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Professor Mike Depledge   Head of Science

EXECUTIVE SUMMARY

- The project is being carried out to develop a tool to evaluate the potential impact of flow and water level criteria on fish species and populations in English and Welsh rivers to allow more scientifically robust and hence defensible assessments to be made. The overall objective of the project is to provide, for various river reach types, generic seasonal flow and water level regime requirements for key life stages of freshwater fish species to advise and influence the management of flow regimes.

- The fish community types in rivers were modelled based on the Agency, fisheries data and complementary environmental data. The models discriminated eight major fish community types that broadly followed the classical zonation theory with river gradient from upland salmonid to lowland cyprinid communities. It is was concluded that the influence of flow and the potential impacts of abstractions and releases should be considered within the context of each of these main fish assemblages, linking key species per community type to their functional ecology and flow requirements. The relationship between the rate of flow, the rate of change of flow, the duration of high/low flow events and their seasonal timing, and their influence over the functioning of fish populations (spawning, recruitment and growth) needs to be considered when evaluating anthropogenic changes to flow patterns.

- Some biases in the dataset were identified, and it was considered important to remove these biases by filling the gaps in information, especially with respect to regions and river reaches poorly represented in the current dataset.

- The preferred habitat characteristics of the predominant fish species found in UK fresh waters was reviewed, but a paucity of data for the lesser species, especially those of conservation value, was found. It was recommended that information about habitat relationships of critical species that drive community structure was required, although, it may be better to develop the guild concept for discriminating the impact of flow regulation on fish and fisheries. Too few studies were found to have examined the wider environmental impacts of adjusting flows, especially the issues associated with maintaining longitudinal connectivity and facilitating passage of fish about obstructions. This is highly relevant to setting environmental flows that allow the free migration of fish during critical periods of their life cycle. It was recommended that these issues are examined and mechanisms for overcoming them are addressed. Similarly little information is available on the importance of relationships between residence times and access to side channels and backwaters from the main river channel for coarse fish species and no information on these characteristics is available for species of conservation value.

- A review of the approaches to assess the impact of modifying flows on fisheries indicated that there were two possible scenarios for development of a tool to meet the requirements for assessing the impact of water resources schemes on fisheries.

    1. A process to arrive at the fish requirements is pursued within the existing framework and objectives of CAMS/RAM: In this case a relatively simple

method of assigning the sensitivity index, such as one of the hydraulic rating, habitat simulation or HABSCORE methods should be developed.

2. If the intention is to manage the hydrological regime in the interest of fish and fisheries, three potential approaches were proposed. 1) The integration of the various biotic components of RAM to produce a single integrated statistic. 2) The development of a RIVPACS type model for fish, e.g. River Fish Environmental Flow Assessment Matrix (RIFEFAM). This model can either be based on species presence or absence, relative species abundance or incorporate those parameters of the fish such as biomass, condition, and growth and survival rates that are needed to manage the fishery. 3) The development of population dynamics models that will assist in predicting the effects on the quality and quantity of the fish population of various alternative hydrological regimes. It was recommended that all three models are examined in detail and the most appropriate for meeting the objectives of the Agency selected for assessing the impact of water resources schemes on fisheries.

- These aspects were developed into a project framework for consideration for future funding to meet the requirements for assessing the impact of water resources schemes on fish and fisheries.

CONTENTS

1 INTRODUCTION

1.1 Background

Flows in rivers in the UK are frequently compromised because of increasing demands for water abstraction. However, little is known about the impact of water abstraction on fish populations and communities, and there is little information upon which to make predictions about whether certain flow regimes will affect the sustainability of fisheries and the habitats upon which they depend. This is of concern because European Directives impose an obligation to protect habitats of threatened species (Habitats Directive) and ensure that all water bodies shall exhibit good ecological status or good ecological potential (Water Framework Directive) by the year 2015. A key requirement for achieving or maintaining good ecological status is the provision of adequate and appropriate flow regimes.

As a result, in England and Wales, the Environment Agency (EA) has the duty to balance the requirements of abstractors for water with the needs of the environment. This is achieved through the abstraction licensing system, which regulates water abstraction. However, the current method of handling abstraction does not address fully the various issues related to fisheries in a thorough and ecologically appropriate way, especially with respect to coarse fish and small-sized species. Coarse fish are an important component of riverine ecosystems in England and Wales that can potentially have high intrinsic value in terms of local and regional economies and in maintaining ecosystem services. Other species, such as lampreys, shads and spined loach, also have high conservation value both in UK and European terms. Unfortunately, there is often a failure to value these fisheries appropriately and rarely are fisheries afforded the protection they deserve. Thus, to achieve the balance between the environment and abstractors more effectively, an improved understanding of the flow and water-level requirements of fish is essential and clear guidelines on flow requirements to protect fish and fisheries are required.

In relation to flow regulation and abstraction there are a number of issues that need to be considered.

- Coarse fish, as well as salmonids, are migratory in their behaviour and move considerable distances up and downstream. They also make lateral movements into and out of floodplain and wetland areas. These movements are usually for spawning and feeding, and they need to be maintained. Abstraction potentially could affect longitudinal and lateral connectivity by restricting access, imposing barriers that disrupt normal life cycles and ultimately reduce stocks.

- Coarse fish are vulnerable to changes in water level caused by abstraction. Long- and short-term fluctuations in water level are known to be detrimental to coarse fish because they disrupt habitat usage, especially by juveniles. This age group uses the marginal areas for feeding and protection from predation, and abstraction often reduces the extent of the littoral margin, thus reducing available habitat. Water level fluctuations can also have deleterious effects on macrophytes, which provide essential feeding and nursery areas for fry.

- Coarse fish are highly vulnerable to entrainment at abstraction points because their swimming capabilities are weak, especially in the younger life stages.

- Little work has been done on the impacts of flows on coarse fish. Nunn, Cowx and Harvey (2003) derived a model that examined the role of discharge in predicting year class strength but this related to gross discharge over seasonal periods. For example, high flows in spring and early summer tend to lead to poor year class strengths, but this is of little relevance because abstractions taking large proportions of the flow at this time of the year are rarely required. By contrast, the influence of abstractions when the river is at or below Q95 in the late summer/autumn, which is also critical to young of the year, is poorly understood, but the need to protect habitat in the littoral marginal areas and maintain longitudinal connectivity, which could be affected by abstractions, is likely to be important.

- Little is known about the habitat preferences of freshwater fish, especially in relation to flow requirements. Copp (1990a, b, c, 1992a, b, 1997) and Garner (1996, 1997b) have examined some aspects of this, but these were studies in highly regulated rivers in Eastern England. The first phase of the NRA Coarse Fish project undertaken by HIFI attempted to draw habitat suitability curves (Cowx 2000, Cowx & Welcomme 1998), but these were based on limited data and somewhat subjective. Furthermore, PHABSIM and IFIM models are poorly developed for British coarse fish and not at all for the species of high conservation value (i.e. lampreys, shad, spined loach and bullheads).

This project is being carried out in order to develop a tool to evaluate the potential impact of flow and water level criteria on fish species and populations in English and Welsh rivers to allow more scientifically robust and hence defensible assessments to be made. The overall objective of the project is to provide, for various river reach types, generic seasonal flow and water level regime requirements for key life stages of freshwater fish species to advise and influence the management of flow regimes.

1.2 Specific Objectives

The overall problem is broken down into a suite of manageable components with the following specific objectives related to the above areas of research.

- To determine the basic habitat requirements of all life stages of fish species in rivers, and the seasonal changes in habitat requirements to maintain population status.

- To determine the critical flow related factors affecting the various life stages of fish species in rivers, and identify the causes and rates of mortality under different ecological and environmental scenarios.

- From the above, identify key seasonal habitat requirements in relation to flows and water levels and present them in a format for dissemination and consideration by water resource personnel.

- Identify key species which are sensitive to extreme flow conditions and could be considered representative in terms of flow requirements of particular species guilds.

- In recognition that species requirements may vary in different types and reaches of rivers, develop an appropriate and relevant river reach classification system for use in developing tools to asses the impact of modifying flow characteristics.

- Identify knowledge gaps and recommend future monitoring and research needs.

- Evaluate options for developing a tool, or suite of tools, to assess the impacts of given flow regimes on fish communities and fisheries.

2 INDICATOR SPECIES FOR CHARACTERISATION OF RIVER HABITAT TYPES

2.1 Overview

The evaluation of fish community types and their indicative species/species assemblages is essential, not only as a key component of the evaluation of flow requirements of fish species in the UK and the development of catchment abstraction water management plans, but also for the implementation of the Water Framework Directive into UK water management. The management of abstractions and river flows requires an understanding of flow requirements and tolerances of individual species (or size classes of species), in terms of swimming speeds and habitat/flow preferences (see Section 3), but also an understanding of how temporal flow characteristics, in terms of quantity and variability, determine the structure of functional fish communities. In respect of this latter issue, the identification of fish community types (characterised by species diversity, structure and abundance of key species), and the determination of the characteristics of flow regimes which regulate the structure and function of the communities, will enable abstraction applications to be evaluated on their potential impacts to fish communities, including both shifts in structural compositions and potential changes in functional ecological processes.

To meet this need, existing typologies for characterising fish communities in rivers were reviewed and limitations with respect to establishing a typology for fish communities in England and Wales were identified. This review was then used as a basis for developing a new typology for English and Welsh rivers using data collated from the EA fisheries monitoring programme. The dataset consisted of representative samples for fish communities of rivers in England and Wales although some areas were notably absent from the dataset (see Section 2.2.1). This dataset was used to discriminate the fish community types present in different river basins and reaches of rivers in England and Wales, irrespective of whether the fish community had been affected by human disturbance or not. In addition, the available data were screened for anthropogenic disturbance to identify the structure of fish communities in relatively undisturbed rivers to enable the natural environmental drivers (catchment characteristics and flow regimes) that determine fish community structure to be identified. In the light of the community typologies developed and the assessment of their relationship with flow regime variables, recommendations are made for the key species per community type and the requirements for further research.

2.2 Existing Characterisation of Fish Communities in Rivers

Each fish species has preferred habitat requirements (Cowx 2001; Tables 2.1 and 3.1) that result in changes in community structure along the upstream-downstream gradient of a river. These habitat requirements have long been recognised and used to classify different zones in a river (Hawkes 1975), such that different fish species with similar habitat preferences are grouped. For example, Huet (1949, 1959) developed a classification based on four zones ('trout', 'grayling' 'barbel' and 'bream') moving from the headwaters downstream. In this classification, it was proposed that "in any

Table 2.1 Distribution, behaviour and preferred habitat of coarse fish in rivers

Species	UK distribution	Behaviour	Preferred habitat characteristics
Salmon	Throughout the UK but restricted in east and south-east England	Anadromous, with adults spawning in natal streams	Juveniles found mainly upper, clear fast flowing rivers with gravelly substratum
Brown trout	Widespread throughout the UK	Anadromous and resident forms, with adults spawning in natal streams	Juveniles found mainly upper, clear fast flowing rivers with gravelly substratum
Bullheads	Widespread throughout England and Wales	Restricted home range	Stony stream and rivers, and some lakes
Stone loach	Widespread but limited in the North of Scotland	Restricted home range	Stony stream and rivers, and some lakes
River lamprey	Throughout the UK but restricted in many rivers	Anadromous	Moderately-flowing streams with areas of silt substratum
Brook lamprey	Fairly widespread and common throughout the UK		Moderately-flowing streams with areas of silt substratum
Eel	Throughout the UK	Catadromous	Middle and lower river reaches and small lowland tributaries
Roach	Throughout UK but limited in south-west England, Wales and Scotland	Home range; migrate to spawn	Lowland rivers; bankside vegetation or open water
Dace	Throughout UK but limited in south-west England, Wales and Scotland	Home range; migrate to spawn	Middle and lower river reaches and small lowland tributaries, sand/gravel/cobble substratum, moderate to high productivity
Chub	Throughout UK but limited in south-west England, Wales and Scotland	Home range; migrate to spawn; shoal when juvenile; larger fish tend to be solitary	Middle and lower river reaches, sand/gravel/cobble substratum, strongly associated with tree and macrophyte cover, large woody debris, rocks, moderate to high productivity
Common bream	Throughout UK but limited in south-west England, Wales and Scotland	Home range; shoal throughout life	Lowland reaches; slow flow, deep backwaters, vegetated areas, mud/silt substrate
Silver bream	Central, eastern and southern England only	Home range; shoal	Lowland reaches; slow flow, deep backwaters, vegetated areas, mud/silt substrate
Rudd	Throughout England and Wales, and parts of Scotland	Forms densely packed shoals; Large fish tend to be solitary	Mainly still waters; slow flowing lowland rivers associated with littoral macrophyte stands

Barbel	Throughout England, although restricted in the north and south-west	Large home range; migrate to spawn; live in small shoals or groups, often solitary when large	Middle reaches; Moderate to fast flow, moderate productivity, high oxygenation, gravel substratum, vegetation and obstructions
Tench	Through England, although restricted in the north and south-west	Solitary; congregate in late spring to spawn; Dormant in winter; live in small groups or sometimes pairs, can be solitary when large.	Lowland reaches, backwaters; mud/silt substrate
Carp	Widespread throughout the UK, except in northern Scotland	Usually found in small groups although larger fish solitary; Peak activity in summer	Slow flowing, vegetation, larger backwaters , productive rivers, occasionally in brackish water
Bleak	Throughout England except south-west, absent from Scotland, and Wales	Form dense shoals	Middle and lower reaches of moderate to high productivity rivers, silt/sand/gravel substratum
Gudgeon	Throughout England except south-west, absent from Scotland	Found in shoals	Middle and lower reaches, slow to moderate flow, silt/sand/gravel substrata, moderate to high productivity rivers
Pike	Widespread throughout the UK	Non-territorial, solitary Peak at activity at dawn and dusk,	Middle and lower reaches; slow-flowing to moderately-flowing, emergent vegetation
Perch	Widespread throughout the UK	Shoal when young, more solitary when older; Peak activity at dawn and dusk, inactive at night; Migrate to spawn	Lowland reaches; slow-flowing, occasionally moderate flow Shallow water with emergent and submerged vegetation Moderately productive water bodies
Ruffe	South, central and eastern England, a few sites in Wales and Scotland	Usually found in high density	Still and slow flowing habitats
Zander	Introduced; present in central and eastern England	Shoal when young; Peak activity in summer; daily maxima at twilight	Lowland reaches and large still waters; prefer shallow, turbid, oxygenated waters, hard substrate

given geographical area, rivers or stretches of river of like breath, depth and slope have near identical biological characteristics and very similar fish populations". Gradient is the primary feature characterising the zones, but width of the stream is also important. Both these characteristics in turn affect stream velocity, water temperature, substrate type, composition and abundance of macrophytes and the composition of benthic fauna. It should be recognised that this is an artificial separation for convenience and the zones are often overlapping because most fish species are able to tolerate a wide range of habitat conditions. However, whether a species thrives or merely survives depends on the individual species habitat preferences at different life stages (see Section 3.2).

This overlapping scenario is prevalent in species-rich river fisheries, which tend to show a continuum of fish community change regulated by the interaction of abiotic and biotic factors (Horwitz 1978), and parameters such as stream order (Kuehne 1962, Harrel *et al*. 1967, Whiteside & McNatt 1972, Zalewski & Naiman 1985), width (Angermeier & Karr 1983), distance from source (Horwitz 1978), habitat diversity (Gorman & Karr 1978, Schlossser 1982), gradient (Hocutt & Stauffer 1975, Zalewski & Naiman 1985, Changeaux 1995), temperature regime (Zalewski & Naiman 1985) and depth (Sheldon 1968, Evans & Noble 1979). As a result, Zalewski & Naiman (1985) developed a more holistic approach based on the River Continuum Concept (Vannote *et al.* 1980). This abiotic-biotic continuum concept is based on functional relationships between abiotic and biotic factors as a mechanism for regulating fish communities. In this approach, there are no sharp limits between zones and a more gradual change in species composition. The functional nature of the concept makes it generally more applicable than Huet's classic zonation. The continuum concept also includes all life stages of fish and interactions between species, thus making it more applicable. The range of the abiotic-biotic continuum varies from a stable predictable environment with a high species diversity, strong interspecific competition and narrow ecological specialisations, to an unstable environment where fish are on the brink of their physiological tolerance with no biotic selection or diversification. Environmental factors relevant to these changes in fish communities include:

(i) distance from source and catchment, which determine habitat diversity and stability;
(ii) temperature expressed in degree-days;
(iii) slope, velocity, substrate type and oxygen content (all of which are assumed to be interrelated).

More recent characterisations include the use of multivariate statistics, such as discriminant analysis, to describe the relationships between the environmental variables and aquatic communities (Moss *et al*. 1987, Weatherley & Ormerod 1987). This has led to a plethora of studies to explain the interactions between coarse fish species and their preferred habitat conditions (Grossman *et al*. 1987, Copp 1990b, c, 1991, 1992a, b, 1993a, b, 1997a, Copp *et al*. 1994, Baras *et al*. 1995, Copp & Garner 1995, Garner 1995, 1996a, 1997a,b, Garner & Clough 1996, Tales *et al*. 1996, Baras 1997, Godinho *et al*. 1997, Jurajda *et al*. 1997, Pilcher & Copp 1997, Watkins *et al*. 1997), but few have examined the characterisation of fish community types in different reaches of rivers.

The methods developed to date, whilst describing the relationships between fish species and habitat variables tend to be over simplistic in terms of characterising the different river types based on fish community structure. Furthermore, the methods available tend to be based on species rich communities and are not necessarily representative of British rivers. The approach developed below uses the multivariate approach to classify river reaches for fisheries purposes. It attempts to identify the categories of river/reach types in English and Welsh rivers and the key species by which these types can be discriminated. These river types can then be used as the basis for environmental management planning in the different systems.

2.3 Overview of Data Available for the Modelling

2.3.1 Fish community data

The fish community data collated for this analysis were sourced from the EA's National Fisheries Monitoring Programme and attempted to have national coverage (Figure 2.1). The majority of the samples were taken from the 2002 and 2001 sampling programme. However, the dataset from the South Anglian area (Essex) comprised many sampling occasions for each site, dating back to 1985. The database contained data for 1062 sites, representing 2141 fishing occasions. Unfortunately, no data were available for South Wales, the south-west of England (Devon & Cornwall), Lincolnshire or Northumberland (East coast rivers, north of the River Tees) (Figure 2.1). Additionally, to remove any variability caused by sampling methodology, these data were restricted to electric fishing surveys. This further restricted the distribution of sampling sites, minimising the number of samples from the main river stem of large rivers where netting, angler catch or hydro acoustics are often the primary sampling approach.

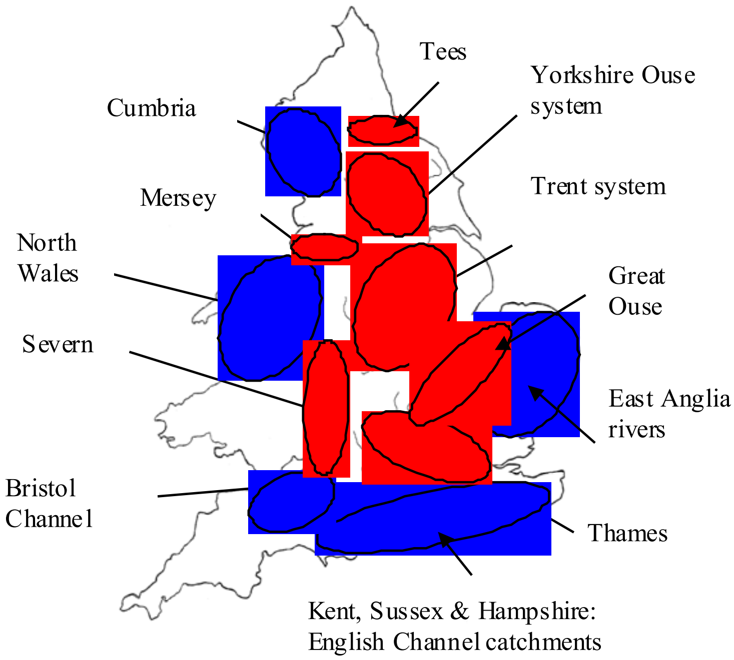


Figure 2.1 Major river basins and regions represented in database used to derive fish-based river typology

2.3.2 Abiotic data

Flow data (Base Flow Index [BFI], mean flow [MF_Nat] and Q5_Nat, Q50_Nat, Q70_Nat & Q90_Nat) and time-series hydrograph data (as mean daily discharge, m^3s^{-1}) were extracted from the Low Flows 2000 dataset (LF2000) and EA hydrological records. Where available, data were extracted per site in the "undisturbed" dataset (Section 2.4.5). Physical characteristics of the site (width, gradient, altitude, catchment size etc.) were either collected on site at the time of the fisheries survey, or extracted from maps or digital river networks. Additionally, each site was assessed by EA Area fisheries scientists against a suite of anthropogenic disturbance criteria including hydrological, morphological and water quality condition.

2.4 Fish Community Analysis

2.4.1 Characteristics of sampling data

Catch data were characterised by generally low fish species diversity, with only 4 to 9 species recorded in the majority of samples. Very few samples recorded more than 12 species (Figure 2.2).

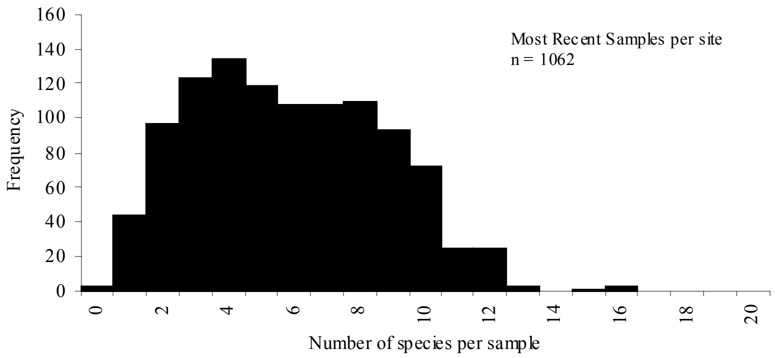


Figure 2.2 Frequency distribution of the total number of species caught per fishing occasion

Eel, *Anguilla anguilla*, and roach, *Rutilus rutilus*, were the most common species in the recent samples for each site in the English and Welsh dataset, each occurring on 52% of fishing occasions. The frequent occurrence of roach, however, reflects the large number of lowland sites reported by the Anglian Region (Southern Area) of the Environment Agency. Where present, stone loach (*Barbatula barbatula*), bullhead (*Cottus gobio*), minnow (*Phoxinus phoxinus*), salmon (*Salmo salar*) and three-spined stickleback (*Gasterosteus aculeatus*) exhibited the highest mean abundances of 1000-2000 individuals per hectare. Maximum abundances per site of over 10,000 individuals per hectare were found for eight species, including eel, roach, salmon and brown trout (*Salmo trutta*). Extremely high abundances (>35,000 individuals ha^{-1}) of stone loach, bullhead, minnow and stickleback were recorded on some fishing occasions (Appendix 1, Table A1.1). Twenty four taxa (22 native plus two alien), plus two types of hybrid, were found in more than 1% of the sites surveyed, whilst 13 taxa (including four alien and three “transitional zone” species) were only recorded in fewer than 1% of sites.

2.4.2 Community-type analysis: Cluster analysis method

Fish abundance data, recorded as catch per unit effort (CPUE) from the first fishing run ($n.ha^{-1}$), were entered into a hierarchical cluster analysis (statistical analyses were undertaken in SPSS version 11.5) to determine various fish community types represented in the dataset. Hierarchical cluster analysis is a statistical method for classifying samples into groups that have taxonomic meaning (Dytham 2003). The approach measures the similarity (phi-square measure) between each combination of the samples (similarity matrix) and then clusters them, based on their similarity, into groups. The clustering method used (Ward's method) is similar to an analysis of variance, minimising the variance in similarity values within groups compared with the variance between groups. When CPUE abundance data were used the clustering was driven primarily by species such as stone loach, bullhead, minnow and eel, which were relatively common but also had a greater range of abundance than many of the key species in both lowland and upland rivers. Therefore, standardised abundance data (maximum value of 1 per species) were used in the cluster analysis, giving a suitable balance between the presence/absence of a species and their abundance. Cluster analysis generates outputs as dendrograms (graphically representing the clustering of samples into similar groups) which were interrogated based on analysis of average community composition. Expert judgement was then used to determine the most appropriate number of distinct clusters.

2.4.3 Fish community types in English and Welsh rivers

Fish community data from the most recent sample per site in the dataset were clustered (Section 2.3.2) to discriminate the main types of fish communities present in English and Welsh rivers. The dendrogram produced from this analysis (Figure 2.3) was characterised by an initial major division of the sites into salmonid upland sites and lowland cyprinid communities. Additionally each of these primary groupings exhibited a single major division into two further clusters. Further divisions identified cluster solutions of six main groups with 15 potential community types (Figure 2.3). Each type was defined as a distinctive species assemblage, with characteristic diversity, composition and key species. However, some types exhibited common characteristics, related to the overall structure of the fish assemblages. Similar types were therefore linked into groups of species assemblages which reflected generic fish communities. The generic fish community groups linked similar types that were distinct in nature because of the geographical distribution of certain species in England and Wales (e.g. barbel, crucian carp etc.) and to the impacted nature of some of the fish communities in these rivers.

Within the 15-cluster solution the salmonid side of the dendrogram comprised three types of salmonid community, reflecting both the presence and abundance of salmon in relation to trout, together with the presence and abundance of complementary species such as bullhead, lamprey (*Lampetra* sp. and *Petromyzon marinus*) and stone loach. Community Type 1 consisted of sites with salmon present, whereas Types 2 and 3 represented sites characterised by trout and complimentary "minor" species such as bullhead, minnow, stone loach and eel (Table 2.2).

Within the cyprinid side of the dendrogram, the major division represented a split between community types, one group characterised by species such as chub (*Leuciscus cephalus*), dace (*Leuciscus leuciscus*), barbel (*Barbus barbus*) and gudgeon (*Gobio gobio*) (Group 2, Types 7, 8, 9, 11 and 12) and the other group by species such as bleak (*Alburnus alburnus*), pike (*Esox lucius*) and tench (*Tinca tinca*) (Group 4, Types 10, 13, 14 and 15) (Table 2.2). The sites belonging to community types in Group 4, characterised by pike, bleak and tench, originated from rivers with generally larger catchment areas, wider wetted widths and were generally deeper than those of Group 2.

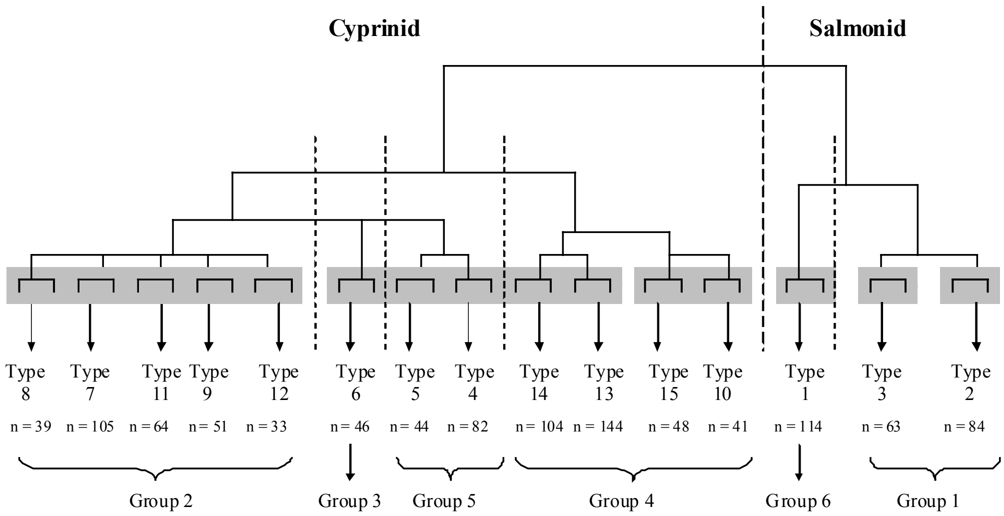


Figure 2.3 **Cluster dendrogram (Ward's method, Phi-squared measure) for fish species abundance (Standardised 1st run CPUE) in English and Welsh rivers. Six major groups, comprising 15 types, are identified.**

The sites from the Hampshire chalk rivers, Test and Itchen, characterised by salmon, trout, grayling (*Thymallus thymallus*) and pike, were clustered together distinctly as community Type 6, even within the six-cluster dendrogram solution (Table 2.2). Although this type is predominantly characterised as salmonid, it was clustered into the cyprinid side of the dendrogram due to the low abundance of salmonids, relative to other salmonid types, and the presence of cyprinid species such as dace, chub, barbel and roach.

Within the 15-cluster solution, the two community types (5 and 4) that formed Group 5 were characterised by high abundances of minnow, stone loach and eels. In particular, Type 4 was formed from sites with extremely high abundances of stone loach, bullhead and minnows. It is possible that this is due to sampling biases arising from non-consistent recording of these species during surveys. Analysis of the abiotic characteristics of these sites indicated that they represented sites with intermediate values of altitude, gradient and width, and may thus reflect intermediate communities between upland and lowland communities (and additionally sites from species poor areas) (see Appendix 1, Table A1.2).

Type 5 was characterised by the presence of coastal species such as flounder (*Platichthys flesus*) and gobies (Gobidae). In addition the average abundance of eels at these sites was very high. These sites reflect short rivers with good connectivity.

Table 2.2 Fish community type characteristics for the 15 community types identified by hierarchical cluster analysis of the fisheries survey dataset. Species marked in bold represent the key species of that fish community type (either the most abundant species for the type or the type within which the species exhibits its highest abundances).

Type	Key species	Complementary species	Abundance/comments
1	**Salmon** Brown Trout	Bullhead Stone loach	Very high abundances of salmon
2	Brown Trout **Bullhead**	Eel **Lamprey**	High abundances of trout and bullhead, salmon occasionally present
3	**Brown Trout**	Bullhead Stone loach	Very high abundance of brown trout, salmon, stone loach and bullhead may be present at low abundance
4	**Bullhead** **Stone loach** **Minnow**	Mixed	Characterised by very high abundance of “minor” species. Complimentary species from both salmonid and cyprinid fish present (represents >1 type?)
5	**Eel** Stone loach	Minnow **Flounder** Grayling	Characterised by very high abundance of “minor” species. Complimentary species from both salmonid and cyprinid fish present (represents >1 type?)
6	**Grayling** **Salmon** Brown Trout	Bullhead Lamprey	Abundance of salmon and trout relatively low but grayling exhibit highest average abundance between community types. Characteristic of sites from the Hampshire chalk rivers Test and Itchen.
7	**Barbel** Stone loach **3 Sp. Stickleback**	**Gudgeon** Roach **Chub**	Highest mean abundance of barbel. High abundance of stone loach and 3-spined sticklebacks. Relatively high abundance of chub
8	**Dace** Chub	Roach Gudgeon	Highest mean abundances of dace between types
9	Roach **Bream**	**Common Carp** Tench	Highest mean abundances of carp and bream between community types
10	Roach **Tench**	**Pike** Perch	Highest mean abundance of tench, relatively high abundance of pike
11	**Roach** Chub **Perch**	Minnow Bullhead	Highest mean abundances roach and perch
12	Roach **Rudd**	Chub Dace Gudgeon	Highest mean abundance of rudd
13	Roach	**Pike**	Relatively low abundances except for roach and pike which exhibited relatively high abundance compared to other types
14	Roach Gudgeon	Chub Dace	Relatively high abundance of dace
15	Roach **Bleak** **Bream**	Barbel Rudd	Relatively high abundance of typical large river/floodplain species

2.4.4 Identification of relatively undisturbed sites

It was recognised that the 15 community types identified in the fisheries survey dataset comprised sites over a range of ecological conditions related to different levels of human disturbance. As such, the 15 community types identified may represent both natural community structures and those fish assemblages of degraded rivers. To assess the relationship between fish community structure and flow it is important to assess flow in relation to natural fish communities in relatively undisturbed sites. For this analysis a sub-set of data from "undisturbed" sites was derived.

To achieve this, each site was scored (from 1 = undisturbed to 5 = highly impacted) for a number of key indicators of anthropogenic impact of the site's hydrological, morphological and chemical status. An assessment of the connectivity status of each site was also made (an assessment of anthropogenic disturbances to the accessibility of the site to migratory fish species, e.g. salmon, eels and lamprey). This information was used to derive a subset of data to determine the typology of natural fish communities. Initial analysis of the dataset identified that even when the exclusion criteria for "undisturbed" data were set at maximum impact scores of 1 or 2 (excluding connectivity) the "undisturbed" dataset failed to represent the range of fish communities which occur in England and Wales. In particular, few sites representative of cyprinid fish communities were retained in the dataset. To overcome this, an initial two-level selection process was established based on:

1. all sites with maximum impact scores of 1 or 2 (excluding connectivity);
2. additional sites with maximum impact scores of 3 – excluding sites that were deemed to be impacted salmonid sites (exclusion based on species composition and gradient).

A total of 417 sites were retained in the initial "undisturbed" dataset following this selection process. A hierarchical cluster analysis was undertaken on the initial "undisturbed" dataset, and the "undisturbed" dataset was restricted further by removing the most impacted sites from each of the major clusters in the dendrogram. This was done to minimise the chance of impacted sites affecting the assessment of each river type, especially for types where sufficient low-impairment data were available. This restricted the "undisturbed" dataset to 312 samples.

Brown trout was the most common species in the "undisturbed" dataset, reflecting the generally higher impact status of many of the lowland sites. Brown trout, bullhead, salmon, stone loach and minnow had the highest mean CPUE values of between 1000 and 2500 individuals per hectare. Where present, the mean abundance of flounder was also high (around 5000 individuals per ha). Dace, roach, chub and gudgeon were the most common lowland species in the "undisturbed" dataset (Appendix 1, Table A1.3). Only three alien species, carp (*Cyprinus carpio*), rainbow trout (*Oncorhynchus mykiss*) and pumpkinseed (*Lepomis gibbosus*), occurred in "undisturbed" samples (Appendix 1, Table A1.3).

The "undisturbed" dataset excluded proportionately more lowland sites and was reflected in the shift in distribution of the numbers of species per sample (Figure 2.5). The majority of samples in the "undisturbed" dataset comprised 2 to 5 species and few samples contained more than 10 species.

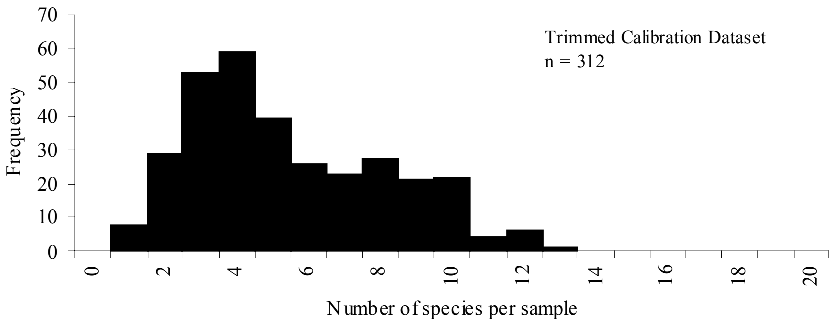


Figure 2.5 Frequency distribution of the total number of species caught per fishing occasion in the "undisturbed" dataset

2.4.5 Typology of "undisturbed" fish communities

Nine major clusters were identified within the resulting dendrogram (Figure 2.6). These were further explored and statistically tested. The nine clusters were assigned into eight fish community types (Figure 2.6). Three adjustments were made to the dendrogram based on ANOVA analysis of both biotic and abiotic variables for the nine major clusters, together with expert judgement (three adjustments correspond to clusters marked by A and B (comprising two changes) in Figure 2.6).

A. This cluster was determined by very high abundances of minnows, bullhead, stone loach and eels. However, analysis showed that both salmonid and cyprinid fish types occurred within this cluster. Therefore, the type was further divided by an additional cluster analysis on these sites to extract the predominantly salmonid sites and the sites characterised by cyprinid fish. These sites were then grouped with the most similar community type in the dendrogram.
B. Review of this cluster indicated that it comprised both lowland cyprinid fish sites and also sites that represented short coastal streams characterised by trout, eels, some cyprinid fish and coastal species such as flounder. Therefore, this cluster was divided by a further cluster analysis into two sub-sets. The lowland fish sub-cluster derived from this discrimination could not be easily distinguished from one of the other major cyprinid fish clusters so they were merged into a single fish community type (Type 6) (Figure 2.6).

Eight fish community types were described on the basis of the cluster analysis, each of which was characterised by both species composition and abundance (Table 2.3). Four salmonid types (including one region-specific type), three predominantly cyprinid types and one semi-transitional type were discriminated (note: sites are coded by longitudinal zonation, i.e. Type 1 highest the gradient). The restricted number of sites in the "undisturbed" dataset restricted the diversity of community structures that could be identified by cluster analysis. However, these eight fish community types for "undisturbed" sites are similar to the major groups of community types identified from all the survey data, particularly the two cyprinid types: viz. Type 6 characterised by roach, chub and gudgeon representing smaller cyprinid rivers; and Type 7, characterised by pike, bleak and roach representing larger lowland rivers (Appendix A1.4).

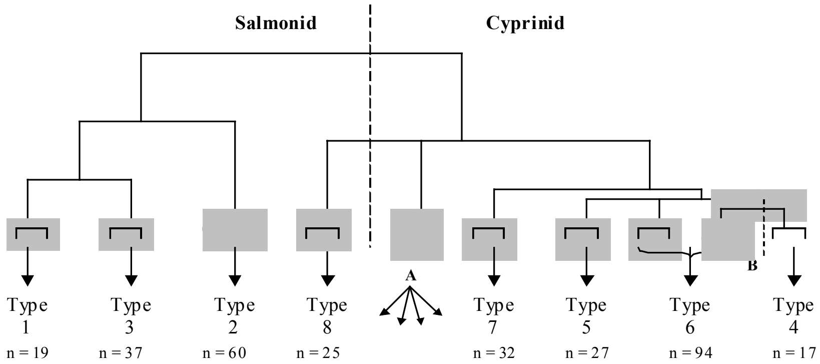


Figure 2.6 Cluster dendrogram (Ward's method, Phi-squared measure) for fish species abundance (Standardised 1st run CPUE) in English and Welsh rivers. Nine major clusters are identified.

Table 2.3 Fish community type characteristics for the 8 community types identified by hierarchical cluster analysis of the fisheries survey dataset from "undisturbed" sites. Species marked in bold represent the key species of that fish community type (either the most abundant species for the type or the type within which the species exhibits its highest abundances).

Type	Key species	Complementary species	Abundance/comments
1	**Brown Trout**	Salmon Bullhead	Very high abundance of trout
2	**Salmon** Brown Trout	Bullhead Stone loach	Very high abundance of salmon
3	Brown Trout	Eel **Bullhead** **Stone loach**	Very high abundances of "minor" species and high abundance of trout
4	Brown Trout **Eel**	**Lamprey** **Flounder**	Relatively high abundance of diadromous species plus some "coastal" species
5	**Barbel** Chub	Grayling Stone loach	Relatively high abundance of barbel, indicative of main river stem of large rivers
6	**Roach** **Bream**	**Gudgeon** **Chub** Dace Perch	Relatively high abundances of common cyprinid species
7	**Pike** **Bleak**	Roach	Presence of bleak and relatively high abundances of pike and bream, indicative of larger lowland rivers
8	Salmon **Grayling**	Brown Trout Pike	Relatively high abundances of grayling, salmon and pike. Constitutes sites from Hampshire Chalk Rivers Test and Itchen.

Again the community type characterised by grayling and salmonids, relating to the rivers Test and Itchen, was distinct within the analysis. Despite being characterised as salmonid, it again was clustered within the cyprinid side of the dendrogram due to the low abundance of salmonids relative to other salmonid types and the presence of cyprinids such as dace, chub and roach.

2.5 Relating Physical River Characteristics and Discharge Regime to Fish Community Types

2.5.1 Flow and site characteristics of rivers in the "undisturbed" dataset

Types 1, 2 and 3, the three main salmonid community types, were characterised by the highest altitudes and gradients (Table 2.4 and Appendix 1, Figures A1.1 - A1.8), whilst groups, 5, 6 and 7, the main cyprinid groups, were characterised by low gradients. Types 6 and 7, the main cyprinid types, differed both in altitude and gradient, with type 7 being characterised by predominantly lower altitudes and gradients. Additionally, despite the scarcity of data, Type 7 sites typically had a greater average depth to the river channel, suggesting this type reflects the lowland, main river stem of large rivers. Type 5 was characterised by large widths and long distances from source, but also by gradients and altitudes of upper lowland types, indicating that this represents the upper lowland type of large rivers, in particular the main river stem.

Table 2.4 Mean values of physical site characteristics and flow variables per "undisturbed" community type.

	"Undisturbed" community type							
	1	2	3	4	5	6	7	8
Size of catchment (km^2)	37.83	56.60	64.86	35.76	600.30	149.03	490.66	329.32
Altitude (m)	121.2	149.8	102.9	78.3	45.5	45.2	19.7	40.3
Gradient slope (m/km)	15.92	12.68	10.62	7.85	1.58	1.92	1.01	1.53
Distance from source (km)	10.67	13.23	13.68	10.24	55.07	23.88	40.75	33.04
Distance to mouth (km)	64.11	71.89	44.76	48.65	78.11	78.26	55.94	52.72
Wetted width (m)	4.21	7.52	4.98	4.25	19.44	6.92	11.26	12.60
Average depth (m)	0.21	0.36	0.36	0.43	0.81	0.63	0.97	1.09
Alkalinity	98.82	40.56	133.64	141.88	156.66	172.29	206.01	204.74
BFI	0.460	0.410	0.548	0.526	0.457	0.510	0.541	0.768
MF_Nat (m^3s^{-1})	0.716	1.957	1.559	0.656	8.075	1.632	3.589	4.962
Q5_Nat (m^3s^{-1})	2.319	6.626	5.220	2.082	28.056	5.438	11.633	13.290
Q50_Nat (m^3s^{-1})	0.417	1.074	0.870	0.388	4.284	0.903	2.069	3.506
Q70_Nat (m^3s^{-1})	0.244	0.623	0.515	0.240	2.572	0.568	1.310	2.455
Q95_Nat (m^3s^{-1})	0.099	0.240	0.226	0.113	1.213	0.301	0.667	1.492

Averages of flow statistics were different between fish community types. In particular, community Type 5 (main river stem of large rivers) and Type 8 (Hampshire Chalk rivers) exhibited the highest mean (MF_Nat, high (Q5_Nat) and low (Q95_Nat) flows. Type 8 was characterised by a high base flow index, reflecting the groundwater-fed nature of these chalk rivers. This was also reflected in the high value of the Q95, the low flow, for this river type. Types 1 and 4 exhibited the lowest high flow values (Q5_Nat), which was related to their small catchment sizes and wetted widths. The two main cyprinid types were different in flow characteristics, with type 7 generally linked to higher discharges resulting from greater catchment size, wetted width and depth.

It should be noted that channelised sections of large rivers, representing impacted river systems, are not included. The functional characteristics of these rivers, especially the flow regime and its influence on habitat and fish communities, will probably be different to that in natural river channels. These channel types need to be considered in the next phase of the project.

2.5.2 Flow characteristics influencing fish community structure

A stepwise, discriminant function analysis was undertaken to determine the abiotic characteristics that determine the fish community type classification. Stepwise discriminant function analysis is a multivariate statistical tool that models which combination of variables best describes the differences between the groupings. Furthermore, the tool determines a probability model for assigning samples to the pre-defined groups based on generating weightings for a number of variables that describe the individual sample. In this way, the model reduces the data for multiple variables into a set of compound functions. The first function explains the most variation between groups, the second function explains the next most variation, and so on. The relationships between the model functions and the individual variables are described by the weightings used to calculate each function from the variables and the correlation between the functions and the individual variables (Dytham 2003). The higher the weighting and correlation values the more important the individual variable is in explaining the variation between groups.

Log_{10} transformed data for gradient, alkalinity, size of catchment, linear distance to source, wetted width, altitude and distance to river mouth, together with calcareous geology and flow variables (Base Flow Index, Mean Flow, Q5, Q50, Q70, Q95) were entered into the analysis. Critical values of Wilks Lambda, $F = 3.84$ and $F = 2.71$, were used to determine variable entry or removal respectively. A seven-step analysis was undertaken and the final discriminant model included gradient, alkalinity, distance to mouth, altitude, Q5, Q70, and Q95 (Tables 2.5 and 2.6).

The first abiotic discriminant function was strongly correlated with the gradient of the site (canonical correlation = 0.867) and (Table 2.5) explained 72.7% of the variance in the discriminant model (Table 2.4). The variables included in the next steps further discriminate individual types. Function 2, which is positively correlated with the variables alkalinity, Q95 and BFI (although BFI is not included in the model) distinguishes Type 8, a type characterised by salmon and grayling, which is mainly driven by the south-coast chalk streams of Hampshire (e.g. rivers Test and Itchen). Function 3, which is positively correlated with alkalinity but negatively correlated with Q95 distinguishes the main salmonid community types - Types 1, 2 and 3. Function 4, which is strongly positively correlated with the flow variables, distinguishes community Type 5, the major lowland river type (characterised by barbel). Functions 5, 6 and 7 effectively aided discrimination between the two lowland cyprinid types, correlating with altitude, distance to mouth, gradient and Q5 (high flows) (Table 2.6).

Table 2.5 Eigen values and canonical correlations for the seven discriminant functions

Function	Eigenvalue	% of Variance	Cumulative %	Canonical Correlation
1	3.040	72.7	72.7	0.867
2	0.463	11.1	83.8	0.563
3	0.346	8.3	92.1	0.507
4	0.211	5.0	97.1	0.417
5	0.077	1.8	98.9	0.267
6	0.043	1.0	100.0	0.202
7	0.002	0.0	100.0	0.041

This analysis suggests that the major abiotic factors driving fish community structure (notwithstanding zoogeography effects) are related to river gradient, flow characteristics and water chemistry (as measured by alkalinity). Gradient discriminated between salmonid and cyprinid communities, illustrated by the x-axis in Figure 2.7 with the sequence from left to right from fish community types 1 and 2 to types 5, 6 and 7. Function 2 (y-axis in Figure 2.7) discriminates Type 8 (grayling, pike and trout) found in south coast chalk rivers, based on Q95 and alkalinity characteristics, reflecting the general high base flow and calcareous geology. Flow variables also discriminated the community (characterised by barbel and chub) found in the main river stem of relatively large rivers (characterised by both high Q5 and Q95 flows, but low base flow). This analysis underpins the importance of understanding the component aspects of catchment hydrology that drive fish community dynamics, and thus the potential effects of abstractions or water transfers thereon.

Table 2.6 Canonical correlation structure matrix between individual abiotic variables and derived discriminant functions

	Discriminant Function						
Variable	1	2	3	4	5	6	7
Log Gradient	-0.767	0.150	0.233	0.197	0.237	0.469	-0.141
Log Alkalinity	0.594	0.425	0.518	-0.195	0.269	0.275	0.107
Log Size of catchment[(a)]	0.487	-0.008	-0.308	0.323	0.065	-0.206	0.147
Log distance to source[(a)]	0.452	-0.114	-0.246	0.306	0.129	-0.220	0.113
BFI[(a)]	0.153	0.487	-0.151	-0.274	-0.091	-0.027	-0.135
Q5_NAT	0.112	-0.027	-0.274	0.774	0.333	-0.400	0.204
Mean flow_NAT[(a)]	0.125	0.048	-0.341	0.766	0.322	-0.380	0.179
Q50_NAT[(a)]	0.139	0.163	-0.432	0.737	0.297	-0.342	0.136
Q70_NAT	0.158	0.236	-0.484	0.717	0.266	-0.301	0.090
Q95_NAT	0.195	0.374	-0.563	0.660	0.175	-0.196	-0.026
Log width[(a)]	0.200	-0.090	-0.273	0.440	0.119	-0.274	0.119
Calcareous geology[(a)]	-0.041	0.167	0.041	0.173	-0.007	0.038	0.120
Log distance to mouth	0.026	-0.398	-0.107	0.204	0.002	0.674	0.577
Log Altitude	-0.393	0.001	0.028	0.286	-0.314	0.551	0.601

a. This variable not used in the analysis.

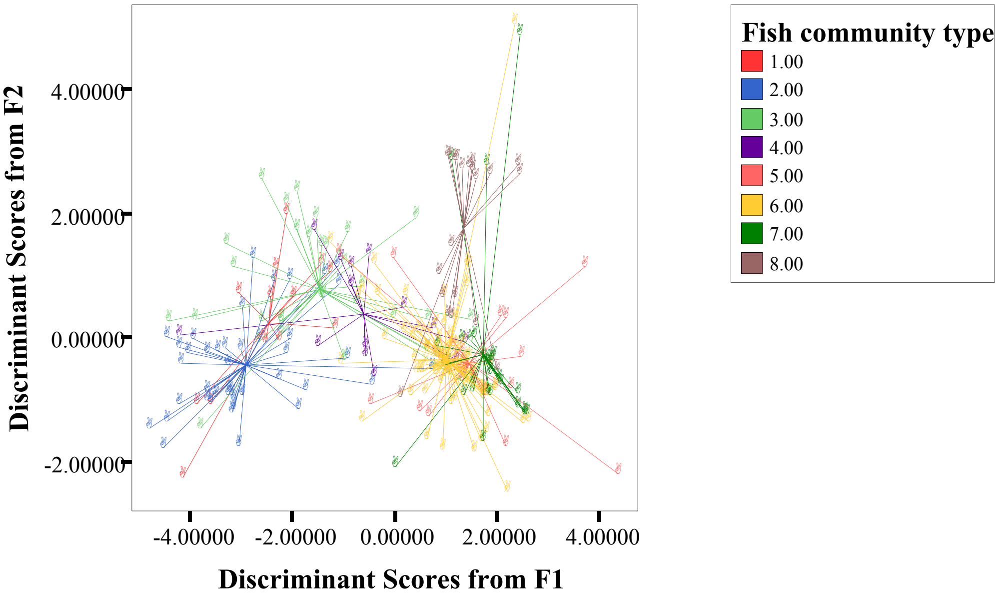


Figure 2.7 Plot of discriminant scores for the first and second discriminant function for "undisturbed" sites. Types are grouped based on the type predicted by the cluster analysis.

It should be noted, however, that this analysis only uses coarse measures of the hydrological regime. As such only the coarsest differences in flow characteristics were linked to differences in community structure, i.e:

- high base flow index and high Q95 discriminating groundwater fed rivers (Type 8, Function 2);
- high discharges discriminating fish communities in the main stem of large rivers (Type 5, Function 4);
- combination of gradient, alkalinity and low discharges (Q95 and Q75) discriminating the three main salmonid communities (Functions 2 and 3).

The use of discharge as a flow surrogate masks the variability of actual water velocities between different channel morphologies, and the use of variables such as Q values does not account for the high temporal variability of flow. It is probable that it is the variability within the hydrograph, the difference between high and low flows, together with the characteristics of the transition between low and high flows that controls the fish community structure (Section 2.5.3).

This type of analysis can be used to develop a predictive tool to classify the types of communities that should be expected in a particular reach of river under given flow regimes. However, for the tool to be useful in setting flow regulation requirements it will need to be further developed with more diverse flow statistics in addition to data for fish communities and flow regimes in rivers impacted by abstractions. As such, a new tool could be developed to predict the effects of abstractions on fish communities and to determine the flow and level criteria which are acceptable to protect the desired community structure (community structure reflecting good ecological status or potential).

2.5.3 Hydrograph characteristics for sites representing different fish community types

Mean flow characteristics, such as mean flow or Q95, do not reflect fully the complexity of inter- and intra-annual variability in flow characteristics, which may determine the structure of fish communities over longer timescales than seasonal or annual effects. Therefore, long-term hydrological patterns need to be considered in relation to fish community types and the potential impacts of abstractions. However, detailed hydrological records are not available for all fishery survey sites. Therefore, two-year hydrographs from EA (plus NRA and local Water Authority) hydrological data for one example “undisturbed” site from each of the fish community types (except for community type 1) (Figure 2.8, note different y-axis scales between salmonid and cyprinid river types) and flow exceedence curves (Burt 1992) (Figure 2.9) are presented to highlight some of the variability in flow characteristics (both within and between basins) that need to be considered for future assessments.

Although the analysis of the hydrographs, in relation to differences between community types, is not possible from single examples, some general features can be identified as potential characteristics of the reaches of rivers supporting the particular communities. Notable characteristics are those that differentiate between headwater catchments, lowland basins and groundwater-fed rivers with high base flows. The main differences are in the physical size of the flows and their variability. In general, the rivers comprising upland salmonid fish communities have much lower overall flows than lowland cyprinid rivers. However, flow in the upland salmonid rivers is much more variable, as indicated by the rapidly fluctuating nature of the hydrographs and the steepness of the flow duration curves, especially around low flows (e.g. comparison of type 3 hydrograph with type 7; Figure 2.8).

Two of the distinctive fish community types, types 5 (main river stem barbel zone) and type 8 (grayling, salmonid and pike community, predominantly from chalk streams) were also characterised by distinctive flow characteristics. Type 5 was characterised by very high flows with relatively high low flows, whilst Type 8 was characterised by ground-water fed flows with little variability and steady fluctuation between high and low flow (Figure 2.8). The link between distinctive fish community types and distinctive flow patterns indicates that it is not only the volume of flow, but a combination of volume, rate, extremes and variability of flow that regulated fish community structure, and probably population structure through habitat requirements of different life stages. One other feature of note is the steepness of the % exceedence example of Type 3. This type was characterised by low abundances of salmon but very high abundances of small-bodied “minor” species. It is possible that the flow regime of these systems may restrict the suitability of habitat for salmon and may be more suitable for small bodied benthic species (bullhead, stone loach and minnow) that can cope with very low flows and avoid high spate flows, and thus thrive due to low competition from larger-bodied benthic species (e.g. juvenile salmon). However, this is an area of research that needs to be investigated in Phase II of the project.

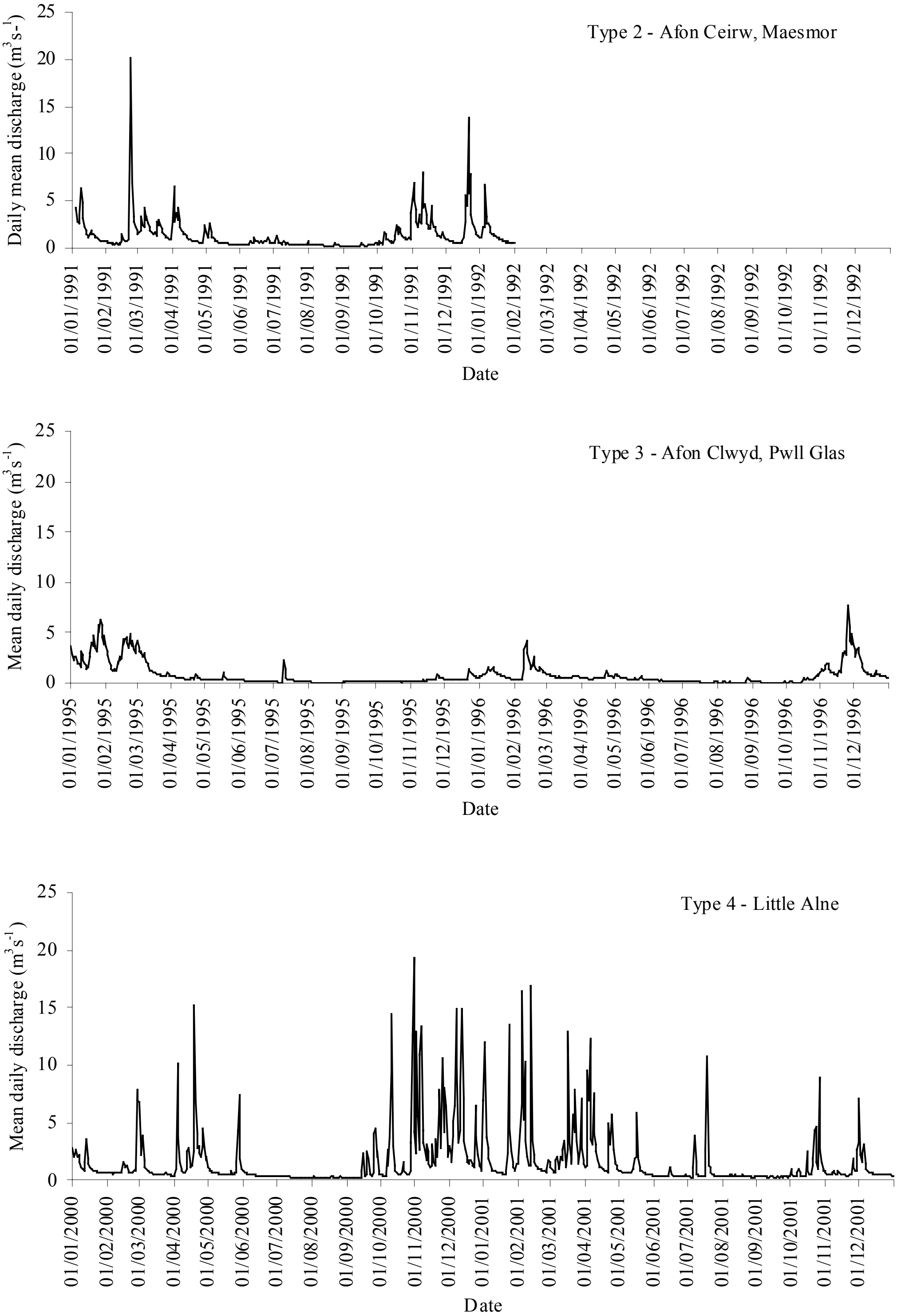
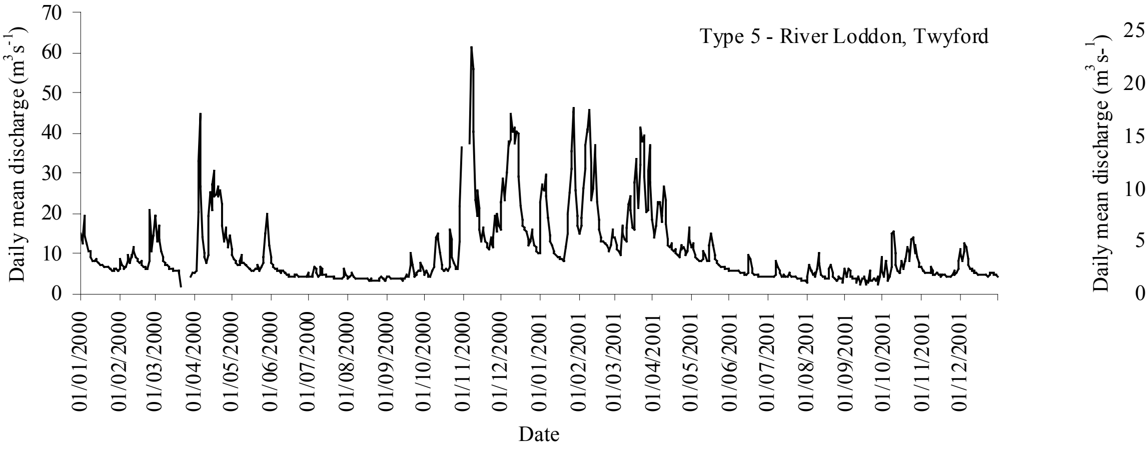
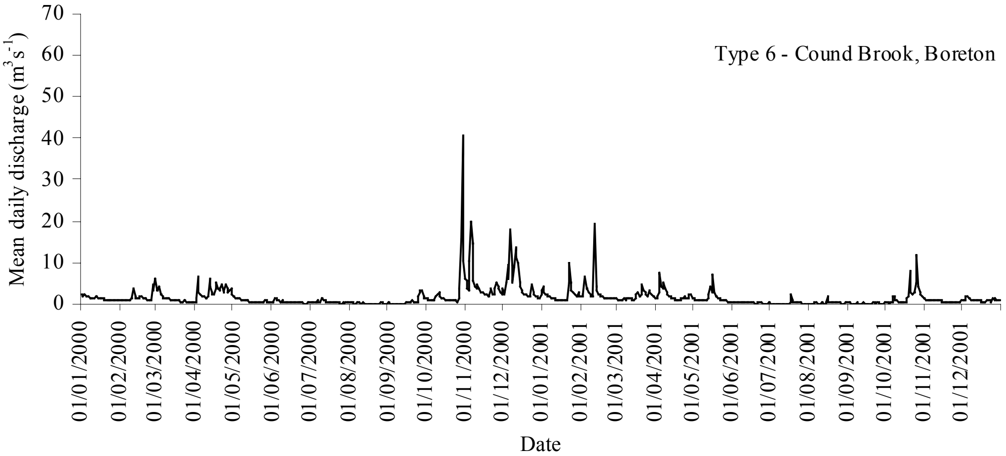


Figure 2.8 Two-year discharge hydrographs for example sites from the three upland fish community types (nb. only one year of data were available for Type 2, Afon Ceirw)





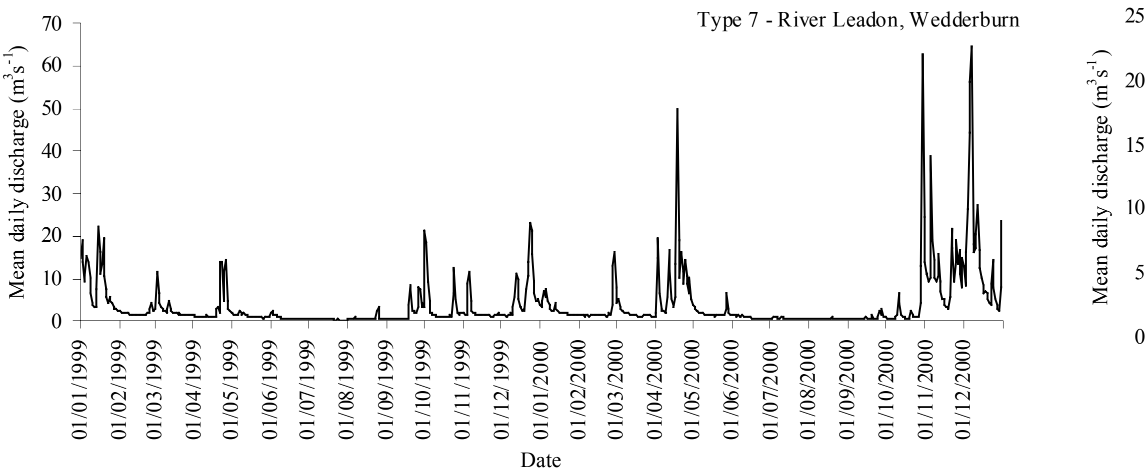


Figure 2.8 continued Two-year discharge hydrographs for example sites from the three lowland cyprinid fish community types

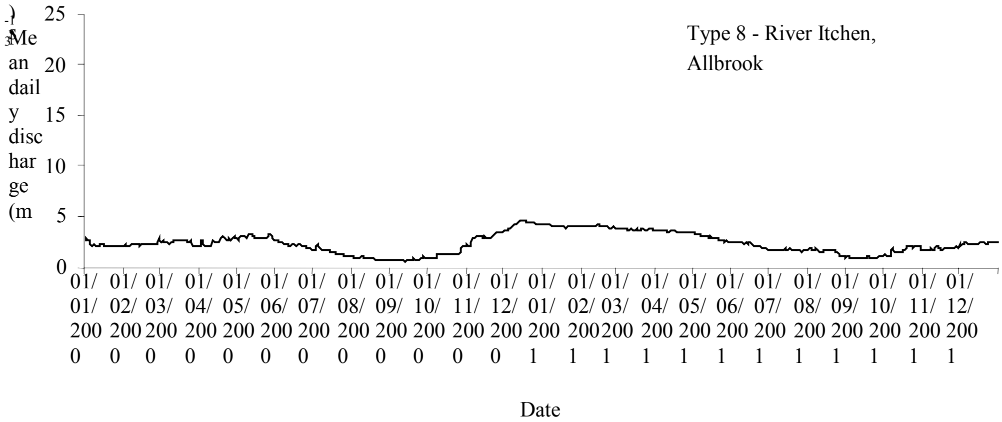


Figure 2.8 continued Two-year discharge hydrographs for an example site from fish community type 8, the River Itchen, Hampshire

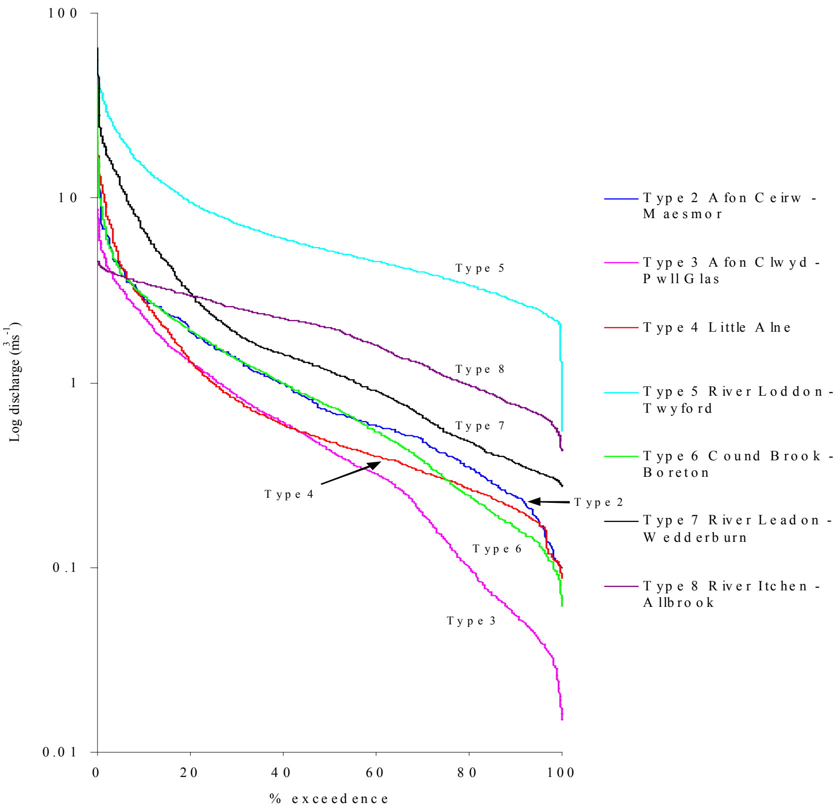


Figure 2.9 Flow duration curves for example sites from seven of the eight "undisturbed" fish community types

2.6 Summary and Conclusions

The fish types derived from both the "undisturbed" and full dataset were broadly similar, reflecting eight major fish community types. The classifications broadly followed the classical zonation theory with river gradient from upland salmonid to lowland cyprinid communities (Huet 1959). The typology identified two main types of salmonid community; those with salmon and those without (although this may also be an artefact of anthropogenic impacts on English and Welsh rivers particularly in relation to longitudinal connectivity issues). The typology identified two main types of cyprinid fish communities, reflecting zonation by gradient from upper lowland communities characterised by rheophilic species (e.g. dace, chub, barbel) to lower lowland communities characterised by eurytopic species (e.g. bleak, roach, bream [*Abramis brama*], pike and perch [*Perca fluviatilis*]), together with classic floodplain species such as tench and rudd (*Scardinius erythrophthalmus*). Additionally, the typology discriminated a fish community characterised by pike, trout and grayling associated with chalk streams (ground-water fed rivers), a fish community associated with the main river stem of large lowland rivers, characterised by barbel, and a fish community type dominated by bullhead, minnows and stone loach. **It is important that the influence of flow and the potential impacts of abstractions and releases be considered within the context of each of these main fish assemblages, linking key species per community type to their functional ecology and flow requirements (see Section 3).**

Within the salmonid communities, the influence of flow regime on the structure of the community is linked to the swimming capacity of different life stages of salmonid species and the availability of habitats (pool, riffle, glide) for different sizes of fish. Additionally, the influence of flow regime on the quality and availability of spawning habitat is critical (notwithstanding the influence of timing and quantity of discharge on the migration of adult salmon and sea trout, and the accessibility of suitable spawning habitat). **The relationship between the rate of flow, the rate of change of flow, the duration of high/low flow events and their seasonal timing, and their influence over the functioning of salmonid populations** (spawning, recruitment and growth) **needs to be considered when evaluating anthropogenic changes to flow patterns.** In most cases, it is probably the influence of abstractions on low flows, the associated compaction and exposure of spawning gravels, together with the reduction in the habitat availability for 0+ fish and the loss of habitat for large fish, which have most influence on the structure of salmonid communities.

Within the cyprinid communities, there are essentially two main types that need to be considered in relation to the flow regime. The split is linked to the functional ecology of the dominant species, particularly their spawning requirements. The upland cyprinid communities, characterised by barbel, chub and dace, are essentially rheophilic/ lithophilic functional communities requiring high flows and clean gravels for spawning, whereas lowland cyprinid communities are essentially dominated by eurytopic/ phytophilic species (roach, bream, pike), which prefer slow flows and spawn on aquatic vegetation. Additionally, lowland cyprinid communities are characterised by limnophilic species (tench), which tend to utilise off-river floodplain water bodies for spawning. Within these two main cyprinid communities is a cyprinid community characterised by barbel, which is associated with the main river stem of large lowland rivers.

For cyprinid communities the influence of flow, specifically, regulation of flow, alters the balance between rheophilic and limnophilic assemblages. Additionally, the regulation of the extent and timing of high and low flow events will influence the extent and timing of lateral connectivity with backwaters and floodplain water bodies, critical habitats for spawning and refuge. It is probable that it is the main channel morphology and the diversity and availability of off-channel habitats that influences the impact of extreme flow events on cyprinid fish communities.

In addition to the types formed by longitudinal zonation, one important regional type, the calcareous rivers of the south coast (especially Hampshire), was discriminated. These sites were generally characterised by low abundances of a community in which grayling, trout and pike were common. **This distinctive fish assemblage was associated with groundwater-fed rivers, with little variability in flow, high base flows and smooth transitions in the flow regime.** However, grayling-based communities are not restricted to rivers such as the Test and Itchen, and further studies will require that data are collected for other rivers with grayling populations.

It is probably that there are biases in the dataset, in particular the under representation of certain geographic areas of England and Wales and the minimal sampling on certain zones of rivers. The absence of data from Lincolnshire, the Yorkshire Ridings rivers, the south-west and South Wales form regional gaps that may hide important regional types or geographic patterns. Furthermore, the limited number of surveys undertaken on main river stems in lowland reaches, together with low numbers of sites reflecting the grayling reaches of rivers in many regions, probably masks some important community types. The limited surveys in the grayling zone from regions other than the Hampshire rivers possibly resulted in the absence of a general grayling zone within the typology. **An important aspect of the next phase of the project is to try and remove these biases by filling the gaps in information, especially with respect to the regions mentioned above and the sections of large lowland rivers not represented in the current dataset.**

Additionally, it is probable that flow statistics such as mean flow and Q values do not accurately reflect elements of the flow regime that influence fish assemblages. **Further work needs to be undertaken to determine the long-term influence of hydrograph characteristics on the fish communities and their dynamics.** For this, gaps in the current dataset need to be filled and additional data about population dynamics (recruitment, size and age structure) of the key species need to be considered, over and above community composition data, to determine the long term influence of flow patterns on recruitment success and life histories (See Section 3).

3 SPECIES HABITAT REQUIREMENTS

3.1 Introduction

Section 2 showed that fish are useful organisms for characterising environmental conditions in streams and rivers. However, to it is important to elucidate the underlying conditions that drive the community characterisation. With respect to flow-related issues this requires:

- determination of the basic habitat requirements of all life stages of fish species in rivers, and the seasonal habitat changes required to maintain population status;
- determination of the critical flow-related factors affecting the various life stages of fish species in rivers;
- from the above, identification of key seasonal habitat requirements in relation to flows and water levels

To achieve this, the flow-related habitat requirements of the various life stages of the main freshwater fish species found in the UK were reviewed (Appendices 2 and 3) and summarised. This information was used to identify which characteristics of flow regimes / hydrographs that have biological significance in determining fish community structure in various river types and limitations in the information for informing policy on water resources schemes.

3.2 Relationship between Habitat and Fish Communities

The biological integrity of fish populations and communities are directly related to the variety and extent of natural habitats and related processes within a river basin. Consequently, a stream ecosystem has to retain its functional complexity to maintain healthy fish populations and community structure. This functional complexity within a stream ecosystem depends mainly on geomorphological and hydrological processes that influence channel morphology and instream biota, and form a mosaic of stream channels and floodplains. Water flow is the main agent responsible for shaping the physical habitat, but rivers and riverine habitats are influenced by interactions with the adjacent environment; more so than are other ecosystems. Thus, functional complexity, usually expressed in terms of habitat diversity, is a major factor in determining the structure and abundance of fish populations, and is in turn a function of depositional and erosion processes, which are directly influenced by the hydrological regime.

Habitat is used here in the sense of the usable range of a stream in which a fish species can live. In its broadest sense, the term habitat defines where a fish species lives without specifying resource availability or use. An individual fish seldom spends its entire life in the same habitat, and even species that are considered as resident in a particular reach (e.g. brown trout) may migrate from spawning grounds in upstream or downstream directions and choose more suitable habitat for particular life stages. In its life history, an individual commonly requires a different habitat with suitable microhabitat conditions for each specific life stage (Figure 3.1).

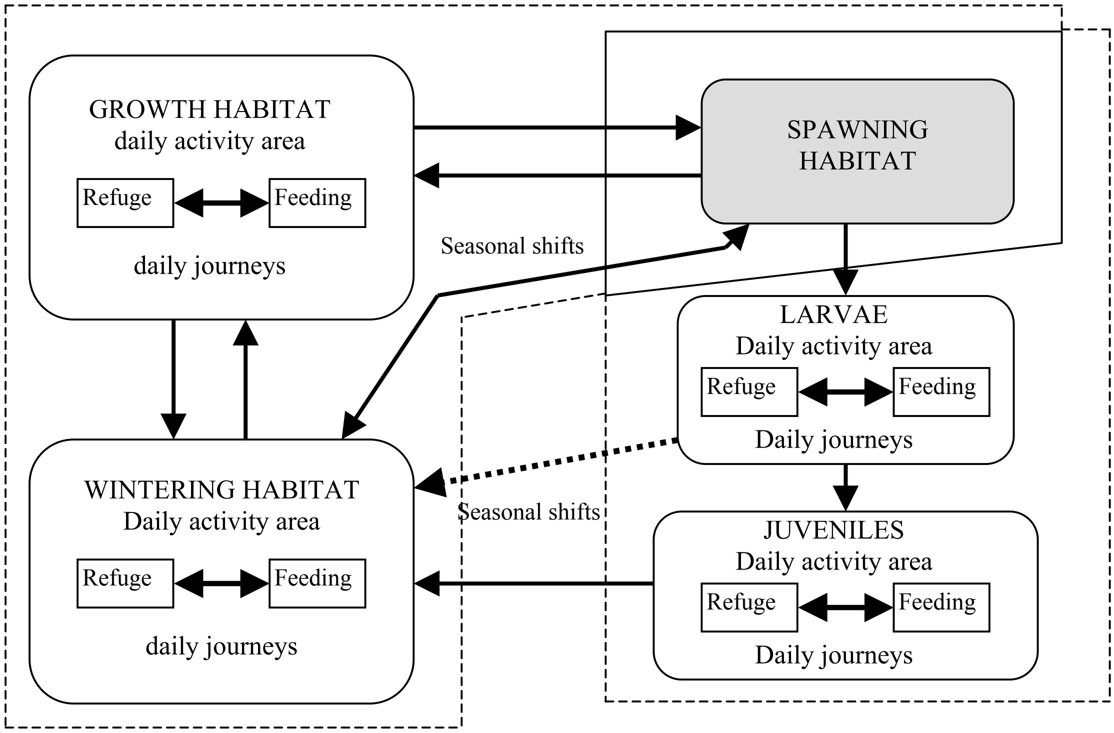


Figure 3.1 Illustration of the functional unit concept for fish habitat requirements (modified from Cowx & Welcomme 1998)

Microhabitat for an individual fish is the localised area where the fish is located at any point in time. Microhabitat characteristics are influenced by structural stream complexity, light intensity, hydraulic variables and stream substratum. Because these elements change over time, microhabitat use has to be considered from a time series perspective. Thus habitat use by an individual fish is a sequence of events, depending on life stage and time (fish may move daily between feeding and resting areas and seasonally between feeding, resting and spawning areas). Seasonal movements may be strongly influenced by discharge. Microhabitat can be characterised by the specific combination of habitat elements, especially depth, water velocity, substrate and cover used, in the place occupied by a fish at a certain time.

3.2.1 Habitat concepts

Fish in rivers depend on undamaged interactive pathways along four dimensions, i.e. longitudinal, lateral, vertical and temporal. Fish display migratory patterns that play an important role in their ecology. To complete their life cycle, fish species need suitable spawning sites. These can be quite close to the areas in which they live as adult fish, but, to optimise reproductive success and counteract flow displacement effects, many fish species return to their natal streams or use upstream spawning grounds or tributaries, side channels and backwaters. Other reasons for migration include optimum feeding strategies, avoidance of unfavourable conditions, or to enhance colonisation (dispersion, Fig. 3.1). The scale of the migration can range from tens of metres (resident fish, e.g. brown trout, bullheads) to tens or hundreds of kilometres (potamodromous migration, e.g. lake or river resident brown trout, barbel), or even to thousands of kilometres (diadromous migration, i.e. sea trout, salmon, eel). Unfortunately, there is no advantage in combining different fish species into migration guilds based on the scale of their migration distances because different patterns of migration may occur simultaneously in components from the same age group, within a single species or even within a single population. For example, spawning migration behaviour for brown trout is highly variable; some individuals show resident behaviour and spawn close to where they live, whereas others carry out spawning migrations of distances up to 50 km. However, there may be advantages in combining reproductive guilds to determine the impacts of altering discharge regimes on fish populations and communities.

Survival and life history are directly related to intact migration pathways, including the possibility of migration into tributaries, side channels and backwaters that are often very important for reproduction, but also serve as rearing areas for larvae and young fish (Fig. 3.1). Each barrier to migration, whether physical or chemical, has an effect on the structure of fish populations and composition of the fish community upstream of the barrier. This barrier-effect view of the way fish communities are distributed in river ecosystems relates to the concept of longitudinal and lateral pathways and is connected to the habitat-centred fish view. Factors that create the barrier effect, such as construction of weirs and dams, channelisation, or altered flow regimes that affect the ability of fish to bypass an obstruction or access backwaters and side channels, will alter the fish community dynamics and functioning.

The lateral dimension refers to the importance of inshore zones, permanently and temporarily connected backwaters, areas that flood during high flows and their linkages with the main river channel River edge and backwater habitats serve not only as preferred feeding and refuge areas but also as spawning areas, depending on the fish species. Young-of-the year fish tend to be associated with the riverbank, especially rheophilic species, which live at or near the stream edges and in off-river channel habitats. Unregulated rivers with unconstrained river channels may provide these required areas under low flow conditions, but flow regulation can alter the area available and constrain recruitment in the population, where reduction in preferred habitat occurs.

The vertical dimension deals with riverine-groundwater interactions and concerns mainly fish species that bury their eggs in gravel depressions called redds (lithophilic fish species, e.g. salmon, trout, grayling). Habitat requirements of eggs and embryos during incubation in substrate interstices are different from those of fish living in the open water. To ensure the development of the embryo, sufficient water must flow through the gravel as deep as the eggs to supply the embryos with oxygen and carry away metabolic wastes. Hydrological processes in the groundwater-river exchange play an important role for successful reproduction of lithophilic fishes. Trout seem to avoid zones of undiluted groundwater upwelling, preferring zones of intermediate surface-groundwater mix. To maintain high intra-gravel oxygen concentrations in spawning areas, high permeability of the streambed is important. Thus, concentrations of fine sediment >15-30% of the total substrate volume will be detrimental to the survival of eggs and embryos of salmonid species.

The fine sediments that clog up the gravels enter mainly by erosion processes, but altered flow regimes affect the deposition processes. Reduction in discharge volume generally reduces the flushing effects of flows and allows great sedimentation. This can both clog the gravels, with potential impacts as described previously, or can overlay the surface of gravels and other spawning substrata. This is particularly detrimental to those species that lay eggs on the surface of or very shallow within the gravels or on aquatic macrophytes, because the eggs are smothered and tend to have low survival. Consequently, natural hydrological processes determine reproductive conditions and survival of embryos of lithophilic fish species.

3.3 Habitat Requirements of Salmonids

The common and widespread river salmonids of the UK are the Atlantic salmon, *Salmo salar* L., which has an anadromous (breeds in fresh water but goes to sea to feed) life history strategy, the trout, *Salmo trutta* L., which has migratory and non-migratory components in its genetic population, and the grayling *Thymallus thymallus* (L.), which exhibits a potamodromous (i.e. migrates within fresh water) life history strategy. Salmonid life cycles have been described in detail in the literature (Mills 1987; Shearer 1992; Crisp 2000), and much of the basic life cycle pattern is common to both *Salmo salar* L. and the anadromous form of *Salmo trutta* L. The habitat requirements to support different stages of this life cycle are complex, but also broadly similar between species (Table 3.1; Appendix 2).

3.3.1 Eggs, incubation and inter-gravel stages

Salmon and anadromous trout return from the sea to their natal river, and seek to spawn in their natal tributary. Some non-anadromous trout also migrate upstream to spawn. The first essential requirement is access to the spawning sites. Low flows in rivers and streams often restrict the upstream migration of adult fish to the spawning habitat, because there is insufficient flow for them to negotiate both natural and artificial barriers (see Sambrook and Cowx 2000). Salmon and trout exhibit spawning site selection which is governed by a complex of environmental cues, including intra-gravel flow, gravel size, depth, stream velocity and cover (Crisp 2000; Table 3.1). These factors are essential for successful spawning, egg survival and hatching.

Male grayling occupy spawning territories and each territory must contain a bed of fine gravel suitable for spawning, hiding places for females before spawning (e.g. overhung banks, large stones) and visual isolation form adjacent territories. The preferred water depths vary from 20 to 65 cm (mean 36 cm), and water velocities of 33-80°cm s^{-1} (mean 54cm s^{-1}). Grayling spawning occurs in water temperatures ranging from 3.5-16.2°C, with a preference for the mid-range. Sudden temperature drops, for example late snowmelts, may inhibit the process.

River bed gravels contain two main components; a framework of larger particles supporting one another in an open structure, and smaller particles described as a matrix that fill the spaces to a greater or lesser extent (Crisp 2000). Sediment characteristics related to flow may affect spawning and subsequent survival of progeny in the following ways:

Table 3.1 Summary of the range of habitat characteristics occupied by different life stages UK river fish species based on literature. Where sufficient data are available, preferred habitat characteristics are identified.

Species	Life stage	Water depth requirements	Flow requirements
Abramis brama	Larvae	20 - <150 cm	<5 $cm.s^{-1}$
	Juvenile	<100 - ~125 cm	<5 $cm.s^{-1}$
	Spawning	25 - ~50 cm	<20 $cm.s^{-1}$
Alburnus alburnus	Larvae	20 - <100 cm	<5 $cm.s^{-1}$
	Juvenile	<20 - >100 cm	<5 $cm.s^{-1}$
	Spawning		<20 $cm.s^{-1}$
Alosa alosa	Larvae	Shallow	Slow
	Juvenile	- 300 cm	
	Spawning	50 - 300 cm	50 - 200 $cm.s^{-1}$
Alosa fallax fallax	Larvae	Shallow	Slow
	Juvenile	- 300 cm	
	Spawning	15 - 300 cm	
Anguilla anguilla	Juvenile	<600 cm	>10 $cm.s^{-1}$
Barbatula barbatula	Juvenile	0 - 20 cm	Still - elevated
Barbus barbus	Larvae	0 - 40 cm	<20 $cm.s^{-1}$
	Juvenile	<20 - 100 cm	Still - 120 $cm.s^{-1}$
	Adult		40 - 100 $cm.s^{-1}$
	Spawning	15 - 40 cm	25 - 49 $cm.s^{-1}$
Blicca bjoerkna	Larvae	>50 cm	
	Juvenile	<50 - >100 cm	<5 $cm.s^{-1}$
	Spawning	10 - 90 cm	5 - 60 $cm.s^{-1}$
Cobitis taenia	Larvae	25 - 45 cm	Still/negligible
	Adult	34.6 cm	<15 - 30 $cm.s^{-1}$
	Spawning	25 - 45 cm	No preference
Cottus gobio	Juvenile	Shallow	Elevated
	Adult	>5 - 40 cm	10 - >40 $cm.s^{-1}$
	Spawning	>5 cm	
Cyprinus carpio	Juvenile	Shallow	
	Spawning	80-100 cm	<5 $cm.s^{-1}$
Esox lucius	Larvae	<150 cm	
	Juvenile	- ~175 cm	Still
	Spawning	50 - 500 cm	<5 $cm.s^{-1}$
Gasterosteus aculeatus	Juvenile	Shallow	Elevated
	Adult	>20 cm	Slow
Gobio gobio	Larvae	Shallow	<20 $cm.s^{-1}$
	Juvenile	<20 - <100 cm	0 - 40 $cm.s^{-1}$
	Adult		<55 $cm.s^{-1}$
	Spawning	5 - 8 cm	2 - 80 $cm.s^{-1}$

Gymnocephalus cernuus	Larvae	50 cm	
	Adult		Still
Lampetra fluviatilis	Larvae	0 - 100 cm	1 - 50 cm.s^{-1}
	Spawning	20 - 150 cm	100 - 200 cm.s^{-1}
Lampetra planeri	Larvae	<50 cm	8 - 10 cm.s^{-1}
	Spawning	3 - 150 cm	30 - 50 cm.s^{-1}
Leuciscus cephalus	Larvae	20 - <100 cm	<5 cm.s^{-1}
	Juvenile	<20 - <100 cm	<5 cm.s^{-1}
	Spawning	>0 - 128 cm	<5 - 75 cm.s^{-1}
Leuciscus leuciscus	Larvae	2 - 50 cm	<2.5 cm.s^{-1}
	Juvenile	<50 cm	Still - elevated
	Adult	17 - 113 cm	0 - 57 cm.s^{-1}
	Spawning	25 - 40 cm	20 - 50 cm.s^{-1}
Osmerus eperlanus	Larvae	- ~250 cm	
	Juvenile	- ~250 cm	
	Spawning	Shallow	Turbulent
Perca fluviatilis	Larvae	<150 cm	
	Juvenile	- ~300 cm	Still or slow
	Spawning	200 - 300 cm	
Petromyzon marinus	Larvae	0 - 220 cm	0 - 17 cm.s^{-1}
	Spawning	13-170 cm	30 - 200 cm.s^{-1}
Phoxinus phoxinus	Larvae	<15 - >40.5 cm	<1.9 - >3.46 cm.s^{-1}
	Juvenile	<34.7 - >53.4 cm	<3.85 - >12.8 cm.s^{-1}
	Adult	10 - >50 cm	0 - >35.9 cm.s^{-1}
	Spawning	10-25 cm	20 - 30 cm.s^{-1}
Pungitius pungitius	Juvenile	Shallow	Elevated
	Adult	>20 cm	Slow - 10 cm.s^{-1}
Rhodeus sericeus	Juvenile	<25 cm	<10 cm.s^{-1}
	Adult	10 - 40 cm	10 - 50 cm.s^{-1}
Rutilus rutilus	Larvae	20 - 150 cm	<5 cm.s^{-1}
		(<100 cm preferred)	(lentic preferred)
	Juvenile	20 - ~175 cm	0 - 40 cm.s^{-1}
		(~50 - 100 cm preferred)	(lentic preferred)
	Spawning	15 - 45 cm	- >20 cm.s^{-1}
Salmo salar	Fry	<10 - 40 cm	5 - 65 cm.s^{-1}
		(=20 cm preferred)	(~15 - 40 cm.s^{-1} preferred)
	0+	<100 cm	5 - 65 cm.s^{-1}
		(<25 cm preferred)	(~15 - 50 cm.s^{-1} preferred)
	Juvenile	5 - 100 cm	0 - <100 cm.s^{-1}
		(~20 - 40 cm preferred)	(~5 - 50 cm.s^{-1} preferred)
	Parr	>10 - <100 cm	4 - <120 cm.s^{-1}
		(~25 - 60 cm preferred)	(~10 - 60 cm.s^{-1} preferred)
	Spawning	15 - 91 cm	>15 - 90 cm.s^{-1}
		(~25 - 50 cm preferred)	(~20 - 50 cm.s^{-1} preferred)

Salmo trutta	Fry	<60 cm	0 - <30 cm.s^{-1}
	0+	<20 - 30 cm (~20 - 30 cm preferred)	<10 - 50 cm.s^{-1} (~10 - 20 cm.s^{-1} preferred)
	Juvenile	5 - 240 cm (~20 - 30 cm preferred)	0 - 44 cm.s^{-1} (<25 cm.s^{-1} preferred)
	Parr	<5.1 - 300 cm (~40 - 75 cm preferred)	0 - 65 cm.s^{-1} (~20 - 30 cm.s^{-1} preferred)
	Adult	9 - 305 cm (~40 - 75 cm preferred)	0 - 142 cm.s^{-1} (~25 cm.s^{-1} preferred)
	Spawning	6 - 91 cm (~25 - 50 cm preferred)	10.8 - 81 cm.s^{-1} (~20 - 50 cm.s^{-1} preferred)
Sander lucioperca	Larvae	>50 - 250 cm	
	Juvenile	- ~250 cm	
	Adult	1.2 - 38 m	0.01 - 0.86 cm.s^{-1}
	Spawning	50 - 300 cm	10 - >70 cm.s^{-1}
Scardinius erythrophthalmus	Larvae	Variable	Still
	Juvenile	>100 cm	Still
	Spawning	10 - 90 cm	<5 cm.s^{-1}
Thymallus thymallus	Larvae	10 - 90 cm	6 - 50 cm.s^{-1}
	Juvenile	40 - 60 cm	<10 - 110 cm.s^{-1}
	Adult	20 - 400 cm	20 - 110 cm.s^{-1}
	Spawning	10 - 50 cm	23 - 91.7 cm.s^{-1}
Tinca tinca	Larvae	No preference	Still
	Juvenile	No preference	Still
	Spawning		<20 cm.s^{-1}

- Framework particles of too small a size may be rejected because of a lack of intra-gravel flow that limits the provision of oxygen and removal of toxic substances from the redd site.

- A large amount of small material known as fines or sand within the bed matrix may lead to siltation or concretion of spawning areas. The influx of fines into the gravel matrix causes a reduced intra-gravel flow, thus slowing the processes of oxygen supply and waste removal. Gravels containing >10% fines are generally considered to threaten the intra-gravel stages of salmon (O'Connor & Andrew 1998, Crisp 2000). In a bed matrix extensively filled with finer sediment, the bed may become compacted. This is termed concretion.

The gravel composition may be determined by the inputs available from erosion; however, the degree of sorting is determined by stream discharge and geometry. Fluctuations in discharge play an important part in shaping stream characteristics. Width, depth, water velocity, intra-gravel flow and sediment characteristics are all functions of discharge. The hydraulic head is considered an important indicator of the suitability of a spawning site as it influences intra-gravel water flow velocity and sediment wash-out during the redd-cutting process. Redds mainly occupy areas of marked change in hydraulic head; either in riffles at the head and tail of pools (Crisp 2000, Moir *et al.* 1998), or at lateral constrictions such as bends or large obstructions (Moir *et al*. 2002). Moir *et al*. (2002), working on salmon, suggested that depth and velocity are related in so far as the deeper the spawning site the greater the velocity must be. This relationship is described by the Froude number, which represents a single descriptive term relating depth to velocity defined, i.e. $Fr = v/\sqrt{dg}$, where Fr is the Froude number, v is the mean flow velocity, d is the flow depth and g is gravitational acceleration (9.81 m s^{-2}). Habitat utilisation was considered to fall between $Fr = 0.2$ and Fr = 0.4. It was also found that no spawning took place in water shallower than 0.15 m, although this control is considered to be due to female size (Crisp 2000).

Patterns of discharge affect the hydraulics of intra-gravel flow, transport, and deposits of fines; under certain conditions, gravel beds may move and salmonid eggs and alevins can be washed out. This is considered a major cause of egg mortality in salmon and trout. Discharge is also important when considering flood events and associated redd wash out. These events can lead to the physical damage of intra-gravel stages as well as physical shock, predation during washout and deposition in sites unsuitable for continued development (Crisp 2000).

Less is known about the incubation process in grayling. As the eggs are, on average, smaller and buried less deeply than those of salmon and trout they are probably more prone to washout and less prone to asphyxiation or entrapment by fines.

3.3.2 Fry and parr requirements

Good salmonid habitat is typified by a great diversity of physical forms within river habitat. Mixed substrate material of a coarse nature, including boulders, cobble and coarse gravel, together with a diversity of flow manifested by riffle and pool sequences are the common features of good salmonid habitat. Young-of-year (YOY) salmonids tend to stay within the area from which they hatched until a primary dispersal period in the first autumn (Egglishaw & Shackley 1977). This is the critical period for both salmon and trout and the point at which cohort size may be established (Armstrong *et al*. 2002). Survival is thus very much determined by the choice of spawning substrate made by the adult returning fish. During the first autumn, YOY fish disperse to winter habitats in deeper water and there is evidence to suggest that they switch to a coarser substratum with large home stones (Rimmer *et al.* 1984). Post-young-of-year (PYOY) salmon and trout predominantly use substratum within the cobble and boulder classes.

Hydrological conditions may be important for young salmonids, particularly during swim-up and dispersal stages. High mortality and weak year classes are correlated with high discharges during swim-up, although high discharges before swim-up and only a week after did not have the same effects (Jensen & Johnson 1999). Experimental evidence, from work on smooth and semi-natural channels, indicates that salmon parr can maintain station in higher water velocities than trout (Ottaway & Clarke 1981, Crisp 2000, Armstrong *et al*. 2002). Young salmon tend to stay close to the bottom except at low flows (<10cm s^{-1}), when they move into positions higher in the water column. The downstream dispersal rate of trout is minimal at about 25cm s^{-1}, but increases at higher and lower velocities. By contrast, salmon fry have a high dispersal rate at water velocities about 7.5cm s^{-1} and much lower rates at 25-70cm s^{-1}. Similar juvenile population densities are found at all velocities, except for salmon, which tend to be less abundant at velocities below 7.5cm s^{-1}. Changes in habitat preference also occur with increase in size, at different seasons of the year, and within and between populations. It is likely that the distribution of, and hence interaction between, juvenile salmon and trout is linked to the topography and flow characteristics of a particular reach of river. Salmon are found in higher gradient streams with more torrential flow regime than trout. Thus, altering the flow regime may affect the distribution patterns of the two species, and potentially increase interactions.

Grayling fry emerge from the gravel at a length of about 2.2cm. They occupy near-surface positions in velocities between 3 and 9 body lengths s^{-1}, until they reach 2.5 to 2.8cm, when they adopt a benthic distribution.

3.3.3 Adults and spawning movements

The habitat requirements of older trout resident in fresh water have similarities with those of younger trout, although allowance must be made for the effects of age/size in modifying the quantitative aspects of the requirements. Adult trout tend to preside more in deeper pools and around undercut banks or are associated with deeper, fast flowing glides. The needs of these fish during upstream migration to spawn are similar to those of sea trout and salmon of similar size.

There is considerable variation in the temporal patterns of upstream movement of potential spawners. It is difficult to quantify the stimuli for upstream migration. Factors that are believed to influence the readiness to move upstream include: physiological readiness of the fish to spawn; river flow; water discolouration; and water temperature.

River flow is considered an important influence on the willingness of migratory salmonids to enter a river and move upstream. Most of the available quantitative information is empirical, and may not be suitable for application outside the region in which it was developed. There are two main conceptual models to describe the influence of flow on upstream movement.

i. Adult salmonids require certain minimum (threshold) flows to be exceeded before they will move upstream, and these flows were defined as percentages of the average daily flow (ADF). For salmon, 30 to 50% of ADF is considered necessary in the lower and middle reaches of rivers (50 to 70% for large spring salmon) and >70% ADF in the headstreams. Trout require 20 to 25% ADF in the lower and middle reaches and 25 to 30% ADF further upstream.
ii. *Salmo salar* and *Salmo trutta* are only thought to move during certain parts of the hydrograph, usually the rising and falling limbs, or the falling limb only, rather than the spate peak. Upstream movement begins when flows reach approx. $0.08m^3\ s^{-1}$, peaks at $0.2m^3\ s^{-1}$, and reduces at higher flows, although differences may occur with river wdth and depth.

Sea trout seem willing to move upstream under a wider range of flow conditions than salmon.

As previously indicated, river flow is also an important parameter dictating how easily fish may pass physical obstructions (caused by dams, weirs, rapids and waterfalls). Fish usually negotiate the obstacle by leaping or swimming directly up inclined surfaces. The ability of salmonids to pass such obstacles will depend upon water velocity over the obstacle, the height of the obstacle (note the pool at the foot of the fall should have a depth of at least 1.25 times the height of the fall) and upon the swimming capabilities of the fish. Leaping ability, as swimming speed, varies with temperature.

However, what is equally critical in negotiating obstacles is that sufficient flow passes over the weir to allow the fish access. At times of low flows, whether natural or regulated, fish can be stranded below obstacles until flows exceed certain thresholds that allow the fish to free passage. Few studies have been carried out on the threshold levels that allow fish free passage (see for example Sambrook and Cowx 2000), but river keepers and bailiffs usually have good local knowledge that can be used to offset this limitation, when setting flow regimes. Further work on flow needs for adult salmon is in progress to alleviate some of the paucity of information in this area (EA R&D project W6-072: Flow protection criteria for adult salmon).

3.4 Habitat Requirements of Coarse Fishes

The preferences and requirements of cyprinid fishes (Table 3.1; Appendix 2) are best described using the functional unit concept. The species functional unit is a spatial entity leading to successful completion of the life cycle (Fig. 3.1). It encompasses the notions of home range, daily activity area, seasonal and/or spawning migrations, as well as activity, seasonal and size-related differences in habitat utilisation, all of which appear critical to the success of coarse fish populations.

In general, evidence suggests that the bottlenecks to the development of cyprinid populations, as with salmonids, relate principally to the presence of, and access to, appropriate spawning sites, to spawning success, and to the growth and survival of newly-hatched larvae. This is largely because adult cyprinids tolerate a wider range of river conditions than their 0-group progeny (Appendix 2). Consequently, an inventory of catchment spawning locations and habitats for each species at different life stages, particularly the 0-group, is imperative for assessing the impact of water resource development schemes. It should be noted that coarse fish species usually occupy the warmer lowland and middle reaches of rivers (See Section 2) and larger rivers, which has implications for the CAMS procedures because these are the regions from which many abstractions occur.

3.4.1 Adult requirements

Coarse fish species found in England and Wales have a range of preferred habitats, but most tend to be fairly plastic, and indeed opportunistic, in their habitat selection, being tolerant to most conditions (See Tables 2.1, 3.1 and Appendix 2). However, it must be recognised that the certain species may only survive, and not thrive, under sub-optimal habitat conditions. For example, barbel are able to survive and grow in still water conditions, but they are unable to breed and generally have lower condition (Taylor *et al.* 2004). Models to determine the optimal habitat characteristics of coarse fish are limited, and largely based on subjective assessment using habitat probability of use curves (see Cowx 2001) derived according to the Instream Incremental Flow Methodology described in Section 4 (e.g. Figs 3.2, 3.3). Fortunately, the models available are for several of the key species that drive the river typology developed in Section 2, i.e. roach, dace, chub, bream and pike.

UK coarse fish species use a range of spawning substrata, which can be used to classify them into a series of reproductive guilds (Table 3.2). These guilds form a useful basis on which to add more detailed, species-specific, information. UK coarse fish can be grouped into 4 categories.

Table 3.2 Habitat preferences and requirements for reproduction of common UK cyprinids. R50 refers to 50% central range of variable utilisation by species. (Data from Grandmottet 1983; Baras 1992; Lelek 1987; Brylinska 1986; Kennedy 1969; Mills 1981a,b,c; Bless 1992; Copp & Mann 1993; Holcik & Hruska 1966; Mann 1996; Cowx 2001)

Species	Depth (cm)	Water velocity (cm s^{-1})	Substratum (∅ mm)	Vegetation	Optimum temperature (°C)
Abramis brama	variable	<20	>5	*Glyceria, Sagittaria, Nuphar*	12 - 20
Barbus barbus	R_{50} = 14 - 22	R_{50} = 35 – 49	R_{50} = 20 - 50	Absent	>14
Blicca bjoerkna	Variable	<20	Indifferent	Hydrophytes Helophytes	16 - 25
Cyprinus carpio	Variable	<5	Indifferent	Submerged riparian or floodplain veg., *Carex, Glyceria, Phragmites*	>18
Gobio gobio		10 – 80	3 - 30	Hydrophytes (occasional)	>17
Leuciscus cephalus	10 - 30	20 – 50 R_{50} = 15 – 75	>5	Hydrophytes (occasional)	14 - 20
Leuciscus leuciscus	25 - 40	20 – 50	30 - 250	Hydrophytes, rootwad (occasional)	6 - 9
Phoxinus phoxinus	10 - 25	>20 R_{50} = 25 – 45	20 - 100	Absent	
Rhodeus sericeus			Unionids		
Rutilus rutilus	15 - 45	>20 R_{50} = 35 – 60	50 - 150	*Fontinalis* moss, *Elodea, Salix, Scirpus*	14 - 18
Tinca tinca	Variable	<20	Indifferent	*Myriophyllum*, submerged riparian or floodplain veg.	20 - 24

Lithophils: Eggs stick to stones and gravel. The larvae are initially photophobic. Optimum gravel sizes and river current velocities vary between species. For example, *Phoxinus phoxinus* (L.) requires 2-3 cm diameter gravel with velocities of 20-30 cm s^{-1}, but have been observed to spawn on finer gravel associated with *Ranunculus* spp. Larger lithophil species, e.g. *Barbus barbus*, spawn on a range of gravel sizes, but are also able to use substratum with larger particle sizes.

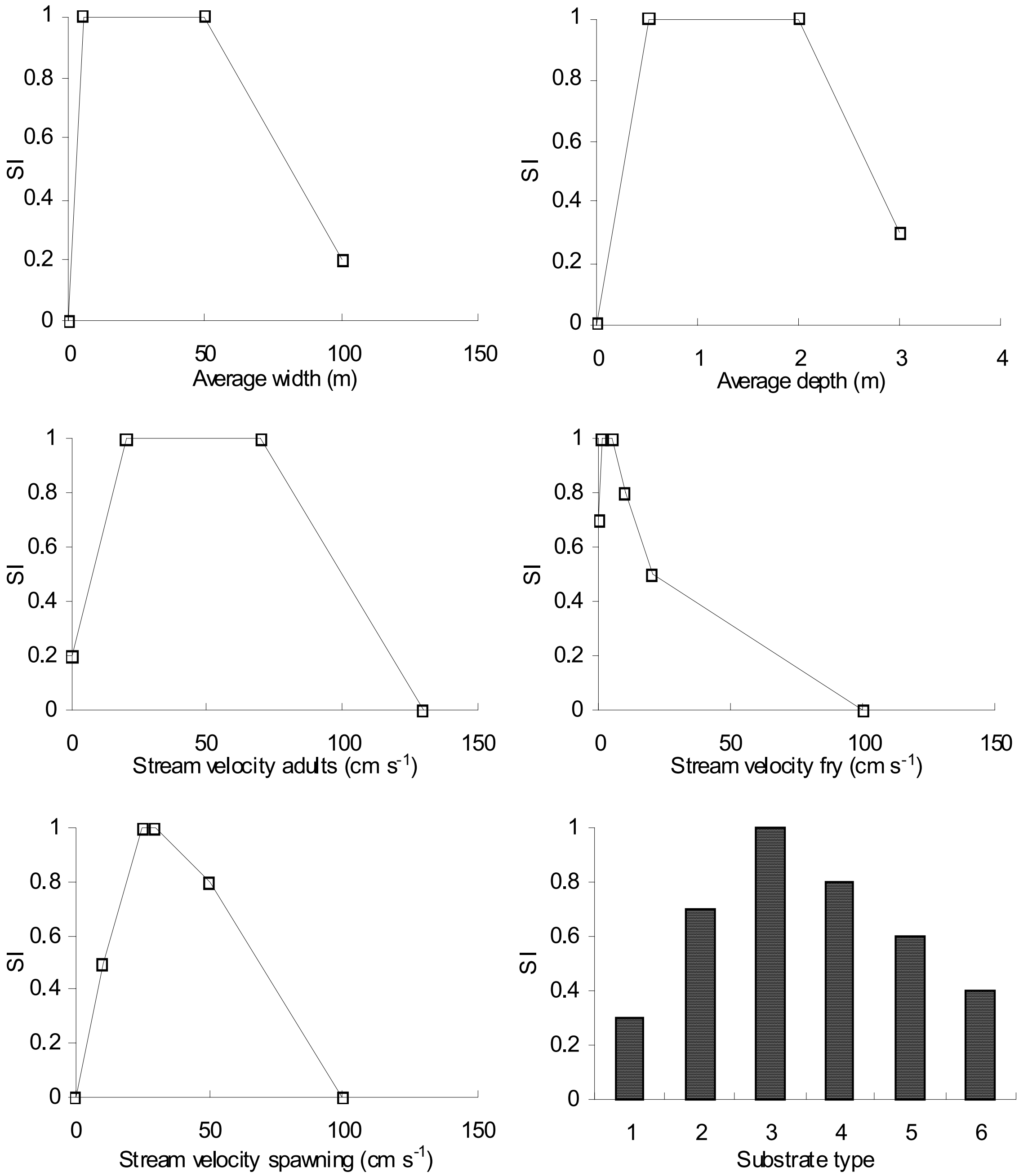


Figure 3.2 Habitat suitability curves for *Leuciscus cephalus* (after Cowx 2001). For substrate 1) - mainly clay or silt > 50%; 2) - equal amount of clay or silt and gravel; 3) mainly gravel > 50%; 4) equal amount of gravel and cobbles; 5) mainly cobbes > 50%; 6) equal amount of cobbles and boulder

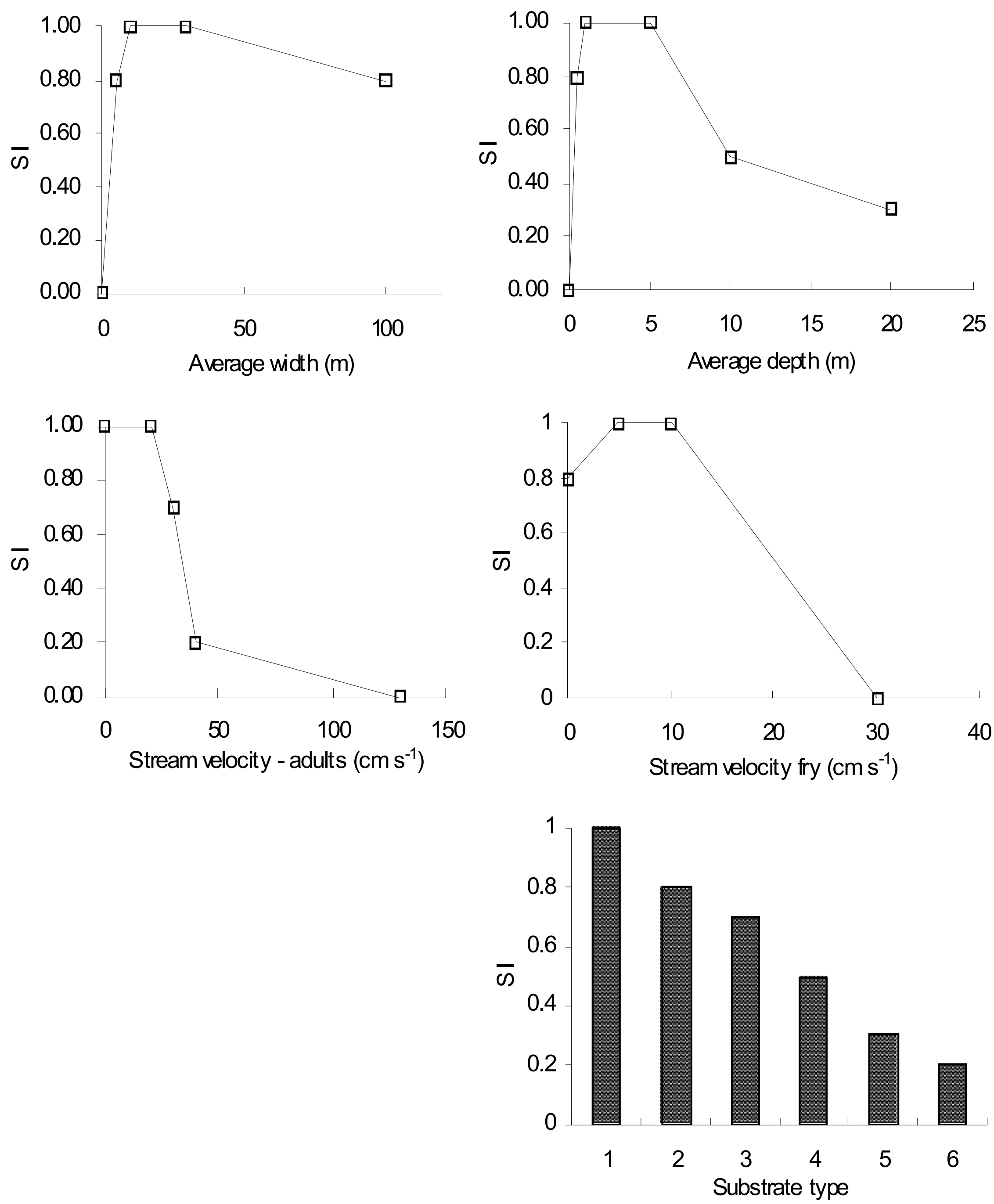


Figure 3.3 Habitat suitability curves for *Rutilus rutilus* (after Cowx 2001). For substrate 1) - mainly clay or silt > 50%; 2) - equal amount of clay or silt and gravel; 3) mainly gravel > 50%; 4) equal amount of gravel and cobbles; 5) mainly cobbes > 50%; 6) equal amount of cobbles and boulder

Phytolithophils: Eggs stick to submerged plants but other substrata are utilised in their absence. Larvae are initially photophobic. These species use a variety of substrata in a range of water velocities. *Rutilus rutilus* (L.) spawns on *Fontinalis* moss, *Elodea* beds, *Salix* roots, *Scirpus*, stones and submerged logs. They prefer spawning in flows
>20cm s^{-1} but also spawn successfully in still water habitats. Although *Abramis brama* (L.) may spawn on stones in lakes, in rivers it utilises only areas with a weak current and macrophyte substratum, e.g. *Rorippa*, *Butomus*, *Sagittaria*, *Glyceria* and *Nuphar*. *Leuciscus leuciscus* (L.) is usually included in this category but most references suggest it prefers spawning on a stony substratum with flow velocities about 30cm s^{-1}.

Phytophils: Eggs adhere to submerged macrophytes. The larvae are not photophobic. *Tinca tinca* (L.) and *Scardinius erythrophthalmus* (L.) spawn among *Myriophyllum* beds, and carp *Cyprinus carpio* L. utilise a range of plants, including *Carex*, *Glyceria*, fresh shoots of *Phragmites* and *Salix* roots. Other species include: *Blicca bjoerkna* (L.) and *Carassius carassius* (L.).

Psammophils: Eggs are laid on sand or fine roots associated with sand, washed by running water. Benthic larvae are photophobic. *Gobio gobio* (L.) have been found to lay their eggs on *Fontinalis* at velocities between 10 and 80cm s^{-1} and among plants on coarser substrata.

Recruitment success in any population is dependent on availability and quality of suitable spawning habitat. Coarse fish are known to migrate considerable distances during the spawning season to access such habitats. Thus, any major obstacle on the migration route could prevent or delay the arrival on the spawning grounds. This can have serious implications on the reproductive success of the species and can lead to a deterioration of the stocks. These issues are discussed in Section 3.3 in relation to salmonids, and the arguments are equally valid for coarse fish, despite the lack of recognition that they move large distances (Lucas & Baras 2001).

Egg mortality

Many phytophils and phytolithophils lay eggs just below the water surface where they are vulnerable to sudden falls in water level. These problems can result from the operation of sluices and navigation locks, or the over-abstraction of water during periods of low flows. The critical time period for avoiding any potential short-term (over a few days) drops (>10 cm) of water level is during the spawning and early larval development phases, which occur from early March (pike and dace) to late summer (tench, chub and carp) (Cowx 2001). Removal of instream vegetation during the spawning period can also have a dramatic affect on spawning success. Weed cutting to prevent flooding can directly remove the eggs of phytophilous fish species, but also have an affect on water levels (e.g. falls in water levels of about 60 cm over 3 days have been observed in rivers following the removal of *Ranunculus* beds), which in turn affects recruitment success by drying out substantial parts of the egg mass.

3.4.2 Habitat requirements of larvae and young of the year growth stages

Young stages represent critical periods in the life cycle of coarse fishes. Critical, sustainable and preferred velocities represent key characteristics in habitat selection and in determining the carrying capacity of the river or stream at various flow regimes. This aspect is particularly relevant to growth and survival of 0+ larvae and juveniles. Critical velocities (CV_{50} [cm s^{-1}] - values that displaced 50% of larvae after 3 min) have been determined for several UK cyprinid species at different sizes and in relation to the ambient water temperature (Table 3.3). This information, combined with the habitat preference curves (Figs 3.2 and 3.3) provide useful guides to the potential tolerance of the larval (and older age groups) stages of various species to displacement from refugia by episodic spates or the effects of abstraction and discharge. However, it should be noted that most larvae select habitats where the flow velocity is well below the critical level. For example, newly-hatched *Rutilus rutilus* and *Leuciscus leuciscus* larvae tend to be confined to riparian habitats where the flow velocity is below 2.0cm s^{-1}, although flow velocities >6cm s^{-1} are required to displaced larvae of 7.5mm (Table 3.3). For older 0-group fish, the maximum current velocity that can be sustained for at least 15 min increases to about 40cm s^{-1}. Consequently, most 0-group cyprinids will be excluded from habitats with velocities >50-60cm s^{-1} until late summer, when they achieve a length at which they are able to cope with these flows.

Table 3.3 Critical, tolerance and preferred water velocities for four UK cyprinids. CV_{50} = critical velocities displacing 50% of the larvae after 3 min. Tolerance and preference limits refer respectively to the P_{95} and P_{75} of the probability of use curves for water velocity (after Cowx and Welcomme 1998)

Species	Fish size (cm)	Velocity limits	Temperature °C
Leuciscus leuciscus	0.9 - 2.5	CV_{50} = 10.3 BL s^{-1}	15 - 16
	4.5 - 8.0	Tol - 10.08 BL s^{-1}	15 – 15
		Pref = 6.33 BL s^{-1}	
Rutilus rutilus	0.6 - 1.5	CV_{50} = 13.3 BL s^{-1}	19 – 20
	0.75	$CV_{50\text{ - }9.20}$ BL s^{-1}	
		Pref = 2.67 BL s^{-1}	
Barbus barbus	1.9 - 3.0	Tol = 11.50 BL s^{-1}	15 - 16
		Pref = 5.67 BL s^{-1}	
	3.0 - 4.5	Tol = 10.81 BL s^{-1}	15 - 16
		Pref = 5.67 BL s^{-1}	
	4.5 - 6.5	Tol = 10.71 BL s^{-1}	15 - 16
		Pref = 5.0 BL s^{-1}	
	7.5 - 12.0	Tol = 6.19 BL s^{-1}	12 – 18
		Pref = 6.57 BL s^{-1}	
Gobio gobio	2.8 - 5.0	Tol = 8.85 BL s^{-1}	15 – 16
		Pref = 5.29 BL s^{-1}	

It should be recognised that this type of analysis effectively limits the carrying capacity of the river for juvenile fish to small areas of the river where suitable flow velocities are found, and may be represented by as little as 2-3% of the river's surface area. This limitation of the actual carrying capacity by critical flows is particularly relevant in regulated, channelised and/or dredged rivers where anthropogenic activities reduce habitat diversity. In relation to this argument, one critical issue that must be recognised is the implications of reduced discharge that restricts the fish's ability occupy suitable habitat in the margins. Reduced flows may dewater these habitats and force smaller individuals into the main channel where the flow is above tolerance.

It has been suggested that water velocity plays only a minor role in determining habitat distribution beyond the critical limits of the fishes' physiological tolerance. However, it also plays a determining factor in shaping the habitat as it conditions most of the attributes of each microhabitat, such as substratum (reduction in particle size with decreasing velocity), vegetation types (e.g. predominance of *Myriophyllum* sp. in lentic habitats) and the food resources. For example, dewatering of shallow areas during abstraction means that algal and rotifer populations, critical food resources for larval stages, are lost.

In addition to river current, other environmental characteristics influence the microhabitat distribution and subsequent survival and recruitment of cyprinid larvae. Cyprinid larvae can be grouped according to their association with various habitat features, especially water depth, channel width and shape, substratum particle size, vegetation cover and type, and water temperature. The habitat preferences curves derived for various species (Figs 3.2, 3.3) help to identify these associations. However, simple observations are equally useful. For example, *Leuciscus cephalus* and *Alburnus alburnus* larvae are often found in lentic water, 20 to 50 cm deep, with a silted gravel substratum and associated macrophytes and woody debris. By contrast, *Leuciscus leuciscus* larvae avoid woody debris but prefer macrophyte and attached periphyton cover in a range of water depths. Initially *Rutilus rutilus* larvae prefer water of 50 to 100cm depth with thick macrophyte growth but, later, they move into shallower areas (20 - 50cm), often in association with *Leuciscus leuciscus* larvae. There is also a strong relationship between shoreline habitat diversity and the number of fish species (0-group). Shoreline slope and diversification are only two factors, and other larval microhabitat features, including proximity of suitable spawning sites, and connected backwaters (feeding areas and refugia during winter floods) are also important. Abstraction can influence all these essential elements of the life cycle of coarse fishes.

3.5 Swimming Performance of Fish

Swimming performance is one of the crucial factors determining the survival of most fish species and has been reviewed by Wolter and Arlinghaus (2003). Predator-prey interactions, reproductive behaviour, especially spawning migrations, habitat shifts and dispersal are of profound ecological importance and depend substantially on the individual's capacity for locomotion (Kolok 1999; Reidy *et al*. 2000). Swimming speed limits and endurance are directly related to food capture, escape from predators, ability to negotiate obstacles and the ability to avoid entrainment and water intakes etc. (Beamish 1978; Videler & Wardle 1991; Videler 1993). Until now, studies of the ecological relevance of swimming performance focused mainly on four topics: environmental influences on exercise performance (e.g. Beamish 1978; Videler 1993; Hammer 1995; Kieffer 2000; Plaut 2001; Turnpenny, et al. 1998), migration abilities (Pavlov 1989; Taylor & Foote 1991; Barbin & Krueger 1994; Zerrath 1996; Toepfer *et al*. 1999; Ellerby *et al*. 2001; Sambrook & Cowx 2000); predator-prey interactions (Howland 1974; Webb 1984b, Fuiman & Magurran 1994; Domenici & Blake 1997; van Damme & van Dooren 1999; Domenici 2001; Bergstrom 2002); and the issues associated with entrainment (Solomon 1992). However, there are substantial differences between the above mentioned topics and how flow manipulation impacts on the fish's tolerance to live in a specific reach of a river or migrate between critical habitats. On the one hand, reduction in flow caused by abstraction may force a fish into the main channel, away from the lower flow marginal areas where they will be subjected to greater physical stresses, and on the other hand, the change in flow/current velocity over obstructions could alter a fish's ability to negotiate the barrier, and thus disrupt its migration pattern.

In general, salmonids exhibit the highest swimming performance. The wide scattering of results in salmonid critical speeds (Appendix 3) are mainly related to the wide time scale between 20s and 1h (Brett 1964) for critical performance exercises. The outlier in burst performance is eel (*Anguilla anguilla*), with a particularly low burst speed, and a few rheophilic cyprinids with surprisingly high burst speeds. The latter findings resulted exclusively from the experiments performed by Zerrath (1996), who reported extremely high burst swimming speeds for small chub (*Leuciscus cephalus*) and gudgeon (*Gobio gobio*) from studies in an experimental fishway. For small fish, the highest burst performance appears to be for fish 50-60mm long.

The summary of the information on swimming performance highlights the limitations of the information in relation to setting environmental flows. **Too few studies have examined the wider environmental impacts of adjusting flows, especially the issues associated with maintaining longitudinal connectivity and facilitating passage of fish about obstructions. This is highly relevant to setting environmental flows that allow the free migration of fish during critical periods of their life cycle. It is recommended that these issues are examined and mechanisms for overcoming them are addressed in the next phase of the project**

3.6 Features of Flow Regimes/Hydrographs Having Biological Significance to Fish

There are several characteristics of river hydrographs that can affect fish and need to be taken into consideration in water management for fisheries. In some respects, this has been illustrated in Section 2 when setting river typology. However, it is deemed important that a review of the features of flow regimes / hydrographs that have biological significance to fish is needed to provide background information when making decisions about water resources management that could affect fish and fisheries.

This analysis draws on a number of publications world-wide reviewed by Welcomme & Halls (2001), Bunn & Arthington (2002), Dyson, Bergkamp & Scanlon (2003), Arthington *et al.* (in press) and Welcomme & Halls (in press). It is intended to illustrate the complex relationships between flow and fish ecology. It is recognised that, while the existing relationships between flow may appear different between the temperate zone and the tropics, this is mainly because many temperate zone rivers have been disconnected from their floodplains as a result of many centuries of river modification. Evidence from surviving floodplain rivers, such as the Rhone and the Danube, suggest that many species of fish respond to seasonal inundations in a similar manner to those inhabiting tropical systems. Any conclusions as to the importance of various aspects of the flow regime (Figure 3.4) therefore remain valid, especially with the emphasis of the EU Water Framework Directive to return rivers to good ecological condition or achieve good ecological potential in heavily modified systems.

In addition, it should be recognised that channel modification has reduced connectivity to former floodplain areas in many rivers. These are essential spawning and nursery habitats for fish, and their loss has restricted essential habitats for recruitment to main channel margins and fragmentary backwater and side channel habitats. Consequently, greater awareness is needed to protect these critical habitats when adjusting flow regimes.

It is unlikely that the different species of fish within an assemblage will respond in exactly the same fashion to flow regimes. Consequently, species assemblages may vary considerably in relative abundance and even species composition, even among similar types of rivers characterised in Section 2.4.6. Decisions surrounding optimal hydrological conditions in a particular stretch of river may therefore be subject to issues surrounding the importance of relative abundance or biomass of different species communities. To compensate for this, it may be necessary to vary the hydrological conditions from to maintain particular assemblages. **The consequence of this issue is that it may not be possible to develop generic flow requirements, but that they will have to be adjusted for specific reach/river types.**

3.6.1 Timing

The timing of changes in hydrological conditions is important to many river fish species because of the synchronisation between physiological readiness to spawn and increases in flow (floods) in the spring (coarse fish) or autumn (salmonids). Most species of river fish have defined breeding seasons centred on a particular hydrological phase (see Lucas & Baras 2001 for a detailed review of migration), and, although some of the species present in English and Welsh rivers are not commonly thought of as migratory, these same species show migratory behaviour elsewhere in Europe. Current research in the UK also suggests that many coarse fish species exhibit more extensive migratory behaviour than previous recognised (see Lucas *et al.* 2000 for example). Migratory species are especially sensitive to the timeliness of the flood because they begin their migration from their downstream feeding habitat during low flow periods and so time their migration to arrive at the upstream spawning site before, or contemporaneously with, the rising flood or high flow conditions. Such species may mature during migration or at the upstream site, postponing the last stages of maturation until the waters begin to rise. Species present in systems where the timing of flooding events is unpredictable are, however, unlikely to rely on changes to flow as a cue for final maturation and spawning. Generalisations concerning the importance of flow conditions for spawning success should therefore be avoided. **For each river type** identified in Section 2, **there is a need to identify which characteristics of the hydrograph act as cues for initiating migration or dispersal**.

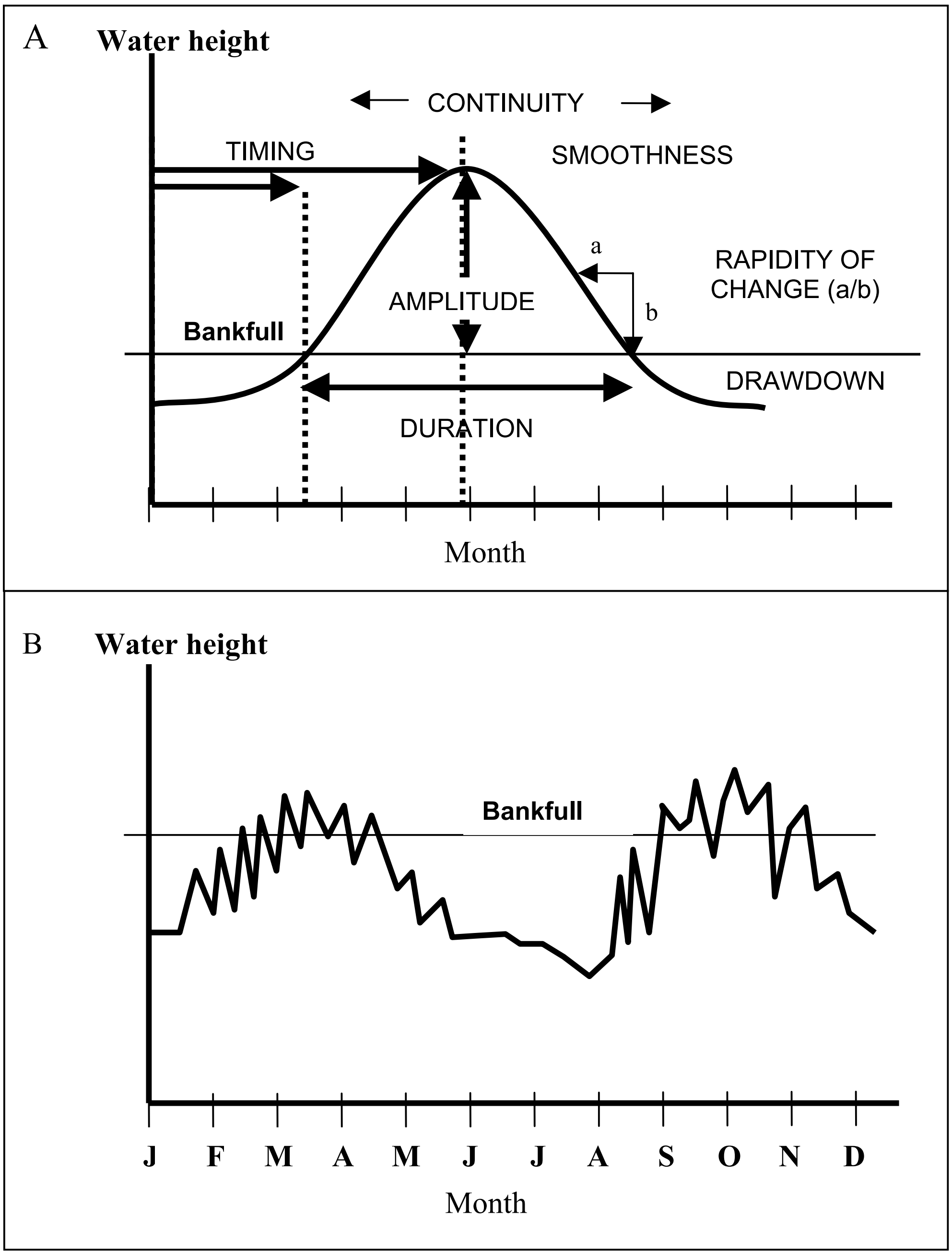


Figure 3.4 Features of hydrology (here represented as changes to water level with time) affecting the behaviour of fish populations. A. Generalised hydrograph; B. Typical temperate zone river hydrograph.

Many migratory fish species have larvae that enter the drift. In Europe these include roach, chub, bream and barbel (Carter & Reader 2000). Little is known as to the flexibility of such behaviour and its tolerance to substantial temporal displacement of the flood phase. Equally, the population dynamics of the drifting fry are little understood with respect to survival, growth and distribution under different flood regimes, although this may be elucidated under the EA Dispersal of stocked coarse fish project (R&D Project W2-086). It is possible that accelerated flows may result in the drifting juveniles being swept past their intended destination, and that flooding failure in floodplain nurseries will result in the loss of a whole year class of fish. The former case is particularly problematic in channelised rivers with reduced refuge habitat where the weak-swimming fry are able to maintain station. In such species, modified hydrographs and artificial flood regimes must fulfil two requirements. Firstly, they must be sufficient and timely enough to induce spawning migrations and behaviour up-river. Secondly, they must be extensive enough to ensure adequate inundation of the nursery floodplains downstream, or provision through off-channel nursery habitat is made available to compensate.

Timing of floods is also important for thermal and behavioural coupling. In many temperate systems a spring flood coincides with rising temperatures (thermal coupling), which favours the growth of young fish by increasing the amount of food available and the rate at which it can be metabolised. Suppression or delay of the spring flood may result either in lower spawning success or in poor growth, and low survival of the young fish due to the lower productivity in the cooler season. The degree to which this applies in English and Welsh rivers is poorly understood (although preliminary studies are being carried out by HIFI), and occupation of the floodplain or the anabranches and backwaters of the main channel in autumn and winter may be more as a refuge from high flow than a feeding and breeding migration.

3.6.2 Continuity

In natural systems, floods may be interrupted by one or more drought periods. Discontinuities are also induced in regulated systems when the primary user places demands on the water that interrupt the smooth progression of flooding. Such discontinuities may be particularly damaging to phytophils and lithophils, which may spawn during the early rises in water level but whose eggs and larvae are then unable to survive because of temporary recessions in water levels. This issue is of particular relevance to abstractors who wish to exploit the high flows following the initial surge of flood water, thus reducing water levels and exposing eggs to the air or limiting availability of refuge habitat because the flow is restricted to the main channel.

3.6.3 Rate of change

The smoothness of the flood is a measure of the steadiness of the rise and fall of the waters. It is the inverse of flashiness, which is the rapidity with which the river responds to local flood events. As smaller streams respond only to rainfall on their immediate basin they are extremely flashy. As the basin area increases the river tends to average out the rainfall over its surface and thus becomes less and less conditioned by local events. English and Welsh rivers tend to be extremely flashy with rapid rises and falls in level (Figure 3.4). This is due partly to the generally small size of their basin and the unpredictability of rainfall patterns, and partly to the historic patterns of land management, deforestation and current farming practice. This can be mitigated to a certain extent in highly regulated systems, at least in so far as low water levels are concerned.

Excessively rapid variation in level can strand attached egg masses of the marginal spawning phytophils and lithophils, resulting in the failure of that batch of spawn. This is a critical issue in Anglian Region of the EA where sluice operation on drains can reduce the water levels over very short time periods. Equally, retreating waters could expose eggs and fry to desiccation. Overly rapid changes in level can also affect fish more directly. The rapid currents associated with such transitions in water current can sweep larvae and eggs of species that deposit their eggs on river margins and species with pelagic and semi-pelagic larvae in the main channel past their appropriate destination. During falling waters an overly rapid retreat of the flood, typically associated with abstraction, increases the risks of stranding of fish in the temporary pools and channels of the floodplain and backwaters resulting in unduly high mortality.

3.6.4 Amplitude

The amplitude of the hydrograph reflects the difference between the water level at low water and the maximum level reached during high water events. In temperate rivers, where the major part of the flow is restricted to the main channel, this can have complex effects on the structure of the submerged substrates. In general, the greater the flow the more fluvial habitats are submerged. However, these may be submerged to a greater depth than is desirable or may be altered by the excessive flow, as in the case of higher vegetation being dislodged and swept away. In rivers that still overbank, the higher the flood the greater the area of floodplain or residual backwater submerged. The role of floodplains in Northern European rivers is far from clear, but evidence from many rivers shows that floodplains, backwaters and riparian pools (including gravel pits and other off-channel structures) are important to fish for shelter, spawning, feeding and fry survival. Here the greater the area flooded the greater the area available for these essential activities, and deeper (higher amplitude) floods produce greater flooded areas that can provide spawning sites, food and shelter for the fish.

The suppression of regular flooding of floodplains in English and Welsh rivers probably has had very much the same effects on fish populations as those of northern Europe, albeit with a smaller number of species. Evidence for the larger European rivers shows that the suppression of flooding of the plain by physical modification and flow control has suppressed both rheophilic and limnophilic species in favour of a small number of eurytopic species (Aarts, Van Den Brink & Nienhuis 2004). Conversely, when high floods have restored the floodplain, such as during the 1997 flooding on the Oder River, the abundance of rheophilic species increased and eurytopic species decreased with an overall increased in species diversity (Bischoff & Wolter 2001).

The role of flooding is upland catchments is less well researched. Out of bank flooding, or the lack of it, probably does not influence directly the reproductive success of headwater species (e.g. salmon or trout). Recruitment dynamics in these reaches are more likely to be influenced by the intensity of the flood flushing away egg and fry life stages, or reduced flows stranding eggs and alevins in the gravels. There is also a potential loss of productivity in low flows because of the reduced wetted area.

3.6.5 Duration

The duration of a deeper water phase influences the time fish in rivers are exposed to favourable river habitats, allowing for better survival, feeding and growth. In tropical rivers, survival, spawning success and growth are enhanced resulting in overall improvement of biomass and production in areas where the floodplain, backwaters and off-channel structures are flooded for significantly longer periods (Welcomme 1985 and 1995 for review). This has not been fully explored in England and Wales or much of Europe, where the role of the floodplain in fish biology has yet to be fully defined, although there is little doubt as to the importance of floodplains and backwaters for 0+ fish of some species (Copp *et al.* 1991; Schiemer & Zalewski 1992). Furthermore, the strong correlations between fish catch and flood intensity in the Danube indicate that similar mechanisms to those of the tropics applied to pristine European rivers.

3.6.6 Relationship of amplitude to duration

Many aspects of fish population dynamics in rivers are influenced more by the volume or area of the system rather than flow, although all three elements are intrinsically linked. The amount of water in the system effectively determines its capacity to support fish populations. It can also regulate the relative abundance of the various component species of the assemblage that is characteristic of any river reach. Models that describe the dynamics of fish populations subject to changes in the amount of water in the system described in Section 4.2.7 below can provide detailed insights into how these features of hydrology can affect fish populations. The combination of amplitude and duration of high water events is often expressed as a flood index. This is often compared with catches of fish in rivers where there is a capture fishery to show a strong correlation between the flood strength in any year and catch in subsequent years. Such relationships have not been established for English and Welsh rivers, but have been demonstrated in European rivers such as the Danube where Stankovic & Jankovic (1971), Holcik & Bastl (1977), Holcik & Kmet (1986) and Holcik (1996) all showed relationships between fish catch and hydrological regime in various stretches of the river. As catches consisted of similar species to those found in UK lowland rivers, there is little reason to suppose that the species would behave differently in England. In the UK, Nunn *et al*. (2003) have linked reduced year class strength in cyprinids to increased discharge in the summer months, presumably because of displacement of 0-group fish and, Sambrook and Cowx (2000) showed catches of salmon are intrinsically linked to discharge regime, with better catches being triggered by increased flows. This is partially linked to the migratory cues of salmonids associated with increased flows (salmon for example, do not run at low flows and are inhibited at high flows), but also to the ability of fish to negotiate natural and artificial barriers. As previously indicated, under low flows, many obstructions are impassable, and it is only when the flows over the obstruction increase that fish can penetrate further upstream. **These relationships are extremely complex and need further elucidation, but the need to define time slots when flow characteristics should be protected** (Sambrook & Cowx 2000) **is paramount**.

3.6.7 Low water levels

The period of low flows in the summer, and to a lesser degree during the winter, is a critical period for the majority of river fish species. At this time most species are confined to a diminished area in the main channels and many backwater and off-channel habitats are inaccessible or diminished in area. Variations in water level at this time can have a great impact on the extent and nature of various habitats, and can influence the amount of, and access to, spawning substratum and dry-season refugia, such as backwaters and riparian vegetation. Flow may virtually cease in the main channel and whole river reaches may become deoxygenated. The high recruitment during the spring spawning seasons has to find space in the much-reduced environment and fish may seek refuge in deep pools within the main channel. Equally, shallow, weedy water inaccessible to predators and with raised temperature ideal for growth, may form under low flows, making them ideal for juveniles. Provisions for environmental flows have to be directed at maintaining adequate water in such habitats. It is recognised, however, that heavily-engineered, channelised lowland rivers are less subjected to loss of habitat as it is only a small proportion of the littoral zone that becomes exposed as the water level drops. Nevertheless, suitable nursery habitat for many species may be limited and any loss caused by flow regulation may be critical to recruitment success to adult life stages.

Modelling of river fish populations (Welcomme & Hagborg (1977) and Halls *et al.* (2000, 2001), indicated that the dry phase is limiting to population densities in most unregulated systems, when significant density-dependent mortality acts as a sort of filter through which the population has to pass to survive into the following year. Preliminary investigations, described in Appendix 5, indicate that these density-dependent mortality processes might also be a feature of population regulation in UK coarse fish species (Figure 3.5). Experience from many rivers indicates that in those that regularly inundate their floodplains, the influence of flooded area during the flood is the most important component of the hydrological regime in determining population size. However, it is likely that in rivers where flow is confined to the main channel, the low water component is more important. It seems reasonable to assume that impacts of lowered dry season volumes would increase exponentially the lower the dry season (summer) flows. This is to some extent supported by Hall's analysis of a hypothetical UK river in which density dependent mortality is operating (Figure 3.6), although care must be taken when interpreting these results as fish populations in UK rivers do not perform as explicitly as the models suggest. For example, low flow periods associated with elevated water temperatures have been show to be responsible for strong year classes in cyprinids (Mills & Mann 1985; Nunn *et al*. 2003). Also, impounded reaches of lowland rivers often support higher abundances of fish at extreme low flows, presumably because of the good growth conditions created by warmer temperatures. Notwithstanding this conflicting information, modelling of this nature may provide an opportunity to explain some of the underlying noise in the recruitment processes operating in English and Welsh river fisheries.

3.6.8 Extreme flow events

At intervals, flood patterns can deliver extreme events that may challenge the capacity of the physical and living components of the ecosystem. Such extreme floods have tragic consequences for human populations whose occupation of the riparian zone of the river is adapted to more normal events, but are very important in channel formation. Nevertheless, living aquatic organisms can be severely affected by both abnormally high and low discharges. High discharges can wash away adult and juvenile fish, especially in rivers that have been hard engineered to contain flow in the main channel (Bischoff & Wolter 2001). Similarly, drifting eggs and larvae can be washed past suitable nurseries and lost to the population.

Extremely low flows may operate mainly on water quality. They can lead to deoxygenation of the water through natural processes, or through the failure of self-purifying mechanisms to correct human-induced eutrophication. In extreme circumstances, low flows can lead to desiccation of much of the riverbed and of an increased percentage of backwaters and off-channel structures.

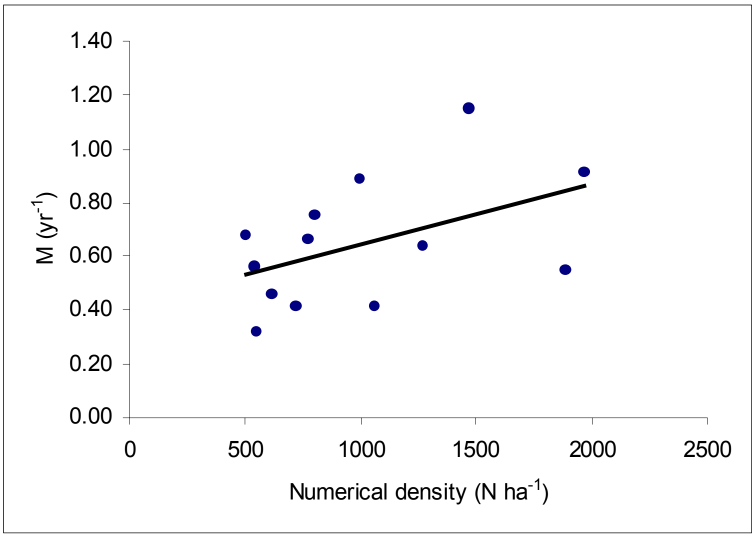


Figure 3.5 **Estimates of the annual instantaneous natural mortality rate, *M* for roach *Rutilus rutilus* plotted as a function of numerical density for UK rivers with fitted linear model: $M = 0.42 + 0.0002\delta$, where δ = Numerical density (N ha^{-1}), $r^2 = 0.23$, $P = 0.09$. Data for Holland Brook is an outlier (δ = 7200 ha^{-1}) and not included. (See Annex 5 for further details).**

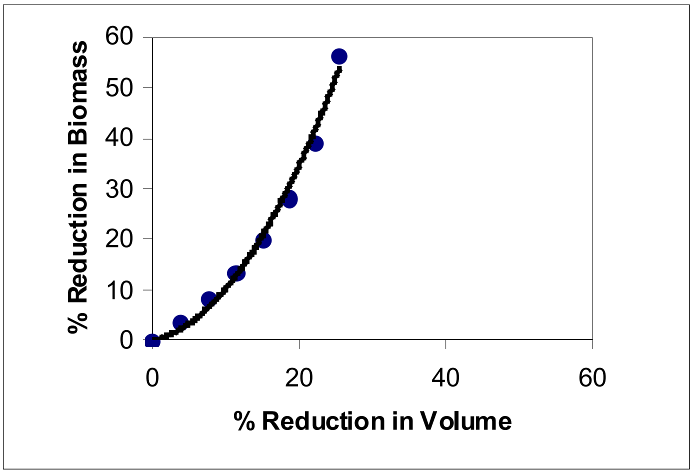


Figure 3.6 Predicted impacts of abstractions on roach biomass assuming density-independent growth and recruitment.

Although these extreme flow events are natural processes and can accrue benefits to the river ecosystem through flushing effects or elevated temperatures improving recruitment and growth potential, in the UK these events are increasing in frequency. Of particular concern is the increased frequency of summer flooding events, which are potentially detrimental to recruitment, and prolonged low winter flows which do not purge the substrate for spawning etc.

3.6.9 Flows and sedimentation and erosion processes

One aspect of flow that has indirect impact on fish and deserves mentioning is that of sedimentation and erosion processes. As already discussed, high flows can cause erosion of bed material and downstream displacement of vulnerable juvenile life stages. Continuous high flows, especially extremely rapid changes in flows, can severely impact on fish recruitment processes. However, of greater concern are the effects of low flows, which potentially, could prevail downstream of abstraction points. During natural summer low flow conditions fine sediments are often deposited in the substratum and can potentially smother aquatic macrophytes, benthic macroinvertebrates and fish eggs, or clog up interstitial spaces in coarse bed material if flushing flows are reduced or lost. Flushing flows in this context refers to flows that clean the gravels of the fine materials that deposit in the interstitial spaces during low flows and make the gravels suitable for spawning and incubation of lithophilic species, especially salmonids. Flow regulation / abstraction can potentially lead to this scenario, especially in upland reaches, resulting in impoverished fish populations. Thus, **it is essential that high flushing flows are protected, especially those occurring prior to the spawning season** (late summer, early autumn for salmonids, and May-June for coarse fishes).

3.7 Types of Flow

Stabilising river flows to an almost constant discharge throughout the year may appear more efficient than retaining a more variable hydrograph because it would avoid much of the complexity associated with variable flows and apparently would lead to more stable fish stocks. However, **the whole hydrograph must be considered as influencing fish and the alternation between dry and wet phases is probably essential to secure a healthy and variable river fish community. All species present in UK rivers evolved to deal with hydrological variation and for some, a degree of variability through the year is probably essential.**

Based on the above, and the work of authors such as Bunn and Arthington (2002), it would appear possible to recognise several types of flow based on the way they operate on fish communities. The following are proposed as an illustration:

- ***Population flows*** influence biomass of the fish community though density dependent interactions. They regulate the volume of water or flooded area in a system. They also change the accessibility of different types of habitat in the channel and access to floodplains, backwaters and off-channel structures. The main criteria are volume, depth or area of water.

- ***Trigger flows*** that cue events such as migration and reproduction. The main criteria here are timing, flow velocity.

- ***Extreme flows*** endanger fish because of excess velocity at high water or through desiccation at low water. These are expressed as extreme flows occurring as isolated peaks in an irregular hydrograph. The main criteria here are flow velocities at peak flows or at extreme low flows

- ***Habitat flows*** maintain environmental quality, including temperature, dissolved oxygen levels, sediment transport and environmental support systems such as vegetation and food organisms. The criteria of volume of water, flow velocity or flooded area in a system that operate indirectly on the fish

3.8 General Indicators of Change in Fish Assemblages

Fish assemblages respond to externally-induced stress by undergoing a series of changes in size, species composition and abundance (Table 3.5). Fishing, eutrophication, diffuse pollution, environmental degradation and alterations to the hydrological regime all tend to elicit varied responses to different perturbations (Welcomme 1995). In the majority of cases, the main changes, to which all others are linked, are a loss of biodiversity and a decline in the mean size of fish in the population. These are caused by the progressive loss of species that are not tolerant to the changing environment or the loss of larger individuals and species and their replacement by smaller ones. The latter scenario is particularly prevalent in heavily exploited fisheries because larger, K-selected species are unable to accommodate to fishing pressure and disappear from the assemblage. This is not an issue in UK river fisheries. The shift to smaller fish also involves a drift from long-lived (K selected) species to short lived (r selected) ones and is a well-known response to stress by animal communities in general (Selye 1973; Barrett, *et al.* 1976). Very often, the larger species are piscivorous predators so the process is accompanied by a decrease in the predator/prey ratio. Some species are able to maintain their place in the assemblage by reducing their breeding size and maturation time. Conversely, some fish communities respond to perturbation, in what can be interpreted as a positive manner. For example, eutrophication tends to result in a decline is species biodiversity but an increase in biomass and growth rates of those species that are able to tolerate the conditions because of better primary and secondary production. The changes indicative of stress in river fish assemblages are summarised in Table 3.5. It should be noted that many of these variables are being used under the European Union FAME project (http://fame.boku.ac.at) to determine the divergence in fish communities for the natural state cause by different anthropogenic activities.

Unfortunately, the generalised response of fish communities to a range of stresses means that it is frequently difficult to separate the impacts of one factor, such as modifications to flow, from others such as environmental modification, temperature and deterioration in water quality. This possible confusion is sometimes used to question the validity of the methodologies used for environmental flow assessment. These issues include the following.

- *Flow and fisheries*: England and Wales have no extractive fisheries, except for certain salmonids and eels, and to a lesser extent shad and lampreys. The problem of confusion between fisheries effects and flow effects does not arise in coarse fisheries, and in salmonid fisheries most estimation methods are aimed at factors dealing with younger year classes before their recruitment to the fishery.

- *Flow and water quality:* Fish assemblages respond to increased eutrophication and low grade, diffuse pollution in a similar manner to those described above and in Table 3.5. However, the water quality of most rivers has been improving steadily in recent decades and any degradation pressure on the fish has been reduced, although not eliminated. The biggest problems of water quality are probably downstream of urban centres, which are intrinsically linked to abstraction issues, although the impact of water quality can probably be discriminated from any associated with flow because the former are more acute.

- *Flow and environmental modification:* It is generally difficult to separate the responses of fish assemblages to flow modification from those resulting from channel modifications where the modification is recent or changing. However, many rivers have not been modified for some years now and hopefully data exist where any changes due to flow can be tested.

- *Flow and temperature:* There are undoubtedly temperature responses, especially in growth, that might be confused with responses to differences in flow, especially where density-dependent, population dynamics models are to be developed and used. The relationships between temperature, growth and year class strength elucidated by Mills and Mann (1985), and Nunn *et al.* (2002) do, however, give a possible mechanism for statistically separating the two factors.

As previously indicated, **the FAME project** is teasing out some of these issues but **will not provide a model for determining the relationship between stressors and impacts. This is an area of modelling that needs further research, but is imperative for the accurate assessment of the impact of water resource schemes on fish and fisheries.**

Table 3.5 Indicators of change occurring in river fish populations in response to fishing and environmental stressors

Indicator	Trend
Level of catch	**1** Falling levels of total catch levels in single species fisheries, however catch levels can be maintained over a wide range of effort and can only be interpreted relative to the catch composition. (Not relevant to UK freshwater fisheries.)
Mean length	**2** Disappearance of larger fish and falling mean size within species **3** Disappearance of larger species and falling mean size of population as a whole
Number of species	**4** Declining number of species diversity
Biomass	**5** Declining biomass in induced mortality is prevalent or reproductive processes compromised. **6** Increased mortality where primary and secondary production of water body increased, e.g. eutrophication.
Type of species	**7** Decline and disappearance of anadromous and long distance riverine migrants **8** Decline and disappearance of native species in favour of exotics where introductions have occurred **9** Decline and disappearance of higher trophic levels (predators) and their replacement by lower food chain species **10** Decline and disappearance of species with high oxygen requirements and their replacement by eurytopic species tolerant of low oxygen [eutrophication]
Population demography	**11** Reduction in the proportion of juveniles in population due to compromised recruitment processes.
Response time	**12** In rivers and river-driven lakes and reservoirs shortened time between flood events and response by population
Other indicators:	**13** Production/Biomass (P/B) ratios rise **14** Mortality rates (z and f) increase **15** Longevity of species declines **16** Drop in condition of individual fish **17** Higher incidence of diseased and deformed individuals [extreme eutrophication and pollution]

3.9 Conclusions and Recommendations

The review of habitat characteristics suggests that data exist to determine the preferred habitat characteristics of the predominant fish species found in UK fresh waters. However, there is a paucity of data for the lesser species, especially those of conservation value. **Consequently, there may be a need to improve on the information about habitat relationships of critical species that drive community structure, although, as indicated, it may be better to develop the guild concept for discriminating the impact of flow regulation on fish and fisheries.**

Survival and life history are directly related to intact migration pathways, including the possibility of migration into tributaries, side channels and backwaters that are often very important for reproduction, but also serve as rearing areas for larvae and young fish. In any decision making tool, therefore, the impact of flow regulation on maintaining connectivity, especially in the longitudinal and lateral dimensions, must be accounted for. **Too few studies have examined the wider environmental impacts of adjusting flows, especially the issues associated with maintaining longitudinal connectivity and facilitating passage of fish about obstructions. This is highly relevant to setting environmental flows that allow the free migration of fish during critical periods of their life cycle. It is recommended that these issues are examined and mechanisms for overcoming them are addressed in the next phase of the project.** Similarly little information is available on **the importance of relationships between residence times and access to side channels and backwaters from the main river channel for coarse fish species, and no information on these characteristics is available for species of conservation value. This lack of information needs addressing.**

One area of research that is imperative, and basically lacking, is the impact of various components of the hydrograph on fish population structure and abundance, as well as community dynamics. These issues are critical for assessing the role of modifying flow regimes on fisheries. However, it appears that different components of the communities behave differently to flow regulation. As these communities vary between rivers and reaches of rivers **it may not be possible to develop generic flow requirements, but that they will have to be adjusted for specific reach/river types.** As indicated, **the FAME project** is teasing out some of these issues but **will not develop a model for determining the relationship between stressors and impacts. This is an area of modelling that needs further research, but is imperative for the accurate assessment of the impact of water resource schemes on fish and fisheries.**

4 REVIEW OF TOOLS TO ASSESS THE IMPACTS OF GIVEN FLOW REGIMES ON FISH AND FISHERIES

4.1 Introduction

For the purposes of evaluating options for developing a tool for use in assessing the impacts of given flow regimes on all fish and fisheries, existing models that have been used to describe and predict these responses are reviewed. This review draws heavily on the syntheses of Tharme (2003) and by the International Water Management Institute (http://www.lk.iwmi.org/ehdb/EFM/efm.asp). The review will be used as the basis for evaluating the most suitable an approach to assess the impacts of given flow regimes on all fish and fisheries (Section 5).

4.2 Approaches to Assessment of Environmental Flows

Recently there has been much interest world-wide in evaluating the different approaches to environmental flow assessment associated with international programmes on water, food and the environment. These have resulted in syntheses such as that of Parsons, Thoms & Norris (2002), Tharme (2003) and Arthington *et al.,* (in press). Tharme identified some 207 methodologies (**E**nvironmental **F**low **M**ethodologies) that have been used or are in active use worldwide (Appendices 3 and 4). Not all of these deal solely with fishery issues and those that do are aimed principally at salmonids in small streams in North America and Europe. The extension of methodologies to other regions of the world, and other groups of fishes, has been relatively recent. Tharme (2003) listed 23 methods as having been used in the United Kingdom (Appendices 3 and 4). These have also not been limited to fish and fisheries.

Tharme (2003) classified the methods used into six main approaches based on observed or predicted responses of fish to changes in hydrology. Tharme's classification tends to be evolutionary, tracing the increasing complexity, scientific rigour and breadth of approach as methodologies emerged to deal with the increasing demands for planning information. A similar temporal approach has been adopted by Arthington *et al.* (2003) in tracing the progression from hydrological and other precautionary environmental flow assessment methods, through holistic scientific panel methodologies to detailed biological-response and ecological-response models. Tharme's (2003) classification is used here to consider the appropriateness of the various approaches to account for fisheries interests in establishing environmental flows and resolving conflicts for water resource use in England and Wales (see also Appendix 4 for tabulation).

4.2.1 Hydrological or look-up table methods

This is the simplest group of methodologies, whereby environmental flow requirements (EFRs) are primarily derived desktop studies that use hydrological data to derive environmental flow recommendations. They may include various hydrological indices, which may be modified to take simple ecological indicators into account. They generally require the setting aside of a fixed proportion of the flow, often a "minimum flow", to protect groups of fish or other selected aspects of the ecosystem, e.g. the Montana minimum and reserved flow methods. More recent approaches, such as the Range of Variability Approach (RVA), are complex, but more flexible. The outputs from hydrological EFMs are rapid and do not require many resources. However, they are relatively inflexible and of low resolution. As such they are particularly suited to general planning of water resources allocation or as preliminary flow targets. They are also best suited to the shallow pool-riffle streams and systems for which they were primarily developed, and have been mainly applied. The methods are extremely cost effective as they use data that are routinely collected for other purposes. They are, however, extremely limited when applied to fish, as there is rarely a temporal dimension to take into account the seasonal needs of fish behaviour. Furthermore, their sensitivity decreases as one proceeds downstream. Upstream, rapid and pool-riffle reaches, typically salmonid habitat, may respond adequately to such indexes as the Q_{95}, but downstream coarse fish reaches are far less sensitive to this measure as the impacts of variations in flow velocity on habitat structure (see Section 3.6) becomes less and less significant.

4.2.2 Hydraulic rating or desktop analytical methods

Hydraulic rating methods use an observed relationship between changes in flow and the extent and quality of different stream habitats available to fish. Simple hydrological variables, such as depth, velocity and discharge, are measured across selected river transects and are used to generate flow-response curves that can be used to establish EFRs. They require limited hydrological and ecological information. The hydraulic rating approach is of low flexibility, confined to one group of organisms and does not allow for negotiation of tradeoffs in flow allocations. The most widespread method in this category is the wetted perimeter method. However, hydraulic rating methods are only adequate for simple, overall planning and have been widely criticised for fisheries purposes. For example, Gippel & Stewardson (1998) commented that use of the wetted perimeter method is problematic, because of its simple approach, and should only be used in conjunction with other techniques. For this reason it has been largely superseded by the more complex habitat simulation approaches, such as PHABSIM (see Section 4.2.3). The methods can include a temporal component, in so far as they can express seasonal deficiencies in specific habitat types. However, the more complete habitat simulation models do this more adequately. Similar arguments also apply to the relative insensitivity of hydraulic rating EFMs for lowland, coarse fish rivers.

4.2.3 Habitat simulation

Methods in this category represent a development of the simpler hydraulic rating methods. They consist of two main components:

(i) A habitat model that provides more detailed instream physical habitat data and computer models that analyse the performance of river habitats under a range of hydrological conditions.
(ii) A database of species-specific habitat suitability criteria, such as responses to dissolved oxygen, temperature, presence or absence of spawning substratum.

In this way simple flow data are transformed into flow-related characteristics of the ecosystem, which can then be matched to the needs of the fish through species-specific habitat suitability criteria. On the basis of these, predictions can be made of the performance of individual species in response to changes in flow and EFRs selected on the basis of the predicted responses of individual species or communities. Habitat simulation methods require considerable amounts of data and expertise and are thus expensive and time consuming. Their data requirements also rise proportional to the area to be considered, due to the increasing number of site analyses that have to be made. Ideally, they should use habitat preferences developed in the water in question, and take account of habitat availability, proportion of carrying capacity and seasonal effects (see Section 3.4.1). This is impractical because collection and analysis of these data are time consuming and generic models of habitat preferences are usually applied. Unfortunately, few habitat preference models exist for UK freshwater fish species (see section 3.4.1), although these include many of the species that are important in driving the river typology identified in Section 2.4. They do, however, provide flexible, scenario-based outputs that can be used to negotiate water allocation among a range of water users. They can also be applied to rivers with economically-important fisheries. The most widespread methodology in this category is the IFIM/PHABSIM approach that has been successfully applied to many temperate river systems. As such the methodology will probably continue to be used in specific systems to manage particular fish communities and fisheries. Information generated from IFIM approaches can also provide an input into holistic methodologies.

PHABSIM is a model designed to calculate an index indicative of the amount of microhabitat available for different life stages of fish and invertebrates at different flow levels through a computer programme, which has two main analytical components: stream hydraulics and life stage habitat requirements.

- The stream hydraulic component predicts depths and water velocities at specific locations on a cross section of a stream. Field measurements of depth, velocity, substrate material, and cover at specific sampling points on a cross section are taken at different flows. Hydraulic measurements, such as water surface levels, are also collected during a field survey. These data are used to calibrate the hydraulic models and then predict depths and velocities at flows different from those measured. The hydraulic models have two major steps. The first is to calculate the water surface elevation for a specified flow, thus predicting the depth. The second is to simulate the velocities across the cross section. Each of these two steps can use techniques based on theory or on empirical regressions. The empirical regressions require a lot of supporting data; the theoretical approach requires much less. Most applications involve a mix of hydraulic sub-models to characterise a variety of hydraulic conditions at various simulated flows.

- The habitat component weights a series of preselected stream cells according to indices that assign a relative value of between 0 and 1 for each habitat attribute. This index indicates how suitable that attribute is for the life stage under consideration. These indices are usually termed habitat suitability indices and are developed using direct observations of the attributes used most often by a life stage, by expert opinion about what the needs are, or by a combination of the two. The hydraulic estimates of depth and velocity at different flow levels are combined with the suitability values for those attributes to weight the area of each cell at the simulated flows. The weighted values for all cells are summed -- thus the term weighted usable area (WUA).

The weighted usable areas for different values of flow are then plotted to obtain a graph which can be used to develop an idea of what life stages are impacted by a loss or gain of available habitat at what time of the year. Time series analysis plays this role, and also factors in any physical and institutional constraints on water management so that alternatives can be evaluated.

There are many variations on this basic approach tailored to different water management criteria, or for special habitat needs. However, the fundamentals of hydraulic and habitat modelling remain the same, resulting in a WUA versus discharge function. PHABSIM provides an index to the availability of microhabitat. It is not a measure of the habitat actually used by aquatic organisms. It can only be used if the preferences for depth, velocity, substrate material/cover, or other predictable microhabitat attributes under in a specific environment of competition and predation are known. The typical application of PHABSIM assumes relatively steady flow conditions such that depths and velocities are comparatively stable for the chosen time step. PHABSIM does not predict the effects of flow on channel change.

As with all methods, PHABSIM has been criticised over its accuracy and lack of verification. However, increasing experience with the method has demonstrated that it can be calibrated and verified (Dunbar 2003). As such, the methodology will probably continue to be used with increasing confidence, particularly to manage abstractions in upland and low flow systems for particular fish communities and fisheries. One of the major strengths of habitat simulation approaches, such as PHABSIM, is that they are adapted to a scenario-based approach where the administrator can select among a series of alternatives for that best meeting the requirements of both abstractor and the natural resource.

4.2.4 Holistic approaches

Holistic methods are essentially data organisation systems that rely on inputs from a range of other methodologies to identify the impacts of changes in flow on a number of river related activities including fisheries. They are the only methodologies that can extend the analysis of benefits beyond a single species, species group or fishery to explore the impacts of other user such as riparian forestry and agriculture. They are generally scenario-based exploring the impacts of changes to the whole hydrograph or to individual components of it. This they can do either through:

- (i) bottom-up methods that involve the systematic construction of a flow regime by adding components, such as incremental increases in overall flow levels, or specific high flow events; or
- (ii) top-down methods that essentially reverse this process by subtracting components of the flow regime. Here EFRs are often defined as acceptable degrees of departure from the reference flow.

The most representative of these methods are the building block methodologies developed in South Africa and Australia (Arthington, Bunn, Pusey, Blühdorn, King, Day, Tharme & O'Keeffe 1992; King and Tharme 1994; Tharme and King 1998; King and Louw 1998). These methods all rely on multi-disciplinary teams of experts who make judgement based on personal experience, or the methods listed above, to establish a consensus of opinion on the results of proposed changes in flow. They have been used mostly to assess the impacts of flow releases from dams at a catchment level. While still being used, these methods have mutated into the more structured frameworks for flow assessment discussed in Section 4.3.

Holistic methods are a fast developing tool that may well be the favoured approach in complex river systems where many interests use the resources. Best professional advice panels can also be used to resolve conflicts where the trade-offs between a number of users or approaches to EFA need to be harmonised.

4.2.5 Combination

Several approaches to environmental flow assessment use several of the above methodologies in combination. These may include partial holistic methods with poorly developed methodological frameworks.

4.2.6 Other

This heading groups together a number of disparate methods that were often not developed primarily for environmental flow assessments but which have been adapted for this purpose. Two methods show particular promise in the United Kingdom.

RIVPACS - River Invertebrate Prediction and Classification System

The RIVPAC model is based on comparisons between high quality reference sites and a site under review to produce an indication of the quality of the reviewed site. A series of high quality reference sites, usually short river reaches, are selected to encompass the full range of running water sites within the region of interest. Biological (macroinvertebrate) and environmental data have been collected at each reference site, on a number of occasions over the year (usually once in spring, summer, autumn and winter) using agreed standard protocols. The parameters for each site are assumed to represent umimpacted sites at the time of sampling, i.e. the site should be unaffected by environmental stresses. Statistical models have been developed to summarise the inter-relationships between the observed macroinvertebrate fauna at the reference sites and their environmental characteristics. This produces a prediction that is strictly validated to assess the quality of the reference sites. The final validated predictive model estimates the macroinvertebrate community to be expected at high quality sites from environmental and physical information. Measurement of these environmental features at a new site leads to a prediction of the macroinvertebrate fauna expected if the site were of high quality. A macroinvertebrate sample at the new site can be compared with the expected fauna and discrepancies between the two are used to assess the biological condition or ‘ecological status’ for that stretch of river. RIVPACS is primarily a diagnostic tool using macroinvertebrates. The principle of RIVPACS has been partially elaborated into a predictive tool for fish under the SERCON programme, and is one of the strategies being adopted under the EU FAME project. This is essentially predicting the fish community composition at high quality reference and deviation from this reference condition is being interpreted as different levels of degradation. Whilst this modelling approach shows promise, the biggest problem is finding reference conditions / sites for lowland rivers, all of which have been modified to some extent in the UK. There are also difficulties in discriminating the principal factors responsible for the degradation, as many rivers are impacted by a multiplicity of activities. It is not known at this time whether the models will have the discriminating power to predict the potential impacts of flow changes on the fish communities, especially in lowland rivers, but the models show promise. **This is one of the areas of future research that should be followed up in the next phase of the project, and should be linked to the EA River Fish Habitat Inventory Phase 3 R&D project.**

HABSCORE

HABSCORE is a visually-based, habitat assessment that evaluates 'the structure of the surrounding physical habitat that influences the quality of the water resource and the condition of the resident aquatic community' (Barbour *et al.* 1999, p5-5). It includes factors that characterise stream habitat on a micro-scale (e.g. embeddedness) and a macro-scale (e.g. channel morphology), as well as factors such as riparian and bank structure which influence the micro and macro-scale features (Barbour 1991; Barbour *et al.* 1999). HABSCORE is composed of ten habitat parameters. To reflect the difference in habitat types between upland and lowland streams, separate assessments have been developed for high and low gradient conditions (Barbour *et al.* 1999). At each site, individual parameters are assessed and rated according to a continuum of scores that represent optimal, sub-optimal, marginal or poor condition. A total score is obtained for each site, and is subsequently used to determine the percent comparability to reference conditions (Plafkin *et al.* 1989). However, the individual parameter scores and the total assessment score also provide an overall assessment of habitat condition at the sampling site. HABSCORE has been used in England for assessments associated with a number of projects, e.g. to advise on habitat quality for setting compensation release flows on the West Ridings of Yorkshire rivers. It is mainly a diagnostic tool used for river rehabilitation assessments and needs modification if it is to be used for assessment of abstraction impacts. It is also only applicable to salmonid streams since the models were not set up for coarse fish in lowland rivers.

A further methodology that is being championed by the Environment Agency is that of River Habitat Surveys (RHS). This methodology enables the characterisation of site habitats with regards to aquatic floral and faunal communities and, when linked to pressure assessment tools, enables the identification of the cause and effects of pressures on natural feature distribution. The methodology is being developed for defining bullhead habitat requirements but needs considerable work to be appropriate for other major species (Colin Bull, EA Northwest region personal communications).

4.2.7 Population dynamics modelling

Most of the above methods use flow as the primary indicator of the likely response of fish populations to variations in hydrological conditions, often through the intermediary of the environment or habitat. The models described in this section are based primarily on the reactions of the fish to changing conditions and are based upon established theories of population regulation. They have been developed, and have so far been successfully used, to predicted changes in fish biomass in tropical lowland rivers and their floodplain water bodies where hydrological variation is described in terms of flooded areas, volumes or flood indices.
The age-structured category of model (Welcomme & Hagborg 1977; Halls *et al.* 2001) can predict how various attributes of the population, such as population biomass, numbers and mean fish weight, vary with time (typically weekly) in response to

changes in flooded area and volume of the flow regime. These models are based upon established theories of population regulation, and are therefore expected to be generally applicable. Their ability to model the responses of the important attributes of populations, such as fish condition to hydrological variation, makes them particularly attractive for managing flows to meet recreational fishery objectives. The age-structured model can be used to explore either single or multi-species responses (without interaction) to changes in hydrological conditions.

Age-structured models require (sub-model) parameter estimates describing how growth, mortality and recruitment in the population respond to changes in population number and biomass and vice versa. These parameters can be estimated from time series analyses, among population comparisons, experimental manipulations and from empirical relationships derived from meta-analysis (e.g. Lorenzen and Edberg 2002). Preliminary investigations (See Appendix 5) indicate that the Environment Agency already holds a considerable amount of data that could be used estimate the parameters of these types of age-structured models.

Whilst the parameters of the biomass dynamics category of model can be easily estimated from a time series of abundance or biomass estimates and corresponding hydrological information, and can be used to model the response of a multi-species assemblage (with interaction), preliminary results indicate that this category of model tends to produce inferior fits compared with the age-structured type (See Appendix 5). The availability of data to fit this type of model should be examined.

4.3 Frameworks for Flow Analysis

The methods described above can be carried out as isolated exercises, especially where assessments at site or short reach level alone are required, but where assessments are required at catchment level, or where the fisheries interests are to be negotiated with a number of other users of the system, the methodology or methodologies chosen are increasingly used within a framework. Frameworks are primarily procedures for the collection and interpretation of knowledge gained through personal experience, directed research or the use of other assessment methods. The three main frameworks most commonly used at present are discussed below.

4.3.1 Instream Flow Incremental Methodology

Instream Flow Incremental Methodology (IFIM) is the oldest of the framework methods. It was developed in the United States and is a legal requirement in some of the States as part of the impact assessment of dams and abstractions. It is a comprehensive process for considering policy and technical aspects of any proposed intervention in the hydrological regime. The methodology consists of five phases.

- *Problem identification:* This involves an analysis of the legal and institutional frame to identify key players, and an analysis of the physical location and geographical extent of any changes that may occur in the system.

- *Planning:* This involves consultations to identify the information needed, the information that already exists and what new information must be obtained. This phase should result in a written work plan

- *Study implementation*: This phase involves collection of data on a range of parameters, which may include temperature, pH, dissolved oxygen, biological parameters, and measures of flow and morphological parameters, such as depth, cover and substrate type. These variables are used to establish the relation between stream flow and stream habitat, and should establish a habitat-time series, which estimates how much habitat would be available for each life stage of each species over time. It provides estimates of the relationship between flow and total habitat derived from models such as PHABSIM.

- *Alternatives analysis*: Stakeholders compare alternative impacts from different flow regimes on the basis of effectiveness, physical feasibility, risk and economics to derive a set of alternative management scenarios.

- *Problem resolution*: This involves choosing between the alternatives in the light of the information on their impact. Attempts should be made to reconcile the interests of the various parties, but this is made difficult because the biological and economic values are difficult to interpret, the data and models are never complete, and the future is uncertain.

IFIM can be used as a scenario-based approach that enables negotiation among various users of the water, but is less suitable for setting flow requirements to meet ecological objectives. The comprehensive nature of IFIM makes it very data hungry and, because of the wide range of issues it covers, it is open to criticism. It is too cumbersome to be used for simpler assessments at site or reach level, and is mainly applicable to catchment level assessments.

4.3.2 DRIFT

DRIFT has been developed mainly to assess impacts of flow reductions from abstractions and dam construction in South Africa and Australia. It consists of four modules (King, Brown and Sabet 2003).

- *Biophysical module:* The river ecosystem is described and predictive capacity developed on how it would change with flow changes.

- *User module:* This module described interactions between stakeholders and river conditions to develop predictive capacity of how river changes would impact the various economic and social activities within the basin.

- *Scenario building:* In the third module, data are collected at selected river sites, each of which is representative of a river reach. These are entered into a custom-built database. Long-term daily flow data for each site are separated into ten flow categories and specialists predict the consequences of up to four levels of change from present condition in each flow category for different components of the river ecosystem. On the basis of these, scenarios are built of potential future flows, and of the predicted impacts of these on the river and the riparian people and economy.

- *Compensation-economics:* This module lists compensation and mitigation costs (King *et al*. 2003).

Although all four modules have been applied to rivers in the subsistence economies of Africa (e.g. King *et al*. 1999; Sabet *et al*. 2002), the first and third modules can be applied alone (e.g. King *et al*. 1999; Brown *et al.* 2000) and are the most developed.

DRIFT is a framework for generating scenarios of impacts of changes to the hydrological regime in rivers. It uses a multi-disciplinary team of specialists in a number of disciplines who build up a picture of predicted change to any presented flow manipulation, starting with channel changes, then water quality and temperature, then vegetation, invertebrates and fish. Each consequence is assigned a Severity Rating, which indicates: (1) if the sub-component is expected to *increase* or *decrease* in abundance, magnitude or size; and (2) the *severity* of that increase/decrease, on a scale of 0 (no measurable change) to 5 (very large change). The scale accommodates some uncertainty, as each rating encompasses a range in percentage gain or loss. The DRIFT procedure has been developed and applied at the catchment level, and provides a scenario based consultative mechanism whereby the interests of many users of the aquatic system can be reconciled. It could be adapted for the UK situation, especially where comprehensive and holistic management of rivers is desired.

As with all expert opinion systems the success of the process depends on the availability and performance of the participating specialists.

4.3.3 The Resource Assessment and Management (RAM) methodology

Resource Assessment and Management (RAM) is an objective-based method developed by the Environment Agency to provide a consistent technical approach to water resource assessment and management at the catchment scale in England and Wales. It is a major component of Catchment Abstraction Management Strategies (CAMS). According to the Environment Agency (2002), the RAM process consists of a series of steps to determine the amount of flow available for abstraction from a river taking account of the sensitivity of a selection of environmental indicators.

- *Characterisation of the river*: The river to be studied is characterised with respect to the sensitivity of the riverine ecology to variations in flow that may be caused by abstraction. The sensitivity of a particular reach of the river is assessed on the basis of its physical characteristics, the dominant fish populations, macrophytes and macro-invertebrates, all of which combine to produce an **Environmental Weighting**. Five Environmental Weighting Bands are used to classify the sensitivity of each river reach to the effects of abstraction impacts – headwaters and the upper reaches withing catchments being the most sensitive.

- *Setting of an ecological river flow objective*: The Environmental Weighting is used with long term natural flow duration data to derive an **Ecological River Flow Objective** and the portion available for abstraction. This may be modified by other in-river flow needs to define the River Flow Objective or flow regime which the Agency is seeking to manage. The River Flow Objective seeks to protect low flows and flow variability by allowing percentages of flow bands to be available for abstraction. The flow bands are derived from long term natural flow duration statistics and the percentage of each band available for abstraction varies according to the Environmental Weighting Band for the river reach. Artificial impacts on river flows upstream of the assessment point, due to both surface water and groundwater abstractions and discharges, are then assessed. These impacts can be calculated for fully licensed volumes and also for recent actual abstraction and discharge rates. Where appropriate, hydrological and water resource or groundwater models may be used for such calculations.

- *Setting of abstraction status*: Flow duration curves that incorporate these impacts are then compared with a River Flow Objective flow duration curve. This indicates whether the river resource status is Water Available (for additional licensing) or No Water Available, or the degree to which resources are already Over Licensed or Over Abstracted.

- *Abstraction licensing and resource management:* The resource assessment is used in conjunction with the CAMS Sustainability Appraisal process, which incorporates stakeholder consultation, and existing Water Resource Strategies to develop a catchment scale Abstraction Licensing Strategy. Once finalised, the Licensing Strategy guides the management of water resources over the CAMS six-year cycle. Should the RAM Framework indicate that resources are over abstracted, further more detailed studies would be required prior to embarking upon any restorative actions.

The default Environmental Weighting (EW) system uses information on four 'ecological' indicators: fish, macro-invertebrates, macrophytes and the physical characteristics of the river reach. It is based principally on the sensitivity of the ecological indicators to changes in river flow. Flow is considered as a simple proxy for a number of related parameters that may have a key influence on habitat (e.g. water depth, flow velocity, wetted perimeter). The EW system provides a default approach which is considered widely applicable to most CAMS at the sub-catchment (>50km^2) and river reach (10 to 30km^2) scales. It can be overridden if better information from previous assessments is available and does not replace the need for more detailed assessment of causal hydro-ecological relationships at a local scale, which will still be required for some abstraction licence determinations.

4.4 Issues

If a tool is to be developed to assess the impacts of given flow regimes on fish and fisheries certain issues need to be explored and clarified. These issues are applicable to both specific and general applications when considering procedures for Environmental Flow Assessment.

4.4.1 Sustainability state

The objective of the sustainability of ecosystem integrity is repeatedly applied to the criteria for assessment. A river may exist in many "sustainable" states depending on the degree of modification and the amount of energy expended in fixing its physical form. Within this context, there is no definition of what is considered the desirable sustainable condition. This is particularly problematic in European lowland rivers that have been systematically modified over a considerable period of time, and in which the benchmark faunas are those that have been selected for by centuries of river training and flow modification. This issue is one of the problems being faced in setting reference conditions of good ecological status under procedures linked to the WFD. It is difficult, if not impossible, to find reaches of rivers, especially in large lowland rivers, that have not been heavily modified, to establish the benchmark conditions. In these cases, there is a need to ensure that the rivers achieve their best ecological potential, which essentially requires cognisance be given to the impact of flow-related changes on the biota. Realistically, **achieving pristine or natural state is unachievable; thus sustainable state must relate to best ecological potential. Defining this condition is a major challenge that needs to be addressed both for the current project and the WFD**.

Within this context, there is an underlying assumption that changes in flow due to abstractions will always be detrimental to the environment and this leads to a mind-set whereby the systematic use of improved flow to rehabilitate and enhance rivers over their benchmark is neglected. The exclusion of flow augmentation in rivers receiving transfer waters or groundwater pumped waters, for example, needs to be considered in the evaluation. Such flow augmentation may be detrimental to fish communities in receiving rivers but may also confer a benefit, in which case the gains in the receiving river may offset any deterioration in the donor river. Such a scenario was identified in the Exe - Taw water transfer scheme, where the River Taw fisheries benefited from increased flows during drought conditions, but no identifiable effect was evident in the Exe catchment (I. Cowx, personal observation).

4.4.2 Environmental objectives

The objectives for the assessment and application of environmental flows have been described as: "determination of how much of the original flow of a river needs to be sustained to maintain specific, valued features of its ecosystem?" However, in this age of extensive abstractions and transfers of water it is now difficult to determine the natural, unperturbed condition of most rivers. The objective is frequently redefined as "to ensure that adequate water is maintained in a river system at all times to protect the species of interest for fisheries or for conservation and the environment on which they depend". The objective identified for RAM - a level for abstraction that does not compromise the capacity of the river to support a sustainable ecology - is similar to this, but begs the fundamental question of why the river is managed, and for what purpose. A river may exist in several sustainable states not all of which are desirable. Equally, different objectives may have very different requirements for the type of hydrological regime chosen. Failure to define clearly the objectives at the outset may result in the wrong choice of environmental flow criteria and flow regimes.

In the case of fisheries, the basic criteria may be conservation-orientated, such as the simple presence or absence of species that are typical of particular river types, to maximise fish biodiversity or to conserve a specific, rare species. They may be economic, such as to maximise biomass of one or more species in support of fisheries, or the need to maximise condition of individual fish rather than produce a dense population of smaller individuals. However, perhaps the most over-riding objective is to comply with 'good ecological status' under the Water Framework Directive. By the year 2015, the UK, like all other EU States, will have to ensure that all rivers, except those that have been heavily modified, achieve this status. As fish are one of the assessment 'quality elements' for designation of status, the importance of protecting fish populations and communities is paramount, and must be at the forefront of decision-making about use of water resources.

Similar arguments apply to other components of the ecosystem. Here a choice has to be made as to whether all ecological objectives have equal rating or should be directed to the support of one or other component of the system. In the case of vegetation, for example, it would be perfectly sustainable for rivers to revert to a heavily vegetated state in the summer but this would not necessarily be compatible with the presence of certain fish species.

4.4.3 The species concept

It is questionable whether the species level is the best biotic unit in measuring responses to flow. It seems that many species have a wide range of adaptive responses to flow regimes that enable the species as a whole to survive sub-optimal conditions as well as profit from favourable ones (Section 3). To overcome this problem, **the guild concept of fish categorisation may be more appropriate for defining responses, as has been suggested for the WFD assessment methodology under the FAME project. This is an issue that needs exploring in the next phase of the project.**

One other fundamental issue that needs to be explored here is that the species' adaptability to survive in sub-optimal conditions is not necessarily acceptable from a conservation / welfare perspective. These fishes may shift their life history strategies to survive the altered environmental conditions, but may be more liable to expiration if additional stressors are introduced at a latter time.

4.4.4 Relationships between biotic components of RAM

There seems to be no mechanism within the environmental weighting component of RAM for considering the potential ecological interactions and responses resulting from seeking optimal flow criteria for discrete elements or components of the river ecosystem (i.e. physical characteristics, fish, macrophytes & macroinvertebrates), each of which contributes an independent score and, presumably, an independent flow objective. However, fish communities are extremely sensitive to the form of the environment, the vegetation present and available food resources (linked to the benthic macroinvertebrate element). Since the independent flow objectives for each element may differ, but equal weighting appears to be assigned to each of the four criteria, these may conflict. In this respect it would be preferable **to establish an integrated biotic score through some holistic methodology than an average score based on four largely independent criteria.** The integration of the various components of RAM to produce a single integrated statistic for input into the spreadsheets would require some process for evaluating the relative importance of, and interrelationships between, the macrophyte, macro-invertebrate, fish and physical habitat estimators. This process would best be carried out using an expert opinion system that groups representatives of the four components. It would give the advantage that any weighting between the components could be adjusted according to the non-flow objectives for the river sector in question (boating, fisheries, general wildlife conservation etc.). [**Assuming that one of the approaches to be developed and tested will be based on the expert-opinion strategy, this is a major component of the next phase of the project.**

4.4.5 The use of flow as a criterion for fisheries

The basic concept that the environmental weighting is based on changes in flow velocity is of concern when applied to fish. The ecological integrity of fish assemblages in rivers is indeed tied to flow, but this most often operates through secondary factors, such as flooded area, volume of water in the system, longitudinal and lateral connectivity, and the subsequent effects on modifying habitat (see Section 3. and 3.6). The most drastic example of such a divergence occurs in rivers that have regular overbank flows, as the flow velocity does not increase after the overbank event, but the resulting changes in the morphology of the system produce drastic changes in the fish communities. In general, the further downstream one progresses the less the integrity of the fish community is a direct correlate of flow, and in lowland rivers, where flow velocity may become minimal, this criterion alone may become insignificant.

There is an implicit assumption in the way flow criteria are calculated, such that the low flow period is often considered the most critical, and that if fish can survive this, what happens during the higher flow periods may not be significant. Modelling the population dynamics of some river fish, especially for salmonids, shows this view to be justified to a certain extent, in that suitable high flows can compensate for deficits created during low flows. However, in lowland rivers that do not regularly inundate the floodplain, the low flow events are probably dominant in structuring the fish communities. Nevertheless, the hydrograph has to be considered as a whole not just as a long-term, flow duration curve. This is because recruitment processes in most UK species are dependent on diversity of flows to condition the habitat (e.g. scouring gravels for spawning salmonids) and potential losses, especially of juvenile life stages, during high flood events, the latter of which we know little about. Most environmental flow assessment methods concentrate on low flow while high flow events, for example in recipient rivers from water transfer schemes, may also affect fish. This is not just a question of increased vulnerability of certain life stages being washed away in low-recurrence flood events, but intermediate flows can also influence the dynamics of fish populations.

4.4.6 The nature of the flow – fish relationship

Coupling of flows to fish ecology: Many fish species are sensitive to the temporal coupling of flows to some biological event. The most prevalent of these is the coupling of peak spring flood flows in many rivers to reproductive cycles of coarse fish species (Section 3.4). Equally many fish use the timely arrival of certain flows as a cue for maturation and spawning (Sections 3.3.and 3.4). Where peak flows have become uncoupled from the biological cycle of the fish because of damming and water abstractions, some fish species have declined in abundance and the fish community has changed to include more generalist species. This is particularly true of migratory salmonid species in the UK, but the extent to which coarse fish species are affected in this way is not clear and needs further study.

The need for temporal variation in flow patterns: There is a general assumption that all fish species characteristic of a particular class of river reach respond to flow in exactly the same way (exemplified by the idea of a single indicator species). It likely that the species present have somewhat different flow requirements and are able to maintain their populations because there is significant year-to-year variation that allows fish to survive but influences year-to-year variations in relative abundance. It may be necessary to build the concept of year-to-year variability of low flow regimes into the criteria for fish. To this end, the work on factors influencing year class strength currently being undertaken in the UK (EA FARRCoF project) may provide significant insight to the drivers of recruitment success. **These parameters, of which discharge at critical periods appears to be a main one, should be thoroughly researched and evaluated.**

The influence of the environment: The form and function of the river system can also influence the choice of environmental flow. For example, the fish in a river that regularly overflows its banks to inundate riparian wetlands will behave differently from those in a river that is confined to a highly regulated and featureless channel. Factors influencing this difference are based on the higher degree of shelter from excessive flows and the greater in-channel diversity that is afforded by the unmodified environments. Furthermore, the hydrograph of less modified systems tends to be buffered against excessive flashiness by the limitations placed on flow velocities by overbanking and by the storage capacity of the floodplain, backwaters and off-channel structures. Considerations of the degree of channel modification, including elements of channel complexity, such as wetland habitats and braided river channels, must be accounted for in any assessment criteria, because the habitats these represent will likely be critical for the maintenance of the fish populations. The assessment criteria must account for connectivity with the main river channel with these habitats is maintained if flow regimes are modified.

The influence of management on fish stocks: Fish populations in many UK rivers have been influenced by stocking for decades. There have also been localised attempts to remove or eradicate unwanted species. These efforts have been aimed at weighting the fish population towards species combinations that are seen as desirable for recreational fisheries, although experience and recent research suggest thatr these effects are often short-lived. This means that some of the fisheries surveyed over the past few years may be distorted with respect to relative abundance and overall biomass by management practices, and may not correspond to those that would be present in un-enhanced situations. Therefore, knowledge of stocking history is essential to interpret correctly presence and absence and relative abundance data, especially when trying to establish base-line conditions or to apply RIVPACS-type procedures. Likewise, any estimates of the impacts of water abstractions in rivers should also consider future stocking programmes and other management measures.

4.5 Conclusions and Recommendations

From the review in Sections 3, 4.2 and 4.3, two possible scenarios for further development seem appropriate.

A A process to arrive at the fish requirements is pursued within the existing framework and objectives of CAMS/RAM: If this scenario is adopted the reduction of flow management for fisheries to a simple 5-grade sensitivity score to water abstractions is missing a huge opportunity to improve the status of UK river fish. However, if the future mechanism is conditioned by the need to manage water abstraction, it may be difficult to establish a rational system of evaluation. In this case a relatively simple method of assigning the sensitivity index could apply, such as one of the hydraulic rating, habitat simulation or HABSCORE methods.

B If the intention is to manage the hydrological regime in the interest of fish and fisheries, and use the approach to encompass assessment of all types of water resource schemes, it is likely that a more sophisticated simulation modelling approach is required. This would overwrite the current environmental weighting default tool within the existing RAM methodology. Here three possible approaches seem to be desirable, the choice of which will depend on the scenario / scheme that is being assessed.

1) An improved methodology to integrate the environmental weighting scores for the various biotic resources to account for the inter-relationships between components and the relative importance of each to each other. This would require some process for evaluating the relative importance of, and interrelationships between, the macrophyte, macro-invertebrate, fish and physical habitat estimators. This process would probably be carried out by some best professional advice system that groups representatives of the four components. It would give the advantage that any weighting between the components could be adjusted according to the non-flow objectives for the river sector in question (e.g. boating, fisheries, general wildlife conservation), which are integrated into the assessment through stakeholder involvement and consultation. This is particularly relevant, as social and ecological issues have often been ignored in the past because of problems defining their importance in the overall appraisal.

2) The development of a RIVPACS-type model for fish, building on the information given in Sections 2 and 3. As it stands RIVPACS is mainly a diagnostic tool rather than a predictive one, but with sufficient information, comparison between rivers with differing conditions and with experience over time, this type of model can be given predictive capacity. This model can either be based on species presence or absence, relative species abundance or incorporate those parameters of the fish such as biomass, condition, and growth and survival rates that are needed to manage the fishery. This tool could be used as an assessment classification tool for use within catchment appraisal as well as within localised studies. It should also provide a generic tool for assessment of all types of water resource development schemes. It is likely that the derivations of the EU FAME project may be used to meet this need.

3) The development of population dynamics models that will assist in predicting the effects on the quality and quantity of the fish population of various alternative hydrological regimes. A population dynamics-based model of responses of fish populations to different flow regimes should also be able to examine the impacts of injection of new individuals of selected species through stocking and transfers of individuals by water transfer schemes.

These scenarios are developed further in Section 5.

5 TOWARDS A SUITE OF TOOLS FOR ASSESSING THE IMPACT OF GIVEN FLOW REGIMES ON FISH AND FISHERIES IN ENGLISH AND WELSH RIVERS

5.1 Introduction

One of the objectives of this project is to evaluate options for developing an approach for use in assessing the impacts of given flow regimes on all fish and fisheries. This process has been undertaken in the preceding chapters. This information is used in this chapter to make recommendations on the most suitable approach for developing a tool that could be used to assess the impacts of given flow regimes for freshwater fish species under data-poor and data-rich conditions. The following chapter attempts to synthesise the information and propose the way forward in developing an assessment tool. The proposals are then taken forward to develop a project submission for further research to meet the Agency's requirements to assess the impact of modifying given flow regimes or water resource schemes on fish and fisheries in English and Welsh rivers.

5.2 Criteria for a Tool to Assess the Impacts of Given Flow Regimes on Fish and Fisheries

Any tool developed for the assessment of impacts of given flow regimes on fish should:

- adequately incorporate the complexity of ecological requirements of all life stages of fish in rivers;
- be capable of being used independently or within the procedures of CAMS – RAM as set out in the Resource Assessment and Management Framework: Report and User Manual;
- be easy to use;
- be cost effective;
- be compatible with expertise available;
- be easy to understand by decision makers;
- be legally robust; and
- be generally accepted by all levels of fisheries and water user stakeholders.

All the methods reviewed previously have inadequacies based on these criteria, and all seem to suffer similar problems of verification. This considered, **it is recommended to develop an existing methodology or methodologies, building on one or more existing tools rather than try to develop a new one(s)**.

5.3 Origin of Data

English and Welsh rivers are characterised by having few abstractive, commercial fisheries for food, excepting certain salmonid species and eels. This means that there are no large fisheries to provide data on catches as a cost-effective sampling tool, although recreational creel surveys may provide some information. Consequently, information on which to base the tool will have to come from routine monitoring or dedicated fisheries surveys compared with historical data sets.

5.4 Choice of Methods

The various methods described in section 4 have different applications and these are summarised in Table 5.1 adapted from Dyson *et al.* (2003).

Table 5.1 Applicability of different methods for assessing the abstraction of water on the aquatic environment for fish and fisheries
* Supplementary method used to extend estimates beyond area of original assessment. e.g. for assessing the impacts of site or reach level abstractions on the river downstream.

Method	Hydrological or Look-up table methods	Hydraulic Rating or desk top analysis methods	Habitat Simulation or functional analysis methods	Visual scoring (HABSCORE)	Holistic approaches	Frameworks
Scoping study or national audit	X					
Basin-scale planning	X	X				
Impact assessment	*					
Site			X	X		
Reach			X	X	X	
Catchment					X	X
River restoration	*					
Site			X			
Reach			X		X	
Catchment					X	X
Multi-sectoral planning	*					X

5.4.1 Site, reach or catchment

The appropriate tools for the assessment of impacts of changes to flow regimes resulting from abstractions and discharges may vary according to the scale of the area to be studied. Cost, manpower requirements and time requirements will all increase as the size and complexity of the study area increases. Conversely, the potential accuracy of the result will decrease with increasing area. However, assessments of limited geographic scope in river basins frequently do not take into account the downstream impacts of an abstraction. Thus, although the impacts at a site may be well established and an abstraction authorised, the validity of such a limited assessment may be called into question, as the effects of the abstraction continue to be felt throughout the rest of the basin in a manner proportional to the percentage contribution of the reach to the total discharge of the system. This means that the impacts of the abstraction will persist, albeit in an increasingly attenuated form, with progression downstream. The concept of hydraulic reach may be useful here, whereby any reach of a river (and a site as a point on that reach) will have a similar hydrograph until a tributary or other form of discharge downstream significantly augments the flow. Normally this would be a river of the same order as the one affected, and would equate the reach to the order length of the stream in question. Because of the attenuation effect, where assessment is required at site or reach level, it may suffice to use some lower order methodology to assess the impacts of this on the river downstream.

5.4.2 Objective-based or scenario-based method?

Objective-based methods set a definite target for abstraction and ecological factors. Any trade-offs that there might be are inherent in the analysis of the data. This has the advantage of presenting a single figure to river basin planners, abstractors and administrators that is readily interpreted for issuing the licence. Furthermore, a standardised objective method applied to several waterways can produce a standard solution that is not vulnerable to attack on the basis of inconsistency. It does, however, rob decision-makers of the opportunity to negotiate among users to achieve the best compromise outcome for all stakeholders. Ideally, there should be a clear threshold between flow and environmental status for objective-based methods to work satisfactorily (Stalnaker 1990; Beecher 1990), but there is insufficient evidence that such thresholds exist. Indeed Dunbar & Acreman (2001) reflected the view that such thresholds are a management concept that has little or no scientific basis.

Scenario-based methods aim to present a range of options to planners by which solutions can be negotiated that satisfy all stakeholders. Because the results of the negotiations in different rivers can result in different solutions being adopted, Dunbar & Acreman (2001) stated that scenario-based approaches have been criticised, particularly in that they can lead to inconsistencies across England and Wales with regard to the granting of licences, with resulting allegations of inequalities between different regions, abstractors and lobby groups. On the other hand, the flexibility of the scenario-based approach and its ability to generate a range of solutions confers an advantage in that it may satisfy the differing, economic, social and biological characteristics of a particular river.

The ultimate choice is between an objective-based method that produces a supposedly standard result, and the greater flexibility of a system like DRIFT that encourages greater stakeholder participation and satisfies multi-sectoral concerns. The CAMS procedure sits across both scenarios, but it is great emphasis towards the latter that is probably appropriate. The procedure by which this is achieved is, however, what needs to be developed, and this is discussed below

5.5 Recommendations for Assessment Methods

5.5.1 Site and reach scale

The most appropriate tool for analysis for lower order upland rivers and probably smaller lowland streams, where the impacts of the abstractions on the fish community are judged to be significant to severe, is a Habitat Simulation or functional analysis method. The reasons for this are: that some of the methods are well known; there is increasing experience with them worldwide, and in Europe knowledge of their limitations is developing; a range of subsidiary approaches can adapt them to local conditions and make them more precise, and habitat suitability indices are available for some of the indicator species, although not all the species, especially those of high conservation value (e.g. lampreys and spined loach). The Agency has also published a manual on the use of one of them (PHABSIM) (Elliott, Johnson, Sekulin, Dunbar & Acreman 1996a, b), which, although out of date, could readily be updated to incorporate more recent thinking. The major criticism of these methods - that little correlation has been shown between the habitat conditions and fish abundance - now appears to be less critical as greater experience is gained with this approach and some authors now appear to think the method is able to be calibrated and verified in some systems (Dunbar 2003).

It is recognized that Habitat Simulation methods are costly and time consuming, but in areas where significant impacts of river fish are anticipated there are probably no financial shortcuts at present. However, for abstractions from smaller rivers where lesser impacts are anticipated, HABSCORE could provide a suitable, more cost-effective, alternative. Further research on ways in which this method can be applied to large systems is desirable.

A model based on the RIVPACS approach could be developed as an alternative or supplementary method, although its eventual use as a decision-making matrix will depend on the development of its predictive capacity through the compilation of a database from the interpretation of existing data and possibly on further research. Such a matrix could be standardised for various types of river reach (as indicated in Section 2 but taking into account the modification status) using the following criteria (indicators):

- absolute species composition (presence and absence);
- relative species composition;
- overall biomass;
- demographic structure;
- mean length at age;
- longevity;
- condition (condition factor and signs of disease or stress).

Assessments should, however, not be confined to the site or reach but should continue downstream using a simpler technology such as an hydraulic-rating method to indicate if impacts are likely to be severe enough to warrant an extension of the habitat-simulation approach further downstream.

There is no technology to assess accurately the impacts of abstractions on large, modified lowland rivers at present. Approaches exist to assess impacts of flow differences in the floodplain rivers of Europe and the tropics, where large-scale abstraction fisheries exist, but rivers that are entirely contained within their channels pose a different sort of problem. This can possibly be resolved by extending Habitat Simulation techniques to such reaches, although the criteria for evaluation will differ between upstream and lowland reaches. Table 5.2 lists some of the major factors that need to be considered in upstream (rhithronic) reaches and in lowland (potamonic) reaches. In lowland regions particularly, the factors selected should predict the responses of the two main types of community – rheophilic and limnophilic. It is assumed that eurytopic will remain relatively insensitive. However, all species are likely to respond to differences in condition and population density according to the characteristics of the hydrological regime.

Table 5.2 Some factors that influence fish assemblages in rhithronic and potamonic river reaches

	Upstream (Rhithronic)	Downstream (Potamonic)
Common criteria	Flow velocity Flow timing Bank structure Emergent vegetation Riparian vegetation Longitudinal connectivity	Flow velocity (rheophilic species) Flow timing Bank structure Emergent vegetation Riparian vegetation Longitudinal connectivity
Unique criteria	Wetted area Pool - riffle area Substratum structure and Area	Depth/area – volume Backwater development Lateral connectivity Floodplain development and flooding Refuge habitat

This type of approach is inherent within the outputs of the FAME fisheries assessment methodologies, where the impact of various environmental stressors can be predicted. Whether the outputs will be sensitive enough to accommodate assessment of the impacts of flow abstraction and regulation remains to be seen, as will the ability to discriminate the roles of different impacts on fisheries, but in both cases they should be tested. The broad scale habitat characteristics used to build the models makes this approach a cost effective and easy to use methodology, and routine Agency surveys may provide most of the data required to make evaluations or develop the baseline reference conditions.

A second strategy, to assess the impacts of abstractions on downstream reaches is the development of population dynamics models that will assist in predicting the effects on the quality and quantity of the fish populations/communities of various alternative hydrological regimes. The findings of the preliminary investigation into the utility of this approach, described in Annex 3, appear promising. This type of model assumes that at least one of the three key processes (growth, mortality and recruitment) regulating population biomass is density-dependent, and can be adequately described using established models. Using roach as a pilot species, the investigation found evidence of density-dependent natural mortality in populations inhabiting UK rivers. However, the influence of environmental factors, such as temperature and discharge intensity at certain times of the year, need to be factored into this analysis.

5.5.2 Catchment scale

At the catchment level, a framework is needed that will integrate information for a number of sites within the river. Here there is a choice between the already established CAMS/RAM framework and the development of a DRIFT approach. Any decision as to which of the alternatives is eventually preferable will rest on the need to incorporate a large range of other stakeholders, such as navigation, pleasure boating, wildlife conservation, water birds, in the decision-making process. Should the issue remain purely one of general environmental quality and fisheries, the CAMS framework is adequate, subject to the caveats expressed in Section 4.8. (It should be noted these caveats are not insurmountable and should be addressed as part of the development of methodologies in the next phase). Should a wider range of stakeholders need to be addressed then the development of a DRIFT type approach may be warranted.

Whichever framework is selected, there is still a need to determine the main method for parameterising the fisheries information. **This can be from individual knowledge and opinion under the best professional advice system, but is best if derived from a 'hard' methodology such as PHABSIM. This is one of the key areas for evaluation under the next phase of the project. This should be based on existing data and tested against existing decision-making criteria.** These are all components of the project proposal developed in the next section.

5.5.3 The assessment index

Most evaluation systems use information graded according to an index of a few levels of community response to changes in flow. In both DRIFT and CAMS, these indices are fed into the more general model. Population dynamics models, however, generate a continuum of information, which can also be reduced to discrete indices by assigning appropriate intervals in the response curve to individual states. A generic scoring system of fish assemblage response to water abstractions and augmentations is proposed:

0 Fish assemblage enhanced in abundance and diversity
I Fish assemblage unchanged
II Fish assemblage slightly reduced in abundance
III Fish assemblage moderately reduced in abundance and condition (condition factor and fish health)
IV Fish assemblage severely reduced in abundance, condition and diversity
V Fish assemblage very severely reduced in abundance, condition and diversity.

How these ratings are scored should be a component of the next phase of the project, but they should relate to the scale of the impact on fisheries well-being proposed under the WFD, as the outputs of the two needs are intrinsically linked.

6 KNOWLEDGE GAPS AND FUTURE MONITORING AND RESEARCH NEEDS

6.1 Introduction

Demands on freshwater resources are continuously increasing, but this utilisation must be done in an environmentally friendly way, such that the aquatic biota and habitats are protected. The latter is particularly important because the European Water Framework Directive requires that all water bodies shall achieve good ecological status or good ecological potential. A key requirement for achieving or maintaining good ecological status is the provision of adequate and appropriate flow regimes. Similarly, aquatic fauna, including fish and their associated fisheries, and habitats need to be protected under provision under the Habitats Directive. Consequently, the Agency has the duty to balance the requirements of water resource use, including abstraction, with the needs of the environment. For water abstraction, this is achieved through the abstraction licensing system. In order to achieve the balance between the environment and abstractors more effectively, the Agency has developed the Resource Assessment and Management (RAM) framework to assess the sensitivity of rivers to abstraction in England and Wales, and to assist the production of Catchment Abstraction Management Strategies (CAMS). As part of RAM, an environmental weighting system has been devised which rates a stretch of river on its environmental sensitivity to abstraction. The final rating is calculated by combining scores from four categories: macrophytes, macro-invertebrates, fisheries and the physical nature of the river reach. Abstraction sensitivities for macrophytes and macro-invertebrates are produced using predictive classification tools, i.e. RIVPACS (River Invertebrate Prediction & Classification System) for macro-invertebrates and a system based on the JNCC classification for macrophytes. However, no comparable predictive capability is available for determining the physical characteristics and the fisheries scores.

In terms of fisheries, a project was carried out to develop a typology of fisheries communities typically found in English and Welsh river systems, review the habitat needs of the key species that make up these communities, and review existing methodologies to assess the impacts of altering flow regimes on fish and fisheries. This proposal builds on this information to develop a tool for assessing the impacts of altering flow regimes, especially abstractions, on fish and fisheries that is scientifically robust, and hence allow defensible assessments to be made. The tool will form part of CAMS procedure, the Habitats Directive review of consents and in determining licence applications, in addition to being a general tool for the assessment of water resources schemes at both the local and catchment scale. The knowledge gained will also assist the Agency in determining the ecology that might be expected under reference conditions and in identifying impacts, as required by the Water Framework Directive.

6.2 Priority Areas for Further Development and Research

6.2.1 Fish assemblage typology

The typology identified eight major fish community types. However, there were probably biases in the dataset; in particular, the under representation of certain geographic areas of England and Wales and the minimal sampling on certain zones of rivers. The absence of data from Lincolnshire, the Yorkshire Ridings rivers, the south-west and South Wales form regional gaps that may hide important regional types or geographic patterns. Furthermore, the limited number of surveys undertaken on main river stems in lowland reaches, together with low numbers of sites reflecting the grayling reaches of rivers in many regions, probably masks some important community types. The limited surveys in the grayling zone from regions other than the Hampshire rivers possibly resulted in the absence of a general grayling zone within the typology. An important aspect of the next phase of the project is to try to remove these biases by filling the gaps in information, especially with respect to the regions mentioned above.

For each of these main fish assemblage types, the influence of flow and the potential impacts of abstractions and releases need be considered, by linking key species per community type to their functional ecology and flow requirements. Additionally, it is probable that flow statistics such as mean flow and Q values used in the modelling do not accurately reflect elements of the flow regime that influence fish assemblages. Thus, further work needs to be undertaken to determine the long-term influence of hydrograph characteristics on the fish communities and their dynamics. For this, gaps in the current dataset need to be filled and additional data about population dynamics (recruitment, size and age structure) of the key species need to be considered, over and above community composition data, to determine the long term influence of flow patterns on recruitment success and life histories.

6.2.2 Fish habitat characteristics

The review of habitat characteristics suggested that data exist to determine the preferred habitat characteristics of the predominant fish species found in UK fresh waters. However, there is a paucity of data for the lesser species, especially those of conservation value. Consequently, there is a need to improve on the information about habitat relationships of critical species that drive community structure.

Too few studies have examined the wider environmental impacts of adjusting flows, especially the issues associated with maintaining longitudinal connectivity and facilitating passage of fish about obstructions. This is highly relevant to setting environmental flows that allow the free migration of fish during critical periods of their life cycle. These issues need to be examined and mechanisms for overcoming them are addressed in the next phase of the project. Similarly, little information is available on the importance of relationships between residence times and access to side channels and backwaters from the main river channel for coarse fish species, and no information on these characteristics is available for species of conservation value. This lack of information needs addressing.

6.2.3 Assessment procedures

From the review of assessment methodologies, two possible scenarios for further development were identified and should be pursued.

A A process to arrive at the fish requirements is pursued within the existing framework and objectives of CAMS/RAM. In this case, a relatively simple method of assigning the sensitivity index, such as one of the hydraulic rating, habitat simulation or HABSCORE methods should be developed.

B If the intention is to manage the hydrological regime in the interest of fish and fisheries and use the approach to encompass assessment of all types of water resource schemes, two main approaches seem to be desirable:

1) The development of a RIVPACS type model for fish, e.g. River Fish Environmental Flow Assessment Matrix (RIFEFAM). This model can either be based on species presence or absence, relative species abundance or incorporate those parameters of the fish such as biomass, condition, and growth and survival rates that are needed to manage the fishery. This tool should be generic in nature and used for assessment of all types of water resource development schemes, both at the catchment scale as well as within localised studies. It is suggested that the possibilities of adapting the outcomes of the EU FAME project be investigated to meet this requirement. Once the tool has been developed, the output can be integrated into the RAM assessment procedures and be compatible with similar procedures for invertebrate and macrophtyes.

2) The development of population dynamics models that will assist in predicting the effects on the quality and quantity of the fish population of various alternative hydrological regimes.

It is recommended that both approaches are examined in detail and the most appropriate for meeting the objectives of the Agency in terms or assessing the impacts water resource scheme is selected.

6.3 Project Proposal

6.3.1 Overall aim

The overall objective of the project will be to develop, trial and evaluate a quantitative tool or suite of tools that can be used to assess the impacts of given flow regimes on fish and fisheries.

6.3.2 Specific objectives

- To review the fish assemblage typology and remove biases due to incomplete or absent datasets.

- To determine the long-term influence of hydrograph characteristics on the fish communities and their dynamics.

- To improve on the information about habitat relationships of critical species that drive community structure.

- To examine the wider environmental impacts of adjusting flows, especially the issues associated with maintaining longitudinal and lateral connectivity and facilitating passage of fish about obstructions and into backwater and refuge habitat.

- To develop an appropriate model or models for assessment of the impact of water resources schemes on fish and fisheries and encapsulate the methodology into the RAM procedures.

6.4 Project Activities

The following activities are required to meet the demands of the specific objectives.

6.4.1 To review the fish assemblage typology and remove biases due to incomplete or absent datasets

To meet this objective requires further interrogation of the Agency databases and revision of the typology modelling to account for the missing data and biases in the output. This will require Agency staff to provide input into determination of the impact criteria at each site and also provision of the biological data in the appropriate format. Further information may be required on habitat variables used in the analysis. The expected output will be a revision of the typology provided in the first phase of the project and greater precision of the fish community types that must be inbuilt into the CAMS assessment procedure.

6.4.2 To determine the long-term influence of hydrograph characteristics on the fish communities and their dynamics.

One of the limitations of the information available is the weak understanding of the drivers between flow dynamics and fish community dynamics. This is fundamental to understand the impact of flow on fisheries and thus modelling of the various components of the hydrograph on fish population and community dynamics is imperative. Two activities are required:

- Collation of data on fisheries population structure and dynamics for each river community type identified previously
- Undertake iterative modelling to advise on the characteristics of the hydrograph that are driving the community dynamics.

6.4.3 To improve on the information about habitat relationships of critical species that drive community structure

Information on habitat relationships of critical species that drive community structure is imperative to underpin the modelling procedure and help define the impact of water resource schemes on fisheries. This component will need to enhance the information database to support the CAMS procedure by:

- collating data on fisheries population structure and habitat characteristics for species that are poorly represented in the literature, especially rare and threatened species.
- updating the databases on fisheries population structure and habitat characteristics for well documented species where considered appropriate.
- undertaking modelling of habitat suitability indices to determine the key habitat drivers for each species and the fish communities as a whole.

6.4.4 To examine the wider environmental impacts of adjusting flows, especially the issues associated with maintaining longitudinal and lateral connectivity and facilitating passage of fish about obstructions, and into backwater and refuge habitat.

To assess the wider environmental impacts of adjusting flows, the following activities are proposed.

- Monitoring the fish community structure along longitudinal and lateral gradients in relation to different flow regimes.
- Modelling the optimal flow regimes to ensure free migration of fish during critical periods of their life cycle.
- Modelling the relationships between residence times and access to side channels and backwaters from the main river channel for coarse fish species and species of conservation value.

It should be recognised that this element of the programme may require acoustic tracking work that has not been costed.

6.4.5 To develop an appropriate model or models for incorporation to assess the impacts of given flow regimes on fish and fisheries.

The ultimate output of the project is to develop a robust, defensible model that can be used to assess the impact of water resources schemes on fish and fisheries to ensure they are protected. Several approaches were identified that all have potential and need further evaluation.

1. Develop and test a simple method of assigning an impact/sensitivity index, such as one of the hydraulic rating, habitat simulation or HABSCORE methods. Existing information and modelling carried out by HIFI for the major coarse fish species can be used as the template for this work.

2. Develop a River Fish Environmental Flow Assessment Matrix (RIFEFAM). This model can either be based on species presence or absence, relative species abundance or incorporate those parameters of the fish such as biomass, condition, and growth and survival rates that are needed to manage the fishery.

3. The results of the preliminary analysis and the model predictions for age-structured population dynamics modelling require validation and field-testing before this approach could be recommended and adopted as part of the water resources assessment procedures. The following activities are proposed:

    - Formally audit fisheries datasets held by the EA, particularly in relation to the availability of length-at-age, numbers-at-age and numerical and biomass density estimates (including their proxies e.g. angler CPUE) of coarse and conservation species of fish.

- Formally audit corresponding hydrological datasets held by the EA, particularly in relation to the time series (weekly) estimates of flooded area and volume.

- Using similar approaches to those described in Annex 5, seek further evidence for density-dependent growth, mortality and recruitment responses among selected species. Also examine the importance of density-independent (abiotic) factors affecting these processes such as temperature and discharge rates, and describe these effects using appropriate empirical models. These could either be based upon temporal comparisons within selected sites/reaches or based upon among site/reach comparisons.

- Predict the affects of abstractions on selected populations and where possible test model predictions by abstracting water from the reach/river and monitoring the response of the resident population(s). Compare observations with model predictions to determine reliability of the model predictions.

The financial/logistical implications of routinely monitoring fisheries and hydrological variables on a reach/site basis to determine the utility of the various modelling approaches and their incorporation into assessment procedures will be undertaken and a decision on the best approach recommended.

6.5 Integrated R&D Strategy

Section 6.3 identified the requirements of three main research questions considered priorities to inform the water resources assessment procedures. These topics are not wholly independent and will probably all be required to provide an integrated assessment of the impacts of given flow regimes on fish and fisheries. Furthermore, given that many of the general project tasks have a similar end-point, and that many of the elements are not mutually exclusive, it is recommended that each element be merged into a comprehensive R&D project strategy. Combining the elements of research in this way will also minimise resource constraints in terms of the number of fisheries required. The time scales and indicative costs presented in Section 6.3.3 reflect that the most appropriate R&D strategy is to merge the individual research elements into one central project.

To achieve the overall objective of the project, i.e. 'To provide a mechanism for the incorporation of fish and fisheries into water resources assessment procedures, there is a need to integrate various elements of the programme to ensure maximum benefits are accrued. This requires comprehensive planning that is summarised within the logical project framework (Table 6.1). This tool is a planning framework typical of that used by major donor agencies such as the European Commission and World Bank, and provides an overview of the objectives, the main activities, expected outputs and measurable indicators that these outputs have been achieved. This framework includes other elements of the project that are not considered as part of the risks being evaluated in this document. However, they are included for completeness, and cannot be divorced from the overall project structure. The logical project framework should be read in conjunction with the design of the project modules (Sections 6.3.1 and 6.3.3), Gantt chart (Fig. 6.1) and budgetary statement (Section 6.3.3) to provide an overview of the project formulation phase.

Table 6.1 Logical project framework identifying objectives, activities and expected outputs for assessing the impacts of given flow regimes on fish and fisheries

	Intervention logic	Indicators of achievement	Verification	Assumptions
Overall Objective	To provide a mechanism for the incorporation of fish and fisheries in the CAMS procedure.	Strategy for assessment of impact of fish and fisheries into the CAMS procedure.	Inclusion in the CAMS procedure.	Willingness to implement recommendations at regional level.
Project purpose	To develop, trial and evaluate a quantitative tool or suite of tools that can be used to assess the impacts of given flow regimes on fish and fisheries.	Introduction of effective strategy to assess the impact of abstraction on fish and fisheries. Adoption of recommendations for proposed assessment strategy. Availability of information base for management decisions.	Uptake of assessment protocols recommended. CAMS manual.	Acceptance of proposals for CAMS procedure by fishery managers. Availability of suitable rivers systems and data.
Results	1) To review the fish assemblage typology and remove biases due to incomplete or absent datasets	Successful completion of data collation and modelling.	Documentation and evaluation.	Availability of resources to support project. Unforeseen interventions affecting the successful outcome of modelling procedures. Recommendations for CAMS policy accepted at regional level.
	2) To determine the long-term influence of hydrograph characteristics on the fish communities and their dynamics.	Successful completion of data collation and modelling.	Documentation and evaluation.	
	3) To improve on the information about habitat relationships of critical species that drive community structure	Successful completion of data collation and modelling.	Documentation and evaluation.	
	4) To examine the wider environmental impacts of adjusting flows, especially the issues associated with maintaining longitudinal and lateral connectivity and facilitating passage of fish about obstructions and into backwater and refuge habitat.	Completion of fishery appraisal.	Documentation and evaluation.	
	5) To develop an appropriate model or models for incorporation in the CAMS procedure to assess the impacts of given flow regimes on fish and fisheries.	Adoption and implementation of procedure in CAMS Strategy.	Project monitoring.	

Activities	1.1. Collate updated database on fish population structures and	*Inputs : Environment Agency*		*Assumes that*

	environmental data 1.2 Revise fish typology framework. --- 2.1. Collation of data on fisheries population structure and dynamics for each river community type identified previously 2.2 Undertake iterative modelling to advise on the characteristics of the hydrograph that are driving the community dynamics. --- 3.1. Collate data on fisheries population structure and habitat characteristics for species that are poorly represented in the literature, especially rare and threatened species. 3.1 Update the databases on fisheries population structure and habitat characteristics for well documented species where considered appropriate. 3.3 Undertake modelling of habitat suitability indices to determine the key habitat drivers for each species and the fish communities as a whole. --- 4.1 Monitoring the fish community structure along longitudinal and lateral gradients in relation to different flow regimes. 4.2 Model the optimal flow regimes to ensure free migration of fish during critical periods of their life cycle. 4.3 Model the relationships between residence times and access to side channels and backwaters from the main river channel for coarse fish species and species of conservation avalue. --- 5.1. Develop and test a simple method of assigning an impact/ sensitivity index, such as one of the hydraulic rating, habitat simulation or HABSCORE methods. 5.2 Develop a River Fish Environmental Flow Assessment Matrix (RIFEFAM). 5.3 Test utility of population dynamics models. 5.4 Revision / adjustments made to CAMS manual and procedures.	*Technical assistance* Support for gaining data for undertaking the project and/or providing access to the contractor to Agency databases. *Materials* Costs of computing and GIS input requirements *Operating expenses* Project operating costs of consultants *Meetings* Project and non project meetings *Consultants* Provision of man-power and appropriate equipment to undertake the project.		Access agreements and clearance for research are timely and not contested. Researchers carry out and report activities as per approved work programmes. Financing and budget approved by project for each activity.

Figure 6.1 Scheduling of inputs and outputs.

Activity	Months of the study																		
	0	2	4	6	8	10	12	14	16	18	20	22	24	26	28	30	32	34	36
Liaison with EA staff																			
Review the fish assemblage typology and remove biases due to incomplete or absent datasets																			
Determine the long-term influence of hydrograph characteristics on the fish communities and their dynamics																			
Improve on the information about habitat relationships of critical species that drive community structure																			
Examine the wider environmental impacts of adjusting flows, especially the issues associated with maintaining longitudinal and lateral connectivity and facilitating passage of fish about obstructions and into backwater and refuge habitat.																			
Develop an appropriate model or models for incorporation in the CAMS procedure to assess the impacts of given flow regimes on fish and fisheries.																			
Submission of interim reports							*						*						
Submission of draft final report																		*	
Submission of final report																			*
Project board review meetings	*			*			*			*			*			*			*

It is anticipated that the Environment Agency and other groups (e.g. Centre for Ecology and Hydrology) are already carrying out research into assessing the impacts of given flow regimes on fish and fisheries, and other biota. The strategic R&D plan for assessing the impacts of given flow regimes on fish and fisheries should make full use of such existing research programmes to inform the assessment.

6.6 Timescale and Indicative Costs

The following section provides approximate timescales and costs of undertaking the proposed units of R&D (Section 6.3.1). The timescale proposed covers a 36-month period to ensure sufficient data are collated (one cycle of routine fisheries monitoring) to allow assessment of the key components of impact of abstraction on a range of fishery types. The scheduling of key activities is reflected in the Gantt chart (Fig 6.1).

Indicative costs (Option 1) are based on the assessment of all water body types over a 36-month period. It is believed that the most cost-effective way of carrying out the research is to have one dedicated research assistant allocated to the project to support the data collection and undertake the bulk of the analysis. The project would be supported by senior experienced staff. Further reductions in manpower costs can be achieved by substituting the Post-doctoral Research Assistant with a Postgraduate Research Assistant registered for a PhD (Option 2). This option is only available if the project is run over a three-year period. The costs would be approximately £14,000 per year stipend (NERC studentship rate) and £3000 per year registration fees (both subject to elements of inflation at about 5%). This not only has the advantage of reducing costs, but also providing more dedicated time (one continuous full-time person for the duration of the project) and training up a person for potential employment by the Agency, allaying the recognised shortfall of highly trained persons for recruitment to Agency professional positions.

The proposed budget does not include any overheads, which are currently running at about 40% for the university sector, and is exclusive of VAT. It should be noted that if the project is established as a research project within the university sector, the financial agreement is not subject to VAT.

No equipment costs are identified as it is expected that the Agency will provide all the data required for the project.

It is suggested that Project Board meetings are held at the start and at six-monthly intervals until the end of the project to establish the project structure, discuss progress and interim results and to review the project outputs, respectively.

6.7 Risks and Constraints

It is recognised that there are two key risks associated with the future R&D project.

- the availability of suitable data relating to fisheries in the key river types;
- acceptability of the output of the research for integration within the water resources assessment procedures.

Option 1 – Costings for Phase 2 over 36 months with Postdoctoral Research Assistant (RA)
Work task manpower indicative costs

Task	Staff Type	Day rate £	No of Days	Total £
1. Project board meetings	Project manager	450	7	3150
	Research associate	300	7	2100
Total			**14**	**5250**
2. To review the fish assemblage typology and remove biases due to incomplete or absent datasets	Project manager	450	2	900
	Research associate	300	10	3000
	Postdoctoral RA	200	20	4000
Total			**27**	**7900**
3. To be undertaken to determine the long-term influence of hydrograph characteristics on the fish communities and their dynamics	Research associate	300	15	4500
	Postdoctoral RA	200	50	10000
Total			**55**	**14500**
4. To improve on the information about habitat relationships of critical species that drive community structure	Project manager	450	10	4500
	Research associate	300	30	9000
	Postdoctoral RA	200	50	10000
Total			**90**	**23500**
5. To examine the wider environmental impacts of adjusting flows, especially the issues associated with maintaining longitudinal and lateral connectivity and facilitating passage of fish about obstructions and into backwater and refuge habitat.	Project manager	450	10	4500
	Research associate	300	40	12000
	Research assistant	200	60	12000
	Technical assistant	100	40	4000
Total			**150**	**32500**
6. To develop an appropriate model or models for incorporation in the water resources assessment procedures to assess the impacts of given flow regimes on fish and fisheries.	Project manager	450	10	4500
	Modelling/flows expert	450	50	22500
	Research associate	300	20	6000
	Research assistant	200	80	16000
Total			**160**	**49000**
Project management		450	9	4050
Grand total (3 years)				**136700**

Daily rates

Staff Name £	Daily Rate £	Proposed Number of Days	TOTAL £
Project manager	450	48	21600
Research associate	300	122	36600
Modelling/flows expert	450	50	22500
Postdoctoral RA	200	260	52000
Technical Assistant	100	40	4000
TOTAL		**520**	**136700**

Total costs

1. STAFF	**136700**
2. TRAVEL & SUBSISTENCE FOR ATTENDING PROJECT MEETINGS	5000
3. OTHER COSTS*	
Computing	4000
Reporting	1000
Total	**146700**

The total cost of the R&D project would be £146700 exclusive of VAT

These costs are indicative of the budget required to undertake the programme of R&D described. The costs do not include overheads, which typically within the University sector are 40%. All costs are exclusive of VAT and are subject to inflation at the Retail Price Index. The costs do not include the following equipment elements which my be required:

Manual radio-tracking equipment (2 receivers and Yagi antennae) Total value: **£3000**
Electric fishing equipment, nets, drysuits, boats, GPS and other survey equipment. Total value: **£15000**

Option 2 – Costings for Phase 2 over 36 months with Postgraduate Research Assistant (PG RA)

Work task manpower indicative costs

Task	Staff Type	Day rate £	No of Days	Total £
1. Project board meetings	Project manager	450	7	3150
	PG RA		7	*
Total			**14**	**3150**
2. To review the fish assemblage typology and remove biases due to incomplete or absent datasets	Project manager	450	2	900
	Research associate	300	10	3000
	PG RA	*	50	*
Total			**62**	**3900**
3. To be undertaken to determine the long-term influence of hydrograph characteristics on the fish communities and their dynamics	Research associate	300	10	3000
	PG RA	*	100	*
Total			**110**	**3000**
4. To improve on the information about habitat relationships of critical species that drive community structure	Project manager	450	10	4500
	Research associate	300	15	4500
	PG RA	*	150	*
Total			**175**	**9000**
5. To examine the wider environmental impacts of adjusting flows, especially the issues associated with maintaining longitudinal and lateral connectivity and facilitating passage of fish about obstructions and into backwater and refuge habitat.	Project manager	450	10	4500
	Research associate	300	20	6000
	Research assistant	*	150	*
	Technical assistant	100	40	4000
Total			**220**	**20500**
6. To develop an appropriate model or models for incorporation in the water resources assessment procedures to assess the impacts of given flow regimes on fish and fisheries.	Project manager	450	10	4500
	Modelling/flows expert	450	50	22500
	Research associate	300	20	6000
	Research assistant	*	153	*
Total			**233**	**33000**
Post graduate research assistant*			660	54000
Project management		450	9	4050
Grand total (3 years)				**124600**

The above pricing structure replaces the Postdoctoral RA staff member (day rate = £200) with a Postgraduate RA (designated * in the table) with an annual salary of £14000 plus annual PhD registration fees of £3000.

Daily rates

Staff Name £	Daily Rate £	Proposed Number of Days	TOTAL £
Project manager	450	48	21600
Research associate	300	75	22500
Modelling/flows expert	450	50	22500
Post graduate research assistant		660	54000
Technical Assistant	100	40	4000
TOTAL		**873**	**124600**

Total costs

1. STAFF	**124600**
2. TRAVEL & SUBSISTENCE FOR ATTENDING PROJECT MEETINGS	5000
3. OTHER COSTS*	
Computing	4000
Reporting	1000
Total	**134600**

The total cost of the R&D project would be £134600 exclusive of VAT

These costs are indicative of the budget required to undertake the programme of R&D described. The costs do not include overheads, which typically within the University sector are 40%. All costs are exclusive of VAT and are subject to inflation at the Retail Price Index. The costs do not include the following equipment elements which my be required:

Manual radio-tracking equipment (2 receivers and Yagi antennae) Total value: **£3000**
Electric fishing equipment, nets, drysuits, boats, GPS and other survey equipment. Total value: **£15000**

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APPENDIX 1

Table A1.1 Summary of catch statistics for the most recent samples from each site in the English and Welsh dataset (n = 1062). Mean abundance data (CPUE n ha^{-1}) are calculated as the average abundance where the species is present.

	% Occurrence	Average CPUE n ha^{-1}	Max
Anguilla anguilla	52.8	303	20409
Rutilus rutilus	52.5	383	10000
Salmo trutta fario	49.3	905	18628
Leuciscus cephalus	47.1	171	5393
Leuciscus leuciscus	44.4	163	2584
Gobio gobio	41.6	216	9000
Cottus gobio	41.1	1577	37454
Esox lucius	41.1	35	373
Barbatula barbatula	39.5	1207	74075
Perca fluviatilis	39.4	75	1850
Phoxinus phoxinus	34.7	1487	37454
Salmo salar	18.4	2166	16214
Gasterosteus aculeatus	15.1	1357	74075
Abramis brama	10.7	31	362
Thymallus thymallus	7.3	130	867
Scardinius erythrophthalmus	6.4	29	140
Tinca tinca	6.3	14	61
Barbus barbus	6.1	35	449
Alburnus alburnus	6.0	151	864
Cyprinus carpio	5.4	29	215
Lamprey all	5.2	414	4167
Gymnocephalus cernuus	3.3	24	175
Platichthys flesus	2.2	531	5840
Rutilus rutilus x Abramis brama	2.1	12	54
Oncorhynchus mykiss	1.7	67	364
Rutilus rutilus x Scardinius erythrophthalmus	1.2	14	69
Carassius auratus	0.6	9	21
Cobitis taenia	0.6	255	925
Carassius carassius	0.5	11	32
Chelon labrosus	0.5	23	35
Blicca bjoerkna	0.4	11	30
Gobiidae	0.4	100	240
Leuciscus idus	0.2	16	21
Pungitius pungitius	0.2	10	16
Dicentrarchus labrax	0.1	8	8
Lepomis gibbosus	0.1	16	16
Pleuronectes platessa	0.1	34	34
Salvelinus fontinalis	0.1	20	20
Sander lucioperca	0.1	18	18

Table A1.2 **Mean 1st run CPUE ($n.ha^{-1}$) per native fish species in each of the 15 types described by cluster analysis. Distinctive species (shaded) / species abundance (bordered) per group are highlighted.**

Fish community type	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**
Abramis brama				0.2	0.2	1.1	1.7	1.5	40.5	2.5	2.6		2.7	1.3	6.3
Alburnus alburnus				0.2			3.5		0.0	0.1	1.3		0.0	3.3	182.6
Anguilla anguilla	92.0	231.4	25.3	75.8	1940.6	83.5	41.3	93.2	68.6	42.2	104.7	70.7	80.1	63.3	38.4
Barbatula barbatula	372.8	67.3	139.4	2612.6	1159.2	85.0	1182.2	312.4	101.9	61.1	237.8	51.9	59.0	93.4	0.9
Barbus barbus				0.5	0.5	1.4	16.4	2.4	0.0		0.1		0.0	0.8	3.8
Blicca bjoerkna							0.1				0.0			0.3	0.1
Carassius carassius												1.4	0.0		
Cobitis taenia							14.3		0.4						
Cottus gobio	782.0	2882.9	271.5	2071.1	1479.8	216.3	305.8	255.5	68.7	39.5	316.2	35.4	78.2	129.7	31.1
Cyprinus carpio							1.9	0.8	20.1	2.1	0.3	1.4	1.0	0.2	0.9
Esox lucius		1.0		1.8	0.4	10.1	6.0	1.4	6.5	40.1	5.5	4.8	50.6	26.5	24.0
Gasterosteus aculeatus	17.5	38.7	2.8	178.9	211.3	23.3	1604.5	51.3	10.5	0.8	154.8	7.6	4.1	37.3	21.0
Gobio gobio	0.1	1.1	1.0	40.9	14.5	16.2	336.0	70.2	95.0	18.6	207.9	164.9	30.6	196.6	54.8
Gymnocephalus cernuus							5.0	0.8	3.6	0.5	0.1		0.1		0.8
Lamprey	26.7	126.6	41.4	13.1	0.4	74.6	13.2	0.9			5.5	0.4	0.0	1.5	
Leuciscus cephalus		0.9	0.4	77.3	19.1	40.8	268.1	146.9	91.8	21.4	140.1	124.6	34.6	152.9	37.0
Leuciscus leuciscus				73.3	26.9	42.0	74.7	411.3	107.2	31.5	70.0	193.7	39.0	176.1	31.3
Perca fluviatilis	0.3	8.1		3.7	0.6	13.5	11.2	7.0	40.3	54.7	190.5	28.6	35.7	31.4	45.4
Phoxinus phoxinus	109.6	117.3	121.3	4273.5	912.4	143.2	531.7	57.6	102.6	33.0	188.9	130.5	91.7	237.5	47.4
Platichthys flesus					256.5		1.0		0.9		1.6		3.1	0.2	4.0
Pungitius pungitius										0.1	0.2				
Rutilus rutilus		10.7	9.0	50.1	19.5	35.8	144.9	190.5	395.2	441.5	784.8	237.6	322.0	282.8	192.4
Salmo salar	3399.8	21.7	123.1	105.3	42.3	269.2	0.3	0.7				0.5	0.0		0.4
Salmo trutta	1296.7	1243.0	2558.1	304.9	84.4	161.0	54.5	104.0	41.8	33.9	35.4	172.9	7.3	8.3	40.3
Scardinius erythrophthalmus		0.2		1.3			0.5		0.6	1.9	1.6	39.5	0.9	0.1	2.9
Thymallus thymallus		1.8		0.6	15.3	192.5	1.0	0.3				0.9	0.1	0.3	0.2
Tinca tinca						0.2	0.0		2.1	16.3		0.8	0.1	0.2	0.7

Table A1.3 **Summary of catch statistics in the "undisturbed" dataset. Mean abundance data are calculated as the average abundance where the species is present.**

	% Occurrence	Average CPUE n ha^{-1}	Max
Salmo trutta fario	70.5	1287	18628
Anguilla anguilla	56.1	550	20409
Cottus gobio	51.6	1688	20409
Barbatula barbatula	43.3	1335	20409
Salmo salar	39.7	2446	16214
Leuciscus leuciscus	36.9	142	1700
Phoxinus phoxinus	35.6	1859	19231
Rutilus rutilus	35.6	374	4518
Leuciscus cephalus	34.0	161	1036
Gobio gobio	31.1	187	1450
Esox lucius	27.6	35	373
Perca fluviatilis	23.1	101	1850
Thymallus thymallus	13.5	142	728
Gasterosteus aculeatus	12.5	620	6098
Barbus barbus	6.1	30	92
Lamprey all	5.1	393	2611
Abramis brama	4.8	38	200
Scardinius erythrophthalmus	3.8	30	96
Alburnus alburnus	3.2	161	864
Gymnocephalus cernuus	2.6	60	175
Cyprinus carpio	1.9	45	154
Oncorhynchus mykiss	1.6	45	79
Cobitis taenia	1.3	375	925
Gobiidae	1.0	126	240
Platichthys flesus	0.6	5054	5840
Tinca tinca	0.6	6	7
Blicca bjoerkna	0.3	30	30
Lepomis gibbosus	0.3	16	16
Pleuronectes platessa	0.3	34	34

Table A1.4 **Mean 1st run CPUE ($n.ha^{-1}$) per native fish species in each of the eight major types in the "undisturbed" dataset, described by cluster analysis. Distinctive species (shaded) / species abundance (bordered) per group are highlighted.**

	Fish type code by longitudinal zonation							
	1	2	3	4	5	6	7	8
A. brama				1.7	0.6	5.0	0.1	2.1
A. alburnus		3.1		50.8	7.6		10.8	
A. anguilla	82.1	116.2	1096.1	1107.7	595.4	94.5	85.3	19.8
B. barbatula	51.6	277.9	1446.1	238.2	718.2	862.2	127.0	8.2
B. barbus					19.4	0.4		0.4
B. bjoerkna							0.9	
C. taenia						15.9		
C. gobio	106.4	845.0	2922.8	619.9	615.3	827.1	78.4	105.9
C. carpio						2.8		
E. lucius		0.6			6.9	8.0	56.5	9.2
G. aculeatus		1.8	2.3	655.9	10.9	107.5	43.8	41.9
GOBIIDAE				22.0		0.1		
G. gobio		0.6		1.4	27.3	165.5	51.8	6.7
G. cernuus						4.9	0.5	
Lamprey all		7.3	24.4	254.9		6.5		
L. gibbosus							0.5	
L. cephalus		1.0	2.7	2.5	72.9	135.9	49.9	20.6
L. leuciscus		0.9		13.6	37.8	136.6	51.3	23.6
O. mykiss	2.4			3.8		0.1	2.5	0.9
P. fluviatilis		1.4	0.9	1.2	6.3	63.6	21.8	10.6
P. phoxinus		93.9	588.4	366.1	850.7	1282.4	829.5	101.3
P. flesus				594.5				
P. platessa				2.0				
R. rutilus		4.0	6.2	35.4	50.7	309.9	266.6	55.2
S. salar	455.9	4262.5	341.3	439.8	15.9	31.9		443.9
S. t. fario	4890.7	1296.3	2138.8	1096.4	33.7	82.1	13.9	173.2
S. erythrophthalmus		0.1		2.6		3.3	0.2	
T. thymallus			5.7		9.5	3.1		207.6
T. tinca						0.1		

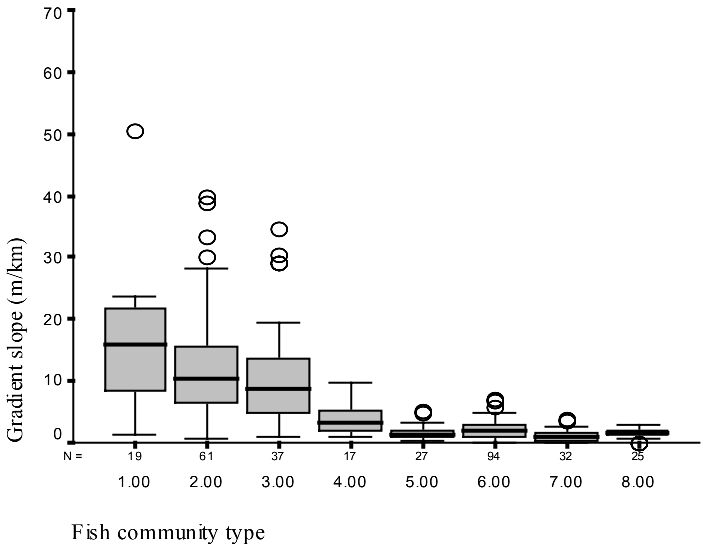

Figure A1.1 Box plot of gradient slope values per fish community type. Black bars indicate median values, shaded boxes illustrate 25th and 75th percentiles, the whiskers indicate the lowest and highest value within 1.5 x the inter-quartile range from the 25th and 75th percentile respectively. Outliers are plotted as circles.

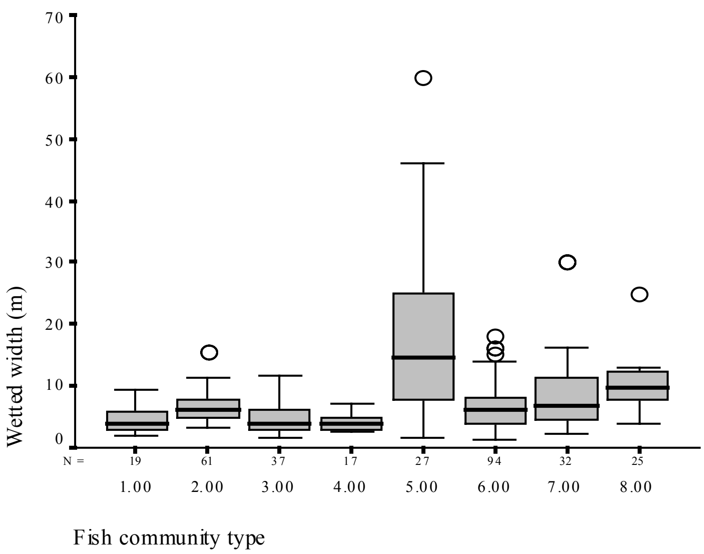

Figure A1.2 Box plot of wetted width values per fish community type. Black bars indicate median values, shaded boxes illustrate 25th and 75th percentiles, the whiskers indicate the lowest and highest value within 1.5 x the inter-quartile range from the 25th and 75th percentile respectively. Outliers are plotted as circles.

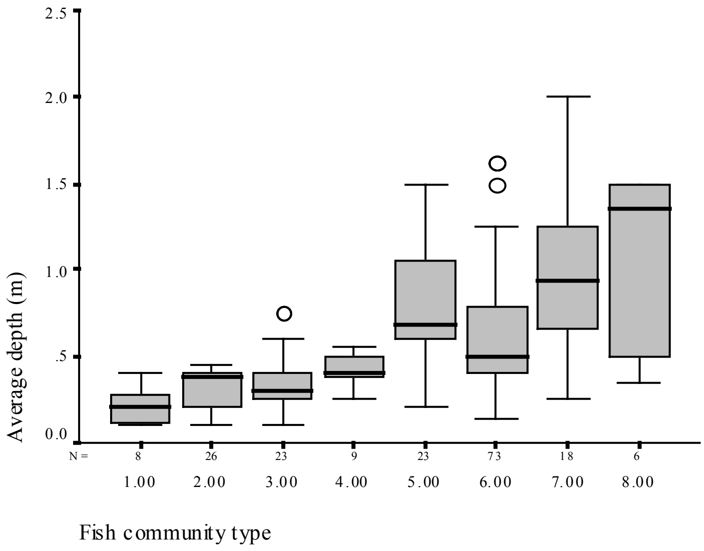

Figure A1.3 **Box plot of site depth values per fish community type. Black bars indicate median values, shaded boxes illustrate 25th and 75th percentiles, the whiskers indicate the lowest and highest value within 1.5 x the inter-quartile range from the 25th and 75th percentile respectively. Outliers are plotted as circles.**

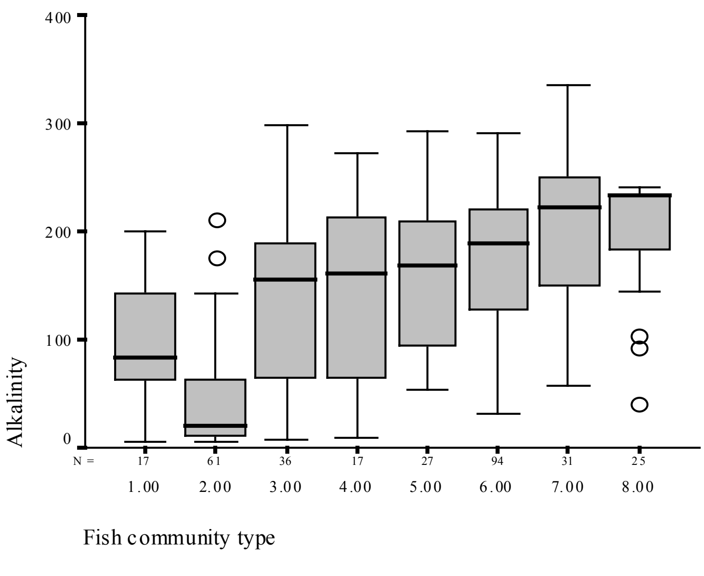

Figure A1.4 **Box plot of site alkalinity values per fish community type. Black bars indicate median values, shaded boxes illustrate 25th and 75th percentiles, the whiskers indicate the lowest and highest value within 1.5 x the inter-quartile range from the 25th and 75th percentile respectively. Outliers are plotted as circles.**

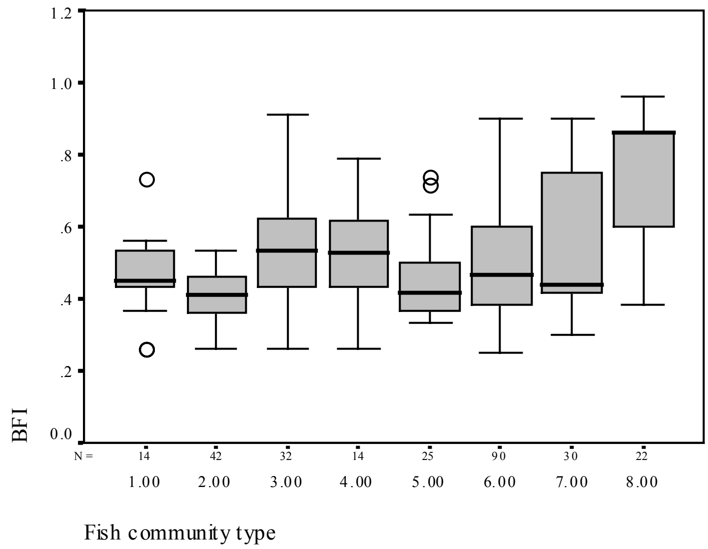

Figure A1.5 Box plot of base flow index per fish community type. Black bars indicate median values, shaded boxes illustrate 25th and 75th percentiles, the whiskers indicate the lowest and highest value within 1.5 x the inter-quartile range from the 25th and 75th percentile respectively. Outliers are plotted as circles.

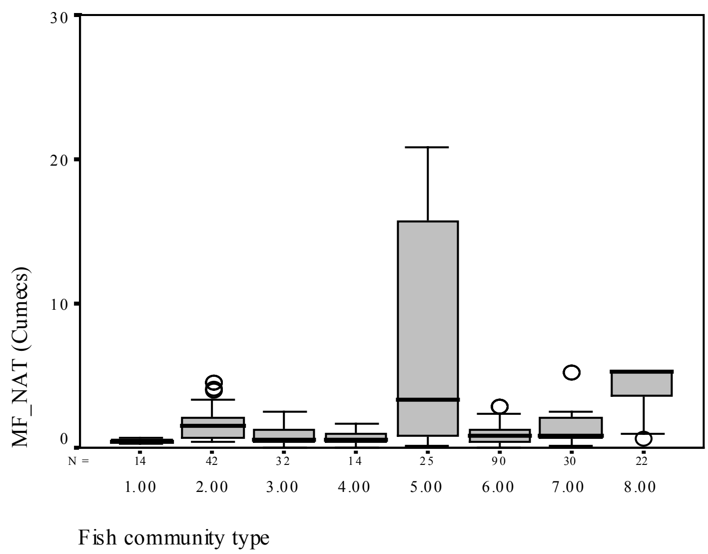

Figure A1.6 Box plot of site mean flow per fish community type. Black bars indicate median values, shaded boxes illustrate 25th and 75th percentiles, the whiskers indicate the lowest and highest value within 1.5 x the inter-quartile range from the 25th and 75th percentile respectively. Outliers are plotted as circles.

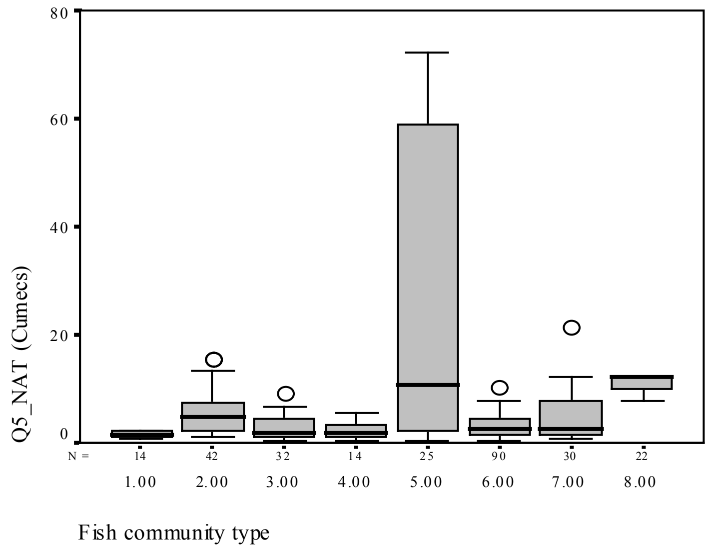

Figure A1.7 **Box plot of site Q5 per fish community type. Black bars indicate median values, shaded boxes illustrate 25th and 75th percentiles, the whiskers indicate the lowest and highest value within 1.5 x the inter-quartile range from the 25th and 75th percentile respectively. Outliers are plotted as circles.**

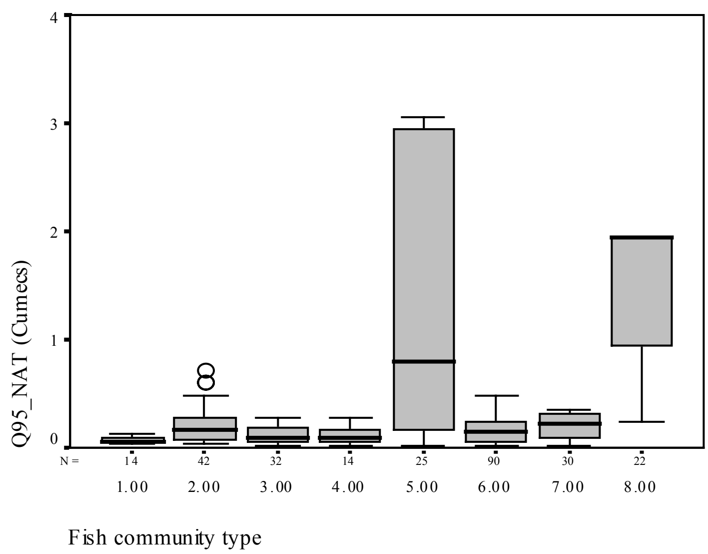

Figure A1.8 **Box plot of site Q5 per fish community type. Black bars indicate median values, shaded boxes illustrate 25th and 75th percentiles, the whiskers indicate the lowest and highest value within 1.5 x the inter-quartile range from the 25th and 75th percentile respectively. Outliers are plotted as circles.**

APPENDIX 2

Table A2.1 Matrix of habitat characteristics of British freshwater fishes

Species	Life stage	Time of year	River type	Water depth requirements	Flow requirements	Reference
Abramis brama	Larvae	May to Oct	A small abandoned channel (Les Nappes) on the upper River Rhône, France	Depths 20-50 cm	Still water	Copp (1993)
	Larvae	May to Oct	River Great Ouse, eastern England	Depths <100 cm	Velocities <5 $cm.s^{-1}$	Garner (1996b)
	Larvae	April to Sept	Lower River Rhine, The Netherlands	Depths <40 cm	Still water	Grift *et al*. (2003)
	Larvae	May to Sept	Kyrönjoki River estuary, Finland	Shallow bays (<150 cm deep)		Urho *et al*. (1990)
	Juvenile	May to Oct	A small abandoned channel (Les Nappes) on the upper River Rhône, France	Depths >100 cm	Still water	Copp (1993)
	Juvenile	May to Oct	River Great Ouse, eastern England	Depths <100 cm	Velocities <5 $cm.s^{-1}$	Garner (1996b)
	Juvenile	Apr to Sept	Lower River Rhine, The Netherlands	No depth preference	Velocities <5 $cm.s^{-1}$	Grift *et al*. (2003)
	Juvenile	May to Sept	Kyrönjoki River estuary, Finland	Depths to ~125 cm		Urho *et al*. (1990)
	Spawning	May to June	Lowland rivers	Depths ~50 cm	Velocities <20 $cm.s^{-1}$	Mann (1996), Cowx & Welcomme (1998), Cowx (2001)
	Spawning	Apr to May	A harbour on the River Meuse, Belgium	Depths 25-50 cm		Poncin *et al*. (1996)
Alburnus alburnus	Larvae	June to Sept	A braided side-channel of the upper River Rhône, France	Depths 20-50 cm	Lentic waters	Copp (1992b)
	Larvae	May to Oct	River Great Ouse, eastern England	Depths <100 cm	Velocities <5 $cm.s^{-1}$	Garner (1996b)
	Larvae	Apr to Sept	Lower River Rhine, The Netherlands	Depths <40 cm	Still water	Grift *et al*. (2003)

Species	Life stage	Time of year	River type	Water depth requirements	Flow requirements	Reference
Alburnus alburnus	Juvenile	June to Sept	A braided side-channel of the upper River Rhône, France	Depths <20 cm (but >100 cm during low discharge conditions)	Lentic waters	Copp (1992b)
	Juvenile	Apr to Sept	Lower River Rhine, The Netherlands	Depths <50 cm	Velocities <5 $cm.s^{-1}$	Grift *et al*. (2003)
	Spawning		Lowland rivers		Velocities <20 $cm.s^{-1}$	Mann (1996)
Alosa alosa	Larvae			Shallow	Slow flowing areas	Maitland & Hatton-Ellis (2003)
	Juvenile			Depths to 300 cm		Maitland & Hatton-Ellis (2003)
	Spawning		River Garonne, France	Depths ~200 cm	Mean Velocities 80-150 $cm.s^{-1}$	Belaud *et al*. (2001)
	Spawning		Range of European rivers	Depths 50-300 cm	Velocities 50-200 $cm.s^{-1}$	Le Clerc (1941), Hoestlandt (1958), Cassou-Leins & Cassou-Leins (1981), Dautrey & Lartigue (1983), Boisneau *et al*. (1990)
	Spawning	Apr to July		Shallow water	Velocities 100-150 $cm.s^{-1}$	Maitland & Hatton-Ellis (2003)
	Spawning		River Garonne, France	Depths 100-150 cm	Velocities ~100 $cm.s^{-1}$	Maitland *et al*. (1995)
Alosa fallax fallax	Larvae			Shallow	Slow flowing areas	Maitland & Hatton-Ellis (2003)
	Juvenile			Depths to 300 cm		Maitland & Hatton-Ellis (2003)
	Spawning		River Wye, Wales	Depths 15-60 cm		Aprahamian (1981)
	Spawning		River Severn, England	Depths up to 300 cm		Aprahamian (1981), Maitland & Hatton-Ellis (2003)
	Spawning		Rivers Wye and Teme, Wales	Depths ~30 cm		Aprahamian (1982)
	Spawning		Range of European rivers	Depths mostly <150 cm		Cassou-Leins & Cassou-Leins (1981), Dautrey & Lartigue (1983), Bracken & Kennedy (1967), Philippart & Vranken (1982)

Species	Life stage	Time of year	River type	Water depth requirements	Flow requirements	Reference
Alosa fallax fallax	Spawning		Rivers Wye, Usk, Tywi and Teme, Wales	Depths >15 to 120 cm (<45 cm preferred)	Flow characterised as 'rippled flow' or 'unbroken standing waves'	Caswell & Aprahamian (2001)
Anguilla anguilla	Juvenile (<15 cm to >45 cm)	Sept	Frémur basin, northwest France	Depths <600 cm	Velocities >10 $cm.s^{-1}$	Laffaille *et al*. (2003)
Barbatula barbatula	Juvenile	Autumn	River Great Ouse, eastern England	Shallow	Elevated Velocities	Copp (1992a)
	Juvenile (<26 mm)	Aug to Oct	Rivers Great Ouse, Rib, Lee and Hiz, eastern England	Depths 10-20 cm	Little or no flow	Kováč *et al*. (1999)
	Juvenile (26-47 mm)	Aug to Oct	Rivers Great Ouse, Rib, Lee and Hiz, eastern England	Depths 0-10 cm	Weak to medium flow	Kováč *et al*. (1999)
	Juvenile (>47 mm)	Aug to Oct	Rivers Great Ouse, Rib, Lee and Hiz, eastern England	Depths 10-20 cm	Moderate flow	Kováč *et al*. (1999)
Barbus barbus	Larvae	May to Jan	River Sieg, Germany	Depths <20 cm	Low current velocity	Bischoff & Freyhof (1999)
	Larvae	Apr to Sept	Lower River Rhine, The Netherlands	Shallow	Velocities <20 $cm.s^{-1}$	Grift *et al*. (2003)
	Larvae		River Danube, Austria	Depths 0-40 cm		Schiemer *et al*. (2001)
	Juvenile (<30 mm)	Apr to Sept	Lower River Rhine, The Netherlands	Depths <50 cm	Still water	Grift *et al*. (2003)
	Juvenile (>30 mm)	Apr to Sept	Lower River Rhine, The Netherlands	Depths to 100 cm	Velocities up to 30 $cm.s^{-1}$	Grift *et al*. (2003)
	Juvenile (<70 mm)	May to Jan	River Sieg, Germany	Depths <20 cm		Bischoff & Freyhof (1999)
	Juvenile (70-89 mm)	May to Jan	River Sieg, Germany		Velocities up to 120 $cm.s^{-1}$	Bischoff & Freyhof (1999)
	Juvenile	Summer	River Danube, Austria		Velocities <50 $cm.s^{-1}$	Schiemer *et al*. (1991)
	Adult	June to Sept	River Nidd, northeast England		Velocities 40-100 $cm.s^{-1}$	Lucas & Batley (1996)
	Spawning	Apr to July		Depths 30-40 cm		Cowx (2001)

Species	Life stage	Time of year	River type	Water depth requirements	Flow requirements	Reference
Barbus barbus	Spawning	May to June	Rivers Hull (northeast England) and Meuse (Belgium)	Depths 15-40 cm	Velocities 28-43 $cm.s^{-1}$	Hancock (1975), Philippart (1987), Baras (1992)
	Spawning				Velocities 25-49 $cm.s^{-1}$	Mann (1996)
Blicca bjoerkna	Larvae	Apr to Sept	Lower River Rhine, The Netherlands	Depths >50 cm		Grift *et al*. (2003)
	Juvenile	May to Oct	A small abandoned channel (Les Nappes) on the upper River Rhône, France	Depths >100 cm	Still water	Copp (1993)
	Juvenile	Sept	Lower River Saône, France	Depths 50-100 cm		Grenouillet *et al*. (2000)
	Juvenile	Apr to Sept	Lower River Rhine, The Netherlands	Depths <50 cm	Velocities <5 $cm.s^{-1}$	Grift *et al*. (2003)
	Juvenile	Aug to Nov	Lower River Rhône, France	Depths <50 cm		Poizat & Pont (1996)
	Spawning		River Sieg, Germany	Depths 10-25 cm	Riffles with Velocities 5-60 $cm.s^{-1}$ (but thought to be sub-optimal conditions)	Freyhof (1998)
	Spawning		Lowland rivers		Velocities <20 $cm.s^{-1}$	Mann (1996), Cowx & Welcomme (1998)
	Spawning	May to June	Lowland rivers	Depths 60-90 cm		Wheeler (1969), Cowx (2001)
Cobitis taenia	Larvae	July	Haaren Creek, northwest Germany	Depths 25-45 cm	Little or no water movement	Bohlen (2000, 2003)
	Adult	May	Grabia River, Poland	Depths 34.6 cm	Velocities 30 $cm.s^{-1}$	Przybylski *et al*. (2003)
	Adult	All year	River Great Ouse, eastern England		Velocities <15 $cm.s^{-1}$	Robotham (1978)
	Spawning	July	Haaren Creek, northwest Germany	Depths 25-45 cm	No water velocity preference	Bohlen (2003)
Cottus gobio	0+		Zieversbeek brook	Shallow	Riffles	Gubbels (1997)
	Juvenile	Autumn	River Great Ouse, eastern	Shallow	Elevated Velocities	Copp (1992a)

			England			
Species	**Life stage**	**Time of year**	**River type**	**Water depth requirements**	**Flow requirements**	**Reference**
Cottus gobio	Adult		Zieversbeek brook		Velocities 10-38 $cm.s^{-1}$ (average 22 $cm.s^{-1}$)	Gubbels (1997)
	Adult		French lowland stream	Depths 20-40 cm	Velocities >40 $cm.s^{-1}$	Roussel & Bardonnet (1996)
	Adult			Depths >5 cm		Tomlinson & Perrow (2003)
	Spawning	Feb to June		Depths >5 cm		Tomlinson & Perrow (2003)
Cyprinus carpio	0+		Backwaters of the upper Mississippi River, USA	Shallow areas associated with flooded vegetation		Sheaffer & Nickum (1986)
	Spawning	May to July	Lowland rivers	Depths 80-100 cm	Velocities <5 $cm.s^{-1}$	Mann (1996), Cowx & Welcomme (1998), Cowx (2001)
Esox lucius	Larvae	May to Sept	Kyrönjoki River estuary, Finland	Shallow bays (<150 cm deep)		Urho *et al*. (1990)
	Juvenile	May to Oct	A small abandoned channel (Les Nappes) on the upper River Rhône, France	No depth preference	Still water	Copp (1993)
	Juvenile	May to Sept	Kyrönjoki River estuary, Finland	Depths to ~175 cm		Urho *et al*. (1990)
	Spawning	May	Upper St. Lawrence River	Depths 200-500 cm		Farrell (2001)
	Spawning	May	Point Marguerite Marsh, upper St. Lawrence River	Depths 50-260 cm		Farrell *et al*. (1996)
	Spawning	March to May	Lowland rivers	Depths 200-350 cm	Velocities <5 $cm.s^{-1}$	Mann (1996), Cowx (2001)
Gasterosteus aculeatus	Juvenile	Autumn	River Great Ouse, eastern England	Shallow	Elevated Velocities	Copp (1992a)
	Adult	Aug to Nov, March	Numerous locations in the Great Ouse catchment, eastern England	Depths >20 cm	Low water velocity	Copp & Kováč (2003)
Gobio gobio	Larvae	Apr to Sept	Lower River Rhine, The Netherlands	Shallow	Velocities <20 $cm.s^{-1}$	Grift *et al*. (2003)
	Juvenile	Autumn	River Great Ouse, eastern	Depths 20-50 cm	Slow to moderate	Copp (1992a)

Species	Life stage	Time of year	River type	Water depth requirements	Flow requirements	Reference
			England		Velocities	
Gobio gobio	Juvenile	June to Sept	A braided side-channel of the upper River Rhône, France	Depths <20 cm	Lentic waters	Copp (1992b)
	Juvenile	May to Oct	River Great Ouse, eastern England	Depths <100 cm	Velocities <5 $cm.s^{-1}$	Garner (1996b)
	Juvenile	Apr to Sept	Lower River Rhine, The Netherlands	Depths <100 cm	Velocities 0-40 $cm.s^{-1}$	Grift *et al*. (2003)
	Juvenile	Aug to Nov	Lower River Rhône, France	Depths <50 cm		Poizat & Pont (1996)
	Adult		Lowland rivers		Velocities <55 $cm.s^{-1}$	Mann (1996)
	Spawning	June	A small rivulet entering the Inniscarra Reservoir, Ireland	Depths 5-8 cm		Kennedy & Fitzmaurice (1972)
	Spawning		Lowland rivers		Velocities 2-80 $cm.s^{-1}$	Mann (1996), Cowx & Welcomme (1998)
Gymnocephalus cernuus	Larvae	Apr to July	St. Louis River, USA	Water depth 50 cm		Brown *et al*. (1998)
	Adult		Lowland rivers		Lentic habitats	Kováč (1998)
Lampetra fluviatilis	Larvae			Depths 0-100 cm, typically 10-50 cm		Entec (2000a, b)
	Larvae				Velocities 1-50 $cm.s^{-1}$, usually 8-10 $cm.s^{-1}$	Hardisty (1986)
	Larvae				Velocities 1-50 $cm.s^{-1}$	Kainua & Valtonen (1980)
	Larvae			Depths <50 cm		Maitland (2003)
	Spawning	March to Apr		Depths 20-150 cm	Velocities 100-200 $cm.s^{-1}$	Maitland (2003)
Lampetra planeri	Larvae				Velocities usually 8-10 $cm.s^{-1}$	Hardisty (1986)
	Larvae			Depths <50 cm		Maitland (2003)
	Spawning			Depths 3-30 cm		Hardisty (1986)
	Spawning			Depths <40 cm	Velocities 30-50 $cm.s^{-1}$	Hardisty & Potter (1971)

Species	Life stage	Time of year	River type	Water depth requirements	Flow requirements	Reference
	Spawning	March to June		Depths 20-150 cm		Maitland (2003)
Species	**Life stage**	**Time of year**	**River type**	**Water depth requirements**	**Flow requirements**	**Reference**
Leuciscus cephalus	Larvae	June to Sept	A braided side-channel of the upper River Rhône, France	Depths 20-50 cm	Lentic waters	Copp (1992b)
	Larvae	May to Oct	River Great Ouse, eastern England	Depths <100 cm	Velocities <5 $cm.s^{-1}$	Garner (1996b)
	Juvenile	Nov to Jan	River Ourthe, Belgium		Mean Velocities <2 $cm.s^{-1}$	Baras & Nindaba (1999)
	Juvenile	Autumn	River Great Ouse, eastern England	Moderate Depths	Slow to moderate Velocities	Copp (1992a)
	Juvenile	June to Sept	A braided side-channel of the upper River Rhône, France	Depths <20 cm	Lentic waters	Copp (1992b)
	Juvenile	June, July, Sept	River Great Ouse, eastern England	Depths <100 cm (day and night)		Garner (1996a)
	Juvenile	May to Oct	River Great Ouse, eastern England	Depths <100 cm	Velocities <5 $cm.s^{-1}$	Garner (1996b)
	Juvenile	Sept	Lower River Saône, France	Depths <100 cm		Grenouillet *et al*. (2000)
	Juvenile	Aug to Nov	Lower River Rhône, France	Depths <50 cm		Poizat & Pont (1996)
	Spawning	May to June	A lowland canal, Germany	Depths >0-128 cm	Velocities <5 $cm.s^{-1}$	Arlinghaus & Wolter (2003)
	Spawning		Lowland rivers	Depths 10-30 cm	Velocities 15-75 $cm.s^{-1}$	Cowx & Welcomme (1998)
	Spawning	May to June	River Spree, Germany	Depths 10-80 cm	Velocities 40 $cm.s^{-1}$	Fredrich *et al*. (2003)
	Spawning		Lowland rivers		Velocities 20-50 $cm.s^{-1}$	Mann (1996)
Leuciscus leuciscus	Larvae (newly-hatched)		River Frome, southern England		Velocities <2 $cm.s^{-1}$	Mann & Mills (1986), Mills (1991)
	Larvae	June to	A braided side-channel of	Depths 20-50 cm	Lentic waters	Copp (1992b)

		Sept	the upper River Rhône, France			
Species	**Life stage**	**Time of year**	**River type**	**Water depth requirements**	**Flow requirements**	**Reference**
Leuciscus leuciscus	Larvae	Apr to May	River Frome, southern England	Depths 2-40 cm	Velocities 0-2.5 $cm.s^{-1}$	Mills *et al*. (1985)
	Juvenile	Nov to Jan	River Ourthe, Belgium		Mean Velocities <2 $cm.s^{-1}$	Baras & Nindaba (1999)
	Juvenile	Autumn	River Great Ouse, eastern England	Shallow	Elevated Velocities	Copp (1992a)
	Juvenile	June to Sept	A braided side-channel of the upper River Rhône, France	Depths <20 cm (but 20-50 cm during low discharge conditions)	Lentic waters	Copp (1992b)
	Adult	March to Apr	River Frome, southern England	Depths 17-113 cm (mean 62 cm)	Velocities 0-57 $cm.s^{-1}$ (mean 6 $cm.s^{-1}$)	Clough *et al*. (1998)
	Spawning			Depths 25-40 cm		Cowx & Welcomme (1998)
	Spawning	March	River Dalua, southern Ireland	Water depth 25-40 cm		Kennedy (1969)
	Spawning		River Frome, southern England		Velocities 20-50 $cm.s^{-1}$	Mann (1996), Cowx & Welcomme (1998)
	Spawning	March	River Frome, southern England		Velocities ~30 $cm.s^{-1}$	Mills (1981a)
Osmerus eperlanus	Larvae	May to Sept	Kyrönjoki River estuary, Finland	Depths to ~250 cm	Water currents are used to disperse the larvae	Urho *et al*. (1990), Urho (1992)
	Juvenile	May to Sept	Kyrönjoki River estuary, Finland	Depths to ~250 cm		Urho *et al*. (1990)
	Spawning	Feb to March	River Cree, Scotland		Turbulent water at the base of riffles	Lyle & Maitland (1997)
	Spawning	Apr to June	Rivers, streams and estuaries of the northern Baltic Sea	Shallow		Urho (1992)
Perca fluviatilis	Larvae	May to Sept	Kyrönjoki River estuary, Finland	Shallow bays (<150 cm deep)		Urho *et al*. (1990)

	Juvenile	Autumn	River Great Ouse, eastern England	Moderate depth	Slow flowing or lentic	Copp (1992a)
Species	**Life stage**	**Time of year**	**River type**	**Water depth requirements**	**Flow requirements**	**Reference**
Perca fluviatilis	Juvenile	May to Sept	Kyrönjoki River estuary, Finland	Depths to ~300 cm		Urho *et al*. (1990)
	Spawning	Apr to May	Lowland rivers	Depths 200-300 cm		Cowx (2001)
Petromyzon marinus	Larvae		River Mondego, Portugal		Velocities 0-17 $cm.s^{-1}$ (range), 10 $cm.s^{-1}$ (mean)	Almeida & Quintella (2002)
	Larvae		River Eamont, northwest England	Depths 0-100 cm, typically 10-50 cm		Entec (2000a, b)
	Larvae			Depths typically <50 cm, sometimes up to 220 cm		Maitland (2003)
	Larvae		Great Lakes watershed, USA		Velocities usually =3 $cm.s^{-1}$ (<60-80 $cm.s^{-1}$ required to enable ammocoetes to burrow)	Thomas (1962)
	Spawning			Depths 13-170 cm (usually 23-51 cm)	Velocities 39.6-158.5 $cm.s^{-1}$	Applegate (1950)
	Spawning				Velocities 30 $cm.s^{-1}$	Beamish (1974)
	Spawning			Depths usually 40-60 cm	Velocities 100-200 $cm.s^{-1}$	Hardisty (1986)
	Spawning	May to June		Depths 20-150 cm		Maitland (2003)
Phoxinus phoxinus	Larvae		Upland and lowland rivers	Depths to >15 cm		Mann (1996)
	Larvae	July	River Frome, southern England	Mean water depth 40.5 cm	Mean water velocity 1.9 $cm.s^{-1}$	Simonović *et al*. (1999)
	Larvae	May to Oct	River Lee, eastern England	Mean water depth 26.86 cm	Mean water velocity 3.46 $cm.s^{-1}$	Simonović *et al*. (1999)
	Juvenile	Autumn	River Great Ouse, eastern England	Shallow	Elevated Velocities	Copp (1992a)

	Juvenile	July	River Frome, southern England	Mean water depth 53.4 cm	Mean water velocity 12.8 $cm.s^{-1}$	Simonović *et al*. (1999)
Species	**Life stage**	**Time of year**	**River type**	**Water depth requirements**	**Flow requirements**	**Reference**
Phoxinus phoxinus	Juvenile	May to Oct	River Lee, eastern England	Mean water depth 34.7 cm	Mean water velocity 3.85 $cm.s^{-1}$	Simonović *et al*. (1999)
	Adult		River Frome, southern England		Velocities 0-10 $cm.s^{-1}$	Garner (1997a)
	Adult	July	River Frome, southern England	Depths 10 to >50 cm		Garner *et al*. (1998)
	Adult	July	River Frome, southern England	Mean water depth 45 cm	Mean water velocity 35.9 $cm.s^{-1}$	Simonović *et al*. (1999)
	Adult	May to Oct	River Lee, eastern England	Mean water depth 36.61 cm	Mean water velocity 6.22 $cm.s^{-1}$	Simonović *et al*. (1999)
	Spawning		Upland and lowland rivers	Depths 10-25 cm	Velocities 20-30 $cm.s^{-1}$	Mann (1996), Cowx & Welcomme (1998)
Pungitius pungitius	Juvenile	Autumn	River Great Ouse, eastern England	Shallow	Elevated Velocities	Copp (1992a)
	Adult	Aug to Nov, March	Numerous locations in the Great Ouse catchment, eastern England	Depths >20 cm	Low water velocity	Copp & Kováč (2003)
	Adult	All year	St. Ippollittis Brook (Great Ouse catchment), eastern England		Areas of dense bank-side vegetation adjacent to faster flowing (10 $cm.s^{-1}$) water	Copp *et al*. (2002)
Rhodeus sericeus	0+	Aug	Lower River Morava, Czech Republic	Depths <25 cm	Velocities <10 $cm.s^{-1}$	Reichard *et al*. (2002)
	1+	Aug	Lower River Morava, Czech Republic	Depths 10-40 cm	Velocities 10-50 $cm.s^{-1}$	Reichard *et al*. (2002)
Rutilus rutilus	Larvae (newly-hatched)	June to July	River Hull, northeast England		Velocities <2 $cm.s^{-1}$	Lightfoot & Jones (1996)
	Larvae (young)	June to Sept	A braided side-channel of the upper River Rhône,	Depths 50-100 cm	Lentic waters	Copp (1992b)

			France			
Species	**Life stage**	**Time of year**	**River type**	**Water depth requirements**	**Flow requirements**	**Reference**
	Larvae (steps 3 to 5)	May to Sept	Upper River Rhône, France	Depths 50-100 cm	Lentic waters	Copp (1990)
Rutilus rutilus	Larvae (step 6)	May to Sept	Upper River Rhône, France	Depths 20-50 cm	Lentic waters	Copp (1990)
	Larvae (old)	June to Sept	A braided side-channel of the upper River Rhône, France	Depths 20-50 cm	Lentic waters	Copp (1992b)
	Larvae (old)	May to Oct	A small abandoned channel (Les Nappes) on the upper River Rhône, France	Depths 20-50 cm	Still water	Copp (1993)
	Larvae		Lowland rivers	Water depth 150 cm	Velocities 0.5-1 $cm.s^{-1}$	Cowx & Welcomme (1998)
	Larvae	May to Oct	River Great Ouse, eastern England	Depths <100 cm	Velocities <5 $cm.s^{-1}$	Garner (1996b)
	Larvae	Apr to Sept	Lower River Rhine, The Netherlands	Depths >50 cm	Still water	Grift *et al*. (2003)
	Larvae	May to Sept	Kyrönjoki River estuary, Finland	Shallow bays (<150 cm deep)		Urho *et al*. (1990)
	Juvenile	May to Sept	A braided side-channel of the upper River Rhône, France	Depths 20-50 cm	Lentic waters	Copp (1990, 1992b)
	Juvenile	Autumn	River Great Ouse, eastern England	Moderate depth	Slow flowing or lentic	Copp (1992a)
	Juvenile	May to Oct	A small abandoned channel (Les Nappes) on the upper River Rhône, France	Depths >100 cm	Still water	Copp (1993)
	Juvenile	Aug to Sept	River Great Ouse, eastern England	Depths 100 cm	Negligible water velocity	Garner (1995)
	Juvenile	May to Oct	River Great Ouse, eastern England	Depths <100 cm	Velocities <5 $cm.s^{-1}$	Garner (1996b)
	Juvenile	Apr to Sept	Lower River Rhine, The Netherlands	Depths <100 cm	Velocities 0-40 $cm.s^{-1}$	Grift *et al*. (2003)

	Juvenile	May to Sept	Kyrönjoki River estuary, Finland	Depths to ~175 cm		Urho *et al*. (1990)
Species	**Life stage**	**Time of year**	**River type**	**Water depth requirements**	**Flow requirements**	**Reference**
	Spawning		Lowland rivers	Depths 15-45 cm	Velocities to >20 $cm.s^{-1}$	Mann (1996), Cowx & Welcomme (1998), Cowx (2001)
Rutilus rutilus	Spawning		River Frome, southern England	Depths 15-30 cm	Velocities to >20 $cm.s^{-1}$	Mills (1981b)
Salmo salar	Fry		Streams	Maximum Depths <10 cm	Minimum Velocities >5-15 $cm.s^{-1}$	Heggenes *et al*. (1999)
	Fry			Depths =20 cm	Velocities 50-65 $cm.s^{-1}$	Hendry & Cragg-Hine (1997)
	Fry			Depths <20 cm	Velocities 25-40 $cm.s^{-1}$	Hendry & Cragg-Hine (2003)
	Fry	Summer	Lower River Bush, Northern Ireland	Mean Depths <20 cm		Kennedy & Strange (1982)
	Fry		Rivers in Nova Scotia and New Brunswick	Depths 20-40 cm	Snout Velocities 5-15 $cm.s^{-1}$	Morantz *et al*. (1987)
	0+		Northern French streams	Depths <23 cm	Water velocity 61 $cm.s^{-1}$	Baglinière & Arribe-Moutounet (1985)
	0+			Depths <25 cm	Mean Velocities 20-40 $cm.s^{-1}$	Crisp (1993, 1996)
	0+	Sept to Oct	Shelligan Burn, Scotland	Depths 0-19 cm		Egglishaw & Shackley (1982)
	0+		Canadian streams	Depths =50 cm		Keenleyside (1962)
	0+	Summer	Lower River Bush, Northern Ireland	Mean Depths <20 cm		Kennedy & Strange (1982)
	0+		Rivers in Nova Scotia and New Brunswick	Depths <25 cm (preferred) and <100 cm (maximum)	Snout Velocities 5-15 $cm.s^{-1}$	Morantz *et al*. (1987)
	0+	Summer	Little Sevogle River, New Brunswick	Depths 24-36 cm	Focal Velocities 10-30 $cm.s^{-1}$	Rimmer *et al*. (1984)

	0+ (<7 cm)		Canadian rivers	Depths 10-15 cm	Mean Velocities 50-65 $cm.s^{-1}$	Symons & Heland (1978)
Species	**Life stage**	**Time of year**	**River type**	**Water depth requirements**	**Flow requirements**	**Reference**
	0+ (>7 cm)		Canadian rivers	Depths 30 cm	Mean Velocities 50-65 $cm.s^{-1}$	Symons & Heland (1978)
Salmo salar	=0+	Nov to March (at night)	Rock River, Vermont, USA		Velocities =19 $cm.s^{-1}$	Whalen & Parrish (1999)
	Young	June, Aug, Oct, Nov	Norwegian river	Depths 30-100 cm	Velocities 10-50 $cm.s^{-1}$	Heggenes & Saltveit (1990)
	Juvenile		Upland and lowland rivers		Velocities 50-60 $cm.s^{-1}$	Crisp (1996, 2000)
	Juvenile	Winter		Depths 40.9-48.9 cm		Cunjak (1988)
	Juvenile		Newfoundland rivers		Velocities 10-30 $cm.s^{-1}$	DeGraaf & Bain (1986)
	Juvenile		Streams	Depths 5-65 cm (range), <25 cm (preferred) and <100 cm (maximum)	Maximum Velocities <100 $cm.s^{-1}$	Heggenes (1990)
	Juvenile	Aug to Sept	Rivers in New Brunswick		Snout Velocities 10-30 $cm.s^{-1}$	Rimmer *et al*. (1984)
	Juvenile		Rivers in northwest England		Flow =0.03 $m^3.s^{-1}$ per metre of channel width	Stewart (1973)
	Juvenile (<7 cm)		Dartmoor, upland area in southwest England		Snout Velocities 0-53 $cm.s^{-1}$ (range) and 4.2 $cm.s^{-1}$ (mean). Most fishes selected snout	Heggenes *et al*. (2002)

					velocities 0-5 $cm.s^{-1}$	

Species	Life stage	Time of year	River type	Water depth requirements	Flow requirements	Reference
	Juvenile (=7 cm)		Dartmoor, upland area in southwest England		Snout Velocities 0-51 $cm.s^{-1}$ (range) and 6.2 $cm.s^{-1}$ (mean). Most fishes selected snout velocities 0-5 $cm.s^{-1}$	Heggenes *et al*. (2002)
Salmo salar	Juvenile (105-165 mm)	Summer and autumn	West Salmon River, Newfoundland	Depths 20 cm (low flow conditions) and 40 cm (high flow conditions)	Velocities 10-20 $cm.s^{-1}$	Scruton *et al*. (2002)
	1+	Sept to Oct	Shelligan Burn, Scotland	Depths >15 cm		Egglishaw & Shackley (1982)
	1+	Summer	Little Sevogle River, New Brunswick	Depths 24-36 cm	Focal Velocities 10-40 $cm.s^{-1}$	Rimmer *et al*. (1984)
	=1+		Northern French streams	Depths >27 cm	Velocities <28 $cm.s^{-1}$	Baglinière & Arribe-Moutounet (1985)
	Parr	Aug to Nov	Todalselva and Vindøla Rivers, central Norway	Depths >50 cm	Velocities 4-10 $cm.s^{-1}$	Bremset (2000)
	Parr		Streams	Depths 20-70 cm	Velocities 10-65 $cm.s^{-1}$	Heggenes (1990)
	Parr		Norwegian stream	Depths >10 and <60 cm	Velocities =10 $cm.s^{-1}$	Heggenes (1991)
	Parr		Streams	Maximum Depths <100 cm	Velocities >5-15 (minimum), >60 (maximum) and 0-20 $cm.s^{-1}$ (range of snout Velocities)	Heggenes *et al*. (1999)
	Parr			Depths 20-40 cm	Velocities 60-70 $cm.s^{-1}$	Hendry & Cragg-Hine (1997)
	Parr			Depths 20-40 cm	Velocities 25-40 $cm.s^{-1}$	Hendry & Cragg-Hine (2003)
	Parr			Depths 25-60 cm	Maximum Velocities	Morantz *et al*. (1987)

					<120 $cm.s^{-1}$, snout Velocities 5-35 $cm.s^{-1}$	

Species	Life stage	Time of year	River type	Water depth requirements	Flow requirements	Reference
	Parr		Rivers in New Brunswick	Depths 25-60 cm	Snout Velocities 10-50 $cm.s^{-1}$	Rimmer *et al*. (1984)
	Parr		Canadian rivers	Depths 25-60 cm	Mean Velocities 50-65 $cm.s^{-1}$	Symons & Heland (1978)
	2+	Summer	Little Sevogle River, New Brunswick	Depths 24-36 cm	Focal Velocities 30-50 $cm.s^{-1}$	Rimmer *et al*. (1984)
Salmo salar	=2+	Autumn	Little Sevogle River, New Brunswick	Depths 24-36 cm	Focal water velocity <10 $cm.s^{-1}$	Rimmer *et al*. (1984)
	Adult			Pools of at least 150 cm depth as holding areas		Hendry & Cragg-Hine (2003)
	Spawning		Canadian river	Depths 17-76 cm (mean 38 cm)	Velocities 35-80 $cm.s^{-1}$ (mean 53 $cm.s^{-1}$)	Beland *et al*. (1982)
	Spawning		Upland and lowland rivers		Between 30 and 50% of the average daily flow (ADF) in the lower and middle reaches of rivers (50-70% for large spring salmon) and >70% ADF in headwater streams for fish to move upstream. Upstream movement begins when flows reach 0.08 $m^3.s^{-1}.m^{-1}$ and peaks at 0.2 $m^3.s^{-1}.m^{-1}$ (i.e. discharge per metre of river width)	Crisp (1996), Cowx & Welcomme (1998)

	Spawning	Oct to Nov	Upland spate river, northern England		Minimum Velocities >15-20 $cm.s^{-1}$	Crisp & Carling (1989)

Species	Life stage	Time of year	River type	Water depth requirements	Flow requirements	Reference
	Spawning			Depths 15-90 cm	Velocities 20-80 $cm.s^{-1}$	Fraser (1975)
	Spawning		Norwegian river	Mean water depth 50 cm	Mean water velocity 40 $cm.s^{-1}$	Heggberget (1991)
	Spawning			Depths 17-76 cm	Velocities 25-90 $cm.s-^{1}$	Hendry & Cragg-Hine (1997)
Salmo salar	Spawning			Depths 15-75 cm	Velocities 50-90 $cm.s^{-1}$	Hendry & Cragg-Hine (2003)
	Spawning			Depths 15-91 cm (range), 24 cm (minimum) and 30-45 cm (optimum)	Velocities 20-81 $cm.s^{-1}$	Jones & King (1950), Smith (1973)
	Spawning		Scottish river	Mean water depth 25 cm	Mean water velocity 53 $cm.s^{-1}$	Moir *et al*. (1998)
	Spawning		Hampshire Avon, southern England		River discharge >9 $m^3.s^{-1}$ in the estuary for adults to enter the river	Solomon *et al*. (1999)
	Spawning		Rivers in southwest England		Between 101 and 284% of the Q95 (the flow exceeded for 95% of the time) needed to induce upstream migration	Solomon *et al*. (1999)
	Spawning	Nov to Dec	A lowland agricultural stream (Newmills Burn, Scotland)	Mean water depth 25.6 cm	Mean water velocity 51.8 $cm.s^{-1}$	Soulsby *et al*. (2001)
	Spawning		A range of rivers in northwest England		Flow =0.084 $m^3.s^{-1}$ per metre of channel width for salmon to	Stewart (1973)

					commence upstream migration	
Salmo trutta	Fry				Velocities 0-20 $cm.s^{-1}$	Bardonnet & Heland (1994)
Species	**Life stage**	**Time of year**	**River type**	**Water depth requirements**	**Flow requirements**	**Reference**
	Fry	Summer	Small streams of the Kings River basin, California	Depths <60 cm	Velocities <30 $cm.s^{-1}$	Lambert & Hanson (1989)
	0+	Summer	Lower River Bush, Northern Ireland	Mean Depths <20-30 cm		Kennedy & Strange (1982)
	0+			Depths <30 cm	Velocities 20-50 $cm.s^{-1}$	Roussel & Bardonnet (1999)
Salmo trutta	0+ (20-40 mm)		River Vojmån, northern Sweden		Velocities <10 $cm.s^{-1}$	Greenberg *et al*. (1996)
	Young	June, Aug, Oct, Nov	Norwegian river	Depths 30-100 cm	Velocities 10-30 $cm.s^{-1}$	Heggenes & Saltveit (1990)
	Juvenile			Depths <20-30 cm		Bardonnet & Heland (1994)
	Juvenile			Water depth <20-30 cm		Bohlin (1977)
	Juvenile				Velocities =25 $cm.s^{-1}$	Crisp (1996, 2000)
	Juvenile	Summer	Small streams of the Kings River basin, California	Depths to 240 cm	Velocities <30 $cm.s^{-1}$	Lambert & Hanson (1989)
	Juvenile		River in Finland	Depths 5-35 cm		Mäki-Petäys *et al*. (1997)
	Juvenile (<7 cm)	Aug to Sept	Dartmoor, upland area in southwest England		Snout Velocities 0-21 $cm.s^{-1}$ (range) and 4 $cm.s^{-1}$ (mean). Most fishes selected snout velocities 0-5 $cm.s^{-1}$	Heggenes *et al*. (2002)
	Juvenile (=7 cm)	Aug to Sept	Dartmoor, upland area in southwest England		Snout Velocities 0-44 $cm.s^{-1}$ (range) and 7 $cm.s^{-1}$ (mean). Most fishes selected snout	Heggenes *et al*. (2002)

					velocities 0-5 $cm.s^{-1}$	
	1+	Sept to Oct	Shelligan Burn, Scotland	Depths >25 cm		Egglishaw & Shackley (1982)
Species	**Life stage**	**Time of year**	**River type**	**Water depth requirements**	**Flow requirements**	**Reference**
	Parr (0+)		Streams		Velocities 20-50 $cm.s^{-1}$	Crisp (1993), Heggenes (1996)
	Parr		A stream		Snout Velocities <20 $cm.s^{-1}$	Bachman (1984)
	Parr			Minimum Depths <5.1 cm	Snout Velocities <20 $cm.s^{-1}$	Baldes & Vincent (1969)
	Parr		A southern UK chalk stream		Snout Velocities <20 $cm.s^{-1}$	Bird *et al*. (1995)
Salmo trutta	Parr	Aug to Sept	Todalselva and Vindøla Rivers, central Norway	Depths >50 to <300 cm	Velocities <50 $cm.s^{-1}$	Bremset (2000)
	Parr	Nov	Todalselva and Vindøla Rivers, central Norway	Depths >50 to <300 cm	Velocities <30 $cm.s^{-1}$	Bremset (2000)
	Parr		River in Finland	Depths 40-75 cm		Mäki-Petäys *et al*. (1997)
	Parr		Six New Zealand rivers	Depths 14-122 cm (range) and 65 cm (mean)	Velocities 0-65 (range), 26.7 (mean) and <20 $cm.s^{-1}$ (snout water velocity)	Shirvell & Dungey (1983)
	Smolt		River Stjørdalselva, mid-Norway		Maximum number of smolts caught when discharge was 70-150 $m^3.s^{-1}$. Few smolts descended at low discharges (<50 $m^3.s^{-1}$)	Hembre *et al*. (2001)
	Adult		Southeast Norwegian stream	Depths >50 cm	Velocities 10-70 $cm.s^{-1}$	Heggenes (1988)
	Adult	Summer	Norwegian and Scottish streams	Depths 9-305 cm (range) and 69 cm (mean)	Velocities 0-142 (range of water column velocities),	Heggenes (2002)

					24 (mean) and 14 $cm.s^{-1}$ (mean focal water velocity)	
Species	**Life stage**	**Time of year**	**River type**	**Water depth requirements**	**Flow requirements**	**Reference**
	Adult		River in Finland	Depths 40-75 cm		Mäki-Petäys *et al*. (1997)
	Adult		Six New Zealand rivers	Depths 14-122 cm (mean 65 cm)	Velocities 0-65 $cm.s^{-1}$ (mean 26.7 $cm.s^{-1}$)	Shirvell & Dungey (1983)
	Adult (>200 mm)	Summer	Small streams of the Kings River basin, California	Depths >60 cm		Lambert & Hanson (1989)
Salmo trutta	Spawning		Upland and lowland rivers		Between 20 and 25% of the average daily flow (ADF) in the lower and middle reaches of rivers and 25-30% ADF in headwater streams for fish to move upstream. Upstream movement begins when flows reach 0.08 $m^3.s^{-1}.m^{-1}$ and peaks at 0.2 $m^3.s^{-1}.m^{-1}$ (i.e. discharge per metre of river width)	Crisp (1996), Cowx & Welcomme (1998)
	Spawning	Oct to Nov	Upland spate river, northern England	Variable water depth, but not less than body depth	Velocities 15-20 $cm.s^{-1}$, up to twice female body length in $cm.s^{-1}$	Crisp & Carling (1989)
	Spawning		Norwegian river	Water depth ~50 cm	Mean water velocity 27 $cm.s^{-1}$	Heggberget (1991)
	Spawning			Depths 15-91 cm (range), 24 cm (minimum) and 30-45 cm (optimum)	Velocities 20-81 $cm.s^{-1}$	Jones & King (1950), Smith (1973)

Species	Life stage	Time of year	River type	Water depth requirements	Flow requirements	Reference
	Spawning	Apr to Dec	River Imsa, Norway		Maximum ascent of spawners when discharge was 7.5-10 $m^3.s^{-1}$. No fish ascended when discharge exceeded 20 $m^3.s^{-1}$	Jonsson & Jonsson (2002)
Salmo trutta	Spawning		Six New Zealand rivers	Depths 6-82 cm (mean 31.7 cm)	Velocities 15-75 $cm.s^{-1}$ (mean 39.4 $cm.s^{-1}$)	Shirvell & Dungey (1983)
	Spawning	Nov to Dec	A lowland agricultural stream (Newmills Burn, Scotland)	Mean water depth 25.6 cm	Mean water velocity 51.8 $cm.s^{-1}$	Soulsby *et al*. (2001)
	Spawning		Southwest Ontario streams	Mean water depth 25.5 cm	Velocities 10.8-80.2 $cm.s^{-1}$ (mean 46.7 $cm.s^{-1}$)	Witzel & MacCrimmon (1983)
Sander lucioperca	Larvae	Apr to Sept	Lower River Rhine, The Netherlands	Depths >50 cm		Grift *et al*. (2003)
	Larvae	May to Sept	Kyrönjoki River estuary, Finland	Depths to ~250 cm		Urho *et al*. (1990)
	Juvenile	May to Sept	Kyrönjoki River estuary, Finland	Depths to ~250 cm		Urho *et al*. (1990)
	Adult	Sept to Dec	River Gudenaa, Denmark	Depths >200 cm		Koed *et al*. (2000)
	Adult	Summer	Pyhakoski Reservoir, Finland	Depths 7-15 m	Velocities 0.01-0.86 $cm.s^{-1}$ (mostly <0.3 $cm.s^{-1}$)	Vehanen & Lahti (2003)
	Adult	Winter	Pyhakoski Reservoir, Finland	Depths 21-23 m	Velocities 0.01-0.86 $cm.s^{-1}$ (mostly <0.3 $cm.s^{-1}$)	Vehanen & Lahti (2003)
	Adult	All year	Pyhakoski Reservoir,	Depths 1.2-38 m	Velocities 0.01-0.86	Vehanen & Lahti (2003)

Species	Life stage	Time of year	River type	Water depth requirements	Flow requirements	Reference
			Finland	(range)	cm.s^{-1} (mostly <0.3 cm.s^{-1})	
	Spawning				Velocities =20 cm.s^{-1} preferred	Balon *et al*. (1977)
	Spawning	Apr to June	Lowland rivers	Depths 50-100 cm	Velocities 10-20 cm.s^{-1}	Deelder & Willemsen (1964), Cowx (2001)
	Spawning		River Gudenaa, Denmark		Velocities >70 cm.s^{-1} and turbulent (but thought to be sub-optimal conditions)	Koed *et al*. (2000)
Sander lucioperca	Spawning	Feb to June	Lowland rivers	Depths 100-300 cm		Lappalainen *et al*. (2003)
Scardinius erythrophthalmus	Larvae	May to Oct	A small abandoned channel (Les Nappes) on the upper River Rhône, France	Variety of Depths	Still water	Copp (1993)
	Juvenile	May to Oct	A small abandoned channel (Les Nappes) on the upper River Rhône, France	Depths >100 cm	Still water	Copp (1993)
	Spawning		Lowland rivers		Velocities <5 cm.s^{-1}	Mann (1996)
	Spawning	May to July		Depths 10-90 cm		Svärdson (1949), Cowx (2001)
Thymallus thymallus	Larvae (17-21 mm)	June	River Kuusinkijoki, northern Finland	Depths 10-30 cm	Velocities <10 cm.s^{-1}	Nykänen & Huusko (2003)
	Larvae (22-25 mm)	June	River Kuusinkijoki, northern Finland	Depths 30-90 cm	Velocities <10 cm.s^{-1}	Nykänen & Huusko (2003)
	Larvae (26-31 mm)	June	River Kuusinkijoki, northern Finland	Depths >50 cm	Velocities 10-50 cm.s^{-1}	Nykänen & Huusko (2003)
	Larvae	Apr to May			Velocities <15 cm.s^{-1}	Bardonnet *et al*. (1991)
	Larvae	Apr to May	River Frome, southern England		Velocities 6-25 cm.s^{-1} (3-9 body lengths.s^{-1})	Scott (1985)

Species	Life stage	Time of year	River type	Water depth requirements	Flow requirements	Reference
	Larvae	Apr to June	River Pollon, France	Depths <40 cm	Velocities <20 cm.s^{-1}	Sempeski & Gaudin (1995b)
	0+ (20-60 mm)		River Vojmån, northern Sweden		Velocities <10 cm.s^{-1}	Greenberg *et al*. (1996)
	0+		Ain river, France (Rhône catchment)	Depths 50-60 cm	Velocities 70-110 cm.s^{-1}	Mallet *et al*. (2000)
	Juvenile				Velocities <20-50 cm.s^{-1}	Janković (1964)
	Juvenile	Apr to June	River Pollon, France	Depths 40-60 cm	Velocities 20-40 cm.s^{-1}	Sempeski & Gaudin (1995b)
	1+		Ain river, France (Rhône catchment)	Depths 80-120 cm	Velocities 70-110 cm.s^{-1}	Mallet *et al*. (2000)
Thymallus thymallus	Adult	Aug to Sept	River Kemijoki, northern Finland	Depths 100-325 cm	Velocities 30-110 cm.s^{-1}	Nykänen *et al*. (2001)
	Adult	Oct	River Kemijoki, northern Finland	Depths 150-400 cm	Velocities 20-80 cm.s^{-1}	Nykänen *et al*. (2001)
	Adult	Late summer	River Kuusinkijoki, northeast Finland	Depths 80-120 cm	Velocities >40 cm.s^{-1}	Nykänen *et al*. (2004)
	Adult	Autumn	River Kuusinkijoki, northeast Finland	Depths 100-240 cm	Velocities <30 cm.s^{-1}	Nykänen *et al*. (2004)
	Adult	June to July	Oulujoki River, northern Finland	Depths 20-155 cm	Velocities 20-45 cm.s^{-1}	Vehanen *et al*. (2003)
	Adult (=2+)		Ain river, France (Rhône catchment)	Depths 100-140 cm	Velocities 70-110 cm.s^{-1}	Mallet *et al*. (2000)
	Spawning	May to June	Hegledbäcken and Svartbäcken creeks, Sweden	Depths <fish body depth to 25 cm		Fabricius & Gustafson (1955)
	Spawning		Rivers Indalsälven and Ammerån, Sweden	Depths 30-50 cm (mean 36 cm)	Velocities 23-90 cm.s^{-1} (mean 54 cm.s^{-1})	Gönczi (1989)
	Spawning			Depths ~50 cm		Janković (1964)
	Spawning			Depths 20-40 cm	Velocities 40-70	Müller (1961)

					$cm.s^{-1}$	
	Spawning	May	River Kuusinkijoki, northeast Finland	Optimum Depths 30-40 cm	Velocities 50-60 $cm.s^{-1}$	Nykänen & Huusko (2002)
Species	**Life stage**	**Time of year**	**River type**	**Water depth requirements**	**Flow requirements**	**Reference**
	Spawning		Rivers Pollon and Suran, France	Depths 10-40 cm	Velocities 25.8-91.7 $cm.s^{-1}$ (mean 48.9 $cm.s^{-1}$)	Sempeski & Gaudin (1995a)
Tinca tinca	Larvae	May to Oct	A small abandoned channel (Les Nappes) on the upper River Rhône, France	No depth preference	Still water	Copp (1993)
	Juvenile	May to Oct	A small abandoned channel (Les Nappes) on the upper River Rhône, France	No depth preference	Still water	Copp (1993)
	Spawning		Lowland rivers		Velocities <20 $cm.s^{-1}$	Cowx & Welcomme (1998)
	Spawning		Lowland rivers		Velocities <5 $cm.s^{-1}$	Mann (1996)

Table A2.2 Critical and burst swimming speeds of freshwater fishes (BL = Body length, TL = Total length, T = Temperature)

Species	Length (mm)	Swimming performance U_{crit} cm s^{-1}	U_{crit} BL s^{-1}	U_{max} cm s^{-1}	U_{max} BL s^{-1}	Time (s)	Temp (°C)	Reference
Abramis bjoerkna	22			30	15			Blaxter 1969[*]
Abramis brama	120/280	87/96					18	Ohlmer & Schwartzkopff, 1959
Alburnus alburnus	19-34	31-53[c]						Pavlov, 1989
Anguilla anguilla	600			114	1.9	2	10	Blaxter & Dickson, 1959
Anguilla anguilla	72	54	7.5			18	11-13	McCleave, 1980
Anguilla anguilla	170-450	-0.82+0.45 ln (TL)				120-180	17	Sprengel & Lüchtenberg, 1991
Anguilla anguilla	220	31.8	1.58				18-22	D'Aoû & Aerts, 1999
Barbatula barbatula	91	53.0/60.8	5.6/5.8		14.2	600	4/20	Stahlberg, 1986
Barbatula barbatula	110	60.8	5.5			180	18	Stahlberg & Peckmann, 1987
*Barbatula barbatul*a	22-42	22.5-38[c]					18-22	Pavlov *et al*., 1972
Barbus barbus	19-30		11.5			180	15-16	Cowx & Welcomme, 1998
Carassius auratus	90			138	15.1	2	10	Blaxter & Dickson, 1959
Carassius auratus	67-213	42-48	6.3-3.8	74-200	11-9.4	1/20	14	Bainbridge, 1960
Carassius auratus	50-210		10				15	Blaxter, 1969
Carassius auratus	180			108	8			Hertel, 1966
Carassius carassius	100				11.4	6	13	Tsukamoto *et al*., 1975
Carassius carassius	18-42	20-49[c]						Pavlov, 1989
Carassius gibelio	230			226	9.8	<1		Komarow, 1971
Chondrostoma nasus	15-45	4.39+0.456 TL (cm s^{-1})					16	Flore *et al*., 2001
Cobitis taenia	34-71	25-42[c]						Pavlov, 1989
Coregnous nasus	60-330[b]	$9.7*FL^{0.45}$				600	12-20	Jones *et al*., 1974
Cottus gobio	16-41	15-34[c]						Pavlov, 1989
Cyprinus carpio	350			236	8.2	<1		Komarow, 1971
Cyprinus carpio	70-130		2.43/2.63		4.1[d]		10/20	Heap & Goldspink, 1986
Cyprinus carpio	36-77	86-98	13.6-15.6	166.4	26.7	3-4.5/1	16-18	Zerrath, 1996
Cyprinus carpio	10-62.5			$3.71*TL^{0.584}$ (m s^{-1}) TL in m		<0.5	20-21	Wakeling *et al*., 1999

Species	Length (mm)	Swimming performance				Time (s)	Temp (°C)	Reference
		U_{crit} cm s^{-1}	U_{crit} BL s^{-1}	U_{max} cm s^{-1}	U_{max} BL s^{-1}			
Esox lucius	425	297					18	Ohlmer & Schwartzkopff, 1959
Esox lucius	170-200			150-210	7.5-12.5			Hertel, 1966
Esox lucius	120-620[b]	$4.9*FL^{0.55}$	600				12-20	Jones *et al*, 1974
Esox lucius	217				7.2	0.115	15	Webb, 1978
Esox lucius	380			397	8.2-10.5	0.1-0.2		Harper & Blake, 1991
Esox lucius	412[b]			280/340		0.13/0.16	8-12	Frith & Blake, 1995
Gasterosteus aculeatus	44.4[a]	108		159		0.05	18	Law & Blake, 1996
Gasterosteus aculeatus	60	66.1	11.1					Taylor & MacPhail, 1986
Gasterosteus aculeatus	55	36.3	6.6			180	18	Stahlberg & Peckmann, 1987
Gasterosteus aculeatus	50	35	7			60	20	Whoriskey & Wootton, 1987
Gasterosteus aculeatus	33-74	80-73	15.8-16.6	145	33.5	4.5-5/1	16-18	Zerrath, 1996
Gobio gobio	61-80	31-37/43-51					3/20	Pavlov *et al*., 1972
Gobio gobio	116	43.0/55.0	3.64/4.66		19.7	600	4/20	Stahlberg, 1986
Gobio gobio	118	45.6/55.0	3.9/4.7			180	4/18	Stahlberg & Peckmann, 1987
Gobio gobio	53-95	106-119	15-16.7	268	36.2	5/1	16-18	Zerrath, 1996
Gobio gobio	28-50		8.85			180	15-16	Cowx & Welcomme, 1998
Gymnocephalus cernuus	105			133	12.7	<1		Komarow, 1971
Leucaspius delineatus	30-41	36-55[c]					18-22	Pavlov *et al*., 1972
Leucaspius delineatus	51	22.7/38.6	4.5/7.7			180	4/18	Stahlberg & Peckmann, 1987
Leuciscus cephalus	48-64	82-92	15.2-17	214.5	37.7	4-6/1	16-18	Zerrath, 1996
Leuciscus cephalus	66-87	96-106	12.5-13.7	376.2	49	4-6/1	16-18	Zerrath, 1996
Leuciscus cephalus	6-43[a]	$0.45 + 0.23\ T + 0.55\ SL$ (cm s^{-1})				180	8-22	Garner, 1999
Leuciscus leuciscus	100-214	46-90	4.4-4.2	110-240	11.2-11	1/20	14	Bainbridge, 1960
Leuciscus leuciscus	8.3-17.3	$-19.9 + 1.46\ TL + 0.9\ T$ (cm s^{-1})				180	13-20	Mann & Bass, 1997
Leuciscus leuciscus	9-25		10.3			180	15-16	Cowx & Welcomme, 1998

Species	Length (mm)	Swimming performance U_{crit} cm s^{-1}	U_{crit} BL s^{-1}	U_{max} cm s^{-1}	U_{max} BL s^{-1}	Time (s)	Temp (°C)	Reference
Leuciscus leuciscus	6-43[a]	-7.4 + 1.0 T + 0.83 SL (cm s^{-1})				180	8-22	Garner, 1999
Lota lota	120-620[b]	30.6*$FL^{0.07}$				600	12-20	Jones *et al*., 1974
Micropterus salmoides	102[b]	50[c]	5[c]			300	25	Farlinger & Beamish, 1977
Micropterus salmoides	142[b]	47.7[c]	5[c]			1800	25	Farlinger & Beamish, 1978
Micropterus salmoides	51				18.8	<1	15	Webb, 1986b
Micropterus salmoides	93-128[b]		2.22/2.9[c]/3.6[c]			1200	5/10/17	Kolok, 1991
Oncorhynchus mykiss	103-280			105-270/32-73	10.2-7.6/3.1-2.6	1/20	14	Bainbridge, 1960
Oncorhynchus mykiss	610-830	430-830	7-13					Weaver, 1963[*]
Oncorhynchus mykiss	48-358[b, d]		9/5.5			60	10	Fry & Cox, 1970
Oncorhynchus mykiss	305.8[b]	66.57				600	12-20	Jones *et al*., 1974
Oncorhynchus mykiss	143				8.5	0.078	15	Webb, 1975b
Oncorhynchus mykiss	96/150/204			202/226/214	21/15/10.5	<1	15	Webb, 1976
Oncorhynchus mykiss	245-387			229-265	9.3-6.8	<1	15	Webb, 1976
Oncorhynchus mykiss	195				8.1	0.114	15	Webb, 1978
Oncorhynchus mykiss	318				8.7	0.125	15-20	Harper & Blake, 1990
Oncorhynchus mykiss	95				13.7	0.074	10	Gamperl *et al*., 1991
Oncorhynchus mykiss	87-100[b]		7.69			900	10	Hawkins & Quinn, 1996
Oncorhynchus mykiss	87[b]		6			180/220	15	Gregory & Wood, 1998
Osmerus eperlanus	50-170	-0.16 + 0.24 ln (TL)				120-180	17	Sprengel & Lüchtenberg, 1991
Perca fluviatilis	100/220	121/126					18	Ohlmer & Schwartzkopff, 1959
Perca fluviatilis	115			145	12.6	<1		Komarow, 1971
Perca fluviatilis	46-64	56-60[c]					18-22	Pavlov *et al*., 1972
Petromyzon marinus	123-148	20-60				1800	5-15	Beamish, 1974
Rhodeus amarus	27-56	24-42[c]						Pavlov, 1989
Rutilus rutilus	120/280	111/112					18	Ohlmer & Schwartzkopff, 1959
Rutilus rutilus	34-62	34-50					18-22	Pavlov *et al*., 1972
Rutilus rutilus	45-100	91-110	13.3-17.2	171.4	32.8	5-6/1	16-18	Zerrath, 1996

Species	Length (mm)	Swimming performance				Time (s)	Temp (°C)	Reference
		U_{crit} cm s^{-1}	U_{crit} BL s^{-1}	U_{max} cm s^{-1}	U_{max} BL s^{-1}			
Rutilus rutilus	6.3-15	-14.06 + 1.38 TL + 0.69 T (cm s^{-1})			180	13-20	Mann & Bass, 1997	
Rutilus rutilus	6-15		13.3			180	19-20	Cowx & Welcomme, 1998
Rutilus rutilus	6-43[a]	-3.64 + 0.5 T + 0.49 SL (cm s^{-1})				180	8-22	Garner, 1999
Salmo salar	750-850			500-600	5.9-8			Hertel, 1966
Salmo salar	25.8	15/19				900	6/14	Heggenes & Traaen, 1988
Salmo salar	27.8	17/19/27				900	7/8/18	Heggenes & Traaen, 1988
Salmo salar	600	176/216				600	12/18	Booth *et al*., 1997
Salmo salar	70[b]			60±1		300	11-13	McDonald *et al*., 1998
Salmo trutta	130/370			137/305	8.2/10	2	10	Blaxter & Dickson, 1959
Salmo trutta	250			380	15			Hertel, 1966
Salmo trutta	26.1	15/19/24				900	6/14/19	Heggenes & Traaen, 1988
Salmo trutta	32.1	23/24				900	7/18	Heggenes & Traaen, 1988
Salmo trutta	13-22/22-66			13-60/25-100[c]		<0.2	12	Hale, 1999
Salvelinus alpinus	355[b]	100.2				600	12-20	Jones *et al*., 1974
Salvelinus fontinalis	110/116[a]		6.17/7.65			90/30	15	Petersen, 1974
Salvelinus fontinalis	112[a]	93	8.3			10	15	Petersen, 1974
Salvelinus fontinalis	24.2	17/19				900	6/14	Heggenes & Traaen, 1988
Salvelinus fontinalis	28[b]			21.2		2		McLaughlin & Noakes, 1998
Salvelinus fontinalis	70[b]			46±1		300	11-13	McDonald *et al*., 1998
Sander lucioperca	420	191					18	Ohlmer & Schwartzkopff, 1959
Scardinius erythrophthalmus	120/280	75/94					18	Ohlmer & Schwartzkopff, 1959

Species	Length	Swimming performance				Time	Temp	Reference
Scardinius erythrophthalmus	42-75	36-60[c]					17-21	Pavlov *et al*., 1972
	(mm)	U_{crit} cm s^{-1}	U_{crit} BL s^{-1}	U_{max} cm s^{-1}	U_{max} BL s^{-1}	(s)	(°C)	
Thymallus arcticus	70-370[b]	$36.32*FL^{0.19}$				600	12-20	Jones *et al*., 1974
Tinca tinca	255			138	7.5	<1		Komarow, 1971
Tinca tinca	22-69	20-42[c]						Pavlov, 1989

[*] Cited in Blaxter (1969); [a]standard length, SL (mm); [b]fork length, FL (mm); [c]estimated from figure; [d]warm adapted (28°C) carp at 20°C; [e]largemouth bass conditioned at 0.35 m s^{-1} for 30 days; [f]anadromous form.

APPENDIX 3

Summary Review of Environmental Flow Methodologies (from Tharme 2003)

EFM Type	Description
Hydrological	(aka fixed-percentage, rule-of-thumb, standard-setting, look-up table, discharge/historical discharge, or hydrological index methods). Typically simple, primarily desktop EFMs that use hydrological data, usually long-term virgin or naturalised, historical monthly or daily flow records, to derive EF recommendations. The EFMs may incorporate various hydrological indices (VHIs)/formulae (e.g. based on hydrological & regionalisation techniques for gauged/ungauged catchments), include catchment variables, or be modified to take account of hydraulic, biological and/or geomorphological criteria. They require only hydrological and some ecological expertise. The flow indices used are commonly selected on the basis of professional judgement and/or by using a combination of statistical analysis and structured observations of rivers of similar hydrological and/or ecological type. A set proportion of flow (often an absolute "minimum flow") represents the EFR intended to maintain river condition (i.e. whole ecosystem), the freshwater fishery or other highlighted ecological features at some designated acceptable level, on an annual/seasonal/monthly basis. Recent approaches (e.g. RVA) are more complex and hence, flexible. As a result of their rapid, non resource intensive, but low resolution outputs, and low flexibility, hydrological EFMs are most appropriate at the planning/reconnaissance level of WRDs, or in low controversy situations where the EFR estimates may be used as preliminary flow targets or as block-booked allocations. Hydrological EFMs may be used as tools within habitat simulation, holistic or combination EFMs. They have been applied in developed and developing countries.
Hydraulic Rating	(aka habitat retention, transect, habitat analysis or standard setting methods). One of two EFM types that utilises a quantifiable relationship between the quantity and quality of an instream resource, such as fishery habitat, and changes in Q, to calculate EFRs (see habitat simulation EFM type). They use changes in simple hydraulic variables (e.g. wetted perimeter, maximum depth, average velocity), usually measured across single (sometimes multiple) river cross-sections, with flow, as a surrogate for habitat factors known or assumed to be limiting to target species/assemblages (typically fish or benthic invertebrates). Cross-sections are placed at a river site where maintenance of flow is most critical or where instream hydraulic habitat is most responsive to flow reduction, and thus potentially most limiting to the aquatic biota (e.g. riffles). A relationship between habitat and Q, developed by plotting the hydraulic variable against discharge (often using hydraulic models), is used to derive the EFR. Commonly, a breakpoint, interpreted as a threshold below which habitat quality becomes significantly degraded, is identified on the habitat-Q response curve, or a minimum EFR is set as the Q producing a fixed percentage reduction in the particular habitat

	attribute. Hydraulic EFMs are combined desktop-field methods requiring limited hydrological, hydraulic modelling and ecological data and expertise. Due to their low-moderate resource intensity and complexity, and low resolution EFR output, they are of low flexibility and most appropriate for application for WRDs where no/limited negotiation of tradeoffs is required, or as a method within a habitat simulation or holistic type EFM. They represent the precursors of more advanced habitat simulation EFMs. They have been applied primarily in developed countries.
Habitat Simulation	(aka (instream) habitat rating/modelling/mapping, hydro-biological or microhabitat methods). One of two types of combined desktop-field habitat-Q based EFMs (see hydraulic rating EFM type). These EFMs derive EFRs through analysis of the quantity and suitability of instream physical habitat available to target species or assemblages (typically fish or invertebrates) under different flow regimes, on the basis of integrated hydrological, hydraulic and biological response data. Typically, the flow-related changes in physical microhabitat are modelled in various hydraulic programs, using data on one or more hydraulic variables, most commonly depth, velocity, substratum composition, cover and, more recently, complex hydraulic indices (e.g. benthic shear stress), collected at multiple cross-sections within the river study reach. The available habitat conditions, simulated using various habitat modelling programs, are linked with information on the range of preferred to unsuitable microhabitat conditions for target species, lifestages, assemblages and/or activities, often depicted using seasonally defined habitat suitability index curves. The resultant outputs, in the form of habitat-Q curves for the biota, or extended as habitat time and exceedence series, are used to predict optimum flows as EFRs. Some habitat simulation EFMs consider ecosystem subcomponents in addition to instream biota (e.g. sediment transport, water quality, riparian vegetation, water dependent wildlife). Data requirements are moderate-high, and include historical flow records, hydraulic variables for multiple cross-sections, and habitat availability and suitability data for various biota. A high degree of expertise in advanced, dynamic hydrological and hydraulic habitat modelling, land surveying, and in physical habitat-flow needs of target species. The EFMs are complex, highly resource-intensive, moderately flexible, and with a moderate to high resolution EFR output. Habitat simulation EFMs are applied in cases of medium/large-scale WRDs involving rivers with economically important fisheries, of high conservation and/or strategic importance, and/or with complex, negotiated tradeoffs among water users. They may comprise tools within holistic type EFMs. They have been applied primarily in developed countries.
Holistic	In holistic EFMs, which are combination desktop-field approaches, important and/or critical flow events are identified in terms of select criteria defining flow variability, for some/all major components or attributes of the riverine ecosystem (e.g. riparian vegetation, geomorphology, floodplain wetland). This is done either through a

	bottom-up or, more common recently, a top-down/combination process that requires considerable multidisciplinary expertise and input (often workshop or expert panel based). The basis of most approaches is the systematic construction of a modified flow regime from scratch (i.e. bottom-up), on a month-by-month (or more frequent), element-by-element basis, where each element represents a well defined feature of the flow regime intended to achieve particular ecological, geomorphological, water quality, and in some cases social or other objectives in the modified river. In contrast, in top-down, scenario-based approaches, EFRs are defined in terms of acceptable degrees of departure from the natural (or other reference) flow regime, rendering them less susceptible to any omission of critical flow characteristics or processes than their bottom-up counterparts. Holistic EFMs range from moderately to highly data intensive, requiring, among other data, within multiple river reaches/sites, historical flow records (virgin, present day), numerous hydraulic variables across multiple cross-sections, and quantitative biophysical data/models of the flow- and habitat-related requirements of all/select biota and ecosystem components. A commensurately high degree of expertise in advanced hydrological and hydraulic habitat modelling, and in the ecology of all individual biota/ecosystem components, is required. The EFMs are of moderate-high resource intensity, complexity and output resolution. The most advanced, highly flexible approaches utilise several tools from hydrological, hydraulic rating and habitat simulation EFMs, within a modular framework, for establishing EFRs, and may also incorporate social (flow related ecosystem goods and services for dependent livelihoods) and economic data. The most advanced holistic EFMs are applied in cases of medium/large-scale WRDs involving rivers of high conservation and/or strategic importance, and/or with complex, negotiated water use tradeoffs. Simpler approaches (e.g. expert panel assessments, intermediate determinations) are appropriate for lower profile cases involving limited tradeoffs. Holistic EFMs have been applied in developed and developing countries.
Combination	Combination (aka hybrid or multivariate statistical) EFMs comprise a diverse array of EFMs that possess characteristics of more than one of the four basic EFM types (hydrological, hydraulic rating, habitat simulation, holistic). They include partial holistic EFMs, which incorporate holistic elements, but within insufficiently developed methodological frameworks. The ecosystem components considered, data, expertise and other resources required, vary among approaches. The level of resolution of the output (EFR), flexibility, and appropriate level of application of the EFM also differ across techniques. The approaches have been applied in developed and developing countries.
Other	Various other disparate methods and analytical techniques not designed for EFAs from first principles, but adapted or with potential to be used for this purpose. The ecosystem components considered, data, expertise and other resources required, vary among approaches.

	The level of resolution and flexibility of the output, as well as its potential for use as an EFR, are highly dependent on the nature of the individual approaches. This group of techniques are sometimes grouped with the multivariate statistical techniques of combination EFMs. They have been applied in developed and developing countries.
Ecosystem component-specific	Often housed within holistic EFMs, are approaches that have diverged from an emphasis on the relationship between instream habitat, biota and flow, to explore other information best suited to specific river components or other connected ecosystems. They include methodologies for non-riverine wetlands, including lakes, estuaries and the nearshore coastal environment, as well as EFMs for riverine and other wetland ecosystem components, e.g. water quality, geomorphology/sedimentology, riparian/aquatic vegetation, aquatic invertebrates, fish, water-dependent vertebrates other than fish, GDEs, social dependence, recreation, aesthetics and cultural amenity. The types of data, expertise and other resources required vary among approaches, as do the level of resolution of the output (EFR), flexibility, and appropriate level of application of the EFM. The approaches have been applied in developed and developing countries.

APPENDIX 4

Environmental flow methodologies used or proposed for use (*) in UK rivers (After Tharme 2003)

Hydrological

Q_{95} [*Q_{90}, Q_{95}, Q_{50}, Q_n - discharge equalled or exceeded 90%, 95%, 50% (= Median Monthly Flow)],* (**DW**F, or proportion or multiple thereof e.g.1.0 x DWF for sensitive rivers or 0.5 x DWF for least sensitive rivers, calculated using MICRO LOW FLOW/other programs [*DWF - Dry Weather Flow*](Bullock *et al.* 1991); [32]Q_{98} (for less sensitive rivers, versus Q_{95}); (Kirmond & Barker 1997); Q_{90}; (Bragg *et al.* 1999) Q_{347} (considered equivalent to Q_{95}) (Dunbar *et al.* 1998);

MAM(7) [*MAM(7) = mean annual minimum 7-day flow frequency statistic*]; (DWF, or proportion thereof, MICRO LOW FLOW/other programs) (Bullock *et al.* 1991; Bragg *et al.* 1999);

Welsh Water Authority Procedure (based on Q_{95}) (Dunbar *et al.* 1998);

NGPRP Method [*NGPRP - Northern Great Plains Resource Program*](Dunbar *et al.* 1998);

Orth & Leonard Regionalisation Method (Petts *et al.* 1995);

Flow Recession Approach (Petts *et al.* 1995);

VHI [*VHI - various simple hydrological indices*](notably low flow statistics) (Gustard & Bullock 1991);

*Ecotype-based Modified Tennant Method (Dunbar *et al.* 1998);

*Hoppe & Finnell Method (Dunbar *et al.* 1998);

*Texas Method (on ecotype basis) (Dunbar *et al.* 1998);

*Regionalisation of seasonal FDCs for river ecotypes (Gustard *et al.* 1987; Young *et al.* 1996; Hardy 1996; O'Grady 1996 quoted in Dunbar *et al*. 1998; Bullock *et al.* 1991);

*RVA [*RVA - Range of Variability Approach*](for England, & particularly Scotland, with modifications) (Dunbar *et al.* 1998; Bragg *et al.* 1999).

Hydraulic rating

Wetted Bed Area-Flow Method (Petts *et al.* 1995);

*R-2 Cross Method (for Wales & Scotland, particularly) (Dunbar *et al.* 1998)

Habitat simulation

IFIM [*IFIM - Instream Flow Incremental Methodology*] (incl. habitat-biomass/population relationships, mesohabitat/biotope HSI curves & modelling) (Bullock *et al.* 1991; Johnson *et al.* 1995; Elliott *et al.* 1996a, b; Dunbar *et al.* 1998; Bird 1996; Gustard & Cole 1998; Gustard & Elliott 1998; Spence & Hickley 2000; Gibbins & Acornley 2000)

*CASIMIR [*CASIMIR - Computer Aided Simulation Model for Instream flow Requirements in regulated/diverted streams*](Dunbar *et al.* 1998)

*Linked statistical hydraulic & multivariate habitat use models (Dunbar *et al.* 1998)

Holistic

River Babingley (Wissey) Method (incl. various eco-hydrological models/methods to determine benchmark flows for EAFR, e.g. PHABSIM & FDC analyses) (Petts & Maddock 1994; Petts 1996; Petts *et al.* 1999)

*Holistic Approach (Dunbar *et al.* 1998; Bragg *et al.* 1999)

*Building block methodology (Dunbar *et al.*, 1998)

*Expert Panel Assessment Method (Dunbar *et al.* 1998)

Combination

Physical Biotopes/Functional Habitats approaches (Padmore 1998);

Expert panel studies (unspecified) (Acreman *et al.* 2000);

DSHHP Method [*DSHHP - Drake, Sherriff/Howard Humphreys & Partners*](associated with SWALP [*SWALP - Surface Water Abstraction Licencing Policy*]) (Dunbar *et al.* 1998);

Holistic elements based on natural flow regime (Sambrook & Petts 2002);

*Combination of IFIM/PHABSIM analyses for target species with holistic elements (Dunbar *et al.* 1998; Gibbins & Acornley 2000);

*Basque Method (Dunbar *et al.* 1998);

*IFIM in association with other flow regime elements (Dunbar *et al.* 1998; Gibbins & Acornley 2000);

*Regionalisation methods based on habitat modelling (Dunbar *et al.* 1998);

Other

PJ [*PJ - (case-specific) professional judgement*](Sheail 1984, 1987);

RIVPACS [*RIVPACS - River Invertebrate Prediction and Classification System, UK*] (incl. *additional development, e.g. flow-related variables & species responses) (Wright *et al.* 1996; Dunbar *et al.* 1998)

Habitat Attribute-BMWP Model [*BMWP - Biological Monitoring Working Party (score)*](Petts *et al.* 1995);

LIFE Method [*LIFE - Lotic-invertebrate Index for Flow Evaluation*] (Extence *et al.* 1999; Dunbar *et al.* 2002)

SWK Method [*SWK - Scott Wilson Kirkpatrick*] (Extence *et al.*, 1999)

Direct use of fisheries population data (incl. migration & spawning activities - Ireland) (Dunbar *et al.* 1998);

Fish Management Models (Bullock *et al.* 1991)

*Jones & Peters Method (Jones & Peters 1977; Dunbar *et al.* 1998);

*HABSCORE (with additional developments) (Dunbar *et al.* 1998)

*Regional regression-based models (unspecified, with additional developments) (Dunbar *et al.* 1998);

*Analysis of raw population data under alternative river management procedures (Dunbar *et al.* 1998).

APPENDIX 5

Evidence for density-dependent population regulation in coarse fish populations – a preliminary investigation based upon the Environment Agency Fish Monitoring Dataset.

Introduction

The age-structured model identified in Section 4.2.7 as a candidate tool to aid decision-making with respect to environmental flows, assumes that at least one of the three key processes (growth mortality and recruitment) regulating population biomass is density-dependent, and can be adequately described using established models.

This report summarises the main findings of a preliminary investigation to seek evidence for these density-dependent processes based upon comparisons of populations of roach (*Rutilus rutilus*) sampled from sections of UK rivers. This approach assumes that discrete populations of roach inhabit these river sections.

Materials and Methods

Evidence of density-dependent regulatory processes was examined using estimates of roach numeric and biomass density sampled at 164 locations within 19 UK rivers between 1993 and 2001. Lower, middle and upper sections of several of these rivers were treated as discrete rivers.

For a given river *R*, and sampling year, *Y*, several population density estimates δ may be available corresponding to specific sampling sites, S. In these cases, a mean density estimate was calculated after $\log_e$ transforming the individual density estimates to ensure that normality assumptions were met before back transformation:

$$\bar{\delta}_{R,Y} = \exp\left(\frac{\sum_{Y=1}^{Y=k} \ln \delta_{R,S,Y}}{n_{R,Y}}\right)$$

where $n_{R,Y}$ is the number of sampling sites, *S* in river *R* sampled in year *Y*. This process reduced the data set to 19 river-year estimates of mean density.

Growth parameter estimates

Length *L* at age estimates, L_t were available corresponding to 14 of the 19 river-year combinations described above[1]. For each combination, the parameters, $L\infty$, *K* and t_0 of von Bertalanffy growth function (VBGF) were estimated using non-linear least squares.

[1] Length at age data for the Colne and Chelmer rivers was not disaggregated by upper and lower reaches.

Natural mortality estimates, M

Numbers of roach *N* at age *t* were also only available for these 14 river-year combinations. For each combination, the total annual instantaneous mortality rate *Z* was estimated as the slope of the regression of $\log_e N_t$ versus *t*. Assuming negligible fishing mortality, it was assumed that the instantaneous natural mortality rate, *M* was approximately equal to *Z*.

Density-dependent relationships examined

The significance of the slope of the regression of L_∞ versus biomass density was used to test for the existence of density-dependent growth among the roach populations following the model of Lorenzen (1996):

$$L_{\infty B} = L_{\infty L} - gB$$

where $L_{\infty B}$ is the asymptotic length at biomass B and $L_{\infty L}$ is the limiting asymptotic length in the absence of competition. The slope parameter g – the competition coefficient describes the amount by which $L_{\infty B}$ decreases per unit of biomass density (Figure 1).

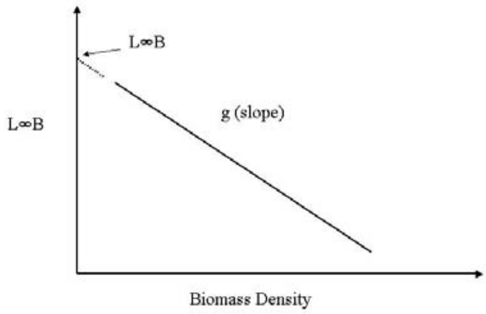

Figure A5.1 The parameters of the density-dependent growth model

A significant negative slope would indicate the existence of density-dependent growth in roach populations.

Density-dependent mortality
The existence of density-dependent natural mortality was tested for in a similar manner, only this time a positive linear or non-linear relationship between *M* and population density would be expected. Population density was expressed in terms of numeric density (N ha^{-1}).

Results

Density-dependent growth

The VBGF was fitted to the length at age data with varying degrees of success. For several river-year combinations, for example the Lower Stour (1994), length-at-age estimates were available only for the first five or six age classes or cohorts over which growth often appeared linear providing little indication of the likely value of $L_{\infty B}$. An ordinal three-point goodness-of-fit score was therefore subjectively assigned to each model fit as follows:

Reasonable or good fits were achieved with only six from a total of 14 length-at-age datasets (Table A1). No significant *(P* < 0.05) relationship was found between the six estimates of $L_{\infty B}$ and biomass density (Figure A5.2).

Table A5.1 Goodness of fit criteria and scores for the VBGF

Score	Description	Comments
*	Poor fit	Few cohorts, linear relationship and no reliable estimate of asymptotic length.
**	Reasonable fit	Length at age data available for more than 7 cohorts with length at age becoming asymptotic.
***	Good fit	Length at age data available for more than 7 cohorts exhibiting a clear asymptotic length.

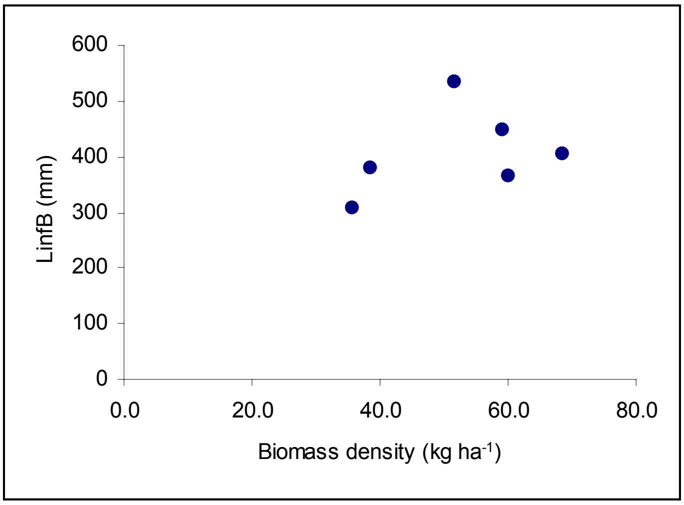

Figure A5.2 Estimates of $L_{\infty B}$ for populations of roach plotted as a function of roach population biomass density

Density-dependent natural mortality

Satisfactory estimates of *M* were obtained for all 14 river-year roach populations although the numeric density estimate for Holland Brook (7200 ha^{-1}) was a clear outlier and therefore excluded. Evidence of density-dependent natural mortality was detected at the P<0.10 level although numeric density explained only approximately 20% of the variation in M (Figure 3).

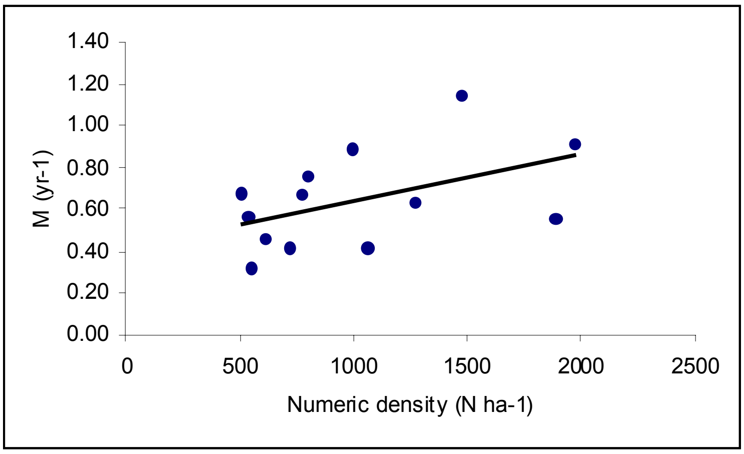

Figure A5.3 Estimates of M plotted as a function of numeric density with fitted linear model: M=0.42+0.0002δ, where δ=Numeric density (N ha^{-1}), R^2 = 0.23, P=0.09. Data for Holland Brook is an outlier (δ =7200 ha^{-1}) and not included.

Conclusions and Recommendations

The existence of density-dependent processes regulating population numbers and biomass have important implications for the management of hydrological conditions since these can have a profound and rapid influence on population density. It is generally accepted that such density-dependent processes can regulate population biomass through their effect on growth and natural mortality rates, and rates of recruitment.

The aim of this preliminary investigation was to test for the existence of density-dependent processes in populations of roach – a common and generally abundant coarse fish species. The results provide some evidence of density-dependent natural mortality in roach populations. The unexplained variation in M, is likely to reflect (i) errors surrounding the estimates of M derived from numbers at age data, (ii) poor congruence between the sites sampled for numbers at age and population density, (iii) imprecise mean population density values estimated on the basis of samples from several separate sites or locations, and (iv) density-independent factors such as water quality.

No evidence of density-dependent growth was detected among roach populations. This may partly reflect the lack of reliable growth parameter estimates. Only six estimates of $L_{\infty B}$ and corresponding biomass density could be compared.

Significant population regulation often occurs during the pre-recruit stage. The available data were inadequate to test for this. It is recommended that length at age and numbers at age data for at least another 15 river-year combinations be compiled to augment the existing dataset for re-analysis prior to drawing any firm conclusions regarding the existence of density-dependent processes regulating coarse fish populations in UK rivers.

Annex to Appendix 5

Annex Table 1 VBGF Parameter and biomass density estimates for the 14 river-year roach populations

River	Date	n	Kg ha-1	Linf	K	t0	Fit
Blackwater	1996	4	59.0	450	0.075	-0.911	**
Blackwater	1999	5	68.3	406	0.113	-0.232	**
Gt Ouse	2001	4	35.5	307	0.178	0.006	***
Stour Upper	1994	9	60.0	366	0.137	0.341	**
Stour Upper	1997	7	38.4	382	0.121	0.253	**
Stour Upper	2000	8	51.6	537	0.067	-0.243	**
Holland Brook	1993	1	236.5	3704	0.008	-1.163	*
Chelmer & Blackwater Canal	1995	3	50.7	1515	0.015	-1.498	*
Stour Lower	1994	2	21.8	249	0.161	-0.750	*
Stour Lower	1997	3	9.6	250	0.286	0.651	*
Stour Lower	2000	2	79.3	695	0.051	-0.727	*
Stour Middle	1994	7	51.2	1195	0.024	-0.877	*
Stour Middle	1997	12	26.0	285	0.227	1.860	*
Stour Middle	2000	10	22.5	379	0.126	1.679	*

Annex Table 2 Estimates of instantaneous natural mortality rate and numeric density for the 14 river-year roach populations. [1] Outlier not included in regression.

River	Date	n	Kg ha-1	N ha-1	Cohorts	M
Blackwater	1996	4	58.96	721	13	0.42
Blackwater	1999	5	68.34	1062	16	0.42
Chelmer & Blackwater Canal	1995	3	50.72	1886	13	0.55
Gt Ouse	2001	4	35.50	550	12	0.32
Stour Lower	1994	2	21.77	1471	5	1.15
Stour Lower	1997	3	9.57	996	7	0.89
Stour Lower	2000	2	79.29	1967	9	0.92
Stour Middle	1994	7	51.24	1270	13	0.64
Stour Middle	1997	12	26.02	802	9	0.75
Stour Middle	2000	10	22.51	506	10	0.68
Stour Upper	1994	9	60.05	772	12	0.66
Stour Upper	1997	7	38.37	540	13	0.56
Stour Upper	2000	8	51.55	614	12	0.46
Holland Brook [1]	1993	1	236.51	7198	6	0.29