehne 1985	√	√	√		
	House 1996	√	√	√		
	Frissell and Nawa 1992		√	√	√	
	Linløkken 1997	√	√		√	
	Karle and Densmore 1994	√				
	Young *et al.*, 1999	√	√	√	√	√
	Davies and Nelson 1994		√	√	√	

Table 2.1. *continued...*

Management focus	Reference	Elements of study design Pre-treatment data	Control	Replication	Statistical evaluation	All 4 elements
In-stream modification	Murphy *et al.*, 1981		√	√	√	
	Rowe *et al.*, 2002		√	√	√	
Riparian modification	Penczak 1995	√			√	
	Dale Jones III *et al.*, 1999		√	√	√	
	Sabater *et al.*, 1998	√	√	√	√	√
	Liljaniemi *et al.*, 2002		√	√	√	
	Blann *et al.*, 2002				√	
	McKergow *et al.*, 2003	√			√	
Proportion of papers with elements of study design		**42%**	**79%**	**85%**	**79%**	**15%**

[2.6] Riparian management for salmonids: research needs.

Interest in riparian research is increasing (Goodwin *et al.*, 1997; Pusey and Arthington, 2003; Shields *et al.*, 2003). An understanding of the general functions of the riparian zone can be applied to develop management to conserve aquatic environments (Vannote *et al.*, 1980; Gregory *et al.*, 1991; Sweeney, 1993; Chen *et al.*, 1998; Tabacchi *et al.*, 1998; Blann, 2002). However, many riparian restoration projects have been conducted without sufficient monitoring and/or experimental design (e.g. Table 2.1). Further research is required to establish the impacts of riparian management on habitats and organisms as well as the mechanisms underlying responses to management (Sweeney, 1992; Vought *et al.*, 1994; Jones *et al.*, 1996). Understanding the factors driving ecological change will become increasingly important as organisms respond to new environmental conditions in the light of climate change (Hulme *et al.*, 2002; Harris *et al.*, 2006).

Management of riparian areas is beginning to move beyond a focus on single species at the reach scale to encompass the whole ecosystem (Kauffman *et al.*, 1997; Muotka *et al.*, 2002; White, 2002; Katopodis, 2005). Research that assesses management impacts on the structure and function of aquatic communities within the context of the wider catchment should now be conducted (Imhof *et al.*, 1996; White, 1996; Landers, 1997). Systematic, scientific design of management projects (that include pretreatment data) and the dissemination of results is now required to further scientific understanding and inform management (Kondolf, 1995; Muhar *et al.*, 1995; Minns *et al.*, 1996; White, 1996; Hendry and Cragg-Hine, 1997; O'Grady, 1998; Rinne, 1999; Bash and Ryan, 2002; Sarr, 2002; Hendry *et al.*, 2003).

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Chapter 3

Standard biotic indices identify catchment-scale priorities for restoration and management in the Wye river system.

[3.0] Abstract

Although biotic indices derived from invertebrates are used often in river monitoring and surveillance, their use in diagnosing problems and prioritising management response is less common. Here, semi-diagnostic biotic indices from archived data were used in combination to characterise catchment-wide macroinvertebrate assemblages in the Wye river system.

Invertebrates from streams (n=55) in three sub-catchments, distinguished on geology, topography and land use, varied significantly in taxonomic composition and feeding guild representation. There were also major variations in indices of organic pollution (BMWP/ASPT) and acidification (AWIC), with acidified and enriched sites both distinct from semi-natural sites across the catchment.

Although the Wye is designated for its conservation importance, these data illustrate the need to mitigate acidification in upland base-poor regions and to better manage diffuse nutrient sources in the lower catchment. These data illustrate how classical and novel biotic indices, applied to routine monitoring data, can aid in prioritising restoration and management. The development of bio-diagnostic tools for identifying sedimentation and habitat degradation would further aid in identifying diffuse management problems in river systems.

[3.1] Introduction

Throughout the world, the ecological status of rivers is assessed using biological indicators (Whiles *et al.,* 2000; Cai *et al.,* 2003; Metzeling *et al.,* 2003; Camargo *et al.,* 2004; Mancini *et al.,* 2005; Uyanik *et al.,* 2005; Padisak, 2006). In some of the most successful of all the developments from ecological monitoring, biotic metrics have identified problems ranging from classical insanitary pollution to the detection of acidification, eutrophication and metals (Rosenberg and Resh 1993; Hawkes, 1997; Kelly, 1998; Davy-Bowker *et al.*, 2005). Effective bioassessment not only detects pollution or impacts, but also appraises the outcomes of restoration or management (Bailey *et al.,* 1998; Stone and Wallace, 1998; Lakly and McArthur, 2000; Purcell *et al.,* 2002). In Europe, the need to identify the effects of pressures on rivers and to maintain or restore good ecological status has been thrown into particularly sharp profile by the Water Framework Directive ('WFD', Directive 2000/60/EC).

Biotic indices have been developed from communities of fish, diatoms, phytoplankton, protozoans and protists but the majority utilise benthic macroinvertebrates in a range of rapid assessment protocols (Resh and Jackson, 1993; Metcalfe-Smith, 1994; Kelly, 1998; Madoni, 2000; Lobo *et al.,* 2004; Cao *et al.,* 2005; Jiang, 2006; Pont *et al.,* 2006). This group has the advantage of being ubiquitous, relatively easy to collect and identify, and diverse enough to include taxa with varying sensitivity to a range of pollutants (Rosenberg and Resh, 1993; Metcalfe-Smith, 1994; Yuan and Norton, 2003). Typically, a standardised kick-sample of invertebrates is collected from the benthos and scores are then assigned to taxa according to their sensitivity to a given range of pollutants (Metcalfe-Smith, 1994; Bradley and Ormerod, 2002a). These scores in turn, are formed into metrics or indices that are considered to vary systematically with water quality (Resh and Jackson, 1993). In some cases modelling methods are also used that detect departures from putative reference conditions (Wright *et al.*, 1998).

Many indices were developed primarily for sanitation purposes, such as organic pollution from sewage effluent and other toxic contamination of drinking water (Cairns and Pratt, 1993). Now, indices based on benthic macroinvertebrate communities are a key feature of assessments of the 'ecological status', 'health' and 'biotic integrity' of rivers (e.g. Reice and Wohlenberg, 1993; Harding *et al.*, 1999;

Griffith *et al.*, 2005; Donohue *et al.*, 2006). In order to fulfil this role, indices should be more than auto-indicative, revealing the wider biological character of communities.

Macroinvertebrate indices are routinely applied in the USA, Australasia, and much of Europe and new indices are being developed elsewhere (e.g. Zamora-Munoz and Alba-Tercedor, 1996; Thorne and Williams, 1997; Mustow, 2002; Czerniawska-Kusza, 2005). Their ability to simplify complex data is widely recognised but questions still arise as to their value in revealing the detail of assemblage structure (Cao *et al.*, 1997; Cao and Hawkins, 2005). It is often reported that single indices cannot fully describe changes or variation in community structure (e.g. Lydy *et al.*, 2000). Therefore, many authors still prefer to complement biotic metrics with multivariate techniques, frequently ordination (Cao *et al.*, 1996; Zamora-Munoz and Alba-Tercedor, 1996; Klemm *et al.*, 2002; Griffith *et al.*, 2003; Herlihy *et al.*, 2005). In contrast to biotic indices, ordination reveals much of the complexity of communities but interpretation can be ambiguous (Austin, 1985). Biotic indices have the advantage of being standardised and comparable.

Many studies have been devoted to identifying the 'best' biotic indicator for a particular site or pollutant (e.g. Cao *et al.*, 1997; Lydy *et al.*, 2000; Solimini *et al.*, 2000) but less attention has been focused on the complementary use of different biotic metrics (e.g. Barton and Metcalfe-Smith, 1992; Chessman and McEvoy, 1998; Klemm *et al.*, 2002). In a study to establish the capacity of biotic metrics to detect stressors, Griffith *et al.* (2003) defined the diagnosis of environmental stressors to lotic ecosystems as the 'use of biological data to identify the presence and relative importance of a particular environmental stressor at individual sites'. Different indices measure different aspects of assemblages and used in combination might not only detect problems but diagnose the causes of impairment.

In the UK there has been particular focus on BMWP (Biological Monitoring Working Party) scores, for which invertebrates are identified to family (Hawkes, 1997; Environment Agency, 1999). More recent indicators include the Acid Water Indicator Community (AWIC), a system for detecting the impacts of acidification based on the same families recorded in the BMWP (Davy-Bowker *et al.*, 2005; Ormerod *et al.*, 2006).

Index scores calculated at the species level often reveal more detailed differentiation between sites and more subtle impacts (Resh and Jackson 1993; Chessman, *et al.*, 2002; Waite *et al.*, 2004). Therefore, some authors claim that family-level data reduces the performance of analysis techniques and can bias assessment scores (e.g. Resh and McElravy, 1993; Gabriels *et al.,* 2005). Others (e.g. Vos *et al.*, 2002; Waite *et al.*, 2004) have demonstrated spatial and temporal variation in family-level data and maintain that identification to the family level is sufficient for biomonitoring and assessment (Verdonschot, 2000). Can the AWIC and BMWP indices reveal structural variation in invertebrate communities within river basins?

This chapter aimed to address some of the questions surrounding the value and applicability of biotic indices to contrasting conditions – and hence potentially contrasting management problems – in the catchment of the Welsh River Wye. In particular, I asked whether family-level data are sufficient to detect variations in environmental quality, reveal assemblage structure and diagnose causes of impairment. Specific objectives were to identify: i) environmental variation among sub-catchments of the Wye, ii) the extent to which biotic metrics reflected environmental gradients, iii) whether biotic metrics could identify spatio-temporal patterns among sub-catchments and through time and iv) whether biotic indices revealed differences between sites in assemblage structure and guild composition.

[3.2] Study Area

The River Wye rises in the Plynlimon Hills in Mid-Wales (at 741 metres above Ordinance Datum) and flows for about 250 km along the Wales / England border before joining the Severn Estuary at Chepstow (Figure 3.1; Edwards and Brooker; 1982; Jarvie *et al.*, 2003). The ecological character of the Wye was extensively described by Edwards and Brooker (1982). The largely rural catchment (4136km^2) is dominated by rough pastoral agriculture and conifer forestry in the upland north west and by arable and dairy faming on the eastern lowlands (Ormerod, 1987, 1988; Ormerod and Edwards, 1987; Edwards *et al.,* 1990; Ormerod *et al.,* 1993; Jarvie *et al.*, 2003). Variation in land use largely reflects geology, with base-poor Ordovician and Silurian formations in the upper catchment and base-rich Devonian Old Red Sandstone forming the lowlands along with an upland massif in the south west (Edwards and Brooker, 1982; Environment Agency, 1998; Brennan *et al.*, 2003). The relatively impermeable geology and large volume of precipitation (2453mm average annual rainfall at Cefn Brwyn 1961-1990) in the uplands results in flashy river regimes. Low precipitation (1011mm average annual rainfall at Redbrook 1961-1990) and a more substantial groundwater supply to lowland tributaries create more constant flow regimes (Jarvie *et al.*, 2003; NERC, 2005).

The upper Wye between its source and Builth Wells has a V-shaped channel, riffles and pools and is fringed by trees. Tributaries in the upper catchment are fast flowing with relatively large bed gradients (Edwards and Brooker; 1982). The Wye has four major tributaries the Irfon (244 km^2) to the north-west, the Ithon (365km^2) in the north-east, the Lugg (1070km^2) to the east and Monnow (433km^2) to the south. The Irfon has high relief, similar to the upper Wye and contrasts with the lower elevation and subdued relief of the Ithon. The Lugg and Monnow are largely characterised by broad valleys and low altitude, except for the upper Monnow which rises in the Black mountains. Downstream from Builth Wells, the valley floor of the main River Wye widens and the channel gradient is more gentle and meanders are interrupted only at gorge sections downstream from Builth Wells and at Ross.

Edwards and Brooker (1982) reported that bryophytes were dominant in the upper Wye (e.g. *Marsupella emarginata*) as were amphibious vascular plants (e.g. *Juncus bufonis* and *Rorippa islandica)* and tall herbs (e.g. *Chrysanthemum vulgare* and

Artemesia vulgare) in the lower catchment. Invertebrate communities in the main River Wye also demonstrated altitudinal zonation. *Phagocata vitta, Chloroperla spp., Sericostoma personatum, Simulium variegatum* and *Eusimulium brevicaule* were restricted to the upper part of Wye (<120m from source) while *Dugesia lugubris* and *Hellobdella stagnalis* only occurred at more than 150 km from the source of the Wye. In addition, geology and water chemistry influenced the distribution of species. *Assellus aquatucus, Gammarus pulex* were only found where calcium concentrations were >9mg/l and *Gammarus* were absent at pH <5.7.

The River Wye and its tributaries are Sites of Special Scientific Interest and a Special Area of Conservation (SAC) under the terms of the European Union Habitats Directive 1992 (Environment Agency, 2000; Environment Agency, 2003). Internationally important species supported by the Wye system include the European otter (*Lutra lutra*), White-clawed crayfish (*Austropotamobius pallipes*) Atlantic salmon (*Salmo salar*), bullhead (*Cottus gobio*), Twaite shad (*Alosa fallax*), Allis shad (*Alosa alosa*), sea, brook and river lampreys (*Petromyzon marinus, Lampetra planeri, Lampetra fluviatilis)* and water crowfoot (*Ranunculus fluitans*) (Environment Agency, 1998; JNCC, 2006).

Restoration programmes on the Wye currently focus on treating the symptoms of acidification by catchment liming in the uplands. Fencing of riparian zones in the middle catchment aims to reduce sediment input and enhance riparian and in-stream biodiversity. Point sources of phosphates from sewage treatment works have also been reduced in the last decade (Environment Agency, 2000).

For this study, three sub-catchments were distinguished as the upper- mid- and lower-sub-catchments after Edwards and Brooker (1982) and Jarvie *et al.* (2003) (Figure 3.1). Only tributary streams (n=55) were used in analyses, main river sites were excluded. Two additional sites, Monk's ditch and Mounton Brook, are technically outside the Wye catchment, but as directly adjacent lower-catchment tributaries of the Severn they were included here (Figure 3.1).

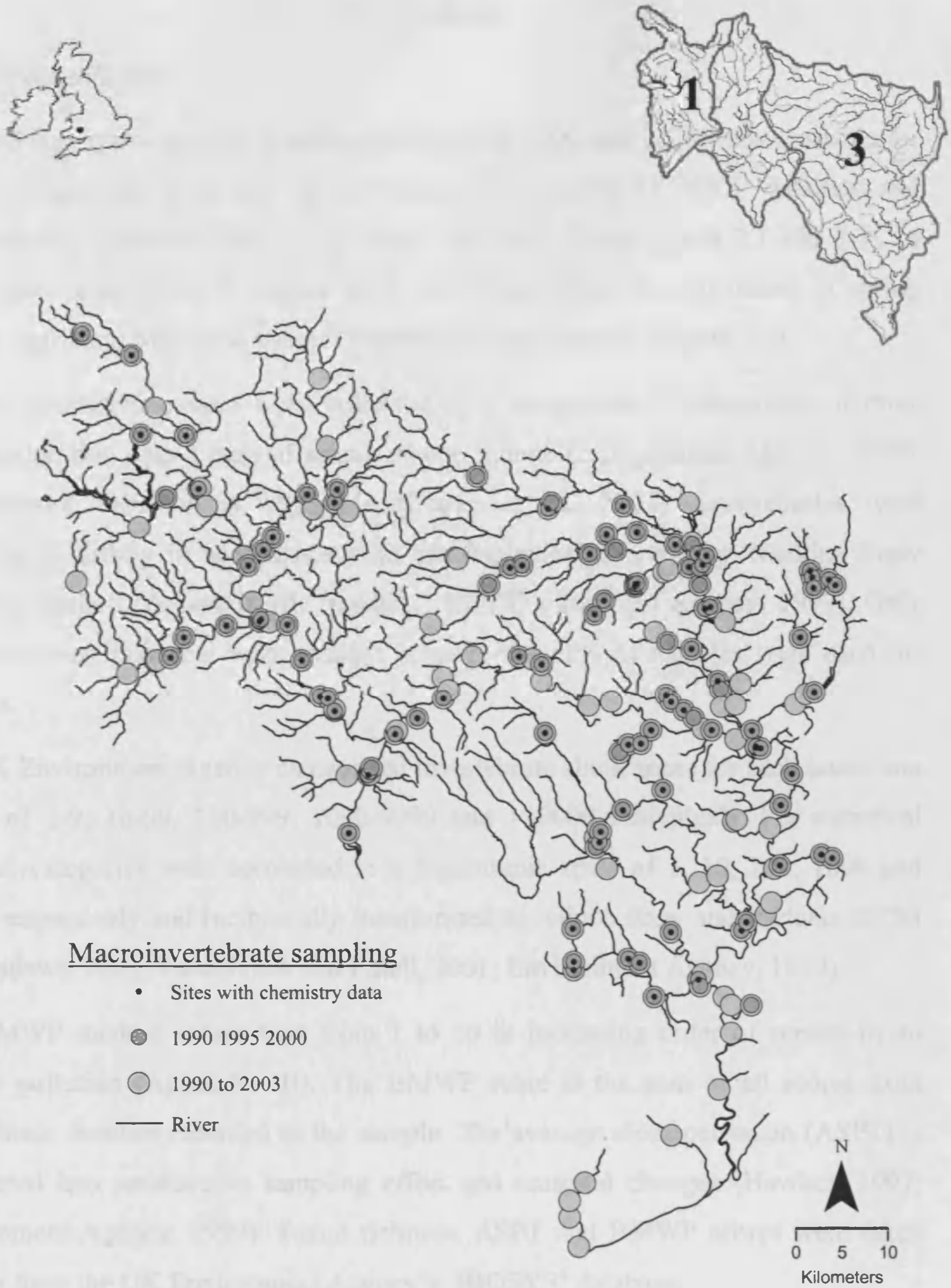

Figure 3.1. Macroinvertebrate sites in the Wye river system (as defined by the Environment Agency) sampled during 1990-2003 (◉). Sites sampled in 1990, 1995 and 2000 (●) where water chemistry data were available are shown (▪). The map was extracted from the Ordnance Survey 1:50000 Scale Colour Raster available from Digimap® (EDINA, Edinburgh). The inset map delineates the upper (1), mid (2) and lower (2) sub-catchments of the Wye

[3.3] Methods

[3.3.1] Data collection

Archived surveys of aquatic invertebrates between 1990 and 2003 were available for the Wye catchment from the UK Environment Agency's 'BIOSYS' database and predominantly reflected sampling in 1990, 1995 and 2000 (Figures 3.1 and 3.2). In those three years 85% of samples from the Wye tributaries were taken in spring (March, April and May) and autumn (September and October) (Figure 3.3).

Benthic macroinvertebrates were collected by a standardised kick-sample of three minutes duration plus a manual search of one minute (Environment Agency, 1999). Inter-operator variation is negligible (Clarke *et al.,* 2002). Invertebrates were identified to family in accordance with the Biological Monitoring Working Party (BMWP) method (Appendix III; Hawkes, 1997; Environment Agency, 1999). Only BMWP-scoring taxa that were present in more than 1% of samples were used for analysis.

The UK Environment Agency categorised invertebrate abundances for each taxon into counts of 1-9, 10-99, 100-999, 1000-9999 and >10000 individuals. For statistical analysis, categories were converted to a logarithmic scale of 1, 10, 100, 1000 and 10000 respectively and reciprocally transformed to reduce skew and kurtosis (SPSS for Windows, 2001; Tabachnick and Fidell, 2001; Environment Agency, 1999).

The BMWP method scores taxa from 1 to 10 in increasing order of sensitivity to organic pollution (Appendix III). The BMWP score is the sum of all scores from invertebrate families recorded in the sample. The average score per taxon (ASPT) is considered less sensitive to sampling effort and seasonal changes (Hawkes, 1997; Environment Agency, 1999). Taxon richness, ASPT and BMWP scores were taken directly from the UK Environment Agency's 'BIOSYS' database.

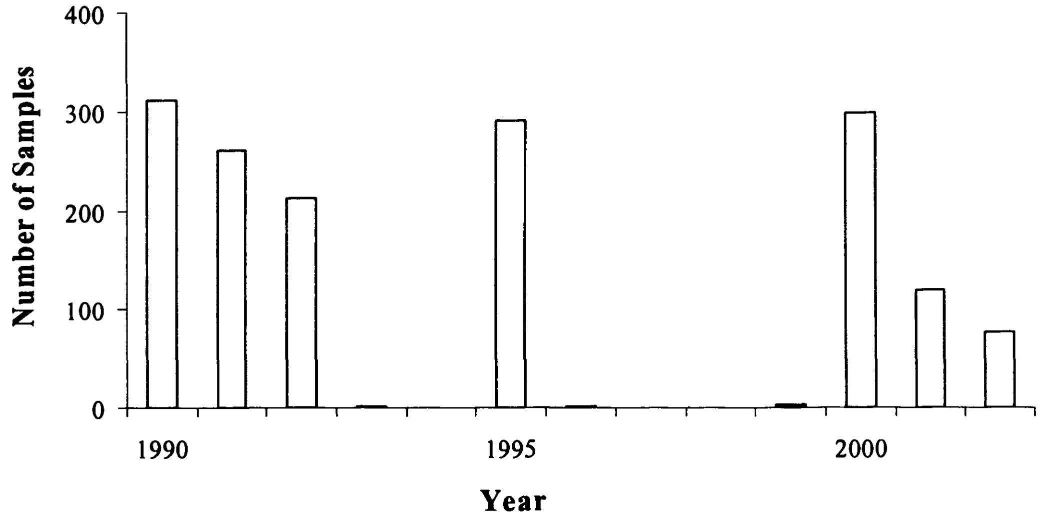

Figure 3.2. The number of invertebrate surveys undertaken on tributaries of the River Wye in each year between 1990 and 2003.

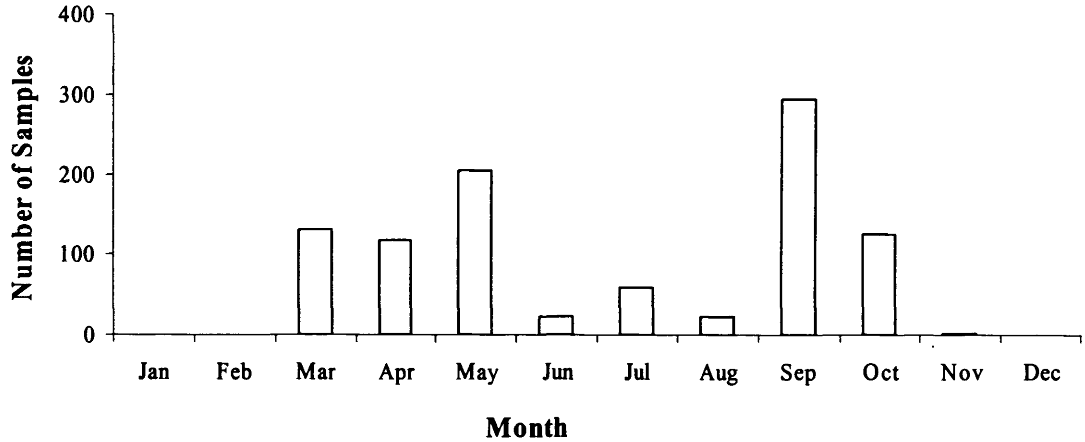

Figure. 3.3. The number of invertebrate samples taken from tributaries of the Wye in each month during the years 1990, 1995 and 2000.

The Acid Water Indicator Community (AWIC) score is currently based on the same taxa and resolution as the BMWP system (Davy-Bowker *et al.*, 2005). On a scale of one to six, high scoring taxa are those most sensitive to acidity (Appendix III). The AWIC score is the mean score of all taxa present in the sample.

Following the initial definition of feeding guilds among invertebrates (detailed in Cummins, 1973; 1974; Cummins and Klug, 1979 and Merritt and Cummins 1984), Moog (1995) redefined a scoring system for functional feeding guilds for the families and genera of Austrian aquatic organisms. Taxa present in each sample form the Wye were given scores to represent their role as shredders (SHR), grazers (GRA), active filterers (AFIL, e.g. Bivalvia), passive filterers (PFIL, e.g. Philopotamidae), detritivores (DET), miners (MIN), xylophages (XYL), predators (PRE) parasites (PAR) and others (OTH) (Appendix II).

Since invertebrates were identified to family and Moog (1995) assigned feeding preference by genera or species, representative scores for each family were calculated. Three different average scores were calculated; the median, mean and a mean weighted by the frequency of occurrence of species during a comprehensive survey of the Wye catchment in 1982 (Ormerod, 1987). There was no difference between scores generated by each method (One way ANOVAs: $P>0.05$ $df_{2,219}$). So, the median score for each family was applied because it was independent of earlier studies in the Wye and it represented the most common score for that family (Appendix III).

The feeding guilds SHR, GRA, PFIL, DET and PRE each accounted for more than one percent of the taxa recorded so were used in further analyses and transformed to reduce skew and kurtosis of the data (SPSS for Windows, 2001; Tabachnick and Fidell, 2001).

Ephemeroptera, Plecoptera and Trichoptera ('EPT') represent key orders of aquatic invertebrates (Ormerod *et al.,* 1993; Wallace *et al.,* 1996). Abundance categories were summed for each order and that score used to approximate the abundance of Ephemeroptera, Plecoptera and Trichoptera respectively.

Chemical data were obtained from the UK Environment Agency's Water Management Information System (WMIS) and historical archives of water chemistry from the river Wye catchment. Monthly ammoniacal nitrogen (mg/l), dissolved oxygen (DO mg/l), Biochemical Oxygen Demand (BOD mg/l), and pH data were available for 94% of invertebrate sites surveyed in 1990, 1995 and 2000 (Figure 3.1).

For those sites, determinand means were calculated for seasons antecedent to invertebrate surveys.

Analysis by the Environment Agency and its predecessors involved standard spectrophotometric methods (SCA, 1978, 1979, 1980, 1981, 1987, 1988; Environment Agency, 2001). Briefly, the methods were as follows: ammoniacal nitrogen ('total ammonia') was the total concentration of ammonium salts (NH_4^+) and free ammonia. (NH_3), derived form the reaction of ammonia with hypochlorite to form monochloramine, in turn measured as a blue indo-phenol complex measured colorimetrically at 656nm. Biochemical Oxygen Demand was analysed in the presence of allylthiourea (ATU) to suppress the uptake of oxygen by ammonia (SCA, 1981, 1988; Environment Agency, 1997). The ATU nitrification suppression variant of the BOD_5 test measured the uptake of dissolved oxygen by the sample during 5 days at 20°C in the dark. Dissolved oxygen concentration was determined from the partial pressure of oxygen in equilibrium with the water at the membrane surface, measured by an electrochemical oxygen sensor covered with a gas-permeable membrane (SCA, 1979). PH was measured electrometrically with a glass electrode in the laboratory (SCA, 1978).

Mean values for each chemical determinand were calculated for the 'winter' (September - February) and 'summer' (March - August) antecedent to each invertebrate sampling period (spring or autumn) after identifying outliers. For each chemical determinand, standard deviates (z scores) from the overall mean were calculated. Samples with standard deviates greater than 5 were removed (Sokal and Rohlf, 1995). Values recorded at the lower limit of detection were given a value of half the detection limit (Sowerby and Brook, 2001). If detection limits reduced through time, the lower detection limit was set to that of the earliest samples in the time series. Missing values were replaced where possible with the mean for the appropriate year, season and section of river; otherwise data were excluded from analysis.

All Geographical and topographical variables were extracted from spatial data layers using the ArcView® Geographical Information System (GIS) package (Version 9, ESRI Inc., California) and analyses were performed in SPSS (Version 11.5, SPSS Inc., Chicago) (see Table 3.1).

Table 3.1. Catchment attributes used to describe local river characteristics within the Wye catchment, Mid-Wales.

Catchment Attribute	Units
Strahler stream order	
Altitude	m OD
Buffering capacity	Low (1), Intermediate (2), High (3)
Slope	m/km
Broad-leaved woodland	km^2 within a 500m buffer
Coniferous woodland	km^2 within a 500m buffer
Neutral grassland	km^2 within a 500m buffer
Calcareous grassland	km^2 within a 500m buffer
Acid grassland	km^2 within a 500m buffer
Arable and horticultural	km^2 within a 500m buffer
Improved grassland	km^2 within a 500m buffer
Built up areas, gardens	km^2 within a 500m buffer
Summer / Winter	Sept - Feb / Mar – Aug

The slope (m/km) of each stream was obtained from the Environment Agency's site registration information (Environment Agency, 1999). Strahler stream orders were obtained from a river network GIS layer supplied by the Centre for Ecology and Hydrology (CEH) at the 1:50 000 scale (Richards, 1982). The altitude of each site was estimated to the nearest 5m above sea level from contours derived from the Ordnance Survey's profile digital elevation model (DEM). The underlying solid geology of each site was ranked according to its buffering capacity; the Ordovician system was classed as a 'low buffer', Silurian as an 'intermediate buffer' and Devonian as 'high buffer' (Hornung *et al.*, 1990; Environment Agency, 1998; Appendix I). Geology surveys were undertaken at the 1: 250000 scale by the British Geological Society and distributed as vector data.

Land use classes were attained from a raster layer of the CEH 'Land Cover Map 2000' (LCM2000) survey. The area under each land use was calculated to the nearest $25m^2$ within 500m buffers around each sample point (each with an area of approximately 0.8km) using the spatial analyst extension for ArcView (version 9) (Fuller *et al.*, 2000). LCM2000 defined 16 'target' classes, comprised of 26 subclasses of land use that were represented by $25m^2$ cells (Appendix IV). More generalised 'target' classes (e.g.) were used for analysis when constituent 'subclasses' were under-represented (i.e. <2% of land use within 500m buffers around sample points).

Target classes that accounted for less than 1% of the land use (e.g. Bracken) were not included in analyses. Land use types used for analysis were Broad-leaved woodland, Coniferous woodland, Neutral grassland, Calcareous grassland, Acid grassland, Arable and horticultural land, Improved grassland and Built up areas (Table 3.1; Appendix IV).

[3.3.2] Data analysis

Principal Component Analysis was used to reveal gradients in macroinvertebrate assemblage structure across the whole of the Wye catchment. PCA was chosen for this purpose because taxonomic change across ordered sites was linear, rather than unimodal (ter Braak and Šmilauer, 1998). Scores were derived from all data available on the BIOSYS database for the Wye catchment. PC scores were then extracted for the surveys undertaken in 1990, 1995 and 2000 and treated as metrics to represent the structure of invertebrate communities.

Variation in biotic metrics between sub-catchments (1, 2 and 3) and years (1990, 1995 and 2000) and interaction terms were revealed through a multivariate, crossed ANOVA, using the GLM. Interactions between years and sub-catchments were further investigated through Tukey pair-wise comparisons and one-way ANOVAs between sub-catchments for each year. River reaches with replicates in 1990, 1995 and 2000 were used in spatio-temporal examination of the data. Often only one site was sampled within a river, but larger rivers had multiple sample sites and comparable reaches were defined as lengths of river with similar Strahler stream order, altitude and geological system. For sites without data in one season, metrics were replaced by mean for that year, season and reach where possible otherwise they were excluded from analyses, thus balancing the number of samples in each year.

The environment of the Wye catchment was characterised by a Principal Component Analysis (PCA) of physico-chemical variables. Environmental differences between the three sub-catchments of the Wye were then identified through a multivariate one-way ANOVA on catchment attributes and water chemistry with streams and years as samples.

Catchment attributes that best explained variation in biotic indices were identified through linear, logarithmic, quadratic and cubic regression. Biotic indices that demonstrated spatial variation across the Wye were regressed against PC axes representing major gradients in geography, topography and chemistry.

All analyses were repeated separately for spring and autumn samples to account for the seasonal nature of macroinvertebrate communities and water chemistry. Results were similar for both seasons. So, only spring surveys are presented here. Analyses were performed in SPSS (version 11.5) and CANOCO for Windows (version 4) with Bonferroni adjustment for multiple tests (ter Braak and Šmilauer, 1998; SPSS for Windows, 2001; Quinn and Keough, 2002). Data were transformed to reduce skew and kurtosis of the data where appropriate. When data were reciprocally transformed ($^{1}/_{[n+1]}$) the sign (+/-) of each value was reversed before Principal Components Analyses (SPSS for Windows, 2001; Tabachnick and Fidell, 2001).

[3.4] Results

[3.4.1] Environmental conditions

Water chemistry, altitude, and land use varied across sub-catchments (Table 3.2). Ordination of environmental variables revealed an altitudinal gradient from steep, upland, acid, wooded reaches to lowland, arable streams with moderately increased BOD and ammonia (Figure 3.4; Table 3.3). Altitude, acidity, slope and the proportion of acid grassland and conifer trees all declined south eastwards from the upper catchment. By contrast, ammoniacal nitrogen, BOD and the proportion of arable farming were greater in the lower catchment. Improved grassland and broadleaved woodland were most extensive in the mid-Wye. Dissolved oxygen (DO) was least in the lower catchment in winter, but did not demonstrate any significant trend in the summer (Table 3.2). Independent of these sub-catchment divisions, there was also a trend from small streams with high ammoniacal nitrogen to large tributaries with relatively high dissolved oxygen (Figure 3.4; Table 3.3).

[3.4.2] Biotic indices and assemblage composition

Biotic indices varied between sub-catchments and demonstrated strong altitudinal gradients down the catchment (Table 3.4; Figures 3.5 and 3.6). For example, BMWP scores were significantly reduced respectively in the acidified upper and lower catchments, reflected also in low ASPT in the lower catchment and reduced taxon richness in the upper catchment (Figures 3.5 and 3.6). AWIC scores declined in the acidified upper catchment (Figure 3.5).

Biotic indices mirrored sub-catchment differences in assemblage and guild composition (Table 3.4; Figures 3.5 and 3.6). Variations among invertebrates closely followed catchment character (Figures 3.7). Assemblages on PC1 varied from those comprised of typical families from upland, fast-flowing streams (Baetidae, Heptageniidae, Hydropsychidae, Lepidostomatidae and Leuctridae) to lowland families such as Asellidae, Planorbidae and Sialidae (Figure 3.7; Table 3.5). On PC2, there was an increase in acid tolerant Perlodidae, Nemouridae and Leuctridae and a decline in populations of acid-sensitive taxa (Gammaridae, Hydrobiidae and Sphaeriidae) (Figure 3.7). PC scores on both these axes, together with feeding guild composition differed between sub-catchments. PC1 demonstrated that communities in the lower catchment were indeed typical of lowland habitats, while PC2 scores declined with acidity downstream. Shredders, grazers and predators were more represented in the uplands while filtering and detritivorious taxa tended to increase in the lower catchment (Table 3.4; Figure 3.5).

[3.4.3] Variations through time

In contrast to strong variations between locations, there were no trends in invertebrate composition through time. Assemblage composition was apparently stable between 1990, 1995 and 2000, and there were only minor variations among some feeding guilds (Table 3.4). Significant interactions between catchment and year were also minor, although there was a reduction in taxon richness in the upper Wye in 1995 and 2000 by comparison with 1990, when values were similar in all three sub-catchments (one-way ANOVA; Tukey pair-wise comparisons, $P<0.05$).

Table 3.2. Variation in environmental variables (mean ±SE) between three sub-catchments of the Wye (GLM; df=2,84; *P<0.05; **P<0.01, and ***P<0.001). Pair-wise differences are indicated by sub-catchment numbers in superscript (P>0.05). Areas under different categories of land use are given as area (km^2) within buffers of 500m radius around each sample point (area of each buffer was approx. $0.8km^2$).

	Sub-catchment			
Catchment Attribute	**Upper**	**Middle**	**Lower**	**F**
Strahler stream order	3.9 (0.3)	3.7 (0.2)	3.4 (0.1)	1.0
Altitude m OD	192 (24) ^{2,3}	143 (14)) ^{1,3}	79 (4) ^{1,2}	**36.6*** **
Buffering capacity	1.8 (0.1) ^{3}	2.1 (0.1)	2.4 (0.1) ^{1}	**7.9***
Slope m/km	5.9 (0.8) ^{3}	10.2 (2.0) ^{3}	4.0 (0.5) ^{1,2}	**7.3***
Broadleaved woodland km^2	0.074 (21) ^{2}	0.129 (20) ^{1,3}	0.060 (7) ^{2}	**9.2****
Coniferous woodland km^2	0.042 (25)	0.029 (8) ^{3}	0.011 (3) ^{2}	**7.2***
Neutral grassland km^2	0.041 (20)	0.008 (5)	0.035 (8)	1.4
Calcareous grassland km^2	0.050 (29)	0.092 (40)	0.048 (8)	0.9
Acid grassland km^2	0.131 (52) ^{3}	0.006 (3) ^{3}	0.000 (0) ^{1,2}	**22.5*** **
Arable and horticultural km^2	0.016 (6) ^{3}	0.049 (11) ^{3}	0.248 (23) ^{1,2}	**17.9*** **
Improved grassland km^2	0.283 (72) ^{2}	0.385 (42) ^{1,3}	0.266 (18) ^{2}	**7.0***
Built up areas, gardens km^2	0.063 (26)	0.028 (10)	0.052 (10)	0.1
NH_4 as N mg/l summer	0.022 (0.006) ^{3}	0.031 (0.004) ^{3}	0.074 (0.01) ^{1,2}	**7.6***
BOD mg/l summer	0.94 (0.09) ^{3}	1.22 (0.10)	1.42 (0.07) ^{1}	**10.2****
DO mg/l summer	10.52 (0.1)	10.2 (0.2)	9.9 (0.1)	2.9
PH summer	7.0 (0.2) ^{2,3}	8.0 (0.1) ^{1}	8.1 (0.0) ^{1}	**71.7*** **
NH_4 as N mg/l winter	0.023 (0.007) ^{3}	0.031 (0.005) ^{3}	0.099 (0.012) ^{1,2}	**10.1****
BOD mg/l winter	0.87 (0.17) ^{2,3}	1.25 (0.08) ^{1,3}	1.59 (0.07) ^{1,2}	**20.4*** **
DO mg/l winter	10.9 (0.2) ^{3}	10.6 (0.2) ^{3}	9.8 (0.1) ^{1,2}	**13.7*** **
pH winter	6.9 (0.2) ^{2,3}	7.7 (0.1) ^{1,3}	7.9 (0.0) ^{1,2}	**71.8*** **

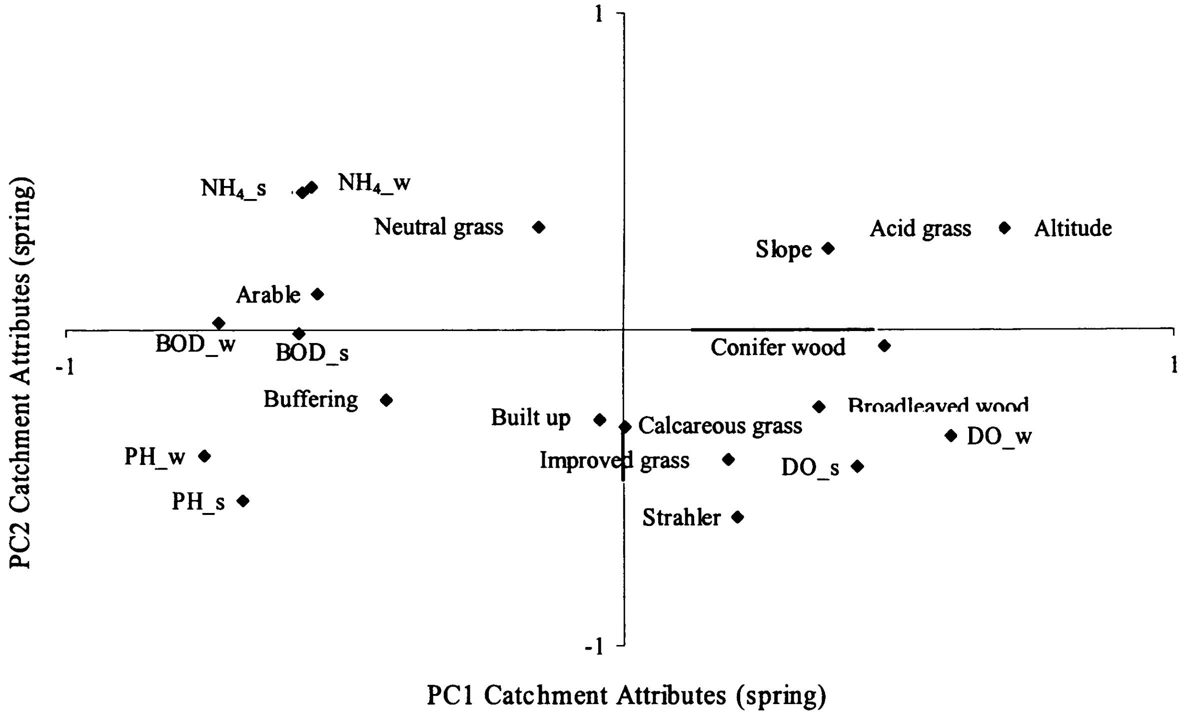

Figure 3.4. Major sources of environmental variation betweens sites sampled in spring, demonstrated by the first and second Principal Component axes. Seasonal means of chemical determinands were calculated for the summer ('_s') and winter ('_w') antecedent to the spring sampling period (see Table 3.1 for full details of catchment attributes and Table 3.3 for analysis).

Table 3.3. Loading of catchment attributes on Principal Component axes describing major sources of environmental variation between sites. Summer ('_s') and winter ('_w') means of chemical determinands were from seasons antecedent to the sampling periods in spring and in autumn. Correlation coefficients greater than 0.5 are highlighted in **bold.**

	Loading of catchment attributes on principal component axes.					
	Autumn			**Spring**		
	PC1	**PC2**	**PC3**	**PC1**	**PC2**	**PC3**
% of Variance	26.9	11.2	9.9	26.0	11.7	10.3
Environmental Variable						
Strahler stream order	0.171	**-0.675**	-0.056	0.208	**-0.596**	-0.006
Altitude	**0.782**	0.287	0.127	**0.753**	0.303	0.210
Buffering	-0.464	-0.176	-0.110	-0.426	-0.226	-0.181
Slope	0.358	0.410	0.148	0.369	0.256	0.058
Broadleaved woodland	0.327	-0.114	0.354	0.355	-0.248	0.366
Conifer woodland	0.490	0.033	0.197	0.473	-0.055	0.197
Neutral grassland	-0.140	0.328	-0.495	-0.153	0.320	-0.451
Calcareous grassland	-0.014	-0.295	**0.640**	0.004	-0.310	**0.634**
Acid grassland	**0.712**	0.293	0.002	**0.691**	0.316	0.059
Arable	**-0.550**	0.097	-0.304	**-0.549**	0.114	-0.255
Improved grassland	0.158	-0.328	**0.690**	0.191	-0.411	**0.625**
Built up	-0.035	-0.229	0.323	-0.042	-0.289	0.329
NH_4_s	-0.493	0.333	0.125	**-0.561**	0.449	0.260
BOD_s	**-0.668**	0.093	0.204	**-0.584**	-0.012	0.164
DO_s	0.412	-0.476	-0.395	0.423	-0.437	-0.450
PH_s	**-0.714**	-0.445	-0.111	**-0.684**	-0.542	-0.120
NH_4_w	**-0.568**	0.483	0.209	**-0.575**	0.432	0.275
BOD_w	**-0.740**	0.103	0.275	**-0.724**	0.022	0.298
DO_w	**0.561**	-0.368	-0.308	**0.593**	-0.337	-0.367
PH_w	**-0.798**	-0.344	-0.056	**-0.752**	-0.402	-0.139

Table 3.4. Variation in biotic indices and invertebrate composition between sub-catchments (1, 2 and 3) and years (1990, 1995 and 2000) in the Wye catchment during spring showing also significant interactions (*P<0.05; **P<0.01, ***P<0.001). See Figure 3.5 for graphical display. Principal Components (PC1 and PC2) were derived from Principal Component Analyses of invertebrate families sampled in spring (see Figure 3.7).

	Sub-catchment	Year	Interaction
BMWP	16.6***	0.6	4.9*
ASPT	60.2***	0.6	2.0
No. TAXA	11.4***	0.2	5.3**
AWIC	106.9***	0.3	1.3
SHR	19.9***	2.2	1.0
GRA	26.1***	8.6*	0.4
PFIL	4.6	1.9	2.6
DET	1.3	1.9	1.7
PRE	38.3***	7.3*	5.1**
Ephemeroptera	16.9***	3.1	0.6
Plecoptera	90.0***	2.1	0.9
Trichoptera	5.4	5.2	2.9
PC1	55.6***	0.6	1.4
PC2	92.0***	0.7	4.2

$df_{\text{sub-catchment, year, sub-catchment * year, error}} = 2, 2, 4, 30$

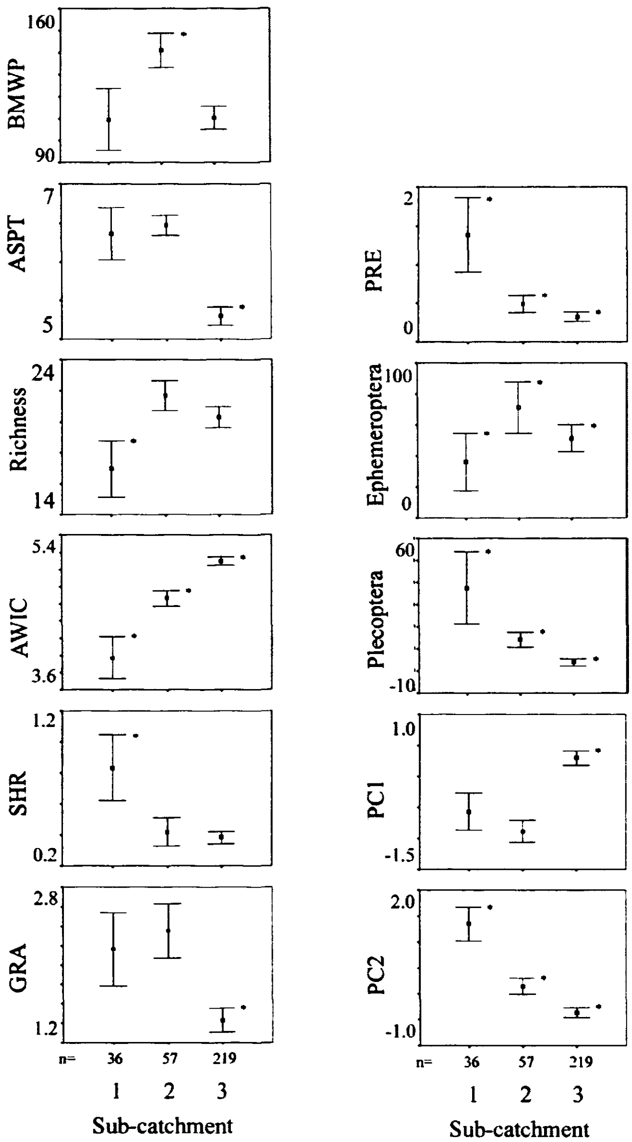

Figure 3.5. Pairwise comparisons of biotic indices and invertebrate assemblage composition between upper (1), middle (2) and lower (3) sub-catchments of the Wye (see Figure 3.1), for the spring sampling period. Sub-catchments for which index scores differed are are indicated by * ($P<0.05$). Principal Components (PC1 and PC2) were derived from Principal Component Analyses of invertebrate families sampled in spring (see Figure 3.7).

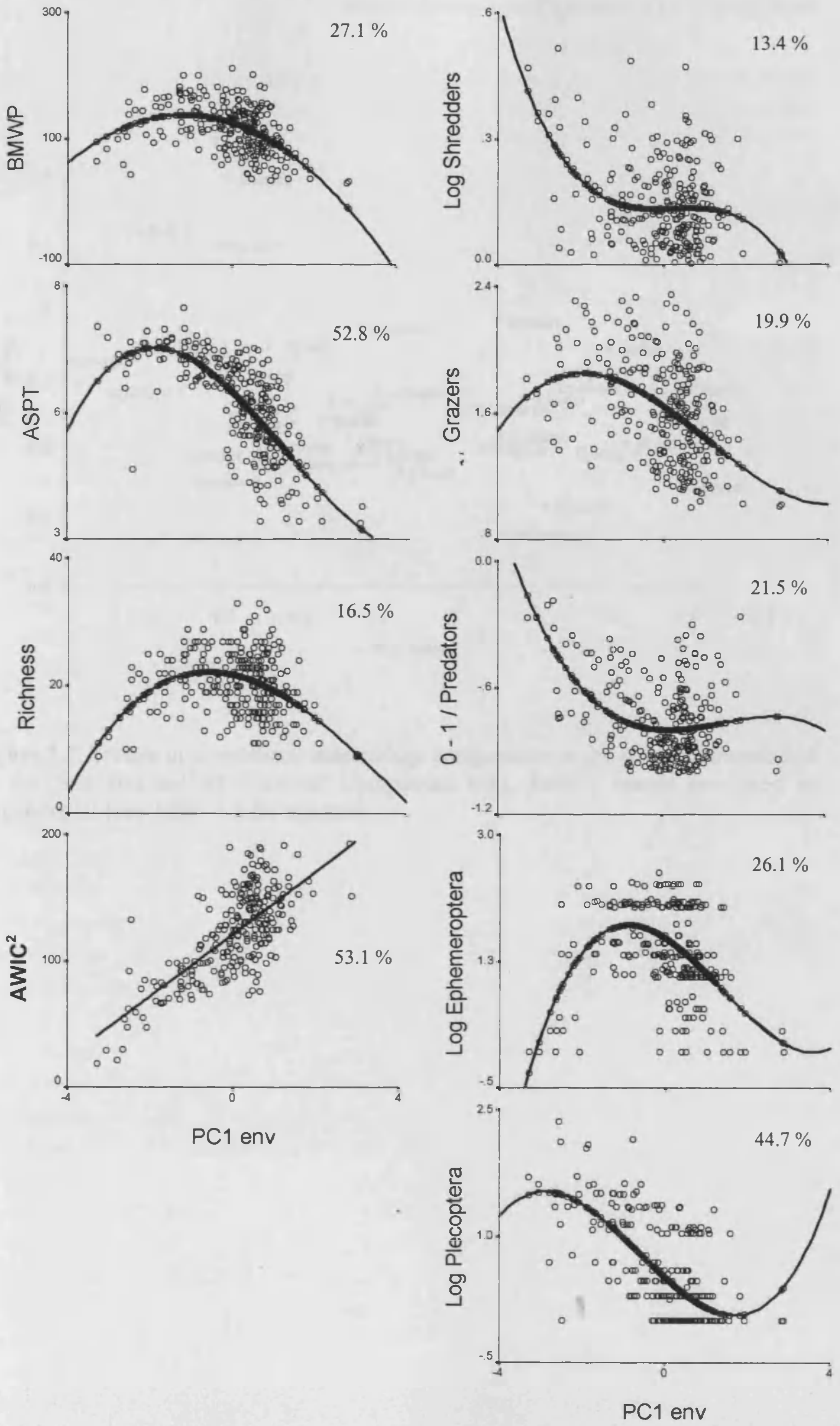

Figure 3.6. Regression of biotic metrics against the major axis of environmental variation (PC1 env) in the Wye catchment. 'PC1 env' was directly proportional to altitude. The percentage of variance explained (%R^2) by each regression is indicated ($P<0.05$).

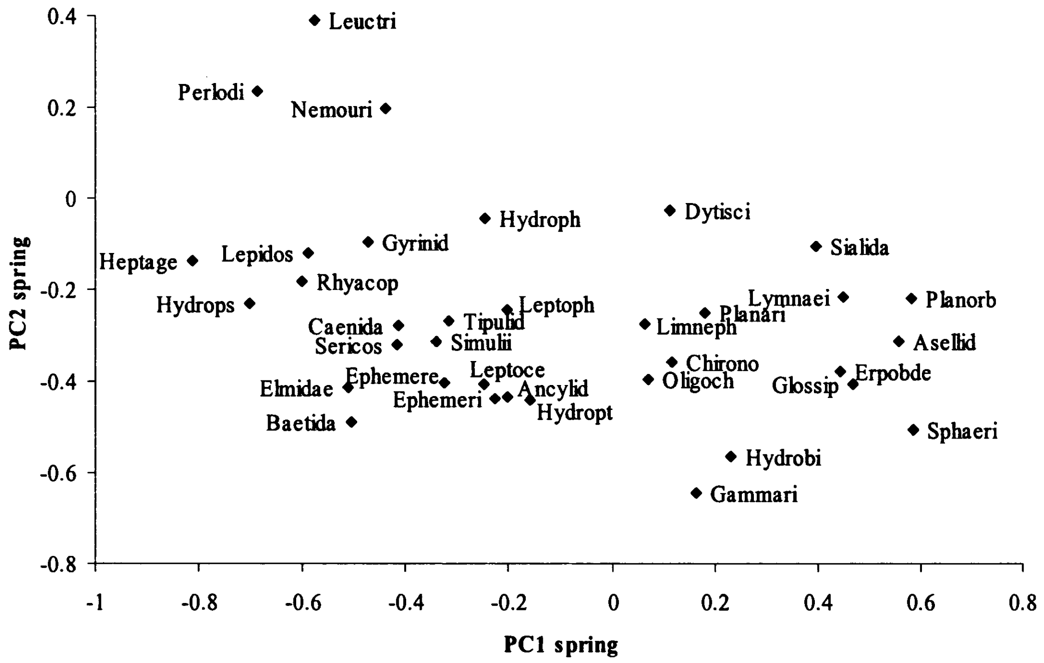

Figure 3.7. Trends in invertebrate assemblage composition in the spring demonstrated by the first and second Principal Component axes. Family names are given in Appendix III (see Table 3.5 for analysis).

Table 3.5. Loading of commonly occurring invertebrate families on Principal Component axes describing major sources of variation in invertebrate communities sampled in the autumn and spring respectively. Correlation coefficients greater than 0.5 are highlighted in **bold.**

	Autumn		Spring	
	PC1	PC2	PC1	PC2
% of Variance	18.9	10.7	18.5	11.6
Family				
Ancylidae	-0.431	-0.405	-0.200	-0.434
Asellidae	**0.503**	-0.413	**0.556**	-0.315
Baetidae	**-0.614**	-0.272	**-0.507**	-0.490
Caenidae	-0.144	-0.201	-0.415	-0.281
Chironomidae	-0.058	-0.267	0.118	-0.358
Dytiscidae Noteridae	0.255	-0.227	0.115	-0.027
Elmidae	**-0.526**	-0.448	**-0.512**	-0.415
Ephemeridae	-0.375	-0.272	-0.324	-0.402
Ephemerellidae	-0.152	-0.340	-0.225	-0.439
Erpobdellidae	0.316	**-0.527**	0.444	-0.379
Gammaridae Crangonyctidae	0.068	**-0.579**	0.166	**-0.643**
Glossiphoniidae	0.494	-0.487	0.468	-0.408
Gyrinidae	-0.422	-0.110	-0.471	-0.098
Heptageniidae	**-0.778**	-0.067	**-0.813**	-0.137
Hydrobiidae Bithyniidae	0.071	**-0.632**	0.232	**-0.566**
Hydrophilidae Hydraenidae	-0.341	-0.097	-0.244	-0.046
Hydropsychidae	**-0.734**	-0.288	**-0.701**	-0.232
Hydroptilidae	-0.221	-0.431	-0.159	-0.442
Lepidostomatidae	**-0.635**	-0.132	**-0.590**	-0.121
Leptoceridae	-0.119	-0.266	-0.247	-0.408
Leptophlebiidae	-0.426	-0.029	-0.202	-0.245
Leuctridae	**-0.645**	0.226	**-0.577**	0.391
Limnephilidae	-0.056	0.005	0.067	-0.275
Lymnaeidae	0.203	-0.441	0.451	-0.217
Nemouridae	-0.332	0.111	-0.439	0.197
Oligochaeta	-0.072	-0.303	0.071	-0.398
Perlodidae	**-0.514**	0.277	**-0.688**	0.236
Planariidae Dugesiidae	0.069	-0.266	0.183	-0.252
Planorbidae	**0.540**	-0.344	**0.582**	-0.222
Rhyacophilidae Glossosomatidae	**-0.592**	-0.037	**-0.602**	-0.184
Sericostomatidae	**-0.518**	-0.330	-0.417	-0.322
Sialidae	**0.534**	-0.180	0.399	-0.107
Simuliidae	**-0.563**	-0.160	-0.338	-0.313
Sphaeriidae	0.478	**-0.574**	**0.586**	**-0.508**
Tipulidae	-0.477	-0.211	-0.314	-0.269

[3.5] Discussion

This study supported previous work in distinguishing ecological differences among regions within the Wye catchment (Edwards and Brooker, 1982; Ormerod and Edwards 1987; Ormerod 1987, 1988; Environment Agency, 1998; Brennan *et al.*, 2003; Jarvie *et al,*. 2003). More importantly, it showed that biotic metrics track the variations across these contrasting catchments, revealing potential management problems: sub-catchments varied in acidity and trophic status, to which different biotic indices responded diagnostically.

In the acidic, oligotrophic environment of the upper Wye acid tolerant orders such as Plecoptera dominated. Plecopteran families such as Perlodidae, Nemouridae and Leuctridae are tolerant of acidification, whereas many ephemeropterans, such as Ephemerellidae and Ephemeridae are highly sensitive to acidification (Elliott *et al.*, 1988; Ormerod, 1992; Davy-Bowker *et al.,* 2005; Appendix III). The BMWP and ASPT scores responded to reductions in ephemeroptera but alone could not diagnose acidification issues in the upper Wye. The new AWIC score was able to clarify the acidification issue. The AWIC score compared favourably with direct pH measurement as found by Ormerod *et al.* (2006) who also demonstrated that AWIC was indicative of calcium concentration, alkalinity and metals toxic at low pH, such as aluminium. AWIC, BMWP and ASPT results described here demonstrate how, when used together, indices reveal more detail of the underlying structure of invertebrate communities. This adds weight to the argument for the use of multiple indices in diagnosis of management issues (Griffith *et al.*, 2003).

The upper Wye has low concentrations of base-cations and high acidity (Edwards and Brooker, 1982; Ormerod and Edwards, 1987). Waters running though the mineral-poor Ordovician Rock of the upper Wye are susceptible to acidification because the base-poor rocks and soils provide little buffering (Appendix I; Reuss and Johnson, 1985; Hornung *et al.*, 1990; Ormerod, 1992). These data reveal that acidification problems are still a widespread feature of the upper catchment, with little sign of recovery so far (see also Lewis *et al.*, 2007 *in press*). Catchment-scale management options are available to treat acidification symptomatically – for example additions of calcium carbonate using various methods (Weatherley *et al.,* 1995; Svenson *et al.,* 1995). The liming of entire catchments is a relatively new technique that aims to increase the pH of runoff before it reaches watercourses (Rundle *et al.,* 1995;

Ormerod *et al.,* 1990; Bradley and Ormerod, 2002b). In the Wye, liming of wetlands within the most acidified areas of the catchment began in 2001 (Borg *et al.*, 1995; Gunn *et al.*, 2001; Lewis, 2006).

Biotic indices showed that invertebrates in the lower Wye catchment were more typical of eutrophic waters, consistent with moderately elevated BOD and ammonia concentrations and the increased proportion of arable land. Indices applied here, however, did not account for river 'type'. Environmental Quality Indices (EQI) have been developed for BMWP, AWIC and richness scores and are proposed for the AWIC index that weight scores according to river 'type'. Cluster analyses is used to classify river reaches according primarily to size and distance along the river continuum (Vannote, 1980; Wright *et al.*, 1998). This enables the prediction of 'expected' biotic scores and comparison of observed scores against them. Water quality is then assessed based on the comparison of observed against expected scores for each river type, according to the RIVPACS (River InVertebrate Prediction And Classification System) model (Wright *et al.*, 1998). Further examination of indices in the lower catchment might therefore be prudent to assess whether water quality in the lower Wye catchment is any different from what might be expected for a river in its lower stages.

Elevated BOD and ammonia are often attributed to arable farming, the predominant land-use within the lower catchment (e.g. Johnes, 1996; Arheimer and Lidén, 2000; Smith *et al.*, 2001; Donohue *et al.,* 2006). In the Windrush catchment in the English Cotswolds, cereal cultivation was one of the greatest contributors to the total loads of nitrogen and phosphorus in the river (Johnes, 1996). In the Wye, catchment-wide trends associating stream order with BOD and ammonia suggested that small streams were unable to dilute ammoniacal inputs. Perhaps special attention should be paid to nutrient loading onto fields in close proximity to smaller streams.

In the lower catchment, nutrient enrichment has been a persistent concern over the last 25 years. In 1982, Edwards and Brooker identified the greatest nutrient concentrations downstream of Hereford and implicated the arable lands of the Lugg basin in the nutrient-rich character of the lower Wye catchment. Later, in 1998, the lower Wye received sewage discharge at Hereford and Ross in excess of 160,000 population equivalent as well as farm effluents and diffuse inputs nutrient from agricultural land (Environment Agency, 1998; Environment Agency, 2000). Designation of the River

Wye from Hereford as a 'eutrophic sensitive area' under the Urban Wastewater Treatment Directive ('UWWTD', Directive 91/271/EEC) improved phosphate removal at sewage treatment works (Environment Agency, 2000). Despite this and the classification of the Wye and its tributaries as Sites of Special Scientific Interest (SSSIs) and a Special Area of Conservation (SAC), nutrient concentrations remain highest in the lower catchment (Jarvie *et al,*. 2003; CCW, 2006a-c; JNCC, 2006). As reported elsewhere in the UK, results suggest that designation was not sufficient to protect water quality (Boon, 1995). A catchment-wide management plan should incorporate reduction of nitrogen and phosphate sources from agricultural land (Environment Agency, 2000; Mancini *et al.*, 2005). Although the BMWP and ASPT scores were designed to detect organic pollution they also respond to other pollutants and forms of impairment. For example, Garcia-Criado *et al.* (2002) found that the Spanish BMWP scores were similar to the tolerance levels of the taxa to coal mining. Chemicals used in sheep dip (e.g. diazinon and propetamphos) also reduce BMWP scores. This study BMWP, ASPT and Richness were reduced in the acidified uplands (DEFRA, 2002). More comprehensive analyses of water chemistry may confirm the impact of plant nutrients and highlight any other contaminants.

Diffuse sources of pollution are inherently more difficult to control than point-sources (e.g. Directive 91/676/EEC 1991). Many landscape-scale initiatives operate in an *ad hoc* manner and vary between catchments. Independent landowners gain grants from the (Welsh) 'Tir Gofal' and (English) 'Countryside Stewardship' schemes in return for managing their farms in an environmentally sensitive manner (Howell and Mackay, 1997; Ovenden *et al.,* 1998; Carey *et al.,* 2001; CCW, 2006d). Rivers Trusts bring landowners together with countryside agencies and scientists to conserve the river environment, often with a focus on salmonid fish. Additionally, in the Wye, the Environment Agency's 'PSYCHIC' model aims to identify sediment and nutrient loss within the catchment, while the 'Wyecare' collaboration of conservation agencies intends to address diffuse pollution (Environment Agency, 2000; English Nature, 2003; Environment Agency, 2003).

In contrast to the upper and lower catchments, tributaries of the Middle-Wye were relatively unpolluted: neither acidification nor organic enrichment were significant issues according to AWIC score (mean > 4.6) and BMWP scores (mean ASPT = 5, BMWP >130) (Hawkes, 1997; Davy-Bowker *et al.*, 2005). The BMWP scores reflect rich communities with an abundance of high-scoring ephemeropteran families (e.g. Caenidae, Baetidae and Ephemerellidae) were well represented which require well oxygenated waters and are often adapted to grazing (Merritt and Cummins 1984; Elliott *et al.*, 1988; Moog, 1995; Hawkes, 1997; Appendix III). Certainly, in terms of invertebrate communities and water chemistry the mid-Wye is of 'very good quality' (BMWP>130) suggesting that current management of the middle Wye is adequate (Pugh, 1997; Environment Agency, 1998). In terms of BOD and ammonia, the upper and middle catchments were in classed as having 'very good water quality', while the lower catchment attained 'good water quality' in terms of General Quality Assessment (Pugh, 1997). However, this study does not reveal the relevance of invertebrate-derived indices to other aspects of the aquatic ecosystem, such as other biotic groups (e.g. fish), physical habitat or ecosystem processes (e.g. production). For example, Griffith *et al.* (2005) found that fish, macroinvertebrates and periphyton differed in their sensitivity to stressors. Conversely, Balestrini *et al.* (2004) reported that the macroinvertebrate based 'Hilsenhoff index' and fish-derived indices of biotic integrity and diversity responded similarly to catchment land use and embeddedness. Surveys of 1874 stream sites across Idaho, USA demonstrated similar responses of macroinvertebrate metrics and salmonids to fine sediment while only macroinvertebrate biotic index scores decreased with increasing copper (Mebane, 2001). Beyond the structural organisation of aquatic communities, more complex information on ecosystem function may require the development of new, specific indices (Lecerf *et al.*, 2006).

Biotic metrics presented here were derived from routine monitoring of invertebrate communities which is undertaken throughout the UK by the Environment Agency (Environment Agency, 1999). Similar surveys are already undertaken throughout much of Europe and will be implemented throughout the continent under the Water Framework Directive (Directive 2000/60/EC). Under the WFD, 'River Basin Management Plans' will establish the ecological status of surface waters and detail programmes of measures to prevent deterioration and to 'protect, enhance and restore all bodies of surface water'.

Some authors, such as Donohue *et al.* (2006) suggest that once relationships have been established between catchment characteristics and biotic metrics, land-cover and chemistry data alone could establish the ecological status of rivers. However, biotic metrics provide direct measures of the ecological nature of rivers. Moreover, in establishing ecological status a baseline can be set against which to measure the outcomes of programmes of measures within river basins (Resh and Jackson, 1993). River restoration, such as liming or nutrient reduction can affect chemical change in the absence of biological change and, as yet, the mechanisms of recovery and the chemical and biological interactions involved remain unclear (Soulsby *et al.*, 1997; Stevens *et al.*, 1997; Raddum and Fjellheim, 2003; Jeppesen *et al.*, 2005). It is therefore important to include a measure of biological response to change and biotic indicators could serve this purpose. However, in summarising trends in macroinvertebrates, biotic indicators inevitably reveal only a part of the variation in communities (Resh and McElravy, 1993). Here, broad differences in communities were revealed at the sub-catchment scale but can indices also detect more subtle impacts within sub-catchments or rivers?

BMWP and ASPT are routinely used in biomonitoring. They can detect small-scale impairment within rivers and are often used up- and down-stream of a pollution incident (Jacobsen, 1998; Camargo, 1992; Perdikaki and Mason, 1999; Wright *et al.*, 2000; DEFRA, 2002). In contrast, the AWIC index is relatively new and there are few examples of its application (Ormerod *et al.*, 2006; Lewis *et al.*, 2007 *in press*). It may also suffer from some of the non-specifity issues surrounding the BMWP. Ormerod *et al.* (2006) point out that the score cannot differentiate between natural and anthropogenic acidity nor acidified (low pH, increased Al) and acid sensitive (low Ca, low alkalinity) streams. They suggest that increased taxonomic resolution of some families might improve the sensitivity of the index. This, along with recording of actual abundance of taxa may improve the ability of biotic indices to diagnose problems and evaluate management practice.

This chapter demonstrates that routinely collected macroinvertebrate data, identified only to family-level, can be usefully summarised as biotic indices that are able to detect sub-catchment variation in the ecological status of rivers. In combination, scores of BMWP, ASPT, AWIC and richness can reveal greater detail of invertebrate community structure. Multiple metrics can then aid diagnosis of problems such as acidification or enrichment within catchments. Here, the three regions of the Wye catchment require different management approaches; work is needed to ameliorate acidification in the upper Wye and reduce nutrient inputs to eutrophic reaches in the lower catchment while the semi-natural reaches of the middle catchment were largely unpolluted. However, the development of bio-diagnostic tools for identifying sedimentation and habitat degradation would further aid in identifying management priorities. Biotic indices provide an elegant means of presenting complex data sets in order to communicate the ecological status of rivers, establish priorities for management, define programmes of measures and evaluate restoration, all of which are set to be major themes in aquatic research in the next decade (Ormerod, 2004).

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APPENDIX I

Geological types of the Wye catchment, their buffering capacity and susceptibility to the acidification of freshwater, summarised from Hornung *et al.* (1990) and Environment Agency (1998).

Geology	Geological period	Buffering capacity	Susceptibility to acidification
Ashgill and Caradoc Upper Division	Ordovician	Little or no buffering capacity	HIGH
Llandovery beds	Silurian	Low buffering capacity	INTERMEDIATE
Ludlow and Wenlock beds	Silurian	Low buffering capacity	INTERMEDIATE
Lower Old Red Sandstone	Devonian	Greater buffering capacity Compared to Ordovician / Silurian.	LOW

APPENDIX II

Functional Feeding Guilds as defined by Moog (1995)

Feeding Type	Abbreviation	Sources of Food
Shredders	**SHR**	Fallen leaves, plant tissue, CPOM*.
Grazers, scrapers, raspers	**GRA**	Epilithic algal tissues, biofilm, partially POM*, endo and epilithic algal tissues, partially tissues of living plants.
Filtering collectors, active filter-feeders, eddy filterers	**AFIL**	Suspended FPOM*, CPOM*, prey. Food in water current is actively filtered. Suspended FPOM*, micro prey is whirled.
Passive filter feeders	**PFIL**	Food brought by flowing water current.
Detritus feeders (gathering collectors)	**DET**	Sedimented FPOM*.
Leaf borers, miners, piercers	**MIN**	Leaves of aquatic plants. Algae and cells of aquatic plants.
Xylophagous	**XYL**	Woody debris.
Predators	**PRE**	Prey.
Parasites	**PAR**	Host.
Other feeding types	**OTH**	Cannot be classified into this scheme.

(* F / C POM = Fine / Coarse Particulate Organic Matter)

Moog (1995) based this classification on previous classifications by Cummins (1973, 1974), Cummins and Klug (1979) and Merritt and Cummins (1984).

APPENDIX III

The Biological Monitoring Working Party (BMWP), Acid Water Indicator Community (AWIC) and Feeding Guild (adapted from Moog 1995; See Appendix II) scoring systems. Commonly occurring families are highlighted in **bold.**

Family	Family code	BMWP	AWIC	SHR	GRA	AFIL	PFIL	DET	MIN	XYL	PRE	PAR	OTI
Aeshnidae	1	8									1		
Ancylidae	2	6	6										
Aphelocheiridae	3	1									1		
Asellidae	4	3	6	2	2			2					5
Astacidae	5	8						3			1		4
Baetidae	6	5	6		5			5					
Brachycentridae	9	1			2		4				3		
Caenidae	10	8	6					1					
Calopterygidae	11	8	6								1		
Capniidae	12	1		6	2			2					
Chironomidae	13	2	4					1					
Chloroperlidae	14	1		1	1			2			6		
Coenagriidae	15	6									1		
Cordulegasteridae	16	8									1		
Corduliidae	17	8									1		
Corixidae	18	5	6					1					
Dendrocoelidae	20	5											
Dryopidae	21	5		5	3			3					
Dytiscidae / Noteridae	23										1		
Elmidae	24	5	6		1								
Ephemerellidae	25	1	6		5			5					
Ephemeridae	26	1	6			1							
Erpobdellidae	27	3	6								1		
Gammaridae / Crangonyctidae	28			1	1			2					1
Gerridae	29	5									1		
Glossiphoniidae	30	3	6										
Goeridae	32	1	4		9			1					
Gomphidae	33	8									1		
Gyrinidae	34	5	3								1		
Haliplidae	35	5											
Heptageniidae	36	1	6		5			5					
Hirudinidae	37	3										1	
Hydrobiidae / Bithyniidae	39												
Hydrometridae	40	5									1		
Hydrophilidae / Hydraenidae	41				3						3		
Hydropsychidae	42	5	4		2		5				3		
Hydroptilidae	43	6	6		5								5
Lepidostomatidae	45	1	2	2	5					3			
Leptoceridae	46	1	6	5	1								
Leptophlebiidae	47	1	6					1					
Lestidae	48	8									1		
Leuctridae	49	1	1	3	3			4					
Libellulidae	50	8									1		
Limnephilidae	51	8	4	5	2			1			2		
Lymnaeidae	52	3	6	3	4			1					2
Mesoveliidae	53	5									1		
Molannidae	54	1						3			7		
Naucoridae	55	5									1		
Nemouridae	56	8	1	4	3			3					
Nepidae	57	5									1		
Neritidae	58	6			1								
Notonectidae	60	5									1		
Odontoceridae	61	1	6	3	3			4					
Oligochaeta	62	1	6					1					
Perlidae	63	1	6		1						9		
Perlodidae	64	1	2		1						8		
Philopotamidae	65	8	6				1						
Phryganeidae	66	1		2	1			1			6		
Physidae	67	3	6	2	5			2					2
Piscicolidae	68	5	6									1	
Planariidae / Dugesiidae	69												
Planorbidae	70	3	6	2	6								2
Platycnemididae	71	6									1		
Pleidae	72	5									1		
Polycentropodidae	73	8	1				1				9		
Potamanthidae	74	1				1		9					
Psychomyiidae	75	8	6		8		1	1		1			
Rhyacophilidae / Glossosomatidae	76							2					
Scirtidae	77	5	6										
Sericostomatidae	78	1	4	1									
Sialidae	79	5	6								1		
Simuliidae	80	5	3		1		9						
Siphlonuridae	81	1						1					
Sphaeriidae	82	3	6										
Taeniopterygidae	83	1	6	3	5			3					
Tipulidae	84	5	4	7				3					
Unionidae	85	6				1							
Valvatidae	86	3	6	1				4					
Viviparidae	87	6			7	3							

APPENDIX IV

Land Cover Map 2000 class variants mapped on to 'Broad Habitats' from Fuller (2000). Broad habitats in bold are those used in analyses.

Broad Habitat (BH)	Class variants of subclasses, which can be combined into 16 target classes
1. **Broadleaved woodland**	deciduous, mixed, open birch, scrub
2. **Coniferous woodland**	conifers, felled, new plantation
4. **Arable & horticultural**	barley, maize, oats, wheat, cereal, cereal (winter), arable bare ground, carrots, field beans, horticulture, linseed, potatoes, peas, oilseed rape, sugar beet, unknown, mustard, non-cereal (spring), orchard, arable grass (ley), set-aside (bare), set-aside (undifferentiated)
5. **Improved grassland**	Intensive, grass (hay/ silage cut),grazing marsh
6. **Neutral grassland**	rough grass (unmanaged), grass (neutral / unimproved)
7. **Calcareous grassland**	calcareous (managed), calcareous (rough)
8. **Acid grassland**	Acid, acid (rough), acid with *Juncus,* acid *Nardus/Festuca/Molinia*
9. Bracken	bracken
10. Dwarf shrub heath	heath dense (ericaceous), gorse
11. Fen, marsh and swamp	Swamp, fen/marsh, fen willow
12. Bog	bog (shrub),bog (grass/shrub), bog (grass/herb), bog (undifferentiated)
13. Standing water/canals	water (inland)
15. Montane habitats	montane
16. Inland rock	Despoiled, semi-natural
17. **Built up areas, gardens**	suburban/rural developed, urban residential/commercial, urban industrial
18. Supra-littoral rock	rock
19. Supra-littoral sediment	shingle (vegetated), shingle, dune, dune shrubs
20. Littoral rock	rock, rock with algae
21. Littoral sediment	mud, sand
22. Inshore sublittoral	sea

Chapter 4

The effect of riparian land-use and management on river habitats in the Wye River system

[4.0] Abstract

As ecotones linking river habitats to their surroundings, the restoration of riparian zones is attracting increasing interest. However, there are questions about i) whether riparian zones can be sufficiently modified to alter riparian habitat and channel conditions; ii) whether riparian management efforts can override the effects of wider catchment pressures and iii) whether these effects can be detected quantitatively.

This chapter introduces a catchment-scale experiment to assess the impact of riparian habitat management in the Wye river system (Wales). In a hierarchically designed field survey, differences in river-habitat character were assessed between tributaries with riparian land-uses that were respectively: recently managed (n=9 streams), unmanaged control (n=12), intensive grazing pasture (n=3) and coniferous woodland (n=3). Riparian management activities along 'recently managed' streams aimed to exclude grazing, enable vegetation to develop and create potential refuges available for salmonids. Riparian zones along the 9 tributaries were coppiced and /or fenced between 1997 and 2003. Habitats were assessed using the Environment Agency's 'River Habitat Survey' (RHS), with habitat variation then quantified using Principal Components Analysis.

Streams draining intensive grazing pasture were characterised by finer substrata and more active channels than elsewhere. Streams within conifer plantations had more rocky substrata, and recently managed streams had less poaching and more filamentous algae compared to controls. Coppicing and riparian fencing appeared to have successfully excluded grazing on banks and increased in-stream vegetation, but substrata, flow type and channel features were not markedly different from control sites, intensive pastoral agriculture or conifer streams.

These data show that the RHS can detect habitat variation among streams with contrasting riparian land use. The effects of recent management in the Wye, although significant, are so far restricted either because insufficient time has elapsed since treatment or because land management within the wider catchment overrides effects at the riparian scale. The biological significance of these results is addressed elsewhere (Chapters 5 and 6).

[4.1] Introduction

The management of riparian zones of rivers to benefit biodiversity is an expanding area of research (Gregory *et al.,* 1991; Mulholland, 1992; Bren, 1993; Naiman *et al.,*1993; Goodwin *et al.,*1997; Naiman and Décamps, 1997; Tockner and Ward, 1999; Gippel and Collier, 2000; Lowrance *et al.,* 2000; Tabacchi *et al.,* 2000; McKergow *et al.,* 2003, also see Chapter 2). The aims are often to buffer aquatic environments against diffuse sources of pollution, such as plant nutrients and sediment, to stabilize banks and create more complex habitats (Goodwin *et al.*, 1997; McKergow *et al.*, 2003; Chapter 2). Techniques involve fencing to exclude livestock grazing from the river bank, management of riparian trees and exclusion of exotic species (Platts and Wagstaff, 1984; Lewis and Williams, 1994). Tree management often comprises a balance of planting, coppicing, thinning and pruning to maximize the structural diversity of the riparian vegetation in order to maximize habitat complexity and improve bank stability (García de Jalón, 1995; Hendry *et al.*, 2003). Such management recognizes the importance of riparian corridors to regional biodiversity in both riverine and adjacent terrestrial environments (Naiman et al., 1993). However, as ecotones linking the aquatic and terrestrial landscape, riparian zones influence the aquatic environment only within the context of the wider landscape (Naiman et al., 1993; Poff, 1997).

In upland environments in Wales, nutrient-poor soils typically support plantation forestry or, following soil treatments, pastoral agriculture (see Chapter 3; Ormerod *et al.*, 1989; Ormerod *et al.*, 1993; Parsons *et al.*, 2003; Liebault *et al.*, 2005). As forestry and agriculture within stream catchments become more intensive, their influence on the riparian environment become more pronounced (Martin et al., 2006). For example, streams draining conifers tend to have more exposed rocks and less riparian vegetation than streams in grassland (Rutt et al. 1989). Conifer plantations also alter the physical structure of the riparian zone, modify vegetation structure, change the hydraulic environment and enhance erosion and sedimentation (Ormerod *et al.*, 1993; Ormerod *et al.*, 1986). Intensive livestock grazing within a river catchment often contributes to increased sediment input to streams (Harding *et al.*, 1999; Zaimes *et al.*, 2004). Pasture land is less able to impede rainfall impaction or surface runoff than scrub or woodland, so that sediment is both more readily mobilized and less easily trapped on catchment surfaces during rainfall events. Increased velocity of overland flow over grassland enables further erosion. Livestock 'poaching' (trampling and churning of the ground) contributes additional sediment to rivers, often directly from the riparian zone. When livestock are allowed to graze in high densities on the river banks, direct destabilization of the banks is likely along side a reduction in the growth of vegetation which might normally give additional support to the banks (Easson and Yarbrough, 2002; Hook, 2003; Evans *et al.*, 2006). Deep, silted, homogenous channels can result (Whol and Carline, 1996; Hook, 2003; Kondolf, 1993).

In Mid-Wales, these effects of intensive riparian grazing were suspected of contributing to declines in salmonid catches (Environment Agency, 2003a, b). In response, riparian management work was instigated in 1997 within the River Wye catchment, and aimed to change the in-stream environment to augment habitats available to salmonid fish (see Chapters 1 and 2). Management was primarily undertaken in the upper and mid-Wye catchment, to the north-west of Hay on Wye in which the predominant land use was pastoral agriculture with exotic coniferous plantations in the northern and western extremities (see Chapter 3). Three key questions are i) whether these changes in the riparian zones are sufficient to alter riparian habitat and channel conditions; ii) whether riparian management efforts can override the effects of wider catchment pressures, and in particular how the effects of such recent management compare with longer-standing basin management and iii) whether these effects can be detected quantitatively.

One of the limitations to understanding the effects of riparian and catchment management on rivers is that effects can be difficult to appraise and quantify. Not only does the evaluation of riparian management require careful data collection, but effects on environments as structurally complex as rivers can be difficult to detect. The River Habitat Survey (RHS), developed to assess the habitat quality of rivers in the UK, could offer one option if the complex data that result could be analysed suitably (Raven *et al.*,1997; Raven *et al.*,1998). Raven *et al.* (2000) suggested that habitat features identified by the River Habitat Surveys, such as submerged tree roots and fallen trees, could be used in post-project appraisal to help predict and confirm the ecological consequences of river management. Moreover, Vaughan & Ormerod (2005) have recently developed methods of synoptically quantifying RHS data to investigate management effects.

However, RHS has rarely been applied to evaluate effects of land use and riparian management within a rural setting (Manel *et al.*, 2000; Davenport *et al.*, 2004). Recent management and contrasting land use within the Wye catchment presented an opportunity to test the scope for using RHS to detect variation in stream habitat with land use.

In this chapter, the River Habitat Survey was used to investigate links between riparian land-use and habitat structure in streams within the Wye catchment. In an attempt to contextualise the effects of recent riparian management, in-stream and riparian habitats were compared across four land use types that were respectively: recently managed riparian habitats (coppiced and fenced), control reaches with trees lining >20% of the (500m) reach length but without recent management, exotic coniferous plantations and intensively grazed pasture.

[4.2] Site description and management

The River Wye drains a large (4136km^2) rural catchment in mid-Wales, an area described in detail in Chapter 3 and by other authors (e.g. Edwards and Brooker, 1982; Ormerod and Edwards, 1987, Ormerod 1987, 1988; Environment Agency, 1998; Brennan *et al.*, 2003; Jarvie *et al.*, 2003). This study focuses on the upper and middle reaches of the Wye catchment (as defined in Chapter 3), which are dominated by pastoral agriculture with some coniferous plantations (Jarvie *et al.*, 2003; Chapter 3).

The Wye and Usk Foundation (WUF) began riparian management in the Wye catchment in 1996 and by 2004 had instigated work on 13 tributaries (orders 2-4) within the Wye catchment, which exceeded 1.5km on 9 tributaries (Figure 4.1). Sites selected for restoration by the WUF were those considered to be good salmon rivers or that had been popular with anglers in the past. Riparian management primarily took place on the lower reaches of tributaries. Management consisted of fencing a riparian buffer from the bank-top (typically to 1-3m from the stream) and coppicing and thinning of bank-side trees (Luxton, 2002). On the Clettwr, Duhonw, Bach Howey, Marteg and Triffrwd less than 33% of the river length was fenced, coppiced or both. On the Hafrena and Llynfi Dulas between 33 and 66% of the bank length was altered. Only the Clywedog and the Edw were almost entirely altered (Figure 4.1; exact locations confidential)

Fencing was designed to prevent poaching by livestock and to encourage the development of vegetation. The intention was to stabilize the banks and to reduce bed siltation (Krog and Hermansen, 1985; Hemphill and Bramley, 1989; Salmon Advisory Committee, 1991; Hendry and Cragg-Hine, 1997). Coppicing and thinning of deciduous trees on the banks was intended to reduce shading and increase the complexity of riparian and in-stream vegetation thus stabilizing banks and creating refuges for salmonids (O'Grady, 1993; García de Jalón, 1995; Growns *et al.*, 2003; Hendry *et al.*, 2003).

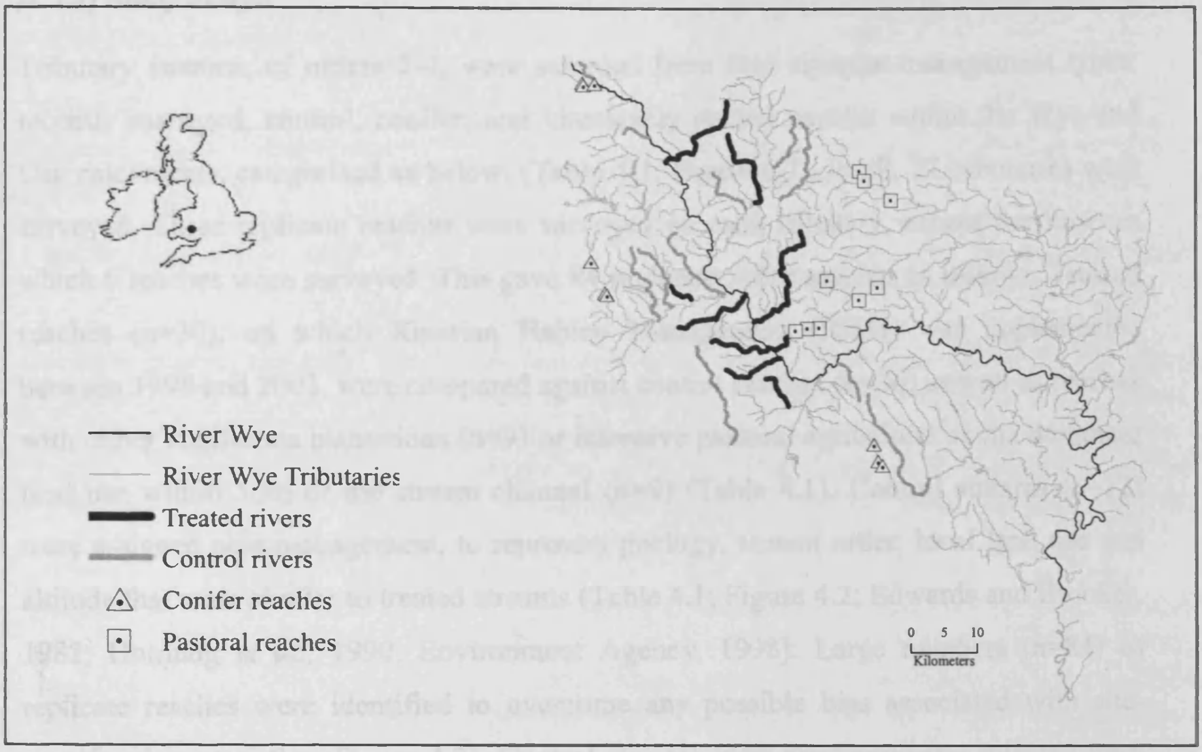


Figure 4.1. Recently treated, control, coniferous and intensive grazing pasture ('pastoral'). reaches of tributaries of the Wye on which River Habitat surveys were undertaken in 2004.

[4.3] Methods

[4.3.1] Study design

Tributary streams, of orders 2-4, were sampled from four riparian management types: recently managed, control, conifer, and intensively grazed pasture within the Wye and Usk catchments, categorised as below, (Table 4.1; Figure 4.2). In all, 27 tributaries were surveyed. Three replicate reaches were surveyed on each tributary, except the Edw on which 6 reaches were surveyed. This gave 84 replicate 500m reaches as follows. Treated reaches (n=30), on which Riparian Habitat Management (RHM) was implemented between 1998 and 2003, were compared against control reaches (n=36) as well as reaches with either coniferous plantations (n=9) or intensive pastoral agriculture as the dominant land use within 50m of the stream channel (n=9) (Table 4.1). Control streams (n=12) were assigned post-management, to represent geology, stream order, local land use and altitude that were similar to treated streams (Table 4.1; Figure 4.2; Edwards and Brooker, 1982; Hornung *et al.,* 1990; Environment Agency, 1998). Large numbers (n=84) of replicate reaches were identified to overcome any possible bias associated with site-specific characteristics (Figure 4.2). Control streams were predominantly selected from the Wye catchment but two tributaries of the directly adjacent River Usk, the Honddu and the Grwyne Fawr, were also considered as appropriate for comparison (Figure 4.2). Surveys were undertaken, where possible, in the upper, middle and lower third of each reference tributary.

Table 4.1. Identification of riparian management types from which the physical habitat of streams was surveyed in 2004.

Treatment	Definition
Riparian habitat management ('RHM')	Riparian trees were coppiced and /or fencing erected to exclude grazing under the Wye and Usk Foundation's scheme.
Conifer plantation ('Conifer')	Conifer plantations dominated the immediate vicinity of the stream and were the predominant land use within 50m.
Intensive grazing pasture ('Pastoral')	Streams passed through intensively grazed fields, identified from maps* as having little riparian tree cover (trees lining <20% of river extent), a regular pattern of small fields rather than expansive areas of open common land, with pastoral agriculture as the predominant land use within 50m. In such reaches, riparian fencing was almost entirely absent.
Control	Control streams were chosen based on land use beyond the river bank which was often pastoral but with tree cover that could be identified from maps*, and was similar to that of 'RHM' streams (trees lining >20% of river extent). Control streams were chosen to represent similar stream order (2 to 4), altitude, and underlying geology to that of RHM streams (see Figure 4.2).

*Maps used to assist the identification of riparian land use were Ordinance Survey maps, scale 1:25 000.

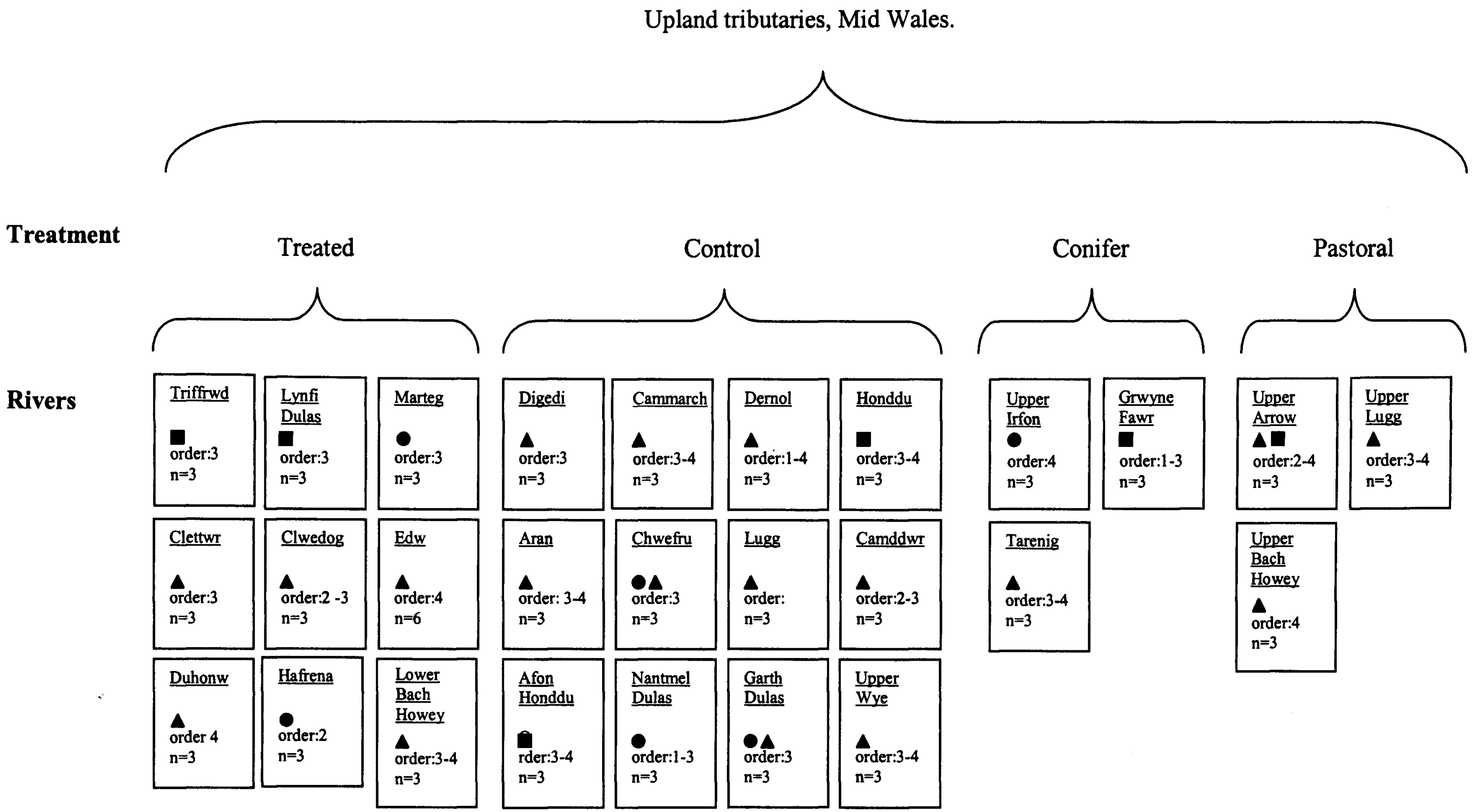
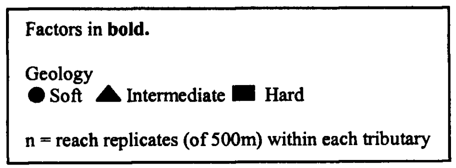


Figure 4.2. Hierarchical ANOVA design used to assess variations in river habitat character between and within tributaries, within management types. Stream order and underlying geology were used to identify appropriate controls in a stratified sampling programme. Management types are defined in Appendix I, geological types in Chapter 3.



[4.3.2] Data collection and treatment

Habitat surveys were undertaken in July and August 2004, using the 2003 version of the UK Environment Agency's River Habitat Survey (RHS) (Raven *et al.*,1998; Environment Agency, 2003c). RHS data are collected as over 100 attributes along a 500m river reach (Fox *et al.*,1998). 10 'spot checks' are recorded at 50m intervals within each reach and a sweep-up set of variables records features over the entire 500m. At each spot check, features of the flow, channel, banks and immediate land use are recorded, typically as the dominant type from a pre-determined series of categories. For example, dominant flow type is chosen from ten possibilities (Environment Agency, 2003c; Vaughan, 2004). Most sweep-up variables are ordinal, often recorded as either absent, present over < 33% of the reach, or present at ≥ 33% (= 'extensive') (Environment Agency, 2003c; Vaughan, 2004).

RHS variables were treated using variable clustering as described by Vaughan and Ormerod (2005). Using this method, the multicolinearity of RHS variables was reduced by identifying groups of similar variables through cluster analysis. Principal Components Analysis (PCA) was then performed on each group and the scores of the first PC axis of variation used to represent that aspect of river character as a 'compound variable' (Vaughan and Ormerod, 2005).

In this study, 87 study sites were surveyed and 10 variables were investigated, resulting in a cases:variables ratio of less than 10:1. In order to reduce this ratio, all of the surveys on the 2005 version of the RHS database for the UK (n=14814) were used to derive principal component scores to represent river habitat characteristics (Harrell *et al.* 1984; Vaughan and Ormerod, 2005; Table 4.2). Surveys undertaken in 2004 as part of this study were added to the data base and the relevant PC scores were extracted for use in further analyses. In accordance with Vaughan and Ormerod (2005), a Principal Components Analysis (PCA) that incorporated optimal scaling was used to account for the ordinal nature of some variables. When a cluster of RHS variables included ordinal variables 'CATPCA' (Categorical Principal Components Analysis) was used to generate scores, otherwise a standard PCA based on correlation was applied (SPSS for Windows, 2001).

Ten variable-clusters were chosen that represented features likely to be altered by management in order to contrast habitat characters of rivers with different riparian land-use. Tree cover, improved pasture and poaching of the bank-tops were used to measure the direct effects of riparian alteration (Table 4.2). Since the object of riparian management was to alter the in-stream habitat, an additional 7 variable clusters were used to represent the indirect effects of riparian change on the channel ('bars, cliffs and pools', rocky, cobbles, bank vegetation, channel vegetation, riffles and filamentous algae). The 'Bars, Cliffs and Pools' cluster largely represented features typical of actively meandering channels, such as unvegetated bars but also stable and eroding cliffs (Table 4.2; Vaughan, 2004). 'Rocky' was indicative of bedrock and boulders and associated flow types such as cascades. 'Cobbles' represented a gradient from cobble through to silt substrata. 'Bank vegetation' described a change from 'uniform' to 'simple' vegetation structure, whilst channel vegetation was derived from the occurrence of macrophytes within the channel at spot checks (Table 4.2; Vaughan, 2004). Vaughan and Ormerod (2005) used the predominant land-use within 50m from the bank top in their clusters. This study investigated pastoral land-use within 5m of the bank-top since treatment aimed to alter the land-use within close proximity to the bank top. Vaughan and Ormerod (2005) found a median Spearman's correlation of 0.75 between land use variables recorded at 5m and at 50m, so the use of their cluster groupings was considered valid in this study.

[4.3.3] Data analysis

A Hierarchical ANOVA was used to assess variations in river habitat between riparian treatments and rivers within treatments, with pair-wise comparisons made by Tukey tests (Iles, 1993; SPSS for Windows, 2001). Data manipulation and statistical analyses were performed in SPSS and Canoco (ter Braak and Šmilauer, 1998, SPSS for Windows, 2001). Bonferroni adjustment for multiple tests was applied where appropriate (Hair, 1995). Since riparian management was largely undertaken in the lower third of tributary streams, any confounding influence of the more lowland locations of treated reaches had to be accounted for. Comparisons of physical habitat between 'treated' and 'control' management types were, therefore repeated using ANOVA confined to reaches from the lower third of tributaries.

[4.4] Results

Although riparian and in-stream habitats were highly variable within the Wye catchment, systematic differences were apparent among rivers and treatments (Table 4.3). Riparian management appeared to reduce poaching on the bank-top while increasing filamentous algae within the channel (Figure 4.3).

The proportion of pasture on the bank top was similar at treated and control reaches but was greater where catchments were dominated by intensive grazing (Figure 4.3). 'Tree cover' primarily represented broadleaved woodland and associated features and was greatest within treated reaches despite efforts to reduce channel shading (Table 4.2; Figure 4.3). Reaches which ran though conifer plantations were not shaded by the canopy and had few of the features, such as underwater roots, exemplified by deciduous trees, such as alder. Moorland was often present within 5m of those streams; consequently moorland-pasture was most commonly associated with conifer-lined tributaries.

No significant differences were detected in the structure of bank vegetation between rivers or treatments (Table 4.3). Channel vegetation varied between rivers and, although more vegetation was recorded in the channels of treated sites, there was no significant treatment effect (Table 4.3). Other in-stream features ('bars, cliffs and pools', rocky and cobble substrata and riffles) were unaffected by riparian alteration (Table 4.3). However, a trend towards finer substrata and more 'bars, cliffs and pools' was evident in 'pastoral' rivers (Table 4.2; Figure 4.3). 'Conifer' rivers differed from others in almost all aspects. They were more rocky with fewer riffles, bars, cobbles and algae (Figure 4.3).

None of these trends were different in analyses comparing treated streams with 'downstream' controls, indicating that the predominantly downstream location of treatment did not bias comparisons of physical habitat (One-way ANOVA comparing poaching, filamentous algae and 'bars, cliffs and pools' $P < 0.05$; $df_{total, error} = 40,1$).

Table 4.2. Loading of River Habitat Survey (RHS) variables on Principal component axes derived from variable clusters.

Tree Cover (X = 10, CATPCA)	
Broad-leaved woodland	35.7
'Complex' bank-face vegetation	30.8
'Complex' bank-top vegetation	35.2
Extent of tree cover	42.9
Shading of channel	42.5
Overhanging boughs	40.6
Exposed bank-side tree roots	39.7
Exposed underwater roots	35.0
Fallen trees in channel	37.3
Woody debris	37.9
Moorland - Pasture (X = 3, PCA)	
Moorland/heath	0.8
Peat banks	0.8
Improved grassland	-0.6
Bars / Cliffs / Pools (X = 7, CATPCA)	
Eroding cliffs ≥ 0.5m	48.0
Stable cliffs ≥ 0.5m	39.0
Number of pools	45.7
Number of unvegetated point bars	56.7
Vertical + toe bank profile	46.5
Marginal dead-water	30.1
Unvegetated mid-channel bars	48.4
Rocky (X = 12, CATPCA)	
Bedrock banks	36.4
Boulder banks	34.0
Earth banks	-24.6
Bedrock channel substratum	32.9
Boulders channel substratum	35.0
'Chute' flow type	31.8
'Broken wave' flow type	27.3
Bryophytes	32.0
Cascades	35.8
Rapids	33.6
Exposed bedrock in channel	35.6
Exposed boulders in channel	38.3

Cobbles (X = 4, PCA)	
Cobble channel substratum	0.8
Cobble banks	0.7
Unvegetated side bars	0.7
Silt substratum	-0.7
Bank Vegetation (X = 4, PCA)	
'Uniform' bank-face vegetation	-0.8
'Simple' bank-face vegetation	0.8
'Simple' bank-top vegetation	0.8
'Uniform' bank-top vegetation	-0.7
Channel Vegetation (X = 7, PCA)	
Reeds/sedges/rushes	0.7
Free floating	0.7
Submerged fine/linear-leaved plants	0.7
Submerged fine/linear-leaved plants	0.7
Emergent broad-leaved	0.6
Submerged broad-leaved	0.6
Amphibious	0.4
Riffles (X = 2, PCA)	0.9
'Unbroken wave' flow type	0.9
Number of riffles	
Runs (X = 2, CATPCA)	85.5
'Rippled' flow type	85.5
Extent of runs	
Poaching (X = 2, CATPCA)	86.1
Composite bank profile	86.1
Poaching of channel margin	
Filamentous Algae	
Unique variable	
Coniferous plantation	
Unique variable	

Table 4.3. Variation in river habitat between rivers within treatments and between treatments (riparian management control, conifer and pastoral reaches), as defined by variable clusters (after Vaughan and Ormerod, 2005; See Table 4.2).

Habitat PCs describing:	River (Treatment)		Treatment	
	F	P	F	P
Tree Cover	10.49	<0.001	34.47	<0.001
Moorland - Pasture	1.72	0.057	9.17	<0.001
Poaching	2.75	0.001	17.59	<0.001
Bars / Cliffs / Pools	1.68	0.064	6.54	0.001
Rocky	3.05	0.001	4.74	<0.001
Cobbles	7.27	<0.001	18.53	<0.001
Bank Vegetation	1.40	0.162	2.06	0.116
Channel Vegetation	2.53	0.003	1.93	0.135
Riffles	2.38	0.006	4.94	0.004
Filamentous Algae	7.75	<0.001	19.53	<0.001
df (Numerator, Denominator)	21, 54		3, 54	

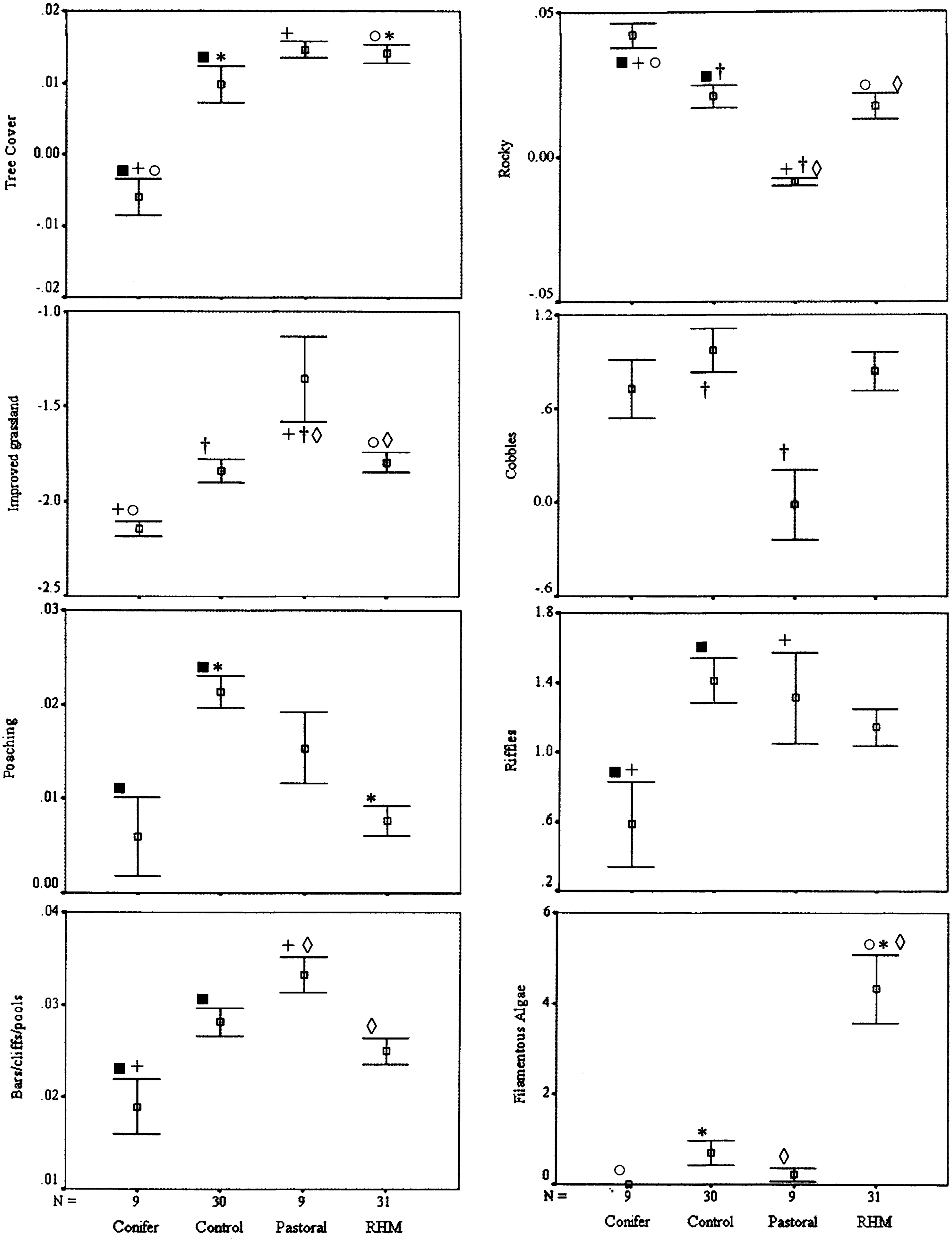


Figure 4.3. Comparisons of river habitat between treated, control, coniferous and pastoral reaches of the Wye represented as mean (± SE) scores of variable clusters after Vaughan and Ormerod (2005). Pair-wise differences between treatments are indicated by *, †, ○, ■, + and ◊. (Tukey tests; P<0.05) (See Table 4.3 for analysis).

[4.5] Discussion

Habitat varied widely across the Wye catchment, between land-use types as well as within rivers. Within rivers, tree cover, poaching, substrata, flow types and channel vegetation all varied. Despite the highly variable habitats within rivers, distinct differences in almost all aspects of habitat were evident between riparian land-use types. Streams draining intensive pasture tended to be characterised by fine substrata and features typical of 'active' channels, recently managed streams by less poaching and more filamentous algae compared to controls. Streams within conifer plantations had more rocky substratum and lacked improved grassland or features associated with deciduous trees.

The variability of habitats within and between river environments demonstrates the importance of exploring the effects of riparian land use at different scales (Townsend *et al.*, 2004). However, many riparian management projects are conducted without adequate monitoring and often with weak experimental design (Kondolf, 1995; Muhar *et al.,* 1995; O'Grady, 1998; Rinne, 1999; Sarr, 2002). Typically, pre-treatment data are few, reference streams or reaches are seldom monitored and the scale of manipulations are limited (Ormerod 2004). Under these circumstances, reaching an evidence-based view on the value of riparian management can be difficult. In instances where no pre-treatment data are collected, survey-based comparisons between treated sites and carefully selected control sites can be used to infer change through time. Although not without difficulties and drawbacks, in some cases this is the only option available (Manel *et al.,*2000). This was the case in the Wye catchment. Here, an experimental design was imposed '*post hoc*' on an already operational scheme which incorporated multiple management techniques, coppice, or fence and coppice and fence, often within the same rivers or reaches. Despite the confounded effects of treatments, the River Habitat Survey was able to detect general differences between recently managed streams, controls and streams running through coniferous plantation or intensively grazed pasture. Some aspects of the different management techniques may have had contrasting effects on the river environment. Across the catchment, reduced tree cover was associated with increased channel vegetation whilst there was often less channel vegetation where bank vegetation was greater. Coppicing and fencing may therefore have had conflicting effects on channel vegetation.

In a study contrasting streams in improved pastoral and native grassland in the South Island of New Zealand, Townsend *et al.*, (2004) also found inter-river variation in riparian and stream vegetation (overhanging vegetation and moss cover) as well as habitat variation between rivers with different riparian land use. In that study, streams running through pasture had less in-stream moss cover, endemic riparian species, lower water velocity and more erosion and sediment. Similarly, in Welsh streams in intensively grazed pasture, substrata tended to be finer along with more 'bars, cliffs and pools'. However, control streams running through what appeared from Ordinance Survey maps (1:25,000) as less intensive pasture land (see Table 4.1) were grazed to the extent that poaching was most evident on the banks here than under any other riparian land-use. Streams with recent riparian management were less poached than controls, indicating success in excluding grazing from banks.

Structural changes in bank-side vegetation following riparian management are likely to develop over time – perhaps even years where the development of woody vegetation is involved. Reports of a lagged response of channel morphology and biota to riparian fencing are therefore not uncommon (Kondolf, 1993, Agouridis *et al.*, 2005). When surveyed in 2004, recently managed reaches had been altered between 1 and 6 years previously, so that limited effects on riparian vegetation and channel structure might be expected. By contrast, the more immediate impact of livestock poaching appeared to be reduced by stock exclusion, while structural changes in the bank vegetation were not detected.

A disadvantage of the inevitable variation in the timing of habitat management was that although deciduous trees were coppiced, after six years there was considerable re-growth. For example, the Clywedog was altered in 1998, and by 2004 the coppice had created 'tunnels' of tree cover (O'Grady, 1993; Hendry *et al.*, 2003). Without maintenance, the coppiced trees had created a more shaded channel which was contrary to management goals (Wye and Usk Foundation, 2006; Hendry and Cragg-Hine, 1997; Crompton, 1994, see Chapter 1).

Tree features recorded by the RHS are predominantly associated with deciduous trees, such as the exposed tree roots of alders. Shading of the river channel and 'extent of tree cover' were important components of the 'tree cover' variable, yet bank-side conifer trees tend not to group with any other RHS habitat variable and were treated as a unique variable (Table 4.2; Vaughan, 2004). Conifer trees lining the stream reaches (orders 1-4, widths approx 2-12m) investigated in this study did not shade the water. In fact, a deciduous tree or, most frequently, a moorland buffer of 1-10m was evident on most reaches. Forestry Commission guidelines suggest that buffers of 10m on either side of small stream channels (1-2m wide) and 20m for larger streams (Forestry Commission, 2003). Although some streams were not buffered to the extent suggested by the current guidelines, trees rarely over-hung into the stream. This contrasts with surveys undertaken in 1990 on smaller streams in the UK (orders 1-3) in which coniferous plantations shaded streams passing through them (Ormerod, 1993). Never the less, the association of 'hard' in-stream features, such as coarse substratum, with 'conifer' streams was common to both studies.

Despite 'tree cover' being greater in recently managed sites than controls, a proliferation of in-stream flora was detected within treated reaches. These results may indicate growth of *Lemanea* and *Rhodophyta* (red macro-algae) which are adapted to shaded conditions and were observed post–treatment on the river Clywedog (Hynes, 1970; Thrib and Benson-Evans, 1983, 1984; Luxton, 2002). Alternatively, the apparent contradiction in shading and algal growth in recently managed streams may simply be a result of the scales at which each feature was observed; 'spot checks' every 50m recorded frequencies of vegetation types within the channel whilst tree cover was estimated for the entire 500m reach (Imhof *et al.*, 1996). The greater resolution of spot check variables may have detected changes 'missed' by 'sweep-up' variables.

The riparian zone is an interface between the wider catchment and the river itself (Malanson, 1993). Although approximately 58 km of river had been coppiced and / or fenced throughout the Wye system by June 2004, this was still only a small part of the landscape that ultimately affected the river channel (Harper *et al.*, 1999; Townsend *et al.*, 2004). By 2004, the majority of riparian management had taken place on the lower third of each tributary. Even on the most extensively managed rivers, the Clywedog and the

Edw, fencing did not reach far into the upper third of the river. This meant that upstream impacts on the channel such as potential sources of sediment remained unchecked.

Many programmes that alter riparian habitat aim to change in-stream morphology by stabilising banks, increasing the width:depth ratio of channels and reducing sedimentation of the substrata thus encouraging the development of riffle-pool sequences (Hendry *et al.*, 2003; Landers, 1997; Chapter 2). However, characters related to sediment supply and substrata are likely to be controlled by topography, hydrology and land use within the wider catchment (Asselman *et al.*, 2003; Opperman *et al.*, 2005; Restrepo and Syvitski 2006). Here, streams with catchments dominated by pastoral agriculture and conifer plantations were geomorphologically distinct from others in terms of substrata, flow type and channel features. This suggests that to achieve any morphological change practitioners should look beyond the immediate riparian interface (Kentula, 1997; Wissmar and Beschta, 1998).

Coppicing and riparian fencing, however, appeared to have successfully excluded grazing on banks and increased filamentous algae within the stream. The consequences of such riparian management for organisms will be investigated in Chapters 4 and 5.

[4.6] References

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Chapter 5

The effects of riparian management on macroinvertebrate communities in the Wye River system

[5.0] Abstract

In upland, rural locations, pastoral agriculture or forestry in river catchments can influence assemblages of river organisms. In addition to wider effects on catchment hydrology and chemistry, intensive livestock grazing can cause bank erosion and increase the sediment supply to streams. Semi-natural riparian buffers are frequently established to offset any negative ecological effects, often for the enhancement of salmonid populations. However, effects on other groups of organisms are less often assessed. Benthic macroinvertebrates, which are ubiquitous across river habitats, and vary in composition with land-use and physico-chemical condition, are ideal indicators in which to appraise management effects.

A programme of fencing and coppicing of riparian trees was implemented on upland tributary streams of the River Wye in Mid-Wales from 1997. In the absence of project-specific invertebrate monitoring, data from the UK Environment Agency's BIOSYS database were used opportunistically to compare assemblages in three recently managed streams and five adjacent control streams. Data were available between 1995 and 2004, enabling some comparisons of pre-and post-treatment assemblage pattern in the recently managed and control streams, respectively.

Over time, in all reaches irrespective of treatment, families typical of lowland, more eutrophic rivers increased in occurrence and taxon richness declined. Post-treatment (2000-2004) assemblages were richer in recently managed streams than in controls, mostly due to effects from taxa typical of channel margins and lowland conditions, but these results were equivocal. This study highlights the importance of the riparian zone and local land-use to stream biota, but also reveals the importance of careful study design in the evidence-based evaluation of riparian management. The reasons for the recent decline in the richness of typical headwater organisms from some tributaries of the Wye require further investigation.

[5.1] Introduction

The management of river catchments has far-reaching impacts on both the physical and biotic character of rivers (Manel *et al.*, 2000; Snyder *et al.*, 2003; Danger and Robson, 2004; Thompson and Townsend, 2004; Townsend *et al.*, 2004; also see Chapter 2). In addition to river hydrology, hydrochemistry and sediment regimes (see Russell *et al.*, 2001; Steegen *et al.*, 2001; McKergow *et al.*, 2003; Peckham, 2003; Evans *et al.*, 2006), the supply of terrestrial organic matter to river ecosystems, such as terrestrial invertebrates and detritus (see Wipfli, 1997; Pretty and Dobson, 2004; Wipfli and Musslewhite, 2004; Baxter *et al.*, 2005), are ultimately under the control of the terrestrial environment. In rural catchments, forestry and agriculture, and their relative intensity can substantially modify biotic communities (Ormerod *et al.*, 1993; Weatherley *et al.*, 1993; Parsons *et al.*, 2003; Buck *et al.*, 2004; Giller and O'Halloran, 2004; Neal *et al.*, 2004; Ormerod *et al.*, 2004; Liebault *et al.*, 2005; Chapter 3).

The responses of aquatic biota to local land-use are often reported as variations in the assemblage composition of benthic macroinvertebrates and fish (e.g. Brewin *et al.*, 1995; Monaghan *et al.*, 2000; Nerbonne and Vondracek, 2001; Meador and Goldstein, 2003; Zimmerman *et al.*, 2003; Sandin and Johnson, 2004; Ormerod *et al.*, 2004). Benthic macroinvertebrates are particularly suited to comparative studies because the relative compositions of species, families and feeding guilds vary closely with local land-use and physico-chemical condition (Rundle *et al.*, 1992; Ormerod *et al.*, 1993; Rosenberg and Resh, 1993; Brewin *et al.*, 1995; Monaghan *et al.*, 2000; Sandin and Johnson, 2004; Townsend *et al.*, 2004). The richness among invertebrates, distinction in habitat requirements, responsiveness to ecological conditions and well-known ecology also make invertebrates excellent subjects in river research.

The presence of trees within a catchment and the proportion of deciduous, coniferous, exotic and native species affects both the quantity and quality (C:N ratio) of organic input to streams (Dobson *et al.*, 1995; Quinn *et al.*, 2000; Elliott *et al.*, 2004; Swan and Palmer, 2004; also see Chapter 2). The relative importance of forestry to riverine productivity varies also with river size. Smaller streams tend to be influenced by riparian trees which can shade the channel directly as well as providing allochthonous matter (Vannote *et al.*, 1980). Invertebrate assemblages in smaller, headwater streams in temperate zones tend to be dominated by shredding and detritivorus feeding guilds with a greater proportion of families typical of stream margins rather than the herbivorous and filter-feeding taxa found further downstream in more 'open' channels (Vannote *et al.*, 1980; Cummins *et al.*, 1989, Sweeney, 1992, 1993; also see Chapter 2).

In contrast to afforested areas, pastoral agriculture is often associated with a reduction in tree and shrub cover both in close proximity to the stream and throughout the catchment (Nakamura and Yamada, 2005). Implications for invertebrate communities include reduced shading, increased algal productivity and increased density of invertebrate herbivores (e.g. Sabater *et al.*, 1998). The proportions of woody debris, detritus, sediment and nutrients within streams are also modified. Vegetation within the catchment can intercept sediment and nutrients and help to reduce additional soil loss through erosion (Easson and Yarbrough, 2002; Hook, 2003). In combination with poaching (trampling) by livestock, loss of terrestrial vegetation can increase the amount of sediment in the benthos (Evans *et al.*, 2006). This can adversely impact on periphyton productivity, the density and diversity of aquatic invertebrates, fish feeding, fish spawning and egg survival (Waters, 1995; Harding *et al.*, 1999; Nakamura and Yamada, 2005).

As the interface between the river and the wider catchment (Malanson, 1993), management of the riparian zone can offer a means of mitigating the effects of wider catchment management on river systems. For example, wooded or grassed riparian buffers can ameliorate some of the negative impacts of agricultural and forestry practices, such as sediment and nutrient release to the stream environment (Petersen *et al.*, 1992; García de Jalón, 1995; Goodwin *et al.*, 1997; Hendry *et al.*, 2003; McKergow *et al.*, 2003; Broadmeadow and Nisbet, 2004; also see Chapter 2). Fenced riparian buffers to exclude livestock grazing, as well as the management of riparian trees, are widely advocated in programmes that aim to improve salmonid fisheries (White, 1996; Hendry and Cragg-Hine, 1997; Roper *et al.*, 1997; White, 2002; Hendry *et al.*, 2003). However, the effects of such programmes on other aspects of freshwater ecology are rarely reported (White, 1996; Kauffman *et al.*, 1997; Chapter 2).

In the UK, across Europe and throughout much of the developed world, invertebrate communities are routinely monitored (Environment Agency, 1999; Rosenberg and Resh, 1993; European Directive 2000/60/EC). Such data are readily available, spatially extensive and in the UK have been collected over at least 15 years. Unfortunately, the application of national archives of ecological data in the assessment of river management projects is seldom published within the academic literature. This chapter demonstrates the application of routinely collected macroinvertebrate data to assess the effects of riparian management on stream biota.

A programme of fencing along with coppicing and thinning riparian trees was implemented on tributary streams of the River Wye in Mid-Wales from 1997 (see Chapter 1). Because macroinvertebrates were not monitored specifically for the project, invertebrate monitoring data from the UK Environment Agency's BIOSYS database were used to compare communities between three recently managed streams and five control streams (Figure 5.1; Environment Agency, 1999). Archived data were also available from 1995 (2-3 years prior to management), thereby enabling additional comparisons of pre- and post-treatment condition in the recently managed and control streams. Specifically, the questions asked were: i) was there any interaction between time period and riparian treatment? ii) were post-treatment assemblages different in treated and control streams? iii) did assemblages change following treatment, in either treated or control streams?

[5.2] Site description and management

The River Wye, its rural catchment ($4136 km^2$) and invertebrate communities were comprehensively described in Chapter 3 and have also been described elsewhere (e.g. Edwards and Brooker, 1982; Ormerod and Edwards, 1987, Ormerod 1987, 1988; Environment Agency, 1998; Brennan *et al.*, 2003; Jarvie *et al.*, 2003). In the middle and upper reaches of the Wye catchment (defined in Chapter 3, figure 3.1) pastoral agriculture is the dominant land-use. Tributaries of the Middle-Wye are relatively unpolluted, being largely unaffected by organic enrichment or acidification (see Chapter 3). Ephemeropteran families, such as Caenidae, Baetidae and Ephemerellidae are well represented in invertebrate assemblages, which are relatively rich, compared with the upper, acidified reaches of the Wye (see Edwards and Brooker; 1982; Chapter 3).

The Wye Habitat Improvement Project (WHIP), implemented by the Wye and Usk Foundation (WUF), aimed to create optimal habitat for salmonid fish in order to boost populations of Atlantic salmon (*Salmo salar*) and brown trout (*S. trutta*) within the Wye catchment (Wye and Usk Foundation, 2006). It involved a programme of fencing to exclude livestock grazing on the river banks (typically 1-3m from the stream bank) and coppicing of riparian trees (see Chapter 1). Riparian management took place between 1997 and 2000 on the Clywedog, Edw and Sgithwen tributaries of the River Wye, of Strahler stream orders 4, 3, and 2 respectively at their confluence with the main river Wye (Figure 5.1; exact locations confidential). The Clywedog and Edw were the most extensively managed tributaries and were coppiced, fenced or both along over two thirds of their length (river length = 15.3 and 24.5 km respectively). The Sgithwen was coppiced close to its confluence with the Wye in 1997, and then additional coppicing took place further upstream in 2000, thus managing less than two thirds of the river's length (Sgithwen length = 9.2 km).

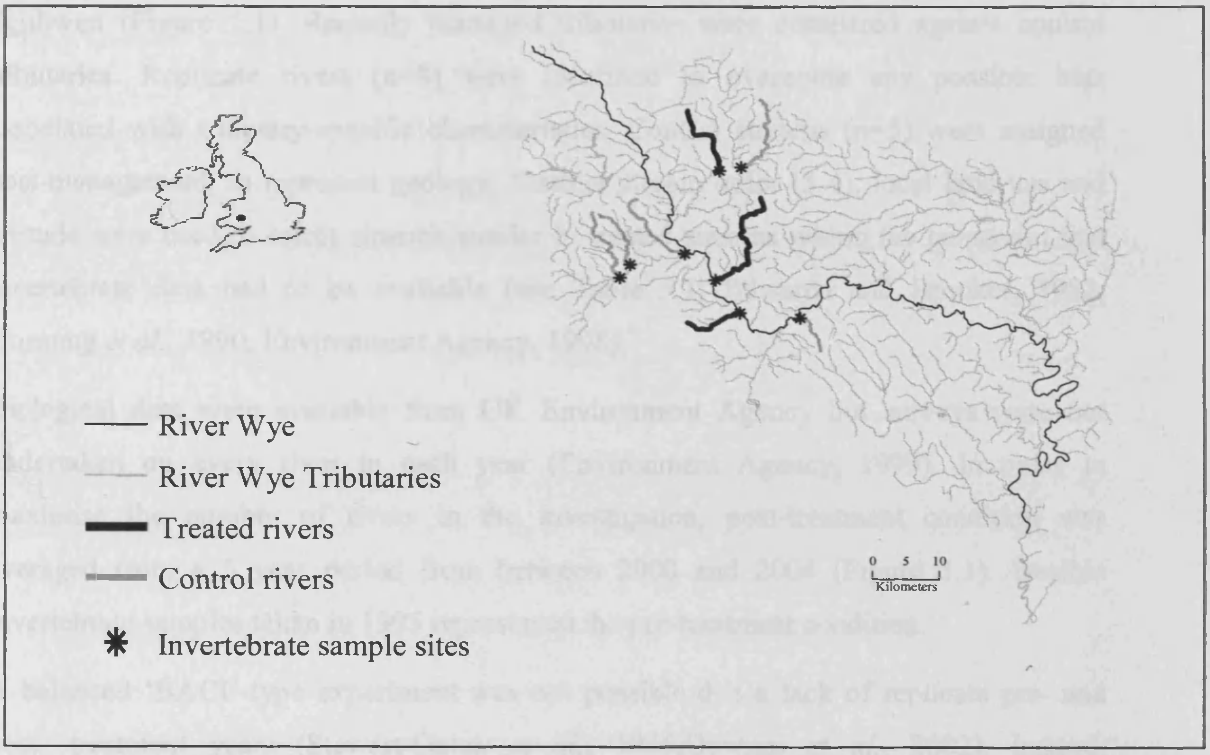

Figure 5.1. Treated and control tributaries of the Wye on which macroinvertebrate surveys were undertaken between 1995 and 2004.

[5.3] Methods

[5.3.1} Study Design

This study investigated the effects of recent riparian management (undertaken between 1997 and 2000) on three tributary streams in the Wye catchment, the Edw, Clywedog and Sgithwen (Figure 5.1). Recently managed tributaries were compared against control tributaries. Replicate rivers (n=8) were identified to overcome any possible bias associated with tributary-specific characteristics. Control streams (n=5) were assigned post-management, to represent geology, Strahler stream order (3-4), local land use and altitude were used to select streams similar to treated streams within the constraint that invertebrate data had to be available (see Table 5.1; Edwards and Brooker, 1982; Hornung *et al.,* 1990; Environment Agency, 1998).

Biological data were available from UK Environment Agency but surveys were not undertaken on every river in each year (Environment Agency, 1999). In order to maximise the number of rivers in the investigation, post-treatment condition was averaged from a 5 year period from between 2000 and 2004 (Figure 5.1). Benthic invertebrate samples taken in 1995 represented the pre-treatment condition.

A balanced 'BACI'-type experiment was not possible due a lack of replicate pre- and post- treatment years (Stewart-Oaten *et al.*; 1986;Downes *et al.*, 2002). Instead, interactions between time period and treatment were tested for. Post-treatment condition was compared between treated and control streams and trends through time were established separately for treated and control groups respectively.

Since no macroinvertebrate surveys were undertaken specifically for this study, the experimental design could only be devised retrospectively using data available from other sources. The effects of this unavoidable constraint on conclusions are discussed below. All streams were surveyed in 2000 but the years in which additional surveys were done varied. Treated streams were largely sampled in 2000 and 2004; whilst the controls were surveyed in 2000 and either 2002 or 2003. Since any treatment effects might be confounded by year effect, analyses were repeated, taking only the surveys done in 2000 to represent post-treatment conditions.

[5.3.2] Data Collection and treatment

Biological data were collected prior to treatment as part of the UK Environment Agency (EA)'s routine monitoring and archived on their 'BIOSYS' database (Environment Agency, 1999). Benthic macroinvertebrates were sampled from reaches close to the confluence with either the main River Wye or two major tributaries of the Wye, the Irfon and Ithon. Invertebrates were collected by a standardised kick-sample, identified to family-level and their abundance recorded on a semi-quantitative logarithmic scale of >1, >10, >100, >1000 and >10000 in accordance with the Biological Monitoring Working Party (BMWP) method (See Chapter 3 for detail; Hawkes, 1997; Environment Agency, 1999).

Macroinvertebrate data were treated as described in detail in Chapter 3 (Figure 5.1). BMWP-scoring taxa that were present in more than 1% of samples were used for analysis and abundances were reciprocally transformed to reduce skew and kurtosis (SPSS for Windows, 2001; Tabachnick and Fidell, 2001). Six biotic metrics were derived from standardised kick-samples of invertebrate families (Environment Agency, 1999; Clarke *et al.,* 2002, also see Chapter 3). Variations in trophic putative structure were assessed from the proportions of 'shredder', 'grazer' and 'detritivore' feeding guilds (see Chapter 3; Moog, 1995). Changes in the structure of invertebrate communities were represented by taxon richness (number of families) and the first two axes of a Principal Component Analysis on family abundance. 'PC1' and 'PC2' reflected gradients in assemblage composition.

In this study, 8 rivers were surveyed and 35 invertebrate families were investigated, resulting in a cases:variables ration of less than 5:1, and hence possible problems in data analysis and interpretation (see Vaughan & Ormerod, 2003). In order to reduce this ratio, surveys on the BIOSYS database for the UK were used to derive Principal Component scores that represented invertebrate assemblages (Harrell *et al.* 1984; Vaughan and Ormerod, 2005). PC scores of samples relevant to this study were then extracted for use in further analyses. Years 2000-2004 and 1995 were used to derive PC scores when interactions between treatments and time periods or temporal changes were investigated (Table 5.3). When post-treatment comparisons were made, data were extracted from 2000-2004 only in order to generate PC scores (Table 5.2). Loadings on the first and second axes were similar in both cases (see Tables 5.2 and 5.3).

[5.3.3] Data analysis

Interactions between treatments (recently managed or control) and time periods (1995 or 2000-2004) were established by a repeated measures ANOVA that crossed treatment with time period (Tabachnick and Fidell, 2001). Post-treatment differences in invertebrate assemblage composition and feeding guild in recently managed and control reaches were then investigated by one-way ANOVA (Sokal and Rohlf, 1995).

To assess the effects of treatment through time, pre-treatment macroinvertebrate data were compared against post-treatment data by repeated measures ANOVA (Tabachnick and Fidell, 2001). These analyses were repeated for recently managed and control reaches respectively.

Data manipulation and statistical analyses were performed in SPSS and CANOCO (ter Braak and Šmilauer, 1998, SPSS for Windows, 2001). Bonferroni adjustment for multiple tests was applied where appropriate (Hair, 1995).

Table 5.1 Definition of riparian management types from which the macroinvertebrate data were collected between 1995 and 2004.

Treatment	Definition
Riparian habitat management ('RHM')	Riparian trees were coppiced and / or fencing erected to exclude grazing under the Wye and Usk Foundation's scheme.
Control	Control streams were chosen based on land use beyond the river bank which was often pastoral but with tree cover that could be identified from maps* that was similar to that of 'RHM' streams (trees lining >20% of river extent). Control streams were chosen to represent similar stream order (3 to 4), altitude, and underlying geology to that of RHM streams (see Chapter 4).

*Maps used to assist the identification of riparian land use were Ordinance Survey maps, scale 1:25 000.

[5.4] Results

[5.4.1] Overall assemblage composition

In a PCA of all samples, families typical of channel margins or slack waters (e.g. Caenidae, Gyrinidae, Leptophlebiidae, Sericostomatidae) were positively loaded on the first PC axis (Tables 5.2 and 5.3). This axis explained 25–29% of the total variance among family and was highly correlated (Spearman's $r_s = 0.9$) with taxon richness. Assemblages shifted from typical upland families (Baetidae, Heptageniidae, Hydropsychidae, Lepidostomatidae and Leuctridae) to lowland families such as Asellidae, Planorbidae and Sialidae on PC2 (Tables 5.2 and 5.3).

[5.4.2] Treatment effects

Interactions between treatment and time period were significant in autumn for PC1, which represented families typical of channel margins (Table 5.4). Marginally significant interactions were suggested for PC2 in spring and PC1 in autumn (Table 5.4). Treatment effects were evident as a shift towards families typical of channel margins and more lowland conditions in treated sites relative to controls (Tables 5.4 and 5.6.).

Post-treatment (2000-2004) invertebrate assemblages were richer in recently managed streams than in controls. Assemblage composition, suggested by PC scores, indicated that recently managed streams held taxa more associated with channel margins (i.e. those loading positively on PC1) and lowland conditions (i.e. those loading negatively on PC2) (Figure 5.2; Table 5.5). There was some tendency towards fewer shredders and more grazers in treated streams but differences between treatments were not significant (Figure 5.2; Table 5.5). Patterns in these invertebrate metrics were similar in spring and autumn (Table 5.5). However, when analyses were repeated with the post-treatment condition represented by only surveys undertaken in 2000 (i.e when all sites were surveyed in the same year) no post-treatment effects were apparent (one-way ANOVA; $df_{total, error} = 7,6$; $P>0.05$).

[5.4.3] Trends through time

Temporal variations in richness, PC2 and feeding guilds were similar in both recently managed and control streams (Table 5.6). Although the representation of different feeding guilds within invertebrate communities did not vary between time-periods (Repeated Measures ANOVA; $P<0.05$ $df_{total, error} = 1,2$ and $1,4$, respectively), lowland families represented by PC2 increased through time in spring samples (Table 5.6). Taxon richness significantly declined over time in autumn (Table 5.6), when reductions in richness were between 15% and 20%. Contrasting trends in PC1 were evident in the two treatments. Communities in treated streams shifted towards families more typical of channel margins while the converse was true in controls (Table 5.6).

Table 5.2. Loading of commonly occurring invertebrate families on Principal Component axes describing major sources of variation in invertebrate communities sampled between 2000 and 2004 in the autumn and spring respectively. Correlation coefficients greater than 0.5 are highlighted in bold.

	Autumn		**Spring**	
	PC1	**PC2**	**PC1**	**PC2**
% of Variance	25	14	23	14
Family				
Ancylidae	**0.503**	-0.167	**0.514**	0.065
Asellidae	0.206	**0.665**	0.236	**0.676**
Baetidae	0.454	-0.415	0.448	-0.271
Caenidae	**0.664**	0.196	**0.646**	-0.099
Chironomidae	0.091	0.016	0.056	0.200
Dytiscidae Noteridae	0.402	0.452	**0.533**	0.282
Elmidae	0.447	-0.272	0.465	-0.283
Ephemeridae	**0.603**	-0.068	**0.573**	-0.031
Ephemerellidae	**0.676**	0.226	**0.658**	0.118
Erpobdellidae	0.267	**0.535**	0.185	**0.581**
Gammaridae Crangonyctidae	0.259	0.307	0.247	0.399
Glossiphoniidae	0.265	**0.689**	0.305	**0.624**
Gyrinidae	**0.721**	-0.003	**0.755**	-0.062
Heptageniidae	**0.634**	-0.487	**0.597**	**-0.574**
Hydrobiidae Bithyniidae	0.333	0.346	0.321	0.469
Hydrophilidae Hydraenidae	**0.588**	-0.037	**0.579**	0.012
Hydropsychidae	**0.571**	-0.467	**0.541**	-0.482
Hydroptilidae	**0.627**	0.130	**0.579**	0.134
Lepidostomatidae	**0.612**	-0.351	**0.588**	-0.339
Leptoceridae	**0.562**	0.189	**0.618**	0.073
Leptophlebiidae	**0.701**	-0.031	**0.603**	0.082
Leuctridae	**0.580**	-0.405	0.473	-0.434
Limnephilidae	0.497	0.139	0.368	0.255
Lymnaeidae	0.314	0.423	0.330	**0.567**
Nemouridae	**0.632**	-0.014	**0.517**	-0.249
Oligochaeta	0.060	-0.011	0.125	0.191
Perlodidae	**0.713**	-0.108	**0.572**	-0.474
Planariidae Dugesiidae	0.332	0.243	0.313	0.344
Planorbidae	0.321	**0.670**	0.331	**0.665**
Rhyacophilidae Glossosomatidae	**0.520**	-0.395	**0.523**	-0.385
Sericostomatidae	**0.633**	-0.162	**0.652**	-0.087
Sialidae	0.449	**0.613**	**0.563**	0.457
Simuliidae	0.458	-0.395	0.296	-0.193
Sphaeriidae	0.171	**0.677**	0.148	**0.750**
Tipulidae	0.414	-0.289	0.423	-0.122

Table 5.3. Principal Components used to explore temporal variation in invertebrate assemblage structure. Loading of commonly occurring invertebrate families on Principal Component axes describing major sources of variation in invertebrate communities sampled in 1995 and between 2000 and 2004 in the autumn and spring respectively. Correlation coefficients greater than 0.5 are highlighted in bold.

	Autumn		**Spring**	
	PC1	**PC2**	**PC1**	**PC2**
% of Variance	28	13	29	13
Family				
Ancylidae	0.485	-0.248	**0.546**	0.016
Asellidae	0.421	**0.583**	**0.554**	**0.509**
Baetidae	0.354	-0.483	0.236	-0.253
Caenidae	**0.743**	0.111	**0.680**	-0.201
Chironomidae	0.085	0.003	0.060	0.179
Dytiscidae Noteridae	**0.525**	0.398	**0.714**	0.090
Elmidae	0.380	-0.317	0.355	-0.291
Ephemeridae	**0.596**	-0.204	**0.550**	-0.180
Ephemerellidae	**0.740**	0.149	**0.710**	0.028
Erpobdellidae	0.392	0.469	0.395	**0.516**
Gammaridae Crangonyctidae	0.323	0.253	0.297	0.424
Glossiphoniidae	0.432	**0.624**	**0.544**	**0.511**
Gyrinidae	**0.740**	-0.028	**0.808**	-0.194
Heptageniidae	**0.567**	**-0.541**	0.429	**-0.694**
Hydrobiidae Bithyniidae	0.400	0.348	0.443	0.482
Hydrophilidae Hydraenidae	**0.614**	-0.122	**0.622**	-0.177
Hydropsychidae	0.485	**-0.547**	0.392	**-0.581**
Hydroptilidae	**0.650**	0.083	**0.595**	0.119
Lepidostomatidae	**0.553**	-0.421	**0.527**	-0.427
Leptoceridae	**0.614**	0.180	**0.673**	-0.009
Leptophlebiidae	**0.745**	-0.083	**0.657**	-0.010
Leuctridae	**0.539**	-0.491	0.440	**-0.624**
Limnephilidae	**0.557**	0.075	0.488	0.237
Lymnaeidae	0.419	0.356	**0.626**	0.346
Nemouridae	**0.687**	-0.032	**0.604**	-0.306
Oligochaeta	0.103	-0.066	0.158	0.197
Perlodidae	**0.741**	-0.161	**0.558**	**-0.549**
Planariidae Dugesiidae	0.407	0.134	0.467	0.236
Planorbidae	**0.572**	**0.538**	**0.667**	0.447
Rhyacophilidae Glossosomatidae	0.466	-0.428	0.434	-0.332
Sericostomatidae	**0.619**	-0.177	**0.682**	-0.102
Sialidae	**0.624**	**0.505**	**0.781**	0.208
Simuliidae	0.383	-0.488	0.154	-0.130
Sphaeriidae	0.352	**0.606**	0.435	**0.655**
Tipulidae	0.406	-0.397	0.384	-0.178

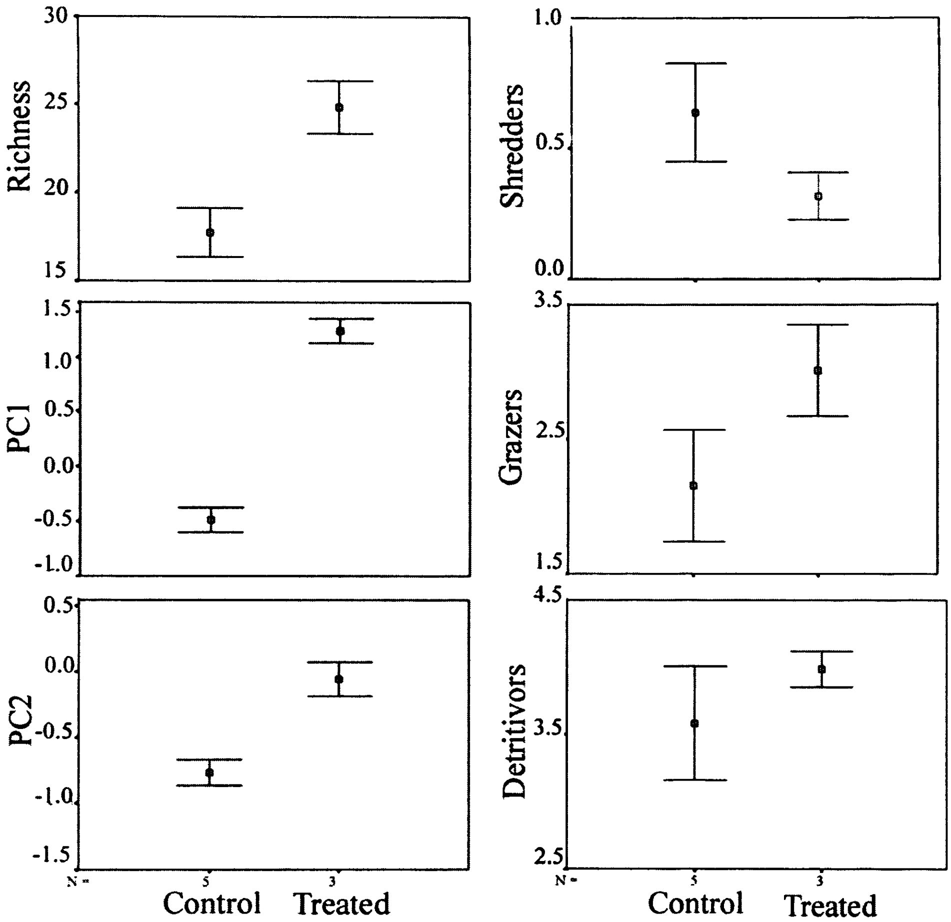

Figure 5.2. Mean (±SE) indicators of invertebrate assemblage composition in treated and control streams surveyed in spring between 2000 and 2004.

Table 5.4. Interactions between treatments (recently managed or control) and time periods (1995 or 2000-2004) identified by repeated measures within-subjects ANOVA ($df_{\text{time period, treatment * time period, error}} = 1,1,6$).

	Spring		Autumn	
	F	P	F	P
Richness	1.22	0.312	24.06	0.228
PC1	8.87	0.025†	22.07	0.003
PC2	5.04	0.066	6.58	0.043†

† No longer significant at 95% CI after Bonferroni correction for multiple tests (P/3)

Table 5.5. Post-treatment variation in the structure of invertebrate communities between treated (n=3) and control (n=5) streams in the River Wye system (2000-2004). $df_{\text{total, error}} = 7,6$.

	Spring		Autumn	
	F	P	F	P
Richness	11.02	0.016	5.40	0.059
PC1	107.40	<0.001	78.85	<0.001
PC2	19.14	0.005	6.87	0.039
Shredders	1.29	0.300	0.29	0.610
Detritivores	0.70	0.436	0.10	0.767
Grazers	1.77	0.232	4.24	0.085

Table 5.6. Temporal variation in the structure of invertebrate communities in a) treated (n=3) and b) control (n=5) streams in the Wye catchment.

a)

	Spring			Autumn		
	Richness	PC1	PC2	Richness	PC1	PC2
1995	25.3 (0.9)	0.3 (0.1)	-1.6 (0.2)	27.3 (0.9)	0.2 (0.3)	-0.6 (0.2)
2000 - 2004	24.8 (1.5)	0.9 (0.1)	-0.5 (0.1)	23.1 (1.3)	0.7 (0.1)	-0.3 (0.3)
df group, error	1, 2	1, 2	1, 2	1, 2	1, 2	1, 2
F	0.14	8.30	12.53	111.08	4.01	11.44
P	0.742	0.102	0.071	0.009	0.183	0.077

b)

	Spring			Autumn		
	Richness	PC1	PC2	Richness	PC1	PC2
1995	21.0 (2.1)	-0.4 (0.2)	-1.1 (0.1)	25.4 (2.4)	0.1 (0.2)	-0.6 (0.2)
2000 - 2004	17.7 (1.4)	-0.7 (0.1)	-0.6 (0.1)	18.0 (1.5)	-0.6 (0.1)	-0.8 (0.2)
df group, error	1, 4	1, 4	1, 4	1, 4	1, 4	1, 4
F	3.52	2.43	31.92	17.61	27.60	2.31
P	0.134	0.194	0.005	0.014	0.006	0.203

[5.5] Discussion

Post-treatment (2000-2004) invertebrate communities were richer in recently managed streams and had taxa more indicative of channel margins and lowland conditions than in controls. However, no 'treatment' effects were evident when considering only surveys undertaken immediately post-treatment in 2000, the only year in which survey data were available in all streams. Significant treatment effects observed between 2001 and 2004 could have been confounded by year effects since post-treatment surveys were undertaken in different years on different streams, notably treated streams were all surveyed in 2004 but controls were not. Alternatively treatment effects may not yet have become evident (Kondolf, 1993, Agouridis *et al.*, 2005), with consequences for in-stream habitat and invertebrates developing only progressively (Nerbonne and Vondracek, 2001).

A comparison of preliminary investigations undertaken in 1998 and 1999 on the Clywedog brook, immediately after management suggested that there was an increase in Perlodidae and Tipulidae and Leuctridae during that first year post-treatment (Ambler, 1998; Williams, 2000). Although this result was not explicitly tested, results from this chapter concur with the assertion that Perlodidae and Tipulidae and Leuctridae were greater in treated streams. All three families were positively loaded on PC1 which was higher in treated streams. So, treatment effects observed during 2000-2004 may have been genuine despite the confounding effects of sample years.

Previous studies on the Clywedog investigated different treatments (fence and coppice, fence only, fenced without trees and controls without a fence) within the same river (Williams, 2000; Luxton 2002). Differences in invertebrate communities between treatments were identified; in 1999 fenced and coppiced sites tended to have more Perlodidae, Baetidae and Tipulidae, whilst *Gammarus pulex* were most abundant in controls (Williams, 2000). Although these results may have been confounded by the pseudoreplication of surveys in close proximity within the same river, they suggest that there were localized treatment effects (Hurlbert, 1984). In this study, evaluation at the whole river-scale may have overlooked any small-scale habitat effects on invertebrate communities.

A disadvantage of using the Environment Agency's macroinvertebrate data was that the resolution of the data was limited to family-level, with abundance recorded semiquantitatively (see section 5.3.2). More precise taxonomy and quantification would instill greater confidence in results, and could potentially have yielded different results. For example, differences in feeding guilds across treatments may become evident if assessed from species rather than family data since not all members of a family share similar feeding preference (Moog, 1995). Averaging the feeding preference across each family, as done here, might therefore mask effects on trophic structure. Similar arguments might be made for families represented by PC axes.

Potential treatment effects on invertebrates, evident as a shift towards families more typical of river margins, imply that riparian alteration promoted marginal habitats. Although River Habitat Surveys undertaken in 2004 did not detect higher occurrences of channel vegetation there was significant variation between rivers and filamentous algae was more frequent in treated streams (see Chapter 4). Changes in amphibious habitat that were sufficient to change stream biota could have been beyond the resolution of the RHS or too subtle to identify amongst the diversity of rivers within the catchment.

Temporal trends in invertebrates across all reaches indicated that families typical of lowland, more eutrophic rivers were favoured at the expense of 'upland' families, thus reducing taxon richness. The lowland shift was most evident in treated reaches which might reflect aspects of the rehabilitation programme that were developed for lowland chalk streams (Giles and Summers, 1996; Summers *et al.,* 2005). In lowland streams, the coppicing of riparian trees is not only designed to allow bank vegetation to develop, thus stabilising the bank, but also to promote in-stream vegetation as energy sources and refuges for fish. In low-gradient streams, macrophytes and algae proliferate and 'channel choking' is not uncommon (Holmes *et al.,* 1998; Environment Agency, 2003). Conversely, in the steep, erosive environment of small upland streams aquatic assemblages dominated by 'shredders' are often driven by allochthonous energy sources derived from bank-side trees (Vannote *et al.*, 1980; Rundle *et al.*, 1992). However, the establishment of in-stream flora may well support invertebrate assemblages that are indicative of more vegètated lowland reaches. Results hint at a shift from shredding to grazing assemblages. This suggests a move from allochthonous sources of energy to autochthonous sources as might be expected under more 'lowland' conditions (Vannote *et al.*, 1980).

Taxon richness tends to be greater in channel margins than in riffles and in Welsh upland streams increased with the amount of grassy vegetation (Ormerod *et al.*, 1993; Bradley and Ormerod, 2002). The increase in families indicative of margins could have slowed the decline in taxon richness in treated sites relative to controls thus causing the disparity between treated and control reaches post-treatment. Nevertheless, the general decline in richness and upland families implicates larger-scale factors in the structuring of stream communities (Bradley and Ormerod, 2001; Malmqvist, 2002; Johnson *et al.*, 2004).

Differences observed in invertebrate assemblages suggest that riparian restoration might have been sufficient to change aquatic biota. Fencing and coppicing regimes appeared to promote richer assemblages with more marginal, lowland species. Results support the role of the riparian zone in maintaining biodiversity but imply a shift away from communities typical of upland streams (Naiman *et al.*, 1993). However, further investigation is required to establish reasons for reduced richness among typical headwater organisms. Unfortunately, only tentative conclusions can be drawn from this study because treatment effects were confounded by year effects. This highlights the importance of experimental design and the need for project-specific data collection in the evaluation of management (Kondolf, 1995; Minns *et al.*, 1996; White, 1996; Bash and Ryan, 2002; Harrison *et al.*, 2004).

[5.6] References

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Chapter 6

The status of salmonid populations in the Wye catchment, Mid-Wales and impact of river management schemes.

[6.0] Abstract

Although salmon abundance is generally declining in England and Wales, local trends may differ from generalised pattern. Local catchment management could make a particularly important contribution to offsetting such effects, but few quantitative case studies are available.

In the UK, the Rivers' Trusts have pioneered ambitious schemes that aim primarily to augment river habitats for salmonids. However, the documentation and evaluation of such projects has been limited. One such project was initiated in 1996 by the Wye and Usk Foundation (WUF) and comprised a catchment-wide scheme of riparian management that aimed to reverse a perceived downward trend in salmonid populations across the Wye system.

The analysis of data from routine monitoring undertaken extensively in the Wye by the Environment Agency between 1985 and 2004 confirmed a decline in populations of Atlantic salmon (*Salmo salar*) and brown trout (*S. trutta*) (of 50% and 67% in juveniles, respectively) but trends varied between species and tributary streams (n = 41). Juvenile (>0+) salmon, juvenile trout and salmon fry (0+) all demonstrated declines but similar trends were not observed in trout fry. Trends through time indicated distinctions between streams with strongly decreasing mildly decreasing or increasing trends in salmonid populations prior to riparian management (1985-1997). Streams in which populations declined were defined as those within which abundance was negatively correlated with time. Declining trends in juvenile trout were evident in 66% of rivers, and declines in salmon were also observed in 66% of rivers in which this species were present during the study period.

The effects of recent riparian management on salmonid populations are so far negligible. Three possible explanations are that i) 6 years is too short a period to detect any response; ii) local trends in salmonids are governed by larger-scale processes or iii) effects are masked by other manipulations, for example the removal of barriers, that have led to wider fish dispersion within treated tributaries.

[6.1] Introduction

The abundances of many vertebrate species, including birds, amphibians and fish, have declined over the last century in response to widespread changes in land use and management (Pitcher and Hart, 1982; Fuller *et al.,* 1995; Noon and McKelvey, 1996; Kirk and Hyslop, 1998; Edwards *et al.,* 2000; Alford *et al.,* 2001). A framework of agreements, Directives (e.g. 92/43/EC and 2000/60/EC) and legislation such as the Ramsar and Bonn Conventions of the 1970's and the Convention on Biological Diversity in 1992 aim to conserve and where possible to boost native populations, often by provision and protection of suitable habitat (Dias, 1996; Rieman *et al.,* 2001; O'Connell and Yallop, 2002; Smith and Hellmann, 2002; Benton *et al.,* 2003). For some organisms however, populations are widely dispersed outside protected areas during some or all of their life cycles (Friedland, 1998; Thiollay, 2006). Moreover, habitats such as rivers and wetlands are often affected by processes over large spatio-temporal extents so that protected areas alone are insufficient to guarantee effective conservation (Buckton and Ormerod, 2002; Allan, 2004; Gaston *et al.*, 2006). For migratory fish in rivers, these two effects are compounded (Durance *et al.* 2006).

One option in these circumstances is to use local management or restoration in an attempt to drive population abundance in directions that deviate from global or regional trends (Pechmann *et al.,* 1991; Alford and Richards, 1999). Such restoration or management schemes should be appropriate to specific populations, while using evaluation methods that are suited to local conditions (Kondolf, 1995; Bash and Ryan, 2002; Downes *et al.,* 2002). In rivers, the catchment is seen as the most appropriate unit for management, with activities sufficiently dispersed to capture as a large a proportion as possible of threatened populations. Ideally, baseline data should be collected to identify whether populations respond in desired ways to management action. In many cases, however, organised data collection is overlooked or considered expensive so that alternative assessment methods must be sought.

Declines in fish populations have been widely reported, and in river systems are often attributed to pollution, climate change, exploitation pressure, habitat loss, effects of exotic species and fragmentation (Mills, 1971; Pitcher and Hart, 1982; Noakes *et al.,* 2000; Ormerod 2003). For example, the numbers of salmon from English and Welsh rivers alive in the sea on 1st January in the first sea winter (the pre-fishery abundance, PFA) and the numbers of returning salmon declined between 1970 and 2004 by 50 and 35 percent, respectively (Environment Agency and CEFAS, 2004).

Salmonid abundance is naturally variable over time both within populations and between populations, in response to environmental change and the genetic predisposition of the fish (Pitcher and Hart, 1982; Milner and Elliot, 2003). It follows that the national decline in salmon over the last three decades might not be reflected in every individual river (Environment Agency, 2003a; Environment Agency and CEFAS, 2004). For example, contrary to national trend, salmonid abundances in the Tyne, Wear, and Taff have increased over the last thirty years (Environment Agency, 2003a; Environment Agency and CEFAS, 2004).

A decline in salmon abundance on the River Wye, Mid-Wales is apparent from rod catches available since 1974 (Gee and Milner, 1980; Environment Agency, 2003a and b). This, along with anecdotal evidence from local angling groups, prompted a scheme of riparian management of tributary streams within the Wye catchment, to promote nursery habitat for salmonids. This catchment-wide management scheme, launched by the Wye and Usk Foundation (WUF) in 1996, aimed to reverse a perceived downward trend in salmonid populations across the Wye catchment. Riparian management undertaken on tributaries of the Wye catchment was based on the assertion that there was a downward trend in salmonid density that might be reversed, at least in managed locations.

This chapter uses routinely collected data to determine whether management effects can be detected against a background of spatio-temporal variations in salmonid density across the Wye catchment. Given likely variation of trends in salmonid abundance across British rivers, and unreliability of rod-catch statistics in general, any evidence for a decline in salmonid density across the Wye catchment was first tested (Bielak and Power, 1988; Mills, 1989). Second, rivers with similar trends in salmonid density between 1985 and 1997 were identified. Then, the effect of riparian management on treated streams (post 1998) was tested against reference streams that had demonstrated similar trends in salmonids during the pre-treatment period (1985-1997). Research questions addressed were therefore i) was there a decline in salmonid density over time? ii) was a general decline salmonid populations evident in all tributary rivers prior to management? iii) did riparian management have an effect on salmonid populations?

[6.2] Study Area and Riparian Management.

The Wye system has been extensively described by Edwards and Brooker (1982), Ormerod (1987, 1988), Ormerod and Edwards (1987) and Jarvie *et al.* (2003) and also in Chapter 3 of this thesis.

Traditionally, the River Wye was famous for its spring-run salmon (Environment Agency, 2003b). Atlantic Salmon (*Salmo salar*) are now explicitly mentioned on the Habitats Directive (Directive 92/43/EEC) and are a species cited in the Sites of Special Scientific Interest (SSSI) and Special Area of Conservation (SAC) designations of the Wye (Environment Agency, 1998; Environment Agency, 2000; Joint Nature Conservancy Committee, 2004). Over the last 30 years, a downward trend in rod catches has caused concern for the salmon fishery in particular (Environment Agency, 2003b). Rod catches of trout (*Salmo trutta*) have also declined in the Wye (Environment Agency, 2003a).

In the UK, techniques to create optimum habitat for salmonids by the management of riparian habitat have been applied primarily on lowland chalk rivers (Salmon Advisory Committee, 1991; Giles and Summers, 1996; Holmes, 2002). Such management was designed to promote spawning and nursery habitat for salmonids and autochthonous production (see Chapter 2). Additional benefits might include reductions in livestock poaching (trampling), and shading of channels that limit bank vegetation and contribute to bank destabilisation (García de Jalón, 1995, Hendry *et al.*, 2003). Siltation of spawning gravels and less complex habitat often result (Hemphill and Bramley, 1989, Salmon Advisory Committee, 1991). Management of the riparian zone to create optimum habitat for salmonids usually involves fencing of a riparian buffer and coppicing or thinning of bank-side trees (Hemphill and Bramley, 1989; Hendry and Cragg-Hine, 1997). The primary aim is to encourage riparian vegetation in order to stabilise banks and reduce silt input to the river from the banks themselves as well as from runoff from the surrounding landscape. Trees are coppiced to promote bank and channel vegetation (O'Grady, 1993). Reduced shading of banks may encourage more complex vegetation structure, thus stabilising banks. In addition, allowing more light into the stream channel can promote the growth of macrophytes and algae within the stream. In-stream and amphibious plants may then increase the complexity of the channel and provide refuges for fry and juvenile fish.

Riparian management work on the River Wye, Mid-Wales, aimed to change the in-stream environment to increase the quality and quantity of habitat available to salmonid fish (The Wye and Usk Foundation, 2006a, b). Management of riparian habitat was implemented on the tributary rivers of the River Wye in Mid-Wales from 1996. This study examines the effect of habitat alteration on salmonid density in two rivers, the Clywedog and the Edw, on which extensive management which began in 1998 and was completed in 1998 and 1999 respectively (Figure 6.1). Alder (*Alnus sp.)* and willow (*Salix sp.)* trees on the river banks were coppiced and approx 1-3m of the riparian zone was fenced to exclude grazing. In 1996, obstructions which may have impeded salmonid migration to spawning grounds were removed from the tributaries of the upper and mid Wye (Figure 6.2; The Wye and Usk Foundation, 2006c). The programme included the removal of debris dams and installation of fish passes where practical.

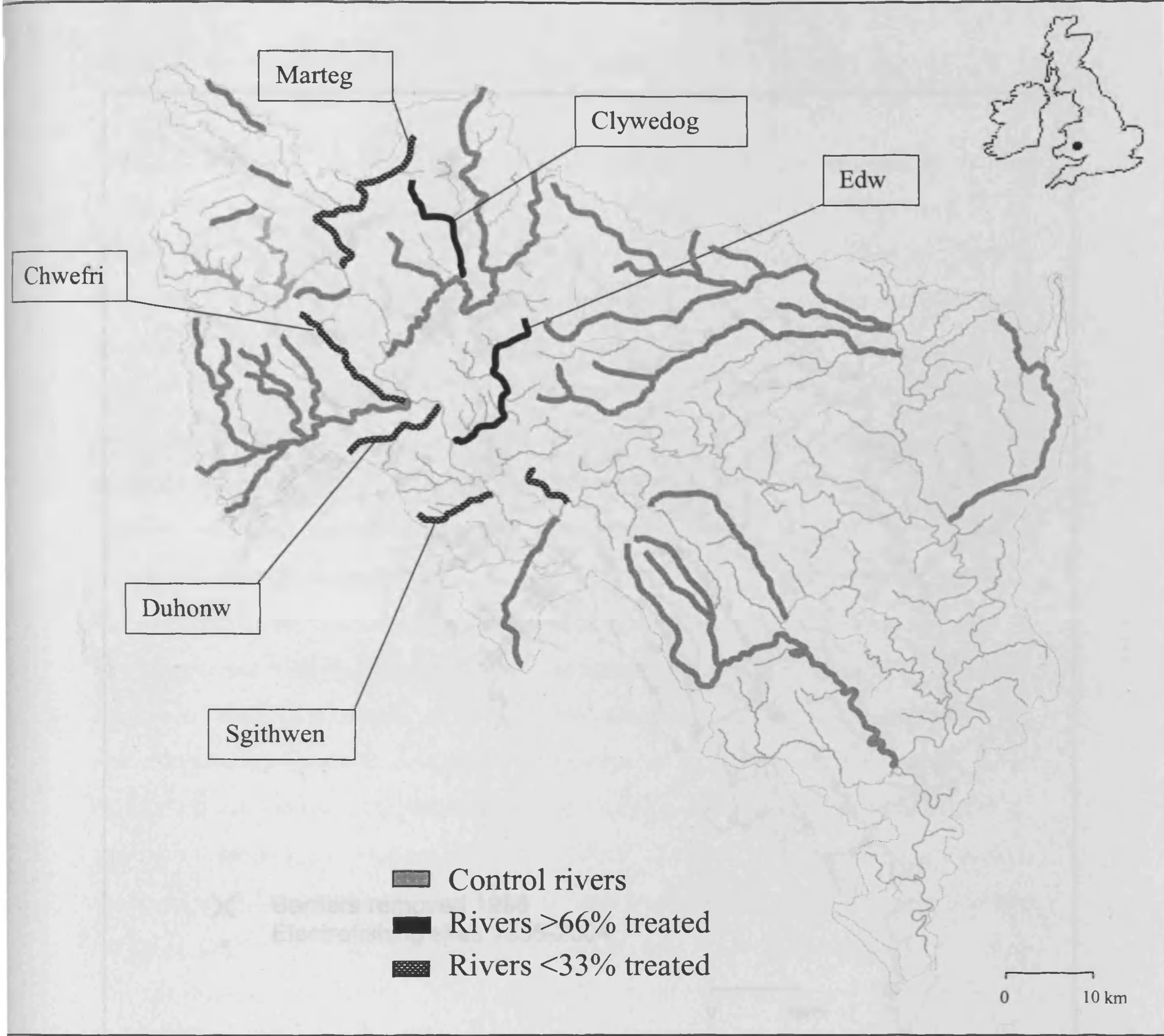

Figure 6.1. The Wye Catchment, Mid-Wales. Location of rivers that were treated with riparian habitat management (1996 to 2003) and rivers used as controls. Treated rivers were classified as those which had been treated along >66% and <33% of their length respectively.

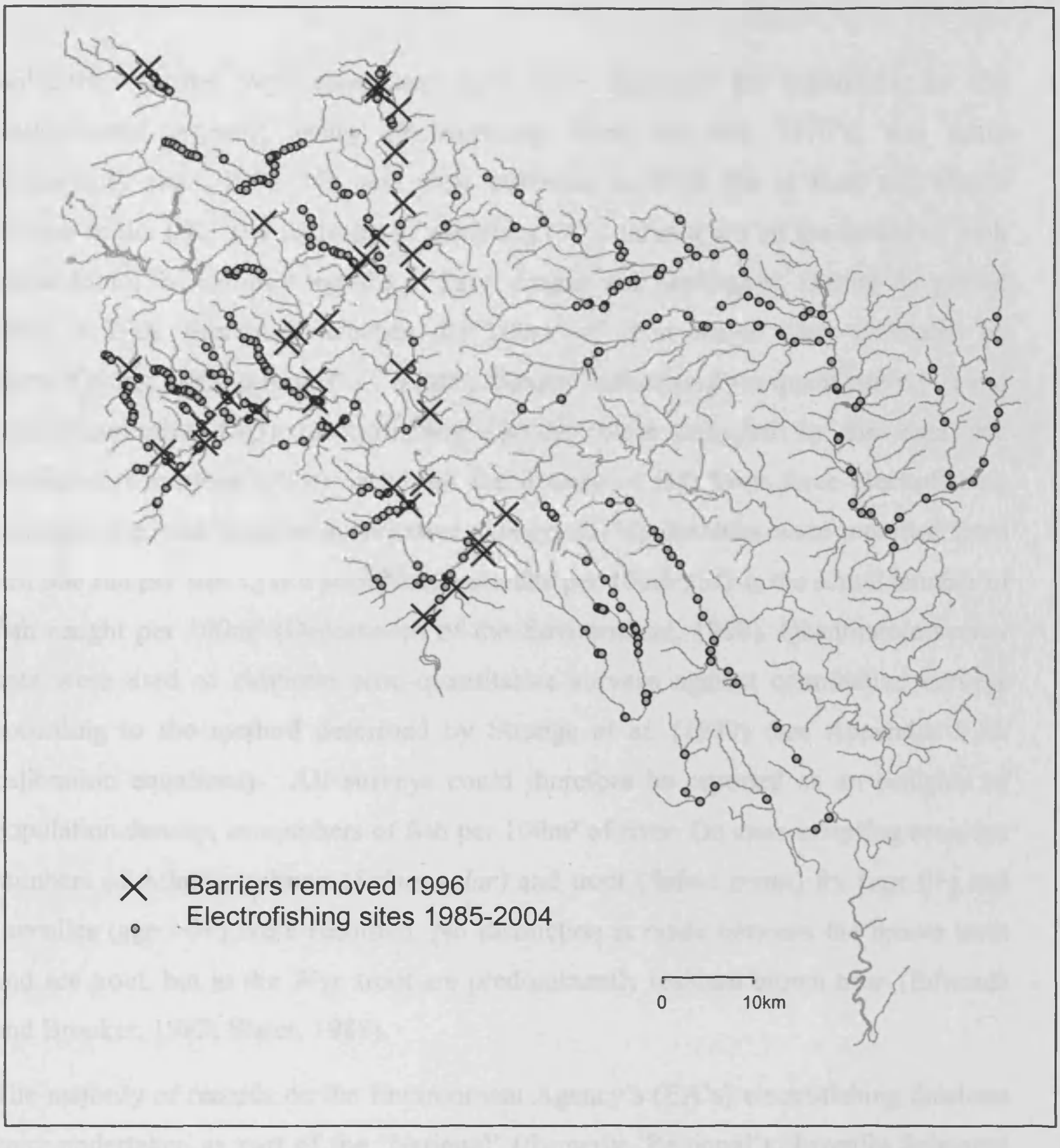

Figure 6.2. Location of obstructions removed from tributaries of the Wye in 1996 and electrofishing sites sampled during the period from 1985 and 2004.

[6.3] Methods

Tributaries in the Wye catchment have been surveyed for salmonids, by the Environment Agency, using electrofishing since the late 1970's, and more consistently since 1985. No data were collected in 2000 due to Foot and Mouth disease in the UK. The majority of sampling (91% of samples on the database) took place during the summer months of July, August and September. During the period 1985 to 2002 salmonid densities per $100m^2$ of river length were estimated by electrofishing using pulsed D.C. output. Density estimates from quantitative (Q) and semi-quantitative (SQ) electrofishing surveys were included in the data set. Quantitative surveys (29%) estimated the density of fish from three electrofishing runs per site, and in semi-quantitative surveys (71%) densities were recorded from just one run per site. Q is a population estimate per $100m^2$, SQ is the actual number of fish caught per $100m^2$ (Department of the Environment, 1988). Quantitative survey data were used to calibrate semi-quantitative surveys against quantitative surveys according to the method described by Strange *et al.* (1989) (see Appendix I for calibration equations). All surveys could therefore be reported as an estimate of population density, as numbers of fish per $100m^2$ of river. On each sampling occasion numbers of Atlantic salmon (*Salmo salar*) and trout (*Salmo trutta*) fry (age 0+) and juveniles (age >0+) were recorded. No distinction is made between the brown trout and sea trout, but in the Wye trout are predominantly resident brown trout (Edwards and Brooker, 1982; Slater, 1988).

The majority of records on the Environment Agency's (EA's) electrofishing database were undertaken as part of the 'National' (formerly 'Regional') 'Juvenile Salmonid Monitoring Programme' (NMP), for which 'temporal' and 'spatial' sites were surveyed across England and Wales. Sites intended for spatial comparison of salmonid populations were monitored on a 5 year rolling programme, and sites which were monitored primarily for use in temporal analyses were surveyed annually (Environment Agency and CEFAS, 2004). Other surveys included those carried out for Environmental Impact Assessments (EIA's) and post-stocking. All such surveys were included in analyses.

One of the aims of this chapter was to identify variation in salmonid densities between the tributaries of the Wye. Data from sites within each tributary were therefore pooled. Grouping data by tributary also helped to overcome some of the difficulties imposed by a non-uniform set of data. The number of tributaries sampled varied from year to year, partly because of the initial design and subsequent modification of the NMP, and partly because of the inclusion of *ad hoc* surveys in the data set. Precise locations of surveys within rivers also altered over the years, which limited the scope for direct comparison of individual sites.

Since long-term trends were of interest over an eighteen year period, any river which had fewer than 5 replicate years of data prior to treatment (up to and including 1996) was excluded from analyses. Similarly, when data obtained before and after treatment were compared, rivers with less than 3 replicate years of data post-treatment (from 1997 to 2002) were excluded. A total of 37 rivers were included in this study, including two (Clywedog and Edw) on which over 66% of the length of riparian habitat was altered from 1998 (Figure 6.1; exact locations confidential). Not every river was sampled in each year, and from 1997 the frequencies of rivers sampled each year were less consistent than they had been in the previous 10 years previously (1985 – 1995) (Appendix II).

The majority of rivers sampled were electrofished just once each year; however some rivers were sampled more than once or had multiple sample sites. A mean annual density was therefore calculated for each river. Analyses were performed separately for each salmonid species age-group, using the SPSS statistics package (Version 11.5). Those rivers for which there were no records of either juvenile salmon or salmon fry were excluded from analyses of juvenile salmon or salmon fry respectively.

Densities of salmon fry, juvenile salmon, trout fry and juvenile trout were normalised by log (n+1) transformation prior to analyses, according to protocols outlined in Fry (1993). Transformed densities of each salmonid species age-group were then standardised by river, to avoid the bias of the results by any one river which may have had numerous fish to start with (Sokal and Rohlf, 1995). Each river therefore had a mean density of 0 and standard deviation was equal to 1. The resulting z scores were used in all analyses.

[6.3.1] Trends in salmonid density over time

In order to establish whether there was a general decline in salmonid density over time in the Wye density data were grouped into 5 subsets of years (1985-1988, 1989-1992, 1993-1996, 1997-1999 and 2000-2004). This removed any autocorrelation of density between years, identified from plots of autocorrelation functions (SPSS for Windows, 2001; Tabachnick and Fidell, 2001). When values of annual density were missing for a particular river, they were replaced with the mean z scores for that river in the relevant time period. Linear relationships between relative salmonid density (standardized by river) between time periods were then tested for by regression. (Sokal and Rohlf, 1995).

Correlation analyses on the frequency of years that each river was sampled and densities of salmonids (standardized by river) in the Wye catchment were used to identify rivers with sampling frequencies that may have obscured trends in density. There were rivers with sampling frequencies that correlated with the general trends observed in salmon fry and juveniles and trout fry densities in the Wye catchment (Appendix II). Regression analyses were then repeated without those rivers to establish whether trends observed were symptomatic of the frequency of sampling.

[6.3.2] Variations between tributaries

Correlation between salmonid densities and time were used to discover whether a general decline in salmonid populations was evident in all tributary rivers prior to management. Spearman's correlation coefficients (r_S) described the monotonic correlation between mean annual salmonid density and year for each tributary during the period 1985 to 2004 (Sokal and Rohlf 1995). Each coefficient was taken to represent the temporal trend in salmonid density for each river. Positive correlation coefficients, approaching 1.0 could be said to represent tributaries with an increasing trend in salmonid density, whilst negative coefficients, approaching -1.0, represented tributaries with declining salmonid populations.

If there was no general trend of salmonid decline across the catchment, a normally distributed population of r coefficients (r_S) centred on a mean of zero change in density would be expected. The expected frequencies of r_S in this 'expected' population were calculated as a normal distribution of r_S with the same standard deviation as the observed population but a mean r_S equal to 0. The normal distribution of 'observed' populations of r_S was confirmed by Kolmogorov-Smirnov tests.

The departure of the 'observed' correlation coefficients from those 'expected' was tested by Chi-squared goodness of fit tests (Sokal and Rohlf 1995). Frequencies of observed r_S values were tested against expected r_S values for four categories of r_S values (-1>-0.5, -0.5 > 0, 0 < 0.5, and 0.5 < 1.0).

If greater numbers of tributaries had densities of salmonids that were negatively correlated with year greater than that which may be expected due to random chance, the general assumption of declining salmonid populations in the Wye catchment would be supported.

[6.3.3] The effects of riparian management

A Before-After-Control-Impact (BACI) method was employed to establish whether riparian management had an effect on salmonid populations. It compares the difference between control and treated (impacted) rivers, before and after treatment (impact).

First, suitable groups of control rivers were identified for each of the treated rivers. Rivers were grouped into those that demonstrated similar trends in density prior to treatment; between 1985 and 1995 for streams from which obstructions were removed and 1985 and 1997 for those with altered riparian habitat (Appendices III and IV). This was done by taking the r_S describing the correlation between annual densities of salmonids and year (from 1985 to 1995) and grouping rivers by four categories of r_S (Table 6.1). Control rivers were then allocated to treatment rivers that were classified similarly prior to treatment.

Mean annual densities of each salmonid species age-group were standardized by river to give a mean annual value of relative density. The 'BACI' analyses was then performed on the relative densities of control and treatment rivers from each group of tributary rivers (Table 6.1; Stewart-Oaten *et al.,* 1986; Downes *et al.,* 2002). A two sample *t*-test compared the mean differences in relative density (density in the 'control rivers' minus the density in the 'treated river(s)') before and after treatment. Divergence of treated rivers from controls post-treatment would demonstrate a treatment effect.

Table 6.1. Method of classification of tributary rivers with similar temporal trends in salmonid density prior to treatment.

Group	Spearman's correlation coefficient (r_S).	Strength of correlation between salmonid density and year.	Inference of trend in salmonid population.
4	0.5 < 1.0	Strong positive	Increasing population
3	0.0 < 0.5	Weak positive	Slight increase
2	-0.5 > 0.0	Weak negative	Slight decrease
1	-1.0 > -0.5	Strong negative	Declining population

If treatment was effective, a lagged response to riparian alteration would be anticipated (Kondolf, 1993). This could result in a relatively large variation in the difference in density (treatment minus control) during the post-treatment phase. Conversely, the control sites were chosen for their similarity to the treated sites and there would be little variation in the difference between them prior to treatment (Table 6.1). The *t* statistics were therefore adjusted for unequal variances in each case (Quinn and Keough, 2002).

Rivers which had only limited riparian alteration (<33% of their length) and tributaries on which obstruction removal had taken place in 1996 were excluded from control groups (Appendix III; Figure 6.1). The Edw and Clywedog therefore represented the rivers with riparian-treatment, which was completed in 1998 on the Clywedog and 1999 on the Edw. To account for multiple tests on species-age groups of salmonids, 'Bonferroni' adjustments were made independently for each river, since treatment occurred at different times (Quinn and Keough, 2002).

The effect of barrier removal on salmonid density was similarly tested. Five tributary streams (the Aran, Cammarch, Chwefri, Sgithwen, and Nantmel Dulas) had sample sites that were all above the point at which a blockage was cleared in 1996 (Figure

6.2; Appendix IV). A second BACI analysis tested for the effects of obstruction removal, by comparison of treated and control streams prior to and after the winter of 1996 (Table 6.1; Appendix IV).

[6.4] Results

[6.4.1] Trends in salmonid density over time

A decline in density was evident in juvenile salmon and fry and juvenile trout between 1985 and 2004 across the Wye catchment (Figure 6.3, Table 6.2). Densities were also highly variable within and between years and suggested some non-linear change through time (Figure 6.3). Trends observed in salmonid densities were not affected by the frequency of rivers sampled in each year (Figure 6.3; Appendix II).

[6.4.2] Variations between tributaries

Declining densities of salmon fry, juvenile salmon, and juvenile trout between 1985 and 2004 were evident in more than half of the streams (Figure 6.4). Observed frequencies of correlation between density and year were normally distributed but mean coefficients (r_s) for salmon fry, juvenile salmon and juvenile trout were negative (-0.4, -0.4, -0.6 respectively). The densities of trout fry did not deviate from those expected due to random chance, with approximately equal numbers of rivers having increasing and decreasing trends in fry (mean r_s of -0.1; Figure 6.4).

Prior to riparian management (1985-1997) on the Edw, juvenile trout demonstrated a strong decline (r_s between density and time = -0.6, in Group 1; Appendix III), and trout fry and juvenile salmon declined less dramatically (Group 2; Appendix III). Weakly declining trends were also observed in salmon densities (fry and juveniles) in the Clywedog. Conversely, weakly positive correlations were observed between year and densities of trout (fry and juveniles) in the Clywedog, and salmon fry in the Edw (Group 3; Appendix III). None of the treated streams demonstrated strong positive increases in salmonid densities in the pre-management period (Group 4; Appendix III).

When trends prior to management were determined from pre-1995 densities, species-age classes of salmonids from treated streams were assigned to similar control groups as they had been when the pre-treatment period extended through to 1997

(Appendices III and IV). Exceptions were juvenile trout in the Clywedog and salmon fry which were placed in either group 2 or 3.

[6.4.3] The effects of riparian management

No positive effects of the removal of barriers were detected in salmonid populations (Appendix V). In fact, a 54% decrease in salmon fry density was observed after 1996 on the Nantmel Dulas. Similarly, riparian management had very little effect on fish density. Only densities of juvenile trout in the Edw suggested a positive response to management, having demonstrated marginally significant treatment effects (Table 6.3; Figure 6.5). In that instance, density continued to decline in control groups, whilst numbers increased in the treated river (Figure 6.5).

Table 6.2. Regression of salmonid densities against five time periods between 1985 and 2004 **a)** within the Wye catchment **b)** excluding rivers for which survey frequency was correlated with salmonid densities in the Wye (Appendix II) (***P<0.001).

a)

Time Period	Salmon Fry	Juvenile Salmon	Trout Fry	Juvenile Trout
1985 - 1988	68.8 (5.9)	7.9 (0.7)	8.6 (1.2)	10.5 (0.7)
1989 – 1992	58.1 (6.1)	6.3 (0.7)	9.6 (2.1)	6.9 (0.6)
1993 - 1995	57.2 (6.9)	5.7 (0.7)	11.8 (2.1)	9.1 (0.9)
1997 - 1999	38.1 (4.1)	4.7 (0.4)	14.8 (2.9)	5.6 (0.5)
2000 - 2004	44.6 (5.2)	3.9 (0.6)	13.5 (3.0)	5.4 (0.6)
df $_{\text{total, error}}$	535, 534	522, 521	554, 553	554, 553
R^2 **%**	12.7	13.8	0.0	7.0
F	77.6***	83.5***	1.27	41.3***

b)

	Salmon Fry	Juvenile Salmon	Trout Fry	Juvenile Trout
df $_{\text{total, error}}$	467, 466	456, 473	474, 473	474, 473
R^2 **%**	14.7	17.8	0.1	8.0
F	80.2***	98.4***	0.3	41.3***

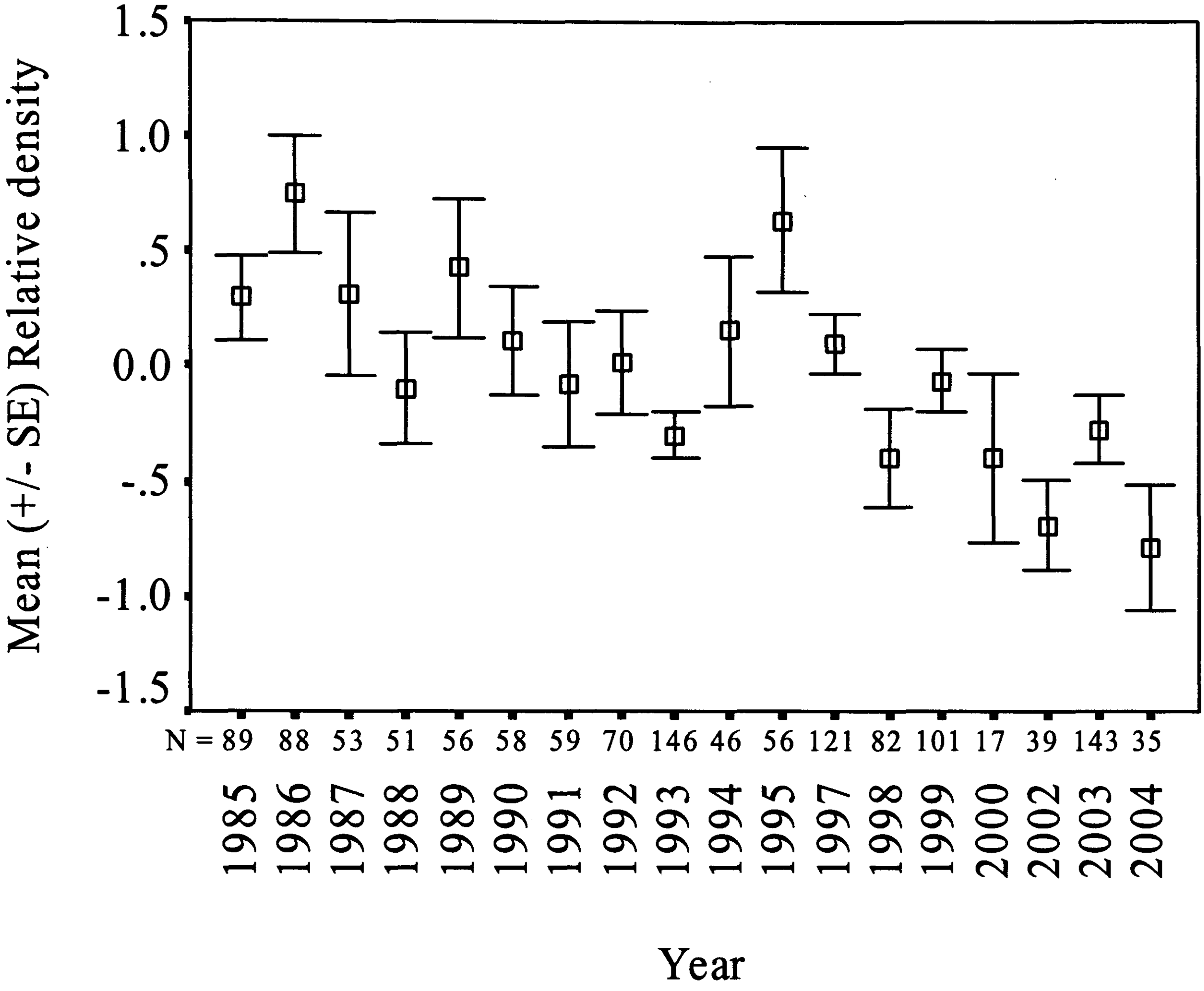

Figure 6.3 a). Relative densities of salmon fry in the Wye tributaries 1985 – 2004. Densities were estimated as number per $100m^2$ of river. Mean (±SE) Z scores, standardised by river are displayed (N indicates the number of rivers that contributed to the mean annual density of the catchment).

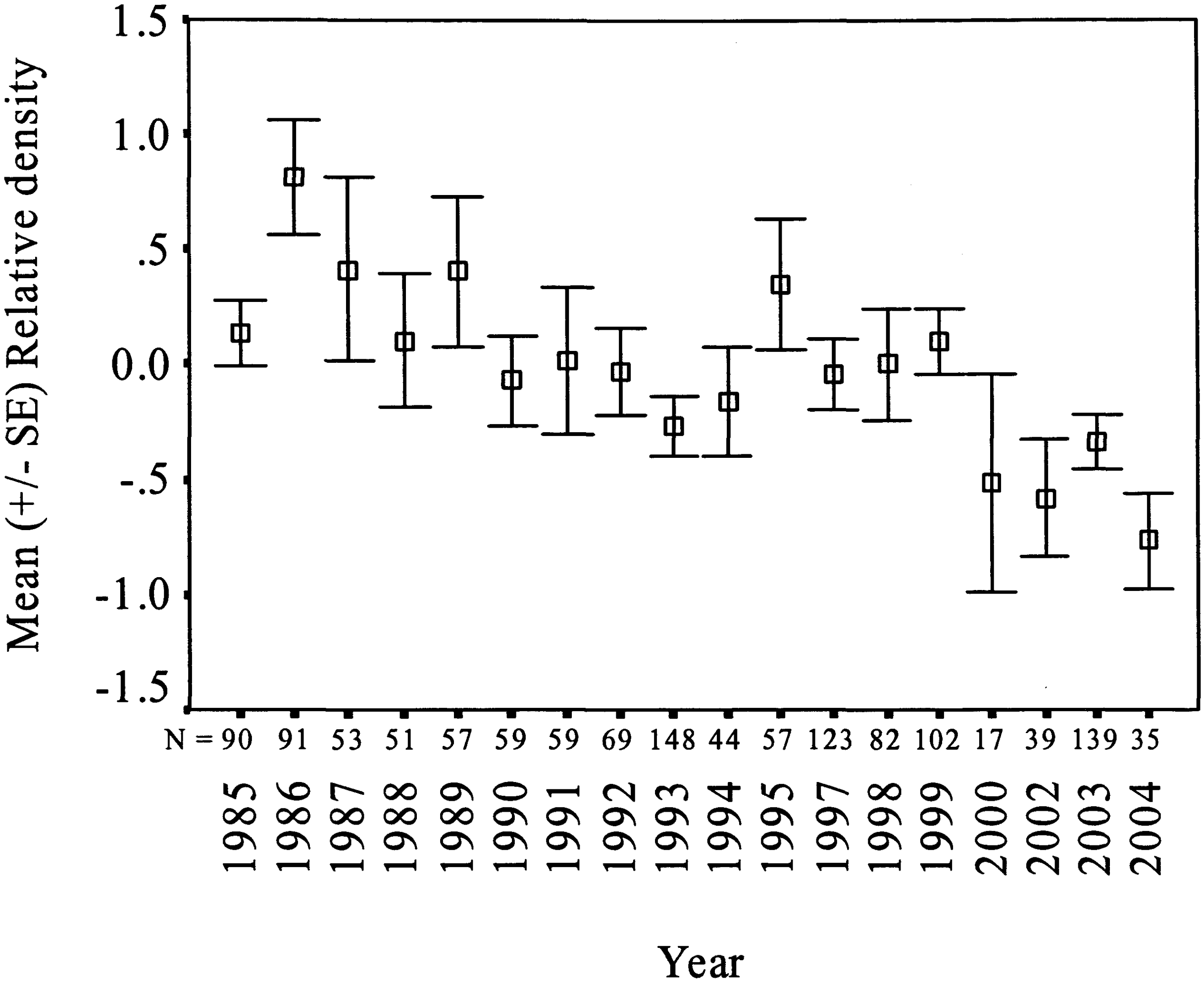

Figure 6.3 b). Annual densities of juvenile salmon in the Wye tributaries 1985 – 2004. Densities were estimated as number per 100m^2 of river. Mean (±SE) Z scores, standardised by river are displayed (N indicates the number of rivers that contributed to the mean annual density of the catchment).

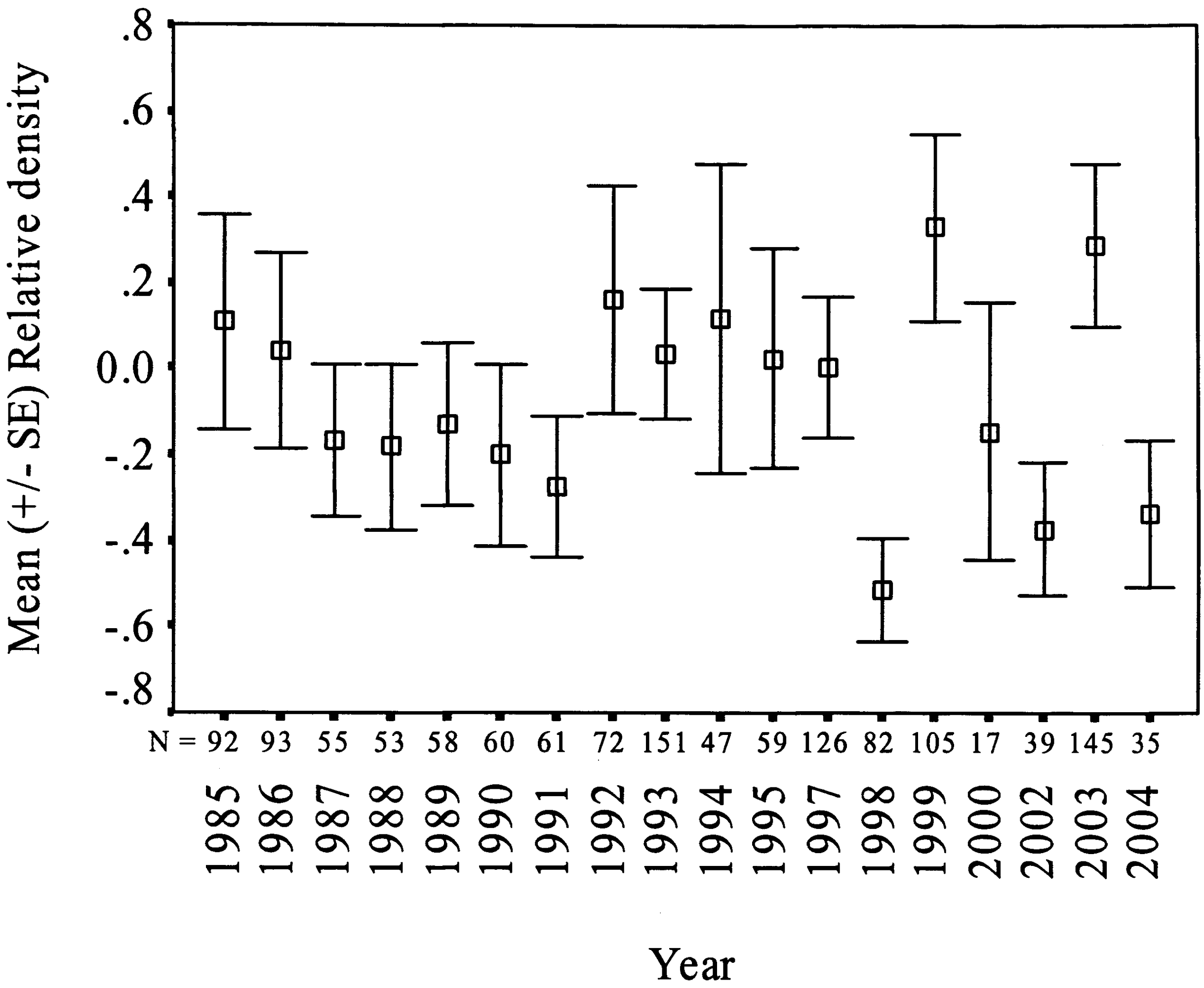

Figure 6.3 c). Annual densities of trout fry in the Wye tributaries 1985 – 2004. Densities were estimated as number per 100m^2 of river. Mean (±SE) Z scores, standardised by river are displayed (N indicates the number of rivers that contributed to the mean annual density of the catchment).

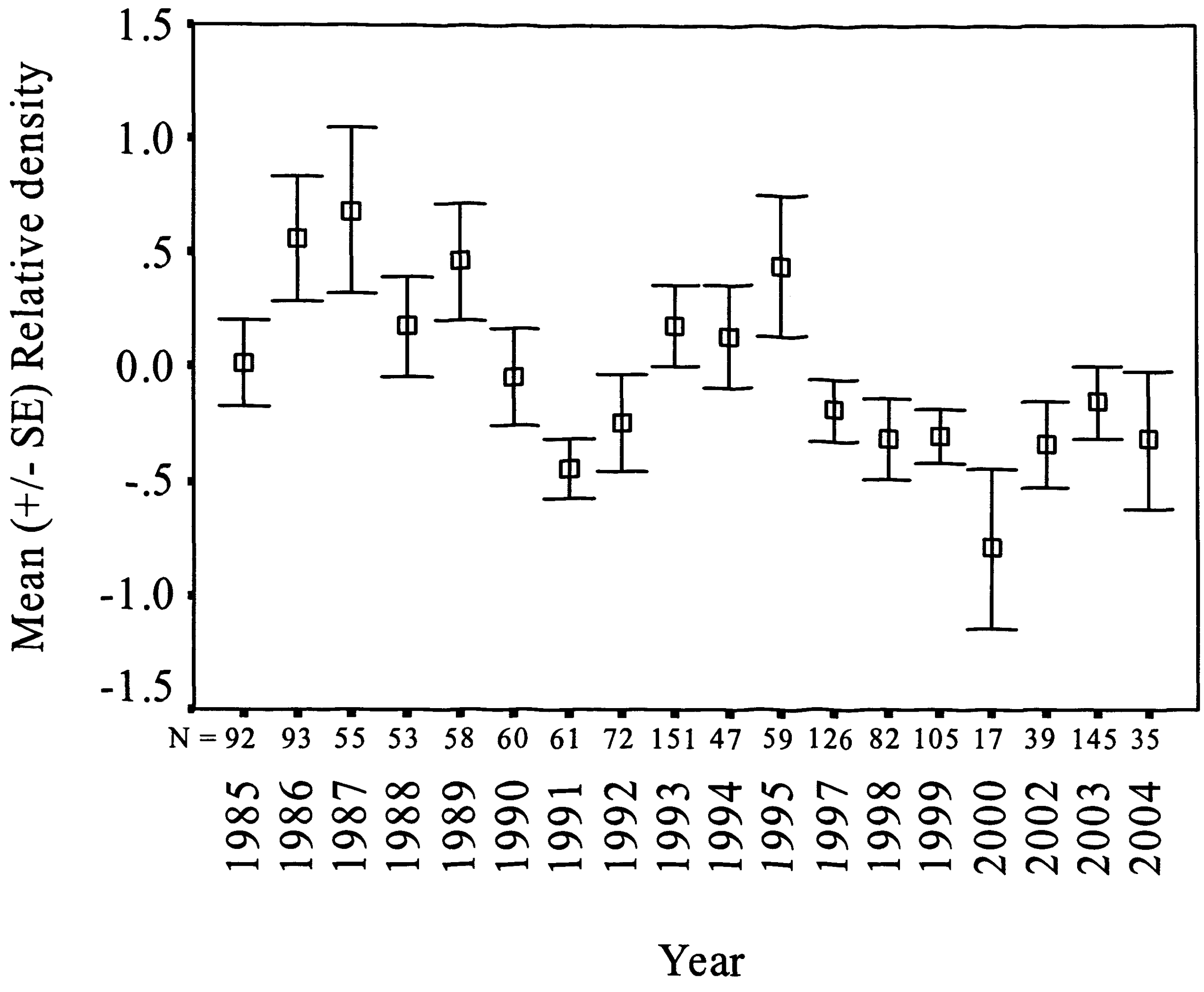

Figure 6.3 d). Annual densities of juvenile trout in the Wye tributaries 1985 – 2004. Densities were estimated as number per $100m^2$ of river. Mean (±SE) Z scores, standardised by river are displayed (N indicates the number of rivers that contributed to the mean annual density of the catchment).

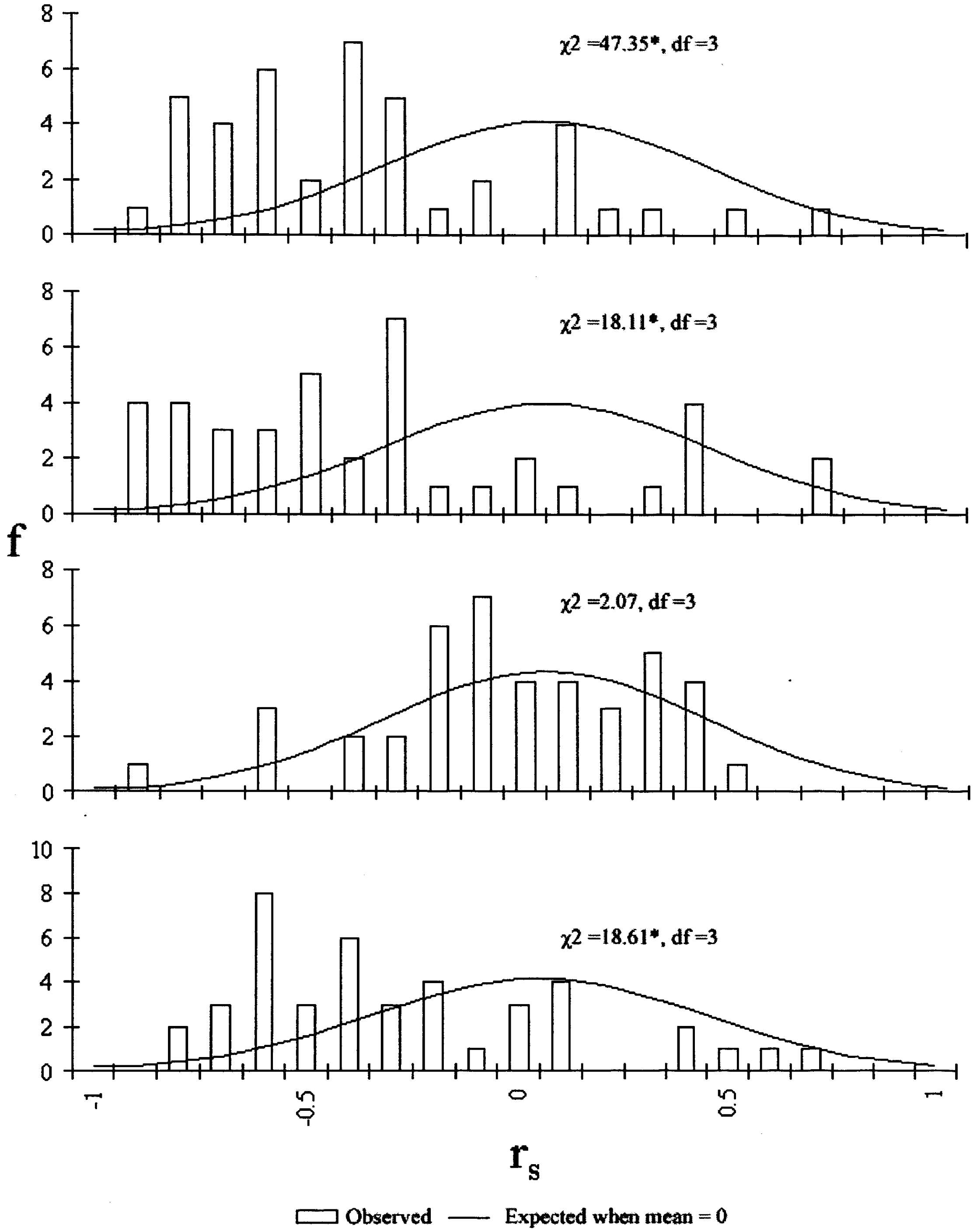

Figure 6.4. Variation across streams in trends in salmonid density prior to treatment. Observed and expected frequencies of rivers (f) with increasing and decreasing trends in salmonid populations over time as defined by Spearman's correlation coefficients (r_S). The expected curve is a normal distribution curve with a mean of zero and the same standard deviation from the mean as the observed data. Chi-squared (x^2) tested the goodness of fit between observed and expected trends indicated by (*P>0.05).

Table 6.3. The density of salmonids (Mean ± SE) before and after the completion of management work on the Clywedog and Edw, in 1998 and 1999 respectively. Differences between the relative densities of salmonids (standardized by river) in treated and control streams (Control minus Treatment) were tested for by two-sample *t*-tests, *t*-values are shown in bold. Degrees of freedom were adjusted for unequal variances when appropriate. P values prior to 'Bonferroni' adjustment for multiple tests are displayed.

		Clywedog		**Edw**	
		Control Group Mean (SE)	Treatment Mean (SE)	Control Group Mean (SE)	Treatment Mean (SE)
Salmon Fry	pre- treatment	48.7 (5.4)	69.2 (7.9)	44.3 (3.1)	156.2 (17.5)
	post- treatment	36.5(3.8)	53.4 (6.8)	40.1 (16.2)	196.7 (37.8)
	Df	7.36		5.185	
	t	**-0.464**		**-2.032**	
	P	0.656		0.096	
Juvenile Salmon	pre- treatment	6.2 (0.4)	7.5 (1.4)	6.2 (0.4)	12.8 (2.1)
	post- treatment	4.8 (1.4)	1.3 (0.6)	5.0 (1.1)	11.3 (4.5)
	Df	16.0		4.7	
	t	**0.570**		**-0.800**	
	P	0.576		0.462	
Trout Fry	pre- treatment	10.2 (2.1)	13.6 (4.6)	7.8 (1.4)	3.6 (1.2)
	post- treatment	17.8 (8.1)	3.5 (3.4)	6.2 (2.3)	4.0 (1.4)
	Df	14.2		9.7	
	t	**2.401**		**-0.730**	
	P	0.031		0.483	
Juvenile Trout	pre- treatment	7.2 (0.7)	4.4 (0.9)	7.9 (1.2)	4.3 (0.8)
	post- treatment	5.3 (0.2)	2.0 (0.8)	2.9 (0.7)	11.5 (3.3)
	Df	11.8		4.368	
	t	**1.004**		**-3.619**	
	P	0.335		0.019 †	

† After Bonferroni adjustment for multiple tests (P/4), treatment effects are evident at the 90% level of confidence.

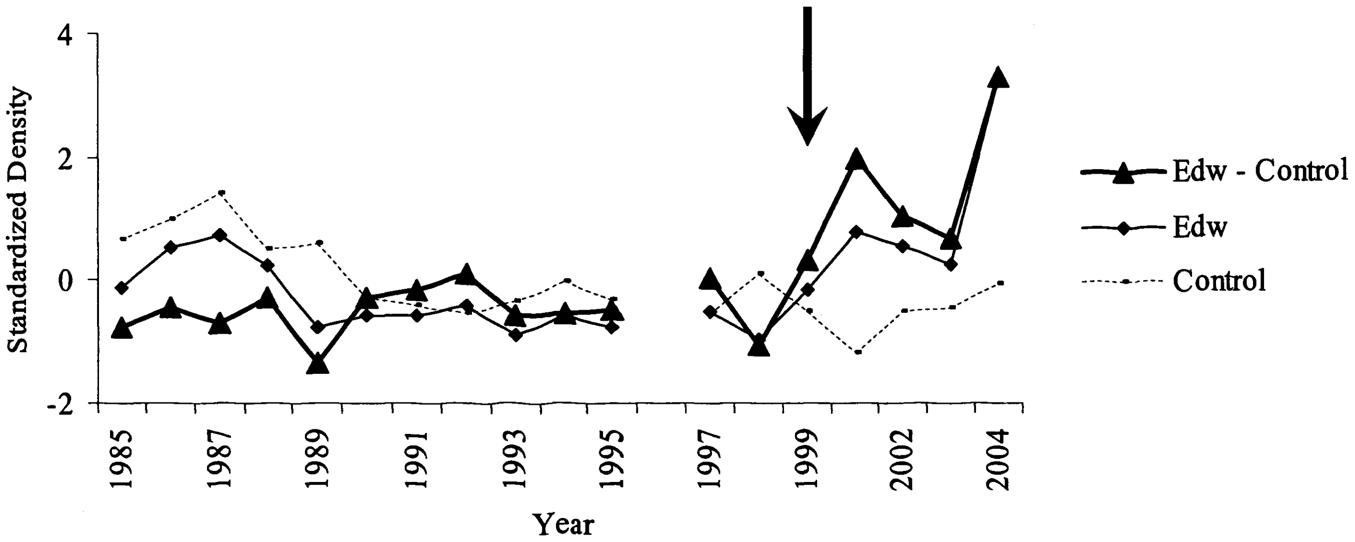

Figure 6.5. The relative density of juvenile trout in the Edw, control streams (Control Group 1) and the difference between the treated and control streams, 'Edw - Control' (1985-2004). The arrow indicates when riparian management took place (1999).

[6.5] Discussion

The primary aim of this chapter was to establish the effectiveness of management works in elevating salmonid populations in the Wye system. The first challenge of this study was to establish a baseline against which to evaluate management in the absence of project-specific monitoring.

Routine annual monitoring was sufficient to detect long-term trends (over 19 years) within tributary streams. Salmon and trout populations were predominantly in decline within the River Wye system during the period 1985-1998, prior to management. This confirmed the premise for intervention (The Wye and Usk Foundation, 2006a; Luxton, 2002). Those baseline data were used to identify suitable controls against which to evaluate management. Attempts to improve riverine habitat did not reverse declining trends in salmonid populations in the Wye catchment within the first 5 years post-treatment.

There are a number of possible reasons why no effect of treatment was detected in the Wye. The (whole river) scale of investigation could have been too coarse to detect the effects of management. Ecological restoration often takes time to effect change and perhaps an effect will become evident in time. Alternatively, either the location of works or the techniques applied may not have been entirely appropriate. A further possibility is that factors driving population declines operated over much larger, regional or global, scales and masked the effects of habitat alteration.

Population densities of salmonids were recorded at one site, often within the downstream reaches of rivers, on an annual basis. On the Clywedog and Edw the vast majority of the river length was fenced, coppiced or both but management was not entire and often dependent on landowner compliance. It is possible that changes affected by management elsewhere along the river may not have been detected at the study reach. Despite being relatively crude, electro-fishing methods employed were able to detect long-term change within populations. If a reversal of decline were affected within the river a change might also be anticipated at the study site. Management aimed to increase populations of salmonids within tributary streams of the Wye catchment. Results suggest that, up to 2004, restoration at the whole-river scale was not in effect.

A lagged response of biota to river management is frequently reported (e.g. Scruton *et al.,* 1998; Moerke and Lamberti, 2003). It takes time for riparian vegetation to develop before it can provide habitat cover, increase bank stability and buffer against siltation (Kondolf, 1993). Even if or when habitat conditions are optimal for juvenile salmonids there may be a lag before subsequent generations are able to take full advantage of those resources. For example, if an optimal habitat were created, egg survivorship in clean gravels might improve, the following year fry emergence might be affected which would in turn impact on juvenile density in the year after that (Armstrong *et al.,* 2003). A significant change in density might not be affected until that first cohort spawned its own young; some 4 years later (Slater, 1988; Klemetsen *et al.,* 2003; Milner *et al.,*, 2003).

It is possible that juvenile trout populations in the Edw demonstrated a lagged response to management. In 2004 post-treatment densities continued to rise, while populations control streams (in which populations demonstrated a similar decline prior to management) continued to decline. Continued investigation of this river could establish whether populations are truly recovering and rule out the effects of random population fluctuations (Pechmann *et al.,* 1991).

In the absence of data both above and below obstructions and without any baseline 'downstream' data for comparison, the effects of barrier removal were particularly difficult to ascertain. Survey sites that were upstream of removed barriers were few (n=5) and often confounded by limited riparian or other work, such as gravel cleaning. No changes in populations were detected at study sites upstream from blockage removal which suggests that barrier removal at those locations did not elevate salmonid populations. However, there were a number of locations (n=33) at which barriers were removed or fish passes installed at which no data were available.

While fish passes at weirs and the removal of barriers that impede salmonid migration can help populations to expand their range, the removal of 'log-jams' and large woody debris from streams is controversial (Hendry *at al.,* 2003). Juvenile and spawning habitats of Pacific salmonid have benefited from the addition of large woody debris to streams in the American Pacific Northwest (House and Boehne, 1985; Keim, 2002). Woody debris also contributes to stream production, particularly in headwaters, through the provision of coarse, particulate and dissolved organic matter (Bilby and Likens, 1980; Gurnell *et al.,* 1995). Further investigation by project-specific monitoring may be better employed to detect effects of the removal of barriers from tributaries of the Wye (Hart, 2002).

Juvenile trout populations in the Edw were the only species-age group to demonstrate any positive response to riparian management and no effect of the removal of barriers to migration was detected. Although lagged responses to riparian management could yet occur, this does raise the question of whether the method and location of management were appropriate.

The Clywedog and Edw were the rivers most extensively managed and the two rivers investigated in this chapter. However, prior to management (1985-1997) their populations of salmon and trout were either slightly increasing or slightly declining (Appendix III). Only juvenile trout in the Edw were classed as strongly declining, and it was only on the Edw that populations suggested some evidence of recovery. This suggests that i) management was not targeted at the worse affected tributaries and ii) perhaps it would have been more effective if applied to rivers where populations demonstrated most dramatic declines.

The worse affected rivers were not targeted since a better chance of recovery was anticipated on tributaries that were traditionally 'good' salmon rivers, such as the Clywedog and Edw (Slater 1988; Luxton, 2002; The Wye and Usk Foundation, 2006b). After 2003, beyond the scope of this study, management was undertaken on the Llynfi Dulas, Duhonw, Triffrwd and Hafrena, all of which demonstrated strongly declining salmonid populations prior to 1997 (Appendix III). In view of post-treatment changes in trout densities in the Edw, management might be more successful here. By the same reasoning riparian management might be appropriate for juvenile populations on the Llynfi and Garth Dulas (Appendix III).

Ecologically sensitive or 'soft engineering' of rivers is still an emerging and largely untested science. In rural areas, exclusion of livestock by fencing is the most established technique used to promote riparian vegetation and stabilise banks. However, buffers that store sediment can also contribute to the river load during high rainfall events (Hook, 2003; Davies *et al.*, 2006). This is not a criticism of fencing *per se* but of land management within the wider catchment that may increase the sediment load in runoff. The buffer alone may not be sufficient to counterbalance soil erosion within the wider catchment.

In pursuit of an improved riparian habitat, advice on the treatment of riparian trees often appears to be conflicting. Planting is recommended, as is coppicing, thinning and pruning (Broadmeadow, 2004). Many authors advocate a balanced approach that is appropriate to river type (O'Grady, 1993; Hendry *et al.* 2003; Summers *et al.*, 2005). Tree roots bind the soil, adding tensile strength to the banks (Easson and Yarbrough, 2002). Conversely, trees with large canopies may be more susceptible to falling during storms, taking large sections of bank with them. During the 18th and 19th centuries at the height of the clog-making industry, alders were coppiced along the banks of the Wye (Slater, 1988). Coppicing retains the root system of the tree whilst modifying the canopy, initially allowing more light to the banks and stream (Rackham, 1986). Prior to the arrival of clog-making in Britain in the 16th century, many of the upland valleys of the Wye remained densely wooded (Slater, 1988). Natural cycles of tree fall would have resulted in a mosaic of light and shady patches along the river. In the absence of sediment sources from intensively farmed land as today, sediment introduced by falling trees was unlikely to have over-loaded spawning gravels. In our managed landscape, rotational coppice can maintain a patchy, diverse habitat and promote understorey growth and stabilise stream banks (Rackham, 1986). For these reasons, the Wye Habitat Improvement Programme adopted coppicing as a restoration technique to stabilise banks but also to promote in-stream cover and production (Luxton, 2002; The Wye and Usk Foundation, 2006a, b).

The use of coppice in the UK as a salmonid habitat improvement technique has been largely confined to lowland streams (e.g. Giles and Summers, 1996) . The river continuum concept describes the structure and function of communities along a river system (Vannote *et al.,* 1980). The concept characterises upland, headwater streams (orders 1-3) as being 'strongly influenced by riparian vegetation which reduces autotrophic production by shading and contributes large amounts of allochthonous detritus'. The ecology of upland streams is therefore adapted to depend on terrestrial inputs and typically invertebrate communities comprise a high proportion of shredders rather than grazers (Vannote *et al.,* 1980). This description differs markedly from that of lowland chalk rivers in the UK on which coppicing and fencing have been successfully employed to boost trout populations (Giles and Summers, 1996; Summers *et al.,* 2005). Summers *et al.* (2005) described one such low gradient English chalk stream as having banks and margins vegetated with tough grasses such as reed canary-grass (*Phalaris arundinacea* L.) and emergent plants like watercress (*Rorippa nasturtiumaquaticum* L.), with 'fool's water-cress' (*Apium nodiflorum* L.) along the margins, and submerged macrophytes, typically *Ranunculus* (sp.) in the mid-channel. In fact, species of algae found within the Wye catchment, such as *Lemanea* and *Rhodophyta* (red macro-algae), are adapted to shaded conditions (Hynes 1970; Thrib and Benson-Evans 1983, 1984; Luxton, 2002). Perhaps opening up the canopy was not appropriate to the upland streams of the Wye?

The increased light afforded by initial coppicing may be short-lived. In 2004, middle reaches of the Clywedog appeared largely 'tunnelled' by 5 year-old re-growth of willow (see Chapter 4). Similar observations were made by Luxton (2002) just one year post-coppice who also observed 'no tunnelling' of bank vegetation prior to treatment. It is possible that neither light interception nor allochthonous leaf-litter input were reduced by coppicing in the long term. However, the thinning and coppicing of approximately 50% of bank-side trees within the Wye catchment probably diversified the age structure of riparian trees, which is generally considered beneficial to salmonid populations, notably trout, as well as general biodiversity (Lewis and Williams 1994; Luxton, 2002; Broadmeadow, 2004).

Functional relationships between habitat variables and fish production remain poorly understood and so responses to riparian alteration are difficult to predict and interpret (Armstrong *et al.,* 2003). Cover is likely to be a key attribute in promoting salmonid abundance and is broadly defined as anything beneath which a fish could be hidden from above such as overhanging and in-stream vegetation, undercut banks, woody debris, rocks, deep or turbulent water (Heggenes, 1988; Milner *et al.,*, 2003). Over-head shade from tree canopies can provide additional cover, allochthonous energy sources and protection from extremes of temperature (see review by Armstrong *et al.,* 2003). Conversely, juvenile trout (>0+) surveyed by Eklov *et al.* (1999) in southern Sweden during late summer / early autumn demonstrated a negative relationship with temperature which implies that shading reduces trout production in cool, temperate regions (Armstrong *et al.,* 2003).

The decline observed in juvenile trout, in the absence of any decline in fry densities suggests that the two life-stages responded to different regulating factors. If habitat were the key factor driving fry densities it seems that it was sufficient to sustain a trout fry population at least. For juvenile trout, and salmon fry and juveniles, either habitat or food were in decline and restricted the river's capacity to hold salmonids or other environmental factors impacted adversely on populations (Steingrimsson and Grant, 1999; Milner *et al.*,, 2003).

Although salmon and trout often inhabit similar high velocity, well aerated rivers, the two species and their life-stages are commonly segregated either through niche separation or inter-year class and inter-specific competition (Bremset and Heggenes, 2000; Armstrong *et al.,* 2003; Milner *et al.*,, 2003). Salmon were consistently more abundant than trout across the catchment as a whole but perhaps their decline favoured trout fry? Trout tend to out-compete salmon except in areas of particularly fast flows (Armstrong, 2003). Climate or land-use changes could have modified river environments to the extent that trout fry had an advantage over salmon fry where salmon were in decline.

Population declines were evident in both migratory and non-migratory salmonid stocks suggesting that factors operating on river systems were instrumental in population declines. Habitat is only one of a vast array of interacting factors that impact on salmonid populations (see reviews by Armstrong, 2003; Milner *et al.*,, 2003 and Ormerod, 2003). Habitat alteration may influence territorial competition and possibly food availability thus regulating population abundance through density-dependent feedback (Elliott, 1994; Milner *et al.,* 2003). Meanwhile, density-independent factors, such as climate, operate over much larger, regional or global scales and can mask the density-dependent effects of habitat intervention (Elliott, 1995; Milner *et al.*, 2003). The effects of landscape and climatic factors on trends in salmonid populations in the Wye are investigated in Chapter 7.

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APPENDIX I

Equations used to calibrate semi-quantitative surveys of salmonid density (actual number of fish caught per $100m^2$ of river) against Quantitative survey data (population estimate of fish numbers per $100m^2$ of river) after Strange *et al.* (1989).

Salmon Fry

Quantitative density = 13.5 + 2.00 Semi-quantitative density
(R-Sq = 87.4%)

Juvenile Salmon

Quantitative density = 2.13 + 1.41 Semi-quantitative density
(R-Sq = 82.5%)

Trout Fry

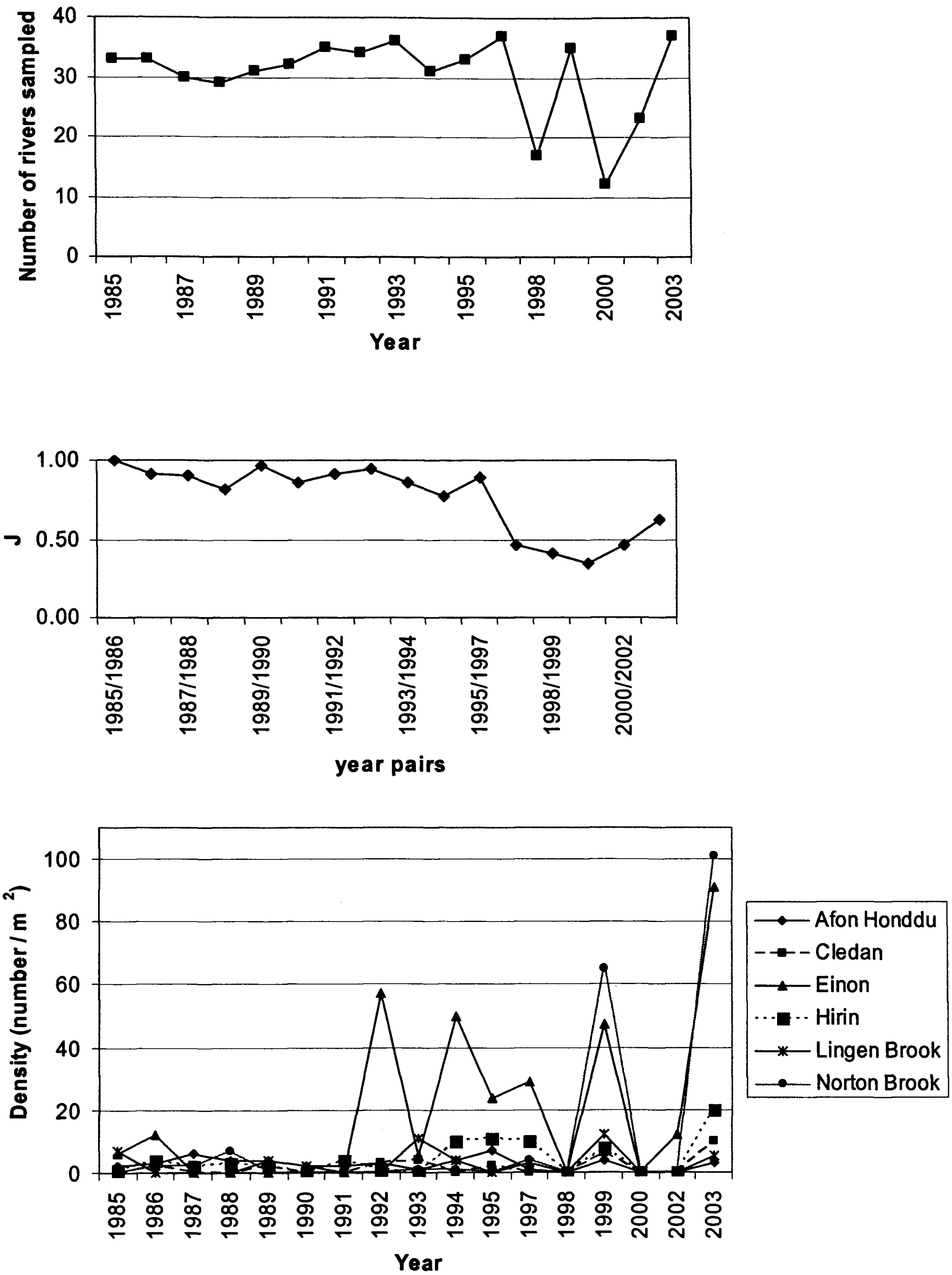
Log_{10} Quantitative density = 0.0808 + 1.16 Log_{10} Semi-quantitative density
(R-Sq = 90.9%)

Juvenile Trout

Quantitative density = 0.703 + 1.35 Semi-quantitative density
(R-Sq = 88.6%)

APPENDIX II

Sensitivity of analyses to identify whether the sampling frequency of rivers within the Wye catchment 1985-2004 influenced temporal trends observed in fish density.



a) Numbers of rivers sampled each year, **b)** similarity in frequency of sampling of rivers in successive pairs of years (Jacquard's coefficient of similarity (J) as described in (Chapman, 1992). **c)** Salmonid densities against year for rivers for which survey frequency was correlated with salmonid densities in the Wye.

APPENDIX III

Groups of tributaries of the Wye that demonstrated similar trends in salmonid density over time during the period prior to riparian restoration (1985-1997). Streams on which riparian habitat was manipulated are highlighted in bold, those with limited riparian alteration are in parenthesis, and those from which obstructions were removed from a point downstream from the survey site are in italic.

Groups of tributaries that demonstrated similar trends in salmonid density over time during the period prior to riparian restoration (1985-1997).

	Salmon fry	**Juvenile salmon**	**Trout fry**	**Juvenile trout**
Group 1	Arrow_ds	*Chwerfri*	Bleddfa	*Aran*
r -1 > -0.5	Cledan	Cnyffiad	Dernol	Arrow_ds
	Cnyffiad	*Duhonw*	*Nantmel Dulas*	*Chwerfri*
	Duhonw	Einon	Hafrena	**Edw**
	Irfon	Garth dulas		Garth dulas
	Llynfi dulas	Llynfi		Hafrena
	(Sgithwen)	Llynfi dulas		Hindwell
	Triffrwyd	*(Sgithwen)*		Llynfi
		Triffrwyd		Llynfi dulas
				Nant Gwynfel
				Norton brook
				Pinsley brook
				(Sgithwen)
				South dulas - Irfon
				Triffrwyd
Group 2	Arrow_us	Arrow_ds	*Aran*	Arrow_us
r -0.5 > 0	Bidno	Arrow_us	Arrow_us	Bleddfa
	Chwerfri	Cledan	*Chwerfri*	Cledan
	Clywedog	**Clywedog**	Cnyffiad	Cnyffiad
	Dernol	Dernol	**Edw**	Dernol
	Dore	Dore	Garth dulas	Dore
	Nantmel Dulas	*Nantmel Dulas*	Gwesyn	*Duhonw*
	Einon	**Edw**	Hindwell	*Nantmel Dulas*
	Frome	Hindwell	Honddu	Gwesyn
	Ithon	Irfon	Llynfi	Honddu
	Lingen brook	Llanwrthwl brook	Nant Gwynfel	Lingen brook
	Llanwrthwl brook	Monnow	Pinsley brook	Lugg_us
	Llynfi	Pinsley brook	*(Sgithwen)*	
	Monnow	South dulas - Irfon	South dulas - Irfon	
	Norton brook		Triffrwyd	
	South dulas - Irfon			
Group 3	*Aran*	*Aran*	Arrow_ds	Bidno
r 0 > 0.5	*Cammarch*	Bidno	Bidno	*Cammarch*
	Edw	Gwesyn	*Cammarch*	**Clywedog**
	Garth dulas	Honddu	Cledan	Einon
	Gwesyn	Ithon	**Clywedog**	Irfon
	Hindwell	Lugg_ds	Dore	Ithon
	Honddu	Lugg_us	*Duhonw*	Llanwrthwl brook
	Lugg_ds	(Marteg)	Einon	Lugg_ds
	Lugg_us	Trothy	Ithon	(Marteg)
	(Marteg)		Lingen brook	Monnow
	Nant Gwynfel		Llanwrthwl brook	Trothy
	Pinsley brook		Llynfi dulas	
			Lodon	
			Lugg_ds	
			(Marteg)	
			Norton brook	
			Olchon	
			Trothy	
Group 4	Hafrena	*Cammarch*	Frome	Frome
r 0.5 > 1		Frome	Irfon	Lodon
		Nant Gwynfel	Lugg_us	Olchon
			Monnow	

APPENDIX IV

Groups of tributaries of the Wye that demonstrated similar trends in salmonid density over time during the period prior to the removal of barriers to migration (1985-1995). Streams from which barriers were removed are underlined, those on which riparian habitat was manipulated are highlighted in bold and tributaries with only limited (<33%) riparian treatment are in italic.

Groups of tributaries that demonstrated similar trends in salmonid density over time during the period prior to the removal of potential barriers to salmonid migration (1985-1995).

	Salmon fry	**Juvenile salmon**	**Trout fry**	**Juvenile trout**
Group 1 r -1 > -0.5	Llynfi dulas Cnyffiad Irfon Triffrwyd Dernol Cledan	*Duhonw* Triffrwyd Einon *Chwerfri* Arrow_us Llynfi	Llynfi dulas *Sgithwen* Bleddfa Nantmel Dulas Dernol Llynfi South Dulas - Irfon	Arrow_ds Aran Norton brook **Edw** Garth Dulas Pinsley brook *Chwerfri* South Dulas - Irfon Llynfi *Sgithwen* Nantmel Dulas Triffrwyd
Group 2 r -0.5 > 0	Arrow_us Monnow Norton brook *Duhonw* Arrow_ds Lingen brook Llynfi *Chwerfri* Dore Bidno Hindwell Llanwrthwl brook South Dulas - Irfon Ithon Frome **Edw** Honddu Nantmel Dulas	Cledan Garth Dulas Monnow Hindwell Cnyffiad **Clywedog** Irfon Dore Pinsley brook Llanwrthwl brook Nantmel Dulas *Sgithwen* Dernol Honddu **Edw** Trothy South Dulas - Irfon	Cnyffiad Hindwell Garth Dulas Triffrwyd Aran Bidno **Edw** Arrow_us Norton brook Pinsley brook Nant Gwynfel Gwesyn	*Duhonw* Hindwell Bleddfa Nant Gwynfel Cnyffiad Arrow_us Dernol Lingen brook Lugg_us Gwesyn **Clywedog** Honddu Dore Lugg_ds
Group 3 r 0 > 0.5	*Sgithwen* Lugg_us Einon Pinsley brook **Clywedog** *Marteg* Garth Dulas Cammarch	Ithon Bidno Lugg_us Lugg_ds Aran *Marteg* Arrow_ds	Cammarch Lingen brook Ithon *Marteg* Honddu **Clywedog** Frome Dore *Chwerfri* *Duhonw* Llanwrthwl brook Lugg_ds Trothy Einon Arrow_ds Lodon Cledan Irfon Olchon	Ithon Cledan Llanwrthwl brook Bidno Cammarch Monnow Irfon *Marteg* Einon
Group 4 r 0.5 > 1	Lugg_ds Gwesyn Aran Nant Gwynfel	Nant Gwynfel Cammarch Llynfi dulas	Monnow Lugg_us	Trothy Olchon Lodon Frome Llynfi dulas

APPENDIX V

The density (Mean ± SE) of a) salmon fry, b) juvenile salmon, c) trout fry and d) juvenile trout before and after the removal of barriers in 1996 from the Aran, Chwefri, Sgithwen, Nantmel Dulas and Cammarch. Differences between the relative densities of salmonids (standardized by river) in treated and control streams (Control minus Treatment) were tested for by two-sample *t*-tests, *t*-values are shown in bold. Degrees of freedom were adjusted for unequal variances when appropriate. P values prior to 'Bonferroni' adjustment for multiple tests are displayed.

a) Salmon fry

		Pre-Treatment	Post-Treatment	df	*t*	P
Aran	Treated	64.2 (7.0)	31.3 (8.0)			
	Controls	15.5 (2.4)	16.6 (5.4)	13	1.642	0.125
Chwefri	Treated	69.6 (13.0)	42.7 (12.6)	14	0.256	0.802
	Controls	31.8 (3.5)	17.3 (1.8)			
Sgithwen	Treated	127.1 (24.7)	57.6 (10.0)	9	1.278	0.233
	Controls	16.1 (2.8)	8.2 (2.7)			
Nantmel Dulas	Treated	10.1 (0.7)	13.5 (0.0)	12	-3.529	**0.004**
	Controls	31.8 (3.5)	17.3 (1.8)			
Cammarch	Treated	116.2 (19.2)	70.7 (17.4)	12	-0.103	0.919
	Controls	16.1 (2.8)	8.2 (2.7)			

b) Juvenile salmon

		Pre-Treatment	Post-Treatment	df	*t*	P
Aran	Treated	4.1 (0.8)	3.5 (2.2)	14	-0.318	0.755
	Controls	4.1 (0.4)	3.1 (0.4)			
Chwefri	Treated	4.4 (0.9)	1.2 (0.4)	14	0.853	0.408
	Controls	16.8 (3.0)	5.0 (1.4)			
Sgithwen	Treated	20.7 (6.5)	7.0 (1.8)	10	0.665	0.521
	Controls	6.5 (0.5)	6.0 (1.4)			
Nantmel Dulas	Treated	4.4 (1.2)	2.3 (0.8)	12	0.087	0.932
	Controls	6.5 (0.5)	6.0 (1.4)			
Cammarch	Treated	12.2 (2.8)	4.0 (1.4)	7	0.412	0.693
	Controls	5.6 (1.9)	5.5 (1.3)			

a) Trout fry

		Pre-Treatment	Post-Treatment	df	*t*	P
Aran	Treated	3.7 (0.9)	2.8 (1.3)	14	0.789	0.443
	Controls	9.9 (2.0)	15.8 (6.2)			
Chwefri	Treated	17.4 (6.7)	15.6 (7.9)	14	0.539	0.598
	Controls	10.8 (2.7)	13.1 (3.0)			
Sgithwen	Treated	5.5 (2.1)	4.5 (1.9)	12	1.584	0.139
	Controls	19.7 (6.5)	23.2 (8.2)			
Nantmel Dulas	Treated	15.2 (5.7)	3.3 (0.9)	10	-0.050	0.961
	Controls	19.7 (6.5)	23.2 (8.2)			
Cammarch	Treated	3.6 (0.9)	3.5 (1.6)	12	0.329	0.748
	Controls	10.8 (2.7)	13.1 (3.0)			

b) Juvenile trout

		Pre-Treatment	Post-Treatment	df	*t*	P
Aran	Treated	8.8 (1.5)	2.7 (1.1)	14	1.812	0.091
	Controls	9.1 (1.6)	3.9 (0.5)			
Chwefri	Treated	9.5 (1.9)	5.2 (0.9)	14	0.746	0.468
	Controls	9.1 (1.6)	3.9 (0.5)			
Sgithwen	Treated	7.8 (2.4)	3.5 (0.9)	10	0.232	0.822
	Controls	9.1 (1.6)	3.9 (0.5)			
Nantmel Dulas	Treated	4.8 (1.4)	4.5 (1.3)	11	-1.194	0.320
	Controls	9.1 (1.6)	3.9 (0.5)			
Cammarch	Treated	5.9 (3.3)	1.7 (0.8)	12	0.513	0.617
	Controls	6.9 (0.8)	6.5 (0.2)			

Chapter 7

Recent trends in juvenile salmonid populations in the Wye catchment, Mid-Wales.

[7.0] Abstract

Populations of Atlantic salmon (*Salmo salar*) and trout (*Salmo trutta*) have declined throughout much of their range over recent decades. Factors such as pollution, density-dependent processes, climate, exploitation and adverse land management are all implicated, but there have been few assessments of longer-term data that help to interpret trends.

The River Wye was traditionally regarded as one of the best salmon angling rivers in England and Wales, but stocks of Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*) have declined over the last 20 years. Consequently, an extensive habitat improvement programme was launched in 1998. By 2004, trends in populations of salmon and trout had not been reversed raising the possibility that population trends might be driven by larger-scale factors – i.e. beyond the habitat scale.

This chapter compares possible regional and local explanations for recent trends among salmonids in the Wye by appraising whether 1) climatic, chemical and hydrological factors could explain long-term trends in juvenile populations and 2) local characteristics of rivers could explain localised trends.

Although some local factors discriminated among rivers that had contrasting trends in salmonid populations, declining trends for both species were evident over a range of environmental conditions. In the Wye catchment as a whole, climate was as a key correlate with population trends; salmonid abundance and population increases were positively correlated with summer rainfall and river discharge, while populations declined when summer temperatures increased.

If substantiated, the widespread decline of salmonid populations in the Wye in response to climate would imply that climatic effects may have masked or offset more local habitat effects, including local riparian management. Climate projections for the UK suggest that altered flow pattern and increasing summer temperatures might exacerbate losses further.

[7.1] Introduction

Populations of Atlantic salmon (*Salmo salar*) and trout (*Salmo trutta*) have declined in recent decades across their range (Slaney *et al.,* 1996; Parrish *et al.,* 1998; Beamish *et al.,* 1999; Environment Agency and CEFAS, 2004). However, the factors involved remain poorly understood despite the large literature on salmonid life histories and environmental preferences (Marschall and Crowder, 1996; Slaney *et al.,* 1996; de Groot, 2002; Armstrong *et al.,* 2003; Milner *et al.,* 2003; Ormerod, 2003). Particular difficulties arise in migratory salmonids because their life cycle is divided between marine and freshwaters environments so that population trends can result from supra-regional, catchment or local (river or reach) effects (Poff, 1997; Poff and Huryn, 1998). In addition, abundance reflects both density-independent processes, such as climate or pollution, and also density-dependent feedback mechanisms, such as territorial competition (Milner *et al.,* 2003). Population decline undoubtedly results from many factors in combination (Marschall and Crowder, 1996; Parrish *et al.,* 1998; de Groot, 2002; Milner *et al.,* 2003). Density-dependence, environmental and demographic stochasticity, climatic forcing and land management are all implicated in current trends (Wilzbach *et al.,* 1998; Parrish *et al.,* 1998; Bjornstad and Grenfell, 2001; Ormerod, 2003). Evaluating among these potentially competing explanations is challenging.

Over regional scales, direct exploitation and disease can alter salmonid populations (Bowker *et al.,* 1998; Ormerod, 2002; Bruno, 2004; Almodovar and Nicola, 2004; Jokikokko and Jutila, 2005; Costello, 2006; Hari *et al.,* 2006; Quinn *et al.,* 2006). Commercial fishing of Atlantic salmon (*Salmo salar*) mostly occurs in near-shore areas including Ireland and the UK and over-fishing has previously been implicated in decline (Mills, 1989; Parrish *et al.,* 1998; Common Fisheries Policy, Council Regulation 2371/2002/EC; Potter, *et al.,* 2003). The widespread reduction of salmon has also prompted some authors to cite regional climatic factors, such as rising sea surface temperatures, as key factors in population change (Noakes *et al.,* 2000; Beamish *et al.,* 1999; Beaugrand and Reid, 2003; Tolimieri and Levin, 2004). In addition to directional climate change, the North Atlantic Oscillation (NAO) has been linked to changes in fish production in both marine and freshwater ecosystems in the Northern hemisphere (Bradley and Ormerod, 2001; Parsons and Lear, 2001).

At catchment scales, barriers to migration, habitat fragmentation, diffuse pollutants, agricultural intensification and industrialization can have adverse effects on stream habitat and water quality and hence salmonid populations (Cazemier, 1994; Naiman and Turner, 2000; McCarty, 2001; Kauffman *et al.,* 2004). Salmonids require well aerated gravels for spawning and are sensitive to organic enrichment and acidification (Herrmann *et al.*, 1993; Kondolf and Wolman, 1995; Armstrong *et al.,* 2003). In the UK, point-sources of organic pollution, such as sewage, have been largely controlled (Ormerod, 2003; Dudgeon *et al.*, 2006). Legislation, such as the Freshwater Fish Directive in Europe (78/659/EEC; soon to be consolidated within the Water Framework Directive, 2000/60/EC) specifies standards water quality for salmonids e.g. dissolved oxygen (>9mg/l), total ammonium (1.0 mg/l) and pH (6-9) as well as guidelines for suspended solids (25 mg/l) nitrites (0.01mg/l) and total ammonium (0.04 mg/l) (78/659/EEC; Annex 1). In rural catchments, particular attention has been given to diffuse pollutants such as acid deposition, eutrophication and sediments (Parrish *et al.*, 1998; Ormerod, 2003; Merz, *et al.*, 2006; Suttle *et al.*, 2004). For example, increased exports of sediment to water courses may decrease the growth and survival of juveniles by altering the availability of prey and increasing foraging effort in addition to the well-documented clogging of spawning gravels causing suffocation of eggs within redds (Suttle *et al.*, 2004).

At the reach scale, many authors have identified habitat as a major influence on salmonid abundance (e.g. Borsuk, *et al.,* 2006). Consequently, management to restore salmon populations is often directed towards creating optimal habitats for salmonids at the river- or riparian-scale (e.g. Kondolf, 1993; Moerke and Lamberti, 2003; Opperman and Merenlender, 2004; see Chapter 2). In reality, however, the effects of such local activities on populations could easily be subsumed or offset by larger-scale processes. Understanding how best to direct such local efforts in the face of larger-scale impacts is thus a key question.

Although long-term data sets are increasingly available through which possible explanations for salmonid declines might be assessed, evaluations are still few (Thomas, 1996; Bjornstad and Grenfell, 2001; Daufresne *et al.,* 2004; Hulme, 2005; Crozier and Zabel, 2006). The River Wye provides a particularly important case study because riparian and in-stream management to benefit salmonids have been attempted. The Wye was traditionally one of the best salmon angling rivers in the UK

(Figure 7.1; Slater, 1988; Gough *et al.,* 1992). However, in recent decades, stocks of Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*) have both declined (See Chapter 6; Gee and Milner, 1980; Gough *et al.,* 1992; Environment Agency, 2003). Catchment-wide habitat management has yet to reverse trends in populations of salmon and trout, possibly because of the limited timescale so far involved (see Chapter 6). However, an alternative explanation is that larger-scale pressures have greater effects on the Wye's salmonids than local management, and this possibility requires assessment.

This chapter aimed to identify any large-scale environmental correlates with salmonid density which could have masked, offset or subsumed any local habitat effects. Specific objectives were to i) describe long-term variations in water chemistry and climatic variables in the Wye catchment, ii) establish whether local river characteristics can discriminate between rivers with different trends in salmonid densities and iii) identify any correlation between climatic, chemical and hydrological factors and juvenile salmonid populations in the Wye catchment. Greater effects of ii) rather than iii) would imply that local management might be able to offset population decline, while the reverse would support the importance of larger effects.

[7.2] Study Area

The Wye catchment is primarily located in mid-Wales before the river crosses the English boarder to join the Severn Estuary at Chepstow (Figure 1). The Upper Wye is typical of upland, high-velocity rivers in the UK. To the west of the catchment, where exotic conifers were extensively planted, streams are also acidified (Ormerod *et al.,* 1989; Edwards *et al.,* 1990). Land-use in the upper and mid-Wye is dominated by pastoral agriculture, while arable farming predominates in the fertile Lower Wye valley (Edwards and Brooker, 1982). The general ecology, land use and water quality of the Wye catchment are described in Chapter 3 and elsewhere (Edwards and Brooker, 1982; Ormerod and Edwards, 1987; Ormerod, 1988; Edwards *et al.,* 1990; Environment Agency, 1998; Brennan *et al.,* 2003).

The temperate climate of mid-Wales is generally wetter than the average for England and Wales for which precipitation averaged 912mm per annum between 1941 and 1970 (Hughes and Morley, 2000). Average annual precipitation across the Wye catchment ranges from of approximately 700 to 2500mm per annum, with higher values in the upper and western catchment (recorded at Cefn Brwyn) (Jarvie *et al.,* 2003; NERC, 2005).

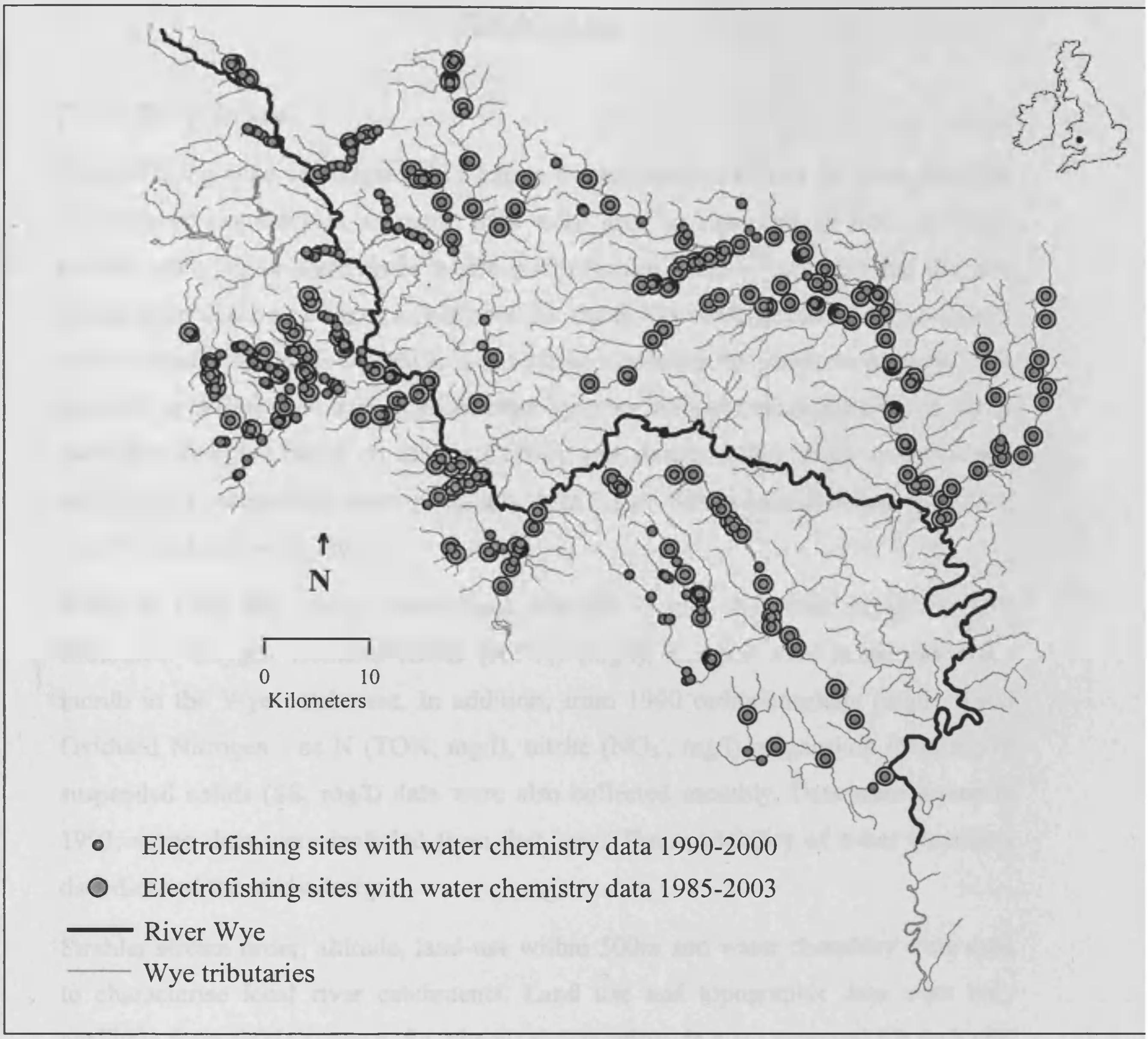


Figure 7.1. Location of electrofishing sites within the River Wye catchment, Wales for which chemical data were also available:
i) orthophosphate, total oxidised nitrogen, nitrite, magnesium and suspended solids, ammoniacal nitrogen, dissolved oxygen, BOD and pH between 1990-2000 (used to describe local river character)
ii) ammoniacal nitrogen, dissolved oxygen, BOD and pH between 1985-2003 (used to identify temporal trend and correlates with salmonid density).

[7.3] Methods

[7.3.1] Study Design

Electrofishing sites investigated in Chapter 6 were used as a basis for these analyses of salmonid populations. Climatic data were used to represent all sites and site-specific corrections were made where possible (see section 7.3.2.3). Land use and topographic data were also available for the entire Wye catchment. Water chemistry and electrofishing data sets, whilst both obtained from the Environment Agency, were sampled at different locations. Electrofishing sites were therefore matched to water chemistry samples based on their proximity and stream order. Some electrofishing sites had no comparable water chemistry data (i.e no data within the same river) and were excluded from analysis.

Between 1980 and 2003, ammoniacal nitrogen (mg/l), dissolved oxygen (mg/l), Biological Oxygen Demand (BOD [ATU]) (mg/l), and pH were monitored every month in the Wye catchment. In addition, from 1990 orthophosphate (mg/l), Total Oxidised Nitrogen - as N (TON, mg/l), nitrite (NO_2^-, mg/l) magnesium (Mg, mg/l), suspended solids (SS, mg/l) data were also collected monthly. Data were scarce in 1993, so no data were included from that year. The availability of water chemistry data dictated the study design.

Strahler stream order, altitude, land-use within 500m and water chemistry were used to characterise local river catchments. Land use and topographic data were only available from single surveys. So, chemical determinands were averaged for each site. To maximise the array of chemical determinands used to describe sites, seasonal means for chemical determinands were calculated from data recorded between 1990 and 2000. This enabled the inclusion of orthophosphate, TON, nitrite, magnesium and suspended solids. The power of river characteristics to discriminate between local trends in salmonid densities was established from 182 sites (reaches) within the Wye catchment (Figure 7.1).

In order to investigate temporal trends in salmonid densities and environmental variation a longer data set was preferable. Salmonid data were available from 1985 onwards. Chemical variables available for 1985-2003 (i.e. ammoniacal nitrogen, dissolved oxygen, BOD and pH) were therefore used in analyses to identify any correlation between climatic, chemical and hydrological factors and juvenile salmonid populations in 27 rivers of the Wye catchment (Figure 7.1).

[7.3.2] Data collection and treatment

[7.3.2.1] Salmonids

Electrofishing data recorded for the Wye system from 1985 to 2003 were provided by the UK Environment Agency. Quantitative and semi-quantitative surveys of salmon (*Salmo salar*) and trout (*Salmo trutta*) juveniles and fry were undertaken during the summer months, most frequently in July. Quantitative surveys (29%) involved three electrofishing runs per each (approx. 50m) stretch of river to yield a population estimate as density of fish per m^2 of river. Semi-quantitative surveys (71%) involve just one run per site and data were converted to equivalent 'quantitative' population estimates according to calibration equations given in Chapter 6, according to Strange *et al.* (1989). Densities were log transformed to achieve normality. Missing data were treated as described in Chapter 6.

[7.3.2.2] Water chemistry

Chemical data were obtained from the Environment Agency's Water Management Information System (WMIS) for the river Wye catchment. Chemical analyses were undertaken by the Environment Agency and its predecessors using standard spectrophotometric methods (SCA, 1977, 1978, 1979, 1980 a, b, 1981 a, b, 1987 a, b, 1988, 1992; Environment Agency, 2001). Methods used to determine ammoniacal nitrogen, dissolved oxygen concentration, Biological Oxygen Demand, and pH are detailed in Chapter 3. Nitrite (mg/l) was measured as an azo dye at 520 nm and nitrate was reduced to nitrite to ascertain the concentration of total oxidised nitrogen (TON mg/l) (SCA, 1981b; Environment Agency, 2001). Magnesium was determined from the acidified sample treated with a lanthanum salt and aspirated into the flame of an atomic absorption spectrophotometer at 282.2 nm (SCA, 1977, 1987b). The concentration of suspended solids (SS mg/l) was obtained by filtration of the sample through a glass-fibre paper, then drying at 105°C and weighing the recovered matter (SCA, 1980).

Initially, the concentration of orthophosphate was determined from the reaction of orthophosphate with acidic molybdate reagents to form a reduced phospho-molybdenum blue complex which was analysed discretely at 724 nm and 660 nm.

Total phosphate was determined when all forms of phosphorus were converted to reactive phosphorus (orthophosphate) by mild oxidative digestion (SCA, 1980). From 1994, air segmented flow analyses were used for the determination of total- and ortho-phosphate. The method oxidised all forms of phosphorus to reactive phosphorus which was then measured colorimetrically at 760nm/880nm (SCA, 1992). To overcome any effects of methodological change, detection limits for determinands were set to half that at the beginning of the time series and outliers were excluded from analyses (as described in Chapter 3). Missing values were replaced, where possible, with the mean for the appropriate year, season and section of river. 'Summer' (April - September) and 'winter' (October - March) means were calculated for seasons antecedent to electrofishing.

[7.3.2.3] Climatic variables

Mean monthly minimum (tmin) and maximum (tmax) temperatures (°C) and rainfall (mm) were obtained for the meteorological station at Ross-on-Wye (Met Office, 2004). Temperature was adjusted for altitude according to an environmental lapse rate of 6.5 °C per 1000m (Ruddiman, 2001). Median winter (October -March) and summer (April - September) discharge (m^3/s) data from the gauging station at Plynlimon were supplied by Cascade Consulting. The North Atlantic Oscillation (NAO) index is the normalized pressure difference between the Azores and Iceland. The index is associated with changes in the surface westerlies across the North Atlantic to Europe and represents dry, cold winters when negative and warm wet winters when positive (Hurrell, 1995). Monthly mean NOA index values between were obtained from the Climatic Research Unit and used to calculate summer and winter means (Climatic Research Unit, 2005; Osborn, 2005).

[7.3.2.4] Land use, geographic and topographic data

Land-use classes from the 'Land Cover Map 2000' were used in analyses and included broad-leaved woodland, coniferous woodland, neutral grassland, calcareous grassland, acid grassland, arable and horticultural land, improved grassland and built up areas. Geographic and topographic data were obtained from the Centre for Ecology and Hydrology (CEH), the British Geological Society and the Land Cover Map 2000 survey and were treated as described in Chapter 3. All Geographical and topographical variables were extracted from spatial data layers using the ArcView® Geographical Information System (GIS) package (Version 9, ESRI Inc., California).

[7.3.3] Data analysis

Spearman's coefficients of correlation (r_s) between environmental variables and year indicated temporal trends in physicochemistry between 1985 and 2003. Temporal variation in environmental variables was characterised using separate Principal Components Analysis on mean 'climatic' and 'water quality' variables for summer and winter respectively. Scores of the first PC axes were used as measures of environmental variation (Table 7.2).

To establish trends in salmonid densities in rivers between 1985 and 2004, rivers were grouped according to the coefficient of correlation between salmonid density and year (as in Section 6.3.3.). Rivers in group 1 demonstrated a marked decline in salmonid populations (r_s 0.5 < 1.0), group 2 demonstrated a slight decrease in density (r_s 0.0 < 0.5), group 3 a slight increase (r_s -0.5 > 0.0) and populations in rivers in group 4 increased strongly (r_s -1.0 > -0.5). Catchment attributes and water chemistry were then used to discriminate between groups of tributary rivers that demonstrated similar trends in salmonid density. For this purpose, discriminant analysis was used to identify environmental factors that might account for variation in salmonid population trends between rivers (Tabachnick and Fidell, 2001). Chi-squared statistics, transformed from Wilks λ, established the significance of each discriminant function in predicting local trends in salmonid populations. Variables retained in the discriminant model were those that demonstrated significant discriminating power between rivers with differing trends in salmonid populations.

'Variable clustering' was adopted to reduce the number of variables used in discriminant analysis and multicolinearity of environmental variables (after Vaughan and Ormerod 2005; see also Chapter 4). Three observations per variable per group are suggested as the minimum number of variables used in discriminant analysis. In this study there were 182 sites and 4 discriminant groups (Williams and Titus, 1988). So, ideally, just 15 variables would have been investigated to achieve stability in discriminant function loadings. In this study, 27 environmental variables were available for analysis. Hierarchical agglomerative clustering was used to establish groups of variables that similarly described variation between river reach characteristics. Average, single, centred and centroid linkage methods were applied to euclidean distance and Pearson's similarity matrices in order to define variable clusters. Average-linkage (between groups) clustering of a euclidean squared distance matrix are presented here (Appendix I). Variable groups were those clustered at less than 10 standardised distance units. Principal Components Analysis (PCA) was then performed on each group and the scores of the first PC axis of variation used to represent a single compound variable in discriminant analysis. The model therefore analysed 16, rather than 27 environmental variables on 182 sites (reaches).

Potential effects of environmental variables on the density of juvenile salmonid populations between 1986 and 2003 in the Wye was established from a multi-level linear regression model (also called mixed effects models) using the SPSS (v12) 'MIXED' procedure. Multi-level regression models contain both fixed and random effects. Random coefficients are generated for each river to model the (covariance) structure of the data in order to establish the fixed effects of the variables of interest. A two-level design whereby observations were clustered within rivers in a 'random intercept' model accounted for the autocorrelated nature of the repeated observations on rivers (Twisk, 2006). The random intercept regression model was a better fit than a model that included a random slope and examination of the residual variance indicated that that further modelling of any additional variance was not required. The predicted change in density with actual climatic variables was estimated by examining raw variables values at the 5, 25, 75 and 95 percentiles of PCA scores (section 7.4.1) with the corresponding predicted density of salmonids. Spearman's correlations were also used to identify the degree of correlation between raw variables that contributed to PC axes that were significantly related to salmonid density in the regression model.

Analyses were performed in SPSS, version 11.5 (SPSS Inc., Chicago), except multilevel analyses which were performed in version 12 due to the unstable nature of mixed-models in earlier versions (SPSS for Windows, 2001; Tabachnick and Fidell, 2001). Data were transformed to reduce skew and kurtosis of the data when appropriate (SPSS for Windows, 2001, Tabachnick and Fidell, 2001). Bonferroni adjustment for multiple tests are reported where appropriate (ter Braak and Šmilauer, 1998; Quinn and Keough, 2002). Analyses were repeated for juvenile salmon (>0+) and juvenile trout (>0+).

[7.4] Results

[7.4.1] Long-term trends in environmental variables and salmonid density

Summer climate was well described in PCA by an axis that reflected high maximum temperatures and sunshine when positive but increased rainfall and discharge when negative (Table 7.2). In contrast, an axis describing winter climate largely reflected warmer temperatures with higher rainfall and discharge (Table 7.2). Axes of variation in water quality reflected trends from high ammonia and biochemical oxygen demand to high dissolved oxygen. pH increased along this axis, particularly in winter (Table 7.2).

Annual densities of salmonids in the Wye system either declined or remained stable in more than half of the years between 1986 and 2003 (Figure 7.2). During this period, biochemical oxygen demand and concentrations of ammonia generally declined. By contrast, ambient temperatures increased, particularly during the summer (Figure 7.3).

[7.4.2] Local river characteristics and trends in salmonid density

The majority of rivers (75% and 69% for salmon and trout respectively) in the Wye system demonstrated either 'strongly' or 'mildly' declining population trends in salmonids during the period 1985 to 2004 (Table 7.1). Although populations of both salmon and trout juveniles increased in less than 30% of rivers, river groupings based on population trend were not consistent across the two species (Table 7.1).

Variable clustering of river characteristics yielded 12 unique variables and 4 compound variables that could potentially discriminate among rivers with different population trends (Appendix I). The first principal components describing each compound variable explained over 75% of the variation in constituent variables (Appendix II). Respectively, they represented i) pH, magnesium and total oxidized nitrogen, ii) orthophosphate, nitrate and winter BOD, iii) altitude and acid grassland and iv) dissolved oxygen (Appendix II).

Different characteristics discriminated between rivers with increasing or declining population trends of salmon and trout (Table 7.3). {Phosphate-Nitrate-BOD}, ammonia, {pH- Mg-TON}, arable land use, stream order, summer suspended sediment and BOD best discriminated between rivers with varying trends in salmon populations (Table 7.3).

When trends in salmon were considered against river character, the first and second functions discriminating among rivers accounted for 66% and 27% of the variance in river character respectively (Figure 7.4; Table 7.4). The first discriminant function represented higher concentrations of suspended solids and large streams when negative and higher phosphate, nitrate and BOD when positive (Figure 7.4; Table 7.4). The second function described a gradient from high ammonia, BOD and proportion of arable land when negative through to high pH, magnesium, total oxidised nitrogen, suspended solids and larger streams when positive (Figure 7.4; Table 7.4).

River characters that discriminated among rivers with differing trends in trout populations differed from those that characterised differences observed in salmon trends. For trout, suspended solids concentrations in winter, calcareous grassland, stream order and {pH-Mg-TON} accounted for variation in population trend across rivers (Table 7.3). The first discriminant function accounted for 67% of variation in river characters, the second 26%. In this instance, the first discriminant function described higher suspended solids through to larger streams with higher concentrations of phosphate, magnesium and total oxidised nitrogen (pH- Mg-TON) (Figure 7.4; Table 7.4). The second axis indicated a higher concentration of suspended solids when negative (Figure 7.4; Table 7.4).

Despite the significance of discriminant functions and the correct classification of 42% and 56% of rivers for salmon and trout respectively, there was much overlap in environmental conditions among river groups at which populations were either declining or increasing (Figure 7.4; Table 7.4). For salmon populations, rivers with strongly declining trends in salmon (Group 1) were virtually ubiquitous (Figure 7.4a). Rivers with slightly declining salmon populations (Group 2) tended towards higher suspended sediment, pH and larger streams but their ranged overlapped with Groups 1 and 3 in discriminant space (Figure 7.4a). Rivers with slightly increasing populations of salmon (Group 3) occurred where streams were smaller and there was less suspended sediment (Figure 7.4a). The remaining 4 rivers in which salmon populations strongly increased (Group 4) generally had a higher nutrient status, less suspended sediment and were more alkaline than other streams (Figure 7.4a).

There was even greater overlap in the range of discriminant scores for rivers with decreasing and slightly increasing trends in trout populations (Groups 1-3) (Figure 7.4b). Rivers with strongly decreasing populations of trout (Group 4; n=4) tended towards higher concentrations of suspended sediment, higher acidity and were smaller than in other river groups (Figure 7.4b).

Together, the widespread tendency toward salmonid population decline and weakness of local factors in explaining trends suggested that larger-scale factors could be more important.

[7.4.3] Climatic and water quality correlates with salmonid density

The multilevel regression model of environmental factors on juvenile density indicated that correlates with temporal trends in salmonid population densities were similar in trout and salmon (Table 7.5). Densities of juvenile salmonids declined when PC1 was higher, i.e. when antecedent summer climate was sunnier with higher maximum temperatures and reduced rainfall and discharge (Table 7.2; Table 7.5).

According to the model, a change across the inter-quartile range for PC scores representing summer climate would equate to a 0.3 °C increase in maximum temperature and a 10.1 mm decrease in monthly rainfall. This would yield a reduction of 1.7 salmon (per 100m of river) and 1.8 trout (per 100m of river) respectively (Table 7.6).

Spearman's correlations between raw climatic variables and densities of trout were weak but significant for maximum temperature (r_s -0.254, P=0.02), sun (r_s -0.186 P<0.001), rainfall (r_s 0.118, P=0.026), and discharge (r_s 0.124 P=0.020). For salmon correlations were significant for density with sun (r_s -0.168 P=0.002) and with maximum temperature (r_s -0.126 P=0.018).

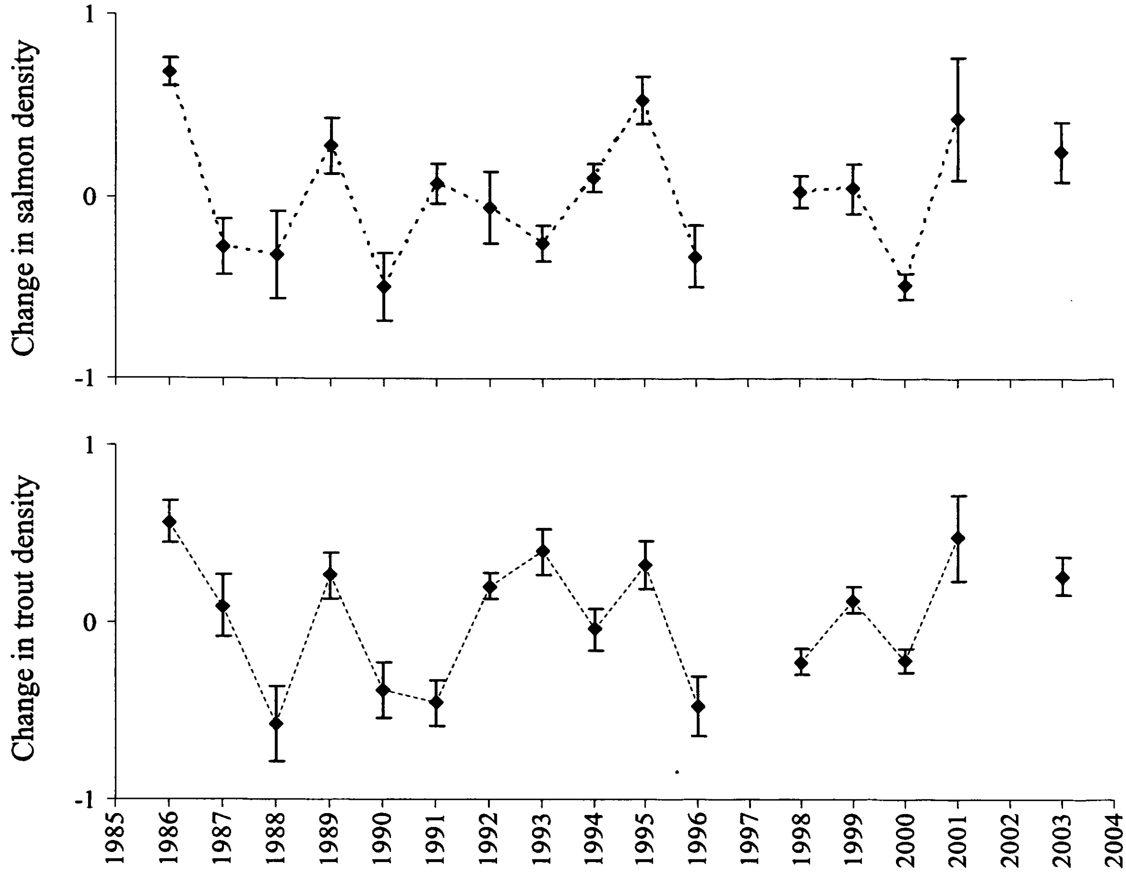


Figure 7.2. Annual change in mean (±SE) densities of salmonids (number of fish per $100m^2$ river stretch, standardized by river) between 1985 and 2003 in the Wye catchment.

Table 7.1. Groups of tributaries of the Wye that demonstrated similar temporal trends in the density of juvenile salmonids during the period 1985 to 2004. Salmonid trends were defined by Spearman's correlation between density and year (r_s). Rivers were grouped into 4 categories: Group 1 (strong decline in salmonid density), Group 2 (slight decrease in density), Group 3 (slight increase) and Group 4 (strong increase).

	Salmon	Trout
Group 1	Einon	Sgithwen
r_s -1 > -0.5	Sgithwen	South dulas - Irfon
'strong decline'	Lodon	Olchon
	Chwerfri	Hafrena
	Triffrwyd	Llanwrthwl brook
	Clywedog	Cammarch
	Olchon	Marteg
	Duhonw	Ithon
	Llanwrthwl brook	Llynfi
	Llynfi	Dernol
	Arrow_us	Pinsley brook
	Cnyffiad	Hindwell
	Monnow	Lugg_ds
	Irfon	
Group 2	South dulas - Irfon	Nant Gwynfel
r_s -0.5 > 0	Arrow_ds	Dore
'mild decline'	Cledan	Monnow
	Nantmel Dulas	Bidno
	Pinsley brook	Clettwr
	Ithon	Arrow_ds
	Hindwell	Clywedog
	Aran	Edw
	Lugg_ds	Honddu
	Dernol	Garth dulas
	Lugg_us	Lodon
	Cammarch	Lugg_us
	Garth dulas	Frome
	Edw	Cnyffiad
	Clettwr	Aran
	Trothy	Duhonw
		Bleddfa
Group 3	Marteg	Norton brook
r_s 0 > 0.5	Llynfi dulas	Arrow_us
'mild increase'	Dore	Chwerfri
	Honddu	Irfon
	Nant Gwynfel	Triffrwyd
	Bidno	Dulas brook
		Cledan
		Lingen brook
Group 4	Dulas brook	Nantmel Dulas
r_s 0.5 > 1	Frome	Llynfi dulas
'strong increase'	Hafrena	Einon
	Gwesyn	Gwesyn

The suffix 'ds' or 'us' denotes the downstream or upstream section of larger tributaries respectively.

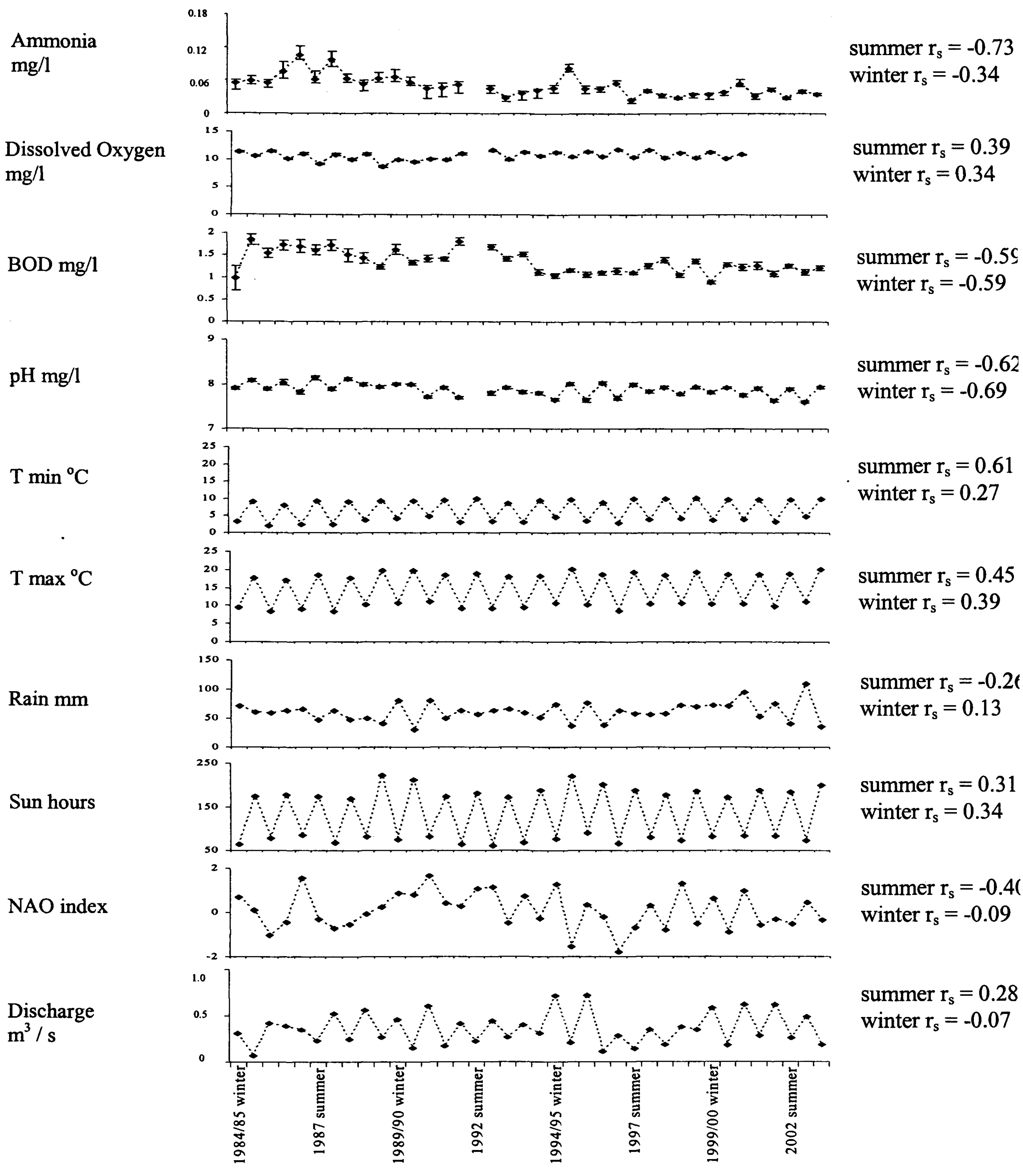


Figure 7.3. Temporal variation in (seasonal mean ± SE) climatic and water chemistry variables and (median) stream discharge relevant to the Wye catchment between winter 1984/5 and summer 2003. Spearman's coefficients of correlation (r_s) between environmental variables and year represent temporal trends in physicochemistry between 1985 and 2003, in summer and winter respectively.

Table 7.2. Principal Component (PC) axes representing variation in 'climatic variables' and 'water quality' relevant to the Wye system recorded between 1985 and 2000 in summer (April - September) and winter (October - March), respectively. Eigenvectors indicate the loading of variables on the first PC axis extracted in Principal Component Analyses.

Summer 'climate' Component

48.9 % variance explained

Sun	0.907
Discharge	-0.856
Rain	-0.851
Tmin	0.081
Tmax	0.755
NAO	0.278

Winter 'climate' Component

45.0 % variance explained

Sun	0.278
Discharge	0.702
Rain	0.619
Tmin	0.857
Tmax	0.885
NAO	0.479

Summer 'water quality' Component

40.3 % variance explained

AN	0.722
BOD	0.762
DO	-0.639
pH	0.317

Winter 'water quality' Component

47.3 % variance explained

AN	0.739
BOD	0.606
DO	-0.696
pH	0.704

Table 7.3. Equality tests of group means for different site characters for juvenile salmon and trout. A small Wilks λ and high χ^2 indicate greater discriminating ability.

Site character	Wilks λ	x^2	P
Salmon			
{Phosphate-Nitrate-BOD}	0.667	29.32	<0.0001
Ammonia (winter)	0.753	19.22	<0.0001
{pH- Mg-TON}	0.775	17.07	<0.0001
Ammonia (summer)	0.819	12.98	<0.0001
Arable and horticultural	0.879	8.11	<0.0001
Strahler stream order	0.889	7.30	0.0001
Suspended sediment (summer)	0.917	5.32	0.0016
BOD (summer)	0.920	5.11	0.0021†
Trout			
Suspended sediment (winter)	0.824	12.28	<0.0001
Calcareous grassland	0.910	5.70	0.0010
Strahler stream order	0.949	3.10	0.0281†
{pH- Mg-TON}	0.950	2.99	0.0324†

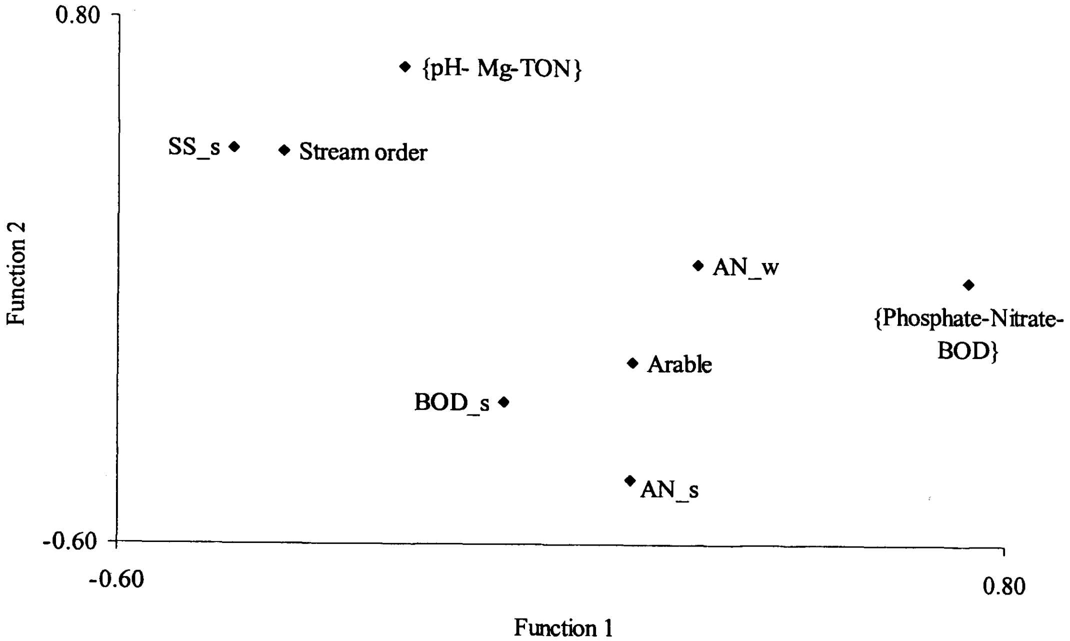
† Significant at 90% CI after Bonferroni correction for multiple comparisons.

Figure 7.4. Variables that discriminated between local trends in populations of juvenile a) salmon b) trout within the Wye catchment.

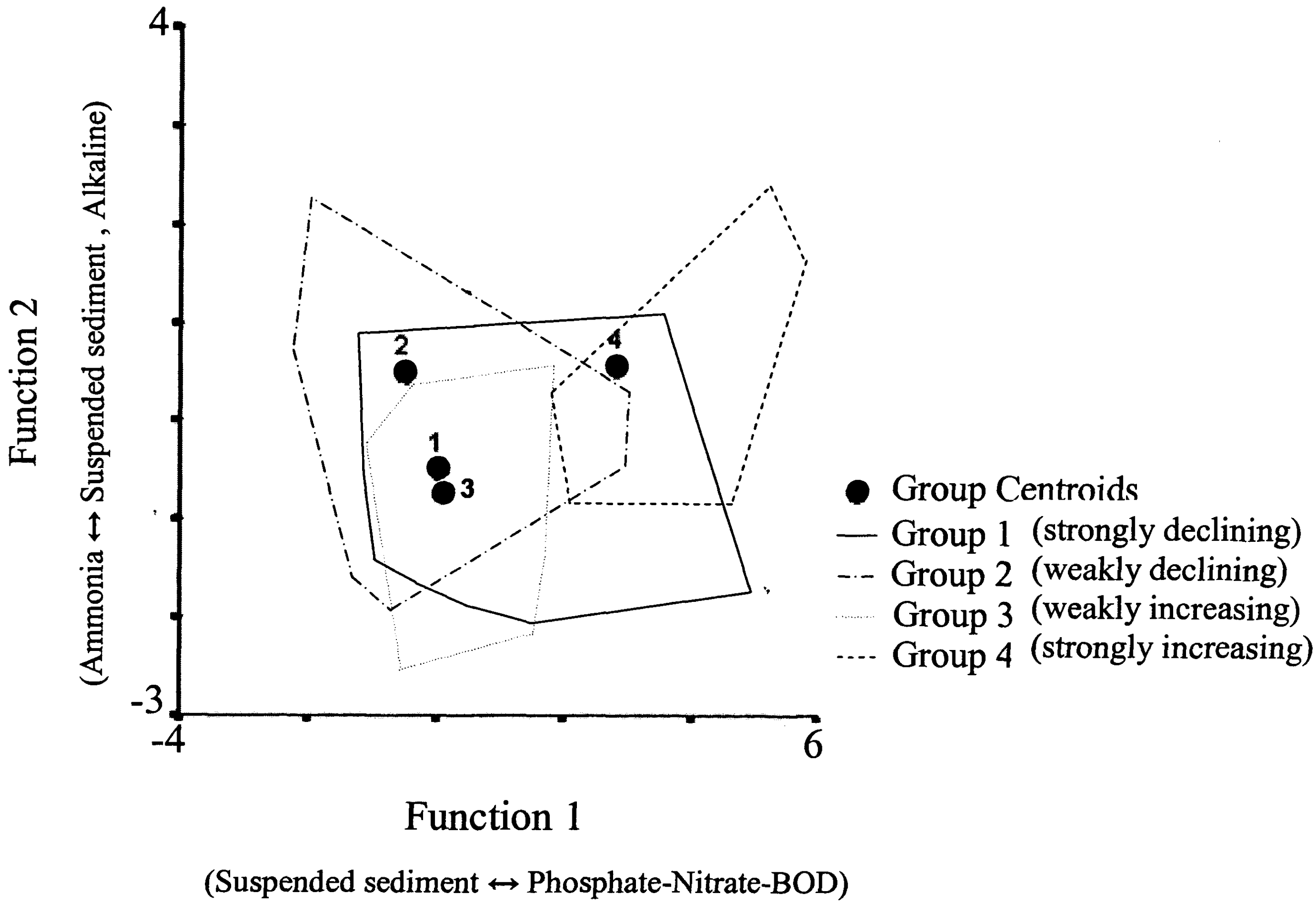
i) Standardized canonical discriminant function coefficients of environmental variables. '_s' and '_w' indicate chemical data recorded in 'summer' and 'winter' respectively.

ii) The range of site scores and group centroids of standardized canonical discriminant function coefficients for discriminant functions 1 and 2. Groups represent rivers with strongly increasing salmonid populations (4), mildly increasing populations (3), mildly decreasing populations (2) and strongly declining populations (1).

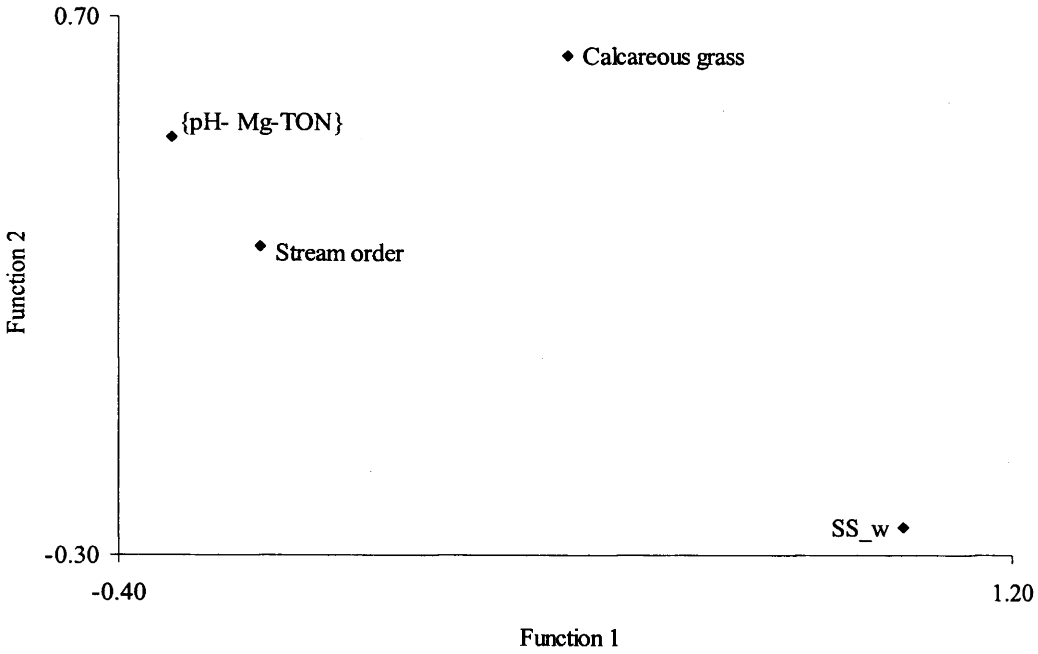
a) i)


0.80
-0.60
{pH- Mg-TON}
SS_s
Stream order
AN_w
{Phosphate-Nitrate-BOD}
Arable
BOD_s
AN_s
-0.60
0.80
Function 2
Function 1

ii)


4
-3
-4
6
Function 2
(Ammonia ↔ Suspended sediment , Alkaline)
Function 1
(Suspended sediment ↔ Phosphate-Nitrate-BOD)
1
2
3
4
Group Centroids
Group 1 (strongly declining)
Group 2 (weakly declining)
Group 3 (weakly increasing)
Group 4 (strongly increasing)

b) i)


0.70
-0.30
-0.40
1.20
Function 2
Function 1
Calcareous grass
{pH- Mg-TON}
Stream order
SS_w

ii)

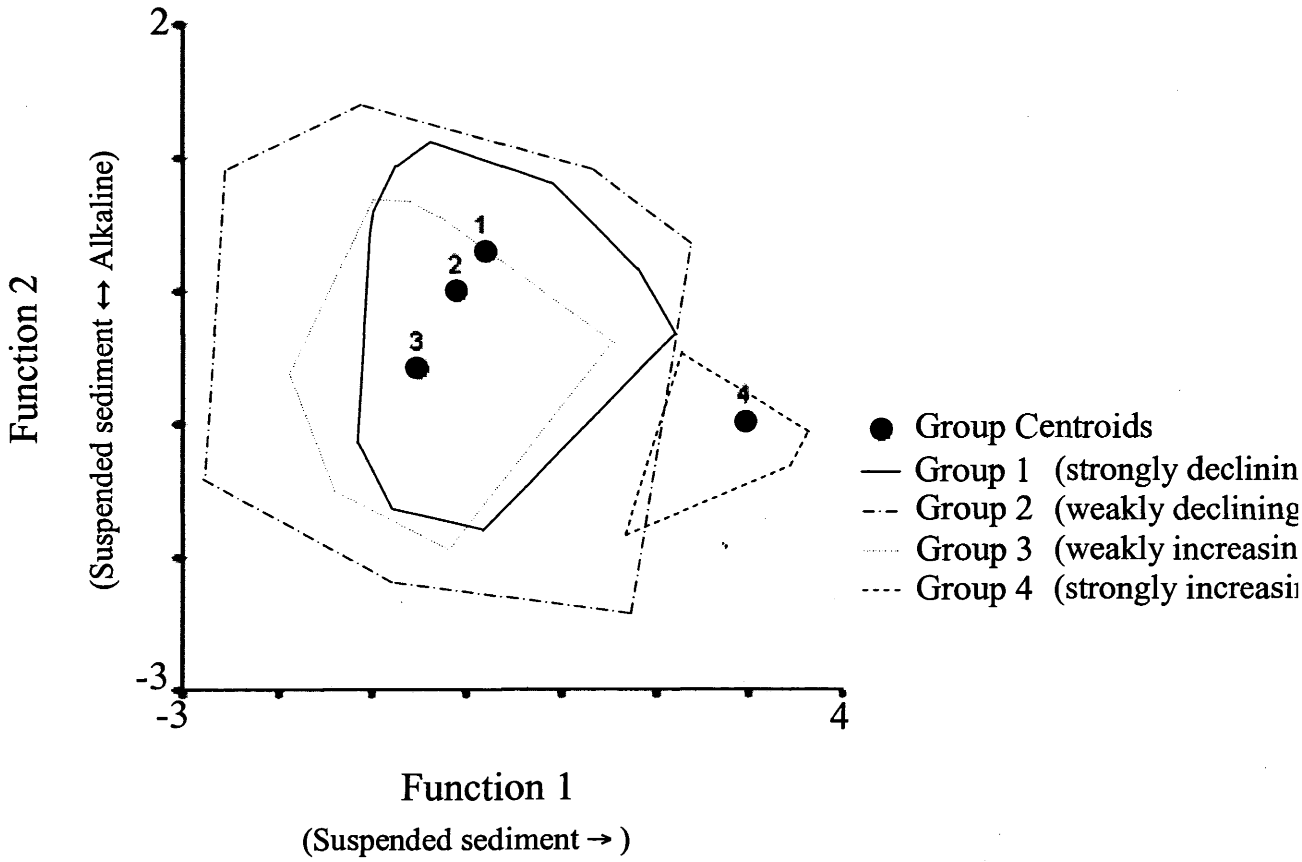

2
-3
-3
4
1
2
3
4
Function 2
(Suspended sediment ↔ Alkaline)
Function 1
(Suspended sediment →)
Group Centroids
Group 1 (strongly declinin
Group 2 (weakly declining
Group 3 (weakly increasin
Group 4 (strongly increasi

Table 7.4. Standardized canonical discriminant function coefficients of environmental variables that discriminated between trends in salmonid populations in tributaries of the Wye. A small Wilks λ and high χ^2 indicate a greater ability to discriminate between streams of varying salmonid status.

Salmon			
Function	**1**	**2**	**3**
% of variance explained	66.3	27.1	6.6
Arable and horticultural	0.206	-0.113	-0.406
Strahler stream order	-0.343	0.443	0.296
{pH- Mg-TON}	-0.157	0.670	-0.154
{Phosphate-Nitrate-BOD}	0.742	0.100	-0.310
Ammonia (summer)	0.202	-0.425	0.552
Ammonia (winter)	0.309	0.148	0.350
BOD (summer)	0.002	-0.220	0.322
Suspended sediment (summer)	-0.421	0.452	0.561
Wilks λ	0.411	0.719	0.924
$\mathbf{x^2}$	153.8	57.0	13.8
P	**<0.001**	**<0.001**	**0.032**†
Original Model % of cases correctly classified		56.1	
Cross-validated Model % of cases correctly classified		51.7	

Trout			
`Function	**1**	**2**	**3**
% of variance explained	67.1	25.5	7.4
Suspended sediment (winter)	1.001	-0.249	0.290
Calcareous grassland	0.397	0.631	-0.220
Strahler stream order	-0.151	0.276	1.010
{pH- Mg-TON}	-0.306	0.478	-0.561
Wilks λ	0.696	0.879	0.974
$\mathbf{x^2}$	62.0	22.0	4.4
P	**<0.001**	**0.001**	0.109
Original Model % of cases correctly classified		44.9	
Cross-validated Model % of cases correctly classified		42.0	

†No longer significant at 95% CI after Bonferroni correction for multiple comparisons

Table 7.5. Structure of the multilevel regression model of environmental factors on log transformed juvenile density in a) salmon and b) trout in the River Wye catchment detailing regression coefficients (Estimate ± SE), 95% confidence intervals (CI) and corresponding ρ-values.

a)

Variable	Estimate (SE)	df	t	P	95% CI
Intercept	0.67 (0.04)	25.48	17.14	0.000	0.59 - 0.75
Summer 'climate' last year	-0.03 (0.01)	213.11	-2.50	0.013	-0.06 - -0.01
Winter 'climate' last year	-0.01 (0.01)	214.02	-1.02	0.309	-0.04 - 0.01
Summer 'water quality' last year	-0.01 (0.02)	236.66	-0.36	0.720	-0.05 - 0.04
Winter 'water quality' last year	-0.02 (0.02)	236.21	-0.87	0.386	-0.07 - 0.03

b)

Variable	Estimate (SE)	df	t	P	95% CI
Intercept	0.69 (0.04)	26.81	17.78	0.000	0.61 - 0.77
Summer 'climate' last year	-0.05 (0.01)	214.96	-3.58	0.000	-0.08 - -0.02
Winter 'climate' last year	-0.01 (0.01)	216.05	-0.35	0.729	-0.03 - 0.02
Summer 'water quality' last year	-0.03 (0.02)	238.32	-1.48	0.141	-0.08 - 0.01
Winter 'water quality' last year	0.03 (0.03)	231.94	1.02	0.308	-0.02 - 0.08

Table 7.6. Values of raw climatic variables (monthly average) at 5, 25, 75 and 95 percentiles of Principal Component scores derived from a Principal Component Analysis of summer climatic variables relevant to the Wye river system (see Table 7.2 for loadings).

PC1summer percentiles	PC1summer scores	Rain fall (mm)	Sun (hours)	NAO	Discharge (m^3 / s)	Tmax °C	Tmin °C
5%	-1.469	62.4	173.3	-0.131	0.346	18.2	9.3
25%	-0.828	48.2	169.8	-0.588	0.264	18.6	10.1
75%	0.761	38.2	202.1	-0.222	0.130	18.9	9.0
95%	1.762	36.0	213.6	0.340	0.124	20.5	10.0

[7.5] Discussion

At local scales, some environmental factors discriminated among sites with increasing densities of salmonids from others with declining or more stable populations. One implication would be that local factors play some role in wider population trends. Thus, for example salmon appeared to have increased in more productive streams where suspended sediment was lower. These results are consistent with the well-documented negative effects of sediment on salmonids documented (e.g. Kondolf and Wolman, 1995; Platts *et al.*, 1989; Suttle *et al.*, 2004). However, trout populations apparently increased under slightly more acid conditions where suspended solids were higher, conditions usually considered adverse to salmonids (Parrish *et al.,* 1998; Armstrong *et al.,* 2003; Poleo *et al.*, 1997). Sediment concentrations in the Wye were below (summer median = 7.5 mg/l, winter median = 11.0mg/l) guideline concentrations acceptable for salmonid fish (average suspended solids $\leq$25 mg/l) given in the Freshwater Fish Directive (78/659/EEC; Lloyd, 1992). Concentrations of suspended solids in the Wye were therefore within the tolerance range of both species. Salmon and trout are adapted to different ecological niches and there is also evidence that species interact and compete (Armstrong *et al,* 2003; Milner *et al.,* 2003). Chapter 6 suggests that trout and salmon fry might respond differently to changing environmental conditions; unlike salmon fry, trout fry density did not demonstrate declines over time. Streams where trout populations tended to increase were also characterised here by lower stream orders. Trout populations could be doing better in smaller streams despite relatively higher sediment and acidity, possibly mediated by niche separation and competitive release from salmon. However, few streams had increasing salmonids numbers (n=4) and the dominant trends were of decline. The risk of spurious factors affecting such small numbers if sites is clearly large and the more important result in this chapter therefore involves the widespread and long-term reduction.

Despite some apparently local patterns that distinguished among rivers with different trends in salmonid populations, declining trends for both brown trout (*Salmo trutta*) and Atlantic salmon (*S. salar*) were evident over a wide range of sites and environmental conditions. This suggests that declining trends might be explained by some catchment-wide effect (Rieman *et al.*, 2001; Regetz, 2003). Possible effects of climate and water quality were therefore investigated. Climatic variables, rather than water quality appeared to describe long-term trends in populations of juvenile salmonids. Salmonid abundance and population increases were positively correlated with summer rainfall and river discharge, while populations declined with increasing summer temperatures.

At this point is perhaps prudent to outline some of the limitations of this study. In attempting to identify likely environmental correlates with salmonid density linear species-environment responses were assumed. However, species do not necessarily respond linearly to environmental gradients (Armstrong *et al.,* 2003; Austin *et al.*, 1990). For example, growth rates are temperature-dependent and reach an asymptotic optima of 15.9 and 13.1 °C in salmon and trout respectively implying a Gaussian response of salmonids to temperature (Elliott and Hurley, 1995, 1997). The model presented here rather oversimplifies species-environment interaction but can provide a guide as to the potential responses of salmonid to variation in climatic and water quality if it is assumed that they exist within their optimal range, which is likely given that Wales is well within the native range of *S. salar* and *S. trutta* (Mills, 1989; Elliott, 1994; Maitland, 2004). Modelling non-linear responses of organisms using more advanced mixed models may improve the fit of models presented here (Twisk, 2006).

Models were also simplistic in the sense that they took environmental conditions in the year preceding fish sampling to explain salmonid response. Salmonid life cycles operate over multiple years and are complex, involving migration to accommodate the different habitat preferences of each life-stage. For example, specific requirements for spawning, such as minimum spawning depths of approximately the same depth as the fish (often averaging between 25 and 50cm), and egg development in redds (oxygen concentrations ≤ 7.0 mg l^{-1}, temperature 12.5 °C, water velocity ≤ 100 cm h^{-1} for survival circa 100%) are important to salmonid survival (see Armstrong *et al.*, 2003; Crisp, 1996). The effects of environment on salmonids are therefore likely to be integrated across the entire lifespan of the fish as well as that of the previous generation. Gee *et al.* (1978) observed that in most cases low values for salmon production could be attributed to poor spawning. Climatic effects operating on the fry of one cohort may not be evident in juvenile populations until the next generation, some 4 or more years later. The life cycle of salmon in the Wye system requires at least 4 years from egg to adult spawner, more in the case of multiple-sea-winter fish which made up approximately 90% of rod catches in 1977 (Edwards and Brooker, 1982). However, the 'critical period' in the dynamics of salmonid populations is reported as that between emergence and the establishment of feeding territories (Armstrong *et al.*, 2003; Nislow *et al.*, 2004). Climatic variables presented here operated over that critical period in the development of the juvenile salmonids in the

Wye. Correlations between climatic variables and salmon densities were generally greater for the preceding year when considered against the each of the preceding 7 years respectively.

Environmental factors other than those climatic and water quality variables examined are also likely to have influenced fish populations. These may include anthropogenic influences such as exploitation and the effects of other pollutants such as pyrethroids (Muir *et al.,* 1994; Moore and Waring, 2001). Factors driving population change also operate at multiple spatial and temporal scales. Infrequent, 'transient' events, such as episodic acidification, can shape populations (Bulger *et al.,* 2000; Rice, 2001; Hastings, 2004; Rodriguez-Arias and Rodo, 2004). Also, the importance of hydraulic, climatic or habitat variables may vary depending on season or time of day (Heggenes *et al.,* 1999). More complex models may be able to account for some of these factors.

Two important anthropogenic factors that can influence the population size and demographics of salmonids were not investigated in this study; exploitation at sea through commercial fishing and angling in freshwaters. Both have increased over the last century making fishing and climate effects difficult to dissociate (Gee *et al.*, 1978; Marschall and Crowder, 1996; Finney *et al.*, 2000). Exploitation effects on fish populations are most frequently reported as a change in age-class structure, with artificial selection against larger fish (Gee and Milner, 1980; Brana *et al.*, 1992; Quinn *et al.,* 2006). In the Wye, numbers of multiple-sea-winter (MSW) and spring salmon stocks have declined (Environment Agency, 2003b). In response, byelaws were introduced from the 1990's to reduce the catch of MSW salmon and the Wye and Usk Foundation have a voluntary catch and release scheme in order to increase spawning escapement (Mawle and Black, 2003).

In the Usk, a catchment adjacent to the Wye, a reduction in estuarine drift netting in from 1992 was linked to an increase in rod catches implying that commercial fishing had an adverse effect on Welsh stocks of salmon (Bowker *et al.*, 1998). Similar effects have been noted elsewhere (Jutila, 2005; Potter *et al.*, 2003). Sediment records from Alaskan lakes have been used to identify trends in Pacific salmonids over 300 years using stable nitrogen isotopes (Finney *et al.*, 2000; Finney *et al.*, 2002). Trends in salmon that were probably driven by climatic forcing were identified prior to and after the onset of commercial fishing. Authors suggested that both commercial fishing and climatic variation can influence salmon.

In the Wye, Atlantic salmon (*S. salar*) were migratory but brown trout (*S. trutta*) were mostly non-migratory (Edwards and Brooker, 1982; Slater, 1988). Correlates of both

S. salar and *S. trutta* with climate therefore imply that factors operating on the river environment were important in salmonid population declines.

Climate-related variables, such as rainfall and temperature, can impact directly on the river habitats preferred by salmonids therefore influencing their behaviour, growth and survival. In rivers, fry (0+) and juvenile (<0+) *S. salar* and *S. trutta* require well-oxygenated (5.0–5.5 mg l^{-1}), cool (optimal between 13 and 16 °C), relatively fast flowing water (approximately 10–30 cm s^{-1} and 20–50 cm s^{-1} respectively) (see review by Armstrong *et al.*, 2003). Both species inhabit deeper water as they grow, with post-young-of-the-year (PYOY, age >0+) salmon and trout typically inhabiting water depths of 25–60 cm and 20–70 cm respectively (Armstrong *et al.*, 2003).

The role of river discharge and water temperature in determining the composition and population dynamics of freshwater fish communities is well recognised. Classical river zonation describes the transition from upland to lowland river environments according to fish assemblage, the 'trout zone' being defined as narrow, shallow and fast flowing with clear waters (Hawkes, 1975; Aarts and Nienhuis, 2003). More recently, a study of coarse fish in the Upper Rhône River demonstrated that low flows and high temperatures favoured thermophilic fish species (e.g. *Leuciscus cephalus and Barbus barbus*) at the expense of northern, cold-water fish species (e.g. *Leuciscus leuciscus*), a trend which was also reflected in the macroinvertebrate fauna (Daufresne *et al.,* 2004).

As poikilotherms, the foraging behaviour and habitat use of salmonids is strongly influenced by temperature (Armstrong *et al.,* 2003). Elevated temperatures in temperate climates may extend the growing season for salmonids but possibly increase susceptibility to disease (Elliot *et al.,* 1998). For example, in alpine streams in Switzerland increased water temperatures induced higher incidences of proliferative kidney disease in brown trout at higher altitudes (Hari *et al.,* 2006). Changes in the intensity and frequency of rainfall affect river flow regimes and have implications for the accessibility and suitability of spawning grounds (Armstrong *et al.,* 2003; Hakala and Hartman, 2004). In addition, the confounded effects of temperature and discharge affect the concentration of pollutants and oxygen in river water (Whitehead *et al.,* 2006).

Considering the correlation of salmonid densities with climatic variables identified in the Wye against the growing background of evidence on environmental effects on salmonids, this study supports the assertion that climate is a key factor in regulating salmonid populations (Beamish *et al.*, 1999a; Beamish *et al.*, 1999b; Beaugrand and

Reid, 2003; Tolimieri and Levin, 2004). Potentially adverse effects of reduced rainfall and river discharge and high temperature on migratory (*S. salar*) and non-migratory (*S. trutta*) salmonid populations, suggest that climatic factors operating on the river environment are important in the regulation of both species, although this does not preclude any oceanic effects on anadromous populations (e.g. McFarlane *et al.*, 2000; Finney *et al.*, 2000).

In the Wye, summer rainfall and river discharge were positively correlated with increases in juvenile salmonids and elevated temperatures and low flows with declining populations. If substantiated, the widespread decline of salmonid populations in the Wye in response to climate would imply that climatic effects may have masked more local habitat effects (Poff, 1997; Poff and Huryn, 1998). This includes the effect of local riparian management.

The UK climate is likely to be between 2 °C and 5 °C warmer by 2080 (Hulme *et al.,* 2002). In Wales, winters are likely to become more wet and summers drier (Conway, 1998; Hulme *et al.,* 2002). Studies on the Upper Wye experimental catchment suggest that the occurrence and amount of precipitation will decrease in summer and autumn, while evapotranspiration will increase, resulting in more frequent 'low flows' (Pilling and Jones, 2002). Extreme drought events, as seen in the in 1970's (Edwards and Brooker, 1982) may become more frequent. Such changes might further exacerbate climate effects on salmonids, potentially altering access to spawning grounds, availability of nursery habitat (e.g. pools), and the concentration of pollutants in river water. The importance of river discharge to salmonids in the summer suggests that water conservation might be a priority for the management of fish stocks within the catchment in the future. In addition, riparian management might have an important role in providing shade to reduce potentially lethal summer temperatures (Rutherford *et al.*, 2004; Watanabe *et al.*, 2005).

Acknowledgements

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[7.6] References

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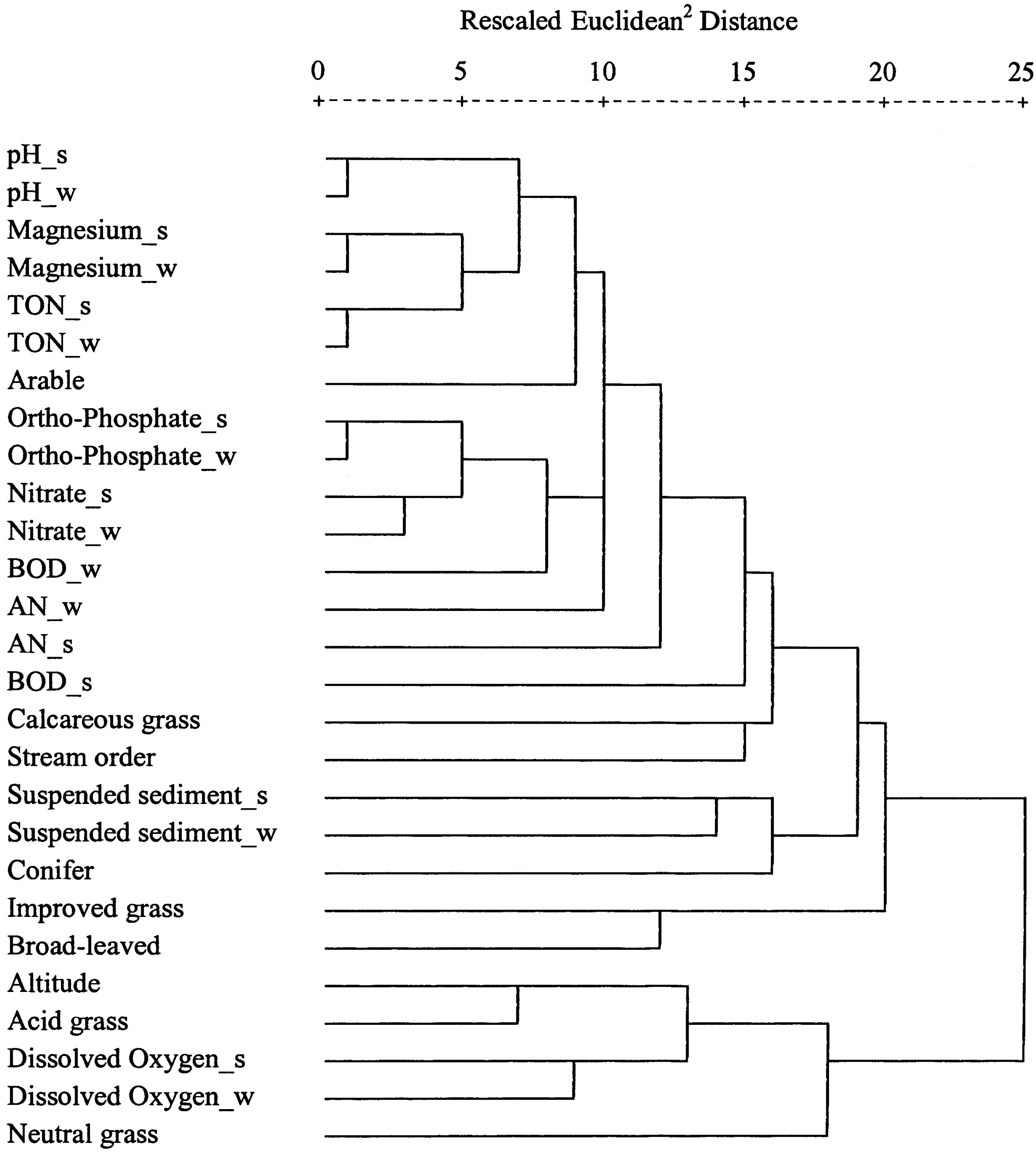
APPENDIX I

Hierarchical average-linkage clustering of environmental characteristics of river reaches of the Wye catchment. Variables with rescaled (squared euclidean) distance units of less than 10 were considered as 'variable groups' in further analyses (Appendix II).


Rescaled Euclidean² Distance
0
5
10
15
20
25
pH_s
pH_w
Magnesium_s
Magnesium_w
TON_s
TON_w
Arable
Ortho-Phosphate_s
Ortho-Phosphate_w
Nitrate_s
Nitrate_w
BOD_w
AN_w
AN_s
BOD_s
Calcareous grass
Stream order
Suspended sediment_s
Suspended sediment_w
Conifer
Improved grass
Broad-leaved
Altitude
Acid grass
Dissolved Oxygen_s
Dissolved Oxygen_w
Neutral grass

APPENDIX II

The first axes extracted by Principal Components Analysis on groups of environmental variables that similarly described variation between rivers of the Wye system as identified from cluster analysis, after Vaughan and Ormerod (2005) (Appendix I). Eigenvectors define the loading of river characteristics on component axes.

{pH- Mg-TON} Component

77.5 % variance explained

pH summer	0.859
pH winter	0.875
Magnesium summer	0.887
Magnesium winter	0.928
Total Oxidized Nitrogen summer	0.846
Total Oxidized Nitrogen winter	0.886

{Phosphate-Nitrate-BOD} Component

77.6 % variance explained

Ortho-Phosphate summer	0.924
Ortho-Phosphate winter	0.932
Nitrate summer	0.861
Nitrate winter	0.917
Biochemical Oxygen Demand winter	0.758

{Altitude-Acid grassland} Component

81.5 % variance explained

Altitude	0.903
Acid grassland	0.903

{DOs-DOw} Component

76.2 % variance explained

Dissolved Oxygen summer	0.873
Dissolved Oxygen summer	0.873

Chapter 8

General Discussion:

Recent trends among aquatic biota in the catchment of the River Wye (Wales) and the effects of riparian management.

[8.1] Introduction

The role of the riparian zone in river ecology has received increased attention over the last 20 years (Chapter 2). Consequently, riparian management programmes that aim to increase the quality and quantity of in-stream habitat for salmonids are currently being implemented in the UK and elsewhere (Krog and Hermansen 1985; Hemphill and Bramley 1989; Salmon Advisory Committee 1991; O'Grady 1993; Imhof *et al.,* 1996; Kauffman *et al.,* 1997; Hendry *et al.,* 2003; Pusey and Arthington 2003). However, experiments evaluating such management have been few, especially those investigating effects on other aspects of aquatic ecology (Liljaniemi *et al.*, 2002; Kiffney *et al.*, 2003; Wright *et al.*, 2003; Lepori *et al.*, 2005). Riparian management undertaken within the Wye system is an example of a catchment-wide programme intended to promote salmonid populations. The Wye catchment is an important conservation area, designated as a Site of Special Scientific Interest and Special Area of Conservation (CCW, 2006 a-c; JNCC, 2006). It was traditionally considered as one of the best salmon rivers in the England and Wales but declining rod catches over recent decades caused concern for the fishery (Edwards and Brooker, 1982; Environment Agency, 2003a). This prompted habitat management work to mitigate against the perceived effects of intensive pastoral land use within the catchment (Wye and Usk Foundation, 2006a, b, c). Routine monitoring data and standard techniques were used throughout this thesis to demonstrate trends in aquatic habitats and biota in space and time within the Wye catchment. This was used as a basis against which to evaluate the effects of land use and riparian management on habitats and salmonids. Because no suitable project-specific data were collected, routine monitoring data and river habitat surveys were used in the best *post-hoc* experimental designs applicable (e.g. 'Before-After-Control-Intervention'; ANOVA among treated and reference sites; multiple regression). Benthic invertebrates were used to indicate effects on aquatic communities and water quality. This brief concluding section discusses the main findings of the thesis, considers the limitations, reviews the relevance to wider issues of river and riparian management and considers further possible work.

[8.2] Main findings

Notwithstanding the challenges of study design, this thesis identified i) large spatial variations in ecological quality between tributaries of the Wye, ii) long-term and widespread trends over 20 years in biota and water chemistry, iii) some effects of management on habitats and invertebrates in expected directions, but no treatment effect on salmonids and iv) potential larger-scale effects, linked to climatic variation, on fish populations that might mask the effects of local habitat management.

Ecological quality varied widely among the Wye's tributaries reflecting different management pressures (Chapter 3). Combined biotic indices supported the need to mitigate acidification in some upland streams and reduce diffuse nutrients in the lower catchment (Chapter 3). Streams in intensive pasture had finer substrata and more active channels than elsewhere supporting the assertion that agricultural practice is linked to sedimentation of spawning gravels in the Wye (Chapter 4; Harding *et al.*, 1999; Armstrong *et al.*, 2003; Zaimes *et al.*, 2004).

Changes in the water quality and aquatic communities of the Wye over time were evident from routinely collected data. Between 1985 and 2003, biochemical oxygen demand and concentrations of ammonia generally declined (Chapter 7). By contrast, ambient temperatures increased, particularly during the summer (Chapter 7). Invertebrate taxon richness in the mid-Wye fell from 1995 to 2004, with families typical of lowland, eutrophic rivers favoured at the expense of upland taxa (Chapter 5). Salmonid fry and juvenile populations fell between 1985 and 2004 throughout the catchment (Chapter 6).

Riparian habitat management appeared to reduce bank poaching (trampling) and increase filamentous algae by comparison with reference streams (Chapter 4). Post-treatment (2000-2004) invertebrate communities were richer in recently managed streams than in controls due mostly to taxa typical of channel margins and lowland conditions (Chapter 5). However, there was no evidence that management reversed the decline in salmonid populations (Chapter 6).

The typical life-cycle of salmonids in the Wye, ranging from 2-7 years from egg to spawner including multi sea-winter fish, might delay response to management (Edwards and Brooker, 1982; Elliott, 1994). This effect cannot be evaluated with only six years' post-treatment data. Alternatively, local effects could be masked or offset by larger-scale trends. For example, changes in water quality (e.g. biochemical oxygen demand and ammonia) could not explain changes in fish populations (Chapter 7). However, salmonid abundance in the Wye declined significantly with increasing summer temperatures, decreasing summer rainfall and decreasing summer discharge (Chapter 7).

All of these results have implications for the management of land and water resources within the catchment, given its importance to fisheries and conservation (Edwards and Brooker, 1982; Environment Agency, 1998; Environment Agency, 2003b; JNCC, 2006). Declines in salmonids alongside reductions in invertebrate taxon richness over recent decades suggest that such measures should be designed to protect and, if possible, restore ecosystems of the Wye catchment.

[8.3] Limitations and caveats

The constraints on studies presented here in terms of their design were outlined from the outset (Chapter 1). The management programme in the Wye was already underway at the start of this investigation; treatments had been applied at different times, were not randomised across rivers, and no project-specific data on habitats or river organisms had been collected. Critically, no data on habitat structure, local salmonid densities or invertebrates had been collected prior to riparian management at most of the managed sites.

Routine monitoring and survey techniques provided standardised and therefore comparable data that were at least semi-quantitative. Many of the drawbacks of using such methods have been discussed throughout the scientific literature and are well-known (Resh and McElravy, 1993; Crozier and Kennedy, 1994; Raven *et al.*, 2002). For example, all the invertebrate data available involved family level identifications, thus missing potential important variations at more specific levels. Offsetting the cost of the level of identification of invertebrates against the time spent on identification has often been debated (Resh, 1994). In this study, over-simplistic generalisations may have resulted in Chapters 3 and 5. However, the detectability of variations among sites and treated/reference even at crude taxonomic levels implies that such effects may be substantial.

Freshwater ecologists, practitioners and government organisations will increasingly have to consider the challenges of appropriate identification and data availability under the Habitats Directive (Directive 92/43/EEC), Water Framework Directive (Directive 2000/60/EC) and the developing Environmental Liability Directive (Directive 2004/35/CE) pervade the management of the river environment. The UK is in a better position than most when it comes to archived data on its freshwater systems and at the forefront in the development of tools adapted to interpret them e.g. RIVPACS (ZamoraMunoz and AlbaTercedor, 1996; Wright *et al.*, 1998; Mustow, 2002). Studies of the type presented here demonstrate how such resources might be utilised in setting priorities for management (see Chapter 3). They also demonstrate some of the limitations of those data sets and highlight a need for project-specific monitoring. Whilst, long term datasets are useful in setting a background against which to identify management issues, given the broad nature of their remit they may not always be sufficient to identify the effects of management projects against specific aims (see Chapter 2; Imhof *et al.*, 1996; Minns *et al.*, 1996).

This thesis provided a broad over view of the biotic and abiotic character of the Wye, given the resources available. Each chapter therefore provides a background against which entire projects might be developed in their own right. For example, in Chapter 5, declines in taxon richness were identified in the mid-Wye but only examined here in the context of confounding the effects of management. Further investigation of these results would be invaluable in revealing detail on the trends and factors driving them in the Wye. For example, comparisons of data presented here against an earlier study of the invertebrate communities in the Wye undertaken to species level by Ormerod in 1982 (Ormerod and Edwards, 1987; Ormerod *et al.*, 1987) alongside a re-survey of Ormerod's sites could explore changes over 25 years. Such a study might also incorporate experiments to establish some of the ecological processes that might underlie the responses of communities to riparian alteration, such as by contrasting leaf processing in treated and control streams (Royer and Minshall, 2003; Danger and Robson, 2004).

Elsewhere in the thesis the organisms and environmental variables and anthropogenic factors investigated were, by necessity, limited. Biotic responses to management were exemplified here by benthic macroinvertebrates and salmonid fish which by no means encompasses the range of aquatic, terrestrial and amphibious species which have lifecycles and requirements intricately linked to riparian habitat (e.g. Bodie, 2001; Semlitsch and Bodie, 2003; Zoellick *et al.*, 2005). A number of studies on birds in particular have demonstrated that the management of riparian habitats have far-reaching implications for the ecological integrity of aquatic and terrestrial systems (Tyler and Ormerod, 1991; Naiman *et al.*, 1993; Kinley and Newhouse, 1997; Shirley and Smith, 2005;; Lussier *et al.*, 2006). Broader studies might include other species and perhaps investigate interactions between them.

The scale at which evaluation was undertaken here was designed to match the scale of management. The management programme attempted to reverse declining rends across the catchment through extensive riparian habitat alteration. So, replicate control and treated tributaries were compared in a catchment-scale investigation. However, this may have meant that reach-scale treatment effects were missed (see discussions in Chapter 4, 5 and 6) and that larger-scale factors operating beyond the catchment, particularly those concerning the migration costs on Atlantic salmon populations, were not fully investigated (Chapters 6 and 7). Chapter 7 attempts an examination of potential environmental drivers of change and demonstrates the challenge of evaluating among correlated environmental data and potentially competing explanations of salmonid decline using weak inference from correlation and regression statistics. It sets the scene for a more detailed investigation into salmonid population dynamics and causes of change (see Chapter 7 for a more detailed discussion).

[8.4] Management of freshwaters and the constraints of scale

The importance of large-scale factors, such as climate, on populations of salmonids might bring the value of local habitat management into question. Alternatively, these larger-scale pressures might be taken to imply that local management is even more crucial. Documented effects of intensive land management practices on aquatic systems and the importance of the quality and availability of habitat to salmonids suggest that without such undertakings population decline might be even more rapid (see Chapters 2, 5 and 6). One key need is to identify where population pressures on salmonids might reflect climate, and act accordingly at the scale required (Durance *et al.* 2006). At local scales, riparian trees have marked effects on thermal regimes and riparian management could provide methods for adapting to climate change effects by providing shade and preventing lethal summer temperatures (Rutherford *et al.*, 2004; Watanabe *et al.*, 2005; Durance & Ormerod *in press*).

Rivers are increasingly being managed from an ecological basis, by viewing species within the context of the wider biotic community and environment (Harper and Smith, 1995; Ward, 1998; Solomon, 2003). The management of riparian habitats on the Wye demonstrates a habitat focus for species protection. However, riparian management is intermediary between boosting populations of single species, for example fish-stocking, and broader conservation which protects the entire catchment (Wissmar and Beschta, 1998; Harper *et al.*, 1999; Verdonschot, 2000; McGinnity, 2002; Aprahamian *et al.*, 2003; Solomon *et al.*, 2003; Donohue *et al.*, 2006). A significant reversal of salmonid decline in response to management given the size of the restoration programme in relation to the size of the catchment might not be expected. For example, upstream effects on the lower reaches of tributaries, where most management was undertaken, may still affect habitat quality and impede population movement (Chapter 4; Dudgeon *et al.*, 2006; Evans *et al.*, 2006). Similarly, larger catchment-scale effects can offset the effects of local riparian management (Ormerod *et al.*, 1993). Prior to the WFD, such effects could not be so effectively managed, so that riparian measures offered one of the few practical means of influencing the river environment. The conservation of freshwaters at the catchment-scale, involving interconnected landscape units is widely accepted by freshwater scientists as necessary for the conservation of biodiversity and is now being incorporated into legislation (Ormerod, 1999; Logan and Furse, 2002; Ormerod, 2004; Dudgeon *et al.*, 2006).

Local management activities need to be appropriate to a given catchment, or even sub-catchment given the contrasting management strategies required in the Wye (Chapter 3). The applicability of habitat management undertaken in the Wye to its predominantly upland character given that much work of this type is undertaken on lowland systems was questioned in Chapter 6. Further, scientific investigations into the effects of riparian management and land use on upland systems in particular are warranted. Reporting and dissemination of results of studies such as this, alongside the evaluation of similar projects undertaken by River's Trusts, will help to broaden current understanding of how different systems respond to habitat work.

To date, the project undertaken by the Wye and Usk Foundation is one of the largest in the UK. It demonstrates the importance of habitat in the protection of a single-species for economic as well as environmental benefit to mid-Wales. Recently the term 'ecological reconciliation' has emerged from the scientific literature (Rosenzweig, 2003). It describes the compromise that is often made between restoring an ideal and the objectives of multiple stakeholders. In an ideal situation conservation, restoration, or environmental management would not be undertaken without an 'holistic understanding of natural river ecosystems' (Ward, 1998). Given the reality of 'anthropogenic landscapes' (Rosenzweig, 2003) management, based on the best available ecological knowledge, can benefit ecosystems and if scientifically designed can inform future management (Dudgeon *et al.*, 2006). The concept of ecological reconciliation underlines the importance of maintaining a dialogue between landowners, practitioners, scientists and the general public, such partnerships are exemplified by projects such as WHIP (The Wye Habitat Improvement Project) and PHISH (Powys Habitat Improvement Scheme) (Wye and Usk Foundation, 2006b, c).

[8.5] Conclusions

Riparian management in the Wye catchment had some of the desired outcomes at the reach or tributary scale, supporting the case for riparian management. However, reversal in salmonid declines is not yet evident. Limits on management effects imply a need for increased protection of river systems over wider spatial scales. Salmonid numbers in the Wye potentially reflect climatic effects of the type observed elsewhere (e.g. Byrne *et al.*, 2003; Byrne *et al.*, 2004; Crozier and Zabel, 2006) and this implies the need to consider climate-change in future management action at all scales.

This thesis demonstrates the importance and relevance of in-situ, replicated, catchment-scale experiments to understanding how organisms and habitats respond to environmental conditions, particularly in the event of a changing climate. The publication and dissemination of this work and similar projects will help to direct future management.

[8.6] Recommendations for future management

The riparian zone is an essential element in the conservation of freshwater organisms and ecological functions (Naiman *et al.*, 2005; Chapter 2). It supports aquatic species through the provision of allochthonous energy sources and by buffering against diffuse pollutants and provides habitat for terrestrial and amphibious species (such as invertebrates, birds, bats and otters) that contribute to regional biodiversity. The two-way energy exchange that occurs between streams and their surroundings is globally important by virtue of the sheer extent of stream networks, with over 20,000 km in Wales alone. However, informed management of this ecotone requires both better evidence of the effects of various management techniques and improved understanding of riparian processes.

In the Wye, reduced bank poaching along streams with stock exclusion suggests that riparian management work can alter bank habitats and may be a precursor to further development of bank vegetation. Although significant changes in fish populations have not yet been observed (Chapter 6), effects on invertebrates illustrate links between management action and ecological response (Chapter 7). The results of thesis therefore support the case for continued riparian fencing.

Elements of the current management programme that require further investigation are the opening up of the canopy and removal of large woody debris from streams. Temperate streams, like those in the upper Wye system, typically represent shaded conditions characterised ecologically by allochthony rather than authochthony (see discussion in Chapter 6). In fact, tree planting on river reaches where trees are absent represents a closer return to semi-natural conditions than selective thinning or coppicing. Aquatic macrophytes are unlikely to develop within the channels of upland streams – where production is dominated by bryophytes and algae - to the same extent as in lowland streams, for which coppicing is advocated (Giles and Summers, 1996). The importance of woody debris to fish production is increasingly recognised, and its removal may have negative effects on local fish production (Gurnell *et al.*, 1995; Keim, 2002).

One other element to be borne in mind with respect to stream management is that climate is increasingly changing the context in which management takes place and the priorities for action. Should reduced discharge and higher temperatures in summer have negative impacts on fish stocks (as suggested in Chapter 7) or exclude other key organisms adapted to cool-water regimes, shaded channels with well vegetated banks could offer important benefits. Shading may reduce lethal temperatures in summer (Rutherford *et al.*, 2004; Watanabe *et al.*, 2005).

In addition, the legislative and policy context for stream and river management is also changing. Not only the Water Framework Directive, but also policies encouraging Catchment Sensitive Farming are increasingly emphasising whole river basins in management planning (Directive 2000/60/EC; DEFRA, 2004). By contrast, much previous work by NGOs (e.g. Rivers Trusts) and statutory organisations (e.g. country agencies such as Countryside Council for Wales, Scottish National Heritage and Natural England) has focused on the riparian zone of tributaries because of the difficulties of work at larger scales. Upstream expansion of fences along streams, such as the Edw, may help to tackle diffuse sources of sediment. Perhaps more importantly, catchment sensitive farming practices, supported through broad-scale agri-environment schemes and the designation of sensitive areas (e.g. nitrate sensitive areas), would be beneficial. Better links between riparian and catchment-scale management offer an important way forward, but also will occur at a scale that means that evaluation will be challenging. A shared vision of river management objectives between National Governments, the Environment Agency, the Countryside Agencies, the Rivers' Trusts and farming agencies would clearly help to develop catchment-wide programmes of measures to protect and enhance aquatic ecosystems in accordance with the Water Framework Directive.

[8.7] References

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