PHYSICAL BIOTOPES IN REPRESENTATIVE RIVER CHANNELS: IDENTIFICATION, HYDRAULIC CHARACTERISATION AND APPLICATION

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ABSTRACT

There is a need to predict the river flow conditions necessary to support instream (and river corridor) biota. Many rivers have their flow regulated in some way and the field of 'habitat hydraulics' (or 'ecohydraulics') has grown up to encourage classification and characterisation of lotic physical environments. This thesis, based on field measurements at eleven sites in North-east England (which are nationally representative), presents the data normally collected for the calibration of the PHABSIM model in alternative ways, to identify and characterise 'physical biotopes'. These are segregated on the basis of subjectively (visually)-defined flow types. Statistical validation of biotopes as hydraulically discrete units shows them to be defined by a characteristic range of hydraulic variables, especially the Froude number.

An investigation of the spatial and temporal variability of biotopes in different channel types and at different flows was carried out (1993-1995), which included the most extreme floods and droughts on record at some sites. The 'biotope approach' is shown to have potential as a reconnaissance approach to the assessment of habitat quality. Despite the success of the biotope approach e.g. in River Habitat Surveys, methodological improvement is still required, notably in the area of hydraulic characterisation by velocity measurements. Hydraulically rough channels and those with substantial macrophyte growth present problems; a logarithmic velocity profile cannot be assumed. A relationship between biotope diversity and stream ecosystem health is suggested; biological surveys are required to determine the direct relationship.

This thesis represents a shift from the geomorphological riffle-pool theory to a broader classification of instream hydraulic units. It provides a framework for testing the debate in ecological theory regarding the role of patch dynamics within the continuum of the river environment. The study indicates that an 'environmentally acceptable flow regime' is one which creates a range of biotopes and maintains 'critical biotopes' at periods coinciding with key life-stage events. Flood events are beneficial for their role in 'resetting' the system.

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1. RIVER FLOW MANAGEMENT AND THE ROLE OF CHANNEL PHYSICAL HABITAT

Chapter Overview

This chapter introduces the context of the research from a river management perspective. River regulation and management of flows for instream biota are discussed with reference to recent environmental concern and low flow problems. The need to consider both ecology and fluvial geomorphology in the management of river channels is emphasised (1.1). The scientific rationale for considering flows within the highly variable environment of a river channel focuses on the standard identification of hydraulic habitat units or 'physical biotopes'; these are considered in light of contemporary stream ecosystem theories, notably 'hydraulic stream ecology' and 'patch dynamics' (1.2).

A review of the methods currently available and operational for the provision of instream ecosystem flow requirements considers simple discharge based methods through to sophisticated modelling techniques including PHABSIM (1.3). National inventories of instream physical habitat are being developed to produce channel typologies; these may be applied to assist in the choice of segment-scale 'ecologically acceptable flows' which support ecosystem structure and functioning. The chapter concludes with an introduction to the potential applications of river channel typologies (1.4) and the relevance of this study in the broader context of sustainable, policy-oriented river management (1.5).

1.1 CONTEXT OF THE STUDY

Increasing demands on water resources and growing concern for environmental protection during the last decade, have led to an increasing need to allocate water to instream activities, including fisheries, recreation, conservation, navigation and pollution dilution (Petts and Maddock, 1994). Integrated management of water resource and aquatic ecosystems is the concern of both managers and researchers (Cohen *et al.*, 1996). In the UK the 1991 Water Act attempts to provide the necessary legal framework to balance the various instream flow demands, by requiring all abstractions (except those less than $20m^3$/day) to be licensed by the former National Rivers Authority (now Environment Agency)* (POST, 1993). However, in order to reallocate water to other uses many existing licences must be revoked and revised; fair allocation of water between the seven different river management functions of the EA requires a precise definition of an **Ecologically Acceptable Flow** (EAF) which is legally binding.

Almost all major rivers in the UK are regulated to some extent (Petts, 1988); in theory this allows flows to be manipulated to minimise the adverse effects of low flows. Recent summers have focused official attention on low flows notably the 1988-92 droughts and, more recently, the very low flows of the 1995 summer (Institute of Hydrology, 1995). The ability to manipulate flows provides the opportunity to fulfil the flow requirements of instream biota at critical life stages and to maintain ecological processes in the channel and riparian environment. In practice, however, these demands must be traded-off against the requirements of abstraction, return flows, HEP production and recreation, which often do not coincide with 'optimum' flows for riverine habitats and biota. There is little or no consideration of the river basin channel network as an ecosystem through which flows and flow variability can be strategically set by co-ordination of other regulating factors. The 1991 Water Resources Act requires increased allocation for fish and effluent disposal, but the statutory mechanisms for ensuring this often fall short of their objectives (see section 1.3). Schemes for river regulation, abstraction and inter-basin transfers are forecast to increase (Petts *et al.*, 1995), which necessitates a rapid, cost-effective technique for impact assessment. Methods for developing such objectives are discussed in section 7.2.

The 'marriage' of ecology and geomorphology is proposed by Petts *et al.* (1995) as a means of providing a more holistic, basin scale assessment of instream flow requirements. These authors advocate a focus on flow management to maintain *habitat* rather than individual species or communities, as predictions of individual biological responses are difficult (Armitage, 1994):

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* During the concluding phases of this research the National Rivers Authority (NRA) - the official river management organisation for England and Wales - became incorporated in the Environment Agency (EA). Readers may find both referred to according to the date of origin of publication or data.

> *"... the objective assessment of habitat change in relation to flow contributes valuable information to the decision making process"*
>
> Petts *et al.* (1995, p15)

Newson (in press) supports the collaboration between geomorphologists and ecologists recently advocated in an international review of river ecology (Cummins *et al.*, 1995), but goes on to show how infrequently this occurs in practice. The past failure of geomorphologists and ecologists to integrate their *scales* of research to provide information at a level appropriate to water resource managers has hindered the development of catchment-scale EAFs. This chapter reviews the methods currently available for quantifying the response of habitats to discharge, and considers alternative methods of determining 'critical biotopes' (see section 1.2.2 for a definition of the term biotope) i.e. those which are most sensitive to fluctuations in flow and which are likely to be the most significant hydraulic units in a reach, in terms of geomorphological and ecological processes.

1.2 ECOLOGICAL PRINCIPLES

The debate over the relative importance of biotic and abiotic factors in determining community structure has pervaded the ecological literature since the 1960s (Pennack, 1971; Ormerod, 1988; Minshall, 1988). Much ecological literature exists in support of both biotic (Allan, 1983; McAuliffe 1983, 1984; Hart, 1992) and physical factors (Ward and Stanford, 1979; Statzner, 1981; Stevenson, 1984; Wright *et al.*, 1984, 1993). A comprehensive review of biological influences is provided in Barnes and Minshall (1983); these are discussed within the context of developments in stream ecosystem theory (Minshall, 1988). Lake and Barmuta (1986) note that the relative importance of biological and physical factors is dependent upon the scale of the study, which is reflected by Gibbins (1996), who concludes that:

> *"...biological interactions may influence community structure on smaller spatial and temporal scales, but their influence on whole river reaches and over long time-periods remains equivocal"*
>
> Gibbins (1996, p18)

In the case of fish, salmonids tend, generally, to be influenced by abiotic factors, whereas most coarse fish populations are believed to be density-dependent (Mann, 1995). This reflects a contrast between upland streams which have frequent and sudden variation in discharge and therefore hydraulic habitat, and stable, lowland habitats where biotic factors play a more important role (Zalewski and Naiman, 1985). Dudgeon (1992) makes a distinction between systems with variable or unpredictable flow regimes where abiotic processes dominate, and physically stable systems where biological interactions are more significant determinands of invertebrate communities.

It is beyond the scope of this thesis to define units which are associated with discrete or specific biotic assemblages. The main objective is to describe and define physical units which relate to "potential" rather than "functional" habitats (Harper *et al.*, 1992). As de Jalon (1992) notes:

> *"...physical habitat factors are generally more predictable, less variable and more easily measured than biological ones, and are thus preferable descriptors of streams."*
>
> de Jalon (1992, p364)

However, as Harper *et al.* (1992) discover, the definition of instream habitats remains ambiguous:

> *"...it has long been recognised that distinct habitats exist in streams, but few studies have shown the existence of a definitive list"*
>
> Harper *et al.* (1992, p224)

The major components of abiotic habitat are considered to be substrate, hydraulics, aquatic vegetation (cover), and water quality or chemistry (Helm, 1985; Gordon and McMahon, 1992). Water quality appears to be the dominant influence in acidic streams (Ormerod *et al.*, 1987; Wade *et al.*, 1989; Rutt *et al.*, 1990). However, in neutral or alkaline streams physical characteristics are considered to be more important (Cushing *et al.*, 1980). It is not the intention of this study to consider water quality; a national classification of all main river watercourses already exists (Department of Environment, 1986). At the reach or basin scale, substrate type, light and temperature are additional physical determinands of the distribution and abundance of stream benthos (Young, 1992); but can be assumed to be constant at the mesoscale which is adopted for this study. At the scale of the "riffle-pool" sequence (Frissell *et al.*, 1986), Mosley (1985) notes:

> *"...there is widespread agreement that, given the appropriate water quality, the distribution and number of fish and invertebrates in a stream are primarily controlled by the flow regime and the associated water velocities, depths, substrate and cover"*
>
> Mosley (1985, p502)

This thesis therefore focuses on substrate and hydraulics, specifically the interaction between substrate, velocity and depth. At the time work for this thesis commenced the Physical Habitat Simulation Model (PHABSIM) was being developed for UK rivers as part of an Institute of Hydrology R&D project (see section 1.3 for a discussion of the merits and limitations of PHABSIM). Depth, velocity and substrate together with cover are considered to be the main physical factors influencing biotic populations (Bovee, 1978; Gore, 1978; Gorman and Karr, 1978; Orth and Maughan, 1982; Shirvell and Dungey, 1983) and are the key variables modelled in PHABSIM. Cover is not formally treated in this thesis; the role of instream vegetation as cover, physical habitat and for its influence on hydraulics is discussed in section 6.2.2.

1.2.1 River Continuum rivals: stream hydraulics and patch dynamics

The River Continuum Concept (RCC) was proposed by Vannote *et al.*, (1980) as a theoretical framework to explain longitudinal changes in the biotic and nutritional status of freshwater ecosystems. The role of fluvial geomorphology in providing a template for freshwater ecology was first stated by Hynes (1970, 1975). Interactions between flow and substrate are known to create a mosaic of habitat patches (Hynes, 1970; Moss, 1980; Statzner *et al.*, 1988). The ecological importance of hydraulics is scale dependent; in a debate over the significance of hydraulics to stream ecology, Cummins *et al.* (1995) conclude:

> *"...at the local reach or tributary scale this seems to be an important argument, but whether this can account for the observed change along entire longitudinal profiles of river systems seems unlikely."*
>
> Cummins *et al.* (1995, p6)

At the mesoscale, flow and its interaction with substrate is recognised as a major determinand of benthic invertebrate distributions (Frissell *et al.*, 1986). At the basin scale Statzner and Higler

(1986) believe stream hydraulics to be the most important factor determining the distribution of benthic invertebrates. The literature regarding abiotic influences on ecological communities has recently been reviewed in the context of major developments in stream ecosystem theory, notably 'patch dynamics' (Naiman *et al.*, 1988) and the role of physical disturbance (White and Pickett, 1985; Hildrew and Giller, 1994).

Early studies on the influence of abiotic factors on invertebrates focussed on flow characteristics within the boundary layer (Ambuhl, 1959). More recently studies of shear stress and the development of hemispheres to estimate bed shear stress (Statzner and Müller, 1989) represent significant advances in 'hydraulic stream ecology.' The strongest relationships between benthic invertebrate distributions and hydraulics are found with bed hydraulic variables (shear stress and boundary Reynolds number) under baseflow conditions (Quinn and Hickey, 1994). However, it is not the objective of the present study to investigate near-bed hydraulics and associations with invertebrate distributions. Broader ecological management must consider the instream environment for fish and other aquatic organisms as well as habitat maintenance *per se*. Differences in mean hydraulic variables are presented as a gross comparison between visually distinct units. The basic instream hydraulic unit is the **'physical biotope'** (see section 1.2.2 for a definition). Hydraulic characterisation of physical biotopes provides ecologists the opportunity to determine their suitability for a range of freshwater biota, or to assess general habitat conservation value based on biotope diversity. Water resource managers may then manipulate flows to maintain particular biotopes or 'biotope diversity' (see section 5.2), a factor which is not formally considered under present flow management.

The recent establishment of a International Symposium on 'Habitat Hydraulics', (the first of which took place in Trondheim, Norway (August, 1994) and the second ('Ecohydraulics') in Quebec, Canada in June, 1996) marks the growing interest in 'hydraulic stream ecology' (Statzner *et al.*, 1988). Some authors state that, at a basin scale, hydraulics form the basic template to which biotic communities are adapted (Southwood, 1977). Scale links between biota, channel units and flow are best addressed by stream hydraulics (Hildrew and Giller, 1994). In an overview of stream ecosystem theory, Cummins *et al.* (1995) support Statzner and Higler's (1986) critique of the River Continuum Concept (Vannote *et al.*, 1980), rejecting the view that the stream ecosystems and biotic associations exist as a series of physical *gradients*. Instead they advocate:

> *"...geomorphological-hydrological characteristics form the fundamental templet (along intact catchments) upon which biological communities become and remain adapted"*
>
> Statzner and Higler (1986, p129)

This notion is supported by empirical evidence; species assemblages are more strongly influenced by this segment-level physical template than longitudinal gradients in environmental variables (Brussock and Brown, 1991). Further evidence is provided by Brown and Matthews (1995) in a study of the relationships between fish species assemblages, riffle-pool sequences and stream order.

The RCC has been modified to relate downstream zonation patterns of benthic invertebrates to changes in stream hydraulics associated with breaks in slope, major tributary confluences and discharge categories (Bruns *et al.*, 1984; Statzner and Higler, 1986). This view is supported by Higler and Verdonschot (1992) for a single organism (the Caddis larvae), but is rejected for whole ecosystems at the catchment scale (Petersen and Sangfors, 1991). Many ecological studies have tended to focus on gradients in continuous environmental variables in a longitudinal direction (Hawkins and Sedell, 1981; Cushing *et al.*, 1983; Wright *et al.*, 1984; Ormerod *et al.*, 1987; Sedell *et al.*, 1989; Brussock and Brown, 1991), in order to test empirically the River Continuum Concept (RCC). The former study sampled only riffle habitat (see section 1.2.2), to reveal a change in relative abundance of invertebrate functional groups as predicted by the RCC. Brussock and Brown's study compared riffle and pool fauna and concluded that local geomorphology and hydraulics 'disrupt' biotic trends associated with longitudinal gradients in environmental variables. Cummins *et al.* (1995) describe how hydraulics are more likely to relate to local geomorphological influences which are superimposed upon broad longitudinal trends to produce a series of 'patches' of hydraulic and morphological units. This is the notion underlying the spatial dimension of the 'patch dynamics' theory of stream ecosystems (Townsend, 1989). (The temporal, discharge-related dimension is discussed in sections 5.3 and 7.3). The debate continues over whether habitat hydraulics and patch dynamics are the major determinands of biotic distributions at the scale of entire catchments, or whether mesoscale physical biotopes need to be considered alongside nutritional and biotic changes down longitudinal profiles in accordance with the RCC. This can only be tested by exploring relations between biota and physical biotopes at a range of sites from different locations within a catchment to incorporate both geomorphological and longitudinal (discharge-related) change.

1.2.2 Biological (invertebrate) sampling strategies

Until recently, ecologists have tended to concentrate on differences between riffle and pool fauna, (Logan and Brooker, 1983; Ormerod and Edwards, 1987; Brown and Brussock, 1991; Statzner and Borchardt, 1994) relating changes in riffle biota to factors which show continuous gradients in a longitudinal direction (see section 1.2.1). Invertebrate sampling at a national scale traditionally sampled only one 'habitat' in order to permit comparison of results between sites (Wright *et al.*, 1984). Sampling focused on riffles due to ease of sampling, and because small changes in discharges can significantly affect habitat availability (Brooker and Hemsworth, 1978; Poff and Ward, 1991), as riffle biota are more sensitive to changes in oxygen concentrations (Gibbins, 1996). Traditionally, 'riffle' has tended to include any shallow, fast flowing area with a broken or wave-dominated water surface (see reviews in Wadeson, 1994). Perception of the importance of marginal habitats has increased recently (Rutt *et al.*, 1989; Petts *et al.*, 1995; Humphries *et al.*, 1996); as has that of macrophyte vegetation in lowland rivers (Smith *et al.*, 1990; Wright *et al.*, 1994; Harper and Smith, 1995). More recently, "all habitat types" have been sampled in the River Invertebrate Prediction and Classification System (RIVPACS) (Wright *et al.*, 1993; Wright, 1995), although in RIVPACS no formal procedures exist for the identification of different habitat types. Units defined by their discrete hydraulic properties have, until this study and that of Wadeson (1995a, 1995b), not been recognised as standard habitat units which may be identified in a range of geomorphological channel types.

The ecological basis for this study therefore centres on the assumption that nutrient levels, competition and predation are not the dominant influence on biota and that physical diversity is associated with high species diversity (Hynes, 1970). It is likely that units with characteristic hydraulic properties equate to patches in the 'patch dynamics' theory (Townsend, 1989). This thesis focuses on the **'physical biotope'** (also known as the 'hydraulic biotope' - Wadeson, 1995b) as the basic hydraulic unit or 'patch'. Cummins *et al.* (1995) state that the critical characteristics of 'hydraulic stream ecology' are mean velocity, depth, substrate and surface slope. At a local scale slope does not directly influence habitat hydraulics, thus a 'physical biotope' may be defined as:

> "a unit with a characteristic range of values for hydraulic and substrate variables."

The term physical biotope is adopted in this study to avoid confusion with the ecologically defined term ‘biotope’. Lelek and Kohler (1990) used the latter to define areas of a larger spatial scale with discrete *physical and chemical* characteristics e.g. the Rhine between Basel (East Germany) and Emmerich (Dutch-German border). Fuchs and Statzner (1990) adopted the definition of biotope proposed by Illies (1978), which includes both features e.g. rivers, brooks, streams, springs, lakes, pools, ponds, peat bogs; and descriptive terms e.g. brackish water, freshwater, lentic water. **Physical biotope assumes a smaller spatial scale (within which water quality is assumed to be constant) broadly corresponding to the “riffle-pool” system (Frissell *et al.*, 1986), but with local spatial variation according to substrate size distributions.** It has been stated that the biotope is independent of scale (Armitage *et al.*, in press); in this study physical biotopes are used to describe mesoscale units which exist within morphological units. The previous, ecological definition of ‘biotope’ describes the abiotic environment of a particular ecological *community* (Price, 1975; Ward, 1992); as yet the ecological relevance of physical biotopes across a range of sites is unproven. The term ‘physical biotope’ is adopted from Wadeson (1995b) to produce a standard terminology for an internationally developing field, but for brevity may be referred to simply as biotope. However is should be emphasised that no assumptions are being made regarding the ecological significance of, or biological communities associated with, these hydraulic units.

The main objective of this thesis is the standard identification of the range of physical biotopes which exist in different channel types, and their hydraulic characterisation at different discharges. Hydraulic diversity may explain differences in invertebrate distributions between physical biotopes; biotope diversity over a larger scale (the reach or segment) may be taken as one index of freshwater habitat quality or *"conservation status"* (O’Keeffe, 1996). It may be considered together with national water quality indices (Department of Environment, 1986) and other features recorded in River Habitat Surveys (Raven *et al.*, 1996) as a means of evaluating the overall quality of a particular reach.

1.2.3 Temporal changes and 'patch dynamics'

The above discussion has concentrated on spatial trends in patch dynamics. Seasonal changes in hydraulics and the flood pulse provide the physical template to which biotic communities become adapted (Poff and Ward, 1990; Brown and Matthews, 1995). The temporal changes considered by 'patch dynamics' are related to a river's flow regime; the description of biotope sequences across a range of channel types and discharges (see sections 5.2 and 5.3) may be related to summary flow statistics in order to predict the range of biotopes, and their probable annual occurrence frequency. The implications for 'patch dynamics' and biotic communities are discussed in section 7.1.

Extreme hydrological events may alter stream ecology if they are of greater magnitude than those predicted by the 'normal' flow regime. From an ecological perspective, the intensity, timing and frequency determine whether an event constitutes 'disturbance' or simply natural system variability (Resh *et al.*, 1988). Another influential factor is the capability of a system to tolerate a disturbance i.e. its resilience (Milner, 1994). The flood events of January and February 1995 and the drought of summer 1995 in Northern England may be discussed in the context of disturbance and resilience of biotope type and distribution and the implications for 'patch dynamics' (see section 7.1.3). In rivers regulated during periods of sustained summer low flows (e.g. those which occurred between 1988-1992 and 1995), an instantaneous increase in discharge brought about by a reservoir release may represent an unpredictable event which affects the temporal 'patch dynamics' and subsequently stream biota. Impacts are likely to depend on the regularity and magnitude of releases throughout summer compared to 'natural' low flows; this is discussed further in section 7.1.3.

1.3 FLOW MANAGEMENT, POLICY AND INSTREAM METHODOLOGY

The statutory mechanism by which flows downstream of reservoirs are managed to protect the requirements of instream users is the **Minimum Maintained Flow (MMF)** (Gustard *et al.*, 1987), defined as *"a specific discharge below which flows must not fall"* (Gibbins, 1996). The methods by which MMFs are set have been criticised as lacking in scientific rationale; being based more on biologists' estimates or engineer's 'rules of thumb' (Fraser, 1972; Armitage and Petts, 1995). The traditional approach to residual flows is to prescribe flows at key locations in the river (usually immediately below abstraction points). Prescribed minimum flows are based on hydrological statistics, and are commonly set at the 95th percentile flow (Q_{95} i.e. the flow which is exceeded for 95% of the annual recorded flows). This value tends to fulfil the requirements of historical industrial and agricultural abstractors rather than instream biota (Gustard *et al.*, 1987; Petts *et al.*, 1995).

Minimum Acceptable Flows (MAF) are set in regulated rivers for fish and effluent disposal (Sheail, 1984, 1987). Rees and Williams (1993) define a 'minimum acceptable flow' as one which:

> *"...provides adequate protection for riverine and fisheries environments and ensures adequate dilution and degradation of effluent discharge."*
>
> Rees and Williams (1993, p38)

The difficulty of establishing such values is recognised in view of uncertainty over the environmental impact of an individual abstraction, and relative importance of different uses. The latter requires value judgements and is subject to individual perception.

Fixed percentages of **Average Daily Flow (ADF)** have also been used as an easily implementable method, based on calculations of yearly averages of mean daily flow statistics. The Montana Method is the best documented, in which 10 % ADF is considered the *minimum* flow requirement for instream needs (Tennant, 1976). Percentages are set to correspond to a 'level' of habitat protection: 30% of Annual Daily Flow (ADF) was considered to provide *optimum* fish habitat in small streams; 20% offered habitat *protection*; 10% ADF corresponded to a *threshold minimum flow*, below which habitat degradation would occur (Orth and Leonard, 1990). Although easy to adhere to, percentages of ADF make no allowance for the variety of channels at all scales. Additionally, neither fixed percentages nor flow percentiles provide consideration of seasonal species requirements e.g. migration, or differing flow requirements to maintain the hydraulic characteristics of physical biotopes which are dependent on channel morphology and flow regime. Additionally, Tennant's recommended flow percentiles apply only to northern hemisphere streams with stable regimes which are snowmelt dominated (Tennant, 1976).

In a review of compensation flows in the UK (Gustard *et al.*, 1987), it was concluded that there is a need to develop quantitative relationships between biota and physical and chemical variables "*at a* *scale appropriate to the river reach".* At present 'optimum' flows for biota may be determined by the application of PHABSIM; adopted in Britain by the Institute of Hydrology (Johnson *et al.*, 1994) from a suite of US models which constitute the major Instream Flow Incremental Methodology (Bovee, 1982). The model combines hydraulic modelling using defined hydraulic parameters with habitat simulations based on 'habitat suitability criteria', to predict changes in 'available habitat' with discharge (Bullock and Gustard, 1992). 'Combined Suitability Indices' are calculated from observations of biota and the associated hydraulic conditions, assuming these indicate species' hydraulic preferences; these are combined with hydraulic simulations to determine changes in area of available physical habitat or Weighted Usable Area (WUA) with discharge (Bullock *et al.*, 1991). However, simulations are specific to an individual species, or indeed life-stage (often an invertebrate species which is known to be sensitive to fluctuations in flow). The ecological credibility of the model has received much criticism for the assumption that the simulated values of hydraulic variables (depth, velocity and substrate) in a combined index constitute the main influences on invertebrate distributions (Morantz *et al.*, 1987; Scott and Shirvell, 1987; Orth, 1987; Swales and Harris, 1995). The assumption that mean depth and velocity are of independent and equal influence (a criterion for constructing habitat suitability indices and calculating WUA) has been criticised (Shirvell and Dungey, 1983; Mathur *et al.*, 1985; Morantz *et al.*, 1987). In reality, habitat selection and preferences vary depending on spatial and temporal changes in habitat availability (Heggenes, 1990, 1996; Heggenes and Saltveit, 1990; Heggenes *et al.*, 1994), predation risk and foraging rate (Harvey, 1991). Beschta and Platts (1986) describe how salmonids often select spawning gravels with below optimum substrate and hydraulic conditions, to ensure spawning adjacent to high quality rearing pools.

PHABSIM simulations are based upon several assumptions about channel morphology, which as this study has revealed, are not met in reality:

1. Channel substrate is accurately represented by the substrate classification scheme; the influence of vegetation on hydraulics does not vary significantly between sites (or plant species)
2. Substrate distributions do not change between calibration flows
3. Cross-sectional morphology is constant between calibration flows

4. A reach selected to be 'representative' will contain all the morphological units (and in the same proportion) as they occur in the wider reach or sector (after Frissell *et al.,* 1986).

Methods are being developed to incorporate the influence of vegetation on channel hydraulics into PHABSIM simulations (Hearne and Armitage, 1993; Hearne *et al.*, 1994). The assumptions of substrate stability have been criticised for braided channels (Mosley, 1982b; Glova and Duncan, 1985). Morphological change due to large flood events (see section 5.4) also results in different substrate distributions, which would alter habitat at the microscale and invalidate previous calculations of WUA.

Although the model successfully combines the disciplines of hydrology and ecology, it does not, at present, incorporate geomorphological influences on flow. The failure to utilise geomorphological information on flow-channel interactions i.e. to incorporate biotopes and their hydraulic response to discharge (section 5.3) is a fundamental criticism of the model. In a review of channel habitat inventory and instream flow assessment, Mosley (1985) concludes that geomorphological input is essential:

> *"...to ensure that investigations are carried out in sections of the river which are representative, so that valid inferences may be made about the river as a whole, and when changes in flow regime may cause changes in channel morphology"*
>
> Mosley (1985, p518)

PHABSIM guidelines for selecting representative reaches state:

> *"The process of selecting a representative reach requires the identification of the variety of different habitat types present in the larger stretch of the river...having identified the variety of different habitat types..we proceed to choose a reach which contains examples of all these habitat types"*
>
> Johnson *et al.* (1994, p10)

The guidelines state that a full survey using aerial photography would be required to ensure these criteria are met, but in practice assessment of features at a larger scale is often performed from bridges. A typical length for a representative reach is suggested to be 500m. This does not consider the spacing of morphological units related to channel width, or account for differences in morphological (and biotope) heterogeneity, related to channel type (see section 1.4).

PHABSIM was developed in the US to provide a flow recommendation to protect fisheries, and has been used mainly in litigation cases (Lamb and Doersken, 1987). However, the options available in the habitat simulation programmes allow a range of values of WUA to be calculated, which allows manipulation of results to produce a discharge which fulfils political and economic requirements (Gan and McMahon, 1990). The complexity of PHABSIM both in the simulation procedures and interpretation of outputs are further barriers to its wide acceptance and use. An indication of the number of hydraulic models and required simulation procedures is provided in a report describing the application of PHABSIM in two British rivers (Bullock *et al.*, 1991). To date PHABSIM has been applied to approximately 50 sites in the UK (Dunbar *et al.*, 1996); however applications are mainly limited to assessments of flows to protect fish in lowland chalk streams; it has not been used to set any MAFs (Gibbins, 1996; Petts, personal communication). This is partly due to the species-specific nature of habitat suitability indices (Armitage and Ladle, 1991; Mountford and Gomes, 1991; Petts *et al.*, 1995) and the problem of selecting representative reaches, which limits the application of the model to site-specific flow assessments. The model's complexity and the specialist technical expertise required to perform hydraulic simulations were just one of the criticisms of PHABSIM outlined by South African researchers (King and Tharme, 1993). Problems of extrapolation of results and the cost of simulating the model are the main practical factors preventing the wider implementation of PHABSIM.

The final limitation of PHABSIM is its restricted output; as Petts *et al.* (1995) reveal, a minimum flow requirement for invertebrates does not necessarily correspond to one for a particular fish species, and this in turn differs for another species of fish. A true **Ecologically Acceptable Flow (EAF)** would accommodate the flow requirements of the overall biotic community, rather than a single 'target' species. Fausch *et al.* (1990) recommend that flows should be set to maintain native fish populations as these are acknowledged to be good indicators of overall environmental quality. In order to achieve this objective, more general habitat criteria than the distribution of microhabitat variables are needed. As Belaud *et al.* (1989) note, fisheries management involves the division of rivers into "typical flow zones" e.g. pool, rapid etc.

Factors other than flow may bring improvement to overall habitat, for example physical habitat restoration (Petts *et al.*, 1995). Willow roots provide a different micro-environment to cobble substratum, despite similarities in hydraulic conditions at channel margins (Jenkins *et al.*, 1984). Jowett (1992) used biological modelling to determine the habitat requirements of adult brown trout. He concluded that the most important variables were adult trout habitat, food production , instream cover and water temperature. The habitat requirements of adult trout were defined as *"deeper areas or pools"* and food (invertebrate) production dependent on *"shallow fast-flowing water with coarse substrate"*; criteria which no not require sophisticated instream flow incremental modelling to identify a 'good trout stream'. Geomorphologists and ecologists have much to offer water resource managers in the identification of features of value to ecology or fisheries (Swales and Harris, 1995). In practical river management, the application of PHABSIM is just one option which should only be carried out when the costs both ecologically and financially) of misallocating flow are high (Petts *et al.*, 1995). Where flow management is appropriate but the costs of a PHABSIM simulation are prohibitive, an **environmental weighting (EW)** system is advocated, based on sensitivity of subcatchments to flow abstraction (Drake and Sherriff, 1987). In their study, environmental weighting was assessed by consideration of six environmental factors, and the resulting score used to calculate maximum permitted abstraction and minimum river flow for each subcatchment. For management of flows across a range of channel types, geomorphology offers a classificatory framework within which the response of channels to flow may be compared, and ecologically acceptable flows set for a particular channel type. This is currently being developed as a methodology for assessing abstractions within the NRA's national research and development programme, using macrophyte and invertebrate data to assess sensitivity to flow in sixteen classes of channel, defined by 'river channel properties' (width:depth ratio, stream order, altitude, substrate, base-flow index) (Sir William Halcrow and Partners, 1995).

1.4 INCORPORATING GEOMORPHOLOGY IN FLOW MANAGEMENT: CHANNEL TYPOLOGIES

An environmental weighting scheme must incorporate channel morphology and habitat type for accurate assessment of MAFs. Swales and Harris (1995) outline the benefits of an "Expert Panel Assessment Method" which allows specialist geomorphological and ecological knowledge to be directly communicated in the assessment of flow percentiles. It does not require extensive field measurement, but is site specific, therefore extrapolation of flow percentiles is not possible. An appropriate scale for extrapolation of prescribed flows is one in which channel morphology and hydraulics are broadly similar. The corollary of this is described by Petts *et al.* (1995):

> *"...sectors with different characteristics require different flows to maintain habitat for target species."*
>
> Petts *et al.* (1995, p8)

Classification is a means of organising information on different rivers for water resource managers (Petts, 1994). In order to consider processes which occur over different spatial and temporal scales, a hierarchical channel classification ranging from the drainage basin to the microhabitat is proposed (Hawkins *et al.*, 1993; Frissell *et al.*, 1986; Wadeson and Rowntree, 1994). Classification to assess a rivers conservation potential has been outlined by Naiman *et al.* (1992). To assess instream flow requirements, these authors propose three hierarchical scales; types, sectors and reaches. Types are defined as 'upland', intermediate' and 'lowland' depending upon their location within a river network; a division which roughly corresponds to Davis' division of rivers into 'youth', 'maturity' and 'old age' (Davis, 1899). Sectors may be defined as sections of river with equal sediment loading and hydrological regime. In the field a change of sector may be identified by tributary confluences, by changes of valley and channel morphology, or by changes in riparian vegetation (Petts, 1994). Reaches exist within sectors; their boundaries occur where channel morphology and riparian vegetation alter.

Naiman *et al.*, (1992) propose that flows should be managed at the reach scale. Reconnaissance fieldwork carried out as part of site selection for this study suggested that, within any reach, certain biotopes would be 'critical' locations for flow management within the reach. These views were reflected by Kershner and Snider (1992). Several authors describe a classification of habitat types at a scale corresponding to Frissell *et al.* (1986)'s "riffle-pool" system (Bisson *et al.*, 1982; de Leeuw, 1981). The fundamental research issue is how to establish, if possible, where reach boundaries will occur from classifications based on map-derived indices, and how these relate to smaller scale, field-recognised physical habitat features. In an attempt to operationalise a classification of rivers based on channel morphology and physical features the Environment Agency (EA) are developing a river typology based on a national inventory of the physical structure of UK rivers (Raven *et al.*, 1996). The progress of this empirical typology has influenced the broad directions taken at several stages of the research reported in this thesis; this study has also made inputs to the River Habitat Survey (RHS), as discussed in section 7.4.

Analysis of the national inventory of RHS sites has been used to develop a preliminary typology of 'semi-natural' rivers in England and Wales consisting of eleven segment types, based on map-derived variables which are considered to be the major influences on physical features: solid geology, altitude, slope and size expressed as flow category (Raven *et al.*, 1996). The typology has recently been modified to give nine segment types with characteristic physical features (Fox *et al.*, 1996). RHS uses the term "features" rather than physical biotopes, mainly to encourage a standard, descriptive assessment of in-channel hydraulic habitat, based on consistent recognition of features by non-specialist surveyors (see section 2.4). A likely application for the typology is to provide a framework for assessing the following river research and management issues:

1. Habitat Quality Index (proposed by EA)
    - the degree of channel 'naturalness' or modification;
    - the probable impact of physical modification,
    - appropriate sites and priorities for river rehabilitation/ restoration

2. Flow management:
    - ecologically acceptable flows for 'critical biotopes' or reaches and 'representative reaches'
    - possible impacts of water transfers.

3. Empirical tests of stream ecosystem theories and links with segment types:
    - River Continuum Concept
    - Patch dynamics and 'hydraulic stream ecology'

The incorporation of physical biotopes and morphological units within hierarchical, catchment scale models has been discussed by several authors (Frissell *et al.*, 1986; Minshall, 1988; Stalnaker *et al.*, 1996; Wadeson and Rowntree, 1994). Links between scales and the practical problem of extrapolating information between scales and disciplines are discussed by Newson (in press). It is recognised that the NRA's segment typology is based on map-derived variables, rather than field-recorded physical features. The physical biotope provides a means of integrating between the traditionally different scales of ecology and geomorphology; its has relevance to both 'patch dynamics' and fluvial processes. Preliminary RHS results indicate that the biotope is one of a range of characteristic (field-based) physical features which relate to segment type (Raven *et al.*, 1996). As Newson (in press) explains:

> *"The range of river channel features and phenomena now reported in the literature has become sufficient for geomorphologists to attempt channel classifications and typologies... Objective classification and subjective typologies at the channel scale then offer a basis for hierarchical switching between large and small scale habitat assessment and management needs."*
>
> Newson (in press)

1.5 IN-CHANNEL FLOW: CONVERGENCE OF APPROACHES AND APPLICATIONS

This thesis was prepared during a period when a convergence of three factors produced new opportunities to progress links between studies of water management, freshwater ecology and fluvial geomorphology:

1. Demand for extensive and effective information/techniques.
2. Debate over relationships between process and form: channel classification and application.
3. Debate over patch dynamics versus river continua: the rise of physical habitat studies/surveys.

The work described in this thesis has therefore developed from the needs of those with urgent applications for the work (it began at the time of proposals for transfers of Kielder Water to the rivers of Yorkshire - a project revived by the 1995 drought). Those concerned with flood defence, both capital schemes and maintenance, have also been pleading for detailed guidance on altering channel form and substrate, as opposed to flow. River restoration schemes, both large scale (the European LIFE funded River Restoration Project, RRP) and small scale (The Tweed Foundation) have moved from the assumption that "adding riffles" constituted habitat enhancement to expressing a need for gains in instream habitat diversity. The river conservation function of water management bodies, has, internationally, sought improvements to corridor surveys incorporating concepts such as "habitat quality indices" (see sections 5.2 and 7.4). In the UK, Scottish Natural Heritage have developed SERCON (System for Evaluating Rivers for Conservation). Both SERCON and RHS have developed during the period of research presented here to address the following issues:

- Statutory responsibilities
- Impact assessment / development scenarios
- SSSI selection
- River Restoration schemes
- EC Habitats Directive
- Ecological Quality of Water (EC directive)

In addition to these pressing needs for applications, there has clearly been a stimulating theoretical environment in which the geomorphological detail of river channels has become highly relevant to ecologists. Transfers of information, properly "packaged" between geomorphology and ecology have never been more appropriate (Newson, in press). It is hoped that the research reported here will become part of a process of exchange between two disciplines and between application and theory. The aim of this thesis is to develop the core scientific investigation but to stress direct applications, or links to applied fields, where appropriate. The primary aims and objectives are listed in the final section of this chapter.

1.6 AIMS AND OBJECTIVES

The main scientific aims of the research described in this thesis were decided at the beginning of the project and are summarised in Table 1.1. The broad overview in Table 1.1 has been followed throughout the course of the research and is described in a logical sequence in this thesis. The work in Chapter 6 developed initially as a result of problems encountered during the collection of field data and the need for accurate representation of hydraulic conditions. Secondly, the research described in Chapter 6 developed in response to calls from ecologists (especially invertebrate biologists) to add an ecological dimension to the work. Thus it is not a central aim of the thesis but an extension of the work. Figure 1.1 illustrates the major components of the research presented in this thesis, the first of which is described in the following chapter.

Core scientific objectives	Reference Chapter(s)
1. Identify the types of morphological units and the distribution of physical biotopes which are present in a range of geomorphological channel types. Select a range of sites to include the full range of existing features.	2, 3
2. Develop standard techniques for the identification of physical biotopes at different sites and discharges, based on a subjective classification scheme.	3
3. Test the subjective classification of physical biotopes using objective statistical analyses.	4
4. Describe how flow 'fills the channel' in terms of mesoscale physical biotopes, considering both their spatial and temporal distribution.	5
5. Use the spatial and temporal distribution of physical biotopes to develop a methodology for manipulating flows in different geomorphological channel types.	5, 7
6. Attempt to provide a method of extrapolating site-specific results to catchment-scale assessments of the distribution of instream features (physical biotopes).	7

Table 1.1: Core scientific objectives of the research described in this thesis.

Additional studies in which the biotope approach has been applied, but which are not included in the main objectives of the thesis are described in section 7.2 and Appendix B3.

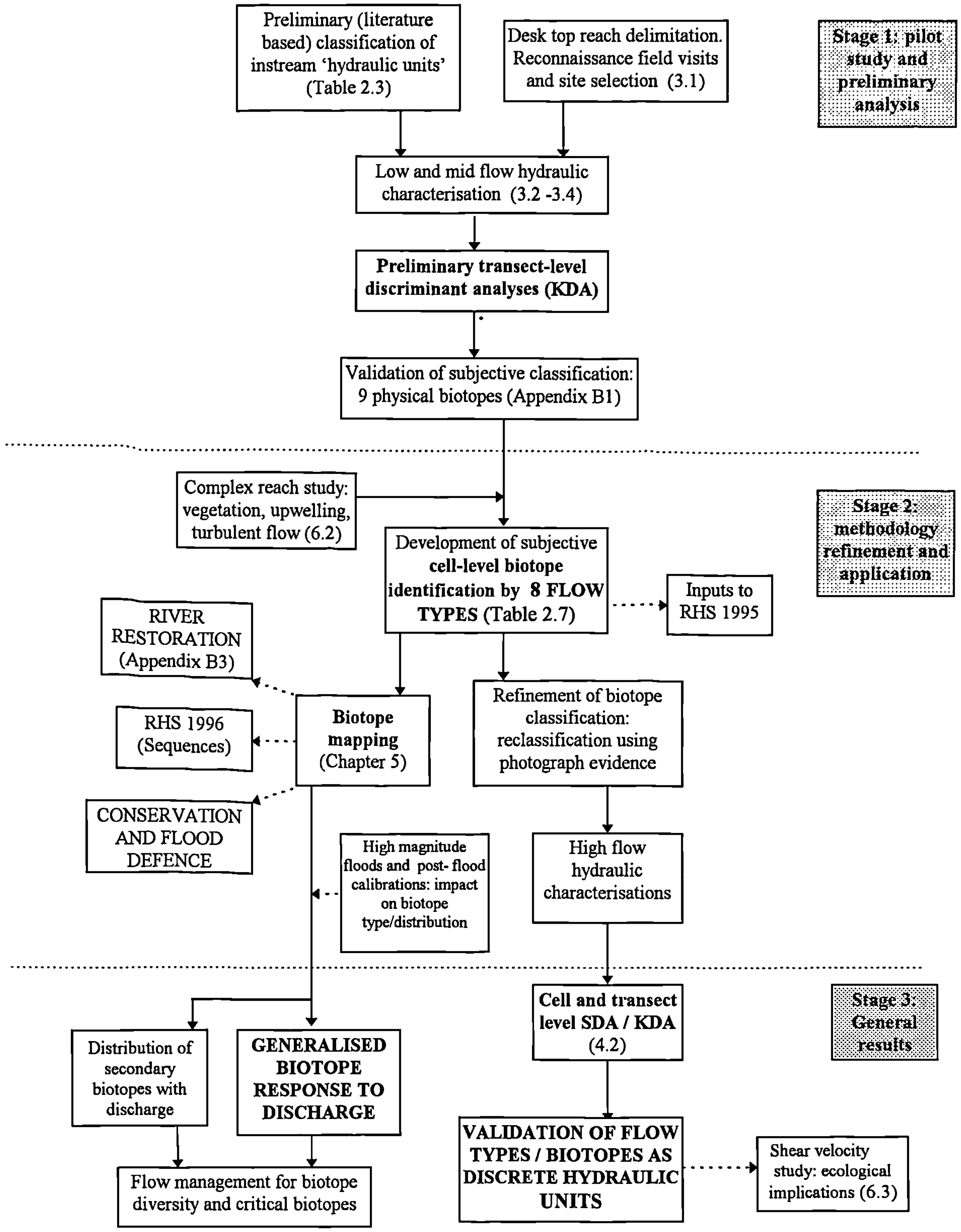


Figure 1.1: Schematic diagram of the major thesis components. Bold capitals indicate the major research findings; capitals represent practical applications to NRA/EA; bold lower case describes techniques/ methods. Numbers in brackets refer to sections or tables.

2. IDENTIFICATION AND CHARACTERISATION OF INSTREAM HABITAT FEATURES: A LITERATURE REVIEW

Chapter Overview

This chapter reviews the ecological basis for a study of instream physical habitat, together with the literature regarding the identification and description of instream habitat for fish and invertebrates (2.1). The influence of fluvial geomorphology on ecological paradigms and research practices is discussed, together with recent interest in, and development of, biological sampling within a range of biotopes (2.2). The identification of biotopes as the basis of surveys of mesoscale hydraulic habitats for fisheries management is described (2.3).

The need for a standard approach to the identification of 'instream habitat' is discussed in the context of national river surveys and habitat inventories within the UK. A glossary of terms used to describe morphological and hydraulic units in this, and other studies is presented (2.4). A summary of the biotopes recognised in this study is presented, together with descriptions for standard field identification. The background to national habitat surveys and the development of a UK river typology to the selection of study sites for this thesis is discussed. Links between this study and use of PHABSIM in the assessment of ecological flows are presented (2.5).

2.1 INSTREAM PHYSICAL HABITAT: THE SEARCH FOR KEY VARIABLES

In a description of physical habitat, Bovee (1982) makes a distinction between macrohabitat and microhabitat variables. The former include temperature, water quality, sediment size and load, channel shape, slope and flow regime, and such factors differ at the sector or catchment scale (Frissell *et al.*, 1986; Petts, 1994). Microhabitat variables include distributions of depth, velocity, substrate and cover, and it is these variables which contribute to 'habitat hydraulics' and influence 'hydraulic stream ecology' as described in section 1.2.1. However these operate at smaller spatial scales; the ecologists' concept of patch dynamics can be perceived over a range of scales from the riffle-pool to individual substrate particles (Minshall, 1984; Pringle *et al.*, 1988; Palmer *et al.*, 1995). The latter offers detailed ecological information, but is not readily capable of being extrapolated or included in catchment-scale surveys and evaluations of instream habitat.

Percival and Whitehead (1929) were amongst the first ecologists to recognise the importance of **substrate** on benthic invertebrate distributions, relating species associations to seven substrate classes. The importance of bed structure and vegetation to substrate stability and invertebrate habitat are emphasised in addition to mean grain size; e.g. "loose stones" are shown to have different biotic assemblages compared with "cemented stones". Pennak and Van Gerpen (1947) visually identified four substrate types and described their associated benthic fauna. Other workers described the influence of substrate on individual species (Linduska, 1942; Thorup, 1966) during an 'era of discovery' in stream ecological theory (Minshall, 1988). Later during the 'era of refinement and experimentation' several researchers found correlations between benthic invertebrate distributions and substrate size classes (Mackay, 1969; Cummins and Lauff, 1969; Allan, 1975; Harman, 1972). Cummins and Lauff concluded that particle size and food supply were of primary importance in explaining microinvertebrate distributions, with current velocity and temperature of secondary importance. Reviews of substrate-biota relationships are provided by Cummins (1962), Hynes (1970) and Minshall (1984). In an extensive literature review, Cummins (1974) concluded that no single factor has greater ecological significance in the structure and function of stream ecosystems than the physical nature of the substrate. However, the reach scale microhabitat variables like substrate tended to be subordinated in the nutrient-based models of the 1970s and early 1980s e.g. the River Continuum Concept (Vannote *et al.*, 1980).

During the 1980's interest in the role of substrate on invertebrate distributions was revived, partly due to interest in instream flow incremental methodologies (IFIM) (see 1.3); but with a new emphasis on substrate heterogeneity rather than mean particle size (Williams, 1980; Minshall, 1984; Erman and Erman, 1984). Geomorphological descriptions of substrate size conventionally record the intermediate diameter (b axis) of standard sample of 100 pebbles (Wolman, 1954), and describe the distribution of substrate as a size (or phi scale value) at which a percentage of the sample is finer (usually D_{50} or $D_{84,}$ corresponding to those substrate sizes at which 50% and 84% of the sample are finer). Such distributions have been used to predict invertebrate abundances (Armitage, 1989). Although providing a standard description of substrate size distributions, precise phi values have little relevance to ecologists who refer to *invertebrate* preferences for substrate in terms of descriptive classes such as 'gravel' and 'pebble'. Cummins (1962) modified the Wentworth particle size classification into the classes outlined in Table 2.1. Cummins' classification (with the addition of a bedrock category) was used to illustrate invertebrate-substrate relationships by Minshall (1984). By comparison, a classification of substrate size for *fisheries* management suggests an upper size limit for gravel of 64mm compared to 16mm for invertebrates (Milner *et al.*, 1985). What is considered to be 'small gravel' for fish includes coarse, medium and fine gravel for invertebrates. At a larger scale e.g. for comparisons of substrate size distributions between sites (often in different reaches or sectors), coarse and fine gravel categories are combined with sand to produce a single 'gravel/sand' class in River Habitat Surveys (National Rivers Authority, 1995; see section 2.4). Different class boundaries depend on both the nature of the discipline and scale of study, and reveal a lack of standards for descriptive substrate classes.

Particle size range (mm)	*Invertebrate substrate classes*	*Fisheries substrate classes*	*Reach scale substrate classes*
-	Bedrock	Bedrock	Bedrock
> 256	Boulder	Boulder	Boulder
128 - 256	Cobble - large	Cobble - large	Cobble - large
64 - 128	Cobble - small	Cobble - small	Cobble - small
32 - 64	Pebble - large	Gravel - large	Gravel - coarse
16 - 32	Pebble - small	Gravel -medium	"
8 - 16	Gravel - coarse	Gravel - small	Gravel - fine
4 - 8	Gravel - medium	"	"
2 - 4	Gravel - fine	"	"
0.06- 2	Sand	Sand	Sand
0.04 - 0.06	Silt	Silt	Silt
< 0.04	Clay	Clay	Clay

Table 2.1: Modified Wentworth-scale substrate size classification (after Cummins, 1962; Minshall, 1984; Milner *et al.*, 1985; National Rivers Authority, 1995).

The classification of substrate size into gravel and cobble categories is somewhat academic; species composition tends to show gradual changes with increasing sediment size (Rabeni and Gibbs, 1980; Sheldon and Haick, 1981; Barmuta, 1989). Moreover it is the interaction of flow and sediment which are more likely to directly influence species distributions via factors including hydraulics and deposition of organic matter. Nevertheless, substrate size is a major descriptor of channel geomorphology and river type (Clark *et al.*, 1995), and remains a key variable in all studies of instream habitat.

Substrate and its direct influences appear to be the most important single influence on stream ecosystems. Allan (1995) states that velocity and its associated physical forces collectively represent the most important environmental variables for instream biota. However, the role of **current** has, historically received less attention than that of substrate, possibly due to the lack of sophisticated equipment for measuring velocity. Research has been undertaken regarding the influence of mean current velocity on general invertebrate distributions (Statzner, 1981; Stevenson, 1984) and fish habitat (Binns and Eisermann, 1979). Benthic communities are adapted to the complexities of near-bed flow, but there exist difficulties in characterising flow in a biologically meaningful way. Since the early studies (Ambuhl, 1959; Smith, 1975), research into near-bed hydraulics has been performed by relatively few authors (Statzner and Higler, 1986; Davis and Barmuta, 1989; Carling, 1992). More emphasis has been placed on the preferences of velocity and depth for individual species of economic importance, e.g. brown trout (*Salmo trutta fario*) (Krog and Hermansen, 1985); trout and Atlantic salmon (*Salmo salar* L.) (Crisp and Carling, 1989). However, hydraulic geometry studies have shown only modest increases in velocity (especially downstream) with significant changes in discharge (Leopold and Maddock, 1953; Wolman, 1955), and as Hynes (1970) suggests, abrasion and substrate shifts are more important than velocity *per se*. Studies have focused on the effects on individual species, and with the development of instream flow incremental methodologies (IFIM), on particular life-stages (see section 1.3).

IFIM attempts to incorporate the combined influences of microhabitat variables to predict changes in habitat with discharge (Bovee, 1978; Gore, 1978; Gorman and Karr, 1982; Orth and Maughan, 1982; Shirvell and Dungey, 1983). Much emphasis has been placed on determining 'habitat suitability' for various life-stages of brown trout (*Salmo trutta fario*) e.g. Belaud *et al.*, (1989); Chaveroche and Sabbath, (1989). However, 'Weighted Seabee Area' produced in PHABSIM simulations is not readily understandable without specialist training; nor can it easily be transferred to other sites. The development of output in conceptually simpler terms to WUA is a collaborative R&D priority of the PHABSIM User Forum Group which includes representatives from the Environment Agency, the Institute of Hydrology and Higher Education Institutes. The majority of water management issues must be implemented with limited financial and human resources, which illustrates the need, in the majority of cases, for a more rapid methodology for the assessment of flow changes on habitat.

It is clear that the microhabitat variables (depth, velocity and substrate) cannot be treated in isolation (Huryn and Wallace, 1987; Quinn and Hickey, 1994); the growing consensus is that the interaction between substrate, hydraulics and depth is of greater ecological importance than any one variable (Brown and Brown, 1984; Nowell and Jumars, 1984). Individual variables may be more relevant than others in different cases, e.g. depth may have an important non-hydraulic influence on collector-browsers (Quinn and Hickey, 1994) and also on fish requiring shelter sites. This cannot be accounted for in PHABSIM, which assumes equal and independent influence of the three microhabitat variables. A methodology is therefore needed which integrates current and substrate without combining them into a single index, has ecological relevance, and importantly, which can be readily understood by practitioners across a range of disciplines. Rabeni and Jacobson (1993) recognise the benefits of managing hydraulic habitat units, rather than attempting to create appropriate microhabitat conditions:

> *"An advantage of hydraulic habitat units as a management tool is that several habitat conditions usually occur together in a unique combination that fishes favour, and managing for particular hydraulic habitat units automatically includes appropriate combinations of several habitat variables".*
>
> Rabeni and Jacobson (1993, p215)

The standard identification of "hydraulic habitat units" would therefore provide a significant contribution to the assessment of physical habitat for instream biota.

2.1.1 The biotope as an integrator of physical factors

Opinion on community theory has traditionally been divided between the organismic and individualistic concepts. A comprehensive description and review of their influence on stream ecosystem theory and research is provided in Lake and Barmuta (1986). In support of the organismic view, Shelford (1911, 1913) proposed that there were discrete "rapids" and "pool" communities, consisting of species with similar environmental preferences. Later, Moon (1939) classified the freshwater environment into visually distinguishable habitat units, making a division between erosional (coarse substrate, rapid flow) and depositional (fine sediment, reduced flow) environments to explain their discrete biotic associations. In the context of this study, Berg (1948) made a significant statement regarding community theory, namely that ***the habitat or biotope is the 'unit of the system' and that each habitat would support a distinct biota distinguished by 'characteristic' species***. Characteristic species may be classified by functional feeding groups; Huryn and Wallace (1987) identified 'scrapers', 'engulfers' and 'collector-filterers' in association with riffles, pools and bedrock respectively. Collector-gatherers were ubiquitous in all three biotopes, which emphasises the need to identify characteristic species. More recently, Barmuta (1989) classified invertebrate functional feeding groups from an upland stream into depositional and erosional sets, by the relative abundance of species which are tolerant of turbulent flow regimes and able to cling to large substrate. He concluded that the substrate size distributions and range of hydraulic conditions over the normal discharge regime are likely to influence biotic assemblages for the site as a whole. Clearly, however, there is more spatial (and temporal) variability of instream hydraulic conditions and habitat than is suggested by such simplistic divisions. Recent developments in stream ecosystem theory concerning 'patch dynamics', are prompting invertebrate ecologists to consider substrate and hydraulic heterogeneity at a scale smaller than Frissell *et al.'s* (1986) "riffle-pool" system (Minshall, 1984; Pringle *et al.*, 1988; Death, 1995; Palmer *et al.*, 1995):

> *"Traditionally, aquatic ecologists have designated riffles, pools, runs etc. as relatively homogeneous units for study ... patchiness within streams at scales perceived and/or exploited by stream organisms is a reality that has often been ignored."*
>
> Pringle *et al.* (1988, p504)

There is an inherent contrast in the scale of descriptions applied to physical habitat units by those specialising in fish and invertebrates. Fish biologists have tended to identify and classify units at a scale larger than that normally considered by invertebrate ecologists, and are responsible for the belief that biotopes are "homogeneous". Fisheries managers perceive rivers and streams as 'habitat types' within broader scale reaches and sectors (Minshall *et al.*, 1988). Even during the phase of description and early quantification (mid 1920s - 1950s), fish habitat was conceptualised at the mesoscale i.e. by the identification of ecologically relevant *habitat units* rather than substrate or current descriptions. Allen (1951) made a pioneering attempt to hydraulically characterise visually identified habitat units (see Table 2.3), on the basis of depth and velocity distributions, together with current descriptions.

It is evident that the scale at which ecological studies are conducted (and therefore the choice of habitat descriptors) is dependent upon the organism and process of interest. Localised substrate size *per se* is of interest to studies of macrophyte, invertebrate and fish distribution; boundaries for substrate size classes depend on whether the descriptions refer to spawning gravels for fish, attachment sites for invertebrates or reach-scale site descriptions (Table 2.1). Larger scale 'substratum patches' are appropriate for studies of mobile macroinvertebrates and fish (Minshall, 1988). If 'habitat units' are to be incorporated into larger scale, hierarchical models of reaches, sectors and catchments (Frissell *et al.*, 1986; Minshall, 1988; Wadeson and Rowntree, 1994), Minshall's habitat type is the smallest practical unit. The benefits of adopting a mesoscale approach to habitat identification are summarised by Harper *et al.*, (1995):

> *"Accurate identification of distinct habitats in the aquatic environment has an intuitive value, since habitats link the impacts on the natural environment and its inhabitants. Moreover, habitats have considerable management potential for the simple reason that they can be recognised visually on the river bank whereas many species and ecological functions cannot".*
>
> Harper *et al.* (1995, p222)

Adoption of a mesoscale approach to habitat identification provides the geomorphologist a prime role in the integration of ecology and hydraulics. The next section discusses the terminology used to describe biotopes in the context of ecological, geomorphological and fisheries research. A literature review of instream habitat features was divided between invertebrate ecology and fisheries references. The influence of geomorphological terminology on ecological research theory and practice is discussed. This differs for reasons given above, from the South African literature review by Wadeson (1994), in which invertebrate ecology, geomorphology and fisheries references were combined into one comprehensive table of "biotope terms".

2.2 INVERTEBRATE HABITAT: SIMPLE UNITS FROM GEOMORPHOLOGY

By far the dominating reference to physical habitat features by invertebrate biologists is the riffle-pool sequence. The riffle-pool sequence has its roots in fluvial geomorphology, with riffles and pools being described as natural channel feature which form as a result of flood scour (Wolman, 1955; Leopold and Wolman, 1970; Brookes, 1994). The earliest cited reference to riffles and pools appears to be in 1927, although this reference was privately printed by Pettis and is therefore unavailable. As Leopold *et al.* (1964) note, these terms were originally used by fishermen:

> *"At low flow the water surface over a pool and riffle sequence tends to consist of alternating flat reaches of low gradient and steeper reaches often involving white water. This appearance of smooth water over the "pool" and "riffles" over the bar - terms well known to trout fishermen - led us to use these terms in describing the feature".*
>
> Leopold *et al.* (1964, p206)

Differential patterns of entrainment, transport and deposition of sediment during floods shape the distribution of substrate particles into morphological units known as riffles and pools, at a more or less regular spacing of 5 to 7 times channel width (Stuart, 1953; Leopold *et al.*, 1964; Yang, 1971; Lisle, 1979; Knighton, 1984). Ferguson (1981) suggests 3 to 10 channel widths is a more realistic spacing interval. Field process studies of the riffle-pool sequence have, until recently, dominated geomorphological research (Newson, 1995), investigating aspects of sediment size (Milne, 1982; Clifford, 1994), hydraulics (Carling, 1991; Clifford and Richards, 1992), sediment transport (Campbell and Sidle, 1985; Sear, 1992) and applications to assessing the impacts of channel or riparian development (Newson and Sear, 1994; Sear *et al.*, 1995). A natural riffle-pool sequence is considered to provide a variety of flow conditions and habitats required by invertebrates via heterogeneous substrate size distributions and cross-sectional morphology; loss of this sequence through channelization creates unsuitable hydraulic habitat (Huggins and Moss, 1974; Moyle, 1976; Brookes, 1994). Riffle-pool spacing has been used as an indicator of channel stability and degree of 'naturalness' (Brookes, 1994; Wood-Smith and Buffington, 1996), with the ratio of riffle:pool being used to evaluate potential fish habitat at different life stages (Beschta and Platts, 1986; Platts *et al.*, 1983, 1987; Bayley and Li, 1992). Studies of hydraulic conditions within riffle-pool sequences are discussed in the context of sediment transport and near-bed boundary conditions for invertebrates (Carling, 1992a, 1992b).

This study, focusing on a range of channels at various positions within catchments of different geologies (see Table 3.1); has revealed a range of biotopes previously 'ignored' or poorly classified by ecologists. Geomorphologists recognise that riffles and pools may be poorly developed or even absent from boulder-bed streams (Miller, 1958; Leopold *et al.*, 1964), where they may be replaced by a step-pool sequence (Whittaker and Jaeggi, 1982). Sand or silt channels with little substrate heterogeneity tend not to form riffles and pools (Leopold *et al.*, 1964). However, the geomorphological riffle-pool classification was adopted by invertebrate ecologists to reflect the terms 'lentic' and 'lotic' (Young, 1992), meaning 'fast-flowing water' and 'slow-flowing water' respectively (Maitland, 1990). The terms 'lentic' and 'lotic' were developed to distinguish between riverine and lacustrine systems; mis-use of these terms in river systems has reinforced a simplistic, two-way classification of instream habitats. As Cummins (1992) states:

> *"...there has been little substantive modification of the separation of running-water habitats into erosional and depositional, usually referred to as riffles and pools, since the classic paper of Moon (1939)"*
>
> Cummins (1992, p239)

Invertebrate sampling at a national scale (RIVPACS) has focused on sampling within riffles (see 1.2.2), as an indicator of water quality at the catchment scale. A recent version of RIVPACS collects data from "all habitat types" (Wright *et al.*, 1994), but these are not formally defined. In practice this usually involves the riffle and any marginal areas which are considered to be important (Cowen, personal communication). Local scale studies of habitat 'patchiness' at the mesoscale have focused on the comparison between riffle and non-riffle fauna; for example, Huryn and Wallace (1987) classified channel units into bedrock outcrops, riffles and pools, on the basis of distinct substrate size distributions and mean velocity. A literature review of the terms used to describe instream habitat units, together with units which are recognised and sampled in invertebrate studies is summarised in Table 2.2.

Biotope	Number of references
Riffle	30
Pool	27
Run	8
Glide	1
Cascade	1
Rapid	4
Bedrock outcrop	4
Marginal deadwater / macrophytes	5

Table 2.2: References to biotopes in invertebrate ecology literature (30 papers).

2.2.1 Ecologically recognised habitats

It is evident that riffles and pools dominate both the sampling strategy for comparison of species distributions and the perception of instream habitat units (biotopes) within stream ecosystems (Minshall, 1988). Carling (1995) makes an ecological distinction between step-pool systems in steep, mountain channels (Chin, 1989) and the lowland riffle-pool sequence (Logan and Brooker, 1983). With the realisation that the riffle-pool classification is insufficient to describe the range of instream habitats, invertebrate ecologists began to recognise additional biotopes. River Corridor Surveys (National Rivers Authority, 1993) made a significant contribution to 'observational surveying' which revealed the limitations of a riffle-pool classification. The instream habitat most frequently referenced and sampled by invertebrate ecologists, after *riffles* and *pools*, is the *run*. This term has existed in fisheries literature (Allen, 1951) and fictional references to sport fishing (Duncan, 1983), but is not properly defined and therefore much understated in invertebrate ecology. The subdivision of habitats beyond riffles, pools and runs is lacking (Smith *et al.*, 1990). *Glides* tend to be classified as either pools or riffles, depending on relative depth and velocity; *rapids* and *cascades* are considered together with riffles (Cummins, 1992).

More recently, invertebrate ecologists have shown interest in a wider range of biotopes, particularly *marginal deadwaters* and their associated vegetation (Rutt *et al.*, 1989; Harper and Smith, 1995; Humphries *et al.*, 1996). The role of marginal deadwaters as refugia has been illustrated (Lancaster and Hildrew, 1993a; Gibbins, 1996). Macrophyte vegetation *per se* is considered to be a major habitat type by invertebrate ecologists, although this applies mainly to low gradient channels where vegetation increases biotope heterogeneity and patchiness (Rooke, 1984; Hearne and Armitage, 1993;

Dallas *et al.*, 1994; Newall, 1995). Brooker (1982) includes both 'hydraulic units' and categories of vegetation i.e. tree roots, grass roots, marginal species (e.g. *Callitriche* spp.) in his classification of 'habitat types'. Armitage *et al.* (1995) describe eight mesohabitats which include a combination of substrate and vegetation types. Shingle banks provide habitat for many rare UK invertebrates (Fowles, 1988) as do emergent river sediments i.e. gravel bars which become exposed at low flows (Eyre, personal communication). Bedrock outcrops offer important habitat for filter-feeding caddisflies (Freeman and Wallace, 1984; Smith-Cuffney and Wallace, 1987; Huryn and Wallace, 1988) as does chute flow in cascades for simulid larvae (Palmer, 1991). In general, however, invertebrate sampling strategies tended to define 'habitats' on an *ad hoc*, site-specific basis (within a broad riffle-pool intuition) according to the nature of the habitats present. The ecological potential of the biotope is recognised (Dallas *et al.*, 1994; Fozzard *et al.*, 1994), but few studies have quantified invertebrate-biotope relations.

2.3 FISHERIES HABITAT: CLOSE OBSERVATION, BROAD DESCRIPTION

Some authors have adopted the geomorphologists pool-riffle description to describe fish habitat (Herrington and Dunham, 1967), identifying and distinguishing between these two units in terms of relative depth. Traditionally, however, anglers and fisheries managers have use a broader terminology to describe 'hydraulic habitat units' or biotopes e.g. riffle, rapid, run, glide, pool and eddy (Duncan, 1983; Walton, 1906). Attempts to classify instream habitats for fisheries into riffles or pools have proven unsuccessful. In one stream habitat survey, (Platts *et al.*, 1983), data recording criteria required transects to be classified as one of two habitats (riffles and pools) to overcome difficulties of distinguishing between, for example, glide and run. Reconnaissance fieldwork for this thesis involved the mapping of 'riffles and pools' in a planned attempt to establish an appropriate, representative survey length. However, field observations at a range of sites indicated that this classification was insufficient for the range of instream habitats identified (see Table 2.3, p48-49). Lack of precise characterisations and associated descriptions for the range of identified biotopes, has often resulted in the past **inappropriate adoption of the invertebrate riffle-pool classification, and failure to accurately identify other biotopes in a standard manner.**

Qualitative description of hydraulic habitats does exist in the scientific literature (e.g. Kennedy, 1984), but offers no standard procedures or guidelines for their identification. The descriptions of pool, glide, run and riffle by Platts *et al.*, (1983) use *relative* depth and velocity together with water surface gradient. For example, the description of *glide* is: *"too shallow to be a pool and too slow to be a run"*; a *run* is characterised by *"rapid, non-turbulent flow - too deep to be a riffle, too fast to be a pool"*. These subjective descriptions may allow identification of biotopes at a given site (providing all biotopes are present for comparison), but cannot be easily transferred to another location, especially if surveys are conducted at different times. Use of relative depth and velocity comparisons necessitates the collection of several point readings, to ensure that the identification of a particular biotope is within a standardised range of depths and velocities. One of the first attempts to identify biotopes by quantified, ranges of microhabitat variables (depth, velocity and substrate) was conducted on New Zealand trout streams (Allen, 1951). Hydraulic characterisations are based on the range of recorded depths and velocities, together with current descriptions e.g. *riffles* are defined as:

> *"Shallow water with a rapid current and usually a broken flow. Such conditions are often described as 'ripples', 'rapids' or 'riffles'. Velocity more than 38 cm sec^{-1}. Depth less than 23 cm".*
>
> Allen (1951), in Wadeson (1994, p3)

Such 'threshold' hydraulic conditions are based on data averaged over a number of data points, and are likely to be guidelines for biotope characterisation, rather than absolute transitional values. Some attempts to identify and characterise instream habitat by physical variables (Milner *et al.*, 1985) describe habitat in terms of both catchment scale variables (e.g. altitude and catchment area) and site features (depth, substrate and cover). Individual variables of site features were grouped according to fish habitat requirements at different life-stages (e.g. depth 0-20cm), and the percentage of each class estimated to predict fish abundances. However, when attempting to predict biological populations these conditions only apply to *"uniform groups of streams"* and their associated species. Moir *et al* (1996) relate the location of Atlantic salmon spawning habitat (redds) to hydraulic (velocity, depth and Froude number) and sedimentary (particle size) distributions, but do not indicate how these conditions can be identified in the field without detailed measurements.

In general scientific studies, the most successful identification schemes e.g. for wild flowers or invertebrates are based on visible, easily recognised features, rather than characteristic, intrinsic conditions. As Frissell *et al.* (1986) explain:

> *"...each habitat type may have a* ***characteristic pattern*** *of flow velocities, depths and sediment dynamics, which should be of prime importance in determining its suitability as habitat for different organisms"*
>
> Frissell *et al.* (1986, p207)

If this is the case, it may be possible to identify discrete 'biotopes' by surface manifestations of the three variables described above. In a description of 'habitats', Allen uses the terms rapid and riffle to describe the current or flow types. However, in invertebrate ecology these terms are recognised as habitat units (see Table 2.3, p48-49), rather than descriptors of hydraulic conditions. **The problem of confusing features (or biotopes) with flow types existed in many early descriptions, which prevented the development of a standard terminology and identification procedure.** In their description of the riffle-pool sequence Leopold *et al.*, (1964) use flow types and biotopes interchangeably, e.g. 'riffle' is used to describe flow conditions rather than the hydraulic unit (see Table 2.3). The confusion between channel features and flow was also evident in prototype RHS forms (National Rivers Authority, 1994); e.g. riffles (a biotope) were described as a flow type (see Table 2.7, p44).

Description of fish habitat structure in terms of microhabitat variables (depth, velocity, area of cover) tends, like the invertebrate literature, to simplify the concept of biotopes and focuses instead on *riffles* and *pools* (Herrington and Dunham, 1967; Gorman and Karr, 1978). However, the management of instream habitat for fisheries and conservation (rather than maximising production of one commercial species), and the need for methods which allow the rapid assessment of river's conservation value led to the development of a more standard terminology for a wider range of biotopes. As ecological research moved away from the totally quantitative paradigm, the identification of fish habitat moved towards a more qualitative, visually based approach. Identification of habitat units for small (first and second order) streams developed as a means of assessing the value of salmonid habitat (Bisson *et al.*, 1982; Platts *et al.*, 1983; Sullivan, 1986). Comprehensive survey methods were developed as a means of evaluating conservation value and the degree of channel modification, to enable prioritisation of management plans. In some studies the biotopes in Table 2.3 are used but not formally defined e.g. Blyth (1983). In other studies, the 'habitat units' identified have distinct morphological and hydraulic characteristics, defined by reference to slope, substrate size range and depth-velocity ranges (Platts *et al.*, 1983). This approach is subjective and dependent upon position within the channel as well as discharge. Other authors produce icons to assist the identification of 'hydraulic habitat units' on the basis of gradient, substrate size and relative roughness (degree of substrate exposure) (Sullivan, 1986), or topography, low water surface slope and hydrodynamic pattern (Bisson *et al.*, 1982). Both these approaches recognise that biotopes are discharge-dependent and that the surface flow type is characteristic of the 'hydraulic unit'.

Grant *et al.* (1990) used the percentage of flow classified as sub- or supercritical, relative roughness and degree of step development to distinguish between 'channel units' in steep gradient, step-pool channels in Oregon. It is clear however that the percentage area with supercritical flow will change with discharge. Moreover, within sub-critical flow, many discrete hydraulic conditions exist. These are now identified as **flow types** (see Table 3.6, p66). Mosley (1982a) defines six instream 'habitats', using a water surface classification to distinguish between them. Surface flow, described as smooth, rippled or wavy, breaking waves or aerated whitewater is used to identify 'habitats'. Descriptions of the surface manifestation of flow have been used to characterise instream features since the nineteenth century. Probably the most eloquent is provided by Ruskin in his essay 'Of Water as Painted by Turner' (Ruskin, 1843; in Barrie, 1987). Ruskin claims that Turner was the only painter to "accurately represent the surface of calm water or the force of agitated water". His descriptions indicate the usefulness of observational science in identifying hydraulically discrete features (Table 2.4, p36).

Waterfall:	*"...it tumbles over a high fall like this; and it is when we have lost the spring of the fall, and have arrived at the plunge of it, that we begin really to feel its weight and wildness."*
Chute:	*"...water will leap a little way; it will leap down a weir or over a stone."*
Cascade:	*"...it pauses after every leap, and curdles about, and rests a little and then goes on again; and if... it meets with any rock or stone, it parts on each side of it with a little bubbling foam, and goes round."*
Rapid:	*"...in water which has gained an impetus, we have the most exquisite arrangements of curved lines, perpetually changing from convex to concave; all in unison of motion."* *"...if it meet a rock... it will often neither part nor foam, but clear it in a smooth dome of water, so that the whole river has the appearance of a deep and raging sea, with this only difference, that the torrent-waves always break backwards, and sea-waves forwards".*
Run:	*"...if water be rippled, the side of every ripple next to us reflects a piece of the sky, and the side of every ripple farthest from us reflects a piece of the opposite shore."*
Glide:	*"when water is abundant, the effects caused by the hindrances on the bottom are not manifested at the surface, since under such conditions only those due to the effects of the bank are apparent"* * *"...the immense width of the river at this spot makes it look like a lake or sea, and it was therefore necessary that we should be made thoroughly to understand and feel that this is not the calm of still water, but the tranquility of a majestic current."*
Pool / deadwater:	*"...there is hardly a road-side pond or pool which has not as much landscape* in *it as above it."*

Table 2.4: Historical descriptions of flow and associated biotopes (all from Ruskin, 1843: in Barrie, 1987; except *Gugliemini: in Levi, 1995).

It is interesting to note that in the entire volume dedicated to the movement of water, there is no description which corresponds to unbroken standing waves (characteristic of ***riffles***). Ruskin's description of *run* (rippled flow) may have been intended to include both runs and riffles; however it is evident that *riffles* do not dominate the either the author's or artist's perception of flow-related features. These features have, in the main, been retained in descriptions of fisheries habitat; the influence of geomorphology on contemporary views of instream habitat is discussed below.

2.3.1. Geomorphology, fish habitats and a standard terminology

It is clear that much of our appreciation of the diversity of instream physical habitat has come from the fisheries literature, backed by the close observations made by non-specialists e.g. artists. Scientific geomorphology has made a restricted but highly influential impact on invertebrate ecologists but we now ask whether channel morphological units can be linked to, or provide names for the broader range of habitats revealed from section 2.3 above. We need to avoid hybrid terminologies and circular arguments between form and process. **The key to 'morphological units' is their controlling influence on the *gross* properties of local flow; 'hydraulic units' or 'biotopes', are defined by the detail of flow direction, velocity and relative roughness within that local flow.**

Table 2.3 (p49-50) summarises the terms identified in the literature regarding fish habitat conservation and management. Early studies refer to 'habitat' as the basic unit; however, the descriptions indicate that these refer to flow dependent 'hydraulic units' rather than 'morphological units' which are independent of flow. Wadeson (1994) describes *"the widespread and often indiscriminate use of the term habitat by many ecologists"*, and advocates use of the term **'hydraulic biotope'**, which refers to the abiotic environment of a community rather than species. Sullivan (1986) is the first author to recognise the interactions of discharge and morphology, and their influence on hydraulic conditions; she adopts the term *"hydraulic habitat unit"*. The importance of links between morphology and discharge is reflected in later studies: Kershner and Snider (1992) advocate the use of "*fluvial features*" as habitat descriptors; Malavoi (1989) describes techniques for the identification of "*morphodynamic units*", suggesting a temporal, discharge-related change of 'habitat'. Physical biotopes will clearly change with stream discharge because controls on flow will vary as the relative water level changes and morphological units drown out.

The terminology used to describe instream physical habitat depends on the nature of the classification and the specialism of its developer (Bisson *et al.*, 1982; Church 1992). Confusion relates to the separate terminologies which exist within the fields of geomorphology, invertebrate ecology and fisheries (see Wadeson, 1994). Practitioners in these fields developed separate terminology and procedures for identification which resulted in similar biotopes being described differently, or common terms being used to describe different biotopes. As Dallas *et al.*, (1994) discover, the naming of biotopes is "*frequently haphazard and inconsistent*". Church (1992) notes that different terminology is used depending on the purpose of the classification. In geomorphological terms, 'morphological units' are described within the context of hierarchical classifications (Frissell *et al.*, 1986; Grant and Wolman, 1990). A geomorphologist's 'riffle-pool' would be classified as 'riffle-glide' 'for habitat purposes" (Church, 1992). Links between the units used for 'habitat' classifications and the 'morphological units' of geomorphological classifications are explained clearly by Church (1992); those 'habitat units' which are not distinguished in strict morphological terms are often omitted from geomorphological classifications, to produce a simplistic riffle-pool classification. In RHS the term 'channel feature' was originally adopted by the NRA, although this includes both in-channel and bankside features (NRA, 1994), which represent two scales of habitat, namely hydraulic and morphological units (see Table 2.5, p39). With the realisation that different hydraulic units existed within morphological or channel units, the term **physical biotope** (biotope for brevity) was adopted. This is equivalent to Wadeson's (1995b) 'hydraulic biotope'. A glossary of the terms used in this thesis, together with those used by other authors is given in Table 2.5.

Waterfalls are described in only 4 of the 20 papers reviewed; in most cases waterfalls are viewed as a barrier to fish passage and are not included as 'habitat'. Confusion arises over the terminology used to identify steep, headwater channels with boulder-dominated beds. The characteristic morphological units of such channels form a step-pool sequence consisting of a series of more or less regularly spaced 'steps' or ***cascades*** with high velocity over the step and small plunge or ***scour pools*** below the steps (Whittaker and Jaeggi, 1982; Jarrett, 1984; Grant *et al.*, 1990). Some authors make a geomorphological distinction between step-pools and cascades (Malavoi, 1989; Grant *et al.*, 1990; Bayley and Li, 1992). The former refer to steps with boulders organised into 'ribs' across the channel (Malavoi, 1989; Grant *et al.*, 1990); the term cascade is restricted to units with non-organised boulders. In 'habitat hydraulics' terms, the step-pool represents the *morphological sequence* which is associated with ***cascade*** and ***pool / glide*** biotopes. ***Chute*** is recognised by two authors, both describing French rivers (Malavoi, 1989; Rabeni and Jacobson, 1993). In the UK, *chute* is regarded as a discrete biotope where large areas of exposed bedrock occur or where water flows over boulders in a the 'step' of a step-pool sequence or cascade. The terminology for *chutes* has been recently refined (see section 4.3). In hydraulic terms chute flow over a 'step' or within a *cascade* is likely to be similar, which explains why Wadeson (1994) groups *cascades* with 'step-pools'. In British rivers the majority of boulder bed channels do not have sufficient gradient to form 'step-pools', so *cascades* and step-pools are considered to be discrete; the former representing a biotope, the latter a morphological sequence. In the 23 papers reviewed, there are only 5 references to step-pools, compared with 12 cascade references.

Term adopted	*Other terms*	*Description*
Physical biotope (biotope)	Hydraulic unit Fluvial feature Morphodynamic unit Channel feature	Unit with a characteristic range of substrate sizes and hydraulic conditions. Discharge dependent: identified by surface flow types (see Table 2.7).
Channel feature	-	In-channel depositional features e.g. point bar, lateral (side) bar, mid-channel bar.
Morphological unit	Channel unit	Geomorphological channel feature with characteristic bedform or structure: identified in a sequence by substrate size (relative roughness), local gradient and hydraulics (sub- or supercritical flow).
Biotope sequence	"Pool/riffle" system	Sequence of biotopes i.e. units with characteristic bedforms and hydraulics. Biotope sequences repeat, at positions corresponding to breaks in bed profile slope.
Reach	-	Length of channel with similar channel slope, local side slope, planform morphology and floodplain features (bank material and riparian vegetation). Usually have a characteristic biotope sequence, or a "predictable spatial association of pool/riffles".
Segment	Sector	Length of channel between tributary junctions with similar long profile slope, lithology and valley side slopes.

Table 2.5: Glossary of terms adopted for the standard description of instream physical habitat, at increasing scales within a river basin (descriptions modified from Frissell *et al.*, 1986).

Rapids are referred to in 14 papers, which reflects their ease of identification, which has traditionally used flow types, albeit described as 'whitewater' or 'torrential' rather than 'broken standing waves' which is proposed as standard terminology (see Table 2.7, p44). The term is used by both geomorphologists and ecologists, although Wadeson (1994) uses the term to describe a morphological unit rather than a biotope. *Rapids* often become *cascades* at lower discharges, and most authors recognise either both features or neither. The dynamic nature of biotopes is suggested by several authors in the papers reviewed (Sullivan, 1986; Malavoi, 1989; Kershner and Snider, 1992; Wadeson, 1994).

Riffles and ***pools*** are identified in all papers reviewed, although pool is defined differently by various authors. Differences in local topography and position within the main channel define plunge, scour, trench and dammed pools (Bisson *et al.*, 1982; Sullivan, 1986; Malavoi, 1989). In this study, the term *pool* is restricted to biotopes which span the full width of the channel; deadwater is used to describe biotopes with scarcely perceptible flow which occur at a smaller scale e.g. at channel margins or bends (see Table 2.6, p43). The term pool is over-used and often mis-used to describe any biotope with relatively slower, less turbulent flow located between riffles, cascades or rapids (Malavoi, 1989; Grant *et al.*, 1990); in hydraulic terms such biotopes are ***runs***. 'Intermediate' reaches are characterised by riffles, rapids, cascades, glides and pools (Church, 1992); these have traditionally been classified by geomorphologists as either riffles and pools. The fact that less than half the papers recognise ***glides*** suggests that some glide biotopes have possibly been identified inaccurately as pools. In the context of lowland channels surveyed in RHS (1994), glides have been described inaccurately as pools or 'slacks' (National Rivers Authority, 1994). Failure to precisely define runs is likely to explain their appearance in only 12 of the 20 papers; preliminary RHS data indicate that runs are present in all channel types (Clark *et al.*, 1995).

There appears to be a subdivision of both pools and ***deadwaters*** in several papers. Whether these have any ecological justification is, as yet, untested generally; research has been carried out to compare marginal biotopes with 'main-channel' biotopes, but on an *ad hoc* site-specific basis e.g. Gibbins (1996), Using the definition of a pool to be a biotope which spans the full channel width (see Table 2.7, p44), several of the 'pools' in the reviewed papers would be classified as deadwaters e.g. eddy and dammed pools. It is likely that features such as macrophyte vegetation and degree of cover are likely to influence biotic communities in deadwaters in addition to their hydraulic conditions.

In view of the confusion and ambiguity of terms defined in the literature, it became clear that a standard procedure was necessary for identifying biotopes in the field, which could be adopted by geomorphologists, ecologists and non-specialist surveyors. The use of **flow types** have provided the key to successful recognition of biotopes; development of the flow type methodology is discussed in section 3.5.1.

2.4 CATCHMENT SCALE RIVER HABITAT SURVEYS

Whilst the identification and characterisation of instream features needs to be rooted in the scientific (particularly ecological) literature, there is a severely practical dimension too. An example of this is that the riffle definition employed by invertebrate biologists correlates well with safe access to the river and efficient kick sampling. In this section the practicalities of a hydraulic habitat classification system based on numerical data linked to a qualitative description of habitat units are reviewed. Only by providing tools for surveys does the broader future of instream habitat (see Frissell *et al.*, 1986) become a reality.

In the UK, national scale surveys developed originally with objectives for specific groups of organisms. RIVPACS uses actual invertebrate distributions compared to expected communities as an indicator of water quality (Wright *et al.*, 1994). HABSCORE is a similar model which predicts salmonid fish populations from assessments of physical habitat, but is restricted to streams sufficiently small to be electrofished i.e. 15m or less (Milner *et al.*, 1985; 1993). The first survey technique developed to incorporate physical habitat for a range of organisms was **River Corridor Survey (RCS)**. This technique was developed in the 1980s as a method for recording instream and riparian habitats and plant assemblages. Their principal aim was to identify sites with rare or degraded habitats, in order to target sites for conservation management, or habitat enhancement (National Rivers Authority, 1992). To allow prioritisation of sites on a national and regional level, comparison of habitats and plant assemblages with surveys at other sites is essential. However, the method used to record information (essentially sketches of a 500m reach) was very open-ended and subjective, with interpretation of 'important' features dependent upon the specialism of the surveyor. The standard of RCS varied considerably between individual surveyors, and the validity of comparing surveys was criticised (Raven *et al.*, 1996).

River Habitat Surveys (RHS) developed from RCS in response to the need for a national classification of rivers based on the *objective* assessment of river habitat quality (Raven *et al.*, 1996). Physical structure is regarded as a key component of habitat quality, and is believed to reflect intrinsic properties of the channel and associated processes which contribute to ecological functioning. Physical features are recorded at 10 spot checks spaced equally within a 500m sample reach; variables including in-channel features (e.g. gravel bar, vegetated island), substrate type (e.g. cobble, boulder), flow types (unbroken standing wave, scarcely perceptible flow) are recorded on a matrix to ensure objectivity in data collection. Channel features have been modified from the original (1994) RHS forms, which confused flow types and features in some cases, and included flows which rarely exist in natural channels e.g. 'approximately laminar' (see Table 2.6, p43). Existing definitions of flow types are provided in Chapter 3; flow types and associated channel features are listed in Table 2.7 (p 44). Involvement with RHS included the appointment of Professor Newson to the Technical Group, assisting the adoption of flow types developed during the course of this study in 1995/6 RHS data collection. This was supported by the attendance of Professor Newson and the author as Technical Assistants at a national field training course, where the standard identification of biotopes by flow type was taught in the field to RHS staff responsible for the regional training of surveyors.

The RHS network of sites was selected on a stratified random basis, selecting a 2 x 2 km site from each 10 x 10 km grid square (National Rivers Authority, 1996). Three sites from each 10 x 10 km grid square were sampled between 1994 and 1996, to provide a database of 4569 RHS sites (Raven *et al.*, 1996). Sites were surveyed at a standard flow level (i.e. summertime low flow) to allow comparisons between different channels in network, regional or national inventories. The RHS database has been analysed to determine 'characteristic' physical features of a particular (semi-natural) segment type. Preliminary results indicate that there is a good relationship between channel type and flow types.

PREDOMINANT FLOW TYPE	DESCRIPTION
Cascade/ waterfall	Distinct vertical drop in water
Torrential/ whitewater	Rapidly flowing water with severely broken surface in steep (5^0-89^0) channel sections (normally boulder/cobble substrate). Includes rapids.
Riffle	Fast flowing shallow water (normally over gravel) with distinctly broken or disturbed surface (max 5^0) slope).
Approximately laminar	Water with largely undisturbed surface other than occasional swirls or eddies. Includes glides and runs. NB Weeds can cause considerable flow variation in glides.
Static	Area of water with no perceptible flow due to natural or artificial ponding (slack). No eddies or swirls. A stick placed in a slack will not create turbulence.
Pool	A distinct feature of deeper water with either no perceptible surface flow or slight eddying/reverse flow. Never longer than three times channel width.
Ponded	Water ponded by natural or artificial obstruction downstream.

Table 2.6: Original descriptions of physical attributes in the 1994 version of RHS (National Rivers Authority, 1994).

One objective of RHS is to compare any watercourse with the reference sites, and predict the distribution of physical features for a given segment type. This is being developed as a scheme for assessing habitat quality, based on the degree of channel 'naturalness' or modification, compared to the distribution of physical features or characteristic "fingerprint" for unmodified channels. Flow type/biotopes and in-channel features (substrate and depositional bars) are two of the seven criteria which will contribute to the Habitat Quality Index (Raven *et al.*, 1996). The role of biotope 'patchiness' and 'diversity' are central to the development of a Habitat Quality Index; these are developed in section 5.3. The ecological relevance of these features is discussed in section 7.1.

PHYSICAL BIOTOPE	ASSOCIATED FLOW TYPE (S)	DESCRIPTION
Waterfall	Free fall	Clearly separates from the backwall of vertical features.
Spill	Chute	Low curving fall in contact with substrate ~ sometimes associated with step-pool sequences.
Rapid	Broken standing waves	White-water tumbling must be present.
Cascade	Chute Broken standing waves	A mixture of the two above flow types ~ chute flow as water passes over the upstream side of the substrate; broken standing waves on the downstream side of the substrate.
Riffle	Unbroken standing waves	Upstream facing wavelets which are not broken.
Run	Rippled	No waves, but general flow direction is downstream with a disturbed rippled surface.
Boil	Upwelling	Heaving water as upwellings break the surface ~ secondary flow evident as vertical and horizontal eddies.
Glide	Smooth boundary turbulent	Perceptible downstream movement is smooth (no eddies) ~ a floating object placed in the water will move downstream. Reflections are distorted by turbulent eddies.
Pool	Scarcely perceptible flow	No net downstream flow ~ a floating object placed in the water remains stationary. Reflections are not distorted. Pools occupy the full channel width.
Deadwater	Scarcely perceptible flow	Flow type as for pool, but do not occupy the full channel width. Associated with channel margins, bends or immediately downstream of point bars and other channel obstructions (e.g. overhanging trees).

Table 2.7: Field based identification of biotopes (modified from 1996 RHS; based on Newson and Padmore, unpublished).

SERCON (System for Evaluating Rivers for Conservation) developed in response to a lack of standard terminology and opinion regarding the criteria which contribute towards quality. It uses both physical channel and species data to determine a score for the overall conservation value of a particular river, based on classical conservation criteria (Ratcliffe, 1977). Six criteria are used in the scoring system, namely: physical diversity; naturalness; representativeness; rarity; species richness. It operates on a larger scale to that adopted by RHS, focusing on Evaluated Catchment Sections

(ECS), which are equivalent to channel segments (Frissell *et al.*, 1986). Unlike RHS, rivers are not classified into segment types, but 'conservation scores' are compared between all sites. SERCON's broad aims for river conservation have been described in a demonstration seminar (Boon, personal communication):

- Fully functional ecological systems
- Natural features retained
- Maximise biodiversity in a geographical context
- Ensure typical and rare types are protected

Links between RHS and SERCON and the validity of extrapolating RHS data to the segment scale is being explored in a joint research initiative by the National Rivers Authority and SERCON.

2.5 TRANSLATION OF BIOTOPE IDENTIFICATION TO FIELD SITES FOR CHARACTERISATION

The literature reviewed in Sections 2.1 - 2.4 allowed a preliminary selection of ecologically important channel features together with descriptions for their field identification. In order to test the validity of these descriptions for channel features in rivers of north-east England, the practical components of the PhD study therefore became:

- identification and definition of channel features in the field
- choice of a network of field research sites to demonstrate the full range of, variability within, and sequences of these features
- field measurements of the key variables
- liaison with those conducting national surveys as a means of establishing a potential to extrapolate the results

The literature reviewed substantially aided the first of these components; these are summarised in Table 2.6. The choice of field sites raised considerable problems. The need for *a priori* judgement on the replicability of site-specific measurements in the absence of a channel typology proved especially problematic. In the context of the developing ‘river segment typology’ it was decided to select a range of natural, characteristic features from a range of geomorphologically discrete channels. In the north east of England there exists a concentration of different lithologies, gradients and altitudes which are the main controls on reach scale channel morphology (Church, 1992; Rowntree, 1994). Geomorphological guidance regarding ‘potential’ sites was tested by reconnaissance fieldwork which involved ‘walking the channel’ for a standard length (corresponding to roughly 50 times the channel width). Biotopes (then considered as ‘channel features’ or ‘hydraulic units’) e.g. riffle, glide etc. and morphological features e.g. point bar, vegetated island were recorded on 1:10 000 scale OS maps, to establish common or typical features for a channel in a particular location within a drainage basin. Sites selected without the benefit of the RHS typology have been compared with the preliminary typology for England and Wales (Raven *et al.*, 1996); this indicates that the study sites are representative of nine of the eleven channel types (exceptions, being true mountain streams and large lowland rivers). The following chapter goes on to discuss precise site location, layout and the fieldwork methodology.

A major consideration before progressing with the field data collection for this project was the eventual use of the calibrated habitat characteristics. Of particular importance was the ability to initiate a new approach to habitat evaluation based on fluvial geomorphology, which could be linked to existing IFIM techniques. Links between this study and IFIM are illustrated in Figure 2.1. Habitat hydraulics provides a simple assessment by field observations based on standard definitions (Table 2.7 and 3.6). Physical biotopes describe the various flow types created by channel geomorphology and operate at a scale which is likely to be relevant to ecology via application of the patch dynamics concept. Scale considerations have been given special weight (e.g. extrapolation to other reaches via catchment scale hierarchical models) because even the most successful empirical investigations often become “rooted” to their hillside, forest or river of origin. Thus, the first important decision was that taken on the network of field sites (see Figure 3.1). Subsequent relationships with the NRA’s River Habitat Survey (RHS) initiative have been mutually beneficial; RHS incorporating standard, objective recognition of biotopes and returning a channel typology which has vindicated the choice of sites and operating scales in this research. One aim of the RHS segment typology is to predict the characteristic channel features and hydraulics of any particular stretch of river from its type. This would provide a means to cross scale boundaries from local-scale hydraulics to the habitat properties of river networks.

The approach laid out as "the biotope approach" in Figure 2.1 is at present largely strategic but its potential guided the programme described in this thesis. It has also provided a potential for further research in extending the approach to whole channel networks (section 7.4). Such an approach would be compatible with, or provide an alternative to existing methods of determining the ecological effects of varying river flows and channel sections, notably IFIM techniques, using PHABSIM. This model has been accepted as an international standard by water managers, if not so by ecologists. It successfully integrates ecology and hydraulics, but at present, does not incorporate geomorphological trends which operate over different spatial scales in channel networks. There are other inherent problems with PHABSIM as described on section 1.3 (p12, 13). The mesoscale survey and typology approach described in Figure 2.1 provides a modular approach to river habitat via a hierarchical river basin model; the approach could be used in the assessment of critical biotopes or reaches, or to improve PHABSIM's selection of representative reaches used in the extrapolation of its outputs.

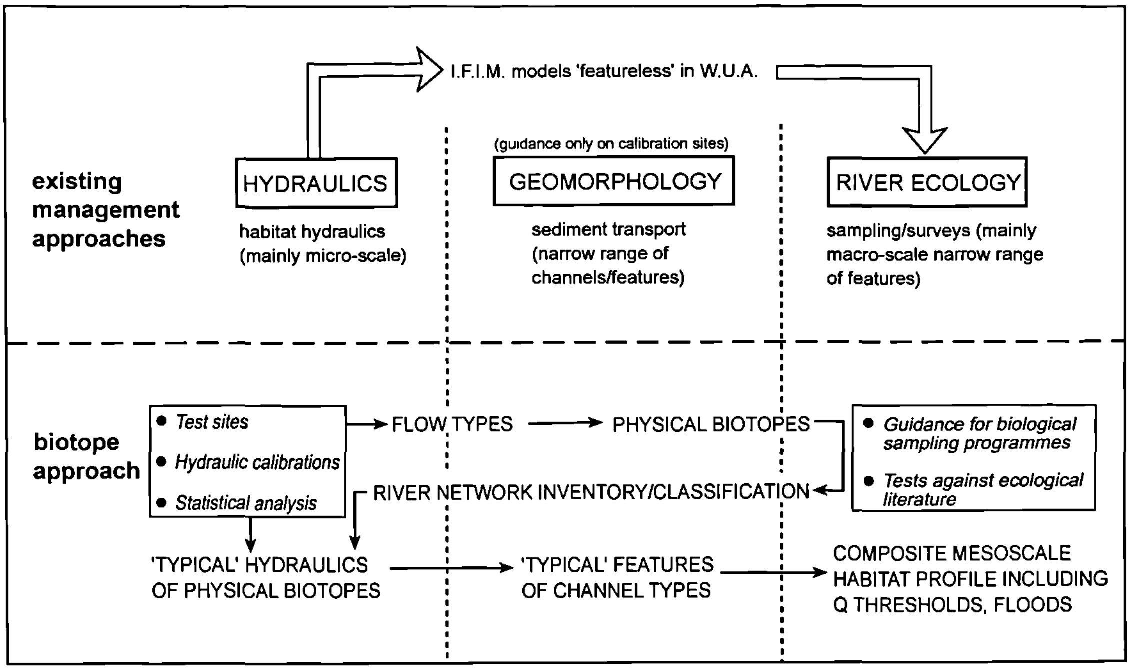


Figure 2.1: Hydraulics, geomorphology and river ecology, linked at present (upper level) by IFIM but conducting separate research agendas. The lower level indicates the biotope approach advocated in this paper.

Author (Year)	**Habitat term**	**Water-fall**	**Step-pool**	**Chute**	**Rapid**	**Cascade**	**Riffle**	**Run**	**Glide**	**Pool**	**Dead-water**
Allen (1951)	Habitat				✓	✓	✓ Stickle	✓	✓ Flat	✓	
de Leeuw (1981)	Hydraulic unit	✓					✓		✓	✓	✓Slough/ditch
Mosley (1982a)	Habitat				✓ Torrent		✓	✓		✓	✓Side channel; backwater
Bisson *et al.* (1982)	Habitat	✓			✓	✓	✓		✓	✓(Scour; plunge; trench dammed)	✓ Backwater
Blyth (1983)	Habitat	✓			✓	✓	✓	✓		✓	✓ Pocket-water
Platts *et al.* (1983)	Habitat						✓	✓	✓	✓	✓Margin
Cummins *et al.* (1984)	Habitat				✓	✓	✓	✓	✓	✓	
Beschta and Platts (1986)	Habitat						✓			✓	
Frissell *et al.* (1986)	Habitat	✓			✓	✓	✓	✓	✓	✓(Scour; plunge)	✓Side-channel ; backwater
Sullivan (1986)	Hydraulic habitat unit				✓	✓	✓		✓	✓(Scour; plunge; eddy; dammed)	✓Alcove; lotic channel
Malavoi (1989)	Morphodynamic unit		✓	✓	✓	✓	✓	✓		✓(Scour, plunge; dammed; alcove)	

Table 2.3: Terminology used to describe instream habitat for fisheries.

Author(Year)	Habitat term	**Water-fall**	**Step-pool**	**Chute**	**Rapid**	**Cascade**	**Riffle**	**Run**	**Glide**	**Pool**	**Dead-water**
Grant *et al* (1990)	Channel unit		✓		✓	✓	✓			✓ Scour; plunge; trench; dammed	
Church (1992)	Morphological unit		✓		✓	✓	✓			✓ As above	✓Back-water; side-channel
Cummins (1992)	Habitat				✓		✓	✓	✓	✓	✓Edgewater; fore-water; backwater
Jowett (1992)	Habitat						✓	✓		✓	
Kershner and Snider (1992)	Fluvial features		✓				✓	✓	✓	✓	
Bayley and Li (1992)	Habitat		✓		✓	✓	✓			✓	✓Slack/ back eddy
Heggenes*et al.* (1993)	Habitat						✓	✓		✓ Pocket water; stillwater ; backwater	
Rabeni and Jacobson (1993)	Hydraulic habitat unit			✓			✓	✓		✓	✓Backwater
Newbury and Gaboury (1993)	Hydraulic habitat				✓	✓Boulder riffle	✓			✓	
Dallas *et al.* (1994)	Biotope				✓	✓	✓	✓		✓	
TOTAL		4	3	2	14	12	20	12	9	20	11

Table 2.3 continued.

3. METHODOLOGY

Chapter Overview

This chapter describes the criteria used to select study sites (3.1) and the procedures by which sites were laid out for data collection and future relocation (3.2). Data collection procedures are described at two scales: transects and individual cells. The advantages and limitations of electromagnetic current meters are discussed, and field safety procedures outlined (3.3). Procedures for the hydraulic characterisation of sites are described, methods for calculating discharge are discussed and hydrological conditions during the study period are summarised (3.4).

Development of biotope identification by flow types is discussed in detail as one of the major contributions of this thesis to the integration of ecology and fluvial geomorphology in instream habitat research (3.5). The chapter ends with a description of the data storage and manipulation procedures, together with methods for preliminary data exploration and representation. Formulae for the calculation of combined hydraulic indices are listed; these provide the basis of the data analysis (3.6).

3.1 SITE SELECTION

One of the aims of this study was to provide hydraulic characterisations of a range of geomorphological channel types; a range which is not available within one basin in the North-east of England. To set up the sample of repeatable, characteristic combinations and sequences of biotopes, sites were selected to span the geomorphological variability of channel morphological units. This proceeded with a desk-top study of geology and slope maps together with NRA Catchment Management Plans and summary flow statistics, to identify rivers with different combinations of rock type, slope and flow regime. Site selection involved preliminary field checks of typical rivers in

North-east England within these subjectively defined groups. This reconnaissance study revealed the following points which are consistent with hierarchical, catchment based classifications (Frissell *et al.* 1986, Wadeson and Rowntree, 1994):

- Morphological units which repeat in sequence form the basic hydraulic controls on (low) flow
- The hydraulic characteristics of these units are (visually at least) dependent on substrate size, gradient and discharge

At higher levels of such classifications, namely the relationship between reaches, segments and catchments the following observations which are not in agreement with Frissell's classification became apparent:

- Reach breaks are not easily identified by breaks in slope or geology
- A repeating sequence of morphological units is not always present
- The spacing of channel features and the associated biotopes is related more to local geomorphology than to position within a reach or sector, or to text-book predictions (e.g. riffle-pool spacing)

It was seen to be important to this study to forge early links with developing national schemes of survey for channel features; a standard terminology and national inventory was anticipated as being essential to wider use of the North-east England data. River Habitat Survey (RHS) forms (National Rivers Authority, 1994) were completed during this pilot tour, to test whether the features observed on-site corresponded with the categories of Instream Channel Features proposed by RHS. It was clear that these features (see Table 2.6) did not adequately describe the range present in the rivers of North-east England. A dialogue was established between this study and RHS, with Professor Newson joining the RHS Technical Group of NRA. Until RHS, no nationally agreed standard procedures existed for the accurate identification of, and discrimination between channel features.

Sites were selected to include a range of channel features which were clearly acting as general or local hydraulic controls. The precise location of study sites was decided using the following additional criteria:

- Proximity to a gauging station
- Straight, natural reach
- Reach with a history of research yielding supplementary information

Proximity to a gauging station was considered to be important to relate biotopes to discharge exceedence percentiles, and so field calculations of discharge may be compared with gauged discharges. Straight reaches were selected as it would be inappropriate to extrapolate data from the complex hydraulics associated with bends (Bathurst *et al.*, 1979). PHABSIM guidelines for site delimitation were also followed (as defined by Johnson *et al.*, 1994). At this stage it was anticipated that the study would bring more geomorphological credibility to the PHABSIM model, which had hitherto selected sites on an *ad hoc*, problem-oriented basis. The location of study sites is illustrated in Figure 3.1. Summary data for the sites are listed in Table 3.1.

The site name refers to the river and, (in brackets) either the nearest landmark on O.S. maps (where the site is not directly adjacent to a gauging station) or the station name in registers of hydrological data. Future reference to sites will use only the river name, with the exception of those rivers with more than one study site; i.e. the Wear (Stanhope) and the Wear (Wolsingham). Sites are listed in order of decreasing substrate size; D_{84} was used as D_{50} was found to be a misleading representation of substrate size distributions, particularly at those sites with a bimodal substrate distribution of boulders with gravel and small cobble in clustered bedforms (Brayshaw, 1984) e.g. Harwood Beck and the Wear (Stanhope).

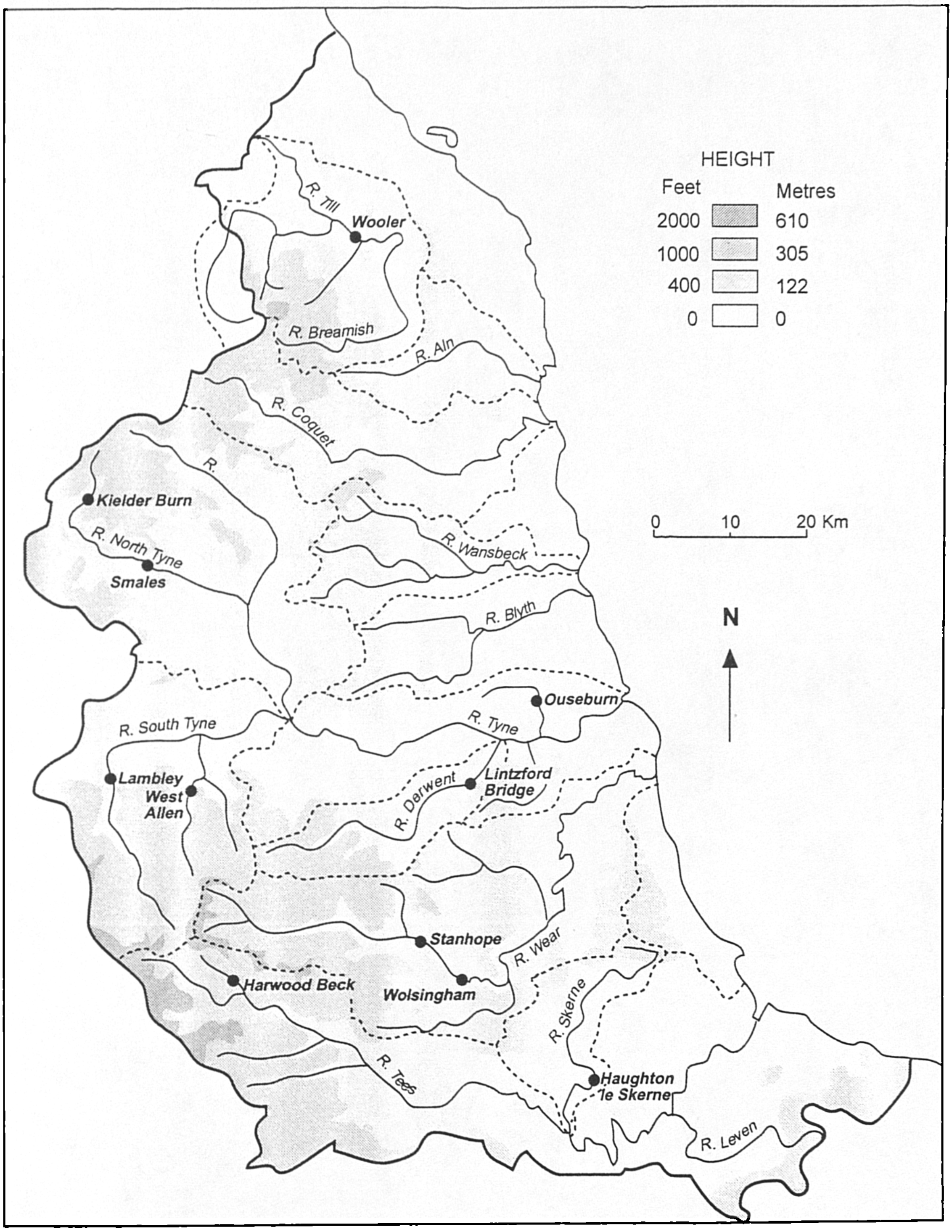


Figure 3.1: Location of study sites within North East England.

Site name	**Grid reference**	**Lithology**	**Catchment area (km^2)**	**Slope**	**D_{50} (mm)**	**D_{84} (mm)**	**D_{90} (mm)**	**Width depth ratio**
West Allen (Hindley Wrae)	NY 781568	Millstone grit and limestone	75.1	0.0108	123^B	422*	500*	45.6
Wear (Stanhope)	NY 984391	Lower limestone	171.9	0.0152	178^B	389*	500*	22.4
Kielder Burn (Kielder)	NY 643946	Cementsone	58.8	0.0113	138	270	350	26.5
South Tyne (Lambley)	NY 672605	Millstone grit and limestone	286.0	0.0019	123	215	256	62.2
Harwood Beck (Harwood)	NY 848310	Middle limestone	25.1	0.0098	85	205	320	42.5
North Tyne (Smales)	NY 737857	Scremerston coal group	267.4	0.0011	106	185	216	65.8
Wear (Wolsingham)	NZ 064372	Millstone grit and limestone	265.8	0.0031	90	153	187	69.7
Derwent (Lintzford Bridge)	NZ 146571	Lower carboniferous	203.5	0.0052	62	125	150	47.0
Ouseburn (Salter's Bridge)	NZ 255686	Middle carboniferous	36.0	0.0015	4	100	130	23.0
Till (Doddington Bridge)	NU 001307	Fell sandstone	315.0	0.0026	24	40	44	28.9
Skerne (Haughton Bridge)	NZ 304158	Magnesian limestone	177.4	0.0023	.43	2	3.5	17.5

Table 3.1: Summary catchment and morphological data for study sites (^{B} indicates a site with bedrock; * indicates that substrate statistics are calculated for the alluvial material present. Width-depth ratios are calculated from high flow *wetted* widths and depths).

3.2 SITE CHARACTERISATION

In order to characterise the in-channel features at each site, a study section was defined as the length of channel which included one complete sequence of characteristic 'morphological units' (Church, 1992) and their associated flow dependent (Wadeson, 1994; 1995b) biotopes. The range of biotopes identified are those listed in Table 2.7. In small streams, namely Kielder Burn and Harwood Beck hydraulic calibration of just one sequence of morphological units would have produced a dataset with fewer than 100 points; suggested as the minimum sample size for PHABSIM calibrations (Johnson *et* *al.*, 1994). Two *cascades* were therefore sampled at these sites to increase the number of data points measured in units with high variation in hydraulics (evident from the diversity of surface flow types). Site boundaries were located at the transition between biotope sequences, with the exception of the heavily engineered Skerne and Ouseburn. At both these sites the *glide* extended several hundred metres beyond the study section; the section was ended when it was considered sufficient data points had been measured to describe the hydraulics of the site. Study sections were divided by transects placed every 5, 10 or 20 m upstream, depending on the hydraulic variability within biotopes. This is a modification of PHABSIM standard data collection procedures (Johnson *et al.*, 1994), in which transects are placed at equal intervals upstream. To ensure sufficient data points are collected to record the range of hydraulics, transects were placed every 5m in biotopes with a high visual degree of variation in velocity, and at 10 or 20m in biotopes with less variation. Table 3.2 summarises the transect spacing in different biotopes.

Transect Spacing (m)	**Physical biotopes at low flow**
5	Cascade, boil, deadwater
10	Riffle, run, glide (channel width<30m)
20	Glide (channel width>30m), pool

Table 3.2: Transect spacing in low flow physical biotopes.

Transects were placed perpendicular to the direction of primary flow and their location marked with painted wooden pegs secured in the bank above bankfull height, or by painted markings on adjacent trees, walls or rock faces where vegetation growth would have made relocation of wooden pegs difficult. Each transect was photographed along its length and bank features including trees and fences were noted to ensure that the exact transect location could be determined at subsequent visits. All information was recorded on a 'Site characterisation' form (Appendix A2).

Biotopes were initially identified at the transect level, as geomorphologists tend to assume channel morphology and flow to be relatively uniform across a cross-section, except at meander bends. In reality the transition between biotopes may occur laterally across the channel, or diagonally across two transects especially downstream of lateral and point bars. Initially transects were classified on the basis of the dominant biotope (which is consistent with RHS data recording procedures). This procedure was followed during stream surveys in the United States (Platts *et al.*, 1983); transects were simply assigned to either 'riffle' or 'pool' class, depending on mean velocity. However, it became clear in this study that allocation to only two types of morphological unit was inadequate to describe fully the range and distribution of biotopes. Furthermore, classification at the transect level overlooked important details of the flow pattern. This led to the development of cell level classification by flow types (see 3.5.1).

3.3 TRANSECT AND CELL LEVEL DATA COLLECTION

Variables recorded at each transect include wetted width during flow calibration and bankfull width, surveyed using a total station Electronic Distance Meter (Leica TC 400). Substrate size and the hydraulic variables depth and velocity were recorded, as these are considered to be the main physical habitat variables (Johnson *et al.*, 1994). The PHABSIM variable 'cover' was not formally recorded; instead macrophyte vegetation was given special measurement consideration (see section 6.2.2). Data collection involved the recording of variables at one metre intervals across each transect, to ensure standard sampling within subjectively selected reaches. Metre intervals were selected to be consistent with PHABSIM procedures (Johnson *et al.*, 1994). Sampling a standard number of equally spaced points at each transect was considered, but was rejected for the following reasons:

1. Cross-channel hydraulic variation tends to be greater in wider rivers
2. Sampling an equal number of points for each site requires assumptions to be made regarding the minimum number of data points necessary
3. Regular interval sampling is compatible with computerised visual data representation e.g. the Microsoft 'Excel' spreadsheet package.

The variables recorded are depth (cm), velocity at 0.6 depth ($m.s^{-1}$) and intermediate substrate diameter (b axis, mm). Depths and velocities were recorded with an electromagnetic current meter (EMCM) (Marsh McBirney, Flo-Mate 2000) attached to a graded gauging rod. EMCMs work by movement of an electrically conductive fluid through a magnetic field which generates a force normal to the magnetic field and fluid motion. The advantages of EMCMs compared to propellor meters are summarised by Lane *et al.*, (1993). The main benefits are the absence of moving parts which lead to inaccurate readings in rapidly fluctuating velocities, and prevent readings to be taken directly next to substrate particles; and their robustness in high/flood flows. Additionally the Flo-Mate 2000 is capable of recording flow in the opposite direction to the main current as a negative velocity. It is not capable of measuring more than one orthogonal direction, but this study is concerned with mean velocity comparisons rather than turbulence and pulsating flows. Limitations of the model are that it can only measure to within approximately 5mm of the substrate, therefore is not capable of recording near-bed turbulence. Only the dominant component of secondary flows could be measured which led to non-representative velocities in upwelling flow cells (see section 4.2.1). Readings were taken facing the direction of maximum flow and averaged over 30 seconds. In steady flow (Smith, 1975) units, namely *glides*, *pools* and *deadwaters* this was reduced to 15 seconds after pilot studies showed these units to show little fluctuation in velocity over this period. Rapidly fluctuating velocities in fully turbulent flow were averaged over 60 seconds but may still produce inaccurate readings due to the magnitude of fluctuations. The accuracy of measurements in highly turbulent flow would benefit from the use of a EMCM with a spherical sensor head, which considers both the mean and higher order moments of flow components (Aubrey and Towbridge, 1985).

The intermediate diameter of the particle on which the base of the current meter rested was measured to the nearest millimetre (Wolman, 1954). Substrate measurements were taken after depth and velocity readings had been taken, to ensure that these readings were not affected by the removal of the substrate. In channels which were too deep to reach the substrate, the size was estimated where the channel bed was visible. Where this was not possible a missing value was recorded for substrate. Substrate data were recorded at only the first calibration at each site, assuming the overall substrate distribution of the reach would not change under 'typical' annual discharges. Calibrations following high magnitude flood events recorded substrate at each cell, to determine the impact of floods on both substrate distributions and morphological units, as summarised in Table 3.3.

All data were recorded on a 'Hydraulic characterisation' form (Appendix A3) working from the downstream transect. Measurements were taken from the true left bank to ensure consistent data input. Data collection began during the Spring and Summer 1994, which allowed 'mid flow' hydraulic calibrations to be performed at some sites before the low summer flows of 1994. 'Low flow' calibrations were made during the months of July and August when the hydrological conditions produced unusually low flows (see section 3.4.1).

'High flow' hydraulic calibrations were more difficult to achieve. In practice, the highest flow in which safe working conditions could be assured was taken as the 'high flow'. In channels where the product of depth and velocity exceeded 1 $m.s^{-1}$, safety procedures involved the field worker wearing a dry suit and harness. The transect was positioned by a rope marked at metre intervals and the worker 'towed' to each sample point by another team member stationed on the bank upstream of the sample transect. Readings in very deep channels e.g. the South Tyne were taken from an inflatable boat, stabilised at each sample point by a rope held by members of the field team working from the banks.

Fieldwork planning and efficient use of labour was facilitated by calling the NRA's regional 'River Line' (telephone data service). This automated message produced by the NRA gives daily stage levels at all the main gauging stations, relative to a summer 'base level'. Many of the 'high flow' calibrations were achieved by a combination of careful judgement and persistence; sites were commonly visited and found to be either too high for fieldwork to be carried out safely, or just below the stage (and hydraulic conditions required). The technique of biotope mapping (section 5.1) developed partly to make good use of these site visits, but more importantly to fill in the 'snapshot' hydraulic pictures produced by the full calibrations.

3.4 HYDRAULIC CHARACTERISATION

Three calibration discharges were selected to hydraulically describe the sequences of units which exist at different flows. As discharge increases morphological controls on flow alter, resulting in different **biotopes** and sequences. Sullivan (1986) concluded that three discrete sequences may be observed within a typical annual flow; these equate to seasonal flows, 'baseflow', 'summer low flow' and 'winter high flow'. Three calibration discharges are also the minimum requirement in PHABSIM for constructing stage-discharge and hydraulic geometry relationships.

Discharge data were obtained from NRA flow records for the nearest gauge (Merrix, personal communication). Flow duration tables were referred to for flow exceedence percentiles of discharges present during hydraulic calibrations and biotope mapping (see section 3.5.2 for an explanation of biotope mapping). Two of the sites chosen, namely the Till and the West Allen were not adjacent to a gauging station, but were included for very different nature of their channel units. In addition to NRA discharge records calculations were made from field measurements. Each calibrated transect yielded cross-section and velocity data appropriate for instantaneous discharge measurements. The method for calculating discharge is that proposed by Gordon *et al.*, (1992), p163. With between 13 and 20 discharges calculated for each site calibration the following calculations were made: transect averages, biotope averages (classified at transect level) and site discharge. The latter was calculated for those transects with a relatively uniform cross section and substrate size distribution (i.e. those most suited to the conventional choice of cross-sections for discharge measurements). This includes those transects within depositional morphological units (Moon, 1939), and includes pools, glides and runs. For a further discussion of the morphological unit see section 3.5.

At low, moderate and high discharges data were collected according to the specifications of PHABSIM, which was the main model being used to establish changes in *"available habitat"* with discharge when this study commenced (Johnson *et al.*, 1994). It works on the assumption that the range of depths, velocities and substrate sizes are the main determinants of physical habitat, and that depth and velocity may be predicted for other discharges on the basis of data recorded at one calibration discharge to specify the overall hydraulic conditions at all flows. At the start of the study it was anticipated that the data would be entered into the PHABSIM programmes. The role of macrophyte vegetation both as habitat *per se* (Wright *et al.*, 1994; Harper *et al.*, 1992; 1995) and in its influence on hydraulics has been cited by several authors (Hearne and Armitage, 1993; Newall, 1995). For this reason vegetated cells were noted and velocity profiles taken in vegetated reaches. Velocity profiles were taken at those sites and cells where it was unlikely that the velocity at 0.6 depth would be equivalent to mean velocity. These include the following reaches:

- Channels with seasonal growth of macrophyte vegetation
- Steep gradient, boulder bed channels
- Units dominated by secondary flows (upwelling)

In order to more accurately characterise the hydraulics of these units, velocity readings at five depths were taken. Standard velocity profiles include readings at the surface, 0.2 depth, 0.6 depth, 0.8 depth and bed velocity (Gordon, *et al.*, 1992). Detailed hydraulics and the management implications for such reaches are presented in section 6.2.

3.4.1 Hydrological conditions and flows

The data were collected between March 1994 and May 1996. During this period the North-east of England experienced flood events of high return periods (Table 3.3) and prolonged drought. The impact of high magnitude floods on channel morphology is summarised in Table 3.4. At the other extreme, the period from April to August 1995 was associated with the lowest total rainfall in England and Wales for over 200 years; intensification of the 1995 drought from mid July to late August produced mean flows well below the historical monthly average. In the South Tyne summer (June to August) flows were 33% of the monthly average; in August the absolute monthly minimum flow occurred i.e. 14% of the monthly average (Institute of Hydrology, 1995). Detailed biotope distributions after flood and during drought flows are illustrated in Figure 5.1, and their implications for 'patch dynamics' are discussed in section 7.1.3.

Station	**Nearest study site**	**Peak flow (cumecs)**	**Estimated return period**
Featherstone	South Tyne	348.3	200^+
Haydon Bridge	West Allen	760.9	150
Stanhope	Wear (Stanhope)	287	500^+
Witton Park	Wear (Wolsingham)	286.3	35
Harwood	Harwood Beck	56.1	25

Table 3.3: Flood flow statistics for significant (> 10years) events of 31.1.95 and 1.2.95 in rivers of North-east England (source; National Rivers Authority, Newcastle).

Scale affected	Reach scale change	Biotope change	Site affected
Substrate	1. Change of substrate volume by deposition	Infilling of pools, runs and glides	Harwood Beck Wear (Stanhope)
	2. Change of substrate size by sediment transport	Altered relative roughness and local bed slope	West Allen Harwood Beck Wear (Stanhope) Wear (Wolsingham)
Morphological unit	1. Movement in position	Reach flow control e.g. migration of riffles	South Tyne Wear (Wolsingham)
	2. Change in extent	Relative extent of riffle pool, run	South Tyne West Allen
Planform	1. Change in width/depth	Restriction of pools, glides and runs by lateral/point bar deposition	Wear (Stanhope)
	2. Change in planform	Meander migration and bank erosion	South Tyne

Table 3.4: Flood geomorphology and biotope change: evidence from Northern England, 1995.

3.5 CHANNEL UNITS, FLOW TYPES AND BIOTOPES

As discussed in section 2.3, there is, in the UK, a need for common terminology and identification of units based on rapid field survey. The lack standard identification procedures was evident in preliminary (1994) River Habitat Survey forms which included only a limited range of 'features' and flow types, and confused these with sequences e.g. 'riffle-pool' and 'step-pool' were listed as individual features. These descriptions apply to the scale of morphological units and biotope sequences, within which physical biotopes exist at a smaller scale (see Table 2.5). A literature review of channel 'units' and 'features' carried out during site selection revealed that a more consistent and rigorous terminology was required in the field of habitat hydraulics. Early site calibrations used descriptions and terminology from Table 2.3 to identify channel 'features' at the transect scale. The lack of standard identification procedures and terminology became evident through dialogue with NRA staff involved with RHS and overseas researchers at the First International Symposium on Habitat Hydraulics (Trondheim, Norway, August 1994). Out of 47 papers discussing the broad theme of habitat hydraulics, only 5 used habitat terminology other than riffles and pools. The continued adoption of old terminology which originates from geomorphological process studies (section 2.2) indicated the need for a new, standard international definition of 'hydraulic habitat units' or biotopes.

Identification of biotopes by the **dominant flow type** developed as a means of distinguishing between *different biotopes which occur within the same morphological unit at different discharges*. Grant *et al.* (1990) first used flow types to identify riffles, rapids, cascades and pools in step-pool sequences in Oregon. The proportion of supercritical flow or standing waves was used to distinguish between riffles and rapids; however no distinction is made between broken and unbroken standing waves. Only a small proportion of the authors reviewed by Wadeson (1994) referred to flow type (see Table 3.5). **Flow types are a surface manifestation of flow energies within the channel; this method of identification therefore developed as a standard, visual classification of biotopes.** In ecological terms flow type is important to the condition or quality of fisheries, as it affects stream-bed oxygen concentrations and biological processes related to feeding (Heede and Rinne, 1990). However very few ecological studies have explored the link between surface flow types and biota; this is examined in section 6.3.

"Instream flow environment"	a) Formally defined references	b) References to flow type
Riffle	8	4
Run	7	4
Cascade	6	4
Glide	4	1
Waterfall	3	0
Backwater	3	0
Rapid	2	1
Chute	1	1

Table 3.5: Instream flow environments with a) a formal definition, and b) those which refer to flow type (23 papers, after Wadeson, 1994).

3.5.1 Biotopes: identification by dominant flow type

The detailed use of flow types to distinguish between flow dependent biotopes was developed when it became evident that certain biotopes would be overlooked with transect-level classifications. In the early calibrations each transect was classified and recorded (see Appendix A2). Other 'features' were simply noted as present and their location recorded. Those biotopes which were consistently overlooked with transect level classification include:

- Marginal *deadwaters*
- Shallow *runs* and *glides* adjacent to *riffles* and *cascades*
- *Glides* and *deadwaters* downstream of point bars
- High flow 'refugia' i.e. relatively slow flow patches which occur downstream of obstacles, point bars and overhanging trees and between mid-channel islands

Identification of flow types at the cell level as opposed to classification of transects by the dominant biotope is illustrated in Figure 3.2. The two modifications to identification are:

1. Individual cells are classified rather than transects.
2. Cells are classified primarily according to flow type before being assigned to the (literature-defined), biotope.

Cell level classification also developed as a result of preliminary discriminant analyses (see section 4.2). Discriminant analysis is a statistical technique which objectively tests the visual classification of 'biotopes'. The analysis operates on *individual data readings,* which resulted in a large proportion of 'misclassifications' in the early analyses which were based on transect level classification.

Cell level classification according to flow type began in November 1994 at the Harwood Beck site under high flow conditions. The 'biotope sequence' (see section 5.2 for the definition) was rapid-run, but small scale 'patches' of slower, rippled flow were present in the wake of large boulders. The flow type at each sample point or 'cell' was thus classified, in addition to transect level classification. **This allowed biotopes to be identified by flow type at the ecological 'patch' scale, and also at the mesoscale according to the dominant flow type across a transect.** As a general rule for classifying transects by the dominant biotope, the flow type which occurs in more than 50% of cells was suggested; this is less open to subjectivity and allows a gross comparison of biotope sequences, so may be more readily applied to national surveys such as RHS.

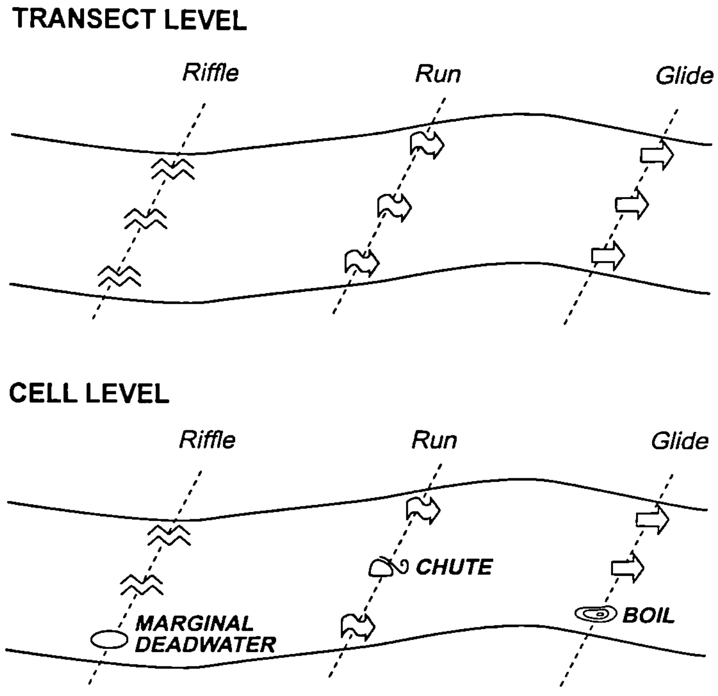


Figure 3.2: Cell and transect level identification of flow types.

The use of flow types to identify biotopes was formalised by British and South African researchers involved in water resources management at the Citrusdal Workshop, South Africa, January 1995 (Rowntree, 1996a). Identification of biotopes by flow type has since been adopted by both the National Rivers Authority for RHS (Raven *et al.*, 1996) and South African water resource managers (Rowntree and Wadeson, in press). Wadeson (1995b) has since adopted the term ***hydraulic* biotope** to indicate that temperature and water quality are excluded from the definition of flow dependent channel units (Rowntree, personal communication).

The physical biotope referred to in this study is equivalent to Wadeson's (1995b) hydraulic biotope. Formal definitions of flow type are based on the existing flow hydraulics literature, which distinguishes between e.g. sub- and super-critical (Davis and Barmuta, 1989). A review of hydraulics literature is provided by Wadeson (1995b), so will not be repeated. The use of flow types to identify biotopes assumes the hypothesis that different flow types are hydraulically discrete (see sections 4.1- 4.3). It may be thus inferred that visually identified flow types are characteristic of a particular combination of substrate and hydraulic conditions. Table 3.6

summarises the flow types identified in this study, providing a description for their field identification. These are a modification of the biotope descriptions agreed at the Citrusdal Workshop, with more emphasis being placed on flow types than substrate, as rivers in England and Wales have less varied substrate types than those in South Africa. These descriptions of flow types replaced those used in 1994 RHS, which included types which rarely exist in natural channels e.g. "approximately laminar". The revised flow type classification has been adopted by the NRA for RHS and was implemented for the first time in the 1995 data collection programme. A modified version of the flow types in Table 3.6 has been included in the RHS Field Methodology Guidance Manual (National Rivers Authority, 1995 - see Table 2.7, p44). The flow types and physical biotopes described in this thesis are illustrated in Appendix A1. More recently, the identification of flow types has been assisted by the production of a training video by Newcastle University Geography and Audio Visual Departments. This is being used primarily by NRA surveyors involved in RHS, to ensure standard identification of flow types. The video has been used as part on an accreditation scheme for field surveyors (Raven, personal communication).

Attendance at a training course for RHS staff in April 1995 revealed the only flow types which were mis-identified to be smooth boundary turbulent and scarcely perceptible flow. Identifying the transition between *glide* (smooth boundary turbulent) and *pool* (scarcely perceptible flow) was best achieved by looking for downstream movement of surface debris 'bubbles' or distorted reflections on the water surface. Alternatively, throwing a floating marker into the channel and observing 'significant' downstream movement distinguishes *glides* from *pools*, as does a 'rod' placed vertically in a *glide* which causes downstream disturbance to flow.

Flow Type	**Description**
Free fall (FF)	Water falls vertically and without obstruction from a distinct feature, generally more than 1m high and often across the full channel width.
Chute (Ch)	Fast, smooth boundary turbulent flow over boulders or bedrock. Flow is in contact with the substrate, and exhibits upstream convergence and downstream divergence.
Broken standing waves (BSW)	White-water 'tumbling' waves with the crest facing in an upstream direction. Associated with 'surging' flow.
Unbroken standing waves (USW)	Undular standing waves in which the crest faces upstream without 'breaking'.
Chaotic flow (CF)	A mixture of the three flow types described above, occurring in a non-organised manner in the channel.
Rippled (Rip)	Surface turbulence does not produce waves, but symmetrical ripples which move in a general downstream direction.
Upwelling (Up)	Secondary flow cells visible at the water surface by vertical 'boils' or circular horizontal eddies.
Smooth boundary turbulent (SBT)	Flow in which relative roughness is sufficiently low that very little surface turbulence occurs. Very small turbulent flows cells are visible, reflections are distorted and surface 'foam' moves in a downstream direction. A stick placed vertically into the flow creates an upstream facing 'V'.
Scarcely perceptible flow (SP)	Surface foam appears to be stationary and reflections are not distorted. A stick placed on the water's surface will remain still.
No flow (NF)	Dry river beds associated with prolonged drought or ephemeral channels.

Table 3.6: Descriptions of flow types used to identify biotopes in the field.

Links between flow types and biotopes classified at the *transect scale* are indicated in Table 3.7. For the purposes of RHS the dominant flow type across a transect is recorded. The dominant flow type is that which is present in 50 percent or more of the transect. Where two flow types are present in equal proportions the 'faster' flow type is recorded as dominant. *Cascades* are an exception to this rule as chute flow (upstream of boulders) is normally associated with broken standing waves (downstream of boulders). Thus no single flow type dominates the transect, but chute flow, broken and unbroken standing waves occur in approximately equal proportions. In RHS these flow types are classified together as chaotic flow. Another exception is the '*run-rapid*' a high flow transitional biotope which has unbroken standing waves and rippled flow in approximately equal proportions. This biotope was not included in the RHS Field Methodology Guidance Manual (National Rivers Authority, 1995) as data collection for RHS, at this stage, took place in low to moderate summer flows. Section 4.2.3 provides a more description of the '*run-rapid*', based on its characteristic hydraulics, as revealed by discriminant analysis.

Dominant Flow Type(s)	**Minor flow type(s)**	**Biotope**
Free-fall	-	**Waterfall**
Chute / Broken standing waves	Upwelling	**Cascade**
Broken standing waves	Unbroken standing waves	**Rapid**
Unbroken standing waves	Rippled	**Riffle**
Rippled	Unbroken standing waves Smooth boundary turbulent	**Run**
Smooth boundary turbulent	Scarcely perceptible flow	**Glide**
Scarcely perceptible flow	-	**Pool** **Deadwater**
Upwelling	Rippled Smooth boundary turbulent	**Boil**

Table 3.7: Transect-level identification of biotopes by dominant flow type (where two flow types are listed as dominant, these occur in approximately equal proportions across the transect). Minor flow types are present in low percentages across a transect.

Preliminary results of the 1995 RHS data collection and analysis suggest that the adoption of strict flow type observational criteria has led to better definition of biotopes compared to 1994, and greater flow type diversity between and within segment types. Some trends between flow type frequency and segment types exist, National Rivers Authority, 1996a); however better discrimination is expected between biotope *sequences* and river segment types (see section 7.4).

As Sullivan (1986) noted, biotopes ("*channel units*") alter with discharge. Several authors have classified habitat units on the basis of visually distinguishable areas (Allen, 1951; Thorup, 1966;

Mackay, 1969; Barmuta, 1989), but without reference to the discharges under which they occur. More recently, Wadeson (1995b) has used the term hydraulic biotope to describe the 'instream flow environments' in a range of South African rivers and discharges. The RHS field guidance manual states that surveys should be carried out at low summer flows, to allow comparison of flow types and biotopes between sites. The identification of biotopes at different discharges may provide a tool for establishing ecologically acceptable flows and is discussed in sections 5.3 and 7.3. It may also be a practical tool in identifying the operation of 'patch dynamics' in freshwater ecological theory.

3.6 DATA STORAGE AND MANIPULATION

With the exception of the South Tyne site, three hydraulic calibrations were performed at all sites, corresponding to a low, mid and high flow. High flow data were not collected at the South Tyne site, as the high flows observed (see Table 3.1) exceeded those in which data could be collected safely. A second low flow calibration was performed at five sites affected by the flood events of January and February 1995 (see section 5.4). Table 3.8 summarises the level of data collection on which the analyses for the following chapters are based.

Level of data collection		Number of sample points
Site		**11**
Transect		**170**
Cell:	Pre-flood	2155
	Post-flood	1100
	Total	**3255**
Individual measurements: e.g. velocity	Pre-flood	6317
	Post-flood	1100
	Total	**7417**

Table 3.8: Number of sample points and level of data collection in the hydraulic dataset.

The first year of the research described in this thesis involved site selection, data collection and preliminary data visualisation, exploration and analysis. Due to the large number of data points and the need to perform various analyses, hydraulic data were entered into the Statistical Analysis System (SAS, 1985). Data from each calibration site and discharge were stored as individual site files and then combined into three larger datasets, corresponding to 'low', 'mid' and 'high' flow. This is simply a means of comparing the three calibration discharges, as the flow exceedence percentiles differed between sites. Low, mid and high flow datasets were grouped separately to determine the hydraulic significance of biotopes at different flows and the likely loss of hydraulic information by mapping only low flow biotopes. In 1994 and 1995 all RHS surveys were collected at low to moderate summer flow (ten sites were sampled bi-monthly in 1996).

There is an implicit hierarchy within the data which required an early decision about management. Whilst there is an observation for each sample point across the transect, the analyses require information about the characteristics of the transect itself. A file format was derived which obviated unnecessary duplication of data entry with the intention of minimising time required and opportunity for input errors. A numerical code was assigned to each flow type and physical biotope as SAS requires data input in this format. For those sites with both cell and transect level classification, separate files were created. These essentially contained the same data, with the 'type' variable entered for individual cells or once only for the transect-level biotope classification. For discriminant analyses (section 4.3) the three datasets ('low', 'mid' and 'high') were combined into two larger files containing the entire transect and cell-level datasets. In order to map biotopes at a smaller spatial scale, the 'transect classified' files were modified further to include the widths of dominant and secondary biotopes in addition to total wetted width. **Secondary biotopes** include the types listed in Table 2.7 which are present for a small percentage of the transect width, and thus constitute a small, but nevertheless significant, percentage of the total wetted area. Mapping of secondary biotopes is explained in detail in section 5.5.

3.6.1 Data visualisation: shear velocity plan distributions

All univariate and multivariate analyses were performed by SAS running on a Unix system. Since the start of the analysis SAS has become available on Microsoft Windows, which would have reduced the data analysis period. The Unix version of SAS requires a significant 'familiarisation period' and specialist statistical advice, both in program writing and interpretation of output. Graphics were drawn in Microsoft 'Excel' (5.5) after the SAS datafiles were transferred to 'Excel' using the Rapid Filer. With hindsight it would have been easier to enter the original data into 'Excel' and transfer files to SAS as the spreadsheet facility allows entire rows or columns to be inserted or deleted. This is not easily accomplished in SAS, so modification of files required individual data points to be entered or deleted separately.

According to Church (1992) disaggregated distributions of velocity and depth within a reach provide the best description of physical habitat. The ranges of depths and velocities at different sites provide some indication of the hydraulic conditions which are associated with different biotopes. Bovee (1996) describes how visualisation techniques can be used to identify ecologically significant habitat heterogeneity at small spatial scales, and its potential application to two-dimensional habitat modelling. Velocity distributions provide an indication of shear stress distribution within the channel; where isolevels are close together, velocity gradients and therefore shear stresses are higher (Wadeson, 1995b). Shear stress is believed to influence benthic invertebrate distributions and may be estimated from water surface slope or velocity profiles (Statzner and Müller, 1989). Alternatively shear velocity may be calculated from individual depth and velocity readings using the formula in section 3.6.2. Maps of shear velocity were drawn in Excel (Version 5) once the original datasets were transferred from SAS using the Rapid Filer. Data from all sites are presented as 3D surface diagrams (essentially site plans with isolevels joining cells of equal shear velocity) to provide a visual representation of the range of near-bed hydraulic variation across different sites under three calibration discharges.

To allow comparison of the hydraulic conditions at 'low', 'mid' and 'high' flow, the plans were drawn with the same shear velocity interval or band. At those sites where the area of exposed gravel changed at low flow, areas of exposed gravel were recorded as zero depth, and the lowest depth interval was plotted from a negative value to zero, to ensure exposed gravel areas were assigned to a different interval than wetted areas with very shallow flow. Maps of shear velocity are presented rather than attempting to produce PHABSIM simulations of Weighted Useable Area. Such plans may be easily interpreted by practitioners of several disciplines; for example geomorphologists may identify potential erosion sites; invertebrate ecologists may relate species distributions to patterns in shear stress, or estimate areas of high shear stress. These plans also illustrate several points of ecological relevance:

- Hydraulic 'patches' have spatial heterogeneity at a smaller scale to that of mesoscale units i.e. Frissell *et al.*'s (1986) 'riffle-pool' scale
- The location of low velocity 'patches' is related to features in RHS e.g. overhanging trees (cf. Skerne) or point bars (cf. West Allen)
- The number of hydraulic 'patches' generally decreases with increasing discharge, with possible implications for 'patch dynamics' theory
- Areas of low velocity persisting at high flow may serve as refugia in high or flood flow
- Vegetation causes hydraulic diversity and elevated depths at low flow in otherwise uniform glides

The technique of visualisation may have potential in future to investigate the sensitivity of different assemblages of morphological units and their related biotopes to changes in discharges. The use of Geographical Information Systems (GIS) in such studies is also growing, as described in section 6.4.

3.6.2 Data analysis

The first stage of analysis was to make all datafiles compatible with SAS programs. Depth and substrate measurements were then converted to metres within SAS. Substrate data was classified according to the reach scale classes listed in Table 2.1 and percentages calculated for individual sites and biotopes. Combined hydraulic indices were calculated for individual cells. Where substrate data were not available, a missing value was recorded. Statistical analyses required a numerical value of substrate size; bedrock was therefore entered as a value of 1000mm (1m).

Summary statistics of hydraulic indices were calculated for the cell-level database. The data were grouped by flow type and a univariate analysis performed in SAS. The mean, standard deviation, minimum and maximum values were calculated for each hydraulic index. Results are summarised in Appendix B2. The Froude number is a dimensionless index which incorporates variables that describe channel morphology, including depth, velocity and acceleration due to gravity (Newbury, 1984). It is calculated by the formula given by Statzner *et al.*, (1988):

$Fr = Vm / (gd)^{1/2}$

Vm = mean velocity at a sample point (ms $^{-1}$)

d = water depth at a given point (m)

g = acceleration due to gravity (9.81 ms $^{-2}$)

Relative roughness is an index of the effect of substrate size and water depth on hydraulics, calculated by the formula (Statzner *et al.*, 1988):

$R = s / (d \times 1000)$

s – substrate size at a sample point (intermediate diameter in mm)

d – water depth at a given point (m)

Shear velocity incorporates the variables depth, velocity and substrate by the formula (Statzner *et al.*, 1988):

$V^* = Vm / 5.75 \log_{10} (12.3d / (s/1000))$

Other innovative indices (i.e. proposed by the author rather than using existing references) which integrate all three recorded variables were calculated at the cell level: shelter index (SI), relative exposure (RE) and turbulence (TI). Shelter index was included as a possible discriminator between low and drought flow types, to determine whether, for example, deadwaters and shallow runs were hydraulically discrete. Relative exposure was included to determine whether flow types could be distinguished by indices calculated from depth and substrate measurements, to avoid having to take repeated velocity readings. Turbulence index was thought to be a good predictor of flow types and biotopes at high discharges. All the above indices are dimensionless so allow comparisons between different rivers. These additional indices are calculated as follows:

$SI = (s / 1000) / Fr$

Fr = Froude number

$RE = d / (s / 1000)$

$$TI = d \times Vm / (s / 1000)$$

All indices were used in combination to determine whether visually identifiable biotopes are hydraulically discrete units. These form the main statistical analyses of the thesis, which are the subject of Chapter 4.

4. STATISTICAL ANALYSIS: THE SIGNIFICANCE OF HYDRAULICALLY DEFINED BIOTOPES

Chapter Overview

Discriminant Analysis is a multivariate technique which provides an objective, statistical test of the subjective classification based on flow types. The principles and methodology of Stepwise Discriminant Analysis (SDA) and Kernel Discriminant Analysis (KDA) are described. In these analyses SDA determines which hydraulic indices best discriminate between flow types, whilst KDA uses significant discriminating indices to objectively allocate field data to flow type classes (4.1).

Two scales of analysis are performed; on data from individual sample points or 'cells', and on data "lumped" across transects. Input data consist of several combined hydraulic indices calculated from the basic field measurements of depth, velocity and substrate size, which discriminate between flow types and which represent macro- and micro-flow environments. SDA at the cell level determines a) which hydraulic indices most successfully discriminate between different flow types and b) which indices best describe individual flow types. KDA allocates individual observations to flow type classes. Cell level analysis validates the notion of biotopes as hydraulically discrete units. Transect level analysis establishes the degree of hydraulic information lost when physical biotopes are described by transect-based surveys, notably River Habitat Surveys (4.2).

Comparisons are made with similar studies in New Zealand and South Africa (4.3). Analysis of variance of Froude number distributions for different physical biotopes is performed to determine a) whether biotopes have characteristic hydraulics which may be extrapolated to other locations in similar reaches, and b) to identify sites with unique hydraulics for a given biotope type (4.4).

4.1 PRINCIPLES OF DISCRIMINANT ANALYSIS

Discriminant Analysis is a multivariate technique which provides an objective, statistical test of the subjective classification of physical biotopes by flow type. Using indices calculated from field data the validity of the visually classified biotopes may be tested statistically. Two issues may be addressed by discriminant analysis: firstly, whether physical biotopes are hydraulically discrete units, and secondly whether biotopes have characteristic hydraulic properties which may be extrapolated to similar biotopes within a reach. A related objective is to determine the appropriateness of the physical biotope as a unit for habitat inventories such as RHS.

Discriminant analysis is an appropriate statistical technique for the discrimination of flow types as it classifies data on the basis of multiple variables, considering hydraulic indices in combination. Unlike other multivariate analyses which test the hypothesis that groups or classes have *no significant difference in the distribution of a particular variable or variables* (e.g. analysis of variance), discriminant analysis tries to provide evidence that a particular observation belongs to an *a priori* defined group. The procedure is similar to cluster analysis where individual observations are classified by their combination of hydraulic variables, and each observation assigned to the most probable flow type class. It therefore provides a hydraulic classification on the basis of calculated hydraulic and substrate indices.

In order to compare data from different sites and discharges, dimensionless indices were used as input variables in the analysis. Both site-specific (e.g. substrate) and flow-dependent (e.g. depth and velocity) variables were used to calculate combined hydraulic indices which are likely to discriminate between flow types (and thus biotopes). Indices representing both the macro- and micro-flow environments were calculated based on a distinction made by Wadeson (1995b). Wolman (1955) first used the Froude number to define habitat types. As biotopes are subjectively identified and defined by surface flow types, the Froude number is likely to be the most useful hydraulic parameter for discriminant analysis, as this index was developed to distinguish between sub- and super-critical flow (Davis and Barmuta, 1989). Statzner *et al.* (1988) concluded that the Froude number is the best predictor of hydraulics identified on the basis of water surface roughness. The Froude number has shown evidence for its success as a good discriminator of physical biotopes in New Zealand and South Africa (Jowett, 1993; Wadeson, 1994; 1995b).

The Froude number represents the macro-flow hydraulic conditions which may influence fish distributions. Calculations for all indices are based on velocity readings taken at 0.6 depth, following the protocol of PHABSIM (see section 3.3). The current meter used (Marsh McBirney Flo-Mate 2000) cannot accurately measure near bed velocities. It is however, these micro-flow hydraulic conditions to which invertebrate biota are adapted (Statzner *et al.*, 1986; Statzner and Higler, 1988; Carling, 1992a). In order therefore to provide some ecological credibility, the indices listed below were calculated in addition to the Froude number. Shear velocity is advocated by Wadeson (1995) as a measure of the shear stress of a given area, expressed in velocity units; it represents near bed hydraulic conditions and is especially significant for benthic invertebrates (Davies *et al.*, 1994). Complex hydraulic characteristics are most useful in modelling relationships between the distribution of stream benthos and the physical habitat (Statzner *et al.*, 1988). Combined hydraulic indices which may discriminate between biotopes and have ecological relevance include the following (formulae are given in section 3.6.2):

- Shear velocity
- Relative roughness
- Relative exposure
- Shelter index
- Turbulence index

Statzner *et al.* (1988) use a wider range of combined hydraulic indices, including the Reynolds number and 'roughness' Reynolds number. These indices were not calculated in this study due to their poorer discriminating capabilities (Jowett, 1993; Wadeson, 1995b). Wadeson (1995b) concludes that the Reynolds number is of little value for a quantitative classification of biotopes. Substrate size data were included as variables in discriminant analysis, as early ecological studies focused on biotic assemblages associated with differences in substrate size distribution (Percival and Whitehead, 1929; Pennak and Van Gerpen, 1947; Kamler and Riedel, 1960). Substrate size *per se* may not influence biotic associations at the biotope scale; the combined influence of hydraulic and substrate variables on biotic assemblages has been emphasised by several authors (Allen, 1951; Thorup, 1966; Mackay, 1969; Barmuta, 1989). Traditionally, and until the Citrusdal Workshop which took place in South Africa in January 1995 (Rowntree, 1995a; Wadeson, 1995a), no formal methodology existed for describing characteristic associations of substrate and hydraulics; this is now possible with the development of a visual typology of conditions (i.e. "flow types").

4.1.1 Discriminant analysis methodology

The principal aims of discriminant analysis in this study are to determine:

i) which of the above indices are the best discriminators of flow type; are any of the indices redundant?
ii) which indices most successfully discriminate between, and therefore describe, individual physical biotopes?
iii) which flow types are hydraulically unique?
iv) does a particular flow type have characteristic hydraulics regardless of location?

Combined hydraulic indices are calculated in SAS (SAS, 1985) for individual observations or cells. Discriminant analysis (like cluster analysis) considers a range of indices and their associated values in order to classify each cell to the group which its hydraulics most closely resemble. The groups to which observations may be allocated in discriminant analysis are the subjectively identified flow types. Preliminary analyses used single indices and combinations of just two indices (Padmore *et al.*, 1995a). These had some success in discriminating between flow types, but multiple indices have been shown to produce better results (Wadeson, 1995b). The analysis was consequently repeated to include all the hydraulic indices. The first stage of analysis is a procedure called **Stepwise Discriminant Analysis (SDA)** which is employed to determine which, if any, indices are redundant i.e. do not discriminate between flow types. In SDA hydraulic indices are weighted according to their relative success in discrimination of flow types. The SDA procedure is similar to a stepwise multiple-regression analysis, in which the proportion of variation between flow types attributed to each index is calculated. Those indices which are significant discriminators are listed in a SDA output file (significance was defined at the 0.001 level). An F value is calculated for each index, indicating the relative contribution of each index as a flow type discriminator. SDA determines which, if any of the indices are redundant; and gives an indication of the most significant discriminating indices. SDA is used here as an alternative procedure to that used by Wadeson (1995b) for determining the relative contribution of different indices as flow type discriminators. Wadeson repeats a multiple range analysis using the Froude number as a single variable, and also in combination with the Reynolds number and 'roughness' Reynolds to determine the degree of improvement in discrimination when additional indices are calculated. SDA determines which of a set of variables are the best discriminators of flow types.

Kernel Discriminant Analysis (KDA) is the second stage of analysis. KDA tests whether *a priori* defined flow types (see Table 3.6) which are mapped by RHS field surveyors are significantly discrete in terms of their combined hydraulic indices. It is used in preference to other multivariate tests (such as those adopted by Wadeson, 1995b) which assume a normal distribution of residuals. When this assumption is not met, as in the case of the hydraulic variables calculated in this study, an Epanechnikov kernel analysis is the most appropriate (Silverman, 1986). KDA allocates each cell to a particular flow type on the basis of its hydraulic indices. If a particular observation does not resemble any of the flow types it is allocated to a 'miscellaneous' class. In effect this identifies cells with unique hydraulics within the sample dataset. The KDA software calculates the percentage of cells which are correctly classified for each flow type, and the percentage of different flow types to which misclassified cells are assigned. A list is produced in an output file indicating the location of misclassified cells (identified by reference to the transect number and reading across the channel e.g. 1.4: the fourth reading from the true left bank of the downstream transect), and the flow type to which it is allocated. This enables those sites or flow types which are hydraulically unique for a particular flow type to be identified. Additionally flow types which are commonly confused (and therefore not statistically discrete in terms of their combined hydraulic indices) are identified. The data analysis procedure is summarised in Figures 4.1 and 4.2.

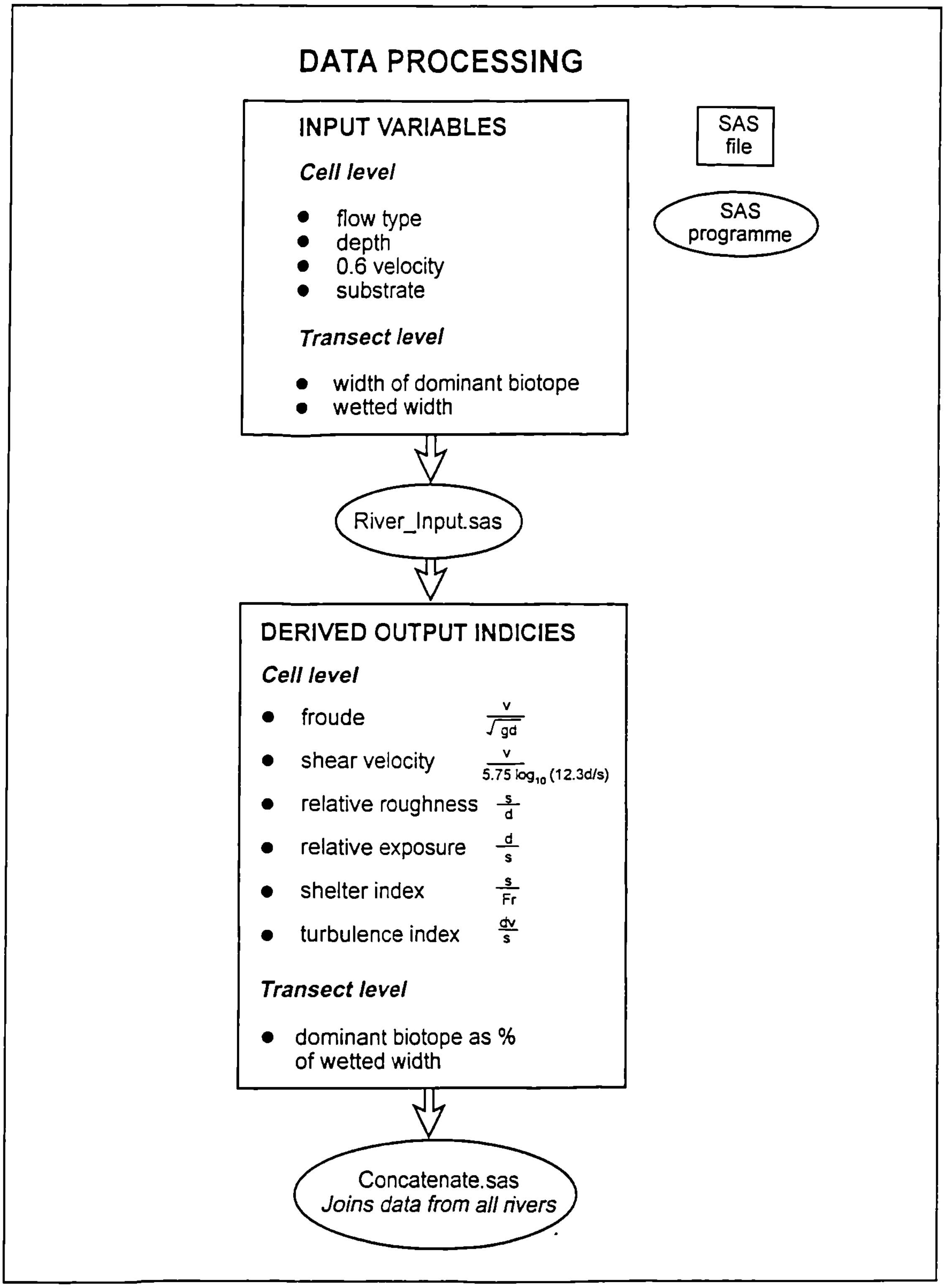


Figure 4.1: Data storage and processing within SAS (Statistical Analysis System).

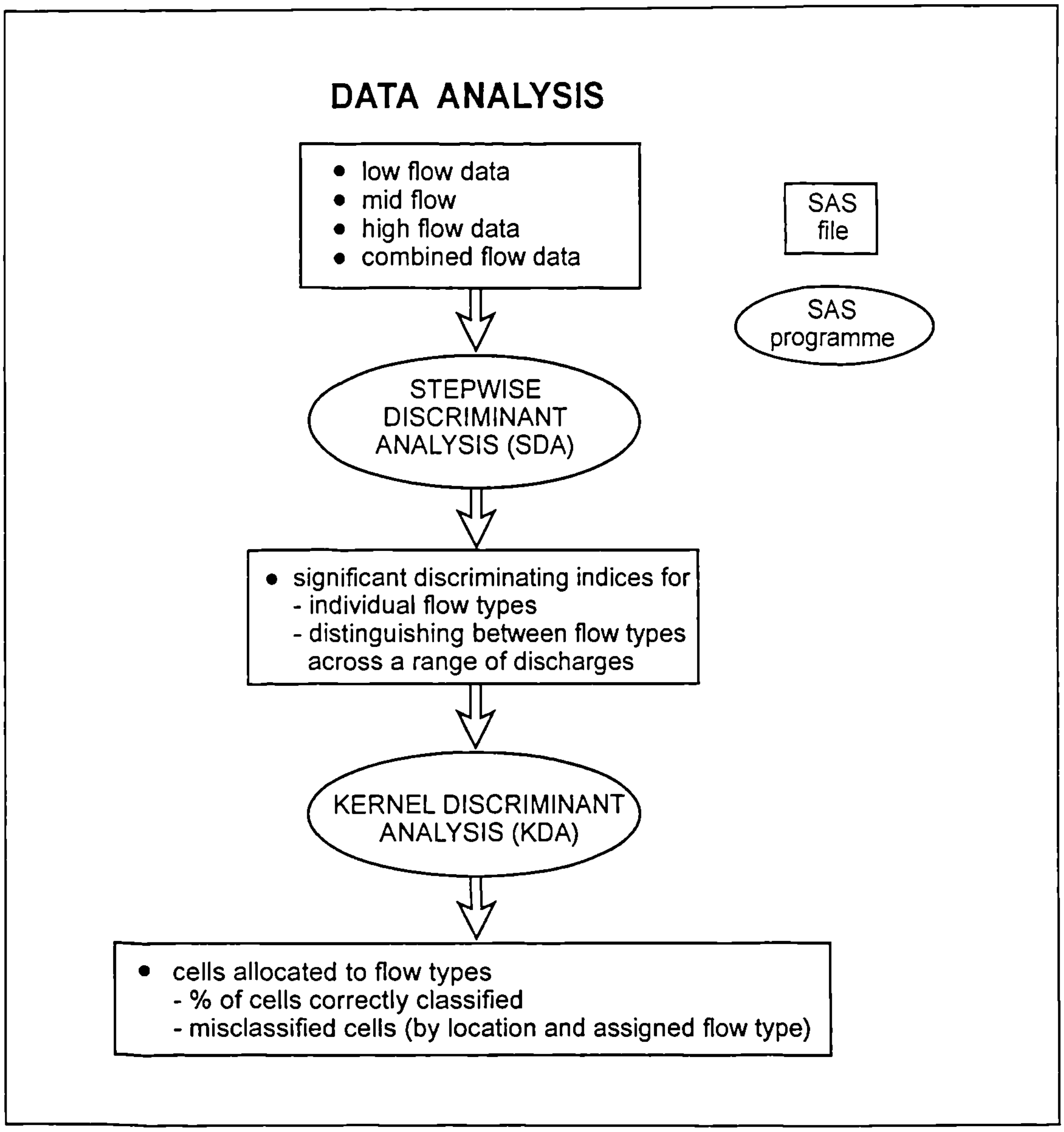


Figure 4.2: Discriminant analysis procedure in SAS (Statistical Analysis System).

4.2 SCALES OF RESOLUTION: CELL AND TRANSECT

With the development of the biotope concept in this study, and the realisation that transects are not uniform in cross-sectional morphology and hydraulics, it became clear that cell level classification was necessary to test the hypothesis that physical biotopes (rather than units delimited by channel-width transects) are hydraulically discrete units. Cell level classification was developed partly in response to the need for more accurate biotope mapping, and partly to improve preliminary discriminant analyses which used data from biotopes classified at the transect level. A significant number of cells were misclassified in earlier analyses (Padmore *et al.*, 1995a; 1995b - Appendix B1) as the KDA applied to individual cells, whilst transect-level analysis assigns one code to all cells within a transect corresponding to the dominant biotope for the transect. Wadeson (1995b) recognises the need for *"cell by cell classification of hydraulic biotopes rather than a general classification of transects"*. (Wadeson, 1995b, p86).

When using transect-level classification of biotopes, the SAS program codes each cell within the transect with the flow type which dominates the transect. Consequently several cells were assigned a different flow type in KDA to the flow type coded in the input data. Commonly 'misclassified' cells are associated with the following biotopes:

- **Secondary biotopes**: where more than one biotope is present across the channel width for a minor percentage of the total wetted width (includes biotopes associated with point or lateral bars, submerged mid channel bars or stable, vegetated islands)
- **Marginal biotopes**: located adjacent to the channel bank within the first few metres of the wetted edge
- **'Refugia' biotopes**: located in the wake of boulders or other features which alter local flow hydraulics e.g. overhanging tree roots, point bars etc.

Cell level classification was therefore adopted to improve the predictive success of KDA. Classification of flow types is described in section 3.5.1. A reduced dataset was available for cell-based KDA, including only those calibration sites and discharges with flow types identified for individual cells. As several of the sites were re-surveyed after high magnitude flood events, a cell-classified database existed which contained all sites and the full range of biotopes (albeit without three calibration discharges at each site). The cell and transect level *datasets are listed in Table 4.1*).

Site	**Level of data collection and discharge category**					
	Transect			**Cell**		
	Low	**Mid**	**High**	**Low**	**Mid**	**High**
South Tyne	✓	✓		✓	✓	✓
North Tyne	✓	✓	✓	✓	✓	
Wear (Stanhope)	✓	✓	✓	✓	✓	
Wear (Wolsingham)	✓	✓	✓	✓		✓
Harwood Beck	✓	✓	✓	✓		✓
West Allen	✓	✓	✓	✓		✓
Kielder Burn	✓	✓	✓			✓
Derwent	✓	✓	✓			✓
Till	✓	✓	✓			✓
Ouseburn	✓	✓	✓			✓
Skerne	✓	✓	✓			✓

Table 4.1: Transect and cell level classification of calibration sites.

4.2.1 Cell level analysis to discriminate between flow types

Eighteen site calibrations had data classified at the cell level. Stepwise discriminant analysis was performed on the total cell level dataset which includes all sites and a range of flows from Q_{100} to Q_{15}. Those indices which are considered to be significant in the model are listed below. The F value is a measure of the relative power of an index to discriminate between flow types. The index with the highest F value is most significant and shaded and/or bold printed to indicate this. Results of the SDA for the cell level dataset are listed in Table 4.2.

Significant discriminating index	F value
Froude number	**2520**
Shear velocity	100
Turbulence index	72
Relative roughness	47
Shelter index	27
Relative exposure	12
Substrate	11

Table 4.2: SDA of indices which discriminate between flow types classified at the **cell-level** across a range of sites and discharges.

It is evident that the Froude number is the best single discriminating index across a range of sites and discharges. This result is predictable, as the Froude number was developed to distinguish between flow types and essentially indicates the roughness of the water at the surface (Statzner *et al.*, 1988). Those indices which include substrate are more successful flow type discriminators, as particle size directly influences local hydraulics. Shear velocity is the second most successful discriminating index, which supports studies of invertebrate distributions in relation to shear velocity or shear stress (Statzner *et al.*, 1988; Statzner and Müller, 1989; Quinn and Hickey, 1994). Turbulence index is a relatively good discriminator of flow types across a range of discharges as it integrates the three main variables, depth, velocity and substrate.

Those indices considered to be significant discriminating indices in SDA (listed in Table 4.2) are used as input variables in KDA, to determine (a) the percentage of correct classifications and (b) the flow types to which misclassified cells are allocated. In similar work on New Zealand rivers (Jowett, 1993) approximately 65% of riffle, pool and run biotopes were accurately classified using the Froude number, which was considered to be an acceptable margin of error. With improved identification of biotopes a greater proportion of observations should be accurately classified for a given flow type. Hence 75% is selected as an arbitrary cut-off level in this study for the subjective classification to be considered robust.

When flow types were identified across a range of discharges the classification of rippled flow as one type led to a significant number of misclassified cells in the original KDA. Rippled flow is associated in the field with three physical biotopes:

i) *shallow runs*: at margins or local hydraulic 'patches' within riffles and cascades
ii) *runs*: biotopes spanning the full channel width, dominated by rippled flow
iii) *run-rapid* 'transitions': full-width biotopes, having both rippled and unbroken standing waves in approximately equal proportions

In *run-rapid* transitions unbroken standing waves form due to the momentum of flow rather than from roughness created by the substrate. Waves tend not to break as relative roughness is insufficient at these depths. Hydraulically, *run-rapid*s have lower velocities than *rapids*, but are generally faster than *runs* at low to moderate flow. In the original KDA a significant percentage of cells which were identified in the field as rippled flow in *run-rapids* were allocated to the unbroken standing waves flow type. Rippled flow in marginal biotopes was frequently classified as smooth boundary turbulent in KDA. Consequently, a distinction was made between rippled flow types in order to improve the discriminating success of the analysis.

Once a distinction had been made between shallow and deep rippled flow types, the main flow type to which 'misclassified' cells were consistently allocated was upwelling. In secondary flow cells velocity at 0.6 depth is not representative of mean velocities (Jarrett, 1984; Bathurst, 1988) so it is not legitimate to compare upwelling flow with other flow types defined by unidirectional velocities. The range of hydraulic indices (calculated from velocities at 0.6 depth) for upwelling flow show much variation in calculated indices; as a result this flow type acted as the 'miscellaneous' class described in 4.2. For the purpose of statistical analysis cells with upwelling flow were therefore removed from the dataset and the analysis repeated to produce the results in Table 4.3.

Flow type	Percent correctly classified	Flow types commonly allocated to misclassifications	Percent misclassifed and allocated flow type						
			BSW	Ch	USW	Rip	SBT	SRip	SP
Broken standing waves	100		X						
Chute	100			X					
Unbroken standing waves	93	BSW, Rip	3.0	0.4	X	3.0	0.6		
Rippled	85	SBT, USW	1.6		4.1	X	7.8		1.5
Smooth boundary turbulent	89		0.3		0.5	1.7	X	1.2	7.3
Shallow rippled	100							X	
Scarcely perceptible flow	98						1.0	1.0	X

Table 4.3: **Cell level** Kernel Discriminant Analysis of flow types (excluding upwelling) from a range of sites and discharges. (Abbreviations are listed in Table 3.6).

Cell level analysis permitted successful classification of flow types across a range of discharges and sites in the majority of cells. Misclassified cells may be explained in one of two ways: firstly, across a range of discharges hydraulic variation within a particular flow type may lead to some overlap in the hydraulic conditions of certain biotopes. Those cells identified in the field as rippled which are classified as smooth boundary turbulent in KDA (7.9 %) exemplify this point; rippled flow at low discharges being hydraulically similar to smooth boundary turbulent flow at higher flows. This reasoning applies to those cells identified as smooth boundary turbulent, but classified in KDA as scarcely perceptible flow; low flow *glides* being hydraulically similar to *deadwaters*. The second probable explanation is the influence of adjacent cells with a different flow type e.g. a significant proportion of cells identified as rippled are adjacent to riffles and have hydraulic conditions similar to adjacent unbroken standing waves. The remaining cells which were identified as smooth boundary turbulent but classified as scarcely perceptible flow are mainly adjacent to marginal deadwaters.

In summary, although a physical biotope may exhibit a variety of hydraulic conditions at the cell level, most cells of a particular flow type are within a statistically similar hydraulic range. This does not apply to biotopes where secondary flow cells dominate (upwelling flow), as velocities at 0.6 depth are not representative of mean values. Upwelling flow is considered to define a discrete biotope (*boil*) for its probable role in increasing oxygen concentration or food availability; boils may therefore be zones of high food production or availability. With the exception of upwelling flow, the majority of misclassified cells may be explained by the influence of adjacent cells with a different flow type on local scale hydraulics. These results imply that the physical biotope (as defined by flow type) is an appropriate scale for description and inventory of hydraulic units. Detailed velocity profiles are required to more accurately characterise *boils* i.e. biotopes dominated by upwelling flow (see section 6.2.3).

4.2.2 Hydraulic indices as descriptors of flow types

Having validated the use of flow types as hydraulically discrete units, the next stage of analysis is to determine which indices best describe a particular flow type. This may provide insight into links between physical biotopes and biota; hydraulic indices which characterise a particular flow type may be, at least in part, responsible for the associated biotic fauna. The use of hydraulic indices to discriminate between hydraulic units was first used by Allen (1951), who delimited units in terms of combinations depth, velocity and substrate ranges. SDA was performed to determine which hydraulic indices best characterised different flow types for data classified at the cell-level. Separate analyses were performed for each flow type, to ascertain which of the hydraulic indices accounted for the greatest hydraulic variation. The hydraulic indices listed in 4.1 were used as input variables; results are summarised in Table 4.4.

HYDRAULIC INDICES	FLOW TYPE							
	SPF	SRip	SBT	Rip	USW	BSW	Ch	Up
Froude number	1626	1305	1784	1301	1512	1504	2031	1327
Shear velocity	72	19	12	18	42	77	190	19
Turbulence index	19	11	67	17	24	18	18	17
Relative roughness		163		10			44	
Substrate						23	20	15
Shelter index	128				15			
Relative exposure			55		9			

Table 4.4: SDA for individual flow types. Numbers shown are calculated F values for a particular hydraulic index and represent the relative importance of the index in characterising the flow type (denoted by shading).

The Froude number is, by far, the best discriminator of all flow types. However, it can be seen that the combinations of hydraulic indices which best distinguish a particular flow type from the rest differ. Scarcely perceptible flow (*deadwaters*) is best characterised by the Froude number, shear velocity and shelter index, whilst smooth boundary turbulent flow (*glides*) is best defined by macro-hydraulic indices, namely the Froude number, turbulence index and relative exposure. A combination of macro- (Froude number) and micro-hydraulic indices (shear velocity) best characterise the higher energy flow types i.e. chute flow, broken and unbroken standing waves. The combination of hydraulic indices which distinguish particular flow types from the others may provide some indication of the type of organisms which are likely to be associated with each flow type, and could guide ecological sampling strategies e.g. invertebrate sampling. As flow types have different combinations of significant hydraulic indices they may be associated with different fish and invertebrate assemblages. The ecological relevance of flow types and biotopes is discussed in section 7.1.1.

4.2.3 Transect level analysis between biotopes

An issue relevant to the scale at which biotopes are identified is that of data "lumping". From a management perspective it is pertinent to establish what level of hydraulic detail is lost if biotopes are identified and described at the transect level, which is the most practicable level for extensive habitat surveys. This section attempts to address the issue, by "lumping" data at the transect level and re-running the discriminant analyses. All sites were classified at the transect-level which provides the basis of biotope mapping (as described in Chapter 5). A practical methodology for biotope inventory at the reach scale requires a procedure which permits accurate representation of biotopes without extensive field survey. Identification of features at 10 metre wide transects, spaced equally within a 500m length of river has been adopted by the National Rivers Authority in River Habitat Surveys (National Rivers Authority, 1996). Discriminant analysis of physical biotopes identified at the transect level has been performed to determine if a site is accurately described by classifying transects on the basis of the dominant flow type (and thus biotope). Transect-level analysis identifies whether cell-level information confirms or disrupts a field assessment of physical biotopes based on the dominant (or "full-width") biotope. It also provides an indication of the degree of information on cross-transect hydraulic variation which is lost by classifying transects by the dominant flow type.

Three separate SDAs were performed for 'low', 'mid' and 'high' discharge datasets representing summer drought flow, baseflow and winter high flow respectively. All variables which are significant at the 0.0001 level in SDA are considered to be appropriate input variables for **Kernel Discriminant Analysis**. The low and mid flow datasets include those biotopes which would be present in RHS data collection which is typically carried out at summer base-flow. A fourth SDA was performed on a combined dataset including all sites with 'low' and 'mid' flow calibration discharges, to investigate which indices best discriminate across a wider range of flows (and biotopes). Results are summarised in Tables 4.5-4.8.

Significant discriminating index	F value
Froude number	**209.5**
Shelter index	**84.4**
Relative roughness	36.9
Shear velocity	34.9
Relative exposure	30.7
Turbulence index	16.6
Substrate	13.9

Table 4.5: SDA for indices which discriminate between flow types classified at the **transect-level** for ‘low’ flow calibrations.

Significant discriminating index	F value
Relative exposure	**61.6**
Froude number	**52.8**
Substrate	28.8
Turbulence index	19.6
Relative roughness	15.6
Shear velocity	13.3
Shelter index	3.0

Table 4.6: SDA for indices which discriminate between flow types at the **transect-level** for ‘mid’ flow calibrations.

Significant discriminating index	F value
Froude number	**203.5**
Shelter index	**110.6**
Shear velocity	49.3
Relative exposure	47.8
Relative roughness	42.8
Substrate	36.8
Turbulence index	24.9

Table 4.7: SDA for indices which discriminate between flow types at the **transect-level** for low-mid’ flow calibrations.

Significant discriminating index	F value
Relative exposure	**197.7**
Turbulence index	**113.6**
Froude number	**112.8**
Relative roughness	56.7
Substrate	49.3
Shelter index	7.1
Shear velocity	3.3

Table 4.8: SDA for indices which discriminate between flow types classified at the **transect-level** for ‘high’ flow calibrations.

It is apparent that at the transect level the success of a particular index varies with discharge. The Froude number is the best discriminator at 'low flow', followed by the shelter index. Shear velocity, relative roughness and relative exposure contribute less, but approximately equally to the discrimination of flow types at drought discharges. At moderate flows ('mid' flow) relative exposure has slightly better discriminating power than the Froude number and no single index is capable of discriminating between flow types. As relative exposure is independent of velocity; it appears that velocities (and velocity-related indices) are less distinctive between flow types associated with moderate discharges. Variation in velocities within 'mid-flow' biotopes may show greatest overlap with other biotopes which will reduce the discriminating power of the Froude number and other velocity-related indices. At moderate flows it is therefore necessary to use all calculated indices to discriminate between flow types in KDA. When flow types associated with a wider range of flows from drought to baseflow are compared ('low-mid' flow), results are similar to the low-flow analysis. Biotopes associated with very low discharges, namely *deadwaters*, *shallow runs* and *glides* have low Froude numbers and a high shelter index which distinguishes them from biotopes associated with moderate flows (*glides*, *runs*, *riffles* and *cascades*).

At 'high' flow relative exposure and turbulence index are better discriminators of flow types than the Froude number. The discriminating power of these indices may be explained by the decreasing influence of bed morphology on hydraulics with increasing discharge; where the effect of individual particles diminishes and roughness becomes more related to overall channel dimensions (Morris, 1955) or the larger particle sizes, i.e. D_{84} (Maizels *et al.*, 1984). As a result, recorded velocities become more homogeneous between biotopes at higher flows, reducing the discriminating power of velocity related indices. Within 'high' flow biotopes a greater proportion of cells will have supercritical flow so the Froude number is a less successful discriminator than at low flows (Jowett, 1993; Padmore *et al.*, 1995b). Wadeson (1995b) explains that the transition between sub- and supercritical flow does not necessarily occur at a Froude number of 1 when individual velocity readings are taken rather than cross-sectional averages. In reality, the range of Froude numbers associated with a particular biotope may show considerable overlap with other biotopes (Padmore *et al.* 1995b; Appendix B2; Wadeson, 1995b). Additionally, the success of the Froude number in discriminating between flow types is dependent on accurate velocity readings. In fully turbulent flow associated with high discharges, velocity fluctuations may lead to some non-representative velocity readings

(Lane *et al.*, 1993); additionally, velocity at 0.6 depth is not representative of mean velocity in flow types associated with boulder bed channels i.e. cascades and rapids (Jarrett, 1984; Bathurst, 1988).

In summary, at the transect level all the calculated indices contribute to the discrimination of flow types, though the relative success of any one index is discharge dependent. The Froude number is the most useful single index across a range of discharges, but as Wadeson (1995b) notes, a *range* of hydraulic variables are required for classification of biotopes by multivariate statistical analyses. Wadeson, however, makes no distinction between the discriminating capabilities of indices at different discharges. The selection of indices for KDA should depend on the range of flow types and discharges. In this study multiple indices are especially significant in the classification of biotopes associated with moderate discharges. None of the indices calculated was found to be redundant in the SDA model (significance level = 0.001). Consequently all the calculated indices are included in the Kernel Discriminant Analyses.

The transect level classification of sites was used to assign transects to the biotope type which dominated the wetted width. Transects with both dominant and secondary biotopes present (see section 5.4) were classified only by the dominant biotope. All transects were assigned to one of the biotope classes listed in Table 2.7. KDA of the 'low' flow dataset shows a greater percentage of correct classifications for all flow types compared to the results of Jowett (1993), who distinguished between riffles, runs and pools. The majority of cells within a transect are correctly classified, although the average percentage of correctly classified observations is lower than for the cell level analysis (Table 4.9). *Riffles* and *glides* have the lowest percentage of correctly allocated observations. *Riffle* 'misclassifications' result from either chute flow over individual boulders at one or two points within the transect i.e. local scale *cascades*, or *marginal deadwaters* which are characterised by shallow rippled flow. Observations which are classified as *glides* in transect-level analysis but are allocated to a different biotope in KDA are associated with *marginal deadwaters* (shallow rippled flow or scarcely perceptible flow) which occur within *glide*-dominated transects.

Biotope	Percent correctly classified	Biotopes commonly allocated to misclassifications	Percent misclassified and allocated biotopes					
			Cascade	Riffle	Run	Boil	Glide	Pool
Cascade	96		X		2.7	0	1.3	0
Riffle	77	Run, cascade	7.2	X	9.7	0.2	3.5	2.4
Run	88	Glide	2.9	0.4	X	1.7	4.9	2.1
Boil	100		0	0	0	X	0	0
Glide	77	Pool, run	2.7	3.0	7.6	0.4	X	9.3
Pool	97		0	0	0.8	0	2.2	X

Table 4.9: Kernel Discriminant Analysis of the 'low' flow dataset.

Discriminant analysis of the 'mid-flow' calibrations representing moderate flows (Q_{35}-Q_{82}) produces the highest percentages of correct classifications overall (Table 4.10). The exception is for observations within *run*-dominated transects, where fewer cells are correctly classified compared with the low-flow results. *Runs* are associated with both riffle and pool morphological units, which accounts for an approximately equal allocation of misclassified cells within *run*-dominated transects to *glides* and *riffles*. Generally, however, biotopes tend to span almost the full channel width at moderate discharges and have a lower proportion of secondary biotopes at the channel margins. Secondary biotopes explain the majority of 'mid' flow misclassifications in *glide*-dominated transects (approximate equal allocation to cascades and riffles), and occur in transects located at a boundary between two biotopes which is diagonal to the direction of flow. Examination of the SDA list of 'misclassified' cells shows the majority of them to be located within the first few metres from the water's edge, and usually associated with adjacent cells or 'patches', rather than isolated cells within the transect.

Biotope	Percent correctly classified	Biotopes commonly allocated to misclassifications	Percent misclassified and allocated biotopes						
			Rapid	Cascade	Riffle	Run	Glide	Pool	Boil
Rapid	100		X						
Cascade	98	Run, cascade		X	0.4		1.6		
Riffle	95	Glide	0.4	2.1	X		1.7		0.8
Run	81		0.7	4.8	5.0	X	6.1		2.4
Glide	87	Pool, run	1.3	3.5	3.7	2.0	X		2.5
Pool	100							X	
Boil	100								X

Table 4.10: Kernel Discriminant Analysis of the 'mid' flow dataset.

Combining the 'low' and 'mid' flow datasets leads to a higher percentage of misclassifications for *riffles*, *runs* and *glides* (Table 4.11). When comparing moderate discharges with very low flows a significant proportion (10.5 %) of points within low flow *runs* (classified by rippled flow dominating the transect) are hydraulically similar to *glides*; this further supports the division of rippled flow into 'runs' and 'shallow runs' (see section 4.2.1). In the latter, the water surface is disturbed due to its close proximity to the channel bed, but velocities are sufficiently low that the biotope resembles *glides* in terms of its overall combined hydraulic indices. This argument is true also for *glides*; low flow glides (with scarcely perceptible flow) may be flowing so slowly that they are hydraulically similar to *pools* or *marginal deadwaters* when compared with biotopes across a range of discharges.

Biotope	Percent correctly classified	Biotopes commonly allocated to misclassifications	Percent misclassified and allocated biotopes						
			Rapid	Cascade	Riffle	Run	Glide	Pool	Boil
Rapid	100		X						
Cascade	93		1.9	X	1.9		2.8		0.4
Riffle	82	Cascade, run	1.5	5.9	X	3.8	3.0	1.7	2.1
Run	70	Glide, riffle	2.1	5.3	7.0	X	10.5	1.5	3.6
Glide	73	Pool, run	2.5	3.5	4.7	5.8		6.9	3.6
Pool	93		0.5		0.7		4.4	X	1.4
Boil	100								X

Table 4.11: Kernel Discriminant Analysis of the combined 'low' and 'mid' flow datasets.

As a result of early KDA analyses a distinction was introduced between *runs* and *run-rapid* transitions which are present across a wide range of discharges. In the original analyses no distinction was made between low to moderate flow runs (dominated by rippled flow) and high flow runs (rippled flow and unbroken standing waves in approximately equal proportions). As a result, the preliminary high flow KDA correctly classified only 38% of *runs*, with the remainder being allocated to *riffles*, *rapids* or *boils*. It was clear from field observations and photographs that this allocation was incorrect; however in the absence of an alternative class KDA simply allocates cells to the most hydraulically similar flow type. Careful examination of photographs at high flow allowed a distinction to be made (for the majority of transects) between *runs* and *run-rapid* transitions. Some transects which were originally classified as 'run' were re-coded as a discrete 'run-rapid' class in the analysis reported here. This produced a higher number of correct classifications; at 'high' flows more sample points are correctly classified, as flow becomes more homogeneous and fewer flow types exist. Results are summarised in Table 4.12.

Biotope	Percent correctly classified	Biotopes commonly allocated to misclassifications	Percent misclassified and allocated biotopes				
			Rapid	Run-rapid	Run	Boil	Glide
Rapid	98		X	1.3	0.7		
Run-rapid	91	Run, rapid	2.5	X	3.8	1.2	1.5
Run	77	Run-rapid, boil	4.5	10.0	X	5.0	3.5
Boil	100					X	
Glide	99				1.0		X

Table 4.12: Kernel Discriminant Analysis of the 'high' flow dataset.

Even after re-classification of 'runs' and 'run-rapid' transitions 10% of sample points within 'runs' are more hydraulically similar to 'run-rapids', and approximately 5% allocated to *rapids* and *boils* by KDA. This indicates that the 'run-rapid' is truely a transitional biotope with hydraulic conditions characteristic of both *runs* and *rapids*. The majority of misclassified 'run' sample points are from the Derwent (36% of all misclassifications) and the North Tyne (16%). Submerged stands of vegetation alter the hydraulics at the Derwent site; misclassified 'run' cells are assigned to *riffles* or *boils*. In the regulated North Tyne site reduced roughness from the armour layer (Sear, 1992) may lead to faster than average velocities in *runs*. This hypothesis is supported by the relatively high percentage of sample points at the North Tyne site classified at the transect-scale as *runs*, but which are assigned to either the *run-rapid* or *rapid* class by KDA. The influence of site-specific instream features on the hydraulic conditions of a particular biotope are discussed in section 4.4.

A possible explanation for a small percentage of misclassifications is errors in the subjective, visual classification of physical biotopes. Early hydraulic calibrations identified biotopes on the basis of a literature review of instream features (see Table 2.3). It is probable in the early stages, that transitions between, for example, *runs* and *glides* were inaccurately identified in terms of flow type. Transect-level classifications were checked before the final analysis was run by careful observation of photographs taken at each transect; however, the possibility of some incorrectly classified biotopes remains. Similar reservations were made by Wadeson (1995b), who initially classified the higher energy biotopes (rapids and cascades) as riffles. Wadeson developed the 'chute' biotope after repeated hydraulic calibrations and adoption of the flow-type classification. Identification of biotopes by flow type developed similarly in this study, which may account for some of the misclassified cells.

4.2.4 Comparison of cell and transect level analysis

As indicated in Table 4.1 not all calibration sites have cell-level data. An inherent problem when performing discriminant analyses on data classified at the transect level is the nature of the analysis procedure. All cells within a particular transect are included in the hydraulic characterisation of the flow type which dominates the transect. This could include cells of a different flow type, which may alter the overall hydraulic conditions which characterise the dominant biotope. To permit a legitimate comparison of cell and transect level analysis, SDA and KDA have been **repeated at the transect level for those sites and discharges with both cell and transect level identification of biotopes** (Table 4.1). The cell level analysis has been repeated to include upwelling flow; results are summarised in Table 4.13. These may be compared with a transect level classification of the same sites, which includes all biotopes associated with the flow types in Table 4.14, with the exception of *deadwaters*. Even at the lowest discharges none of the transects had a *deadwater* as the dominant biotope, although smooth boundary turbulent flow is present for small percentages of the wetted width. Results of transect level discriminant analysis are summarised in Table 4.14.

Flow type	Percent correctly classified	Flow types commonly allocated to misclassifications	Percent misclassified and allocated flow types							
			BSW	Ch	USW	Rip	Up	SBT	SRip	SP
BSW	100		X							
Ch	100			X						
USW	91	BSW, Rip	2.9	0.3	X	2.9	1.9	0.5		0.5
Rip	80	SBT, Up	1.6	0.2	4.2	X	5.1	7.5		1.4
Up	100						X			
SBT	88	Up	0.2	0.2	0.4	2.2	0.4	X	1.2	7.4
SRip	100								X	
SP	97						0.6	1.2	1.2	X

Table 4.13: **Cell level** Kernel Discriminant Analysis of sites across a range of discharges, using the dataset from Table 4.3. Abbreviations listed in Table 3.6.

Biotope	Percent correctly classified	Biotopes commonly allocated to misclassifications	Percent misclassified and allocated biotopes							
			Rapid	Run-rapid	Cascade	Riffle	Run	Boil	Glide	Shallow run
Rapid	100		X							
Run-rapid	64	Glide, riffle, rapid	5.5	X	1.4	6.1	4.4	5.5	8.9	4.2
Cascade	77	Riffle, shallow run	2.3	0.3	X	4.8	1.9		1.3	2.4
Riffle	89	Run, shallow run	1.1			X	5.0	1.1	1.2	2.6
Run	60	Riffle, shallow run, run-rapid	1.7	5.9	4.7	13.5	X	1.3	5.2	7.7
Boil	96					1.8	0.4	X		1.8
Glide	66	Shallow run, boil	0.8	2.3	4.4	3.1	5.3	5.7	X	12.4
Shallow run	98						0.6	0.4	1	X

Table 4.14: **Transect level** Kernel Discriminant Analysis of sites across a range of discharges, using the dataset from Table 4.3.

Taking 75% as an arbitrary point above which KDA is considered to successfully classify subjectively defined flow types, is may be concluded that **cell level classification** is successful in the majority of flow types. Rippled flow has the smallest percentage of successfully classified cells; a significant proportion of these cells occur at the Derwent site and are classified as unbroken standing waves or smooth boundary turbulent flow in KDA. Here stands of submerged vegetation are altering local scale hydraulics; as is the case at the Skerne site. Vegetation at the Skerne accounts for most of the cells identified as smooth boundary turbulent in the field which are classified as scarcely perceptible flow in KDA. Upwelling flow cells within transects dominated by rippled flow at high discharges, e.g. the Till and Wolsingham sites, accounts for those cells which are identified as rippled flow, but which are assigned to upwelling flow in KDA. In summary, identification of flow types at the cell level corresponds to the identification of hydraulically discrete 'patches', provided the sample points are not associated with vegetated sites or biotopes dominated by secondary flow. Reaches with upwelling, turbulent or supercritical flow and vegetated channels require more detailed velocity profiles to accurately characterise their hydraulics; readings at 0.6 depth are of little relevance in cells where a logarithmic velocity profile may not exist. This is discussed further in section 6.2.

Comparison with associated biotopes reveal that **classification at the transect** scale is inadequate for biotopes with a range of hydraulic conditions i.e. *run-rapid* transitions, and those which often do not span the channel width e.g. *runs* and *glides*. Transect level classification of biotopes produces a lower percentage of correct classifications than cell level analysis of the same dataset. When transects are classified by the dominant flow type there will inevitably be some sample points within the transect which have a different flow type. These include marginal biotopes and small scale hydraulically discrete 'patches' e.g. *chutes* within *riffles*. The significance of marginal and secondary biotopes for instream habitat is discussed in section 5.5 in the context of biotope mapping and determining appropriate scales for inventories of physical habitat.

4.3 INTERNATIONAL COMPARISON OF RESULTS

Jowett (1993) was the first author to apply the Froude number as a discriminator of visually identified biotopes. However he identifies only three biotopes; the riffle, run and pool, and his results are calculated from a single discharge. Consequently, his 'threshold' Froude number values (0.18 for pool-run; 0.41 for run-riffle) must be interpreted with caution. A better comparison is possible with the work of Wadeson (1995b), who identifies all the biotopes listed in Table 3.7, within a range of geomorphological channel types and discharges.

Wadeson's Froude number distributions for 'glides' are an order of magnitude greater than those calculated for smooth boundary turbulent flow (which characterises glides) in this study. South African 'glides' flow over smooth bedrock (Wadeson, 1994; 1995b), unlike the majority of UK *glides* which occur over small cobble to silt substrate, often in man-modified channels. Thus South African 'glides' are relatively shallow and fast, with Froude number distributions similar to UK chutes; by comparison, UK *glides* are deep and slow with lower Froude numbers. Mis-identification of features which would be classified as *glides* in the UK, but have been identified as pools in South Africa (a biotope with "slow to almost still" velocity is considered to be a 'pool') (Wadeson, 1994) may also explain the higher Froude numbers of South African 'pools' compared to those in the UK. A recent discussion with South African ecologists (King; Tharme, personal communication) has led to a consensus of terminology between the UK and South Africa. The term *glide* will be retained for the smooth boundary turbulent flow associated with UK channel types and substrates. Wadeson (1995b)'s 'glides' will be renamed *spills* which is similar to the classification of 'skimming flow' proposed by Davis and Barmuta (1989) and adopted by Young (1992). Chute flow will be restricted to flow which is funnelled between two substrate particles (usually boulders). No distinction was made in the work reported in this thesis between chutes and *spills* within *cascades*. As they are known to be associated with discrete biotic assemblages (King, personal communication) any further work will adopt this new classification and terminology. This is especially pertinent for headwater streams with extreme relative roughness and flow range (Young, 1996).

Froude number distributions for *runs* (rippled flow) and *riffles* (unbroken standing waves) show a greater range than those calculated by Wadeson, which may be explained by to the increased discharge range over which flow types were identified. Higher Froude numbers are also expected in the UK data set, as calculations are based on observations of a particular *flow type*, not biotope-averaged data. Rippled flow and unbroken standing waves are found (albeit much less frequently) as localised cells within high flow biotopes, notably *rapids* and *run-rapids*. As the South African data is biotope-averaged, rippled flow and unbroken standing waves with higher Froude numbers will be grouped with cascades or rapids. Lower values for rippled flow and unbroken standing waves may also be partly attributable to the inclusion in the present data set of negative velocities, which could not be calculated with the mechanical current meter used by Wadeson (1995b).

Chute flow and broken standing waves in this study show higher Froude number distributions compared to the associated chute, cascade and rapid biotopes in the South African study. This may be explained in one of three ways; firstly a larger number of sample points in the UK study will inevitably produce greater variation in Froude numbers. Secondly, higher discharges result in higher Froude numbers in the UK data; indeed supercritical flow with Froude numbers exceeding unity (Davis and Barmuta, 1989) are recorded. The final explanation for the disparity in results is the possible mis-identification of certain flow types by Wadeson, which would influence the distribution of Froude numbers by flow type. Wadeson's classification of biotopes by use of a matrix of flow types and associated substrate types (Wadeson, 1995b) was formalised at the Citrusdal Workshop, after much of his data collection had been undertaken and biotopes identified on the basis of a literature review (Wadeson, 1994). This retrospective classification which the author concedes may have misclassified some biotopes is likely to explain the discrepancies in Froude number distributions between those in this thesis and those of Wadeson.

With the exception of *chute* flow and *smooth boundary turbulent* flow, the results presented here are within a similar statistical range as those of Wadeson, which substantiates the identification of biotopes (or their associated flow types) as hydraulically similar units irrespective of location. The next section considers the hydraulics of individual biotopes, to determine whether all biotopes are similar regardless of location, or whether a particular site has an anomalous hydraulic distribution, which may influence the results of the discriminant analysis.

4.4 INTER-SITE COMPARISONS OF BIOTOPE HYDRAULICS

Statistical differences in the hydraulic indices of subjectively identified biotopes were investigated in 4.2. This validated the classification of biotopes by flow type, raising the issue of marginal and secondary biotopes which occur as 'patches' within transects. This section of analysis examines the hydraulic characteristics of particular biotopes, to establish if all observations classified as a particular biotope are hydraulically alike. This is the crux of being able, or not, to apply flow type assessments in the field to river network typologies to determine ecologically acceptable flows. If all biotopes identified as a given type are hydraulically similar, hydraulic characterisations produced in this study may be extrapolated to similar biotopes at other locations e.g. as part of hierarchical catchment models. Conversely, if a particular type of biotope has unique hydraulic conditions at a specific site, differences in the distribution of hydraulic indices may be related to local site features.

4.4.1 Froude number distribution: a visual representation of hydraulics

Box and whisker plots of hydraulic indices which characterise flow types provide a visual representation of hydraulic similarities between biotopes, indicating any possible outlier sites. Indices used were those shown in Table 4.4 to be the best descriptors of different flow types. The Froude number has been shown to be the best *single* discriminatory index of flow types across all flow types and discharges, both in this study, in New Zealand (Jowett, 1993) and in South Africa (Wadeson, 1995b). To summarise the distribution of Froude numbers for a particular flow type, a univariate analysis was performed within SAS. Maximum, minimum and quartile (25th and 75th percentile) values of the appropriate hydraulic indices were calculated for flow type groups (at those sites with **cell level** classification of flow types). Results are presented in Figure 4.3.

Within low energy biotopes (deadwaters, pools and glides) the distribution of Froude numbers is similar across all sites regardless of overall discharge. The range of the distribution is small which indicates low energy. Rippled flow and unbroken standing waves show a greater degree of variation, but mean Froude numbers are similar at different sites and discharges. The 'threshold' Froude numbers for runs and riffles of 0.18 and 0.41 respectively (Jowett, 1993) are exceeded in the distribution of Froude numbers for corresponding flow types, which is to be expected considering the greater discharge range. The Skerne site appears to be unique in its Froude number distribution for rippled flow, due to the uniform cross-sectional topography and substrate size distribution which reduce hydraulic variation at this site (see also section 6.2.2 for a review of the influence of vegetation on hydraulic conditions). This is typical of engineered channels.

There is some overlap in the Froude number distributions of rippled flow, unbroken standing waves and broken standing waves. However the interquartile range appears to be significantly different between flow types. Chute flow and broken standing waves show the greatest range and variation between sites, indicating the high degree of turbulence with which these flow types are associated. The distribution of Froude numbers at different sites does not show a common pattern for chute flow and broken standing waves. The Harwood Beck site appears to have a unique distribution for

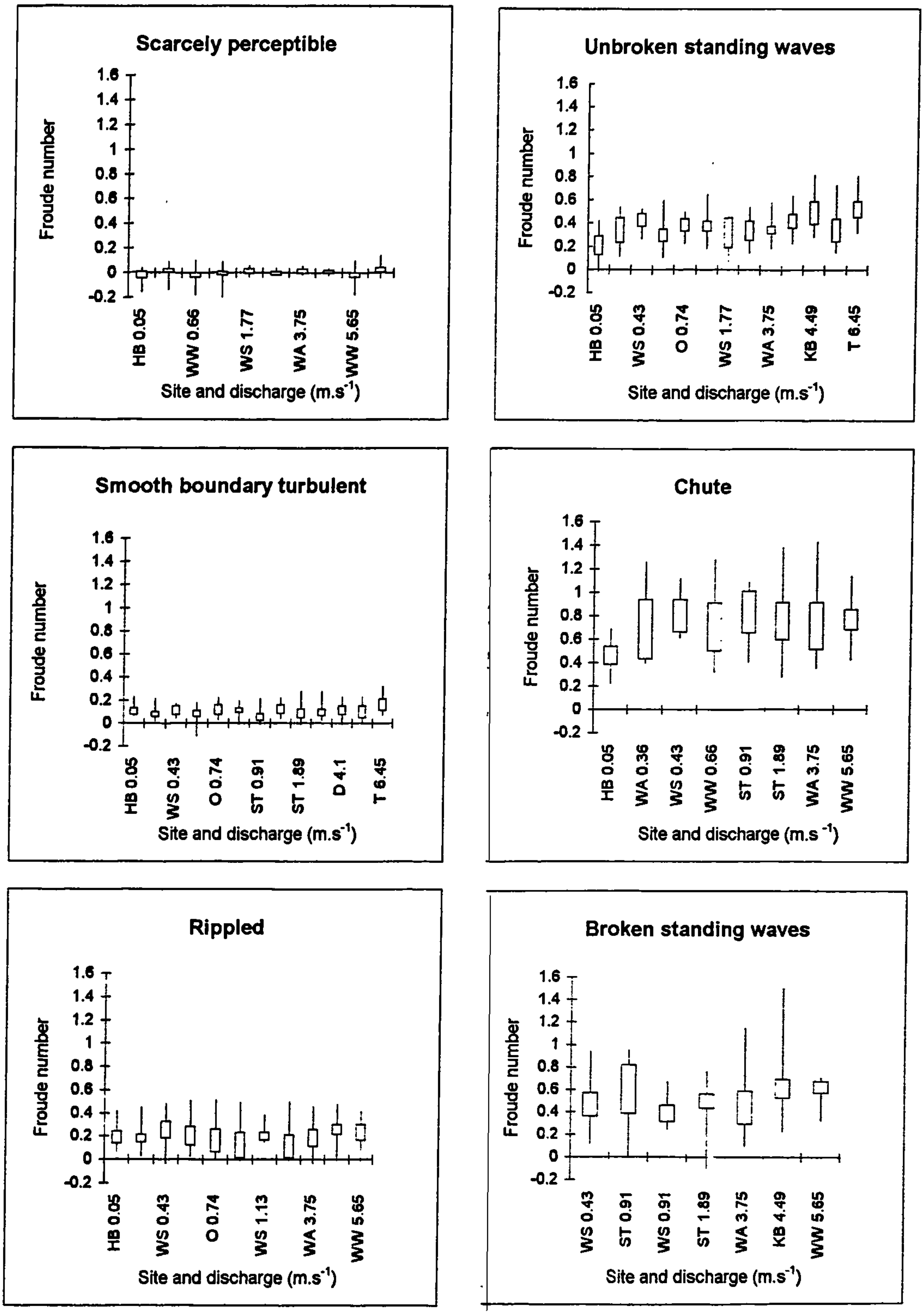


Figure 4.3: Froude number distributions for flow types classified at the cell-level. (HB=Harwood Beck; WW=Wear (Wolsingham); WS=Wear (Stanhope); WA=West Allen; O=Ouseburn; ST=South Tyne; D=Derwent; T=Till; KB=Kielder Burn).

chute flow; this is the only site where *cascades* are recorded in drought flows, as the substrate size (boulders) is sufficiently large to create broken standing waves at very low flows. In order to determine whether distributions of hydraulic indices are statistically similar between sites, analysis of variance may be performed.

4.4.2 Hydraulically unique sites

Analysis of variance tests the hypothesis that the distributions of hydraulic indices for a particular flow type are independent of site and discharge. Unlike discriminant analysis, one-way analysis of variance (ANOVA) is performed on just one variable. From the analysis described in 4.2.2 the Froude number is the best descriptor of just two flow types at the scale of individual biotopes i.e. unbroken standing waves and chutes (see Table 4.4). Relative exposure more successfully characterises smooth boundary turbulent and rippled flow, whilst relative roughness is the best index for upwelling flow. Analysis of variance is performed for the index which best characterises each flow type (the index with the highest F value). Ideally ANOVA should only be performed on datasets in which the subgroups are normally distributed, but there is no agreement over the appropriate test for equality of variance; Bartlett's test has been challenged by Box (1954). However, Earickson and Harlin (1994, p155) suggest that the importance of the assumptions of both normality and homogeneity are overrated. Thus, although these assumptions are unlikely to be met, it was not considered necessary to transform the data. Assuming analysis of variance is a sufficiently robust test, a significant difference at the 0.001 level indicates that *at least one* mean is different from the rest. It does not indicate whether *all* sites are significantly different, or if just one site has a unique distribution of a particular hydraulic variable (Wadeson, 1995b; p124). To overcome this, the Scheffé test is performed; it identifies, for biotopes where one or more site(s) has a significantly different distribution of a given hydraulic index, *which* sites are unique. Using this test, assumptions of normality and equality of variance (which underlie the use of ANOVA) may be relaxed (Scheffé, 1959). The Scheffé test is conservative in testing differences among several means (Clarke and Hosking, 1986), which means that only very dissimilar distributions will be identified. Sites are identified in SAS by a 'rivname', listing the site and relative discharge e.g. Wolsingham 'high'.

Unlike the visual representations of Froude number distributions (Figure 4.3) which are calculated from those sites with cell-level identification of flow types, ANOVA and the Scheffé test were performed on the datatset with biotopes classified at the transect-level biotopes. This enabled all sites and calibration discharges to be included in the analysis. Results of the Scheffé test revealed very few flow types to have distributions outside the range which statistically characterises a particular biotope. Table 4.15 indicates those sites with significantly different distributions of characteristic hydraulic indices for a particular biotope.

The engineered Skerne channel is the only anomaly for smooth boundary turbulent flow. Channel deepening and removal of the natural topographical variation in cross-section and long-profile produce unique hydraulic conditions for the Skerne *glide* compared to the other smooth boundary turbulent observations. Macrophyte vegetation at the Derwent site produces unique hydraulic conditions within the *riffle*. The influence of instream vegetation is discussed in detail in Chapter 6. The distribution of hydraulic indices in upwelling flow cells was unique at each site, due to the complex nature of this flow type. More detailed analysis of the hydraulics of upwelling flow are described in Chapter 6. In general, there were few significant intersite differences between the hydraulic distribution of a particular flow type. This implies that a biotope identified by flow type has the same range of hydraulic conditions to any biotope of this type, regardless of location or channel type. In terms of extrapolation this implies that biotopes and their hydraulic characterisations may be transferred anywhere in a channel network, provided a scaling factor is included (see section 7.4.1).

Biotope	**Exceptional site(s) (Flow)**	**Probable explanation**
Glide	Skerne (low, mid,high)	• Engineered channel which is over-deep, producing a low relative exposure index
Riffle	Derwent (low)	• Macrophyte vegetation
Rapid	Allen (high)	• Bedrock has reduced resistance to flow , producing a high turbulence index
Boil	All	• Upwelling flow complex and cannot be described by a single hydraulic index

Table 4.15: Summary results of the Scheffé test of biotopes' Froude number distributions by site and calibration discharge.

4.5 SUMMARY

Discriminant analysis provides an objective, statistically based classification of individual observations on the basis of combined hydraulic indices. The indices used in the analysis are those which have been shown to discriminate between flow types at different discharges and which are representative of both macro- and micro-scale flow conditions. Combinations of hydraulic indices which best define or characterise individual biotopes vary between flow types. Across a range of flow types and discharges all the calculated indices contribute to the discrimination of flow types, although the relative discriminatory power of different indices changes with discharge and associated flow types. Overall, the Froude number is the most successful single discriminatory index at all discharges.

Cell level discriminant analysis produces successful classification in approximately 95% of observations, when percentages are averaged for individual flow types (Table 4.12). These figures compare favourably with those of previous studies in New Zealand and South Africa, due to the more precise classification of biotopes by flow types. Successful allocation occurs across a range of sites and discharges, thus the identification of biotopes may be applied with care to any river. The major contributions to the understanding of biotope hydraulics provided by cell level discriminant analysis include:

1. Identification of 'patches' *within* biotopes having discrete hydraulic conditions
    - marginal deadwaters / deadwaters downstream of point and lateral bars or boulders
    - vegetated areas (submerged or emergent)
    - chutes over boulders in cascades or riffles
    - man-made channel obstructions or local influences on flow e.g. overhanging trees

2. Identification of biotopes with 'unique' hydraulics
    - glides in engineered channels
    - sites with instream macrophyte vegetation
    - upwelling flow

3. Checks on transect-level data aggregation
    - determination of the dominant hydraulic biotope where two appear to contribute approximately equal areas in any one transect
    - explanation of misclassifications

Transect level classification of biotopes produces a lower percentage of correct classifications than cell level analysis of the same dataset. As discussed in 4.3.3 the reduced success of KDA is attributable to the identification of sample points within a transect having different distributions of hydraulic variables to those of the dominant flow type. The major limitation of data aggregation at the transect level is that flow types which are present, but do not usually dominate a transect are under-represented. These include those flow types listed in 1 and 2 above. The need to refine further the hydraulic characterisations of upwelling flow, supercritical flow in rough channels and vegetated channels is the subject of section 6.2.

As flow types have been proven to be hydraulically dissimilar, they therefore describe a suite of discrete biotopes. In the absence of research to determine their ecological significance, the full range of flow types should be included in field inventories such as those used in RHS. Preliminary RHS analysis to determine characteristic flow types associated with the original eleven segment types, indicates that two flow types are not commonly identified as dominating transects, notably upwelling and chute flow (Holmes, personal communication). The former commonly occurs within *glides* (smooth boundary turbulent flow) or *runs* (rippled flow), and is associated with bends in sinuous reaches and high flows. Chute flow rarely dominates a transect, except in fractured bedrock sites or step-pool sequences. The ecological significance of chute flow within cascades to simulid larvae has been noted by Palmer (1991), which justifies the retention of chute flow in RHS. Both chute flow and upwelling are commonly recorded in RHS inventories by the 'sweep-up' section, which records all features present in the sample reach (Raven *et al.*, 1996). The concept of dominant and secondary biotopes and transect-level classification of biotopes has implications for the use of RHS data in determining Habitat Quality Indices; results are presented in section 5.5 and discussed in sections 7.1 and 7.4.

Comparisons of calculated hydraulic indices by biotope types are consistent with the work of Wadeson (1995b). Discrepancies are explained by the high proportion of bedrock controlled sites in South Africa, which produces higher velocities in smooth boundary turbulent flow (bedrock glides) compared to the alluvial glides which dominate the channel network of England and Wales. Biotopes with unique hydraulics for a particular flow type are associated with channelized rivers, instream vegetation and upwelling flow. The flow type and biotope classification system are likely to continue to evolve as evidenced by the recent international debate about *chute* and *glide*.

5. BIOTOPE MAPPING: THE DYNAMIC LINK BETWEEN GEOMORPHOLOGY, HYDRAULICS AND HABITAT

Chapter Overview

This chapter introduces a simple technique which provides a rapid, field based assessment of instream habitat across a range of discharges. Biotope mapping may be employed to determine characteristic biotope sequences in space and time for semi-natural river channels (5.1). To enable a gross comparison of biotope sequences in representative channels at a scale appropriate for reach-scale mapping, biotopes are classified at the transect scale and simple indices of biotope 'patchiness' and 'diversity' calculated (5.2). These indices are related spatially to channel type and temporally to flow exceedence (5.3).

Biotope patchiness and diversity indices for a channel length standardised by channel width are used to suggest broad flow exceedence percentiles which bring about maximum biotope patchiness. The effects of floods on biotope types and patches are discussed in the context of 'patch dynamics' theory (5.4). Biotope heterogeneity at a smaller spatial scale (within-transect) is examined at a range of discharges to add an ecological dimension and as a contribution to studies of marginal and refugia habitats (5.5).

5.1 INTRODUCTION

An inherent problem when attempting to integrate geomorphological and ecological studies is their different scales of conceptualisation and operation. Ecologists tend to adopt a 'bottom-up' approach from microhabitat variables within 'patches' through mesoscale riffle-pool units and whole drainage basins in the context of the River Continuum Concept. Geomorphologists operate at larger scales, placing morphological units within their planform, valley floor and hillside structure. Fisher (1994) makes reference to the slow progress made by ecologists in presenting their knowledge of freshwater systems at the reach and basin scale:

> *"Aquatic systems are often presented in landscape maps as opaque homogeneous patches. Aquatic scientists have also been slow to borrow the tools and concepts of landscape ecology to resolve these patches".*
>
> Fisher (1994, p587)

The identification of biotopes as discrete hydraulic units has been verified in section 4.3. The role of the biotope as the smallest scale unit to which biotic assemblages are adapted, i.e. the 'functional habitat' (Harper *et al.*, 1992) has been suggested (section 1.2). Townsend and Hildrew (1994) describe the spatial and temporal aspects of habitat heterogeneity as a basis for predicting species traits within a particular habitat type. At a particular site the hydraulic conditions are relatively homogeneous within units of the same biotope type; habitat heterogeneity is produced by the presence of different biotope types. Empirical studies confirm that habitat and biotic diversity are correlated; Wesche (1985) shows a positive association with habitat and fish diversity. At a more general ecological level, the same notion is held:

> *"...stream communities are largely determined by the organization, structure and dynamics of the physical stream habitat"*
>
> Frissell *et al.* (1986, p199)

Bovee (1996) states that the *mosaic* of different habitat types may be of greater importance in some communities than the abundance of any single habitat type. From a management perspective the diversity of physical habitat may be used an indicator of conservation value (Blyth, 1983; Raven *et al.*, 1996). Li and Reynolds (1994) recognise five components of spatial heterogeneity that may be linked to functional ecological responses:

1. Number of patch types
2. Proportion of each type
3. Spatial arrangement
4. Patch shape
5. Contrast between neighbouring patches

in Bovee (1996, p152)

These concepts are borrowed from field ecology; in the case of the fluvial environment points 1-3 are likely to have most relevance and are the focus of section 5.2. The notion of patch shape may be applied in a vertical dimension as depth may be a limiting factor for fish e.g. in terms of resting sites or migration over riffles. Generally, however, hydraulic conditions have more relevance to instream biota. Patch shape will be discussed simply in terms of wetted area (see section 5.5). Contrast between patches is less apparent in the dynamic fluvial environment than in terrestrial ecosystems where environmental variables may vary over small and localised areas. Biotope sequences at some discharges (particularly low flows) show some parallels with this concept, and are discussed in section 5.3.

Bovee (1996) describes the application of two dimensional habitat models which are capable of modelling patterns of habitat availability and use at the mesohabitat scale. At present many obstacles exist to the widespread use of such models, even in the US where river managers regularly employ PHABSIM to determine minimum flow requirements. Biotope mapping essentially provides a low-cost alternative to two dimensional-models, which may be extrapolated to reaches and possibly catchments by the application of geomorphological models (Rowntree, 1996b). The use of geomorphological models is developed further in section 7.4.

The core objective of biotope mapping is to map and quantify the spatial and temporal distribution of biotopes over a range of channel types and discharges. Hydraulic characterisation of biotopes as described in section 6.3 would allow species' preference or tolerance ranges to be inferred, based on the existing literature, PHABSIM habitat suitability curves or empirical data describing associated biotic assemblages. Knowledge of flow exceedence percentiles at which biotope sequences change would allow management of flows to maintain critical biotopes (King *et al.*, 1989) or target sequences. Biotope mapping is also seen as a potential contribution to 'patch dynamics' theory in which patch heterogeneity and recovery following disturbance are considered to be major characteristics of freshwater ecosystems. The role of floods in patch dynamics is discussed in section 5.4.

5.2 BIOTOPE 'PATCHES' AND 'DIVERSITY': LINKS TO CHANNEL TYPE

Hogan and Church (1989) note that the proportion of the channel represented by each habitat changes with flow. Brookes (1995a) states that there is a need to record *sequential* channel changes over a range of discharges. The biotope sequence represents the *characteristic pattern of hydraulic habitat units which repeats in a reach of a particular channel type, and is dynamic with respect to discharge.* Biotope mapping was carried out to establish how the extent of a particular biotope and sequences alter with fluctuations in discharge. An example of biotope change with discharge is illustrated in Appendix A5. As described in section 3.5.1 biotopes were initially identified at the transect scale, on the basis of the dominant flow type. This level of detail is consistent with River Habitat Surveys (RHS) which also record information at the transect scale. RHS is a national inventory of river habitat which includes, amongst other factors, instream physical features, substrate and flow type (National Rivers Authority, 1995; 1996a). Variables are recorded by the dominant type across a metre-wide transect as data analysis requires input in the form of *one class* only for each category. Mapping at the transect scale gives an indication of the spatial biotope sequence characteristic of a particular channel type and its temporal sequence over a range of discharges. It provides an appropriate scale for rapid habitat mapping at the reach scale.

Transect level biotope mapping was carried out by recording the dominant biotope type at each transect, at those discharges when full hydraulic calibrations were made, and also during and following hydrological events likely to bring about a change in the biotope sequence. These included both flood events and periods of prolonged extreme drought, which occurred in January and February 1995 and July to August 1995 respectively (Institute of Hydrology, 1995). This included flows ranging from Q_{99} to $Q_{0.5}$, as indicated in Table 5.1. The 'high' flow percentile relates to the highest flow observed *excluding the floods of January and February 1995.* These floods range in occurrence time between 35 and 200 years (see Table 3.3), so are considered to be outside the flow regime for a 'normal' year. A 30 cumecs flow released down the North Tyne specifically for a canoe rally in November 1995 is also excluded from Table 5.1, although the biotope sequence at this flow is presented. Flow exceedence percentiles for the Till and West Allen sites are not precise values, as they were calculated from historical flow duration curves (the National Rivers Authority has ceased to keep a continuous flow record at this site). Problems of estimating flow percentiles from historical data are discussed in section 7.3.1.

Site	Minimum flow percentile	Maximum flow percentile	Range
Harwood Beck	100	4	96
Kielder Burn	99	15	84
Wear (Stanhope)	100	3	97
West Allen	95*	2*	93
Lambley	99	18	81
Wear (Wolsingham)	95	16	79
Derwent	82	8	74
Smales	77	11	66
Till	96*	6*	90
Ouseburn	100	0.3	100
Skerne	76	3	73

Table 5.1: Range of flows (to nearest percentile) observed at study sites during biotope mapping (* estimated from discharge calculated at the site and historical flow duration curves).

Biotope sequences were recorded at the transect level on a standard form (see Appendix A4). A photograph was taken at each transect to allow standardised identification of biotopes relative to other discharges and sites. Any transition in biotope type was photographed and its location measured as distance upstream of a known transect. Biotopes were classified according to the dominant flow type at a transect, as illustrated in Figure 3.2. Comparisons of *transect-level* biotope sequences at several discharges are summarised in Figures 5.1 (p134-137). At the majority of sites the biotope sequence associated with one complete sequence of morphological units is presented. Exceptions are the Ouseburn site at which a channelized section was mapped immediately upstream of a natural morphological unit sequence. At the Till site two sequences were selected, one in a straight reach, another located on a bend 500m downstream of the straight section. Thus two biotope maps are illustrated for the Till site. At the South Tyne site a *deadwater* associated with backwater is not illustrated in Figure 5.1d, but this biotope is included in calculations of **biotope 'patches' and 'diversity' indices** (described below). For all sites lengths are given as percentages although actual lengths of biotope sequences at the study sites vary from 65m (the West Allen site) to 2km (the Skerne site).

The biotope maps presented represent the full range of flows which occurred during the study period, with the exception of the flood flows in Table 3.3, as discharges associated with flood conditions (and therefore biotope sequences) fluctuated over intervals as short as twenty minutes. Biotope sequences associated with seasonal flow fluctuations during a 'normal' annual flow regime are therefore denoted. Differences exist both in the number of biotope units and range of biotope types associated with one sequence. In order to compare sites representative of different channel types, two indices may be defined which are likely to have different ecological implications:

1. **Biotope 'patches':** the number of biotope units for a given channel length, regardless of their type.
2. **Biotope 'diversity':** the number of biotope *units* for a given channel length, multiplied by the number of biotope *types*.

Biotope 'patches' assumes spatial heterogeneity *per se* provides habitat for biotic diversity, whereas biotope 'diversity' recognises the importance of different biotope types in the provision of different ecological niches. To allow a comparison between sites having different widths (and catchment areas), the number of biotope units was calculated for a standard length of 10 channel widths. The distinction between biotope patches and diversity is made by a simple multiplication. Biotope patches is the *number of units* which are present over a standard channel length (in this case 10 channel widths). The number of biotope patches per site is calculated from Figure 5.1 (p134-137). The number of biotope patches is multiplied by the number of *different biotope types* to give an index of biotope diversity. Those biotopes which are mapped as having isolated exposed boulders are not treated as discrete biotope types, as exposed boulders simply represent a reduction in biotope area without producing different hydraulic conditions (at the mesoscale). Boulders may represent a different habitat in detail; Chutter (1970) made a distinction between *"stones in current"* and *"stones out of current."* However in this study only the hydraulic habitat components are considered.

Biotope patches is summarised in Figure 5.2, as number of biotopes per 10 channel widths. This is simply a means of comparing the *number of changes of biotope* at a particular site, over a range of discharges. The following list represents the study sites in terms of decreasing patches:

1. Unconfined bedrock and boulder channels
2. Steep, headwater boulder dominated channels
3. Mid gradient, wandering cobble-bed channels
4. Fine gravel, actively meandering channels
5. Mid gradient, stable cobble-bed channels
6. Regulated mid gradient, cobble-bed channels
7. Confined bedrock channels
8. Low gradient, engineered channels

This list provides a broad overview of biotope patches across the range of channel types which are representative of the nine river segment types determined from RHS (Fox *et al.*, 1996). In the context of catchment scale habitat hydraulics the following points regarding biotope 'patches' apply to the study sites:

- The high number of biotopes in unconfined bedrock channels identifies them as possible 'critical' reaches within alluvial channel networks.
- 'Patches' is greatest in channels of low stream order with high substrate heterogeneity, which is consistent with the basin scale 'habitat hydraulics' theory (Statzner and Higler, 1986).
- Mid gradient, cobble channels have intermediate numbers of biotopes; their 'patches' is influenced by local channel morphology.
- Fewer biotope sequences are present in regulated channels due to augmentation of natural low flows which drowns out some morphological features.
- Low biotope patches in confined bedrock channels suggests these as possible 'bottleneck' reaches.
- Engineered channels have low numbers of biotopes, being dominated by glides as the result of removal of some morphological features (dredging, regrading).

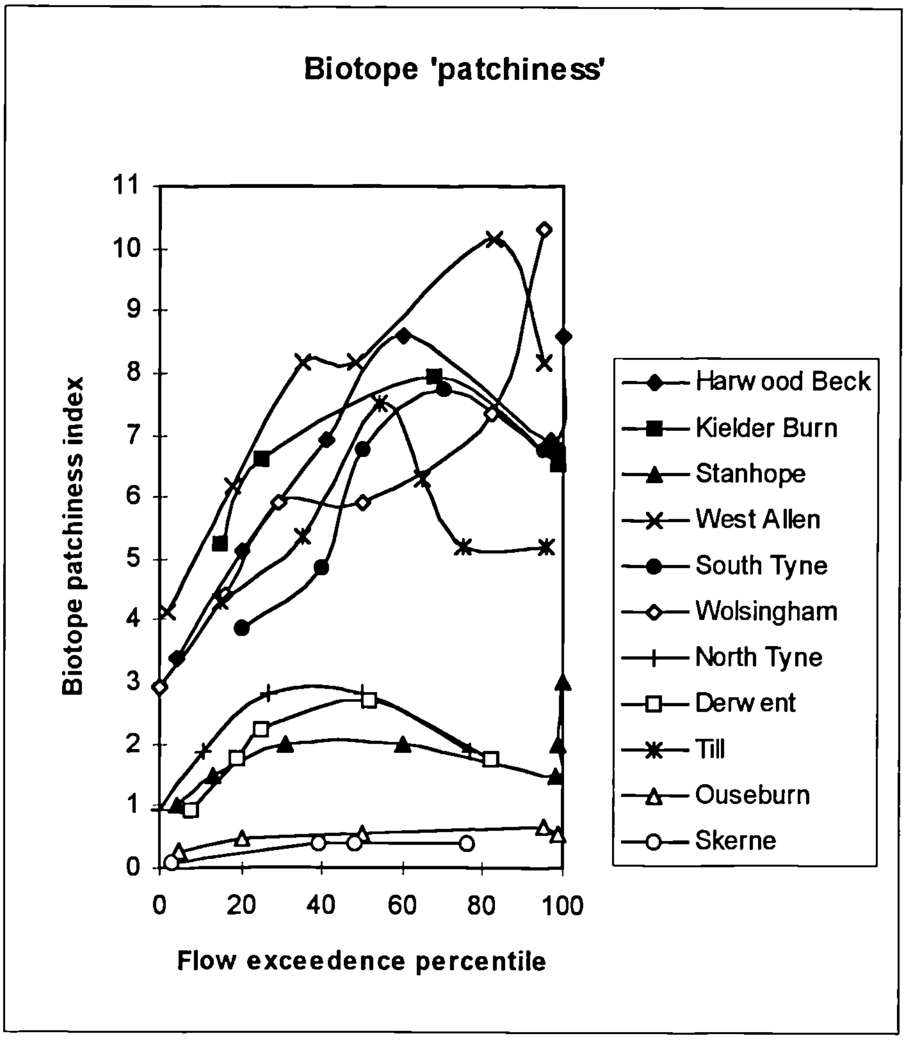


Figure 5.2: Biotope 'patches' index (calculated for a standard length of 10 channel widths).

Biotope patches was greatest in the faster flowing, headwater channels where there is a greater degree of flow variation, and thus substrate variation. This is consistent with observations by Fozzard *et al.*, (1994) in Scottish rivers. A general rule is that channel with more flow variation will contain more biotopes, as a result of substrate and flow heterogeneity. As catchment area (or mean discharge) increases, biotope patches is reduced for a standard channel length (Fozzard, *et al.*, 1994), but this needs to be scaled by channel width to permit legitimate comparisons to be made. The wandering channel at the Lambley site has high biotope patches as it is located in an unstable, depositional zone characterised by a wandering main channel with adjoining backwaters. Mid gradient cobble-bed channels have intermediate biotope patches which is a function of lower substrate heterogeneity and a more stable flow regime. This is consistent with studies of Atlantic salmon spawning habitat in the Nouvelle River, Quebec; the "overwidened braid-like reach" in a wandering channel having considerably more habitat area than a single thread, sinuous reach (Lapointe and Payne, 1996).

Trends in biotic diversity have been related to stream order or catchment area for natural stream ecosystems in the RCC (Statzner and Higler, 1986). The National River Authority's river segment typology (Raven *et al.*, 1996) attempts to link physical habitat types, distribution and diversity to segment types. However, biotope patches may also differ from its expected value based on river type. The regulated sites are associated with lower biotope patches, due to the reduced flow range and lower morphological activity which has produced a more uniform channel in terms of both cross-section and substrates. Low gradient channels have the lowest biotope patches across all flows. This is partly a natural phenomenon, as these channels have finer, more uniform substrates and a more stable flow regime. However channelization for flood defence maintenance has removed most of the natural variation in these channels. The Skerne and Ouseburn sites exemplify this point; removal of the natural biotope sequence by channel deepening and straightening has significantly reduced biotope patches. Links between biotope patches and channel type are being developed as part of the Habitat Quality Index (HQI) proposed by the Environment Agency (Raven *et al.*, 1996). Sites with low biotope patches due to channelization may be used to identify and prioritise potential sites for river restoration or enhancement schemes.

Figure 5.3 illustrates biotope 'diversity', using the above index. Noting the different scales of the two axes in Figures 5.2 and 5.3 it is clear that a greater difference in 'diversity' exists between sites than in 'patches'. This is a potential limitation of the existing River Segment Typology and proposed Habitat Quality Index (HQI) (Newson, personal communication); these systems classify rivers according to the average number of features present in a standard length of 500m (*not* standardised by channel width). Rivers of a particular type with fewer features than average will score lower than those with average or greater than average. This does not take into account the relative habitat quality of different biotope types or the biotope mosaic (Bovee, 1996), which are considered to be an essential component of habitat heterogeneity (Li and Reynolds, 1994). Inclusion of biotope types i.e. biotope diversity is likely to be imperative to assessments of conservation value. The South Tyne site exemplifies this point; it is ranked higher for 'diversity' than 'patches' due to the number of different biotope types associated with a depositional zone of a wandering gravel-bed channel. However, the deep glide downstream of the mid channel bar at this site extends for approximately 250m which leads to fewer biotope units over a width-standardised channel length, despite the fact that biotope 'diversity' is high.

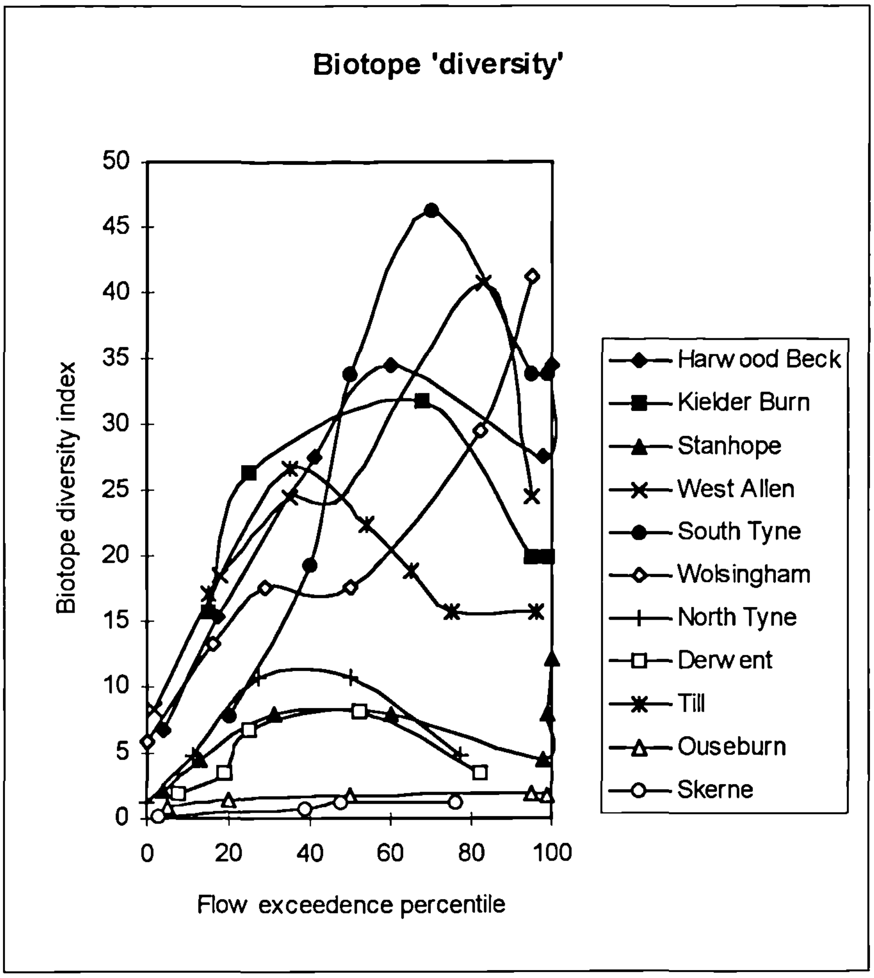


Figure 5.3: Biotope 'diversity' (calculated for a standard length of 10 channel widths).

Site specific features alter biotope diversity, which may be natural or artificial. Biotope diversity at the Ouseburn site (Figure 5.3) must be interpreted with caution as its value represents the average number of biotopes from the natural and channelized sections. Figure 5.4 compares biotope diversity at the natural and channelized sections of the Ouseburn site. Figure 5.5 compares biotope diversity at the straight and curved sections of the Till. The point bar located at the bend produced a shift in flow direction and is associated with an additional *boil* within the *riffle-glide* sequence. This results in several biotope units over a distance of just 30m. The Till results are extrapolated over 100m as illustrated in Figure 5.5.

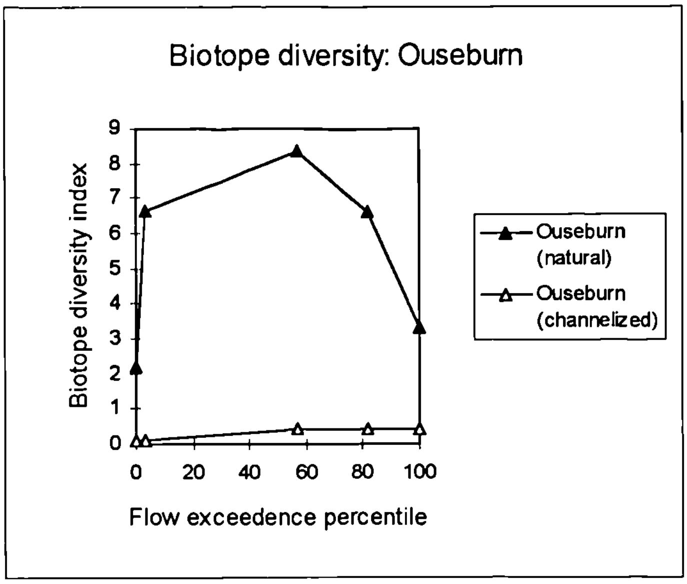


Figure 5.4: Comparison of biotope diversity in a natural and channelized section of the Ouseburn.

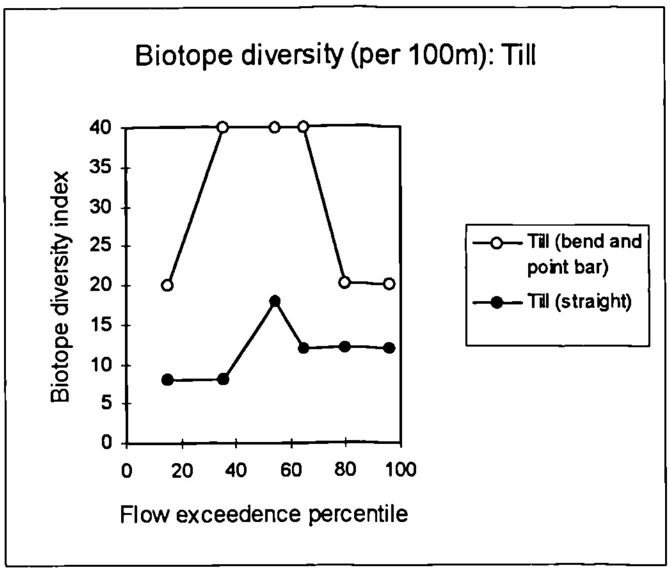


Figure 5.5: Comparison of biotope diversity in a straight section and bend of the Till.

If biotope types are to be incorporated into a Habitat Quality Index in addition to absolute numbers of biotope units, large differences are evident between different representative river types. This does not imply that headwater channels (e.g. the Harwood and Kielder sites) with higher 'diversity' scores in the HQI will have a higher conservation value than low gradient channels, as comparisons will only be made between channels of a particular type. Additionally, it is clear that absolute biotope numbers or 'patches' does not necessarily define an ecologically diverse site or one of high conservation value. At the Ouseburn site (and many other urban watercourses) water quality is the overriding influence on 'habitat potential', despite the relatively high biotope 'patches' in non-engineered sections of channel.

The work presented above needs extending within the context of the Environment Agency's proposed Habitat Quality Index (Raven *et al.*, 1996). Rivers of a particular type are to be scored and ranked on the basis of 'Characteristic' and 'Intrinsic' features. A weighting system is being developed which considers the relative importance of various features. The fundamental question is whether biotope 'diversity' is more important to biotic communities than 'patches', and whether all biotope types should be equally weighted. This can only be addressed by research to determine 'functional habitats' across a range of channel types. It may be that biotope 'patches' is more important in very irregular headwater reaches where boundaries between biotope types are less easy to identify. In these steep,rough channels biotope mapping at the transect scale is impractical and is likely to have little ecological value (King, personal communication). By contrast surveyed biotope 'diversity' or 'critical' biotopes may be more important to ecosystem functioning in 'mid' and 'lower' reaches, where biotopes spanning the channel width are clearly identified and easier to map at the transect scale. The future development of this potential aid to habitat quality assessment will be discussed further in sections 7.1.1 and 7.3.2.

5.3 BIOTOPE CHANGE WITH DISCHARGE VARIATIONS

5.3.1 Response of individual biotopes

The response of individual biotopes to discharge offers a potential means of predicting which biotopes will be present at a given site under various flows. In a sense it is the qualitative equivalent of the backwater hydraulic modelling in PHABSIM. An initial aim of biotope mapping was to describe how biotopes respond to changes in discharge, and if possible, calculate discharges at which the biotope sequence alters. In practice, it has not been possible to determine precise flow percentiles, partly as it was not possible to map sites at all discharges and partly due to the channel change which occurred during large floods, which altered the post-flood biotope sequence at equivalent discharges (see section 5.4). Instead, a generalised biotope response to increased discharge has been established from the field observations. Figure 5.6 does not indicate the actual flow percentiles at which biotopes change, but illustrates changes which take place when increases in discharge are sufficient to cause a change in biotope. This information may be applied to sites with similar biotopes to predict how biotope sequences will change with discharge. A site visit is necessary to map the low flow biotope sequence and its morphological units (which act as hydraulic controls); from this biotope sequences at higher discharges may be inferred.

Morphological units which are the hydraulic controls on dynamic biotope sequences maybe classified into depositional and erosional units. This distinction was first made by Moon (1939) in order to explain differences in invertebrate distributions. Depositional and erosional units broadly correspond with Rowntree's (1996c) classification of five morphological units (riffle, step-pool, cascade, pool and planar bed) into two groups 'riffles' and 'pools', representing topographical high and low points in the channel's long-profile. In this study, a distinction is made between boulder and bedrock channels which have high substrate roughness producing cascade and rapid biotopes, and cobble/gravel bed channels. The influence of substrate on hydraulic conditions and biotopes is dependent upon substrate size and type of morphological unit. In ***depositional units*** (pools and planar beds) relative roughness decreases as flow increases in boulder and cobble bed channels, as particle roughness becomes relatively less significant compared with form roughness i.e. roughness produced by the overall long profile and cross-sectional channel morphology (Maizels, 1984). This is illustrated at the Kielder site; the low flow (Q_{95}) *run* becoming in part *glide* where flow increases sufficiently to drown out the roughness produced by the cobble substrate. At very low flows the planar bed unit is entirely *run*. By comparison, in gravel and sand bed channels, substrate roughness has less influence on hydraulics than channel form and topography, even at very low flows, so *deadwaters* are common. Small increases in discharge result in a shift from *deadwater* to *glide*. Further increases in discharge result in *glides* becoming *runs*. The flow exceedence percentile at which this occurs appears to be related to channel gradient, water depth and substrate size. In general the threshold flows for the transition from *glides* to *runs* are higher in deep, low gradient glides with finer substrate. Shallower *glides* over relatively steeper gradients and coarser substrate tend to become *runs* at lower flows (higher flow exceedence percentiles), as substrate has a greater influence on flow.

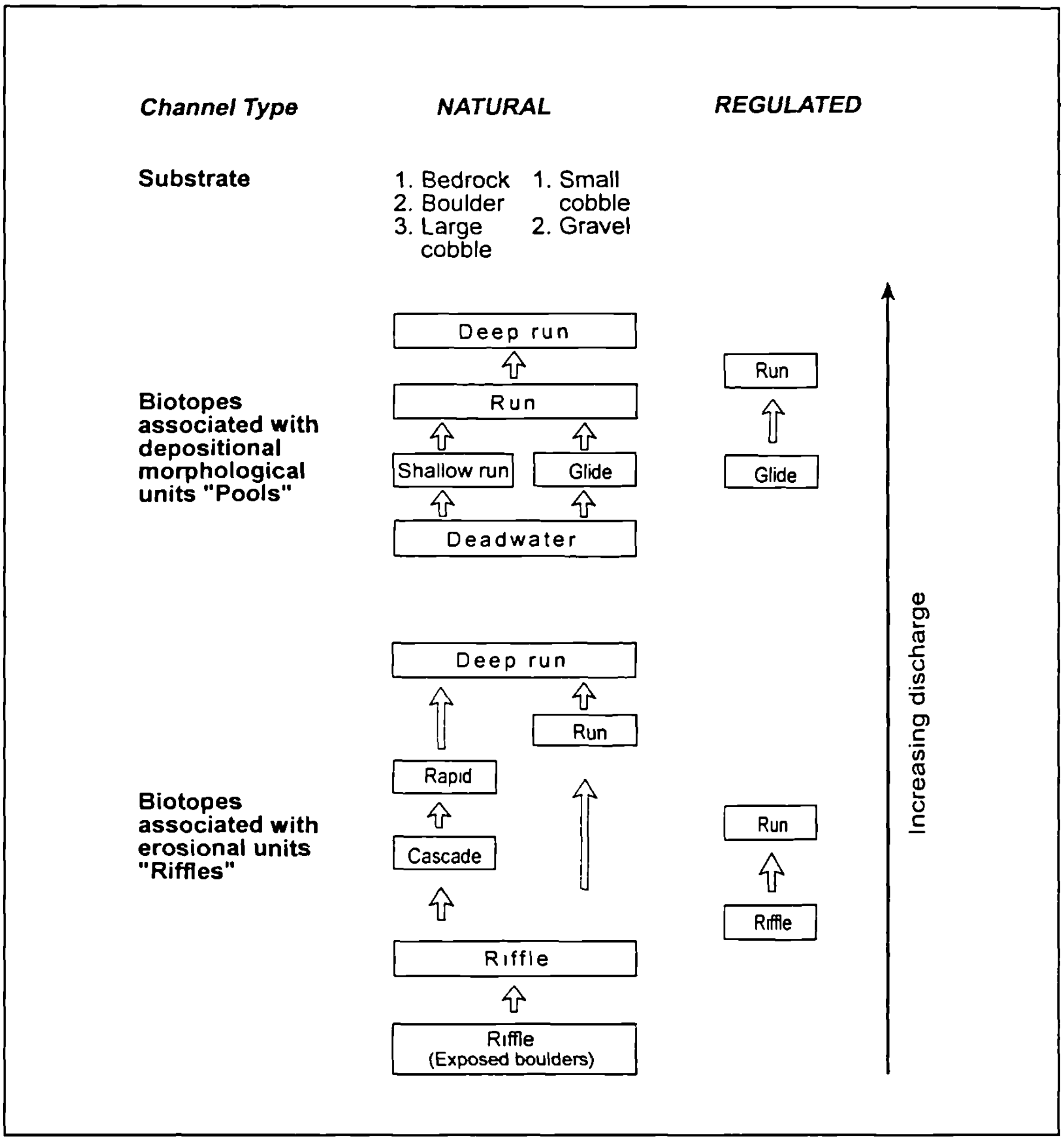


Figure 5.6: Generalised biotope response to discharge.

In ***erosional*** morphological units with cobble or boulder bed (*cascades*) substrate roughness increases with initial increases in discharge. This is illustrated by biotope sequences at Harwood Beck and Kielder Burn. Flow over *cascades* at very low discharges is not sufficient to produce hydraulic drops (chutes) or jumps (broken standing waves) around exposed boulders, and in drought flows deadwaters may become isolated from areas of *shallow run*. With sufficient flow to fill the channel but not submerge boulders, cobbles in the wake of boulders and between clusters (Brayshaw, 1985; Brayshaw *et al.*, 1983) form the major roughness element, and the biotope becomes almost entirely *riffle*. As discharge increases slightly boulders have a more significant influence on flow and a greater proportion of the unit becomes a *cascade*. With further increases in flow boulders are just submerged and the entire unit functions as a cascade. Threshold flows appear to be related to width:depth ratio, channel morphology and gradient. At the Harwood Beck site *riffles* become *cascades* at higher flows (lower flow exceedence percentiles) as the channel has a larger width:depth ratio and increases in discharges therefore result in greater increases in wetted width than depth or velocity (see section 5.4).

Extending the notion of gradient and width:depth ratios to the bedrock controlled sites, it appears that the steeper gradient *cascades* at these sites results in chute flow and broken standing waves even at very low flows. In the less confined West Allen site part of the unit is *riffle* at the lowest flow percentiles, whereas the larger substrate size and steeper gradient at the Wear (Stanhope) produce chute flow associated with *cascades* even at very low flows. As discharge increases further in cascades, this biotope becomes a *rapid*, as the influence of individual roughness elements changes from wake interference to skimming flow (Davis and Barmuta, 1987; Young, 1996). The *rapid* represents an additional biotope not present in erosional morphological units of smaller substrate size (cobble and gravel). In channels with smaller substrate *riffles* over cobble and gravel become *runs* at higher discharges. The boundaries between *runs* and *rapids* become difficult to determine at very high flows, producing transitional *run-rapids*. The formation of *run-rapids* at high flows is dependent upon both local water depth, surface slope and the dimensions of roughness elements. Carling (1992) notes that instream hydraulics reflect channel morphology and bed form at 'low to moderate" flows. Above a certain flow form roughness will no longer be influencing hydraulics, resulting in a *run-rapid* biotope which dominates the reach.

The influence of substrate and channel topography in erosional morphological units of small cobble and gravel sized rivers decreases as flow increases. The unbroken standing waves of *riffles* are produced directly by the interaction of flow and individual roughness elements; as flows increase this hydraulic control is drowned out and riffles become *runs*. In regulated rivers the exceedence percentile at which *glides* and *riffles* become *runs* is lower, i.e. flows are proportionally higher. This may be related to armouring of the bed of North Tyne; an armoured bed being associated with lower relative roughness due to a more uniform roughness height. It is only when water depth increases to cause sufficient turbulence and momentum of the water itself that this regulated *glide* becomes a *run*. Alternatively channel gradient may influence the discharge required for the transition from glide to run, with steeper channels causing sufficient momentum of flow to produce runs at lower discharges. Gradients for the study sites are summarised in Table 3.1.

As flows increase, Moon's separation of erosional and depositional units becomes less distinct and turbulence extends downstream into the pool or planar bed unit. The high flow *rapid* increases in area proportional to the biotope associated with the pool or planar bed, which, at high flows is a *run*. This generalised response occurs in all channels with substrate ranging from large gravel to boulder. Where smooth bedrock outcrops are present roughness is reduced at low flows and small scale *runs* occur within the *cascade*. However, at flows above drought conditions the region of locally reduced turbulence is lost due to the influence of adjacent chute flow and broken standing waves. At high flows the *cascade* becomes a *rapid* when water is sufficiently deep to 'skim' the obstacle created by the substrate (Davis and Barmuta, 1987).

In fine gravel channels the bed topography (Maizels *et al.*, 1984) provides the main roughness factor. At the River Till (bend) the point bar is the major hydraulic control, creating a riffle where flow converges when reaching the bar. As flows increase this creates secondary flows downstream of the riffle, to produce a biotope dominated by upwelling flow, or a *boil*. This flow type is characteristic of meander bends, at the downstream end of depositional features or at the transition between biotopes where flow converges. As water depth increases over a point bar there is sufficient momentum for the downstream component of flow to dominate, producing upwelling within a biotope which is predominantly *glide*. Upstream of the bar there is sufficient water depth and disturbance of flow from the bar to produce a *run*. The upstream section of the River Till site is straight with a more even long profile and cross section, so hydraulics are influenced to a greater extent by substrate roughness. At very low flows much of the gravel was exposed, with possible implications for biotic communities. These are discussed in section 7.3.1.

The impact of low flows on habitat availability has been the subject of several studies (Orth and Leonard, 1990; Tharme, 1996; Gibbins 1996), with PHABSIM often being employed to determine flows below which significant reductions in habitat area will occur (Maddock, 1992; Petts and Maddock, 1994). Reductions in wetted width have been used as a rapid assessment technique (Maddock, 1994; Gippel and Stewardson, 1996), but these give no indication of the biotopes present at very low flows. If depth is insufficient to provide cover or allow passage of fish over riffles, that area will be unsuitable habitat despite being within the wetted perimeter. Gippel and Stewardson (1996) relate width reductions to hypothetical cross-sectional channel profiles; in this study empirical evidence of biotope change and width reductions is presented (Figure 5.7). Sites which showed significant width reductions at low flows all had point bars, as these topographical high points are the first areas to become 'dried out' at low flows. As point bars are recorded in RHS and their frequency predicted for a particular channel type, this database may be used in conjunction with the river segment typology to predict sites which are likely to be sensitive to low flows.

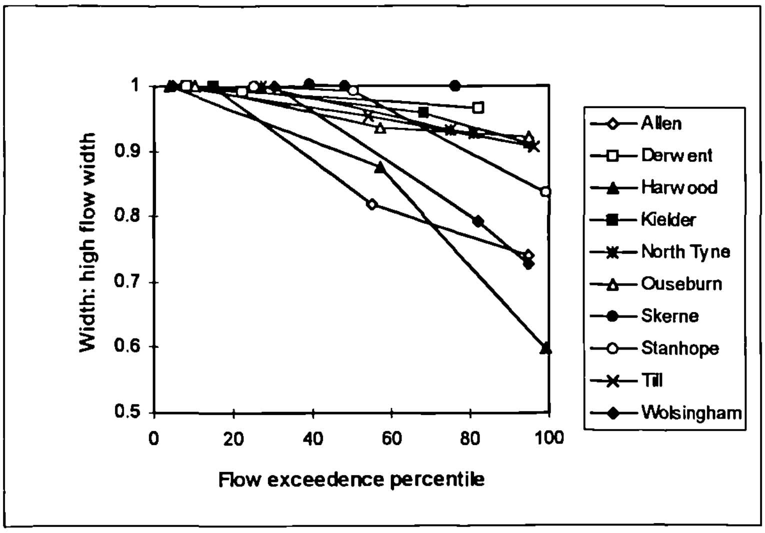


Figure 5.7: Reductions in wetted width with decreasing discharge (as a percentage of high flow wetted width).

Biotope mapping was carried out at all study sites during August 1995, which represented the most intense drought period of that summer (Institute of Hydrology, 1995). A generalised biotope response to drought flow is presented in Figure 5.8. All sites retained marginal deadwaters, although those associated with the steep headwater channels with irregular cross-sectional profiles (Harwood Beck and Kielder Burn) became isolated from the main channel. From a flow management perspective the identification of isolated or dewatered biotopes provides an indication of ecosystem stress, and could be used to determine periods when a drought flow regime should be imposed. This is the basis of the Building Block Methodology being developed for South African rivers (King and Tharme, 1993; King and Louw, in press). To allow practical flow management at the catchment scale critical reaches which are sensitive to low flow must be identified, which requires biotope mapping at the catchment scale within the theoretical framework of hierarchial, geomorphological models. This development is discussed in section 7.4.

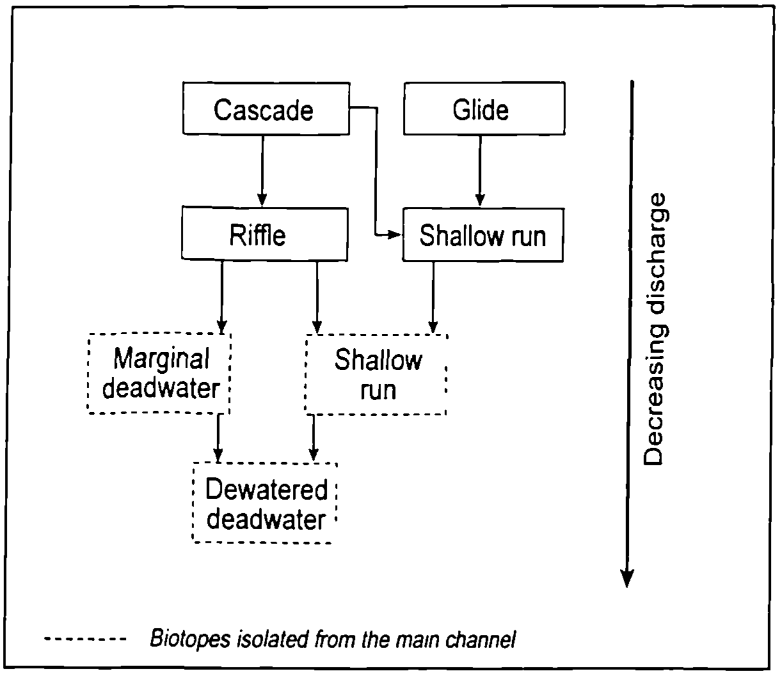


Figure 5.8: Response of biotopes to low flows (dashed boxes indicate biotopes which have become isolated from the main channel flow by deposits of coarse bed material).

5.3.2 Biotope sequences: flow exceedence percentiles for biotope diversity

Biotope mapping of sequences at different discharges has also been used to determine flow exceedence percentiles which bestow maximum biotope patches. In the absence of detailed ecological studies of 'functional' or 'critical' biotopes across a range of channel types, maintenance of maximum biotope patches would provide 'insurance' for a range of species, habitats and ecological functions. Swales and Harris (1995) state that habitat diversity is maximum at intermediate flows and decreases as flows approach extremes. This guidelines refer to flow evaluation for fish habitat (Bain *et al.*, 1988; Leonard and Orth, 1988). Table 5.2 indicates the flow percentile at which biotope 'patches' and 'diversity' are maximum. Values are taken from Figures 5.2 and 5.5 to the nearest 5th percentile. These are not to be interpreted as a single and fixed prescribed flow; it is acknowledged that a variable flow which mimics a natural regime is most beneficial to biota (King and Louw, in press). Indeed it may be via flow variability that biotope diversity is ecologically important, with different biotopes being maintained at key life-stage events. Instead, these values allow a flow regime to be manipulated around exceedence percentiles which create maximum biotope diversity, and which differ by channel type. A detailed discussion is given in Padmore (in press).

Site	Flow percentile giving maximum biotope 'diversity'	Flow percentile range giving moderate-high biotope 'diversity'
Harwood Beck	100* (55)	45-90 (45%)
Wear (Stanhope)	100* (50)	30-70 (40%)
West Allen	90* (55)	55-90 (35%)
Wear (Wolsingham)	(80)	70-90 (20%)
Ouseburn	95 ^{NH}	90-95 (5%)
South Tyne	70	45-95 (50%)
Kielder Burn	70	30-80 (50%)
Skerne	40-80 ^{V}	40-80 (40%)
Derwent	50	40-60 (20%)
North Tyne	40	25-55 (30%)
Till	30	25-45 (20%)

Table 5.2: Flow percentile at which biotope 'patchiness' and 'diversity' are maximum (* indicates a post-flood biotope map and flow; pre-flood flow exceedence percentiles bestowing maximum biotope diversity are given in brackets. ^{V} indicates biotope diversity resulting from instream vegetation. ^{NH} refers to non-hydraulic influences on biotope diversity e.g. tree-roots or in-channel domestic waste).

Excluding the post-flood biotope sequences the majority of sites have maximum biotope diversity at low to moderate flows, which is consistent with the findings of Swales and Harris (1995). At these discharges the influence of channel morphology on hydraulics is greatest which results in high biotope diversity in most semi-natural channel types. In general, at flows above Q_{50} hydraulic controls are 'drowned out' and boundaries between low flow biotopes become less distinctive. The overall result is reduced biotope patches and diversity for a given section of channel. Exceptions are the Till, North Tyne and Derwent sites, which exhibit maximum biotope diversity at higher flows (lower flow exceedence percentiles). At the regulated North Tyne site, power turbine operation constraints at Kielder Dam produce compensation flows and minimum maintained flows in excess of the natural flow regime even when the river is not regulated to support abstraction (Archer, 1993). Thus an elevated, non-natural regime is generally in operation and natural peak flows occur less frequently. Low flows are rarely allowed to drop to natural summer levels, thus the increased 'patches' experienced with low flows at natural sites does not occur. Maximum biotope diversity at the Tyne site was associated with the lowest flow observed, but these represent higher exceedence percentiles in a regulated river. The low flows cited in Table 5.1 refer to discharges experienced during the summer drought, 1995. Although 1995 was an exceptionally dry summer (Institute of Hydrology, 1995), the flows experienced at the North Tyne site were augmented to maintain minimum flows. It is interesting to consider the flow variation which occurred at the regulated site during the drought of August 1995. Although a reduced range of flows are experienced annually rapid fluctuations in flow occur; on 5 August 1995 the lowest flow recorded was 3.1 $m^3.s^{-1}$, with a corresponding biotope sequence as indicated in Figure 5.1 (North Tyne). This had been reduced from 12.2 $m^3.s^{-1}$ over a period of six hours (Merrix, personal communication). During the 'large' release the biotope sequence would be similar to winter flow (i.e. 15.8 $m^3.s^{-1}$ observed in December 1995). The implications of such rapid fluctuations and unusual seasonal flow regime are discussed in section 7.3.1.

The Till site is dominated by gravel substrate, which produces mainly riffles at low flows. Exposed gravel bars are present in the straight section; although they are not counted as a separate biotope for the purposes of biotope mapping, their probable importance as fish spawning habitat (Baras *et al.*, 1996) or invertebrate habitat within the hyporheic zone (Gibert *et al.*, 1990; Petts and Maddock, 1994) is recognised. Channels with fine gravel substrate require higher flows to increase turbulence and hydraulic/biotope diversity. Additionally upwelling flow occurs at higher flows at this site, particularly at the bend where secondary flow cells are created as flow converges over the point bar. Maximum biotope diversity at the Wear (Wolsingham) and Ouseburn sites occurs at very low flows i.e. Q_{95}, but this is due to the presence of random in-channel obstructions creating local hydraulic and biotope diversity. In the channelized Ouseburn site, domestic waste leads to deposition of sediment and the formation of a riffle around the obstruction. Were this obstruction removed (as occurs during floods) biotope diversity would be maximum at moderate flows. At Wolsingham tree-roots and the remains of a concrete groyne increase local hydraulic diversity to produce a *riffle* upstream of the obstruction and a *marginal deadwater* immediately downstream. The importance of man-made structures in increasing biotope diversity is of particular relevance in engineered channels which would otherwise by hydraulically uniform.

5.3.3 Flow ranges for biotope diversity

Moving away from the concept of fixed flows is the notion of flow ranges within which discharges may be manipulated according to local flow conditions, instream seasonal flow requirements and consumptive users of flow. The flow exceedence range producing moderate to high biotope diversity was estimated from Figure 5.3, by 'eyeballing' the flow exceedence percentiles at which biotope diversity dropped significantly. From a flow management perspective a range of flows bestowing moderate to high biotope diversity is more useful than a single value, as it allows flexibility when setting EAFs. Additionally management for biotope diversity can be achieved with biotope mapping and simple stage recordings to determine levels at which biotope sequences change. Thus it may be employed at sites where no continuous flow records exist.

Those sites with a high degree of substrate and biotope heterogeneity (e.g. the headwater, boulder dominated channels and the wandering gravel-bed channel) maintain biotope diversity over a wider range of flows. By comparison sites with uniform cross section or low spatial heterogeneity (e.g. the channelized and regulated sites) have moderate to high biotope diversity only at a restricted flow range. Naturally low substrate heterogeneity at the Till site means that a high, yet narrow flow range (Q_{40}-Q_{50}) is required to produce maximum biotope diversity. Similar observations are made at the Derwent and North Tyne sites, where biotope diversity is associated with higher flows. In regulated channels restriction of the flow range and the absence of flow extremes results in less annual variation in biotope sequences. Biotope diversity at the North Tyne and Derwent sites is low compared with the remaining natural channels (Figure 5.3). It is possible that armouring of the channel bed has produce a more uniform cross-section and reduced substrate heterogeneity. These findings are consistent with Maddock's (1992) studies of invertebrate habitat using PHABSIM; sites with low physical diversity required a high flow to provide maximum useable habitat. One exception to Maddock's rule is the channelized Ouseburn site which has uniform topography and cross-sectional area, yet highest biotope diversity at Q_{95} due to an obstruction in the channel.

In the channelized Skerne site maximum diversity occurs across a range of flows from Q_{40} to Q_{80}, but this is related more to presence of macrophyte vegetation rather than the interaction of discharge and substrate. At low flows there is some evidence of bed morphology 'mimicking' riffles, but the main factor influencing biotope diversity is macrophyte vegetation. Seasonal growth of *Schoenoplectus lacustris* creates *boils* (upwelling flow) in channels which would otherwise be hydraulically homogeneous *glides* across a wide range of flows. When this seasonal growth declines substrate becomes the dominant influence on hydraulics, and *riffles* formed by sediments trapped amongst the vegetation are present. The influence of submerged vegetation in the regulated Derwent produces a low flow biotope sequence similar to those expected at higher flows. Vegetation has mainly a streamlining influence; decreasing hydraulic roughness and producing *runs* at low flow which would be *riffles* in unvegetated conditions. The precise hydraulic effect of macrophyte vegetation is dependent upon the type of vegetation (emergent or streaming). Further discussion of the effects of vegetation on hydraulic diversity is provided in section 6.2.2.

The effects of low flow on biotope diversity are specific to a particular type of river (Armitage, 1995). Alluvial sites with bedrock and boulders maintain a relatively high biotope diversity even at low (non-drought) discharges i.e. Q_{95}. However these sites are sensitive to drought flows, as it is these channels which experienced significant reductions in wetted width and isolation of deadwaters from the main channel. By comparison mid-gradient, cobble bed channels (regulated and unregulated) and the bedrock controlled site experience reductions in biotope diversity at the very lowest flows. In summary, **river- (or channel type-) specific flow management guidelines are required.** These are discussed in detail in section 7.3.

5.4 FLOOD EFFECTS AND BIOTOPE SEQUENCES

Carling (1995) discusses the role of sediment transport in the adjustment of ecological patch dynamics, stating that flows of at least two thirds bankfull are required to mobilise coarse bed sediment. The flooding which occurred over much of North East England on 31 January 1995 and 22 February 1996 will undoubtedly have resulted in sediment transport and associated changes in microscale substrate distribution and hydraulic conditions. At the mesoscale, the impact of high magnitude floods may be determined by comparing pre-flood biotope maps and photographs with post-flood conditions at similar flows. The following are indicators of flood induced morphological change:

- Change of planform
- Change in location of flow controls i.e. morphological units
- Change in biotope sequence and distribution at a similar discharge
- Change in substrate size distribution

These flood events provide the opportunity to describe the impact on channel morphology from an ecological perspective. Sites were observed in the week following the flood, once the floodwaters had receded and instream morphology was visible. All sites had been visited during the three weeks prior to the flood, in order to map biotopes present under winter baseflow and after snowmelt. Evidence is presented here only for changes in biotope sequences. Flood-induced morphological change is listed in Table 3.1, but detailed results not presented. Morphological change is evident where the biotope sequence (at the transect scale) differs from those mapped previously at the same discharge, or for sites mapped at different exceedence percentiles, the sequence predicted by Figures 5.6 and 5.8. Sites were mapped repeatedly during the summer of 1995, in order to compare pre- and post-flood biotope sequences at similar discharges. Post-flood biotope sequences in Figure 5.1 are indicated with an asterisk.

Biotope mapping at the Wear (Stanhope) site provides the best illustration of the effect of floods on biotope patches and diversity, as this site was mapped at the same flow exceedence percentile. Increased biotope 'patches' and 'diversity' are present in post-flood conditions. In riffle morphological units flood sediments have been deposited within the associated biotope (*cascade* or *riffle*) to raise the bed level and produce patches of 'slower' biotope types. At the Wear (Stanhope), West Allen and South Tyne sites deposition of cobbles within the cascade produces 'patches' of riffle. Initially the flood sediments were poorly sorted which meant that the *riffle* biotopes did not occur in full width biotopes, but localised patches. At the Harwood Beck site step formation resulted in increased 'patches' by the reorganisation of boulders into steps which span the channel. Ponding of flow upstream of 'steps' produces runs between steps; in pre-flood conditions this unit was entirely cascade.

In biotopes associated with pool morphological units, deposition leads to raising of the bed level and channel narrowing, which results in a higher energy biotope for a given discharge as velocity increases to compensate for the reduced area. This is illustrated by pre- and post-flood biotope sequences at the West Allen, Harwood Beck, Wear (Stanhope) and South Tyne sites. In the first two sites a pre-flood *glide* has become a *run* at the same or lower discharges after the flood. At the latter sites infilling of pools with cobble changed the biotope to *glide* at the same, or lower, post-flood flows. At the South Tyne site the main deposition of cobbles has occurred in the large pool downstream of the mid channel bar, raising the bed by 1.2 m and causing the shift from *pool* to *glide*. In the bedrock site i.e. the Wear (Stanhope), the low flow *pool* is reduced in area as the riffle morphological unit (and associated *cascade* biotope) has migrated downstream by approximately 10m. This change is less apparent at higher flows, due partially to the reduced influence of substrate on hydraulics, and to a certain extent attributed to the reworking of bed sediments in smaller scale flood events, which occurred between February 1995 and the subsequent resurveys.

In terms of biotope diversity, there appears to have been a redistribution of existing biotopes rather than a change in the type and proportion of a particular biotope type. The only site which shows an increase in the distribution and *area* of biotopes is the South Tyne site; deposition on a shoal caused a greater degree of ponding upstream which led to an increased area of *deadwater*. The increase in area is associated with a secondary channel which is inactive under the normal discharge range. This site is naturally highly mobile and the position of the main channel switches with relatively low magnitude flood events. A small change in biotope area is therefore not unexpected for a highly mobile site whose morphological units are dynamic, even under a normal range of discharges.

Increased biotope diversity after floods is indicated in Figure 5.3. At the Harwood Beck, West Allen, Wear (Stanhope) and South Tyne sites 'kinks' in the biotope diversity curves indicate increased patches as a result of redistribution of sediments during floods. These are particularly evident at low flows (Q_{95} and less). By comparison, sites not affected by floods have smaller biotope diversity indices at the lowest flows. It appears that the influence of a flood is related not only to the magnitude and frequency of the event (see Table 3.3), but the channel geomorphology. Biotopes at the Harwood Beck site are redistributed by a 35 year event whereas the Wear (Wolsingham) site is not affected by an event of a similar magnitude. This supports claims that the floodplain morphology, sediment characteristics and disturbance regimes reflect both the geomorphological setting and the flow regime (Petts and Amoros, 1996). The increased biotope patches following floods has implications for the 'resetting' of ecological communities. This is discussed in section 7.1.3.

5.5 MAPPING CROSS-CHANNEL BIOTOPE AND FLOW TYPE VARIATION

5.5.1 Dominant and secondary biotopes

In natural channels biotopes occur as a series of spatially and temporally changing 'patches' (White and Pickett, 1985; Naiman *et al.*, 1988; Hildrew and Giller, 1994); their boundaries occur where interactions between morphology and flow result in a local change in hydraulics, rather than in units delimited at intervals perpendicular to the direction of flow. Rabeni and Jacobson (1993) describe how the position of the boundaries between main channel ("run") and marginal ("edgewater") biotopes change laterally with discharge. Patches at a smaller scale than that defined by transect level biotope mapping is ecologically significant:

> *"the fish community characteristic of a segment of stream is determined by the complexity of habitats present in the area, especially the horizontal heterogeneity component."*
>
> Gorman and Karr (1978, p514)

Tharme (1996) also says that adjacent biotopes are important for survival strategies during drought, involving vertical and lateral movement. In order therefore, to more accurately map biotope types and their distribution at a scale relevant to instream biota and patch dynamics, transects which did not have cell level data were recoded by both **dominant and secondary biotopes**. This was done by reference to photographs taken at each transect for all sites and calibration discharges. Additionally, recording dominant and secondary biotopes makes it possible to establish the degree of detail lost by recording information at the transect scale in River Habitat Surveys. Cell level classification as described in section 4.3 is necessary for statistical discrimination between flow types; however for applied biotope mapping as a technique for instream habitat inventory and gross comparisons of habitat hydraulics between sites, such small scale classification is not practical. Where more than one flow type occurs in discrete biotopes organised across the channel, it is useful to map dominant and secondary biotopes.

The proportion of dominant and secondary biotopes was estimated for sites which were initially classified only at the transect level. Each transect was re-coded by its dominant and secondary biotope and the proportion of each plotted in Excel. The channel is represented by a series of adjoining histograms to give a schematic view of each site. The dominant and secondary biotopes are labelled on the y axis, which represents distance in metres upstream. Individual transects are not labelled; only where a change in biotope type occurs is this indicated. Dominant and secondary biotopes at two study sites with contrasting proportions of secondary biotope type (at three calibration discharges) are illustrated in Figure 5.9.

The biotope types which are commonly overlooked by transect level classification are listed in section 3.5.1. The most common biotope not recorded by this method is the *marginal deadwater*, which occurs over a range of discharges at channel margins. Marginal deadwaters are considered of increasing importance for their diversity of biota and role as refugia at high discharges (Townsend and Hildrew, 1994; Milner, 1994). In the River Habitat Survey there is scope for recording marginal deadwaters in the ‘sweep-up’, but only as ‘present’ rather than recording total area or extent. In addition to marginal biotopes, the majority of sites had secondary biotopes present at points in the channel where local depressions in bed topography result in a different flow type (thus biotope) at the micro-scale.

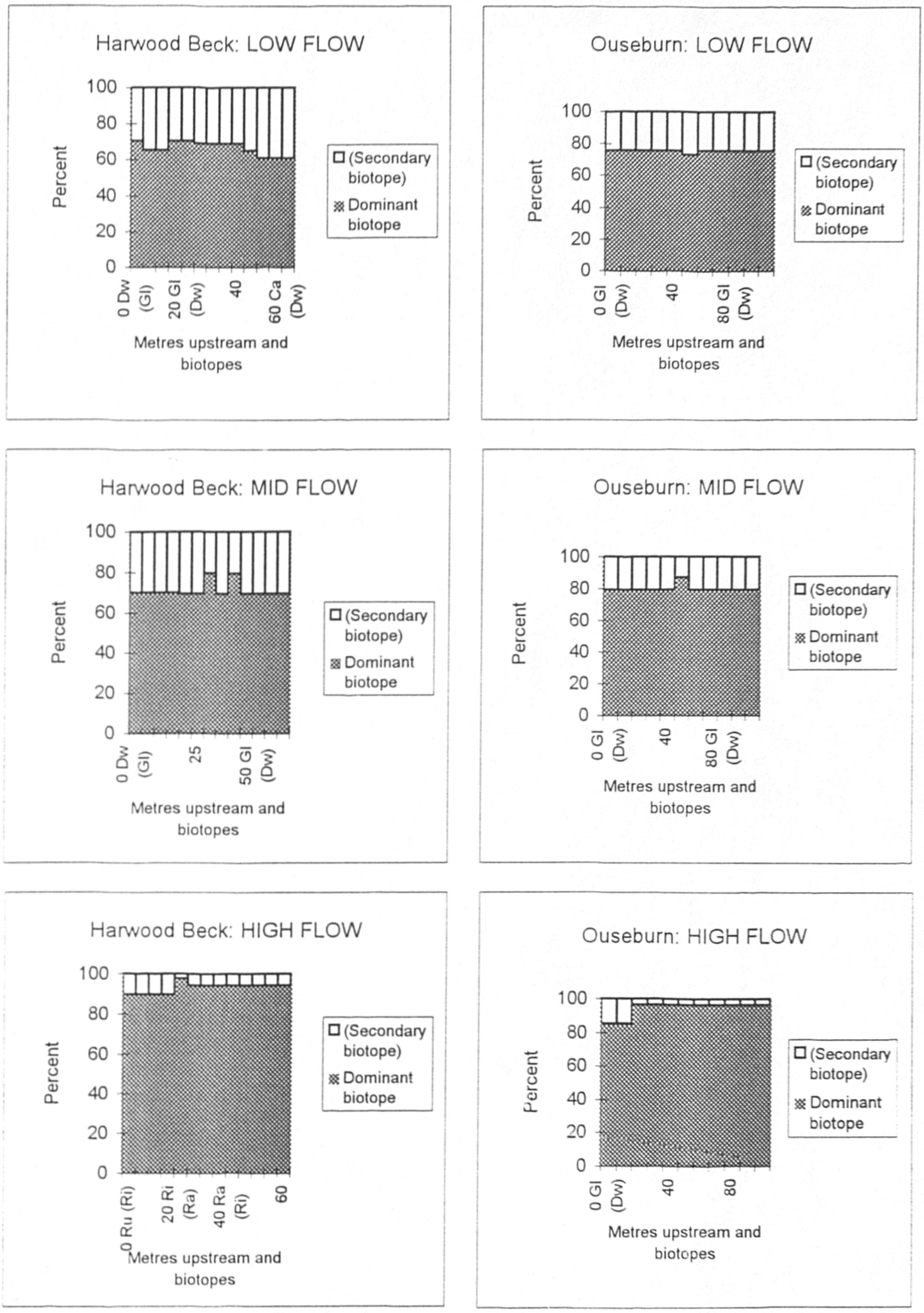


Figure 5.9: Percentages of dominant and secondary biotopes in two study sites at low, mid and high flow calibrations. Ca=Cascade; Ra=Rapid; Ri=Riffle; Ru=Run; Gl=Glide; Dw=Deadwater.

The proportion of the channel width which is dominant biotope varies with site (or channel type) and discharge. In general a greater, and significant proportion of the channel is occupied by secondary biotopes at low flows. Dominant and secondary biotopes correspond with mesohabitats, the borders of which are best defined at low flows (Armitage *et al.*, in press). This has implications for River Habitat Surveys which, at present are carried out during summer low flows and record secondary biotopes in the 'sweep-up' section. The relative proportions of dominant and secondary biotopes and their degree of connectedness may be more important than simply their presence or absence. As Gibbins (1996) states, the combined area of "hydraulic dead zones" in a given reach indicates its potential to act as a refuge area for invertebrates. Lancaster and Hildrew (1993a) show patches with low shear stress to exist even at high discharges. The role of "dead zones" as nurseries and high flow refugia has also been documented for both salmonid (Harvey, 1987; Heggenes, 1988a) and coarse fish (Mann and Mills, 1986). In the latter study deadwaters associated with the channel margins represented only 2-3 % of the river's surface, but were considered to be essential refugia for newly hatched larvae. RHS at higher flows is being carried out at some sites to investigate changes with discharge. For those sites not mapped predictions of biotope changes can be made from Figure 5.6, although it is not possible to calculate biotope areas.

The proportion of channel width which is secondary biotope also varies with channel morphology and substrate diversity; those channels with large boulders and bedrock e.g. the West Allen having a larger proportion of secondary biotopes across the range of discharges examined (Figure 5.8a). Sites with fine, mobile gravel e.g. the Till are also associated with a large percentage of secondary biotopes. Macrophyte vegetation increases the horizontal biotope heterogeneity at the Derwent and Skerne sites. In unvegetated channelized rivers the proportion of secondary biotopes is very low e.g. the Ouseburn site. In the regulated North Tyne site there is little variation in biotopes across the channel other than the presence of marginal *deadwaters*. This may be a result of bed armouring which produces a more uniform cross-sectional area. The ecological implications are discussed in section 7.3.2.

5.5.2 Cell level variation in flow types

The cell level data set was analysed to calculate the percentages of different flow types present at each site. In effect, percentage values indicate the error in classifying flow types at the transect level in RHS. Cell level data may also be interpreted by ecologists working at a smaller 'patch' scale. Percentages are listed in Table 5.3; these results are discussed in the context of patch dynamics theory in section 7.1.1.

Site	Flow type							
	Ch	BSW	USW	Rip	SBT	SPF	SRip	Up
Allen low	3.5		6.5	9	43	33.5	4.5	
Allen high	11	10.5	29	34	8	7		0.5
Harwood low	14		20	31	7	18	10	
Kielder high	4.5	41	39.5		11.5	3.5		
Lambley low	3	2	4	7	32	46	6	
Lambley mid	11	3	5	29	49	3		
Till high			20.5	21	11	17.5	0.5	29.5
Stanhope low	2	7	13	35	35	4	4	
Stanhope mid	2.5	12.5	18	44	2.5	9		11.5
Wolsingham low	6	0.5	16.5	24	21.5	19.5	11	1
Wolsingham high		5	26	39.5	16	13.5		
North Tyne mid				82.5	17.5			
Derwent high			34.5	42	18	5.5		
Ouseburn high-natural			19	28	47	6		
-channelized				93	7			
Skerne high			27		62	11		

Table 5.3: Percentages of flow types recorded at the cell-level classified site calibrations.

At all sites a range of flow types are present in varying proportions. Sites with the greatest diversity in flow types are those with high biotope diversity indices, but the proportion of each flow type (and therefore biotope) can only be estimated with this detailed biotope mapping. The diversity of flow types generally decreases with increasing discharge. In the context of RHS it may be prudent to map, in more detail, *areas* of secondary biotopes (they are simply recorded as present or absent in the current RHS system). Mapping areas of secondary biotopes may only be necessary in channel types with a high diversity index, or at low flows. The future development of this work is discussed in section 7.6.

5.6 SUMMARY

The concept of patch dynamics emphasises that the biotope approach must not be a static one in space and time: biotope heterogeneity can make a contribution to habitat evaluation and flow assessment. Whilst the application of a suite of measures from landscape ecology to channel biotopes is a possible future study, this chapter has illustrated that simple, transect-level mapping of dominant and secondary biotopes allows calculation of meaningful indices: 'patches' and 'diversity'. These indices relate logically to a channel typology and also illustrate the negative aspects of flow regulation and river engineering. The indices change with changing discharges; there is a relatively predictable sequence of biotope types and numbers as water level rises and falls. The technique is, therefore, a qualitative parallel to the backwater hydraulic models within PHABSIM but with direct relevance to habitat evaluation. The technique reveals the 'resetting' effect of major floods on physical habitat and illustrates the importance of moderate, yet varied, flows as environmentally acceptable. It can also be used to refine the proposed Habitat Quality Index based on River Habitat Surveys.

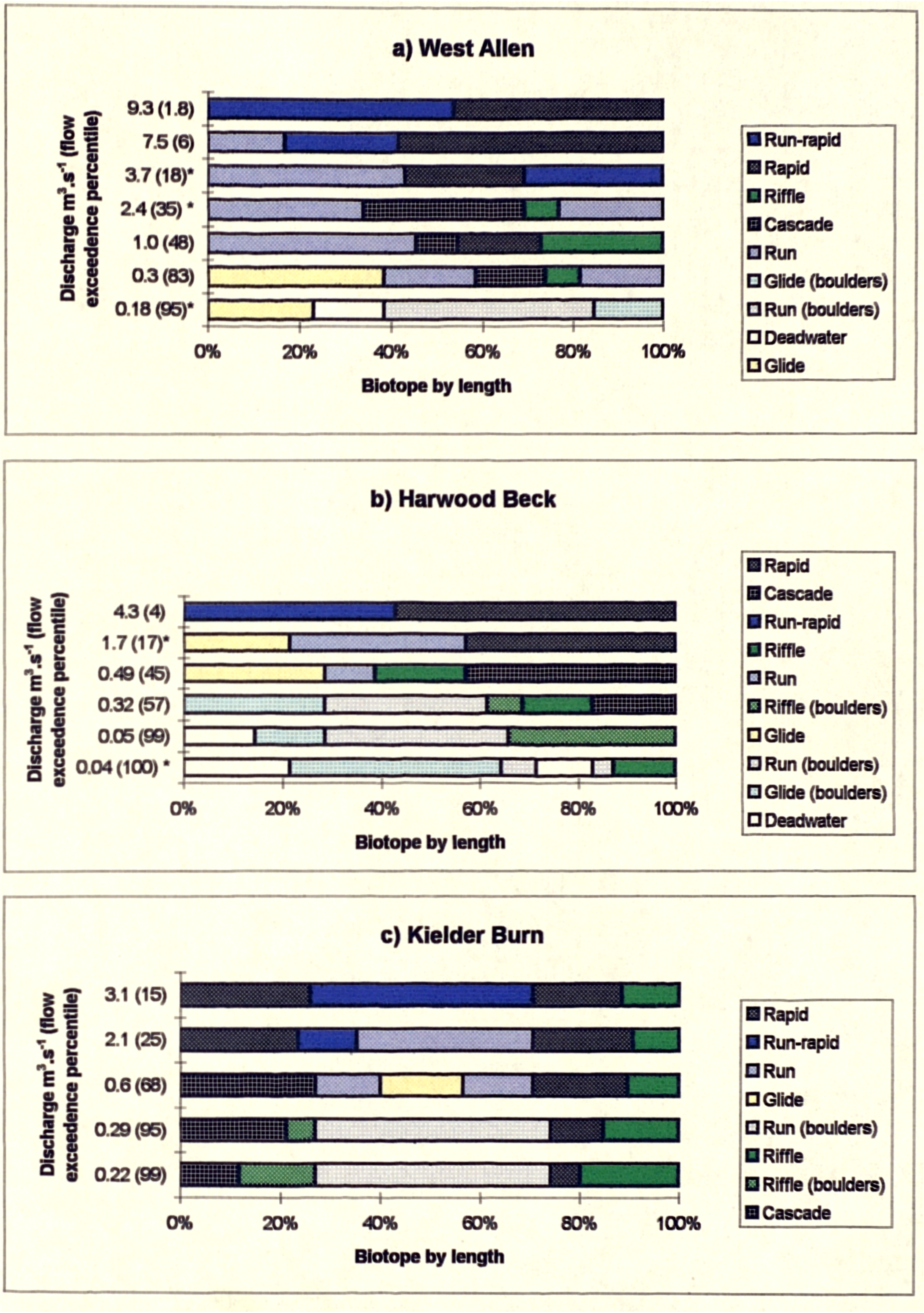
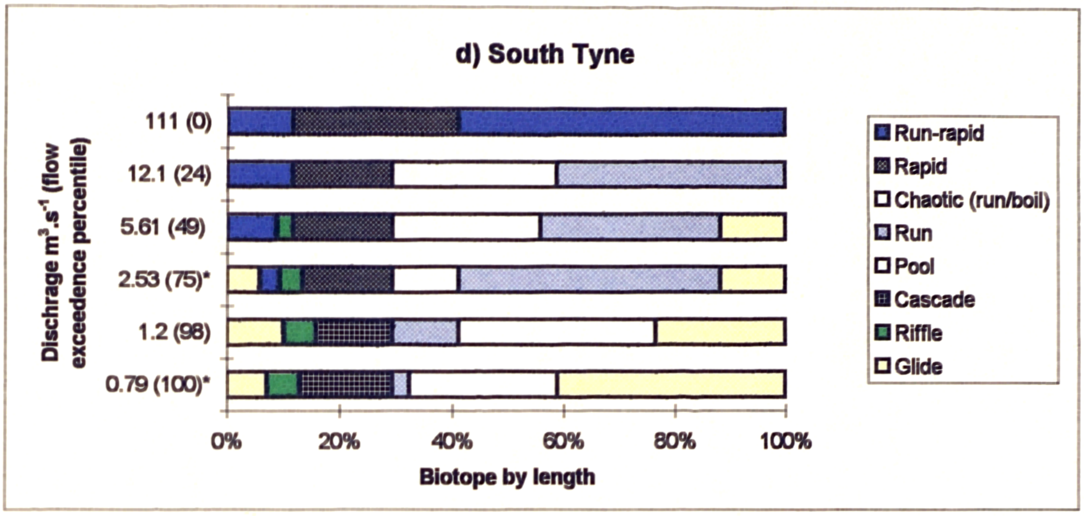
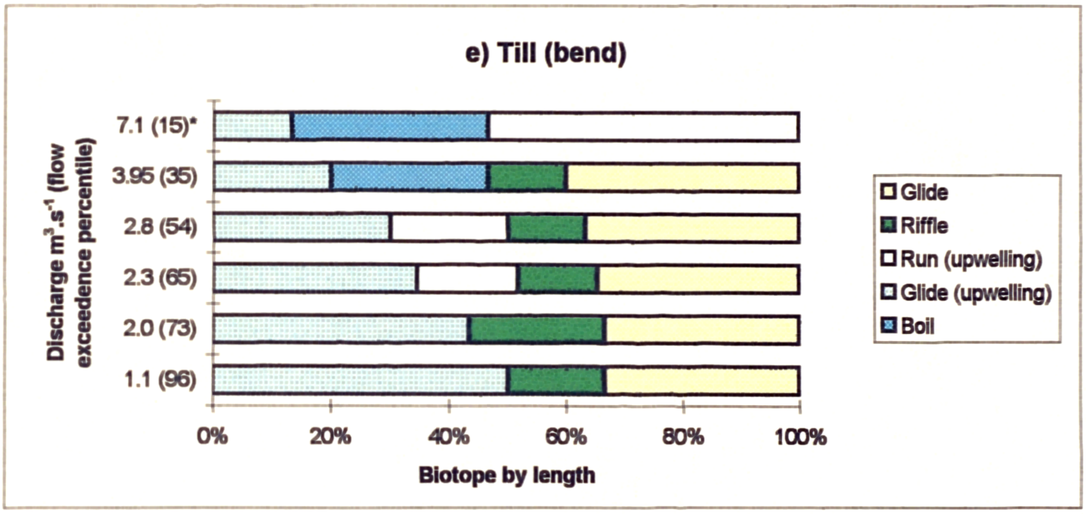
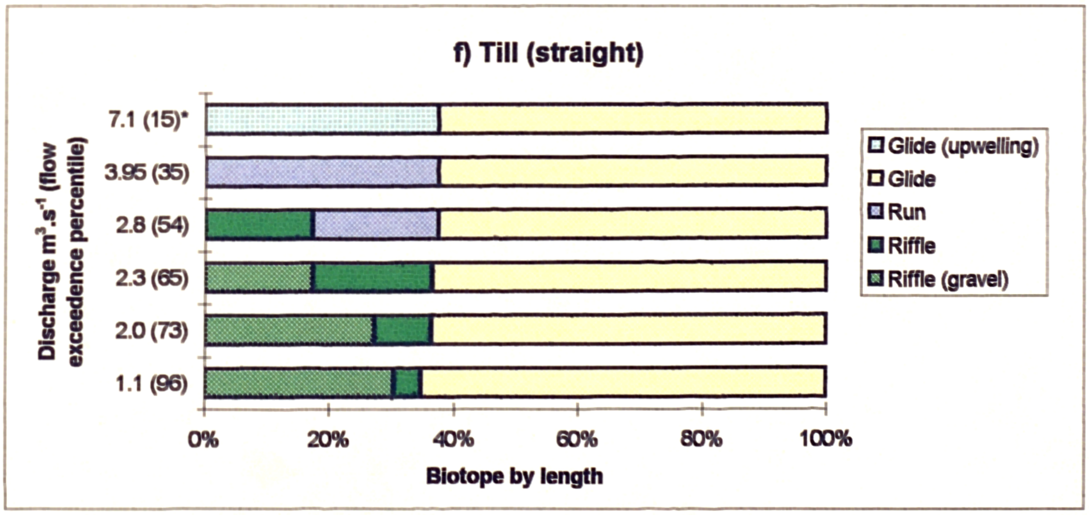


Figure 5.1: Biotope mapping across a range of observed discharges at the study sites (* indicates a post-flood biotope map).







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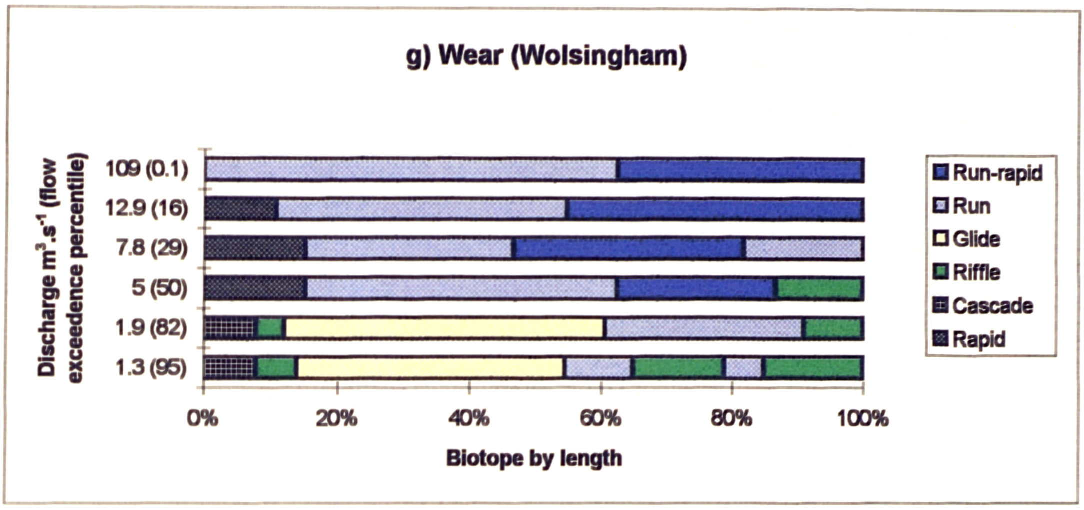

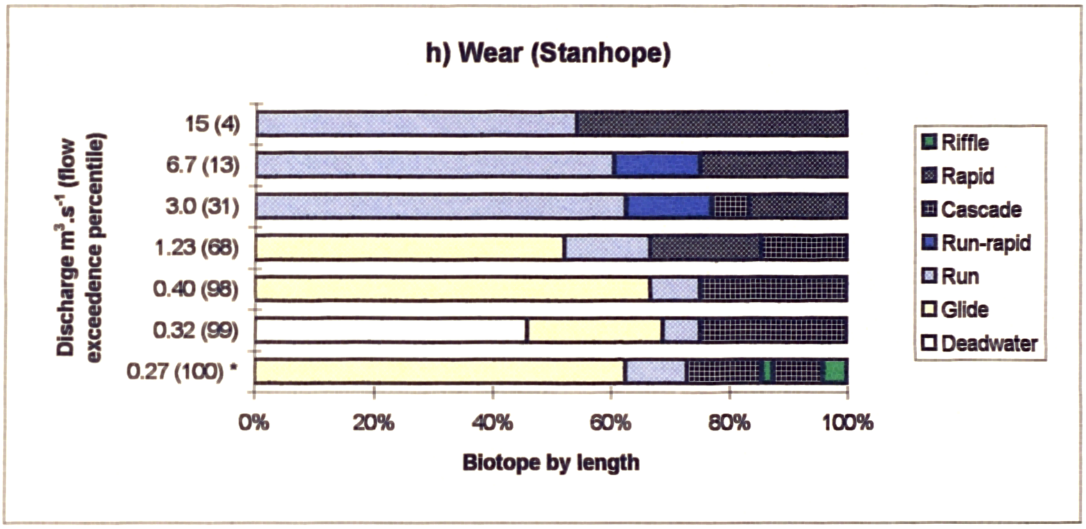

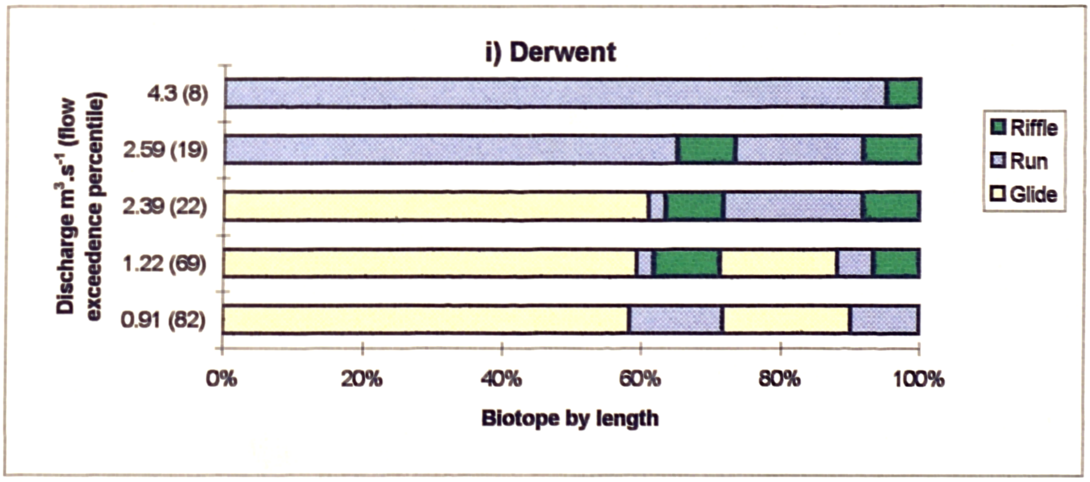

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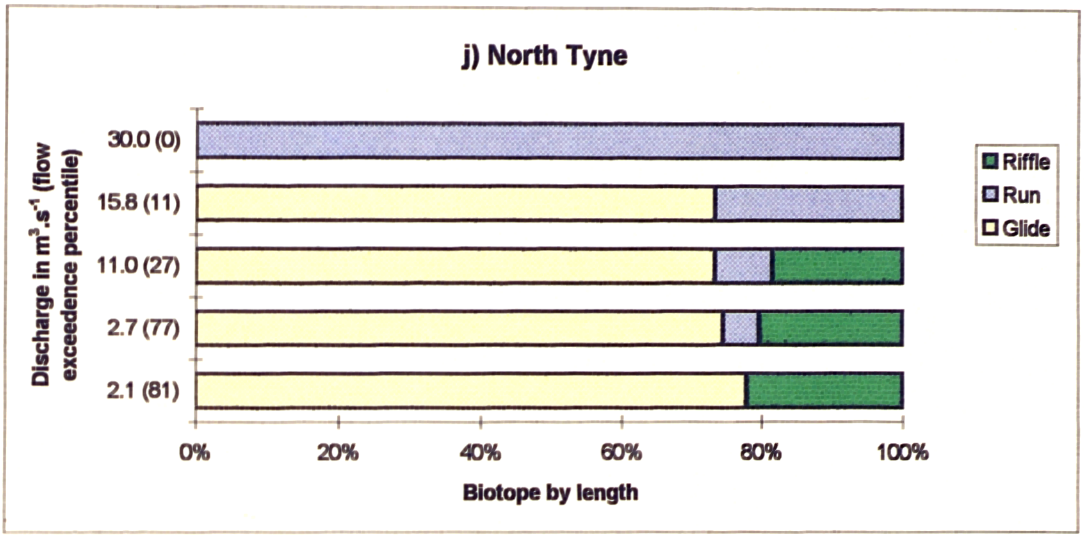
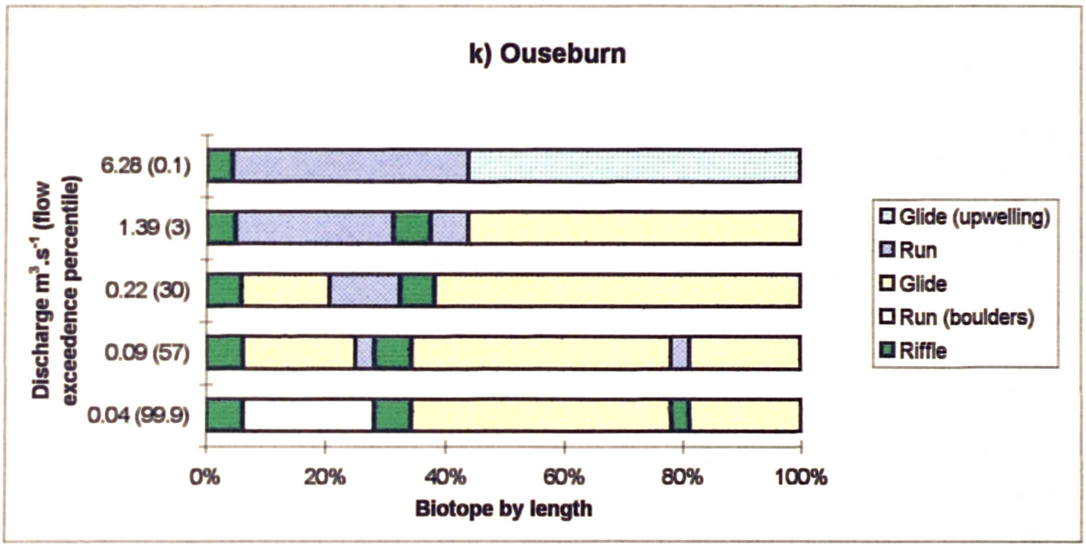




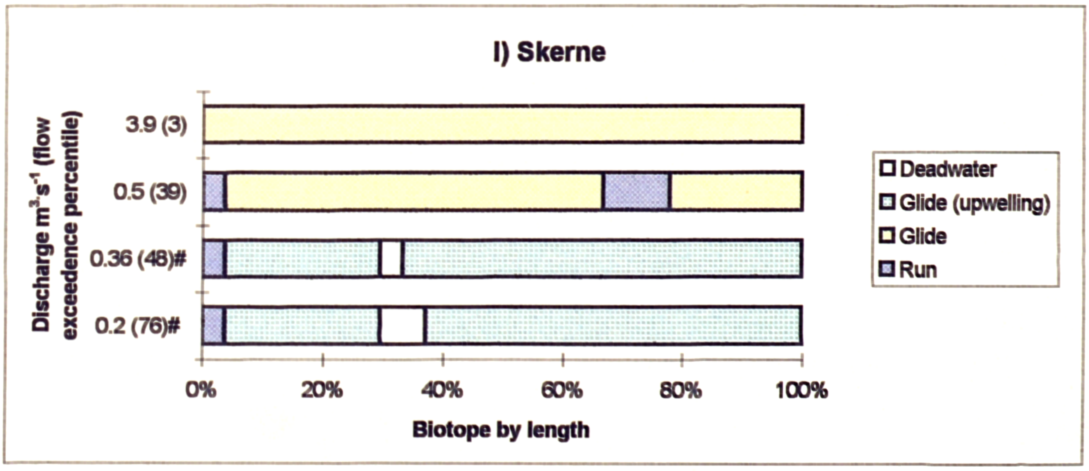


Figure 5.1 cont^{d}. (# indicates a biotope map influenced by instream macrophyte vegetation).

6. HYDRAULIC AND ECOLOGICAL VALIDITY OF BIOTOPE CHARACTERISATIONS

Chapter Overview

This chapter begins with a review of the field-gathered velocity data using calculations of discharge from velocity-area measurements. Discrepancies between gauging station discharges and calculations using 'mean' velocities provide an introduction to the limitations of hydraulic characterisations based on a single velocity measurement (6.1). The velocity profile is discussed for channels where roughness factors are sufficiently great to distort the logarithmic velocity profile. Examples are given from three study sites: one with a boulder bed, a site experiencing seasonal instream macrophyte growth, and upwelling flow associated with a mobile, actively meandering gravel-bed channel (6.2). The implications of these studies are discussed in the context of hydraulic-habitat models such as PHABSIM, and in terms of further research requirements.

The ecological relevance of macro- and micro-scale hydraulic indices is discussed for fish and invertebrates. The influence of simple hydraulic and geomorphic descriptors (depth, velocity, substrate and cover) is considered for their role in fish habitat, whilst shear velocity is concluded to be the most appropriate hydraulic index for describing microflow conditions and invertebrate distributions. Links between flow types and shear velocity are explored, and are advocated as surrogates for the estimation of near-bed hydraulics (6.3). A summary of the implications of these findings is provided (6.4).

6.1 DISCHARGE AND VELOCITY MEASUREMENTS: TESTS OF ACCURACY

Two issues are relevant to the work reported here and general eco-hydraulic studies. Firstly, it is necessary to consider whether the velocity measured at 0.6 depth from the water surface is representative of the range of existing velocities. In channels where the vertical velocity profile assumes a logarithmic shape, velocities measured at 0.6 depth from the water surface are approximately equal to average column velocities (Gordon *et al.*, 1992). Secondly, it has been questioned whether conditions measured in the moving water column have any significance to near-bed hydraulic conditions and the associated biota (Davis and Barmuta, 1989; Carling, 1992a). In order to test the first of these reservations discharges calculated at permanent NRA gauging stations may be compared with values derived from field measurements. Several methods exist for calculating discharge, and these generally fall into four categories as described in Gordon *et al.* (1992). Permanent gauging stations normally use artificial structures with fixed stage-discharge relationships based on the hydraulic head or depth of water passing the weir. At some gauging stations the **velocity-area method** may be employed when several depth and velocity readings are taken across transects to enable calculation of cross-sectional areas. Discharge is then calculated as:

$$Q \quad VA,$$

where Q = discharge ($m^3.s^{-2}$),
V = average velocity ($m.s^{-1}$),
A = cross sectional area of the water (m^2).

Average velocity is dependent upon the velocity profile, originally referred to by Mariotte in the 17th century (quoted in Levi, 1995):

> *"...the water in a river does not advance with equal velocities at the surface and other parts, because near the bottom it is much delayed by encountering stones, weeds, and other irregularities."*
>
> Levi, (1995, p136)

Velocities are generally faster towards the centre of streams and decrease towards the banks due to frictional resistance (Gordon *et al.*, 1992). Discharge is therefore calculated from a number of velocity readings, taken at verticals across transects placed perpendicular to the direction of flow. The British Standard for discharge calculations states that velocities should be measured at intervals spaced not more than 1/15th of the width in channels with regular bed profiles, or 1/20th in beds with irregular profiles (British Standards Institute, 1980). However, the data requirements of velocity-area discharge gauging and PHABSIM calibrations are fundamentally different; the former needing a standard *number* of velocity verticals and the latter a standard spacing (Johnson *et al.*, 1994). When work for this thesis commenced, it was anticipated that recorded hydraulic conditions at different calibration discharges would be compared with PHABSIM simulations. IFIM data collection guidelines were therefore followed as described in section 3.3. Nevertheless at the five sites selected here for comparison of gauged and calculated discharges, the requirements of the British Standard are relatively well met by a 1m spacing. Individual discharge calculations are then summed for the cross-section using the following standard formula:

$$Q = w_1D_1v_1 + w_2D_2v_2 + w_nD_nv_n$$

Gordon *et al.*, (1992, p163)

where w is the width of the part-section in metres, D the depth of the vertical in metres, v the average velocity at each vertical ($m.s^{-1}$) and Q is the discharge ($m^3.s^{-2}$). Average velocities for turbulent flow are calculated from standard velocity profiles based on equations developed by Prandtl and von Karman (Gordon *et al.*, 1992). In channels where the depths are large compared with bed material size, the velocity varies logarithmically with distance from the bed (Bathurst, 1988). This relationship allows the average velocity to be reasonably accurately represented by velocity readings taken at a depth of 0.6 from the water surface (Bathurst, 1988; Gordon *et al.*, 1992).

The selection of an appropriate site for calculating discharge appears to be fairly restrictive. As Gordon *et al.*, (1992) describe:

> *"A fairly straight reach should be chosen in which flow, depth, width, velocity and slope are relatively uniform. Sites should be avoided which have extreme turbulence, upstream obstructions, eddies, dead-water zones, divided channels or regions where the flow path is reasonably curved".*
>
> Gordon *et al.*, (1992, p160)

The transect emplacement for site calibrations in this study is similar to those employed for velocity-area gaugings but in order to study physical habitat the 'ideal" flow conditions for the technique of discharge gauging are deliberately flouted. In natural channels ideal conditions for flow gauging occur less frequently than may be expected. To investigate the representativeness of velocity readings taken at 0.6 depth, calculations of discharge were made at each transect using the velocity-area method described above. A program was written in SAS to calculate discharges at all calibration flows for those sites which were directly adjacent to a NRA gauging station. The calculated discharge at each transect was then averaged for biotope types classified at the transect scale. Comparisons with gauged discharge readings taken at 15 minute intervals, averaged over the period of hydraulic calibration provide an indication of those biotopes in which velocities at 0.6 depth are unlikely to represent average velocities. Identification of hydraulically complex biotopes is partly intuitive based on the degree of surface turbulence; comparisons of gauged and calculated discharges provide a measure of the relative error in assuming readings taken at 0.6 depth to be representative of average velocities. Such a comparison reasonably assumes that field measurement of width and depth is much less error-prone than that of velocity. It also assumes that errors in velocity measurement are in *representing* velocity, i.e. that the current meter is calibrated and operated properly. Figure 6.1 summarises the discharge data for the five study sites where a comparison is feasible. NRA/EA are satisfied with the quality of discharge data from all five sites.

Discharges calculated in *deadwaters* and *glides* are less than those at gauging stations, due to slower velocities in biotopes associated with pool morphological units. Calculated discharges in runs most closely resemble gauged values. Average discharges in *cascades* and *rapids* deviate most significantly from gauged readings, but with no consistent pattern. This implies that velocities recorded at 0.6 depth do not accurately represent average velocities in cascades and rapids; neither do they deviate from average values in a manner which could be described by a simple algorithm. The validity of PHABSIM simulations in hydraulically complex sections is questionable, a reservation made by authors previously (King and Tharme, 1993; Gippel and Stewardson, 1996).

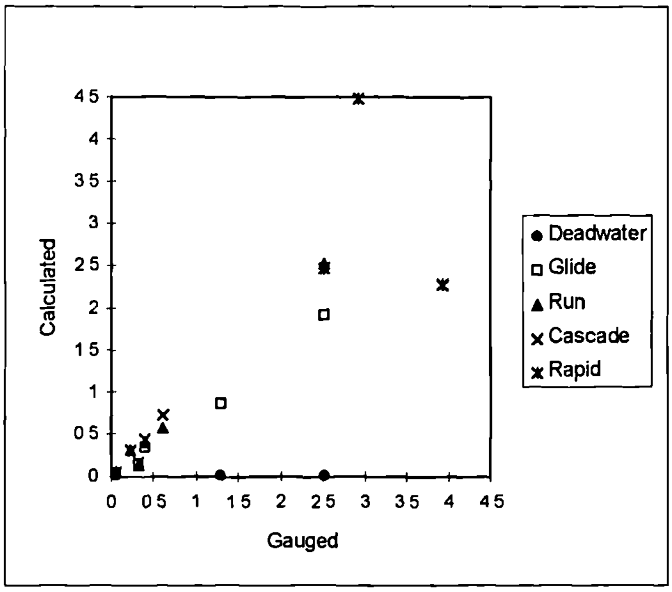


Figure 6.1: Comparison of discharges ($m^3.s^{-1}$) calculated for biotopes at the transect scale with gauged discharges at five study sites.

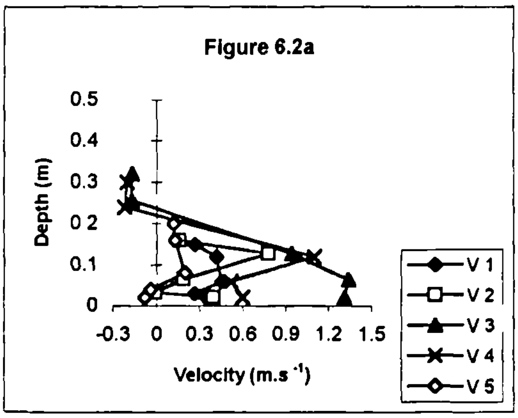
The discrepancy is least for *run* transects, an predictable outcome since these correspond most closely to ideal velocity-area sections. Additionally, it is the suspicion that velocity measured at 0.6 depth is unrepresentative of microflow conditions, especially those near the bed (because of deviation of the velocity profile from the theoretical norm); this hypothesis led to more detailed studies of the most complex flow conditions. The next section goes on to investigate the hydraulic conditions at three sites which almost certainly have non-logarithmic velocity profiles. These include boulder bed cascades, vegetated reaches and those dominated by upwelling flow.

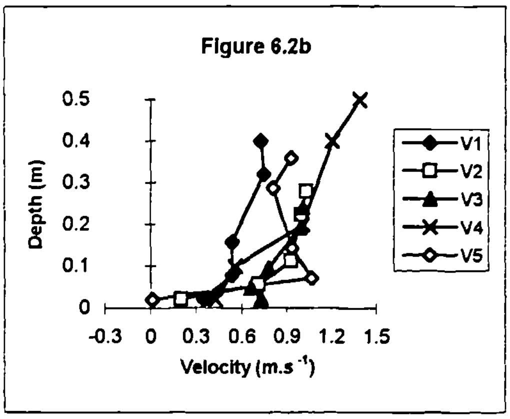
6.2 VELOCITY PROFILE STUDIES

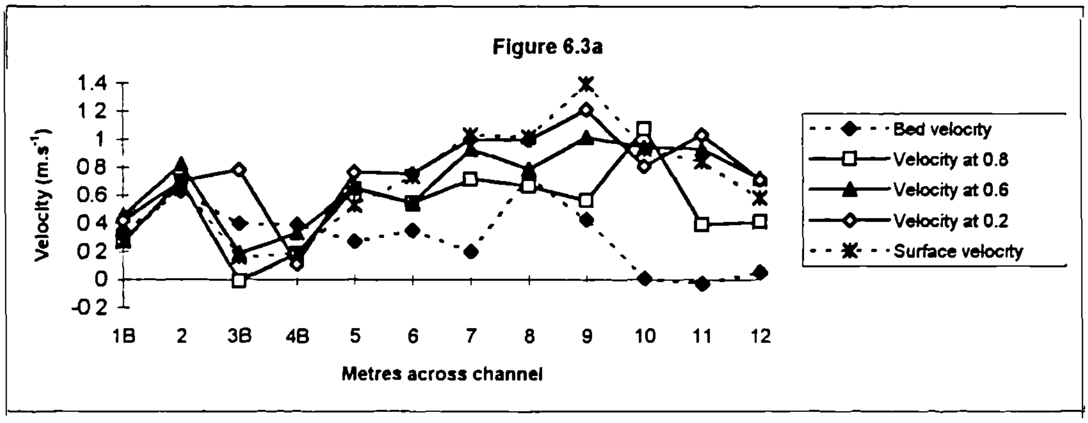
6.2.1 Velocity profiles in boulder-bed cascades

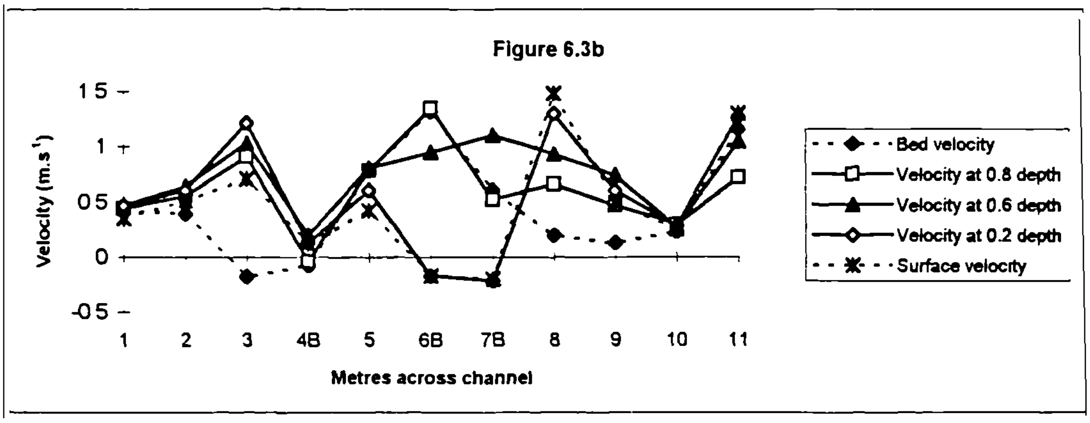
When the depth of roughness elements is high in relation to the depth of water, the classic velocity profile is distorted (Gordon *et al.*, 1992), producing an S-shaped profile in shallow, steep cobble and boulder bed streams (Jarrett, 1984; Marchand *et al.*, 1984). In biotopes where roughness elements protrude through the water surface, the values of velocities taken at 0.6 depth are unlikely to represent the mean velocity (Jarrett, 1985; Bathurst, 1988), and are unlikely to determine near-bed hydraulics. This latter point has been illustrated by Beebe (1996), who uses empirical results to show how calculations based on velocity readings at 0.6 depth can produce large errors in the prediction of velocities at the bed and water surface. Under such conditions, calculations of near-bed hydraulic indices based on measurements at 0.6 depth are unlikely to be accurate, with implications for predictions of invertebrate distributions e.g. Statzner and Higler (1986), Statzner *et al.* (1988). To investigate further the hydraulics of such reaches, the Harwood Beck site was selected to exemplify high roughness conditions, being characterised by a *cascade-run* sequence at low flows and *rapid* biotopes at relatively high flows. This site is very close to the Tees site studied by Bathurst (1988).

In order to describe the more complex hydraulics of *rapids* and *cascades*, and discuss the implications for both predicted shear velocities and hydraulic models such as PHABSIM, velocity profiles were measured within the boulder-dominated *rapid*, at the high calibration discharge. Five velocity readings were taken at standard depths from the water surface, following guidelines suggested in Gordon *et al.*, (1992). These correspond to the following depth intervals measured from the water surface: bed, 0.8 depth, 0.6 depth, 0.2 depth and surface velocity. The data were stored in an Excel spreadsheet and velocity profiles plotted for individual sample points, as illustrated in Bathurst *et al.* (1988). Sample, representative velocity profiles from the Harwood Beck rapid are illustrated in Figure 6.2a and b. Figure 6.2a shows velocity profiles at five verticals within the rapid which are *not* directly above a boulder. In only one vertical does the S-shaped profile described by Bathurst (1988) exist; the majority appear to have an irregular velocity profile. Figure 6.2b illustrates velocity profiles at sample points directly above boulders; at some intervals velocities at the surface are the lowest recorded, whilst bed velocities are the highest in the vertical. This may be explained by the flow patterns in broken standing waves and dissipation of energy at the water surface. Velocities recorded at verticals across sample, representative rapid transects are illustrated in Figures 6.3a and b. It is evident that there is little pattern in velocities, either across the transect at a particular depth interval or within individual sample points. Bed velocities appear to show little relationship with readings taken at 0.6 depth, as illustrated in Figure 6.3a; bed velocity is low but there is little difference in the remaining velocities within the vertical. Those readings which are taken directly above boulders are indicated by a B; these have the most distorted velocity profiles, and unusually low surface or bed velocities.









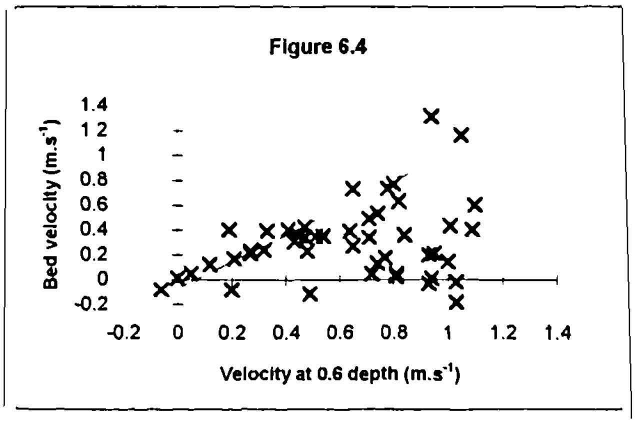


Figure 6.2: Velocity profiles in verticals within a rapid in a) random locations, and b) directly above boulders.

Figure 6.3a,b: Velocity profiles across rapid transects (B indicates the presence of a boulder).

The implications of these findings are twofold: from an ecological perspective it is evident that boulder bed rapids (and cascades) have considerable variation in near-bed velocities; the existence of low velocity zones could provide small-scale refugia from the faster near-bed velocities which are also present within the biotope. Heede and Rinne (1990) describe how low velocity areas downstream of boulders or other obstructions provide resting, feeding and spawning habitat for fish. Young (1996) confirms the need for much more detailed habitat hydraulic calibration of such reaches. The hydraulic implications are relevant to both calculations of Weighted Useable Area (WUA) in PHABSIM (based on readings taken at 0.6 depth, which are assumed to be 'mean' velocity), and predictions of near-bed hydraulics in relation to physiological studies of biota. Hydraulic models such as PHABSIM are unlikely to accurately predict the range of velocities at different discharges (either at the bed, or 0.6 depth), which will obviously affect the validity of WUA estimates. In this study, shear velocity has been adopted as the most appropriate near-bed hydraulic index, as it has relevance not only to invertebrate ecology but sediment transport processes. However predictions of shear velocity using the formula in section 3.6.2 are likely to be inaccurate in biotopes where the hydraulic conditions are controlled by boulders with high relative roughness. Figure 6.4 illustrates the range of measured bed velocities compared to those measured at 0.6 depth for the Harwood Beck rapid; there appears to be much scatter in the bed velocities particularly at higher 'mean' velocities.

6.2.2 Vegetation and velocity profiles

The impact of macrophyte vegetation on in-channel hydraulics has been investigated by several authors (Dawson and Robinson, 1984; Marshall and Westlake, 1990). In channels with regular cross-sectional morphology and steady uniform flow, plants maintain depth in low flow conditions and associated available habitat for invertebrates (Hearne and Armitage, 1993). Associated with this increase in depth is altered hydraulic roughness (Dawson and Robinson, 1984; Gordon *et al.*; 1992), and slower surface velocities where flow is retarded by floating vegetation. Of more importance is the increased hydraulic diversity created by vegetation in lowland channels, especially in terms of the range of velocities within stands of macrophytes (Wright *et al.*, 1994; Padmore, 1995). Newall (1995) found increased diversity of velocities in floating species compared with emergent and submerged vegetation.

In biotic terms vegetation also plays a significant role. Newall (1995) described how Manning's 'n' remains relatively constant with increased discharge, allowing macroinvertebrates to use plants as refuge sites since the plants' resistance to flow is not reduced and low velocities are maintained at all discharges. Various authors have compared vegetated and unvegetated stands in terms of invertebrate density and biomass (Rooke, 1984; Wright *et al.*, 1994; Newall, 1995). Generally, these show increased diversity in vegetated stands of channel margins and deadwaters. Rooke's studies in the main channel of a swiftly flowing riffle-run reach were an exception, in that the vegetated areas were *less* diverse than unvegetated sites in terms of invertebrates. In Rooke's study, vegetation trapped sand or silt substrate, which is known to have a smaller number of invertebrates than gravel and cobble substrate (Hynes, 1970; Cummins, 1973). Links between vegetation, hydraulic and sedimentary conditions are likely to influence the abiotic environment for both fish and invertebrates.

Hearne *et al.* (1994) relate seasonal changes in macrophyte growth to instream hydraulics as a means of determining ecologically acceptable flows. To examine further the seasonal effects of macrophyte vegetation on instream hydraulics, two sites were studied in detail. The rivers Skerne and Cole were selected by the River Restoration Project Board; details of the restoration procedures and pre- and post-project monitoring are provided in section 7.2. This section looks in detail at the effects of macrophyte vegetation on the velocity profile. It also considers the role of vegetation in habitat maintenance at low flows.

The unrestored River Cole at Coleshill, Wiltshire is a low diversity river in terms of biotopes, with slow, hydraulically uniform glides dominating the reach. The restoration reach lies within a section of the Cole designated as an EC coarse fishery. Objectives to improve the physical habitat for fisheries involve improving low flow habitat, especially the presence of 'significant features' (River Restoration Project, 1994a). Prior to the scheme the restoration reach was predominantly *glide*, with the major exception of a 110m pool caused by ponding of flow upstream of the Mill House weir. Hydraulic diversity within the reach mainly associated with *runs*, as *glides* have little hydraulic variation in terms of depths and velocities (see Figure 6.5a). In pre-restoration conditions *runs* are generally associated with 'flams' i.e. deposits of silt/clay where lateral bars would be expected to develop in gravel-bed rivers. Sedimentation reduces depths and encourages vegetation development by providing root attachment sites and relatively shallow water. In the June survey the 'flams' extended approximately 30% of the channel width from both banks due to the development of dense stands of *Schoenoplectus lacustris*. Both the presence of vegetation and larger substrate size contribute to increased hydraulic diversity in *runs*. However, 'flams' were traditionally removed by regular maintenance so do not create permanent hydraulic diversity.

Full velocity profiles were not measured in the Cole due to time constraints (and because this site was not one of the original study sites). The work carried out was essentially a biotope mapping exercise (see section 5.1), together with comparisons of 0.6 velocities in pre- and post-restoration conditions. However, because the seasonal presence of in-channel macrophytes in the Cole is a major habitat factor its gross effect was measured by visiting the reach in both spring and summer, comparing the velocity and depth distributions in vegetated and unvegetated conditions (Figure 6.5). In June, the channel was occupied with submerged e.g. *Myriophyllum spicatum* and floating species e.g. *Nupar lutea*; flow was further obstructed by emergent species including *Sparganium erectum, Schoenoplectus lacustris* and *Typha latifolia.* Figure 6.5b indicates a reduction in velocities in *runs* which had dense growth of floating *Ranunculus penicillatus*, together with *Typha latifolia* at the channel margins. Both velocities and depths were reduced in vegetated *glides*, which had little floating or submerged vegetation but dense stands of emergent *Butomus umbellatus* and *Phalaris arundinacea* extending several metres into the channel. However, these differences may be partly masked by the lower discharges in the unvegetated conditions. In order to remove the influence of discharge and focus on the effects of vegetation on instream hydraulics, results from the unvegetated survey were analysed further. Only in runs was there a sufficient number of vegetated cells to allow statistical comparisons to be made; mean velocities were calculated for unvegetated cells, those with emergent macrophytes and a third group with submerged or floating macrophytes present. Vegetation appears to reduce velocities slightly; a mean velocity of 0.44 $m.s^{-1}$ was reduced to 0.42 $m.s^{1}$ in cells with floating or submerged vegetation, and 0.29 $m.s^{-1}$ where emergent vegetation is present (see Table 7.1 for species lists).

Percival and Whitehead (1929) classified vegetation into four types with varying hydraulic roughness. For example, they contrasted “loose moss” which allows passage of flow and deposition of silt e.g. *Fontinalis spp.*, with “thick moss carpets” which prevent the passage of flow and encourages deposition of fine detritus. It seems that emergent vegetation is having a greater influence on instream hydraulics, which is logical in view of their more robust structure. Whether such slight reductions are significant to the instream biota is questionable. It is more likely that the diversity of habitat offered by the vegetation structure will increase the overall habitat quality or suitable area (Harper *et al.*, 1995). A study of 76 sites on 30 lowland rivers showed that areas with macrophytes present had a significantly higher species' abundance than macrophyte-free gravel and silt areas. Emergent macrophytes had the greatest family richness, followed by submerged and floating vegetation, then non-macrophyte areas (Wright *et al.*, 1994). Such factors must be considered in addition to hydraulics; a factor overlooked by hydraulic models such as PHABSIM.

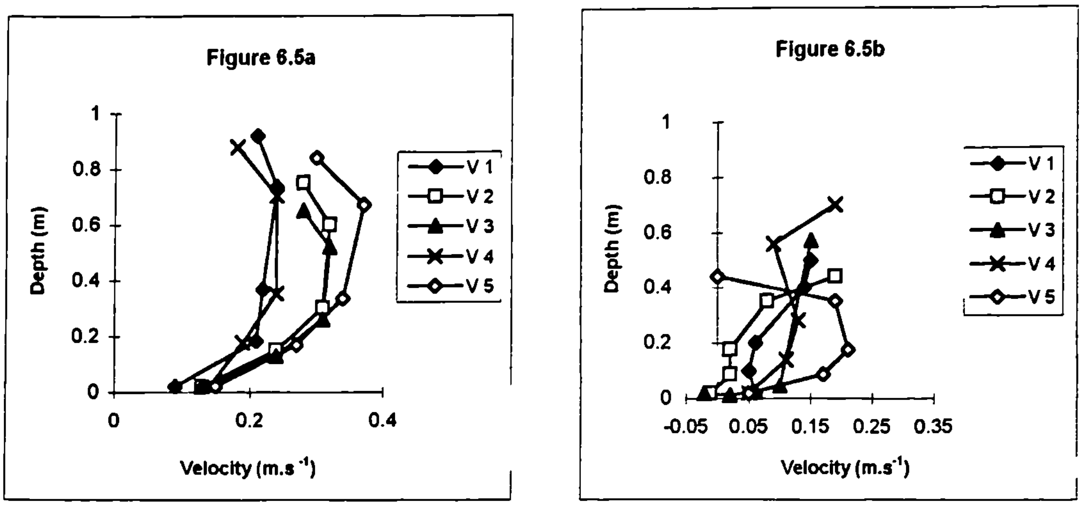
In the River Skerne, macrophyte vegetation dominates the distribution of flow types between June and September. In unvegetated conditions observed in November 1995, no true *riffles* were present as the channel dimensions are artificially deep and wide, due to modification in the 1960's (Newson *et al.*, 1994). This resulted in a reach which was uniform *glide* for approximately 1.2km. In vegetated conditions the glide had upwelling flow present, which indicated greater hydraulic variation. The influence of vegetation on velocity profiles was investigated by taking velocity readings at standard depths, as described in 6.2.1. 'Vegetated' measurements were taken in October 1994; peak biomass occurred in July and was starting to degenerate in October, but these conditions were selected to compare with the unvegetated reach because it was necessary to wait for flows to return to winter baseflow following low summer flows. 'Unvegetated' velocities were measured in November 1994, when flows were similar (i.e. Q_{48} in October, 1995; Q_{39} in November 1995), but the vegetation had died back. Mean velocity is calculated as the average of the five vertical readings at all points sampled. Ratios of the range of velocities to mean velocity give an indication of the influence of vegetation on velocity distributions. Differences are summarised in Table 6.1.

	Mean depth	Bed V	V 0.8 depth	V 0.6 depth	V 0.2 depth	Surface V	Range	Mean	Range: mean
Vegetated	0.51	0.01	0.04	0.05	0.07	0.08	0.09	0.04	2.25
Unvegetated	0.55	0.10	0.20	0.26	0.34	0.24	0.24	0.22	1.09

Table 6.1: Comparison of mean depths (m) and velocities (V) ($m.s^{-1}$) at 56 and 73 verticals in 8 and 9 respective cross-sections in vegetated and unvegetated conditions of the River Skerne.

The effect of discharge on velocity distributions is likely to contribute to some degree to the higher velocities recorded in the unvegetated conditions i.e. a flow of Q_{39} compared to Q_{48} for the vegetated site calibration. These discharges are relative not absolute comparisons as the study site is not directly adjacent to a gauging station, although no major tributaries flow into the Skerne between the gauge at Preston-le-Skerne and the study reach. Despite differences in flow, the effect of vegetation on velocities is evident; mean velocity is five orders of magnitude greater in unvegetated sections compared to vegetated conditions, where mean velocities are reduced due to increased flow resistance. Faster 'threads' are created where vegetation is locally absent which increases the range of recorded velocities in vegetated conditions. In unvegetated conditions the range of recorded velocities is much lower due to the lack of substrate and topographical variation in an engineered channel.

Sample velocity profiles for vegetated and unvegetated sections give an indication of the overall influence of plant biomass on instream hydraulics, as illustrated in Figure 6.5a and b. In unvegetated conditions, the 'logarithmic' velocity profile is slightly distorted at the water surface; at such low velocities the friction of air and water at the surface will be sufficiently significant to reduce surface velocities. The fact that an 'ideal' logarithmic profile is not present raises questions as to the validity of the velocity profile theory; if it exists anywhere, it should be present in a channel with little substrate roughness and a uniform, rectangular cross-section. The presence of vegetation further distorts the velocity profile, such that there appears to be no regularity in the sample profiles. Hydraulic conditions are determined by localised vegetation patches, and their distribution within the water column. The presence or absence of vegetation throughout the entire vertical determines the range of recorded velocities; slower surface velocities occur when vegetation is present throughout the water column, which is consistent with observations by Gordon *et al.*, (1992). From a management perspective it would appear that the range of velocities is related to vegetation biomass (Newall, 1995). Shear velocity is unlikely to constitute a habitat factor affecting invertebrate distribution in lowland rivers, as there is very little variation in either velocities or substrate size. Vegetation enhances the habitat value, by the provision of structures which act as attachment sites (Wright *et al.*, 1992; Harper *et al.*, 1995). This supports the maintenance of instream vegetation to increase hydraulic diversity for invertebrates, and justifies the selective, not complete removal of vegetation for flood defence maintenance.



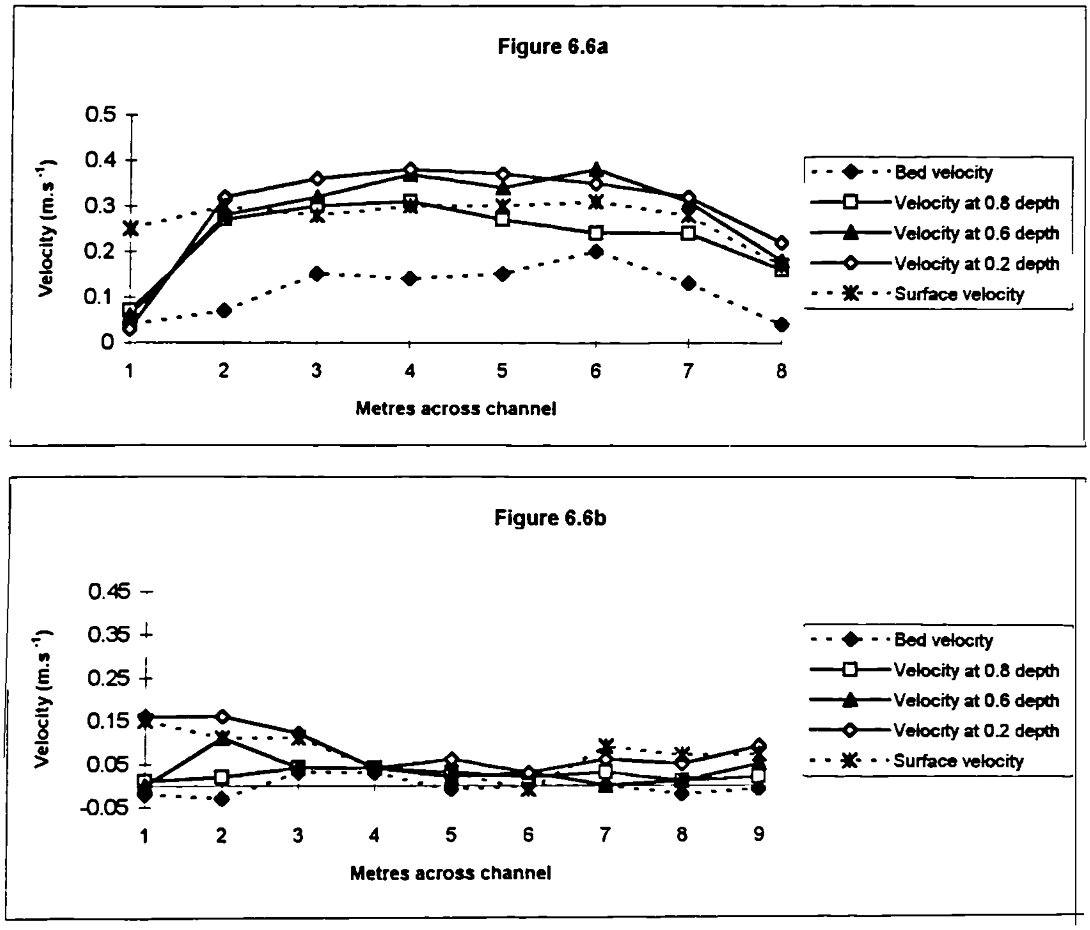


Figure 6.5: Velocity profiles in verticals within a glide in a) unvegetated, and b) vegetated conditions.

Figure 6.6: Velocity profiles across glide transects in a) unvegetated, and b) vegetated conditions.

Velocity profiles across unvegetated and vegetated transects are illustrated in Figures 6.6a and b. Both a reduction in velocities and distortion of the velocity profile are evident in Figure 6.6b. Velocities are measured in the same *glide* transect, so the hydraulic conditions in the vegetated graph may be attributed entirely to increased roughness from the macrophytes. In such conditions it appears that the velocity at 0.8 x depth (from the water surface) is most representative of mean velocities (see Table 6.1). In unvegetated sections, bed velocities are unusually high, which indicates reduced roughness in a modified, silt-dominated channel. No single reading is sufficient to describe the range of velocities or provide an accurate estimate of mean velocities. This has implications for hydraulic modelling in vegetated conditions; PHABSIM estimates of WUA are based on velocities at 0.6 depth, assuming a logarithmic velocity profile exists. Algorithms have been devised to incorporate the increased hydraulic roughness from vegetation in the PHABSIM model (Hearne *et al.*, 1994), but it is likely that the precise hydraulic effects depend on vegetation biomass and the vegetation type.

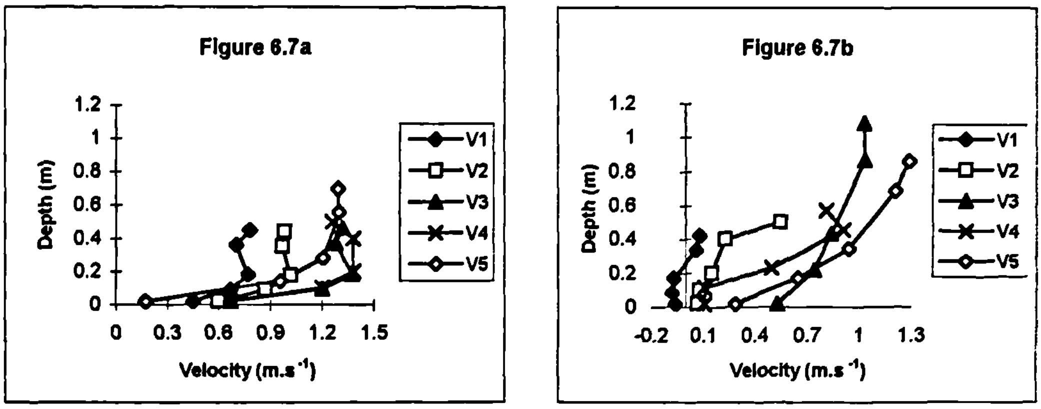
Vegetation has a clear influence on depth of water; under conditions of low summer flow mean depths would be expected to decrease significantly in a channel with a rectangular cross-section. However vegetation in the Skerne caused 'ponding' in the channel; mean depth in unvegetated conditions was 0.55m compared to 0.51m in vegetated conditions, i.e. the vegetated, low flow water level was only 0.04m lower than the unvegetated high flow level. By comparison stage readings at the gauging station were 0.1m lower in October (vegetated) than those measured in November (unvegetated). Thus the vegetation was causing an increase in water level of 0.06m. In some lowland rivers vegetation is used to maintain depths and invertebrate habitat (Hearne and Armitage, 1993). This is not the case at the Skerne site where low water quality is the over-riding influence on biota. In general, however ponding due to vegetation has implications for flood conveyance and carrying capacity; should a high intensity event occur under conditions of peak biomass the risk of overbank flooding would be high. This is less of an issue in the post-restoration Skerne, as the channel has been designed to allow flooding of backwater channels created from the original channel (see Appendix B3).

6.2.3 Upwelling study

Two reaches were selected at the Till site which is a mobile, gravel-bed channel flowing through the Milfield Plain, whose glacio-fluvial deposits produce a substrate dominated by fine gravel (see Table 3.1). The Till is actively eroding and has many unvegetated point bars which alter the direction of flow and biotope sequence. A detailed study site was selected adjacent to a point bar which had a *glide-riffle-glide* biotope sequence at low flow, and a *glide-boil-run* sequence at high flow. Upwelling flow is characteristic of this location within this type of channel. A *deadwater* was present downstream of the point bar and towards the left bank; this remained at all calibration discharges, although its width changed relative to the width of flow over the point bar. At higher discharges, increased turbulence led to a greater area of upwelling flow.

The Marsh McBirney Flo-mate 2000 EMCM is capable of measuring flows in only one orthogonal direction (see section 3.3). Upwelling flow is identified by horizontal eddies on the water surface, and 'swells' where vertical flow cells break the surface (see Table 3.6). Due to the degree of turbulence and complexities of flow direction, it is unlikely that the velocity recorded at 0.6 depth in the direction of dominant flow is representative of average velocities. This was evident when performing hydraulic calibrations, especially at high flow; fluctuation in velocity was so rapid that it was difficult to ascertain the direction of dominant flow. Velocities were averaged over a time period of 60 seconds in as opposed to 15 seconds in the 'slower' biotopes and 30 seconds in *riffles*, *cascades* and *rapids*. Nevertheless, the representativeness of the averaged velocity readings was not considered to be entirely accurate.

Two studies were carried out in the *boil* to investigate the complexities of their hydraulics deriving from the importance of secondary currents. A velocity profile study was performed as described in sections 6.2.1 and 6.2.2. Velocity profiles were measured at high flow; sample velocity profiles from the *glide* and *boil* are presented in Figures 6.7a and b. Additionally results are presented for three transects within the downstream reach at the high flow calibration. Figure 6.8a illustrates the variation in velocities which occur across a transect; metres 1-6 are measured in the deadwater downstream of the point bar, whilst the remaining sample points occur in a *glide*. Figure 6.8b shows the velocity profiles across a boil transect; a full velocity profile could not be established for



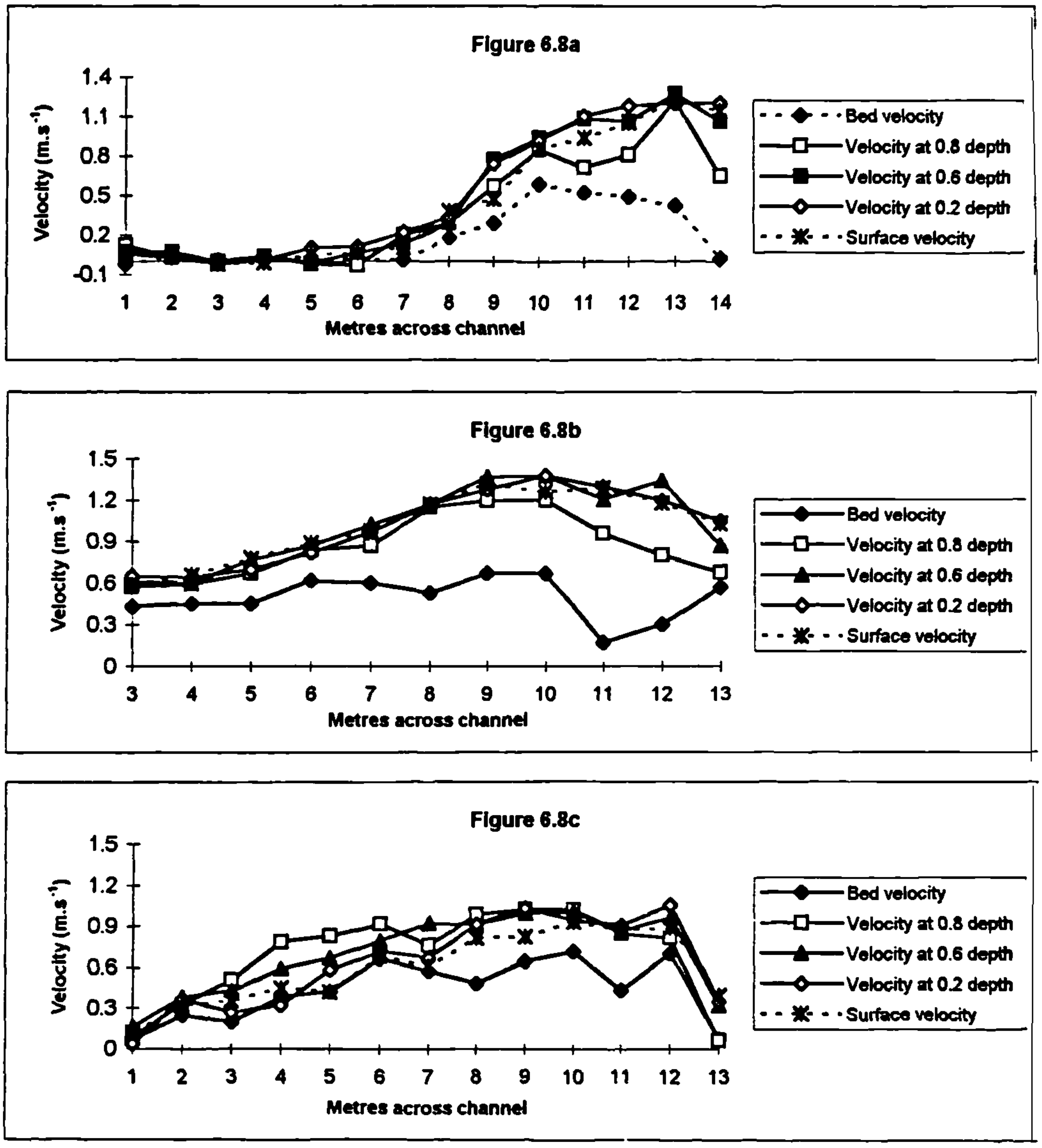


Figure 6.7: Velocity profiles in verticals within a) a glide, and b) a boil in the River Till, low flow.
Figure 6.8: Velocity profiles across transects in a mobile gravel-bed channel (River Till) in a) a deadwater-glide, b) a boil, and c) a glide.

sample points located within the first two metres from the right bank as depth and velocities exceeded those which could be safely measured. At high calibration discharges this reach of the Till site had a mobile bed with further hindered the safe collection of the full datatset. At shallow depths the velocity profile is distorted in the *boil*, as illustrated in Figures 6.7b and 6.8b; (cells at 8-13m flow over the point bar so are more representative of a *riffle*). At greater depths the velocity profile is still distorted, but to a lesser extent. Figure 6.8c is measured upstream of the *boil* in a *glide* transect; it illustrates how the fastest velocities tend to occur at 0.8 depth. It appears that the resistance to flow is reduced at greater depths over fine substrate; however nearer to the surface the turbulence of the flow moving in different horizontal and vertical components creates greater resistance compared to that produced by the substrate.

The British Standard technique for measuring velocities where oblique flow occurs is to record the velocity with the current meter perpendicular to the cross-section, and adjust this value using the following formula:

$$V_{\text{corrected}} = V_{\text{measured}} \cos \gamma$$

where γ is the angle of derivation from the perpendicular. The variation in horizontal flow direction was investigated, to determine the degree of error that may be expected if velocities were measured with the current meter pointing in the direction of surface flow. This was achieved by simply tying a piece of string to the current meter and adjusting its position on the wading rod to a depth corresponding to the velocity vertical being measured. At each depth, the head of the electrode was pointed in the direction of dominant horizontal surface flow, and the string's angle of deviation from horizontal recorded. Results are summarised in Table 6.2 and illustrated in Figure 6.9. Table 6.2 summarises the average angle that the flow deviates from the downstream direction (taken as 180 °), for each transect and depth in the vertical. Figure 6.9 summarises the frequency of measured angles at each vertical depth: the angle of flow is shown as degrees, from 0° to 360° on the radial diagram, the frequency indicated on the vertical axis.

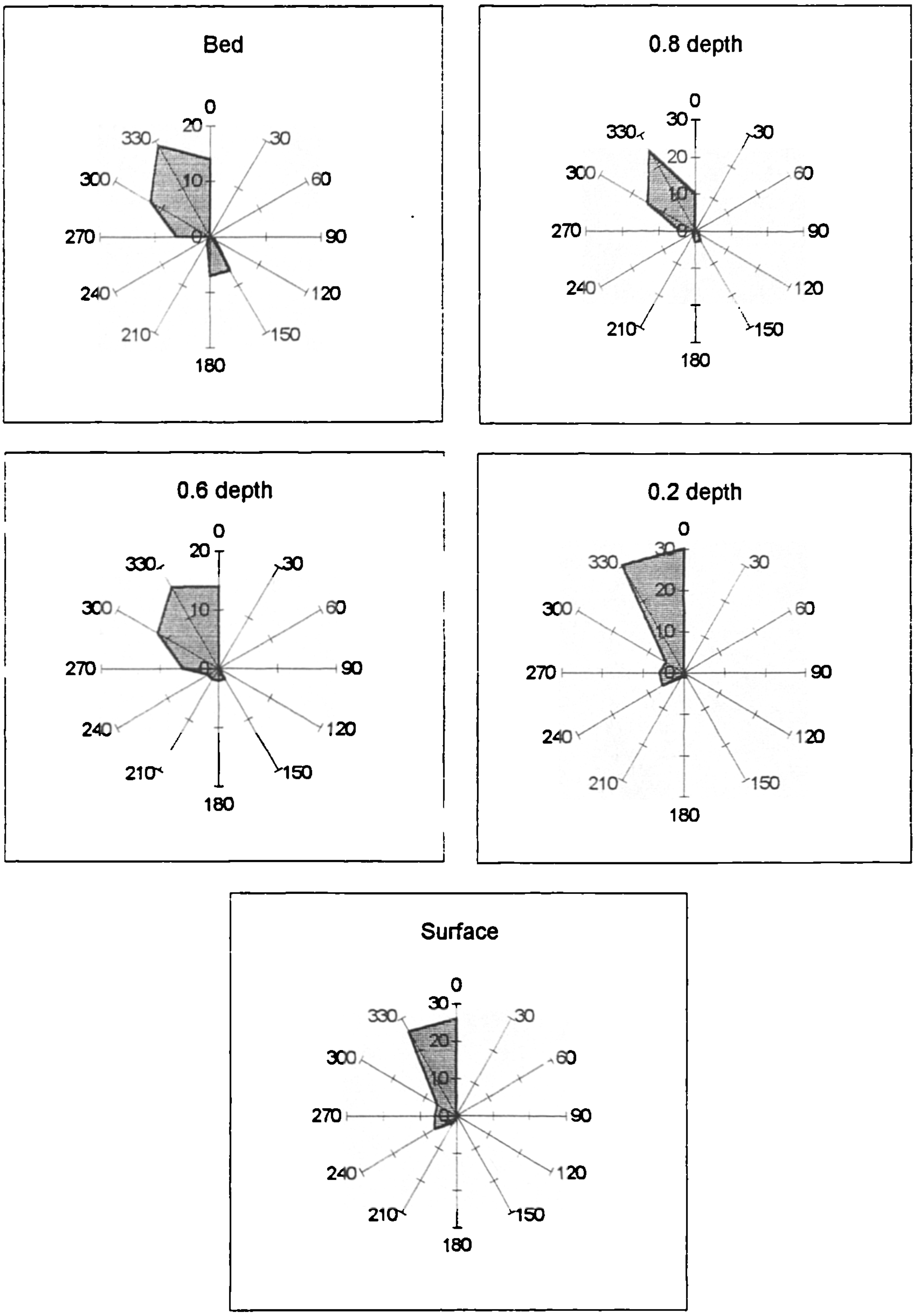


Figure 6.9: Direction of streamline (as indicated by string) showing angle of dominant flow and frequency at standard depths in verticals within a boil.

Transect	Bed	0.8 depth	0.6 depth	0.2 depth	Surface
1	9	59	53	70	85
2	104	112	93	99	99
3	136	133	130	154	141
4	149	143	131	164	148
5	147	141	153	170	171
6	131	133	156	176	177
7	147	146	150	173	176
Average	**118**	**124**	**124**	**144**	**142**

Table 6.2: Angle of deviation from 'true downstream' (180°) at depths within verticals in a boil.

It can be seen that at all depths, the direction of the streamline occurs between 270° and 360°. Moving upwards through the water column towards the water surface the angle of flow deviation from 'true downstream' (180°) increases, as bed roughness has less influence on the flow compared to secondary flow currents. At 0.6 depth, an average flow deviation of 124° results in measured velocities which are approximately 1.8 times smaller than those which would be recorded if the current meter were placed perpendicular to the cross-section (not in the direction of dominant flow). Measured velocities compared to 'downstream' velocities have been calculated using the cosine formula above. These are summarised for all depths within the vertical velocity profile as illustrated in Figure 6.10.

As a final comparison of velocities measured perpendicular to the cross-section and those in the direction of dominant flow, velocity profiles have been replotted using the data measured with the current meter facing the true direction of flow. Figure 6.11 illustrates sample velocity profiles in low flow boils, a horseshoe shaped velocity profile is evident. In general, velocities increase to approximately 0.6 depth, then are reduced by resistance from vertical flow cells. At higher flows secondary flow has a more significant influence on recorded velocities as indicated by the lack of pattern in velocity profiles (see Figure 6.7b).

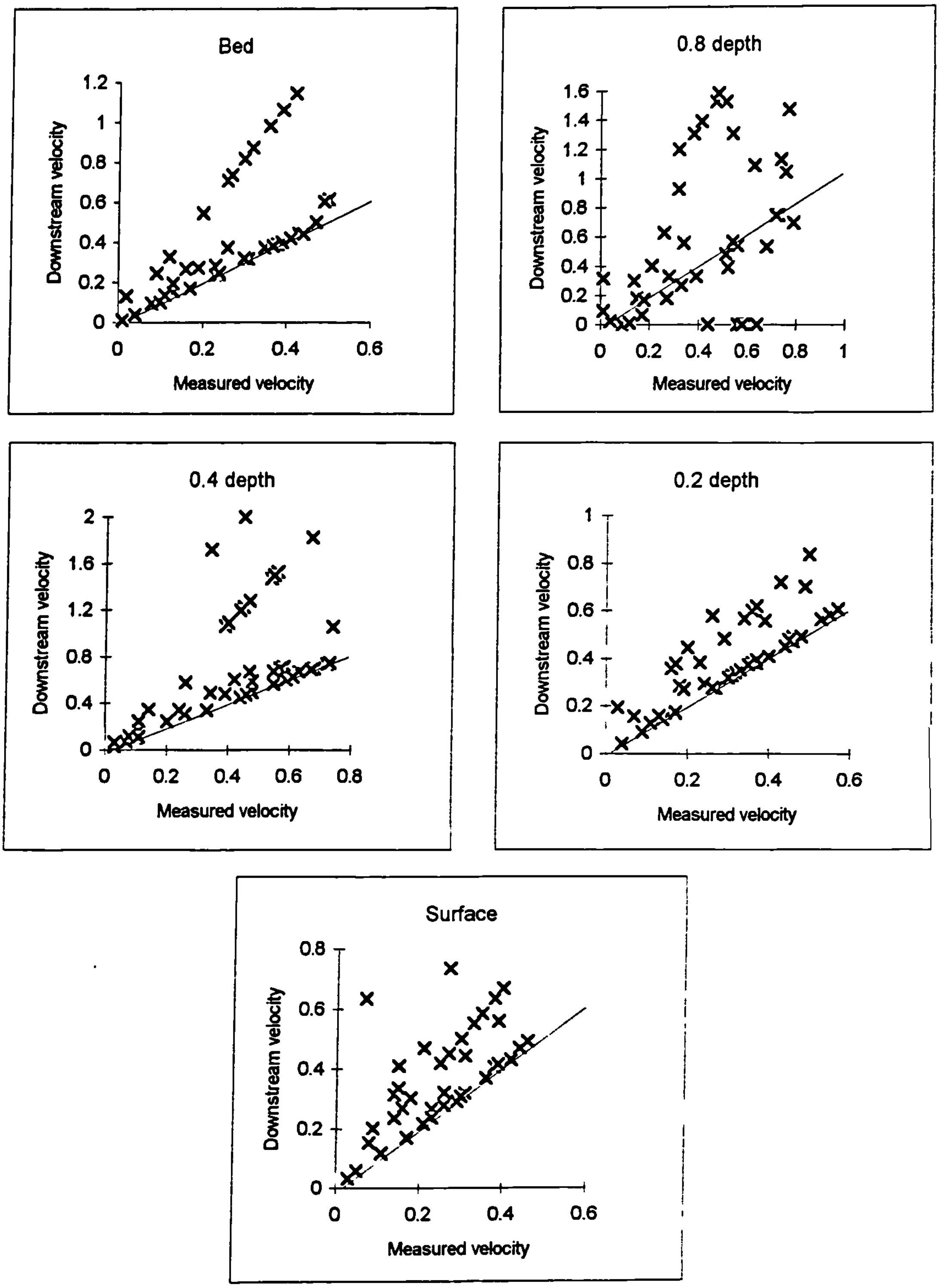


Figure 6.10: Comparison of measured and 'downstream' velocities (predicted using the cosine formula) at depths in verticals within a boil.

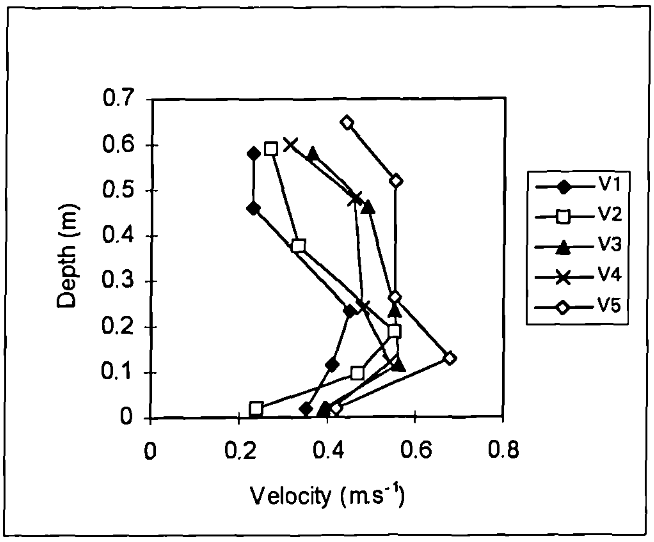


Figure 6.11: Velocity profiles in a low flow boil, measured in the direction of dominant flow.

Summarising the findings of velocity measurements in these 'difficult' biotopes, i.e. extremely rough or vegetated ones and in *boils*, it has been shown that:

1. Velocities measured at '0.6 depth' are frequently a poor guide to the mean column velocity.
2. The assumption of a logarithmic profile in the calculation of indices derived from 'mean' velocity is a *necessarily* inaccurate component of the work presented here (and in most other work on habitat hydraulics, including PHABSIM).
3. There appears to be little hope of applying correction factors to measured velocities without a separate and extensive research programme, using specially designed equipment (e.g. Young, 1992 used a laser-Doppler velocimeter, but in a laboratory flume). Statzner hemispheres (Statzner and Müller, 1989) do not address the column velocity.

Paradoxically, a useful guide to 'true' velocity distributions may well be provided by distributions of organisms, indicating that further research on flow types and velocity distributions must be interdisciplinary (see section 7.6).

6.3 ECOLOGICAL RELEVANCE OF HYDRAULIC CHARACTERISATIONS

Paradoxically, having cast doubt on the validity of velocity measurements at 0.6 depth as indicative of the mean column velocity (in two biotopes and during periods of macrophyte growth) we now need to use these velocity measurements to calculate the shear velocity on the channel bed - as a more refined indicator of hydraulic habitat for benthic organisms. Further elaboration of 'true' velocity profile models is beyond the scope of this thesis, and whilst the shear velocity formula does assume the von Karman model profile the introduction of depth and substrate variables reduces the sensitivity of the velocity term. Further work on the 'true' hydraulic conditions existing near the bed of hydraulically complex biotopes must await field studies of the biological significance of the broad flow-type categorisation of biotopes presented here.

From an ecological perspective it is the hydraulics of, and types of biotope that influence the spatial distribution of biotic communities, rather than discharge *per se*. Biotopes have been shown to be significantly different in terms of simple and complex hydraulic indices (see section 4.3). Consideration of combinations of depth, velocity and substrate, rather than any isolated variable, is recommended by freshwater ecologists (Allen, 1951; Thorup, 1966; Mackay, 1969; Barmuta, 1989), and is the basis for PHABSIM predictions of weighted useable area (WUA) (Johnson *et al.*, 1994). However, a single discharge-dependent value of WUA inappropriately assumes equal and independent influence of the three microhabitat variables (Shirvell and Dungey, 1983; Mathur *et al.*, 1985; Morantz *et al.*, 1987). Disaggregated descriptions of biotope hydraulics may provide more *ecologically relevant* characterisations: depth and mean column velocity being of importance to fish habitat suitability (Binns and Eiserman, 1979; Krog and Hermansen, 1985). For invertebrate studies the microhabitat variables (depth, velocity and substrate) should not be treated separately (Brown and Brown, 1984; Nowell and Juwars, 1984; Huryn and Wallace, 1987; Quinn and Hickey, 1994); shear velocity integrates these microhabitat variables and is a good indicator of 'potential invertebrate habitat' (Davis and Barmuta, 1989; Carling, 1992a; Young, 1992).

Determination of an appropriate scale of study and ecologically relevant variables appears to be especially problematic to ecologists; there is no consensus of sampling strategy with respect to either physical habitat units or appropriate microhabitat variables. With fish habitat modelling, the relative suitability of the microhabitat variables depth and velocity is species and discharge dependent e.g. the increased suitability provided by depth as discharge increases result in less favourable velocities, although the calculated WUA may remain relatively constant (Heggenes *et al.*, 1994). It would be more pertinent to establish, where appropriate, 'threshold' or critical depths and velocities. Depth is known to be more important in small streams (Heggenes 1988b; 1996); with increasing body size trout use deeper areas which offer more space and cover (Wesche *et al.*, 1987; Heggenes, 1988b, 1996). The transferability of habitat suitability curves remains, as yet, dubious. Until general habitat suitability curves have been developed for a range of channel types and biotopes (one of the UK PHABSIM User Forum's R&D priorities), it may be judicious to maximise variability of depths and velocities to accommodate different life-stages. Moreover, depth, velocity and substrate size are not often limiting factors for fish (Heggenes and Saltveit, 1990), except perhaps in spawning redds (Crisp and Carling, 1989); in general, cover is considered to be of equal importance as microhabitat variables (Fausch and White, 1981; Wesche *et al.*, 1987). For this reason, disaggregated depth, velocity and substrate distributions are not presented in this thesis. Instead, maintenance of a diversity of biotopes with varied hydraulic conditions would improve the overall hydraulic habitat for fish and ensure patches with a range of velocities existed; brown trout have been shown to exhibit a narrow tolerance range for snout velocities (Morantz *et al.*, 1987; Heggenes, 1996). Provided a diversity of biotopes exist fish will adapt to natural variation in hydraulic conditions.

The selection of study sites and physical biotopes for this thesis was driven by the need for an inventory of standard, easily recognised instream physical habitat features which relate to mesoscale, geomorphological features. It was not an initial objective to measure physical habitat variables at a scale to which biota are sensitive. Therefore, the PHABSIM data collection procedure (Johnson *et al.*, 1994) was adopted, which does not directly measure near-bed hydraulics. However, if biotopes are to have ecological, as well as geomorphological credibility, their microhabitat hydraulic characteristics must be examined. Jowett (1992) states that the abundance of insects is often related to Froude number, as this index most successfully discriminates between riffles, runs and pools. However it is unlikely that Froude numbers *per se* are likely to influence benthos; calculations of Froude number are based on 'mean' (0.6 depth) velocities (Davis and Barmuta, 1989), were developed to distinguish between surface flow types, and are subject to error in channels where a logarithmic profile does not exist. Moreover, hydraulic descriptions of biotopes based on velocity readings taken at 0.6 depth may not directly determine biotic assemblages; such values provide a means of comparing gross hydraulics, but are of limited ecological relevance as the micro-distribution of stream benthos is influenced by *near-bed* hydraulics (Statzner *et al.*, 1988; Davis and Barmuta, 1989; Carling, 1992). Wadeson (1995b) makes the distinction between macro- and micro-flow hydraulic indices; the latter are more relevant to invertebrates. Davis and Barmuta (1989) identify five 'near-bed' flow types and propose the classification of flows as 'chaotic' or 'non-chaotic', in order to have ecological relevance. In general, it is concluded that complex hydraulic variables relating to near-bed conditions are potentially useful predictors of benthic invertebrate distributions (Statzner *et al.*, 1988; Young, 1992; Quinn and Hickey, 1994).

Young (1993) advocates a classification of near-bed flows to explain benthic invertebrate distributions, as an alternative to sampling at the riffle-pool scale (see section 2.1.1). In earlier studies, the author argues that ecological studies which focus on sampling at the riffle-pool scale, often overlook small scale variation in microhabitat variables at the channel bed; which have greater importance to 'patchiness' and trends in benthic invertebrate distributions (Young, 1992). The fact that biotopes (pool, riffle and run) were *"loosely defined"* in Young's study (i.e. without formal reference to flow type) has probably led to their inaccurate classification, with obvious implications for biotope-invertebrate relationships. With the adoption of the standard identification of biotopes based on flow types (section 3.5), and statistical validation of their discrete gross hydraulic characteristics (sections 4.3-4.6), we may infer that near-bed, microhabitat conditions may be characteristic of a particular biotope type, even if the detailed hydraulics of the near-bed zone (and their biological significance) remain in doubt. As relationships exist between benthic invertebrate distributions and bed hydraulic variables (shear stress and boundary Reynolds number) under baseflow conditions (Quinn and Hickey, 1994), we may be optimistic that flow types will have predictive power for invertebrate distributions.

The indices used to represent near-bed hydraulics and explain or predict invertebrate distributions (Statzner *et al.*, 1988) involve detailed calculations which combine kinematic viscosity, mean velocity, depth and substrate roughness, or estimates using shear stress hemispheres which are expensive to calibrate (Statzner and Müller, 1989). If the classification of biotopes based on flow types is related to distributions of near-bed hydraulic indices, then this visual classification may be used as an indicator, or surrogate, of (ecologically relevant) near-bed conditions. Links between indices calculated from mean (0.6 depth) velocities and near-bed hydraulic indices have been made by previous researchers; Froude number distributions (the most successful, single hydraulic index to distinguish between flow types - see section 4.3) have been shown to relate closely to those of shear stress (Robertson *et al.*, 1995). Thus it follows that biotopes defined by flow type are likely to show strong correlation with shear velocity, and therefore, invertebrate distributions. From an ecological perspective, shear stress is an index of the energy required by organisms to withstand flow (Statzner and Borchardt, 1992). Shear velocity is similar to shear stress, calculated from individual readings of depth, velocity and substrate rather than estimates of water surface slope (Statzner *et al.*, 1988). It is a key influence on the erosion of bed sediment (Dingman, 1984), is especially important to epibenthic 'microcrustaceans' which are vulnerable to erosion and displacement (Palmer, 1992, Richardson, 1992) and is considered to be central to the patch dynamics view of lotic communities (Statzner *et al.*, 1991).

Shear velocity is calculated rather than shear stress, to take advantage of the cell level data collection which allows near-bed hydraulic indices to be related to flow types. Calculations are based on velocity readings taken at 0.6 depth, using the formula given in section 3.6.2, after Wadeson (1995b), from Statzner *et al.*, 1988. Shear velocity distributions are illustrated in Figure 6.12 (p169-173). Where possible the same scale has been used for all calibration discharges at a site to allow easier visual comparisons of the range and distribution of shear velocities. Shear velocity distribution graphs were drawn in Excel, which requires a grid-style data format. Exposed gravel, point or lateral bars were entered as having zero shear velocity; these are included in the lowest class on the distribution diagram, to give an indication of relative changes in wetted width with discharge.

6.3.1 Descriptions of shear velocities by site and discharge

Shear velocities were plotted for all sites except two: the downstream study section at the Till site which was dominated by a *boil* and velocities were measured in the direction of dominant flow in only one of the three calibration sites. For reasons discussed in section 6.3.3 it was considered to be inappropriate to calculate shear velocities from measurements at 0.6 depth in a direction perpendicular to the cross-section. The deep (>3m) *glide* at the South Tyne site was also omitted, on the basis that shear velocities would change very little with discharge at such depths, and moreover, that biota would not be sensitive to such small differences. Maps are presented for five sites with different substrate size distributions.

Shear velocity distributions are closely related to biotope type with cascades and rapids having the fastest recorded shear velocities. It should be borne in mind that these values are based on calculations which assume logarithmic velocity profile, so values are only relative, not absolute. However, results are consistent with early empirical work, where shear velocity was found to be approximately 10 percent of mean velocity in rocky streams, but only 3 percent in fine sediment streams (Smith, 1975). The fact that the fastest and greatest range of shear velocities are measured in rapid, cascade and riffle biotopes may explain the high invertebrate diversities which are commonly associated with these biotopes (Wright, *et al.*, 1984; Brown and Brussock, 1991). As Petts *et al.* (1995) note, the influence of flow on biota is related more to the *range* of hydraulic conditions than any hydraulic parameter *per se.* In the case of stonefly *(Dinocras cephalotes)* and mayfly larvae *(Baetis rhodani)* smaller, i.e. emergent, larvae prefer high shear stresses (Statzner and Borchardt, 1992). There is a continuum of shear velocities in *runs* and *glides* which suggests perhaps that invertebrate assemblages show some overlap between these two biotopes. *Glides, pools* and *deadwaters* form a third group which are associated with low shear velocities; again there is no discrete shear velocity distribution associated with any one of theses three biotopes, which may suggest some overlap in invertebrate distributions.

In general shear velocities increase with discharge, with the exception of the North Tyne site. Here the fastest shear velocities occur at the 'mid' calibration flow, then decrease at higher discharges. This appears to support the velocity-reversal hypothesis for riffle -pool sequences (Carling, 1992b). The floods of January and February 1995 also appear to increase the range of shear velocities, as indicated in Figure 6.13. The Stanhope and South Tyne sites provide good examples of this; at the former, discharges are equivalent for the pre- and post-flood 'low' flow, but the range of shear velocities is much greater in the post-flood cascade. At the Lambley site, the deeper narrower channel produced by incision after the flood leads to a greater range of shear velocities; again at a discharge very similar to the pre-flood 'low' flow. These findings are consistent with the notion that floods are regulators of biotic communities, as discussed in section 7.1.3.

6.3.2 Links between flow type and shear velocity

In order to explore the relationships between flow types (which define biotopes) and shear velocities, distributions of the latter were calculated in SAS for all flow types in the cell-level dataset. An analysis of variance was performed to test the null hypothesis (Ho) that the distribution of shear velocities is independent of flow type. Results are significant at the 0.0001 level, which supports rejection of the null hypothesis and suggests that a relationship exists between flow type and shear velocity. In order to determine which flow types have unique, and which have similar shear velocity distributions, a Scheffé test was performed. Results are summarised in Table 6.3.

Flow type	**SPF**	**SRip**	**SBT**	**Rip**	**USW**	**BSW**	**Ch**	**UW**
SPF	-	✓	x	x	x	x	x	x
SRip	✓	-	x	x	x	x	x	x
SBT	x	x	-	x	x	x	x	x
Rip	x	x	x	-	x	x	✓	x
USW	x	x	x	x	-	x	x	x
BSW	x	x	x	x	✓	-	✓	x
Ch	x	x	x	✓	x	✓	-	x
UW	x	x	x	x	x	x	x	-

Table 6.3: Flow types with discrete and similar shear velocity distributions (a tick indicates a statistically similar ($p<0.0001$) distribution; a cross indicates flow types with dissimilar shear velocities). Abbreviation as listed in Table 3.6.

Results support the separation of rippled flow into shallow and deep (see section 4.3). In terms of shear velocities, and therefore hydraulic conditions for benthic invertebrates, shallow rippled flow provides a similar near-bed environment to scarcely perceptible flow, despite surface turbulence. It is possibly that depths are sufficiently shallow to allow the formation of a boundary layer (Ambuhl, 1959). Near-bed hydraulics appear to be similar in chute flow, broken standing waves and unbroken standing waves i.e. *riffle*, *cascade* and *rapid* biotopes. From an ecological perspective, this may mean that invertebrate distributions are similar in these biotopes, if indeed they are influenced primarily by hydraulic conditions at the bed, and if calculations of shear velocity based on readings at 0.6 depth are accurate. However, studies which compare predicted and actual near-bed hydraulic indices (shear velocity and boundary Reynolds numbers) indicate that calculated values slightly underestimate measured indices in rough boundary conditions (Quinn and Hickey, 1994; Beebe, 1996). *Cascades* and *rapids* are therefore likely to experience greater shear velocities than those illustrated in Figure 6.1. Chute flow has a similar shear velocity distribution to rippled flow based on calculations of velocity at 0.6 depth; however the small depths which characterise chute flow (usually less than 0.1 m) are likely to prevent the formation of a logarithmic velocity profile. Velocities are likely to be similar throughout the small depth of flow over boulders, leading to higher associated shear velocities than those calculated using the formula given in section 3.6.2. Moreover, it is likely that factors other than *near-bed* hydraulic conditions are the dominant influence on distributions of certain species, e.g. the caddisfly *Brachycentrus occidentalis* is strongly associated with near-critical and chaotic flow at convergent zones over the top of cobbles and boulders (Wetmore *et al.*, 1990); the ecological significance of chute flow to one species of *Simulium* (blackfly) larvae has been noted (Palmer, 1991).

6.4 SUMMARY

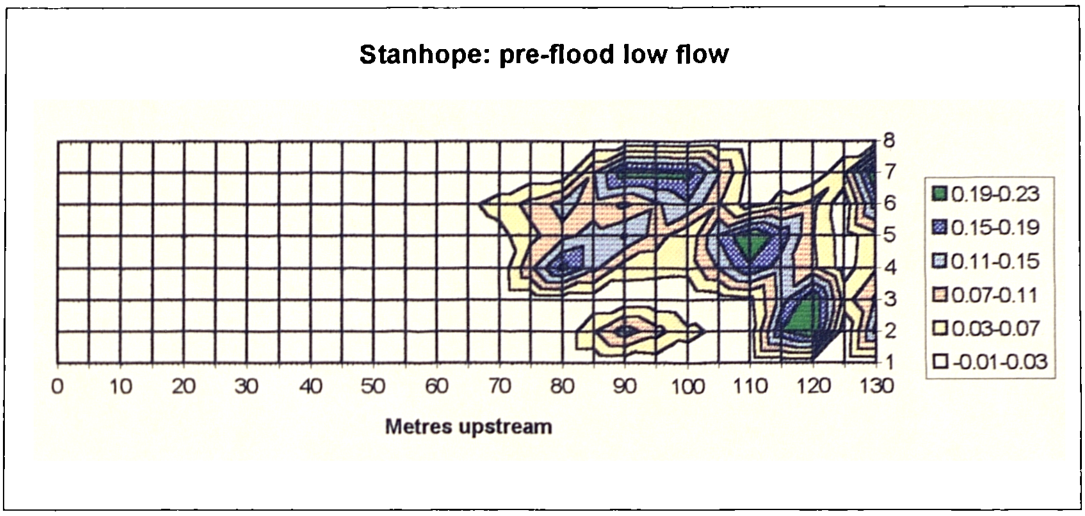
The limitations of hydraulic characterisations based on velocity readings at 0.6 depth have been illustrated for hydraulically complex biotopes. These findings question the general application of PHABSIM in the prediction of available habitat with fluctuating discharge. In channels with irregular beds, vegetation or upwelling flow, PHABSIM simulations are likely to be inaccurate, necessitating a biotope approach to discharge-related changes in hydraulic conditions. Three points may be made in summary of this chapter:

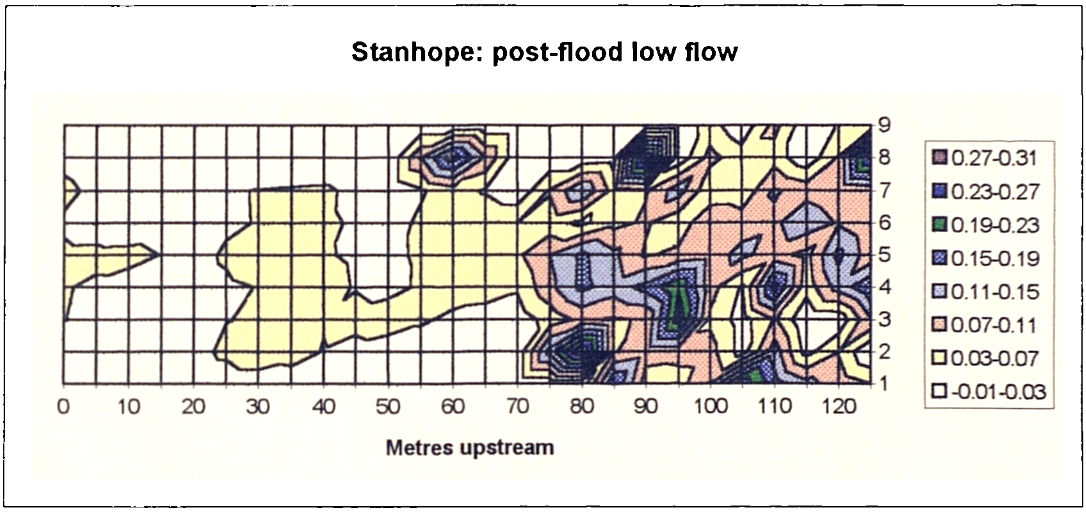
1. In natural river channels the hydraulic conditions are very complex (as indicated by the range of velocity profiles), and can only be approximated by simple models based on velocities at 0.6 depth.
2. In order to make generalisations we need to make simple assumptions. Rapid, cascade and boil biotopes and channels dominated by macrophyte vegetation are the major deviants; in the remaining biotopes velocities at 0.6 depth may be taken as representative of mean values.

3. The Froude number and shear velocity assume average velocities at 0.6 depth and a logarithmic profile, yet they provide practicable and apparently successful indices for the comparison of flow types and near-bed hydraulic conditions respectively.

Financial and human resource constraints make it unlikely that full velocity profiles will be done in practical studies of habitat hydraulics. Increasingly, practical studies will be based on limited fieldwork, which emphasises the need for mesoscale biotopes and River Habitat Survey as intermediate, readily applied technology. The ecological relevance of physical biotopes needs to be developed further to remove the need for continued debate as to the most appropriate indicators of near-bed hydraulic conditions. Knowledge of associated invertebrate and fish communities would provide an indication of those biotopes with high conservation value.

The application of visualisation techniques appears to be a developing field, as described by several authors at the second International Habitat Hydraulics Conference in Quebec, June 1996. A general overview of these techniques is provided by Hardy (1996). The use of Geographical Information Systems (GIS) to link biotic and hydraulic data by means of overlays are described (Muotka, 1996). It is hoped these GIS will enhance the understanding of spatial heterogeneity and scale selection in lotic communities. Bovee (1996) describes how spatially explicit habitat models can be applied to test the importance of habitat juxtaposition to the patch dynamics theory. Cell-level data collection has been used in PHABSIM to calculate composite habitat suitability (Gore *et al.*, 1996). These authors show how cell-by-cell calculations improve the success of predictions of available benthic habitat compared to observed increases in habitat area following the introduction of artificial riffles as part of a restoration project.


Stanhope: pre-flood low flow


Stanhope: post-flood low flow

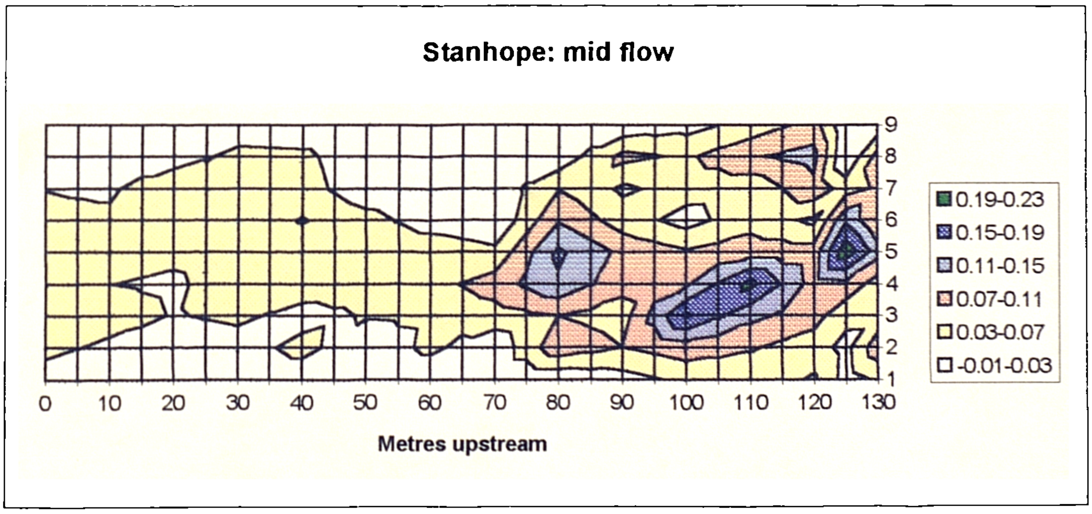
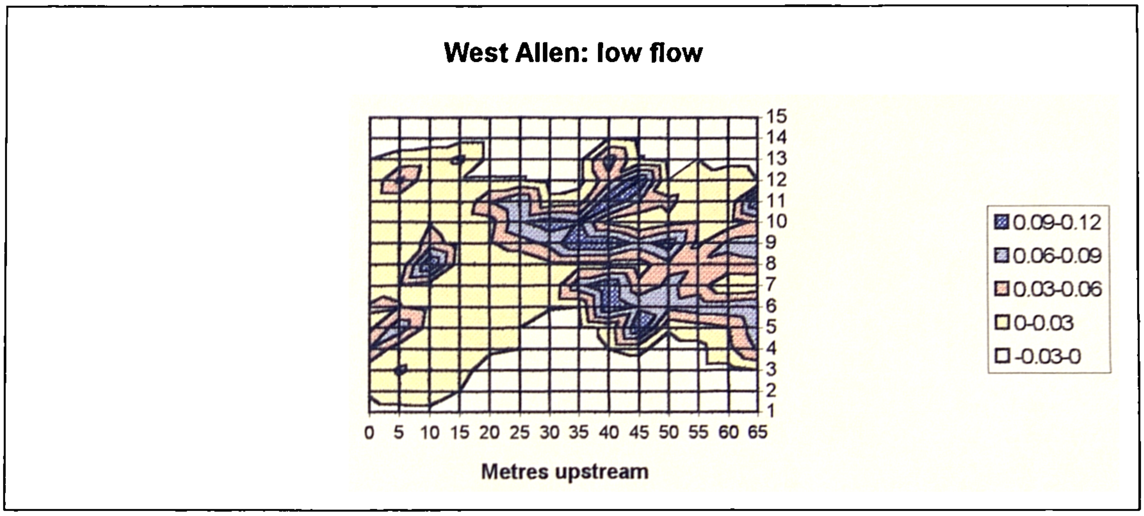
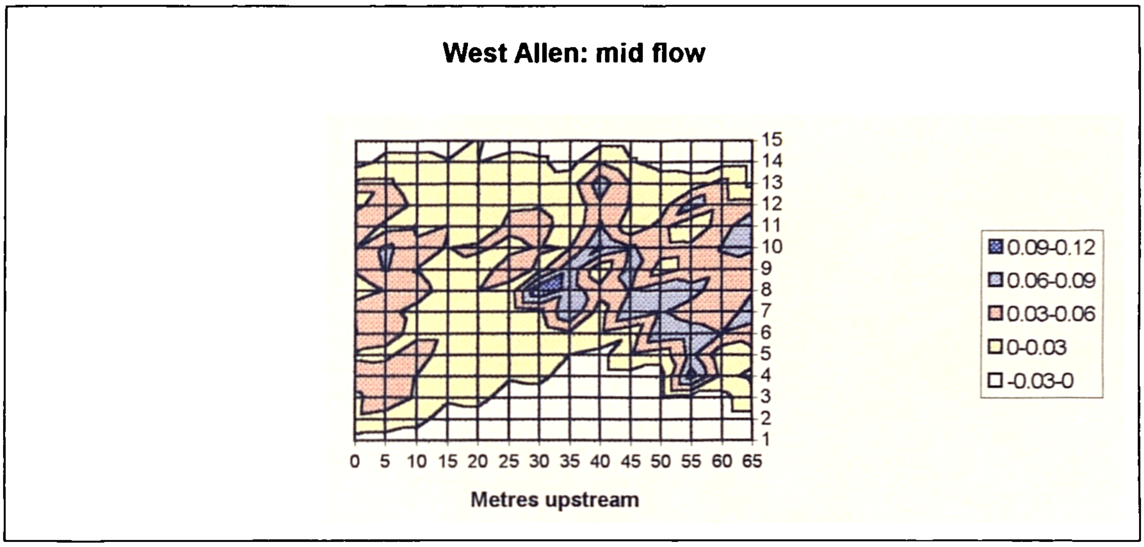

Stanhope: mid flow

Figure 6.12: Shear velocity distributions at the study sites under three calibration flows.





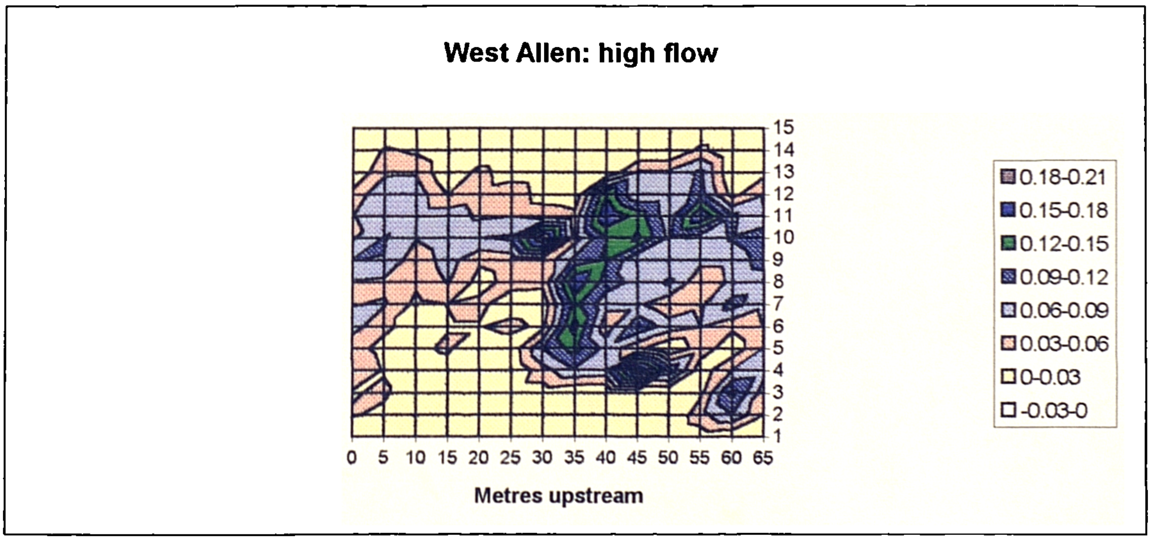
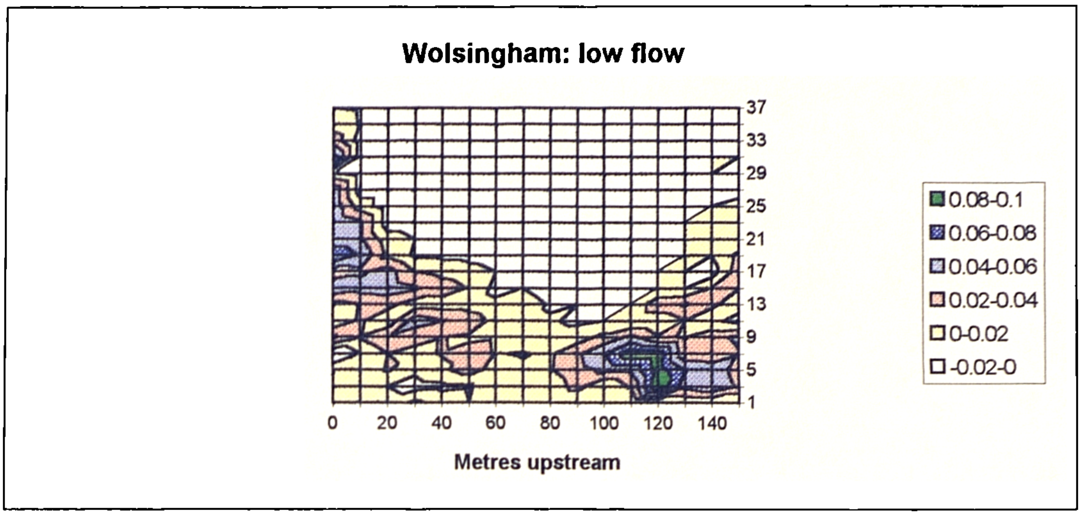
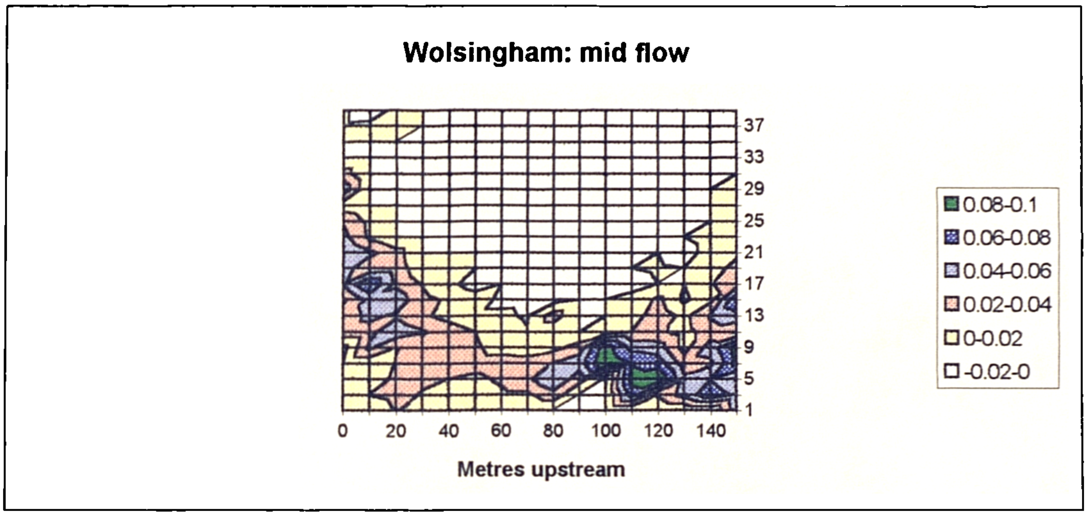
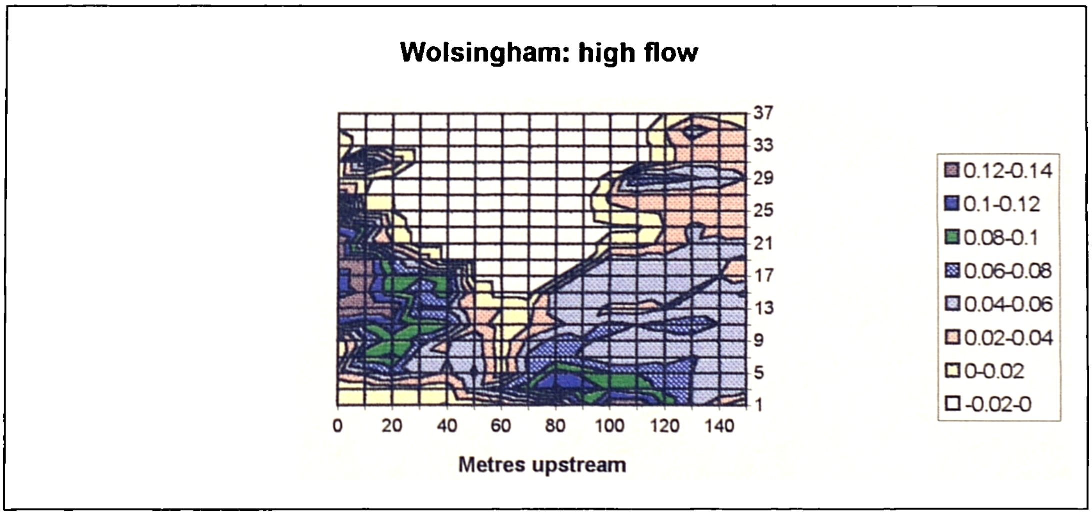


Figure 6.12 continued.


Wolsingham: low flow


Wolsingham: mid flow


Wolsingham: high flow

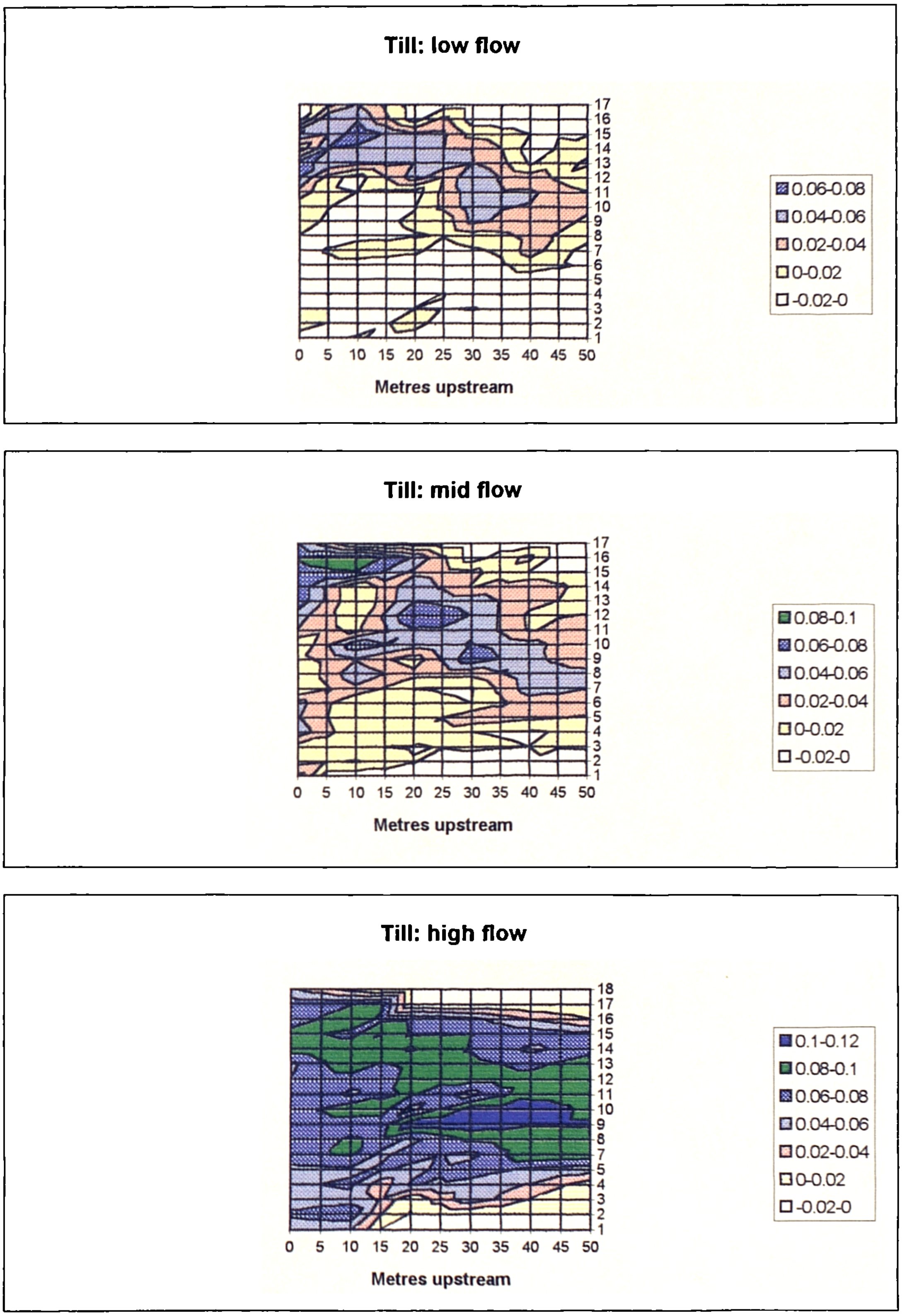
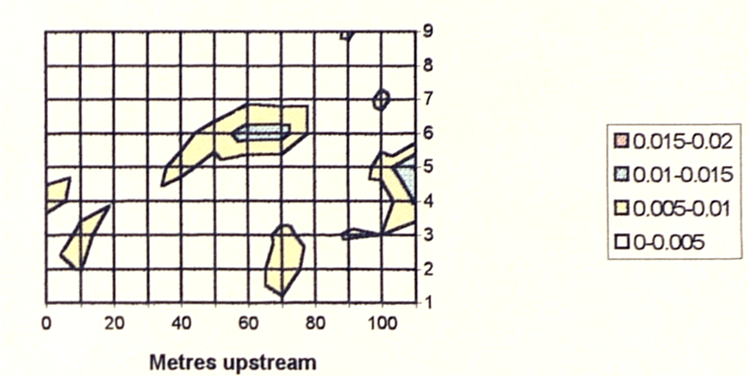
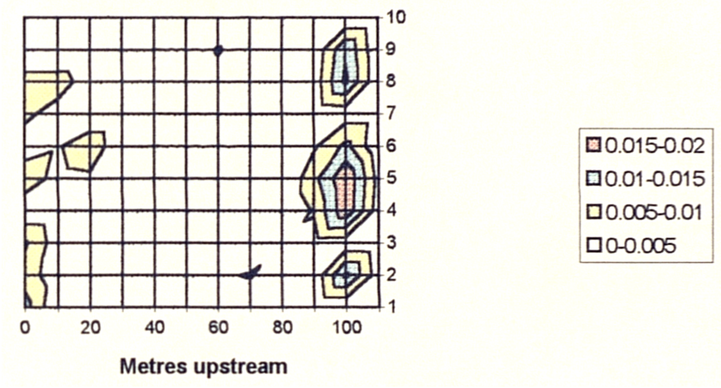


Figure 6.12 continued.

Skerne: low flow



Skerne: mid flow



Skerne: high flow

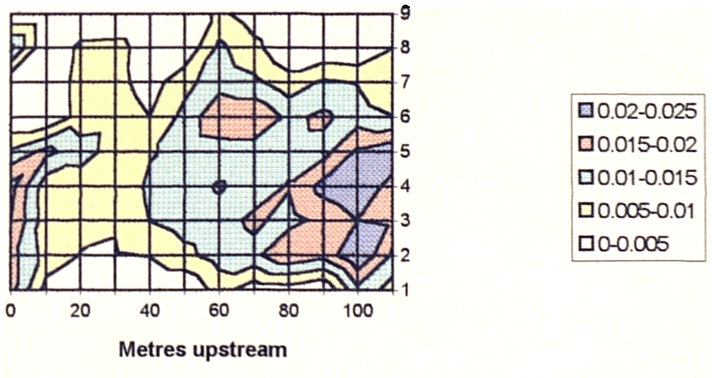


Figure 6.12 continued.

7. APPLICATIONS TO THEORY AND MANAGEMENT

Chapter Overview

This chapter begins by discussing the results of the study within their theoretical context. The possible ecological relevance of physical biotopes across a range of discharges is discussed in the context of stream ecosystem theories, namely patch dynamics and the River Continuum Concepts. The impact of flood flows on biotope patchiness and likely impact on instream biota is also discussed (7.1). The applications of physical biotopes to practical river management are described, with particular emphasis on the Rivers Skerne and Cole, example sites in the River Restoration Project (7.2). The existence of characteristic biotope sequences suggests that flows can be manipulated to maintain biotope diversity. Flow management is considered from a perspective of both critical and representative biotopes (7.3). The chapter concludes with a discussion of the methods by which physical biotopes may be applied to catchment scale models of physical habitat, based on national typologies. It discusses ways of scaling and extrapolating the site-specific results presented in this thesis to the reach scale (7.4).

7.1 RELEVANCE TO STREAM ECOSYSTEM THEORIES

The work for this thesis proceeded within the context described in section 1.5 (p18). The empirical framework for testing theories of patch dynamics and the river continuum concept has been provided by the development of national scale, physical habitat surveys, notably RHS. This chapter now returns to ecological theory, placing the findings of the thesis in a context which has relevance to management for the maintenance of ecosystem diversity.

7.1.1 Patch dynamics: channel type and spatial variation in 'patchiness'

The theory of patch dynamics has both spatial and temporal dimensions, being related to both physical habitat diversity at a particular discharge, and the range of hydraulic conditions which occur over the flow regime. The work described in this thesis can be directly applied to test the patch dynamics theory of river ecosystem structure and functioning. In this section the spatial aspect will be considered, in relation to the range of geomorphological channel types selected as study sites. The empirical results regarding spatial 'patchiness' and 'diversity' are described in detail in section 5.2. To provide a theoretical framework for their discussion, the patch dynamics literature was reviewed.

The patch dynamics theory states that species richness is predicted to increase with spatial heterogeneity (Townsend, 1989). Empirical findings in support of this theory have been provided by Marshall and Westlake (1990) and Armitage (1995), who state that community diversity can be maintained even at low discharges provided that habitat heterogeneity exists. The physical biotopes identified throughout the course of this study are likely to provide a range of habitats which are suitable for different organisms and life-stages. To recap, *riffles* are associated with high invertebrate densities making them good feeding sites for fish (Beschta and Platts, 1986; Crisp, 1993); their use as spawning sites has been widely documented, (Crisp and Carling, 1989; Crisp, 1993). *Marginal deadwaters* are important shelter sites for adult fish; the silt substrate often found in marginal deadwaters allows macrophytes to become established which provides further cover for young-of-year fish (Lobb and Orth, 1991; Bovee *et al*., 1994). *Shallow runs* are found at the margins of riffles and deadwaters. They provide important habitat unit for juvenile fish (Copp, 1989; Rabeni and Jacobson, 1993), and act as 'connecting' biotopes in very low flows by preventing deadwaters from becoming isolated or dewatered. If sufficiently deep, *runs* and *glides* provide shelter from predators, together with feeding or migratory habitat for adult fish. Clearly the more biotopes present, the more species and life-stages the channel will be able to support (given that nutritional and water quality requirements are fulfilled).

Biotope 'diversity' is clearly related to geomorphological channel type as described in section 5.2. Huryn and Wallace (1987) conclude that the functional structure of macrofauna is the result of the relative contribution of the habitat types riffle, pool and bedrock outcrop. Habitat mapping in PHABSIM may be carried out to account for the different proportions of habitat type, which alter the overall calculation of WUA (Johnson *et al*., 1994). The presence or size of individual habitat units appears to be less important than their juxtaposition (Rabeni and Jacobson, 1993). These authors describe more frequent habitat use in *runs* and mid-channel *pools* which are adjacent to marginal pools. The presence of *deadwaters* adjacent to *riffles* is important for emergent fry, which require areas with relatively low velocity and shelter from predators (Snyder, 1990; Scheidegger and Bain, 1995). Salmonids often select headwater *riffles* in favour of those further down the system which have better quality spawning gravels, but poorer quality rearing habitat (Beschta and Platts, 1986). For mobile species the juxtaposition of adjacent biotopes is probably of less importance than the presence of a range of biotopes within a reach.

In the patch dynamics literature, a patch is defined as a spatial unit determined by the organism or problem in question (Pringle *et al.*, 1988). The definition of **biotope 'patchiness'** adopted in this study (section 5.2) refers only to the *number* of units, regardless of their type. The concept of **biotope 'diversity'** considers both the number *and* types of biotopes, and is therefore likely to have greater ecological relevance than biotope patchiness. However, few empirical studies have compared biotope 'patchiness' with 'diversity'. Links between biotopes and fish have been widely documented (see section 2.3) although the term is not widely adopted. For example, Bain and Knight (1996) identify five fish assemblages associated with "habitat types" defined on the basis of combinations of current speed, depth, cover and substrate types. These correspond to biotopes identified in this study, but are not formally recognised.

The distribution of **secondary biotopes** is essential to an understanding of spatial patch dynamics. As Rabeni and Jacobson (1993) observe, the boundaries between biotopes vary less significantly in a longitudinal direction than across the channel at different flows. Gorman and Karr (1978) conclude that fish community characteristics result from the horizontal habitat heterogeneity. Secondary biotope mapping at the study sites is described in section 5.5. In upland channels of the study sites (e.g. Harwood Beck, Kielder Burn), low velocity regions were present in the wake of large boulders. These allow larger-bodied, poorly streamlined animals to co-exist with species tolerant of high shear stresses in turbulent flow (Barmuta, 1989). In mid-gradient cobble and gravel bed channels (e.g. North Tyne, Wolsingham), low velocity regions occur upstream of tree roots and other channel obstructions, or where vegetation is present. Even in lowland channels which have relatively uniform cross-sectional topography compared to upland boulder-bed channels (e.g. Ouseburn, Derwent), secondary biotopes are present (see Figure 5.9, p132). Discharge-related variation is particularly important in low gradient, fine gravel channels which would otherwise be relatively uniform (Armitage, 1995). Regulated rivers show less horizontal variation in biotope types (see section 5.5). Over an evolutionary timescale the reduction in flow range and biotope patchiness may have implications for invertebrate communities associated with marginal biotopes.

7.1.2 Discharge, refugia and disturbance: temporal variation in 'patchiness'

Hildrew and Giller (1994) claim that flow forces *"are undoubtedly the major architects of physical patchiness in streams"*. This differs from the concept of a "shifting mosaic" (Bormann and Likens, 1979), which assumes a uniformity of biotope types and area, but with a different spatial distribution. Temporal change in biotope 'patchiness' is dependent upon the interactions between morphological units and discharge, and the range of hydraulic conditions which exist within and between different physical biotopes. Results of biotope mapping (section 5.3) have illustrated how biotopes at a particular location in a channel alter temporally with discharge. For example, a riffle morphological unit at moderate flows will be a *cascade* biotope; at low flows this unit becomes part *riffle*, part *cascade*. At even lower flows 'patchiness' increases further as *riffle*, *shallow run* and *deadwater* biotopes are present; at very low flows *deadwaters* and may become isolated from the main channel (see section 5.3.1). At high flows the boundaries between biotopes are less distinct, reducing biotope 'patchiness' and 'diversity'. This is consistent with invertebrate studies; Barmuta (1989) observed discrete riffle fauna over a range of discharges throughout a year, whereas *pool* and *run* biotopes became faunally similar at higher flows in winter months.

Links between biotope patchiness, discharge and biota are outlined by Townsend and Hildrew (1994). They predict that biotic populations in habitats with more spatial heterogeneity will be less affected by temporal variations due to discharge. Where biotope heterogeneity exists biota will be less affected by changes in discharge, being able to move to an adjacent biotope with appropriate hydraulic conditions. This definition assumes biotope 'diversity' (as defined in section 5.2) exists; because at any one site, different biotopes will, by definition, have discrete hydraulic conditions. Studies by Gibbins (1996) on the effects of reservoir releases on *Micronectar* spp. (an invertebrate associated with marginal deadwaters and known to be susceptible to disturbance) support this theory; his experiments revealed that margins in alluvial areas (which were associated with substrate heterogeneity) did not show significant reductions in invertebrate numbers, whereas the invertebrate fauna within bedrock margins (uniform bedrock substrate) were 'washed away'. At sites with more spatial (and thus hydraulic) heterogeneity, biota may be morphologically or physiologically adapted to changing hydraulic conditions (Vogel, 1981), and be able to withstand discharge-related changes at a particular site (King *et al.*, 1989).

Spates influence the hydraulic characteristics of patches differentially; those least affected by changes in discharge are termed flow 'refugia' (Lancaster and Hildrew, 1993a; 1993b). The emphasis on riffle sampling in invertebrate ecology studies has possibly masked temporal (discharge-related) shifts in species composition, as *riffles* are the most stable morphological units (Sear, 1992; 1996), despite experiencing a range of hydraulic conditions across various discharges (see Figure 4.3, p99). Associated riffle biota will therefore have adapted to these hydraulic conditions. By comparison, other less hydraulically stable biotopes may show greater temporal fluctuation in biota; as Palmer *et al.* (1995) show, the ratio of copepods to chironimids shifted more during spates in refugium patches (debris dams with fine sediment) than non-refugium patches (mid-channel). Robertson *et al.* (1995) recorded relatively greater declines in species density in 'faster' reaches which showed greater variation in hydraulic conditions in spate flows.

The resistance or **resilience** of ecological systems to disturbance is attributable to the spatial heterogeneity of the stream (Pickett and White, 1985; Hildrew and Giller, 1994). The importance of refugia biotopes to system resilience in the event of hydrological changes has been emphasised (Petts and Maddock, 1994). Deadwaters ("slack waters") may be important refugia for biota during floods (Townsend, 1989). Townsend and Giller (1994) relate the ability of organisms to utilise refugia at higher flows to the spatial variation in biotopes (*"habitat heterogeneity"*). Two scales of physical habitat heterogeneity are identified; **inter-habitat** i.e. diversity of biotope types (mesoscale), and **intra-habitat** i.e. diversity of microscale variables including shear-stress, bed velocity and depth. Variation in the latter is described in section 6.3 and illustrated in Figure 6.12 (p169-173). Animals may fare better in some patches (refugia) than in others due to the persistence, during spates, of patches with low shear velocity (Townsend, 1989; King *et al.*, 1989). In this study, the maintenance of significant percentages of smooth boundary turbulent flow (*glides*) and scarcely perceptible flow (*deadwaters*) in the upland sites, and scarcely perceptible flow (*deadwaters)* in the lowland sites (both which are characterised by low shear velocity distributions) at high flows provides empirical evidence that refugia sites exist in most channel types (see Table 5.3).

Townsend (1989) lists the potential "slack-water" areas which provide refugia during floods for headwater channels, alternating incised/alluvial reaches and alluvial reaches. Refugia appear to promote system resilience; Palmer *et al.*, (1995) review several papers which state that stream invertebrate assemblages are highly resilient to spates due to the presence of refugia. Carling (1995) describes the importance of "dead-zone" environments as nursery environments or refugia against high flows. In this study lower energy biotopes e.g. *deadwaters* and *glides* existed at most sites at high discharges. One exception was the channelized section of the Ouseburn which has a low percentage of slow flow types at high discharges. By comparison, an area of scarcely perceptible flow is present at high flows in the channelized Skerne due to the presence of an overhanging willow tree. The identification of instream physical features and bankside vegetation (both which are recorded in RHS) may thus be used as indicators of high flow refugia sites.

Inter-habitat heterogeneity has been investigated for the study sites in sections 5.2 and 5.5 (biotope 'patchiness' and 'diversity'). Biotope diversity is important for mobile species e.g. fish move to deadwater zones at channel margins to avoid the adverse hydraulic conditions and shear stresses associated with the main channel (Harvey, 1987; Heggenes, 1988a; Jowett and Richardson, 1994). The importance to overall population numbers has been proven; Pearsons *et al.*, (1992) found that relatively fewer fish numbers were lost following flooding of hydraulically complex stream sections than in more simple channels. The existence of refuge biotopes is also essential when predators are present (Harvey, 1991). Under these conditions, smaller fish are found in shallower, vegetated areas which are 'less favoured' by large adults due to depth limitation and increased risk of predation from birds and mammals.

Townsend (1989) discusses the critical role of **'disturbance'** as a 'reset mechanism' which initiates changes in patch dynamics. He recognises spates as a form of disturbance by the removal of organisms and altered spatial distribution of substrates. This differs from the definitions used by White and Pickett (1985) and Resh *et al.*, (1988), who recognise that events only constitute disturbance if they are beyond 'normal' orders of magnitude. Milner (1994) considers both the magnitude of events and temporal duration of impacts, recognising system disturbance only when impacts are permanent. Geomorphologists distinguish between three types of floods: those which disturb biota but not sediment; those which move sediment; and threshold floods in which morphological units and therefore biotopes change type, location and possibly scale (Newson, 1992a). In the context of this study any event which brings about a change in the distribution of substrate and hydraulic conditions (and thus biotopes) qualifies as disturbance. There has been insufficient time to determine the long-term impacts of floods on flow types and biotope patchiness; only the immediate effects of discharge fluctuation and spates have been ascertained by biotope mapping (section 5.4).

Floods or high flows initiate a redistribution of 'patches' within the system (Cummins *et al.*, 1995), and may create patches or biotopes with different hydraulic conditions. This has been described for the study sites at the scale of morphological units in section 5.4. System resilience and species assemblages are influenced not only by the spatial biotope heterogeneity, but by the flow regime; with greater genetic variability being found in streams of high disturbance frequency (Robinson *et al.*, 1992). Those species possessing mechanisms to tolerate disturbance are more likely to be found in streams with a 'flashy' flow regime or high discharge range. If the system is not resilient to discharge fluctuations, **recovery** is the mechanism by which biotic populations adjust to the disturbance (Milner, 1994). Traditionally, recovery has been related to invertebrate assemblages (Minshall *et al.*, 1983; Gore and Milner, 1990; Niemi *et al.*, 1990). The influence of floods on biotope resilience and recovery (rather than species or community), is discussed in the following section.

7.1.3 High magnitude floods, biotope change and implications for habitat survey

The influence of floods upon instream habitat is recognised for its role in structuring riverine ecosystems (Poff and Ward, 1989), but the impact of rare, high magnitude floods on instream habitat across a range of channel types has not previously been considered. Ecological perspectives have concentrated on impacts on individual species or communities e.g. macrophytes (Brookes, 1986) and invertebrates (Minshall *et al.*, 1983; Niemi *et al.*, 1990). In morphological terms channel dimensions are shaped by dominant or formative discharges (Petts and Maddock, 1994), whilst floods less than bankfull may result in 'ecologically significant substratum movement' (Leopold *et al.*, 1964). Events of high magnitude may bring about long term channel change if geomorphic 'thresholds' are crossed (Newson, 1992a).

Floods, are believed to be key regulators of biotic populations (Jowett and Richardson 1994; Stalnaker *et al.*, 1996), by maintaining habitat structure and key life-stages. At lower magnitude floods the basic channel morphology is maintained, fine sediment is flushed from spawning gravels and fish migration is initiated (Reiser *et al.*, 1989; Milhous, 1996). Communities are adapted to floods which occur annually; these events are not considered to represent 'disturbance' to the natural system. By contrast, high magnitude floods present pulse disturbances to river systems (Bender *et al.*, 1984). Milner (1994) defines events of a 50 year return period or greater as sufficient to cause such disturbance. In the context of patch dynamics theory it is pertinent to determine whether flood related disturbance results in a spatial redistribution of biotopes, or a fundamental change to the characteristic biotope types and sequences. Davies (1978) describes how salmon occupy the same 'lies' unless "floods alter the bed of the river"; the subsequent movement of fish to an upstream or downstream pool suggests that floods initiate a redistribution of biotopes.

As a result of the January/February 1995 floods, channel change has occurred at all sites where discharge was sufficient to cross bed thresholds, and where sediment was available for transport. In bedrock reaches deposition of alluvial material resulted in channel narrowing, but the basic planform is not free to adjust, so the distribution of morphological units was unaltered. At most alluvial sites the cross-sectional morphology was altered, but the basic morphological units (which control the basic biotope sequence) remain. Erosion of channel features, for example point bars, is the main alteration to channel morphology. At the unstable, wandering channel at Lambley, erosion of an instream gravel island has led to a redistribution of biotopes. Locally, the proportion of *riffle* and *deadwater* biotopes has increased (see section 5.4) but this is likely to represent a short-term redistribution of biotope 'patches' rather than a change to the overall sediment budget and reach-scale distribution of biotopes. In channels with stable, well vegetated banks the impacts appear to be less severe e.g. at the Wolsingham site erosion of the point bar has had no significant impact on the biotope sequence. Channels with well developed floodplains experience geomorphologically effective events relatively frequently (Pickup and Warner, 1976; Wolman and Gerson, 1978), which explains why the 35 year event at the Wolsingham site has not altered the biotope types and sequence. By comparison the 25 year event at the headwater Harwood Beck site caused significant biotope change. **It appears that the sensitivity of channels to flood-induced biotope change is related to both magnitude of event and channel type.**

A detailed account of flood-induced biotope 'patchiness' is provided in section 5.4. At most sites flood events appear to have an immediate positive impact on instream habitat by increasing the range of biotope patchiness and diversity of hydraulic conditions. This supports the theory that floods regulate biological populations (Jowett and Richardson, 1994; Stalnaker *et al.*, 1996). Disturbance at the study sites is perceived as a reset mechanism as defined by Townsend (1989). The scale of flooding which occurred in January and February 1995 is likely to cause some immediate loss of invertebrates through scour and drift, but rates of invertebrate recovery are rapid, yet varied. From an ecological perspective, sites with high flood frequency are less biologically impacted by floods, and are predicted to have shorter recovery times (Townsend *et al.*, 1992; Milner, 1994). From a geomorphological perspective none of the study sites experienced geomorphological thresholds (Newson, 1992a), which would permanently alter the biotope types and sequences. This is partly due to the small flood frequency growth curves (i.e. low ratio of the 100:2 year event) in UK rivers (Newson, 1994). Additionally, many channels in England and Wales have been shown to be sensitive to relatively small changes in sediment supply and runoff, and adjust their size and shape rapidly and frequently (Lewin *et al.*, 1988). This is likely to have resulted in the evolution of biota which are adapted to a fluctuating flow and sediment regime.

It is highly unlikely that flows which occurred on 31 January 1995 would produce 'suitable habitat' in PHABSIM simulations of Weighted Useable Area versus discharge. However, all the sites affected possessed refugia sites i.e. areas with low shear velocity at relatively high discharges (see Figure 6.12). Whether these persisted at all sites under flood flows is uncertain (observations at the South Tyne and Wolsingham sites suggest they do); what is clear is that all sites showed equal or greater biotope patchiness and hydraulic diversity following the flood (see Figures 5.1 and 5.3). Shear velocity distributions are also altered in post-flood conditions (see Figure 6.12, p169 - Stanhope site). These findings have implications for future RHS surveys and upgrading of the existing national network. It would be prudent to resurvey any UK reference sites which experience large floods, to monitor and assess the impact of large events on biotope distribution. Recovery of the former biotope distribution would indicate a return of the pre-flood physical habitat, but it is recognised that effects on channel morphology and associated physical habitat may manifest themselves further down the system. Moreover, 'third-order' effects on biological populations may take longer to recover (Petts, 1988). Small-scale changes in biotope patchiness as described in section 5.4 are unlikely to be identified by RHS spot checks at 50m intervals, but larger scale change of mesoscale features may be detected with post-flood RHS.

Links between the findings of this research and patch dynamics theory have been described in the above sections. A summary of the empirical-theoretical links is provided in Table 7.1.

Theory (as it affects physical habitat)		**Relevant findings**
1. River Continuum Concept		Difficulty (RHS) in defining channel typology
2. Patch dynamics	- spatial aspects	Cell level analysis reveals potential habitats in biotopes. These mostly equal functional habitats where literature reports.
		Secondary biotopes revealed as common - lateral connectivity.
		Biotope sequences repeatable - longitudinal connectivity.
		Biotope 'diversity' related to channel type.
	- temporal aspects	Biotope mapping reveals characteristic changes with flow for each biotope and sequences of biotopes.
	- refugia	Confirmed by biotope cell approach - diversity of hydraulic patches ensures refugia.
		Low shear velocities persist in patches at high discharges.
	- disturbance/resilience	High magnitude floods mostly failed to change biotope types profoundly: some low-flow phenomena changed e.g. biotope distribution and absolute dimensions high-flow refugia (deadwaters) observed.
3. Threshold channel changes (geomorphology)		High magnitude floods which cross intransitive thresholds likely to change biotope patchiness/ diversity.
4. Sediment mobility (geomorphology)		Riffle-pool sequence has traditionally monopolised process studies- now worthy to investigate other biotopes e.g. runs/glides/deadwaters etc.

Table 7.1: Links between empirical findings from the 'biotope approach' and ecological/ geomorphological theories.

7.1.4 Links with recent ecological approaches to river habitats and biological sampling

The biotope approach allows the rival theories of RCC and patch dynamics to be empirically tested. It may be applied at a particular location to examine temporal changes in patch dynamics, or spatially within a catchment. If the spatial dimension of patch dynamics is being examined, trends within a catchment must also be considered in the context of the River Continuum Concept (RCC). The validation of physical biotopes as hydraulically discrete units provides a new framework for invertebrate sampling, in order to test the ecological significance of these biotopes (or 'patches') across a range of channel types. A recent study of invertebrate assemblages sampled according to biotopes at the eleven sites described in this thesis (Figure 3.1, p52) indicates a main division into upland and lowland sites (in accordance with the RCC), with flow types accounting for local scale variation in biota (Grundy, 1996). These results are consistent with national-scale invertebrate distributions predicted for RIVPACS, in which substrate is the single most important variable influencing biota, as it combines nutritional and water quality influences which relate to the RCC and local habitat variability (Petts, personal communication).

During the progress of the biotope study liaison has been maintained with parallel research with freshwater ecologists in lowland rivers of England and Wales. In these lowland rivers invertebrate assemblages have been related to 'mesohabitats' (Armitage *et al.*, in press) and 'functional habitats' (Harper and Smith, 1995; National Rivers Authority, 1996b) . In lowland channels which have less variation in substrate size, discharge and flow types (these channels are dominated by *glides* - National Rivers Authority, 1996a), greater emphasis is placed on smaller scale variation in substratum patches as mesohabitats, and on the role of vegetation types (alongside morphological units) as 'functional habitats'. The mesohabitat and functional habitat approaches clearly provide more detailed information on ecological distributions and can be directly used as indicators of habitat quality and recent fluvial events. They operate on a different spatial scale to physical biotopes, attempting to define empirically an appropriate scale for ecological sampling. By comparison the scale of physical biotopes is inherent (determined by field observation of flow types) and is only scaled up to the transect level because of the need to map instream physical features at a reach scale for national inventories. Physical biotopes have a potential ecological relevance, but can be more directly applied to assessments of flow management for ecosystems and physical habitats rather than biological communities. Table 7.2 compares the physical biotope approach described in this study with existing ecological sampling strategies.

	PHYSICAL BIOTOPES	MESOHABITATS	FUNCTIONAL HABITATS
River Types	Range of river types including upland coarse substrates (boulder, bedrock, cobble and gravel) and lowland (gravel and silt)	Lowland chalk and gravel	Lowland silt, clay and gravel
Key variables	Flow types (hydraulics) and substrates	Substrates and vegetation. No direct reference to hydraulics.	Vegetation and substrates
Approach	Scale determined by subjectively defined physical biotopes: work towards flow management and ecology from these units.	Attempt to assess an appropriate scale for ecological sampling by simultaneous biological and substrate sampling.	Attempt to assess the function of physically defined units by simultaneous sampling of biota and morphological units/vegetative structures.

Table 7.2: Comparison of physical biotope and river habitat sampling.

In time, a fusion of all three approaches: biotope, mesohabitat and functional habitat is clearly desired. It was the purpose of this study to define the range of physical biotopes which exist, and describe their hydraulic conditions across a range of flows. Ecologists may now determine which of these *'potential habitats'* represent discrete *'functional habitats'* (Harper *et al.*, 1992). A physical biotope may contain several 'functional habitats' depending on vegetation, nutritional and water quality influences. For example, Jenkins *et al.* (1984) revealed four discrete marginal habitats in terms of invertebrate distributions. Maddock and Bird (1996) advocate a distinction between shallow and deep glide for "mesohabitat types" in the River Tavy, Devon. Alternatively some biotopes types may represent the same 'functional habitat' and can be considered as one unit in terms of habitat quality. Much emphasis has been placed on the subject of 'functional habitats' in lowland rivers, but it is recognised that there is a paucity of information regarding upland channels (National Rivers Authority, 1996b). More information regarding the ecological value of biotopes across a wider range of river types is needed, and is being carried out as part of the Environment Agency's R&D projects, as a contribution to their developing Habitat Quality Index.

7.2 APPLICATIONS FOR RIVER MANAGEMENT

7.2.1 *River restoration monitoring*

The River Restoration Project was established as a demonstration project in restoration techniques, and to improve understanding of the effects of restoration schemes on, among other aspects, nature conservation value and visual amenity (River Restoration Project, 1994b). The scheme was designed using the expertise of freshwater ecologists, planners, fisheries biologists and geomorphologists, and aims to achieve the rehabilitation and enhancement of selected river channels. In a paper presented at British Hydrological Society Conference on Floodplain Rivers in Birmingham, June 1996 (Large and Petts, in press) the restoration of ecological functioning and techniques for evaluating the success of restoration schemes to this aim are proposed as one component of sustainable river restoration (Brookes *et al.*, in press). Monitoring "riverine productivity" was suggested as a means of evaluating post-restoration ecological value; it is proposed that biotope patchiness may be used as an indicator of potential ecological productivity since it can be estimated with relatively little time investment.

Many of the first river restoration schemes were retroactive: designed to mitigate the effects of land use impacts e.g. mining and road construction (Gore, 1985). Appropriate techniques developed from theories and local experiences; a review of North American restoration programmes is provided by Gore (1985). Several papers describe the various techniques (Brookes, 1992; 1995b; Osborne *et al.*, 1993; Brookes and Shields, 1996), and a seminar on River Restoration describing UK, Danish and German experiences was held at Hydraulics Research, Wallingford in November 1995. Technically, *restoration* involves a change to the original, natural state of a channel, as described by popular North American environmental literature:

> *A trout stream is only a trout stream when it's flowing between its own two banks, at its own pace, in its own sweet way."*
>
> Duncan (1983, p287)

In practice restoration often equates to habitat *enhancement*, due to financial, political and land use constraints on total channel restoration. Osborne *et al.*, (1993) state that restoration programmes should aim to create a channel with stable geomorphic features, which supports a functionally diverse community assemblage. Features which enhance fish habitat are described by Wesche (1985). In terms of instream physical habitat this would include the retention of natural biotope sequences and bank features which provide habitat:

> *"...the best physical habitat restoration efforts imitate the geomorphology of a reference channel in the hope that natural restoration of biological integrity and water quality will follow."*
>
> Osborne *et al.*, (1993, p191)

The rivers Cole and Skerne reflect the river management practices of the last fifty years, in which channel modification and engineering for flood defence has led to the realignment or resectioning of many UK watercourses to allow the more rapid passing of flood waters and increased flood conveyance (Brookes, 1988). The two sites were selected by the River Restoration Project (River Restoration Project), a non-profit making company established to promote the restoration of rivers. The River Restoration Project was supported by the NRA, government conservation agencies and has received EC 'LIFE' funding (River Restoration Project, 1994a). Site selection and the establishment of the Steering Group began in 1993; restoration works were implemented in 1995.

As described in section 2.3.1 and developed throughout this thesis, the basic unit of instream physical habitat is the *biotope*. To test the success of the restoration project in increasing the diversity of physical habitat, biotope mapping was carried out as part of pre- and post-restoration monitoring on both the Skerne and the Cole. This work was carried out voluntarily by the University of Newcastle, and has been reported internally to the River Restoration Project as part of a pre-project monitoring exercise (River Restoration Project, 1994a). A summary of the immediate effects of the restoration schemes on biotope types and distribution is presented in Appendix B3.

There have been few studies of the precise ecological 'value' of a particular biotope for biological communities (see section 7.1.1). Biotope-biota relationships have been tested in South African rivers for both hydraulic biotopes (Wadeson, 1995b) and 'functional habitats' associated with vegetation types (Dallas *et al.*, 1994). Until more research has been undertaken regarding biotope-community relationships in the UK, judicious river habitat management will aim to maintain existing diversity; river restoration or enhancement schemes should aim to increase the proportion of biotopes which are poorly represented for that type of channel. The ecological benefits of restoring a diversity of biotopes have been indicated; Biggs (1996) describes the recolonisation process for periphyton, which is most rapid in *"pools, slow flowing runs and stream margins"*; similar processes are thought to take place for macrophytes (Fox, 1992). In terms of wider biotic communities, timescales for recovery are related to size of stream and degree of isolation (Fuchs and Statzner, 1990). At both the Cole and Skerne sites a natural section of channel exists upstream of the restoration reaches, which should promote the rapid recolonisation of macrophytes which increase hydraulic diversity, and invertebrates.

In the broader context of catchment scale river restoration or habitat enhancement schemes, it is essential that features are restored which are appropriate to the original unmodified channel. Brookes *et al.* (1996) discusses the historical perspective on restoration and advocates (ideally) a return of channels to their Holocene status. However, practical restoration schemes must represent a balance between the 'ideal' and the realistic. The restoration of channels, or enhancement of features which are present in unmodified, contemporary channels of a similar geomorphological character provides a pragmatic management objective. This emphasises the need for a classification or typology of natural channels, within which the features of similar channels may be compared. Heed and Rinne (1990) advocate the placing of boulders to improve fish habitat; this is common practice within the Tweed Foundation's in-channel salmonid habitat restoration programme (The Tweed Foundation, 1995). These improvement structures are only likely to be ecologically successful if boulders are characteristic of the unmodified channel (whereas channel features to improve *fishing* need less care). The River Keekle in West Cumbria which has been restored following open cast mining in 1994, has very little substrate diversity and only a small number of *riffle* biotopes. The restored channel has been 'lined' with butyl and boulders at its margins, which prevent erosion of sediment but does not allow a more natural biotope sequence to develop. A natural tributary of the Keekle provided the best 'template' for the restoration, but this does not seem to have been considered, despite the fact that there were no constraints on the restoration scheme (e.g. flood damage or infrastructure). The application of a channel typology in determining the 'desired state' or targets for the restoration of physical habitat channels is discussed in section 7.4.

7.2.2 Post-flood channel maintenance

Biotope mapping was applied to a flood defence management issue which arose on the River Swale following large floods which occurred in January and February 1995. Local landowners' calls for protection against further flooding were assessed by walking the affected 2km reach from Catterick to Ellerton-on-Swale. As part of the field assessment, biotope mapping was carried out in May 1995. The greatest biotope diversity was associated with the natural sedimentation zones which had been reworked by the floods, creating a 'wandering' gravel-bed channel. By contrast 'stabilised', revetted single-thread channels are characterised by longer *run-glide* sequences with lower biotope 'diversity' for a standard channel length. Anglers reported a decline in fish catches which coincided with pre-existing erosion management by revetments, attributing the decline to loss of habitat diversity, but particularly to the loss of deep holding pools.

Despite the lower biotope patchiness in the revetted sections, overall, wandering gravel-bed channels are much more diverse than the engineered channels of the Cole and Skerne (see section 5.2). The River Swale has a diversity of biotopes, both in unmodified and protected sections, with no single biotope dominating. Compared with the engineered channels, there is a relatively large area of available channel habitat per metre of channel, due to the bypass channels and backwaters associated with depositional features characteristic of a wandering channel. From a conservation perspective any flood management scheme should aim to maintain and, where possible, enhance biotope diversity. Flood defence activities should allow (and, if possible, encourage) the existence of areas of active gravel, to maximise physical habitat diversity. Braid-like sections in a wandering reach have been shown to provide good salmonid habitat due to a diversity of morphological and hydraulic units (Lapointe and Payne, 1996). These recommendations were included in a report to the NRA by Newcastle University (Newson and Padmore, 1995). As a result the flood defence managers concerned took the decision to increase channel capacity only by 'scalping' gravel from a point bar, rather than moving machinery into the channel.

7.3 FLOW MANAGEMENT AND CRITICAL BIOTOPES

The issue of flow management traditionally focused on the establishment of minimum maintained flows (MMFs), which are usually set as a fixed discharge at just one location in a catchment (see section 1.3, p10). With increasing awareness of, and legislation to protect, aquatic ecosystems and biota (Mellquist, 1992; Boon, 1992; Newson, 1992b), the emphasis has moved away from a minimum flow at a single point in a system, towards the concept of an *ecologically acceptable flow regime* (EAFR) (Petts, 1996). These aim to accommodate not only the seasonal requirements of a range of organisms (spawning, rearing, migration etc.) but the range of flows necessary to maintain channel and floodplain processes (Petts and Maddock, 1994). In order to meet the habitat requirements of a range of instream biota, flow manipulation which maintains the full range of biotopes and, where appropriate, the characteristic biotope sequence is the logical development of the work reported here. However, the differing requirements of individual species, and indeed life stages may or may not be compatible; Humphries *et al.* (1996) state that flows which maintain the inundation of one habitat will not be appropriate for others. In the Sabie River, South Africa, fish and hippopotamus are highly valued from a conservation perspective, yet they have very different instream flow requirements (Gore *et al.*, 1992). Moreover, the non-ecological demands for flows, e.g. water supply, abstraction and recreation are unlikely to be compatible with the flow and physical habitat requirements of others. As King *et al.* (1989) recognise, the needs of all users of water both in and out of channel are unlikely to be achieved in practice. Different uses must be traded off, which indicates the need for a quantification of the relative importance of ecologically-allocated flow at various periods of the year, and at different points within a catchment. This may be combined with ecological information on the conservation value of biotopes to address the following key questions (after King *et al.*, 1989):

1. Do different river types require different flow exceedence percentiles for maintenance of biotope sequences?
2. Can the instream flow requirements for the maintenance of critical biotopes and characteristic biotope sequences be determined?
3. Which biotopes are of high conservation value?

The first issue has been addressed in section 5.3. Issues 2 and 3 need a combination of both habitat hydraulics and ecological information. This thesis has concentrated on the first of these aspects, considering biotope distributions across a range of discharges and channel types. In terms of biodiversity maintenance, **critical biotopes** would be those which are essential to the completion of key life-stages, though they may contribute only a small proportion of the total river network (King, *et al.*, 1989). Importantly for this work, it is these biotopes which most sensitive to discharge e.g. *cascades* and *riffles*. This thesis has explored the notion of biotope 'diversity' maintenance across a range of channel morphologies and discharges.

7.3.1 Maintenance of a dynamic biotope sequence, biotope 'patchiness' and 'connectivity'

Previous studies have considered environmentally acceptable flow as a percentage of annual daily flow (ADF). Orth and Leonard (1990) concluded that 30% ADF provided optimum habitat in small streams. At thirty percent ADF Tennant (1976) describes stream hydraulics as *"satisfactory"*, as the majority of substrate is wetted Bars function as riffles; runs and pools are sufficiently deep to provide fish cover. Invertebrate life is reduced but still sufficient for fish feeding requirements. As flows approach sixty percent of average, a diverse range of habitats exist to fulfil all life-stages of fish and invertebrates and provide good bankside habitat including marginal deadwaters. Section 5.3.2 indicates the flow exceedence percentiles which bring about maximum biotope diversity at the eleven study sites. Ranges of flows within which biotope diversity is 'acceptable' for different sites (representative of different channel types) are given in section 5.3.3. These results may be broadly extrapolated to rivers of a similar type to allow, in rivers which are regulated, discharges to be manipulated according to seasonal requirements of biota, or other instream flow needs. It is acknowledged that biotope diversity will not correlate with biotic diversity in all cases; at the Ouseburn site water quality is the overriding determinant of invertebrate communities (Turnbull and Bevan, 1994). Also of significance in addition to the range of biotopes are other habitat features which contribute to diversity, for example overhanging trees, undercut banks etc. Where these features are present, habitat diversity may be achieved at lower discharges (Petts *et al.*, 1995), which illustrates the benefits of physical habitat restoration in addition to flow manipulation.

In terms of biotope diversity, very low flows (below Q_{95}) result in lower indices than moderate flows (see Figure 5.3). Under current ecological thinking (and in our ignorance of the precise ecological value of biotopes; see section 7.6), a minimum maintained flow would be one which maintains the range of biotopes present in a natural channel of similar type. Some loss of total wetted area is tolerable, provided all biotope types (especially marginal and 'connecting' biotopes) are retained. Under such conditions, the hyporheic zone (Gibert *et al.*, 1990; Petts and Maddock, 1994) and emergent river sediments or gravel bars play an important role in maintaining available habitat in drought for both invertebrates (Eyre, personal communication) and fish (Barbel) (Baras *et al.*, 1996). When the total area of channel is wetted, an optimum discharge would be one which creates high biotope patchiness, and produces critical biotopes at periods coinciding with key life-stage events. An EAFR (Petts, 1996) includes a range of discharges which mimic a natural flow regime, rather than a single, fixed prescribed value (King and Louw, in press). Results from section 5.3 suggest the range of flow exceedence percentiles which provide various biotope diversity indices at different sites (and therefore channel types). An annual flow regime for a regulated river should operate within limits set by the flow exceedence percentiles producing maximum and minimum biotope diversity, to create a flow regime as close to natural as possible.. It has not been possibly to determine precise threshold flows at which biotopes change, but this is not considered to be a limitation as it is unlikely that flow exceedence percentiles which bring about a change in biotope type and sequence at the study sites would be directly transferable to all sites of the same channel type. Additionally, it cannot be stated with certainty that maximum biotope diversity corresponds to the highest indicated on Figure 5.3, as biotope mapping was performed at only selected flow exceedence percentiles. These were chosen to correspond to probable changes in biotope type and distribution i.e. hydrological events, but these values offer guidelines not precise values.

Biotope 'patchiness' and 'diversity' as illustrated in Figures 5.2 and 5.3 are based on biotopes classified at the transect scale, and does not take into account variation across the channel. As described in section 5.5, the proportion of the channel width occupied by the dominant biotope varies with site (channel type) and discharge. It may be that the increased horizontal component of patchiness compensates for a reduction in patchiness in a longitudinal direction. Differences between channel types are also evident and are described in section 5.5. In regulated rivers there is little hydraulic variation other than the presence/absence of *marginal deadwaters*. This may be an important aspect of the ecological impact of armoured beds in regulated channels, as suggested in section 5.1.

Whether secondary and marginal biotopes have equal, or greater importance to biota at low flows is the subject of much ecological research at present (Tharme, 1996; Gibbins, 1996). At very low flows a reduction in total habitat area occurs as biotopes become 'dewatered' i.e. exposed boulders and localised patches of exposed substrate are present. Of all the study sites, Harwood Beck was the most severely affected, although the Till, West Allen and Kielder sites were affected to a lesser degree (see Figure 5.7, p122). Reductions in wetted area have been used as an indicator of overall habitat availability (Tennant, 1976; Maddock, 1994; Gippel and Stewardson, 1996); in this study it appears that width reductions occur at sites associated with point bars or instream vegetation. Information on the probable location and relative frequency of these features within different channel types is available from the RHS database; this provides an indication of reaches which are likely to be affected by reductions in width at low flows.

The ecological implications of drought have previously been discussed for their impact on biotic communities. Frissell *et al.* (1986) state that the length of time a substrate patch is wetted is the most important determinant of its capacity as stream habitat. King *et al.*, (1989) review the causes for and processes by which fauna respond to changes in flow, and conclude that the first significant change to invertebrate communities takes place with a loss of wetted area. Canton *et al.*, (1984) describe invertebrate losses through desiccation, associated with a reduction in total wetted area and riffle habitat. In the context of patch dynamics, Townsend (1989) states that drought (of any duration) constitutes a disturbance to the system, by the temporary elimination, or reduction of *patch types and* wetted area. Of equal, if not greater importance than wetted area is the type of biotopes present. A reduction in discharge below a critical or threshold value results in the range of hydraulic conditions being sufficiently altered to bring about a change in flow type (and biotope) and associated hydraulic indices e.g. shear velocity. These are described in section 5.3.1, and a summary of the effects of low flows on biotopes is illustrated in Figure 5.8 (p123). Armitage (1995) states that such reductions in habitat area are not a major threat to invertebrates unless prolonged; invertebrates appear to be resilient to droughts and spates within the 'normal regime'. In terms of the width reductions that occurred at low flows in *cascades*, the drought of 1995 may have resulted in impoverishment of the invertebrate fauna, as biotope areas will have been reduced and the hydraulic conditions of biotopes altered during the period from June to August.

Both the hydraulic conditions of refugia biotopes and their 'connectedness' will determine the survival of communities. Harvey (1991) showed how shallow riffles prevent movement between 'pools' by adult bass, although most small fish could still migrate locally between biotopes. Sites which were significantly affected during the drought flows between June and August 1995 include the boulder-dominated and unconfined bedrock sites i.e. Harwood Beck, Kielder Burn and the West Allen. Drought flows resulted in biotopes becoming isolated, and a change in biotope type from *cascade* to *riffles*, *shallow run* and *deadwater*. Glides were less affected but nevertheless showed reductions in wetted area. The total biotope area was reduced at all sites, especially Harwood Beck which had no shelter from overhanging vegetation, therefore experiencing larger evaporative losses. The mid and low gradient sites showed a reduction in wetted area, but all biotope types were maintained and remained connected. In terms of flow management, it appears that certain channel types are more sensitive to drought than others, which is consistent with the results of a R&D project to assess, among other things, low flow abstractions (Sir William Halcrow & Partners, 1995). The presence of dewatered areas of gravel, or marginal biotopes (e.g. shallow runs) which have become isolated from the main flow (see Figure 5.8) are an indicator of 'ecosystem stress'. Simple stage discharge relationships at critical points in a catchment would indicate the range of flows when drought flow biotopes are present. This information could then be used to determine periods when a 'drought flow' release from a reservoir would be appropriate.

Until biota-biotope relations are more fully understood, it seems expedient to maintain a range of biotopes, especially at periods when critical biotic life-stages are taking place. Petts *et al.* (1995) emphasise the benefits of water transfer to reaches which would become dewatered during summer low flows. In a study of water transfers from Kielder Reservoir to the River Wear, Gibbins (1996) showed how summer releases did not increase the area of available habitat beyond that present at natural, higher flows; but they did prevent the reductions in habitat area which would occur naturally in drought flows. The effects of the prolonged drought at unregulated sites may have been prevented by augmentation of flows at the regulated North Tyne and Derwent sites to maintain the full wetted area and range of biotopes during the drought period.

Problems of setting minimum flows or ecological flows based on flow exceedence percentiles became apparent when calculating discharges for the two study sites which did not have a continuous NRA flow record. The West Allen gauging station was abandoned in 1981 due to rapid changes of control on measured water levels; the control for the station is effectively the cascade/rapid featured in the biotope calibrations reported here and its sensitivity to floods is proven by the effects of the January and February 1995 events. For the purposes of biotope mapping a stage-discharge relationship was established on the basis of the three pre-flood hydraulic calibrations for which discharges were calculated using the velocity-area method (see section 6.1). Discharges at other occasions when biotope mapping was performed were estimated from stage readings at the abandoned gauging station. These were converted to flow exceedence percentiles using the flow duration curve for the West Allen gauging station for the period 1970-1979. However, it appears that, at the time of construction of the flow duration curve exceedence values were underestimated. The lowest flow at which biotope mapping was carried out was during the summer drought of 1995, when flows were the lowest recorded for over 20 years (see section 5.3.1), yet the calculated discharge corresponded to a flow exceedence percentile of Q[illegible] At all other sites where biotope mapping was performed during this drought, the discharge recorded corresponded to flows between Q_{99} and Q_{100}. This illustrates the potential problem of setting minimum flows based on flow duration data at sites with changing morphology.

7.3.2 Critical biotopes versus biotope patchiness

Management of flow to maintain the hydraulic conditions associated with biotopes can be considered from two perspectives. Critical biotopes are those which are present at only key locations within the catchment or have limited distribution throughout a reach, but are considered to be essential to the completion of a particular life-stage (Kershner and Snider, 1992; King *et al.*, 1989). Representative biotopes are those which form the characteristic biotope sequence for a given channel type, and which contribute to overall biotope patchiness.

Much of the ecological literature focuses on flow manipulation over *riffles*, as these are more sensitive to flow fluctuation (Gibbins, 1996) and are considered to be 'high production' sites (Beschta and Platts, 1986; King *et al.*, 1989). Moreover, as riffles are topographical high points in a longitudinal profile, management to ensure the hydraulic conditions of riffles are met, will mean the hydraulic requirements of *glides* and *runs* are automatically accommodated. Studies in streams of the central United States indicated that riffle assemblages varied more with temporal variation in discharge and thus hydraulics, whereas *pool* assemblages varied more with their spatial position in the catchment as hydraulics in pools are more stable with discharge fluctuations (Brown and

Matthews, 1995). The adoption of *riffles* as critical biotopes has dominated past invertebrate ecology research, as discussed in section 2.2. Emphasis on the precise micro-habitat variables needed by different species for various life-stage events has dominated both ecological studies and water resource management; PHABSIM is both species and site-specific.

Management of flow to maintain 'critical biotopes' requires knowledge of local systems. In the River Allen, the confluence of the East and West Allens may be considered to have 'critical biotopes', due to presence of a high density of rare invertebrates (English Nature, SSSI citation). If this site is associated with unique hydraulic conditions, it would seem that these may, at least in part, explain the biotic assemblage. Maintenance of the hydraulic conditions of this reach would therefore be critical to the survival of this rare invertebrate assemblage. It may be, however, that the high conservation value of this reach is related to the metal content of the substrate geology; in this case the hydraulic conditions are of less significance, unless they maintain a critical metal concentration. The limitations of managing flows for critical biotopes are clearly illustrated; without detailed knowledge of process and causal mechanisms, the conservation value of individual sites cannot easily be maintained.

Increasing awareness of *marginal deadwaters* and their associated vegetative habitats has produced a shift away from the concept of riffles as critical to freshwater systems. It is in slow flowing marginal biotopes that the majority of 'rare' invertebrate species are found (Rutt *et al.*, 1989; Harper and Smith, 1995; Humphries *et al.*, 1996). Low flows should be set to maintain marginal deadwaters and ensure that these do not become isolated or dewatered. Management of flows to maintain a critical wetted width is advocated by Gippel and Stewardson (1996). Threshold discharges at which wetted width significantly declines are suggested as minimum flows in reaches which are susceptible to low flows. These authors describe width-discharge relations for hypothetical channels with different cross-sectional profiles. The objections to this approach have been discussed for rough, irregular channels where biotope type and dewatering of cascades dominate over simple wetted perimeter. Alternative flow manipulation to prevent the prolonged and extensive spatial coverage of drought biotopes has been advocated in section 7.3.1.

The other side of the conservation coin, is whether management should focus on rare communities at site-specific locations, or have broader aims of conserving the range of habitats (and their associated ecological communities) in entire reaches and catchments. Traditionally, instream flow management has focused on fish, as these are assumed to be indicative of a healthy ecosystem, being the top predators in the freshwater food-chain (Fausch *et al.,* 1990; Gordon *et al.*, 1992). However, as King *et al.* (1989) point out, a 'healthy' fish community does not necessarily indicate continued and full ecological functioning. As invertebrates are known to have narrower tolerances to flow than many fish species (Gore and Judy, 1981), loss of invertebrate habitat and numbers may occur with no change in fish habitat or population dynamics. The need to maintain habitats rather than biological communities is becoming increasingly accepted as a management option (Petts *et al.*, 1995). **Guidelines should be based at the reach-scale, to ensure biotopes within different channel types are maintained at periods coinciding with key life-stage events.** The definition of a reach and its implications for managing flow for different river types are presented in section 7.4.

Flow manipulation for habitat maintenance allows management for both 'representative' and 'critical' reaches. Knowledge of sensitive biotopes will allow flows to be manipulated to ensure these biotopes are present within a reach at critical periods for ecological functioning and channel processes. Critical reaches within a system are key locations where the biotope sequence at the reach scale is sensitive to fluctuation in flow, and where a minimum discharge is essential to allow the passage of fish to upstream reaches, for example. Braided reaches are examples of critical reaches; where flow of a given volume is divided between several channels which may impede the passage of fish if flows become too low. A meander cut-off on the River Ehen, Cumbria is another example of a critical reach within this system. Low flows through this reach may be, in part, responsible for recent declines in salmonid stocks (Adams, personal communication); maintenance of biotopes and their connectedness in this reach may be essential for the long-term survival of fish stocks.

7.4 CATCHMENT SCALE EXTRAPOLATION: AN ALTERNATIVE (OR SUPPLEMENT) TO PHABSIM

The standard identification of biotopes and the statistical validation of their different (but not unique) hydraulic characteristics provides a tool for the rapid identification of 'potential habitats' in the field. At present 'functional habitats' may be determined from existing local knowledge of the river system, existing literature or PHABSIM 'species preference curves.' Category Two curves are less site- or species specific, and are therefore considered to be most readily transferred to other sites (NRA, 1996b). Biotopes may be rapidly mapped by field survey if site-specific management is to be implemented. Current applications of 'at-a site' biotope mapping have been described in section 7.2. The more challenging research question is whether biotope sequences mapped at a particular location may be extrapolated within a reach. PHABSIM guidelines state that a length of approximately 500m provides a representative reach (Johnson *et al.*, 1994).

The selection of a 500m length of channel may be appropriate for flow manipulation associated with a particular abstraction, but is only applicable at the reach scale if the 'representative reach' consists of the appropriate (calibrated) biotope types and sequences (or distribution). The definition of a reach adopted here is that used by geomorphologists, namely:

> *"...a length of channel within which the constraints on channel form are uniform so that a characteristic assemblage of channel forms occur"*
>
> Wadeson and Rowntree (1994, p55)

Wadeson and Rowntree consider the reach within a hierarchical classification of river channels, similar to that proposed by Frissell *et al.* (1986). At lower scales in the hierarchy characteristic biotopes may be identified as repeating units; these are located within 'segments' which correspond with those defined by Frissell *et al.* (1986), and which have since been adopted by others (Maddock, 1994; Petts and Maddock, 1994). Both authors use patterns in river margin and floodplain vegetation as a means of identifying reach breaks in addition to morphological units. 'Ground-truthing' of reach boundaries may be employed using the features described above; in most cases these coincide with breaks in slope, making it possible to define reach breaks from maps (de Leeuw, 1981; Frissell *et al.*, 1986).

The significant of reach-scale classifications to instream biota was first stated by Cummins 1984:

> *"In the last two decades there has been the simple, but important, recognition that running waters differing in expression of the basic geomorphic (e.g. size, gradient), hydrologic (e.g. discharge), and biotic (e.g. community organisation) characteristics are linked together in drainage networks and functionally are inseparably linked to the stream-side vegetated zone"*
>
> Cummins *et al.* (1984, p1820)

Links between reaches and instream hydraulic conditions has been described by Statzner and Higler (1986), who observe zones of transition in 'stream hydraulics' from the source to the mouth of a stream. This is a modification of the River Continuum Concept (Statzner and Higler, 1985), which states that the physical structure of rivers does not follow a continuous gradient, but is *related to local geomorphology and slope, and its influence on substrate and flow*, rather than changes in hydraulics which are dependent on discharge and slope (and expressed by stream order in the RCC).

Just as the repeating sequences of morphological units are recognised by geomorphologists to constitute "*important properties of the channel*" (Grant *et al.*, 1990), the biotope sequence represents characteristic instream hydraulic units for a given channel type. The concept of changing sequences or proportional distribution of biotopes was first recognised by fisheries managers:

> *"Runs are almost always present downstream from riffles in the transition to lateral or bluff pools, and increase in importance in a downstream direction as narrow valley sections are replaced by wider valley bottoms with recent alluvium and alluvial terrace deposits"*
>
> Rabeni and Jacobson (1993, p215)

The relative proportion of the channel made of a particular 'habitat' type is influenced by climate (hydrology) and geology (topography, lithology and sediment transport). Morphological sequences produce characteristic assemblages of morphological units and 'mesohabitats' at a regional scale (Morisawa, 1968; 1985; Brussock *et al.*, 1985). At a scale relevant to sediment transport and ecology, hydraulics are determined by history, geomorphology, climate, vegetation and land use (Milner *et al.*, 1985; Statzner *et al.*, 1988). Tributaries, local geomorphic controls and the riparian zone affect the downstream pattern in the RCC, to influence channel morphology at the reach scale (Cummins *et al.*, 1995). Stream gradient influences "habitat structure" i.e. biotope types and their distribution at the local (riffle-pool) scale (Rutt *et al.*, 1989). Additionally, vegetation alters functional feeding groups, resulting in further discontinuities to the RCC (Fisher, 1995). A combination of information regarding the physical habitat and nutritional status of a particular channel may be used to predict the associated biota.

In any catchment-scale application of the biotope approach a channel typology is essential. Cohen *et al.* (1996) have developed a regional model of mesohabitat distribution within and between different geomorphological regions of the Loire basin (France). They show that mesohabitat distributions in alluvial rivers may be predicted by slope and stream order. In the UK, the existing RHS database for England and Wales has been used to develop a preliminary typology of semi-natural rivers. The river segment typology (Fox *et al.*, 1996) use 'controlling' geomorphological features to predict the type and frequency of physical features including biotopes and associated hydraulic conditions. Based on the assumption that morphology is primarily related to flow regime, geology and altitude (Poff and Ward, 1989), the typology was developed based on the variables geology, slope, distance from source and height of source (Fox *et al.*, 1996). Over 4500 sites have been surveyed between 1994 and 1996, which have enabled characteristic physical habitat features to be ascertained for a particular channel type. Preliminary analysis shows that flow types and substrate size developed for this thesis are closely related to channel type, although cannot be used, in isolation, to predict channel type. Of more significance is the sequence and proportional distribution of biotopes.

The study sites in this research programme were selected without the benefit of the NRA's typology but retrospective comparison shows that are representative of the main channel types. Types not covered by the study sites include the steep gradient, boulder dominated headwater streams which are common in the Lake District and very low gradient rivers. These were not excluded intentionally, but are absent from North East England. Fortuitously, the 'Lake District' streams are unlikely to be manipulated to a great degree due to their location in National Parks and the logistics of management. Low gradient streams with relatively uniform cross-sectional morphology are suitable for PHABSIM calibrations; any future management of such channels is likely to continue using this approach, or rapid habitat assessment techniques such as those developed by Maddock (1994). Additionally, the biota associated with big, lowland rivers are more likely to be controlled by biotic factors (Zalewski and Naiman, 1985), water quality or functional habitats at the scale of substrate and vegetation patchiness. Research into functional habitats in low gradient channels has been carried out at the University of Leicester (National Rivers Authority, 1996b). This report notes that there is a paucity of information on functional habitats in upland channels. The research presented here provides information on potential habitats and changes in biotopes (a surrogate for available habitat) with discharge. **Knowledge of the distribution of physical biotopes for a particular type enables the prediction of features for any river within that type.**

This thesis has also addressed changes in characteristic biotope sequences which occur with discharge fluctuations at sites representative of a range of channel types. Information on the frequency of features at various flows, (which may be obtained from a combination of hydrological data and field survey) enables temporal predictions of biotope sequences to be made within reaches. It is assumed that biotope diversity, both at the biotope and reach scale will promote biotic diversity. Flows may therefore be manipulated (or physical structures added to the channel) in order to create maximum biotope diversity or maintain critical biotopes. However, in order to provide a practical cost-effective alternative to the habitat *area* predictions of PHABSIM, a means of extrapolating dimensional, as well as hydraulic results is required. Hydraulic geometry is a possible method for determining useable areas at different flows; its potential application is the subject of the following section.

7.4.1 Extrapolation to reaches by hydraulic geometry

Several attempts also have been made to classify rivers into reaches of similar channel dynamics, to enable appropriate management (Mosley, 1987; Kellerhals and Church, 1989; Rosgen, 1994). One of the most useful classification system to date is that developed for the US Department of Agriculture and Forest Services (Rosgen 1994), which applies principles of hydraulic geometry to link morphological channel types and hydraulics. Hydraulic geometry describes the way in which depth, width and mean velocity change with bankfull discharge (Leopold and Maddock, 1953; Ferguson 1986). Channels are classified according to topographical, morphological and sedimentary properties, and broad hydraulic predictions made for a morphologically defined stream type, based on hydraulic geometry and slope-discharge equations (Leopold and Wolman, 1957; Leopold *et al.*, 1964). 'Downstream' hydraulic geometry describes the way depth, velocity and width increase with catchment area (or mean flow), and can therefore be used to predict changes in biotope *areas* at different locations within a catchment. 'At-a-station' hydraulic geometry describes how these variables alter at a particular site, and is dependent upon cross sectional channel shape. Wadeson (1995b) makes some generalisations regarding the influence of channel shape; at one extreme a narrow bedrock gorge will show a rapid increase in velocity as width increases are constrained by the channel sides. By contrast, a wide, shallow alluvial channel will show a greater increase in width and depth than average velocity.

Traditionally hydraulic geometry equations are based on bankfull flows. If wetted width is used an index of available habitat (Maddock, 1994; Gippel and Stewardson, 1996), hydraulic geometry relations need to be determined for flows less than bankfull. As a means of incorporating scale into predictions of biotope areas for a given channel at a known location within a reach, basin scale hydraulic geometry models were tested. Singh and McConkey Broeren (1989) describe the application of hydraulic geometry relations to predictions of width, depth and velocity for streams within a basin. These authors recognise that reach averaged values are insufficient to describe depth and velocity changes within different "habitats", but conclude:

> *"basin stream hydraulic geometry and flow duration relations combined with relationships defining the distribution of depth and velocity in a reach provide a valuable link for relating flow conditions throughout a basin."*
>
> Singh and McConkey Broeren (1989, p596)

Relationships between the distribution of depth and velocity are defined implicitly by the standard identification of biotopes. In order to extrapolate results from the study sites to points within the same reach, downstream hydraulic geometry may be employed, to investigate the link between catchment area and wetted width. If width can be accurately predicted from catchment area, desk-top estimates of total habitat area can be made for any site based on predictions of wetted width combined with predictions of biotope type and distribution from the RHS/River Segment Typology database. Obviously this technique will by no means match the sophistication of PHABSIM, but may be employed for broad, reach scale estimates of total habitat type and area.

A regression of mean width against a flow surrogate, catchment area (logarithmic values), was plotted for the study sites. As the majority of sites were adjacent to NRA gauging stations catchment areas were listed in the flow statistics records (Institute of Hydrology, 1995). At those sites without a gauging station catchment areas were calculated from 1: 250 000 OS maps. Wetted widths were taken from the mid flow exceedence percentile, but it should be borne in mind that this flow value differs slightly between sites. Harvey (1969) noted that trends in downstream hydraulic geometry studies are complicated by changing flow exceedence percentiles. However, this is less likely to influence the regression than the different geomorphological channel types. Based on just the eleven study sites, catchment area contributes to less than 50 % of the variation in channel widths at the study sites (r^2 =0.469).

The scatter in the original dataset was related to the geomorphology and cross-sectional area of the channels. A sites which is 'over-wide' for its catchment areas is the regulated North Tyne, which may have been widened by erosion below the Kielder Dam, due to an inability to incise the resistant bedrock (the Smales study site is just 4m below the dam). The site is above any major tributaries which would cause channel narrowing as a means of readjusting to a reduced sediment supply and flow fluctuation in the main channel (Petts, 1979; Petts and Thoms, 1987). Sites which are narrower than that predicted by the catchment area include the Wear (Stanhope) site, an incised bedrock gorge which is not free to adjust its channel width at bankfull discharges; width is controlled by the local channel morphology. The channelized Skerne and Ouseburn sites have been deepened for flood defence maintenance in the 1960s (Newson *et al.*, 1994) to create an unnatural, rectangular cross-section, which is narrower than channels with a more natural sloping cross-sectional profile.

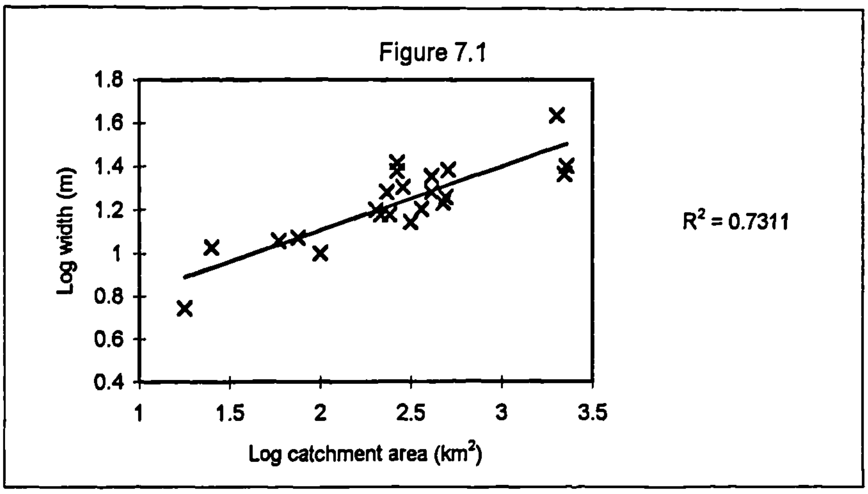
It was considered that eleven points were insufficient to make any firm conclusions from a regression analysis. In an attempt to produce a more robust regression channel widths were measured at several other locations in rivers where the study sites were. Bridges were chosen for easy access to the river and the wetted width measured at the head of a riffle upstream of the bridge. All widths were measured on the same day following a period of dry weather in north east England in an attempt to standardise the flow exceedence percentile. Figure 7.1 indicates that a good relationship exists between catchment area and wetted width, although more sites will need to be included to determine the following:

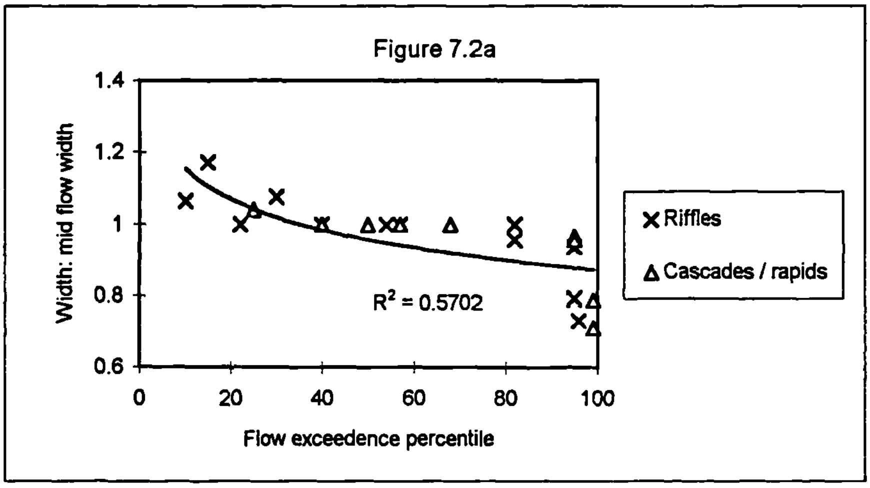
1. Do relationships between discharge, wetted width and catchment area differ between RHS channel types?
2. What features are responsible for anomalous wetted widths - can these be predicted using features recorded in RHS?
3. Do catchment area-width relationships exist for all biotopes e.g. can predictions be made for *glides* on the basis of results from *riffles*?

In attempt to address the first two of these questions with data from the eleven study sites, 'at-a-station' hydraulic geometry was carried out for wetted width measurements. Mean wetted widths were calculated as a percentage of the high flow width, as an indicator of those channels which are width-sensitive to discharge. A better relationship would have been established if widths had been standardised by bankfull width, but it was not possible to record bankfull widths at all the study sites. Figure 5.7 reflects the influence of channel morphology on wetted widths, as sites which show significant reductions in wetted with at low flows i.e. the Wear (Wolsingham), West Allen and Harwood Beck, have large lateral bars present. As flows decrease reduction of the wetted width over the bar significantly reduces the low flow wetted width percentage. The Harwood Beck site shows the largest reduction in wetted width. This site is an unvegetated headwater site which may be 'over-wide' at high or bankfull flows compared to channels with a tree-lined riparian zone (Charlton *et al.*, 1978), resulting in relatively narrow low flow wetted widths. It may be inferred that unvegetated channels and channel types characterised by a high frequency of point and lateral bars are likely to show significant reductions in wetted habitat area at low discharges. The sensitivity of different channel types to flow fluctuation is being developed as part of an 'environmental weighting system' in which assessment of low flows and environmentally acceptable abstractions would vary for sixteen 'classes' of river having different physical characteristics (Sir William Halcrow & Partners, 1995).

In order to address the third question 'at-a-station' hydraulic geometry was carried out for transect-level biotope classes rather than site-averaged widths. In this case because all the data used came from the study sites it was possible to differentiate changes in hydraulic geometry according to biotopes. Wetted widths and depths were standardised by the mid flow exceedence percentile, and a regression line plotted for each biotope class. Biotopes were grouped according to riffle and pool morphological units (Rowntree, 1996c) with the exception of *runs* as these may occur over riffle or pool morphological units depending on discharge. *Cascades* and *rapids* were grouped together as these biotopes were associated with the same transects at different discharges. Results are illustrated in Figures 7.2 and 7.3; they indicate distinctive responses as follows:

Riffles and *cascades/rapids* show more significant changes in width and depth with discharge than *glides* and *pools*. This is consistent with the findings of Tennant (1976), who noted that reductions in wetted area are greater in *"shallow riffle or shoal areas"*. Wetted width in cascades and riffles appears to exhibit a non-linear (logarithmic) relationship with discharge (see Figure 7.2a), with the greatest reductions in wetted width occurring at the lowest flow exceedence percentiles. This is consistent with the findings of Gippel and Stewardson (1996). Depth is a function of roughness, and in many cases roughness does not increase in a manner which may be described by a simple linear relationship (Richards, 1973). For this data a power function best described the relationship between depth and discharge. Width and depth are also functions of channel geometry and bed material which leads to different hydraulic geometries for different channel types and biotopes. Riffles and cascades generally occur in different channel types and have discrete local gradients which show different width and depth sensitivity to discharge fluctuations. Figure 7.4 illustrates wetted widths as a percentage of high-flow widths for *riffles* and *cascades*. Essentially this calculates the reductions in width which occur with decreasing flow. Clearly *riffles* are less sensitive to width reductions than *cascades* for the study sites and range of discharges observed. This clearly shows that before the physical biotope approach can be fully incorporated in the prediction of changes in habitat area catchment area and the selection of environmentally acceptable flows, more data should be gathered to strengthen these relationships. The network of sites established for this research provides an opportunity to refine the reconnaissance use of hydraulic geometry presented here. Such a refinement would be essential to a system-wide application of the biotope approach.





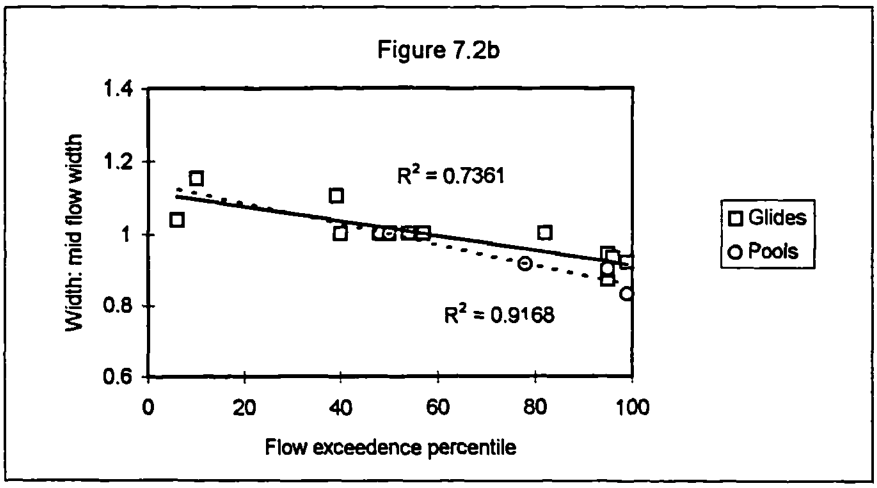
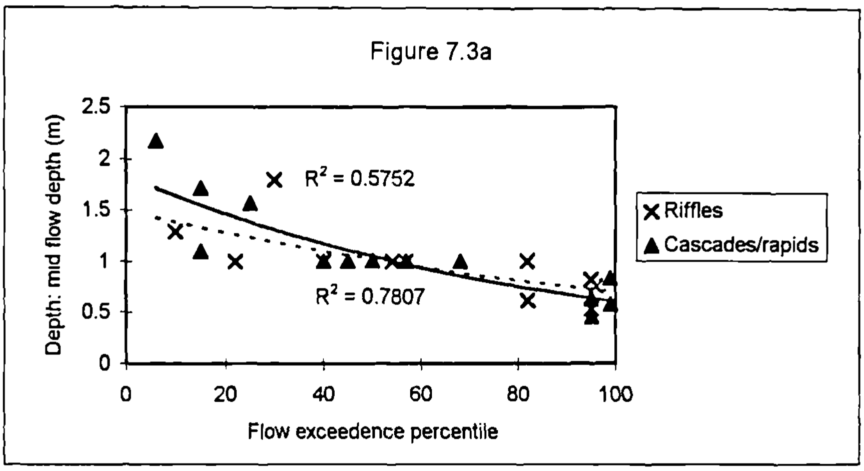
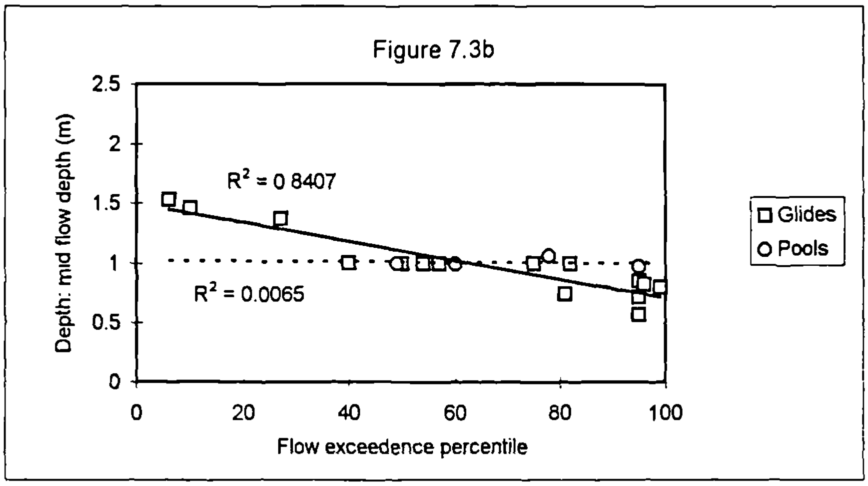


Figure 7.1: Downstream hydraulic geometry relationship between mid flow wetted width and catchment area.

Figure 7.2: Relative changes in wetted width with discharge for biotopes associated with a) riffle morphological units and b) pool morphological units.





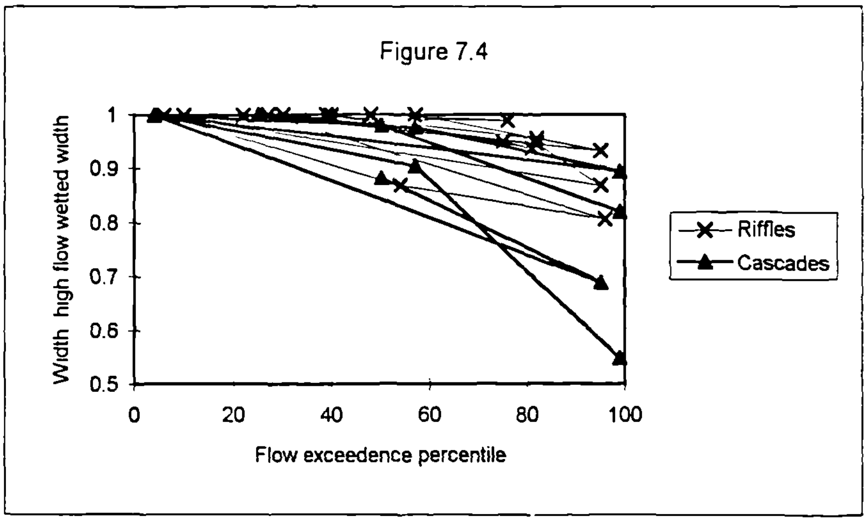


Figure 7.3: Relative changes in depth with discharge for biotopes associated with a) riffle morphological units and b) pool morphological units.

Figure 7.4: Reductions in wetted width with decreasing discharge in riffles and cascades.

8. CONCLUSIONS

Chapter Overview

This chapter forms a summary of the contributions to theory and practice emerging at present from the research reported here. The identification of physical biotopes represents a paradigm shift from the simple, geomorphological interpretation of flows in the riffle-pool sequence. Hydraulic characterisation and validation of physical biotopes as discrete hydraulic units suggests that they should be adopted as standard instream units. Biotopes change in a predictable manner with fluctuations in discharge: the biotope approach therefore has a high potential to contribute to our understanding, and thus sensitive management, of stream ecosystems. To be operationalised there remain a number of refinements to be made, and these are listed as proposed future research.

8.1 CONCLUSIONS

The physical biotope identified by flow type provides an appropriate spatial scale for classifying and comparing hydraulic units which can be recognised by both geomorphologists and ecologists. This study was also concerned with instream physical features at a scale which may be mapped nationally but which has relevance to a range of fluvial geomorphological processes, habitats and organisms which contribute to ecosystem diversity. Characteristic hydraulic conditions result from local geomorphological controls and discharges within a given range to create hydraulic 'patches.' These may be adopted as the basic instream physical/hydraulic unit in which geomorphological process studies, mapping of physical features or biological sampling may be carried out. It is recognised that certain ecological process studies require a smaller spatial scale of analysis, but hydraulic conditions within physical biotopes yield patch units which are likely to be relevant to fish habitat preferences and distributions. The biotope approach allows a mesoscale assessment of instream physical features, which permits comparison between reaches and segments. All classification schemes must be tailored to suit their applications; to be regionally and nationally applicable and to allow links with smaller ecologically recognised scales, the mesoscale biotope approach is advocated. The following points represent the major scientific findings which have emerged from this research:

1. The work for this thesis proceeded in response to calls for a **standard methodology for the identification of instream physical features.** Prior to this research, confusion and ambiguity existed in the description of instream hydraulic habitat. Riffle and pool *morphological units* exist in most natural channels, with flow being controlled by the riffle (or its equivalent in coarse, higher gradient channels) which represents the topographical 'high' point in the channel's long profile. Within morphological units, physical biotopes are present at a smaller spatial scale and dynamic temporal scale. Morphological units usually contain multiple biotopes depending on the local distribution of substrate and flow. This necessitates a procedure for the consistent identification of physical biotopes in the field.

2. **The surface flow type is the best *single* observational feature by which biotopes may be identified** and characterised (see Table 8.1). Flow types have been validated as a means of identifying physical biotopes by discriminant analysis. The Froude number is by far the best hydraulic index for the discrimination of flow types, and has been used together with other hydraulic indices to define quantitatively the hydraulic domains of physical biotopes (see Appendix B2).

3. The use of flow types to identify physical biotopes has been successfully scaled up and adopted at the national scale in the NRA's River Habitat Survey (RHS). Since the application of the flow type methodology depends on large (reach) scale survey techniques, transect-level recording of flow types was incorporated into RHS. This has allowed the broad distribution of physical biotopes to be determined at the reach scale, and the UK distribution of biotopes to be mapped spatially according to geomorphological channel types.

Flow Type	**Description**	**Associated Biotope(s)**
Free fall	Water falls vertically and without obstruction from a distinct feature, generally more than 1m high and often across the full channel width.	**Waterfall**
Chute	Fast, smooth boundary turbulent flow over boulders or bedrock. Flow is in contact with the substrate, and exhibits upstream convergence and downstream divergence.	**Spill** - chute flow over areas of exposed bedrock. **Cascade** - chute flow over individual boulders.
Broken standing waves	White-water 'tumbling' waves with the crest facing in an upstream direction. Associated with 'surging' flow.	**Cascade** - at the downstream side of the boulder flow diverges or 'breaks'. **Rapid**
Unbroken standing waves	Undular standing waves in which the crest faces upstream without 'breaking'.	**Riffle**
Rippled	Surface turbulence does not produce waves, but symmetrical ripples which move in a general downstream direction.	**Run**
Upwelling	Secondary flow cells visible at the water surface by vertical 'boils' or circular horizontal eddies.	**Boil**
Smooth boundary turbulent	Flow in which relative roughness is sufficiently low that very little surface turbulence occurs. Very small turbulent flows cells are visible, reflections are distorted and surface 'foam' moves in a downstream direction. A stick placed vertically into the flow creates an upstream facing 'V'.	**Glide**
Scarcely perceptible flow	Surface foam appears to be stationary and reflections are not distorted. A stick placed on the water's surface will remain still.	**Pool** - occupy the full channel width. **Marginal deadwater** - do not occupy the full channel width.

Table 8.1: Descriptions of flow types used to identify biotopes in the field.

4. Detailed analysis of hydraulic variation at the cell-level has identified the limitations of recording physical biotopes at the transect-level, which include:
   i) Failure to accurately map the distribution and spatial extent of secondary or marginal biotopes.
   ii) Failure to describe biotope 'patchiness' which is a key component for maintaining ecological diversity.

   In order to map spatial units which relate to biological communities cell-level identification and recording should be adopted, which incorporates substrates and vegetation in addition to flow types i.e. "mesohabitats" or "functional habitats".

5. Transect-level recording of biotopes has allowed flow related changes in biotope types and sequences to be determined via **biotope mapping**. Biotope diversity indices (standardised by channel width) provide a means of comparing the response of channels to discharge fluctuations between sites representative of different geomorphological river types. In the absence of a hydraulic model (e.g. PHABSIM), or in situations where the application of a model is inappropriate (e.g. channels with vegetation, high roughness or uneven cross-sections and mobile topography), or when a low-cost, rapid assessment is required, biotope sequences under various discharges (including drought and flood flows) provide an indication of the range of hydraulic conditions and their relative distribution within a reach. Critical biotopes and their critical flows can then be established.

6. The morphology of, and substrates within, a river channel are dynamic. **Geomorphological changes during floods** may, if thresholds are crossed, cause a permanent change to the type and distribution of biotopes. Even without threshold channel change, biotope 'patchiness' is increased, albeit possibly temporarily. This may initiate 'system restructuring' for instream biota and lead to long-term changes in species assemblages and community structure. From a flow management perspective, floods alter the range of exceedence percentiles between which characteristic biotope sequences are present.

7. The ecological significance of physical biotopes is, at this stage, uncertain across a range of sites representative of different geomorphological channel types. **The 'patch dynamics' theory provides a justification for the assumption that physical biotope diversity will promote biotic diversity.** If relationships between biotopes and biotic assemblages do exist, the debate over which variables most accurately represent the hydraulic environment of fish and invertebrates, together with methods for their measurement (e.g. shear stress hemispheres) may be less relevant. Physical biotopes may be used to represent a particular hydraulic domain, without recording dynamic, micro-flow hydraulic variables. From a conservation perspective an assessment of the ecological relevance of biotopes will allow them to be 'weighted' as a contribution to assessments of the Environment Agency's proposed Habitat Quality Index.

8. The biotope and flow type methodology allows a **reconnaissance approach** to flow at a scale recognised by both ecologists ('patches' or 'potential habitats') and geomorphologists (morphological units). **Flow visualisation** may help to define more clearly the sensitivity of biotopes to changes in flow. This can be developed using GIS to help select managed flows to maintain a distribution of physical biotopes according to ecological (biological and river channel) requirements. The reconnaissance approach is also useful for planning and auditing e.g. river restoration, capital schemes and flood maintenance.

9. The central issue still remaining is whether the biotope approach can be used as a practical protocol for determining instream flow requirements and ecologically acceptable flow regimes. Future studies will need to refine the technique; provisionally this study provides the basis for improvements to PHABSIM in terms of the spatial representation of habitat. This requires development of the river segment typology, biotope sensitivity assessment and hydraulic geometry/habitat area studies. These are discussed in the final section as proposed future research.

8.1 FURTHER RESEARCH

The findings of this research have prompted further questions in the field of habitat hydraulics and hierarchical geomorphological models, some of which are to be addressed by the author as part of a post-doctoral study. This work, which will be partly funded by the Environment Agency will attempt to operationalise the concept of habitat hydraulics as a tool for establishing and managing environmentally acceptable flows. The main issues and questions are summarised in Table 7.1.

1. Are depth, velocity and substrate adequate descriptors of physical habitat, both in terms of their biological significance and in terms of their representation by field measurements?
    - How representative is '0.6 velocity'?
    - Is shear velocity a good indicator of near-bed conditions?
    - How do we adequately represent vegetative influences on habitat (macrophytes)?
    - Is substrate intermediate diameter the best descriptor?

2. What are the significant physical biotopes and their associated morphological features across the full range of UK channel types?
    - Do we need all the flow types used by RHS?: can biotopes be 'lumped' on the basis of similar biotic assemblages, e.g. runs and glides in a particular segment type, or runs in lowland segment types?
    - Do some of the remaining types need subdividing (e.g. glides)?'
    - What are special features of sinuous reaches?

3. How should changing flows be incorporated in the proposed synthesis?
    - Are there functional relationships between physical biotopes and flow?
    - Are there thresholds of hydraulic change significant to each biotope?
    - What is the geomorphological/habitat influence of frequent floods?
    - Are there seasonal or other time series properties of physical biotopes?

4. What are the practical problems of assembling physical biotopes via channel features and channel type? Can this be done for large, heterogeneous basins?
    - Can the RHS national typology be extrapolated with a view to modular assembly of biotopes/habitat hydraulic properties; do field-based biotope audits empirically validate the representativeness of biotope sequences in a reach?
    - Are there critical biotopes or sequences within catchments and ecosystems, saving effort on full channel network synthesis?
    - Is biotope 'diversity' or 'patchiness more important? Are longitudinal sequences (mapped at the transect level), or juxtaposition of biotopes (mapped at the cell-level) more important, and does this vary for different channel types?

Table 8.2: Priorities for future research in habitat hydraulics.

In the context of R & D priorities for the development of PHABSIM as agreed by members of a User Forum including Environment Agency (formerly NRA) staff, Institute of Hydrology and members of Higher Education Institutes *"the development of techniques for transfer of habitat availability and habitat distribution data within and between rivers"* was the highest ranked topic area compared with other developments including refinement of the biological and hydraulic modelling (to incorporate sediment transport, vegetation and flow related habitat change). Point 4 in Table 7.4 hopes to address this issue. Work on the ecological significance of biotopes for invertebrate assemblages (Point 2) has commenced for the biotopes defined in this study at the sites selected here. A project at Newcastle University investigating relations between invertebrates sampled within physical biotopes, and excluding site specific factors of water temperature, pH, conductivity and dissolved oxygen has examined both species and family level biotic assemblages. This has been written up as an internal report to the Environment Agency (Grundy, 1996).

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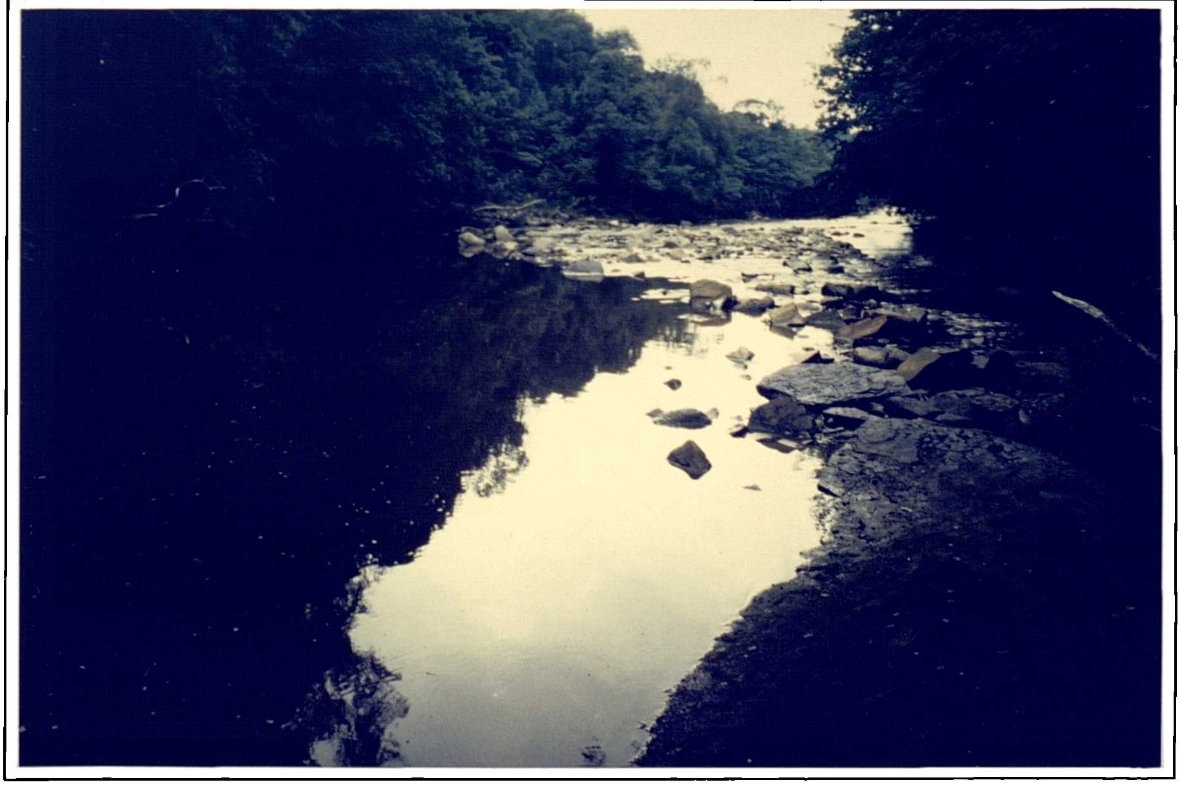
APPENDIX A: DATA COLLECTION AND ILLUSTRATION

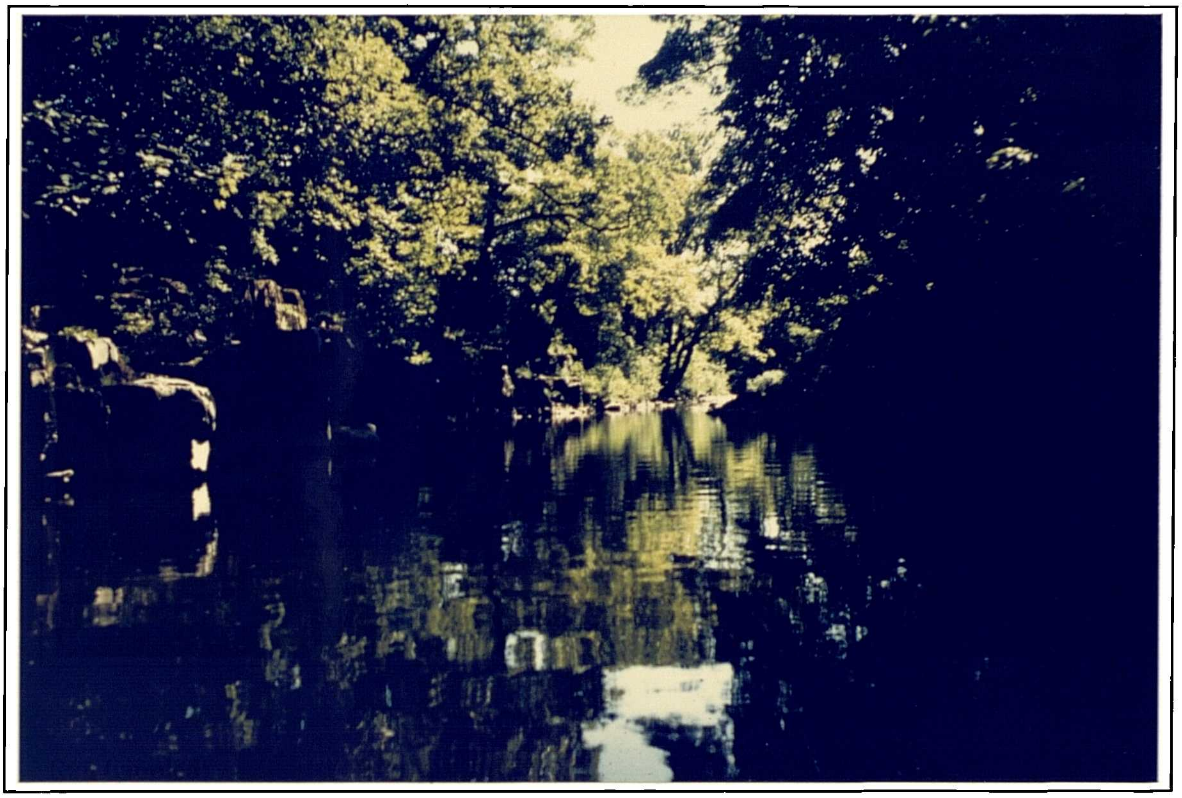
A1: Physical Biotopes and **(Flow Types):** low to high energy

Marginal deadwater **(scarcely perceptible flow)**: within the first few metres of the wetted edge

Pool **(scarcely perceptible flow)**: spans the channel width



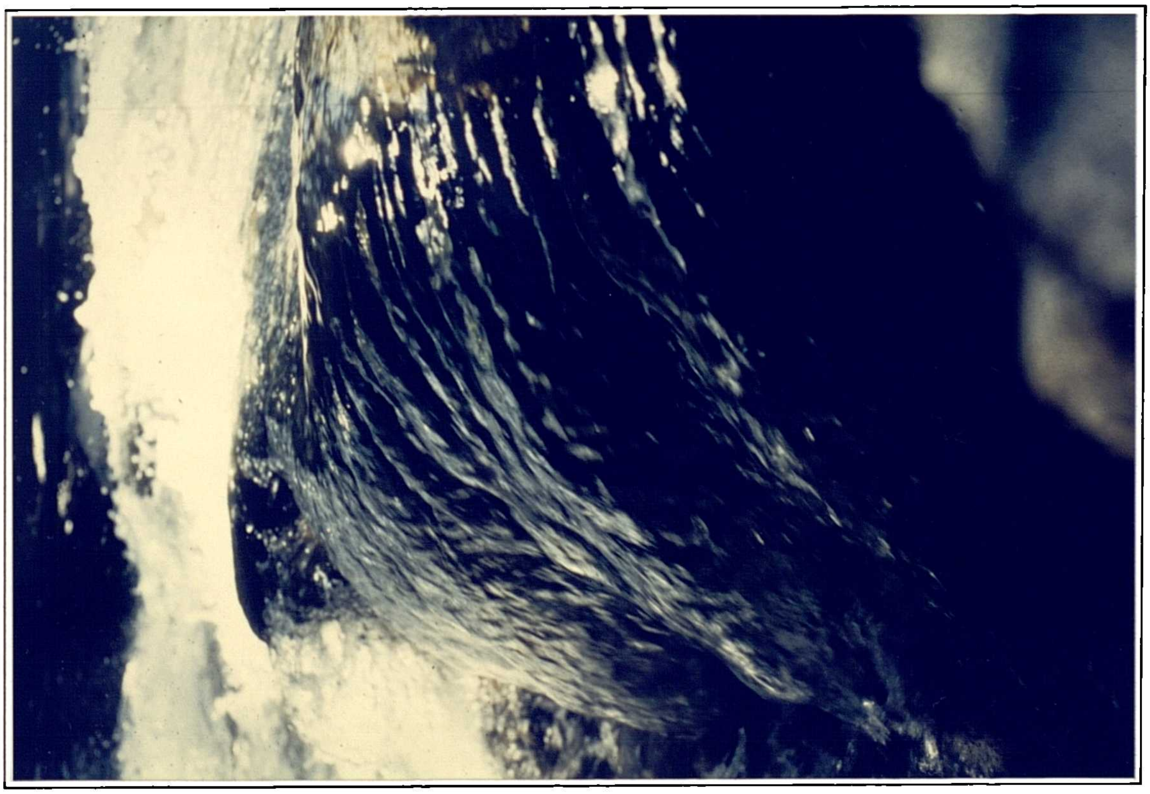
Glide (smooth boundary turbulent)



Run (rippled)

Riffle (unbroken standing waves)

Boil (upwelling)

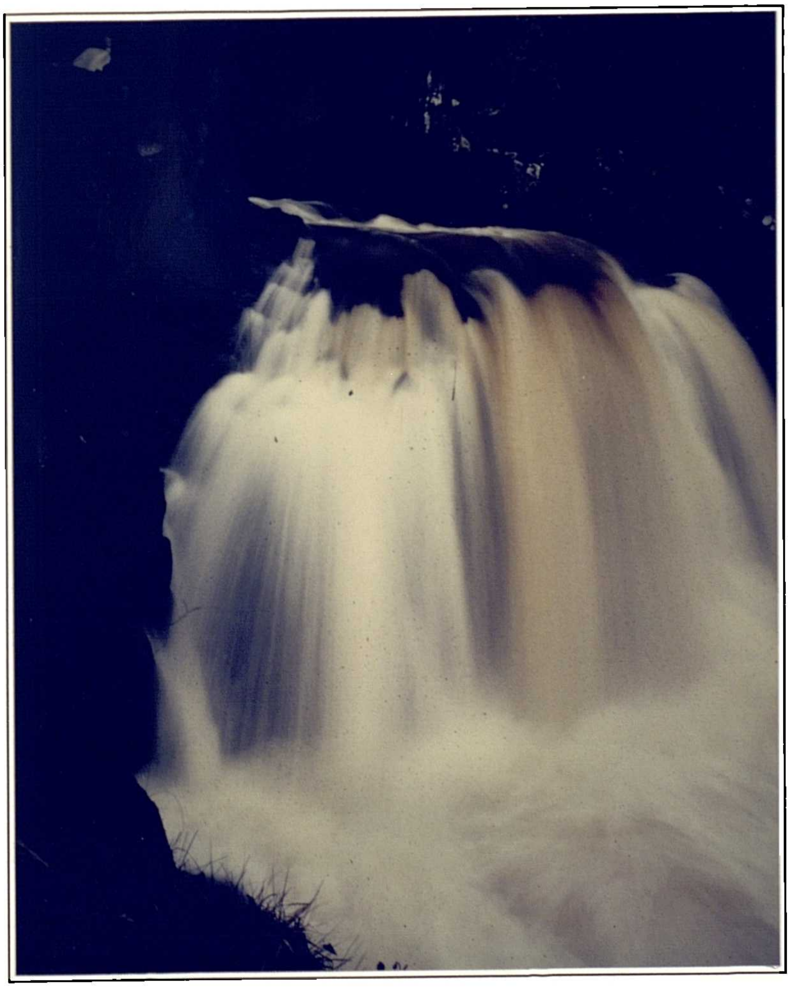


Spill (chute flow)

Cascade (chute and broken standing waves)

Rapid (broken standing waves)

Waterfall (free-fall)



A2: 'Site Characterisation' form

Site name:	Wear (Wolsingham)
Date:	25.11.95
Stage:	0.69m

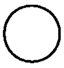
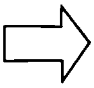
Transect	**Left Bank Landmark**	**Biotope Type**	**Secondary biotope/ other features**	**Right Bank Landmark**
1	First sycamore from bridge	Glide	Marginal deadwater	
2		Glide		Fence post
3		Riffle	Glide	Second sycamore
4		Riffle	Marginal run to left of vegetated island. Island 3m from left bank, 4m wide.	
5	Footpath enters river	Riffle	Marginal run to left of island, as above. Marginal glide to right of second vegetated island 5m wide, 3.5m from right bank.	
6	Top of riffle	Riffle	As above.	

A3: 'Hydraulic Characterisation' form

Site name:	Wear (Wolsingham)
Date:	25.11.95
Stage:	0.69m

Transect	Left bank (m)	Biotope	Flow type	Depth (cm)	Velocity ($m.s^{-1}$)	Substrate (mm)	Right bank (m)
1	0.4	Cascade	NPF	10	0	122	30.6
			NPF	15	0	141	
			NPF	1	0	203	
			NPF	2	0	126	
			NPF	1	0.01	108	
			NPF	3	0.02	98	
			NPF	4	0	109	
			NPF	2	0.01	89	
			NPF	5	0.03	47	
			NPF	7	0.01	143	
			USW	20	0.66	180	
			Ch	18	0.97	99	
			Ch	16	1.26	59	
			USW	20	0.77	65	
			USW	30	0.79	192	
			BSW	20	1	223	
			BSW	34	1.17	49	
			USW	34	1.11	100	
			Ch	32	1.46	296	
			BSW	30	0.96	60	
			Ch	34	1.35	90	
			USW	50	1.05	97	
			BSW	36	1.16	49	
			USW	32	1.35	128	
			Ch	22	1.25	73	
			Rip	10	0.54	30	
			Rip	16	0.05	82	
			SPF	7	0	0.06	
2	0.5	Cascade	SPF	4	0	90	33.7
			Rip	8	0.19	94	
			Rip	0	0	89	
			Rip	8	0.37	86	
			USW	14	0.34	120	

A4: Biotope mapping form

<table>
<tr><td colspan="2">SITE:</td><td colspan="2">DATE:</td><td colspan="4">Flow conditions:</td></tr>
<tr><td colspan="2">STAGE:</td><td colspan="6">Transects with photographs:</td></tr>
<tr><td colspan="8">Transects
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18</td></tr>
<tr><td colspan="8"></td></tr>
<tr><td colspan="3">Biotope sequence:</td><td>POOL
</td><td colspan="2">DEADWATER
</td><td colspan="2">GLIDE
</td></tr>
<tr><td>RUN
</td><td>RIFFLE
</td><td>CASCADE
</td><td>RAPID
</td><td colspan="2">BOIL
</td><td colspan="2">BOULDERS
</td></tr>
</table>

A5: Biotope sequences at the Stanhope site in low and flood flows

APPENDIX B: PRELIMINARY RESULTS AND APPLICATION

B1: INSTREAM HABITAT IN GRAVEL BED RIVERS: IDENTIFICATION AND CHARACTERISATION OF BIOTOPES

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Paper presented at the Fourth International Gravel Bed Rivers Conference: Gravel Bed Rivers in the Environment, Gold Bar, Washington, USA, 20-26 August 1995.

Abstract

This paper addresses the problem of standardised identification and interpretation of instream habitats, which has been hindered in the past by reference to flow dependent, site specific criteria and ambiguous, qualitative terminology. A sample of reaches in North-east England rivers was selected on the basis of achieving an extensive range of morphological units and within these, physical biotopes. Biotopes were subjectively defined on the basis of dominant flow type and their habitat hydraulics measured by data collection procedures consistent with the requirements of the instream habitat assessment model PHABSIM. Classification was initially applied at the transect scale but improvements are gained by classifying each measured cell.

Depth, velocity and substrate data was used to calculate combined hydraulic indices for each sample point. The classification of flow types was then tested by discriminant analysis, to determine whether a priori *defined units are hydraulically and morphologically discrete. Discriminant analysis allocates cells to a particular flow type on the basis of derived hydraulic and morphological indices (Froude number, relative roughness, relative exposure, shelter index and turbulence index).*

Classification at the transect level resulted in misallocation of several cells, due to cross channel variation in physical biotopes. Cell level classification of flow types resulted in more successful classification of flow types across a range of biotopes and geomorphological reaches. Misclassified cells fell into three categories; firstly those cells which have hydraulics similar to adjacent cells of a different flow type. This is common in cascades and riffles, where cells with rippled flow are hydraulically similar to adjacent broken and *unbroken standing waves. From a management perspective this means small scale variation in flow type may be ignored and the unit characterised by the hydraulics of the dominant flow type. Secondly vegetated cells were often misclassified as was upwelling flow, the third group of misclassified cells; in all cases velocity at 0.6 depth is not representative of mean velocities.*

Biotope mapping at the sample reach scale is advocated as a precursor to modelling the impacts of changing flows. Results illustrate how the biotope sequence alters with discharge and applications for the technique are suggested. Flood flows lead to redistribution of substrate within biotopes, change in extent of different biotopes, or new biotopes as morphological units change through erosion and deposition.

Hydraulic characterisation of biotopes raises the prospect that the habitat hydraulics of changing flows can be synthesised for whole river systems via inventories of channel features and channel classification. The national River Habitat Survey (based partly on morphological units) being compiled by National Rivers Authority field surveyors in England and Wales may also form the basis for an economic alternative to PHABSIM for extrapolating the measured hydraulic characteristics of reaches.

1. Introduction

The need for a more unified classification of instream physical habitats has been emphasised recently by several authors (Jowett 1993, Scruton 1994, Wadeson 1994). Wadeson advocates the term 'biotope' as opposed to 'habitat'; the former referring to community rather than species level. The term biotope is retained to maintain consistency with international terminology, whilst stressing that only physical aspects of the local flow environment are considered. Additionally relationships between species assemblages and physical biotopes are, as yet, untested in British rivers. A physical biotope is distinguished by the hydraulics associated with a particular morphology under a range of flows. This approach is essentially that taken by instream methodologies such as PHABSIM (Johnson *et al*, 1994). PHABSIM (Physical Habitat Simulation Model) estimates of Weighted Useable Area (Shirvell 1989) have indicated the importance of cover in addition to depth, velocity and substrate. Thus a biotope may be defined as *a morphological unit with a characteristic range of hydraulic, sedimentary and vegetative variables.*

To provide scale guidance for ecological surveys, catchments may be divided into sectors and reaches; within reaches characteristic morphological units exist (Maddock 1994). A reach is defined as *"a length of channel within which the constraints on channel form are uniform so that a characteristic assemblage of channel forms occur"* (Wadeson and Rowntree 1994). Selection of representative morphological units and characterisation of their biotopes via hydraulic and geomorphic measurements is performed at the reach scale, as channel bed materials within a given reach are relatively constant with fluctuating discharge.

Change in available habitat with discharge is central to the Instream Flow Incremental Methodology. PHABSIM is a computer model which uses a suite of hydraulic and biological models to simulate change in Weighted Useable Area with discharge (Institute of Hydrology 1994). Results are specific to the species and hydraulic conditions of the reach selected; thus, ideally, to enable general application of simulation results, channels must be classified by reach types (Stalnaker 1994). This may permit catchment-scale application, as associated morphological units are known to exhibit unique, yet regular, predictable hydraulic behaviour (Sullivan 1986). However, the choice of representative reaches has always been a problem for PHABSIM users (King and Tharme 1993). Additionally, knowledge of the characteristic hydraulics of biotopes over a range of discharges within the typical flow regime may assist water resource managers in evaluating the conservation status (O'Keeffe 1995) of a particular reach.

The problem treated by this paper therefore has three dimensions:
a) The subjective selection of reaches and use of nomenclature for flow types, morphological units and biotopes,
b) The objective testing of links with dynamic hydraulic properties and processes inherent in the characterisation of biotopes,
c) The use of the geomorphological knowledge base (including physical habitat surveys) to extrapolate properties and processes to the larger scale, eg. catchment, for evaluating channel and flow management on, inter alia, river restoration schemes and in regulated rivers

1.1 Channel classification; reach and morphological unit identification

Several attempts have been made to classify rivers into reaches with similar processes, to enable appropriate management (Mosley 1987, Kellerhals and Church 1989, Rosgen 1992). The most useful classification system to date is that developed for the US Department of Agriculture and Forest Services (Rosgen 1992), which applies principles of hydraulic geometry (Ferguson 1986) to link morphological channel types and hydraulics. Channels are classified according to topographical, morphological and sedimentary properties, and predictions of hydraulics made for a morphologically defined stream type, based on hydraulic geometry and slope-discharge equations (Leopold and Wolman 1957; Leopold, Wolman and Miller 1964). The relationship between habitat hydraulics and benthic invertebrate zonation (Statzner and Higler 1986) indicates how such a classification may be used to predict potential habitat.

The selection of reaches for the present study had to proceed without the benefit of a UK river typology of the type anticipated shortly (NRA 1994). An office-based 'Rosgen-style' classification was attempted for rivers in North East England, but aerial photography cannot identify morphological units. An iterative, field-based reach scale classification was therefore adopted, with reaches selected on the basis of two prime geomorphological factors, slope and substrate size. Additional considerations were to obtain a diversity of channel features, and to fulfil the calibration requirements of PHABSIM so that the data gathered would have strategic value to river managers.

Kershner and Snider (1992) advocate the use of fluvial features as habitat descriptors, including riffles, pools, runs, glides and steps. However these terms include both morphological units and biotopes, as no standard techniques exist for their identification across different sectors or reaches. Wadeson (1994) reviews the terminology used to describe biotopes in twenty one literature sources, and reveals much confusion between morphological units and biotopes. He makes the distinction between static, morphological units and the flow-dependent, ecological units or biotopes.

The recent River Habitat Survey of England and Wales (NRA 1994) describes units in terms of flow pattern, width and depth at summer flows. As these alter with discharge, and absolute dimensions are not transferable between sites, depths are relative and do not permit standardised identification of biotopes. Discussion with fisheries ecologists and geomorphologists, supported by a literature review and field observation assisted the more disciplined identification of biotopes appropriate to UK conditions. These are described in Table 1.

Morphological unit	*Associated biotopes*	*Fluvio-morphological features for field identification*	*Flow type*
Pool	Main channel pool	Deep, slow or still region of water between riffle units.	Scarcely perceptible flow.
	Marginal deadwater	Located at channel margins, adjacent to glides and runs.	Scarcely perceptible flow.
	Dammed pool	Obstacles (boulders, tree roots, vegetated islands) visible.	Scarcely perceptible flow.
	Backwater	Joined to main channel at one end. Often vegetated.	Scarcely perceptible flow.
	Glide	Uniform, moderately fast-flowing water. Even long-profile with some horizontal eddies. Located upstream of riffles or cascades.	Smooth boundary turbulent.
	Run	Deep, fast flowing water occurring at high flows. Usually associated with rapids.	Rippled water surface with undular long-profile.
	Boil	Located at channel margins and meander bends.	Upwelling; horizontal and vertical eddies dominate.
Riffle	Riffle	Turbulent surface with small standing waves over sand to cobble substrate. Shallower than adjacent biotopes.	Small broken or unbroken standing waves.
	Run	Gravel - cobble riffle at high flow. Boulder riffle at very high or flood flows; dominates entire reach.	As above.
	Boil	Turbulent region located between biotopes or in a mobile, unstable bed. Usually small scale; doesn't extend the full channel width.	As above.
Coarse (boulder) riffle	Cascade	Boulders randomly protrude water surface in steep gradient section.	Chute flow and broken standing waves; water passes swiftly over obstructions creating whitewater on downstream side.
	Rapid	Steep gradient reach through cobble, boulder or bedrock at moderately high-high flow. Obstacles submerged.	Broken standing waves, becoming chaotic at very high flows.
	Boil	Small scale turbulent area between boulders or bedrock at high flow.	As above.
Step-pool	Cascade	Boulders organised into steps across the channel.	As above.
	Pool	Ponded reach upstream of a cascade with smaller sediment.	As main channel pool.
Waterfall	Waterfall	Associated with bedrock outcrops and large boulders. Water falls vertically, not always in contact with the rock.	Vertical free fall.

Table 1: Observational classification of morphological units and associated biotopes by morphology and flow type (after Allen 1951, Bisson *et al* 1981, Mosley 1987, Malovoi 1989, Kellerhals and Church 1989, NRA 1993, Wadeson 1994).

Descriptions are also consistent with French terminology, although the term morphodynamic unit is used rather than biotope (Malavoi 1989). Field identification is based primarily on flow type and morphological features as listed in Table 1. The morphological unit 'coarse riffle' is included to describe boulder bed reaches which are not organised into steps across the channel. Wadeson groups the associated cascade biotope with the step-pool morphological unit. However in British rivers the majority of boulder bed channels do not have sufficient gradient to form step-pools, so a distinction is made between these two morphological units. The key to morphological units is their controlling influence on the *gross* properties of local flow; biotopes, however, are defined by the detail of flow direction, velocity and relative roughness within that local flow. From an ecological perspective, it is the combination of substrate size and flow dependent hydraulics which create the hydraulic conditions to which biota are adapted. This paper will address the identification of morphological units at the reach scale, and attempt to determine the relationship between morphological units and discrete biotopes.

Subjective identification of discrete units can be tested and made reproducible by quantifying easily measurable variables which contribute to physical habitat and which are likely to differ between morphological units. As discharge increases the nature and spatial distribution of biotopes is likely to alter as hydraulic controls change. Reaches will tend to become hydraulically more uniform as individual controls drown out, moving the boundaries of active morphological units. This may be investigated by high flow hydraulic measurements and **'biotope mapping'**, as described in section 3.

2 Methodology

2.1 Site selection and field techniques

Channel types were provisionally classified on the basis of low flow biotopes, dominant substrate and flow type. Study sites representative of each channel type were selected on the basis of proximity to gauging stations, existing monitoring information and access. The final choice of study sites is given in Table 2 and Figure 1 (see page 53).

Site (Catchment)	**Grid Reference**	**Morphological units present**	**Biotopes present at low flow**
Stanhope (Wear)	NY 984391	CR-SP-PB	C-P
West Allen (Tyne)	NY 781568	Ri-SP-P	BR-C-G
Harwood Beck (Tees)	NY 848310	CR-P	Ri-P
Kielder Burn (Tyne)	NY 643946	CR-P-CR	C-G-C
Lambley, South Tyne (Tyne)	NY 672605	P-Ri-P	P-Ri-C-P; Bw
Smales, North Tyne (Tyne)	NY 737857	P-Ri	P-G-Ri
Wolsingham (Wear)	NZ 064372	Ri-P-Ri	Ri-Ru-G(Dw)-Ri
Wooler, Till (Tweed)	NU 001307	Ri; P-Ri-P	Ri; G (Dw)-Ri-G(B)-G(Dw)
Lintzford Bridge, Derwent (Tyne)	NZ 146571	Ri-P	VRi-G(Dw)
Ouseburn (Tyne)	NZ 255686	PB; Ri-P	G; P-Ri
Haughton-le-Skerne, Skerne (Tees)	NZ 304 158	Ri-PB	VG(Dw)-VR

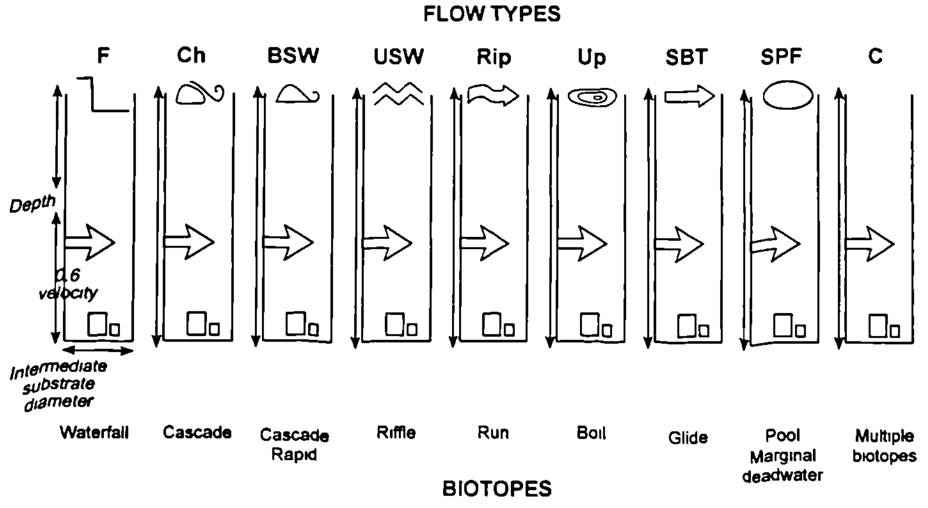
Morphological units: PB = planar bed; P=Pool; Ri=Riffle; CR=Coarse riffle; SP=Step-pool
Biotopes: C=Cascade; BR=Bedrock riffle; Ri=Riffle; Ru=Run; G=Glide; P=Main channel pool; Dw= Marginal deadwater; B=boil; V=vegetated (biotope contained within full channel biotope).

Table 2: Study sites, morphological sequence and low flow biotope sequence

It is clear that morphological units recognised as pools by geomorphologists are associated with a range of biotopes. In fact the true pool biotope as defined by flow type occurs much less frequently than geomorphologists and ecologists have recorded previously.

Transect location follows the requirements of PHABSIM (Institute of Hydrology 1994), with modifications to transect spacing depending on longitudinal hydraulic variability. The downstream transect is located at a hydraulic control point, and transects located upstream at intervals of between 5 and 20m, depending on channel width and the observed hydraulic variability within a given unit. The aim is to place sufficient transects to record hydraulic diversity, without data redundancy. Thus transects are located along one complete biotope sequence as described in Table 2. Their position is marked with wooden pegs or by painting trees or boulders in bedrock reaches. Transects are numbered to ease location when resampling at other calibration flows.

It is recognised that 'within-transect' hydraulic variability can be greater than that in an upstream direction. Thus measurements are taken every metre across the channel, regardless of channel width. Initially biotopes were classified at the transect level, as described in Padmore *et al* (1995). However it is clear that several flow types and biotopes may occur across a single transect, as illustrated in Figure 2 (see page 64). Because of this flow types were classified at the cell level. Flow types illustrated in Figure 3 were recorded at each metre interval sampled. This approach ensures that marginal deadwaters and backwaters are recorded. This is essential as they are considered hydraulically to be *"the richest biological habitat"* (Kellerhals and Church 1989). Additionally it allows a more successful classification of biotopes across a range of rivers as decribed in section 3.



DIRECT MEASUREMENTS

Depth (m)
0 6 velocity (ms^{-1})
Intermediate substrate diameter (mm)

DERIVED INDICES

Froude number
Relative roughness
Shear velocity
Shelter index
Relative exposure
Turbulence index

FLOW TYPES

F = Fall
Ch = Chute
BSW = Broken standing wave
USW = Unbroken standing wave
Rip = Rippled
Up = Upwelling
SBT = Smooth boundary turbulent
SPF = Scarcely perceptible flow
C = Chaotic

Figure 3: Identification of biotopes by flow types

Depth, velocity and substrate size (intermediate diameter) are recorded at each sample point or cell, starting from the left bank of the most downstream transect and completing successive transects upstream. By sediment sampling at each point the total sample exceeds the minimum data required by traditional methods for substrate size analysis (Wolman 1954). Actual sediment size is measured as opposed to an index based on the Wentworth scale, or that used in PHABSIM (Woody-Trihey 1981), in order to allow sedimentary characterisation of biotopes using data analysis techniques appropriate for continuous distributions.

The effective width (EW) is taken as that section with water present; actual channel width (AW) is also measured as the distance between vegetated banks, or that which is regularly covered under a normal flow regime. This allows EW to be expressed as a percentage of AW, as a simple index of available habitat under fluctuating flows (Maddock 1994).

Stage is recorded at each site, either from permanent stage boards at nearby gauging stations, or from specially installed markers at sites some distance from flow gauges. This allows significant changes in morphology and hydraulics at a site to be related to discharge. Sites are surveyed at three calibration flows at which the biotope sequence is different. Sullivan (1986) concluded that habitat hydraulics differed significantly under three flows, notably summer low flow, baseflow and stormflow. To date all sites have been sampled under low and moderate flow conditions. A further six sites have been sampled at high flow. Flow exceedence percentiles are listed in Padmore *et al* (1995). Finally biotopes have been mapped following rainfall ‘events’, as an indication of channel hydraulics at discharges intermediate to those when full surveys were undertaken. ‘Biotope mapping’ is also needed at higher flows in which in-channel measurements are not feasible. Results are presented in section 3.4.

2.2 Statistical data analysis

The data were processed within the SAS system (SAS 1985), and substrate size distribution calculated for both sites overall and biotopes within sites. Width-depth ratio is calculated at the transect level as a means of comparing the planform morphology at the reach scale.

Hydraulic variables calculated are those considered to influence the microhabitat for instream biota, and which have been shown to be good predictors of habitat type or biotope. Most notable is the Froude number (Jowett 1993, Rowntree and Wadeson 1994). Froude number *(Fr)* is a dimensionless velocity-depth ratio, allowing comparisons across different rivers. In hydraulic terms it classifies flow sub-critical ($Fr < 1$) or supercritical ($Fr > 1$) (Davis and Barmuta 1989). Ecologically the Froude number provides a relative measure of stresses within the channel, in terms of the range of depths and velocities. It has been shown to correlate with the distribution of benthic invertebrates (Statzner 1981, Orth and Maughan 1983, Jowett *et al* 1991). The Froude number is calculated by the formula:

$$Fr = Vm / (gY)^{½}$$

Vm = mean velocity at a sample point (ms $^{-1}$)
Y = water depth (m)
g = acceleration due to gravity (9 81 ms $^{-2}$)

Relative roughness is an index of the effect of substrate size and water depth on hydraulics, calculated by the formula:

$$R = s / (d \times 1000)$$

s – substrate size at a sample point (intermediate diameter in cm)
d = water depth at a given point (m)

Three other indices which integrate all three recorded variables were calculated at the cell level: shelter index (SI), relative exposure (RE) and turbulence (TI). All indices listed above are dimensionless so allow comparisons between different rivers. Their success in discriminating between flow types is less than that of the Froude number but nevertheless significant. This is expected as the Froude number was developed to distinguish between flow types. These additional indices are calculated as follows:

$$RE = (d \times 1000) / s$$

s substrate size at a sample point (intermediate diameter in cm)
d – water depth at a given point (m)

$$SI = s / (1000 \times Fr)$$

s = substrate size at a sample point (intermediate diameter in cm)
Fr = Froude number

$$TI = d \times Vm / s$$

s – substrate size at a sample point (intermediate diameter in cm)
Vm = mean velocity (ms^{-1})
d – water depth at a given point (m)

2.2.1. Substrate size distributions by site

Mean substrate distributions were calculated for each site as a statistical indication of gross morphological differences between sites. These are listed in Padmore *et al* (1995a). Analysis of variance (ANOVA) of mean substrate distributions between sites is statistically significant at the 0.001 level. Mean substrate size is an acceptable index of differences in substrate distribution at most sites, with the exception of Harwood Beck. This site has a bimodal substrate distribution of boulders and fine gravels, giving a spurious mean value. Thus D_{84} (the substrate size which 84% are finer than) is suggested as a better index of overall substrate distribution. Additionally it is the D_{84} which has most influence on flow resistance (Maizels 1984); thus more likely to correlate with flow hydraulics.

2.2.2 Discriminant analysis: objective classification of biotopes

Each transect was subjectively assigned to a particular biotope, and coded ***at the cell level*** by flow type to allow hydraulic and morphological indices to be calculated and averaged at this scale. Indices were calculated in SAS for individual cells. This approach differs from the preliminary work reported in Padmore *et al* (1995a); results presented there are for transect-level classifications and are river-specific.

Discriminant analysis provides an objective, statistical test of biotope classification by flow type. It works by assigning each cell to a particular flow type, on the basis of its associated hydraulic and morphological indices. The variables used to allocate cells to a particular flow type are determined by **Stepwise Discriminant Analysis (SDA)**. SDA is a procedure similar to multiple-regression analysis. It calculates, for a given dataset, the percentage of variability between flow types accounted for by each index. In other words it determines which indices most successfully distinguish between flow types. Three separate datatsets were created in SAS: one containing data from all sites at low flow, the second containing all sites surveyed at high flow; thirdly this data was combined into a larger dataset. SDA was not performed for moderate flows as cell level information was not available. However the datasets described included all sites and the full range of flow types.

Once significant discriminating indices have been determined **Kernel Discriminant Analysis (KDA)** is performed, using variables from SDA (see section 3.1). KDA allocates each cell to a particular flow type, calculates the percentage of cell correctly classified for each flow type and lists the flow types to which misclassified cells are assigned. The data analysis procedure is summarised in Figures 4.1 and 4.2 (see pages 79 and 80).

3. Results and discussion

3.1 Significant discriminating indices

Results of Stepwise Discriminant Analysis are listed in Table 3. The partial r^2 value indicates the percentage of variability between flow types accounted for by each of the indices listed.

Flow category	Significant discriminating indices	Partial r^2
Low flow	Froude	0.63
	Shelter index	0.08
	Relative roughness	0.08
High flow	Froude	0.50
	Relative roughness	0.09
	Relative exposure	0.07
	Turbulence index	0.07
Combined flows	Froude	0.53
	Relative roughness	0.12
	Shelter index	0.07
	Relative exposure	0.07
	Turbulence index	0.07

Table 3: Indices which significantly discriminate between flow types

As expected the Froude number was the index which best discriminated between flow types. At low flows shelter index (the relationship between substrate size and Froude number) discriminates between some flow types, notably chute from broken standing waves or unbroken standing waves. Shelter index also distinguishes between smooth boundary turbulent flow of glides over gravel or cobble substrate and scarcely perceptible flow associated with marginal deadwaters which often have fine gravel or silt substrate. At higher flows there is less variation in hydraulics at the cell level, so indices related to the Froude number show less variation between flow types. As a result other indices contribute more to variation between flow types, as indicated by figures above.

3.2 Flow type allocation

Results of Kernel Discriminant Analysis for high and low flow datasets are listed in Table 4. If 75% or more observations are accurately classified for a given flow type, the subjective classification may be considered statistically robust. This is an arbitrary cut-off level, selected partly on the basis of similar work in New Zealand rivers (Jowett 1993). Approximately 65% of riffle, pool and run units were accurately classified using the Froude number, which was considered an acceptable margin of error.

Flow Type	***Percentage correctly classified***	***Percentage misclassified and flow type allocated to misclassified cells***
Unbroken standing waves (USW)	**86.8** *72.0*	8.4 Rip 2.4 BSW 2.4 SBT *11.8 BSW* *8.6 Rip* *3.2 SBT*
Broken standing waves (BSW)	**100** *96.2*	- *3.9 SBT*
Chute (Ch)	- *88.4*	- *6.7 BSW* *2.3 Rip*
Rippled (Rip)	**82.5** *62.9*	9.5 USW 3.8 SBT 2.4 BSW *13.5 SBT* *10.0 USW* *5.9 BSW* *3.5 Ch*
Smooth boundary turbulent (SBT)	**81.0** *75.4*	9.1 Rip 5.1 USW *14.6 SPF* *9.6 Rip*
Scarcely perceptible flow (>0.1 m) (SPF)	**95.0** *85.7*	2.5 Rip 2.5 BSW *5.4 Rip* *3 6 SPF*
Scarcely perceptible flow (<0.1m) (SPF-Rip)	- *87.1*	- *6.8 SBT* *5.3 SPF*

Table 4: Percentage of cells correctly classified by discriminant analysis and flow type allocation of misclassified cells (HIGH FLOW data in plain case; *LOW FLOW* data in *italics*)

Cell level analysis permitted successful classification of flow types across a range of geomorphological reach types. The only flow type which was consistently misclassified is upwelling. In these areas of secondary flow cells velocity at 0.6 depth is not representative of mean velocities, so it is not appropriate to compare velocity-related indices related with other flow types. Such areas are considered as separate biotopes for their role in increasing oxygen concentration; they may therefore be zones of high food production or availability. However for the purpose of statistical analysis cells with upwelling flow were removed from the dataset and the analysis repeated to produce the results in Table 4.

At high flows the majority of cells are correctly classified. At low flows substrate has a greater influence on cell level hydraulics via increased roughness, so there is an increased variety of flow types within any one transect or biotope. This leads to a greater number misclassifications, notably unbroken standing waves and rippled flow types are misclassified in more than 25 % of cells. A list of flow types to which misclassified cells are allocated indicates which flow types are hydraulically similar. For example 11.8 % of cells coded as unbroken standing waves are classified as broken standing waves suggesting some unbroken standing waves are approaching supercritical flow. The majority of misclassified cells may be explained by the influence of adjacent cells on hydraulics. At high flows a rapid may contain some unbroken standing waves at the cell level; however at the transect scale hydraulic may be characterised by that of the dominant flow type. Probable explanations for misclassified cells are summarised in Table 5.

To summarise, although a transect or biotope may contain several flow types at the cell level, most cells within a biotope are hydraulically similar to those of the dominant flow type. The dominant flow type associated with a particular biotope is indicated in Figure 3. From a management perspective these results imply that the biotope is an appropriate scale for description and inventory of hydraulic units. Upwelling flow, boulder bed reaches and vegetated channels require more detailed velocity profiles to accurately characterise their hydraulics; this has been discussed fully in Padmore *et al* (1995).

3.3 Hydraulics of flow types and links to biotopes

Distributions of Froude numbers by flow type for low and high flow datasets are shown in Figure 5. The distributions are statistically different between all flow types ($p<0.001$), with the exception of upwelling flow. Trends in Froude numbers by flow type are similar for both low and high flow datasets, which further supports the notion that the hydraulics of a particular flow type are similar across a range of reaches and discharges.

Mean Froude numbers calculated are within the range of those for South African rivers, with the exception of riffles. South African 'riffles' (Rowntree and Wadeson 1994) would be classified by the present study as cascades, due to the high Froude numbers associated with chute flow and unbroken standing waves. Lower Froude numbers for UK riffles may be explained by the smaller substrate size distributions than those in South African rivers; with unbroken standing waves rather than supercritical flow. Secondly the exceptional low flows experienced during the summer of 1994 and 1995 (Archer personal communication, Institute of Hydrology 1995) lead to below average velocities. Assuming Froude numbers will increase at discharges more representative of the annual flow regime, results are consistent with those of Rowntree and Wadeson (1994) and Jowett (1993).

In this study an electromagnetic current meter was used to measure velocities, which is capable of accurately recording some sections of turbulent flows as negative velocities. This has the advantage of providing more detailed hydraulic information, but limits the use of average Froude values. That upwelling flow does not prove statistically discrete from other flow types exemplifies this point. Mean (0.6 depth) values are of little relevance in such cells, in those with rough boundary conditions (boulder bed) and those with weed growth, because the "normal" logarithmic velocity profile does not exist. We are currently taking velocity readings at five depths in order to improve the hydraulic characterisation of such cells.

A distinction is made between shallow and deep rippled flow types. Despite having the same flow type the two associated biotopes appear very distinct in terms of their potential habitat role. Shallow rippled flow is found at the margins of shallow riffles and areas of shallow deadwater. It is likely to be an important habitat unit for juvenile fish and amphibia, and provide refuge sites in very low flows by preventing deadwaters from becoming isolated and drying out. By contrast the deep rippled flow associated with runs offers habitat for invertebrate species tolerant of faster velocities, and feeding or migratory habitat for adult fish.

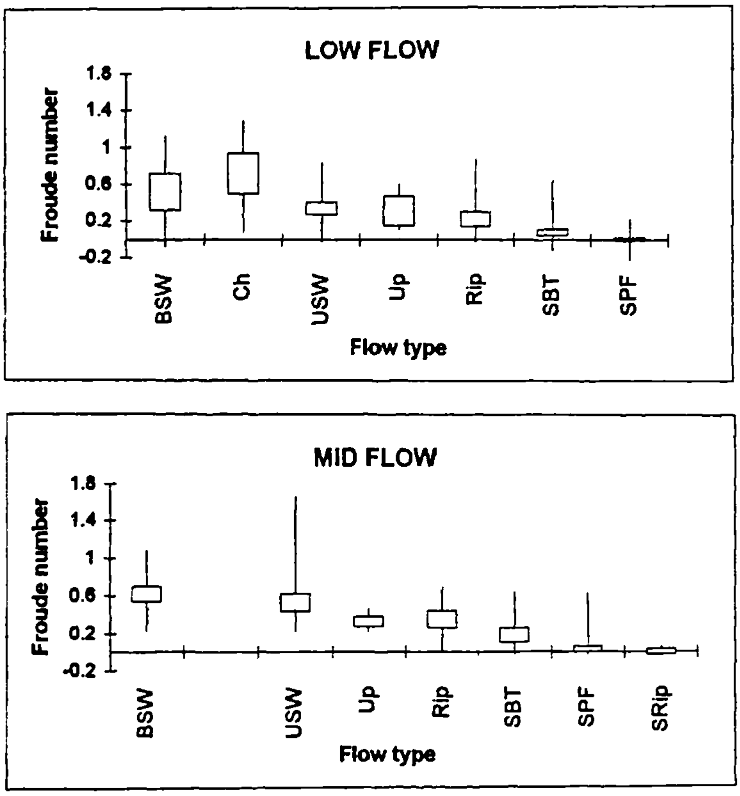


Figure 5: Froude number distributions for flow types classified at the transect level, from sites at low and moderate discharges.

Scarcely perceptible flow is also associated with two discrete biotopes. Shallow deadwater areas offer localised refugia against the faster hydraulics within riffles and cascades. These differ from deep areas of marginal deadwater or full width pools which are important shelter sites for adult fish. The silt substrate often found in marginal deadwaters allows macropyhte roots to become established, creating a biotope very different to that of shallow deadwaters and pools which span the full channel width due to upstream ponding. The latter may provide rest sites for migratory fish, but do not offer the shade provided by overhanging vegetation in marginal deadwaters.

From an ecological perspective the next stage of this research is to determine whether discrete species assemblages are associated with the biotopes delimited. The question is whether biota are sensitive to the hydraulics and substrate size distributions which characterise the differen biotopes, and whether biotopes are associated with discrete species assemblages.

3.4 Biotope mapping

An indication of channel response to increased discharge in terms of the nature and distribution of biotopes is provided by **biotope mapping**. Sites were observed under different flows, stage levels recorded and biotopes mapped. This involved sketching flow type onto a base map of the site, supported by photographs and a record of dominant flow type at each transect. A sample biotope map is illustrated in Figure 6. As water depth increases over coarse riffles hydraulic diversity increases as supercritical flow occurs in cells over boulders. The biotope may then be described as a cascade, characterised by broken standing waves and chute flow. With further increases in discharge, and depths greater than the average boulder height, broken standing waves will dominate; the biotope is then a rapid. At flood flows, fewer broken standing waves will be present as the effect of boulders is drowned out, and the reach may eventually function as a run (rippled flow).

By repeating biotope mapping after rainfall events, it may be possible to determine **threshold flows** ie. discharges which bring about a fundamental change in biotope type and distribution. General objectives of biotope mapping include:

a) establishment of flow percentiles at which biotopes change in different reaches,
b) characterisation of the range of biotopes present for a given reach type under a range of flows,
c) calculation of the probability of occurrence of a particular biotope sequence, both annually and seasonally.

The implications for instream biota are uncertain; it is believed organisms simply move to refuge sites under such conditions. Benthic invertebrates may find refuge in the hyporheic zone (King personal communication), whilst fish simply move to deadwater zones at channel margins to avoid the adverse hydraulic conditions and shear stresses associated with the main channel (Jowett and Richardson 1994).

Floods are believed to be key regulators of biotic populations (Jowett and Richardson 1994; Stalnaker *et al* 1994), by maintaining habitat structure and the sedimentary and hydraulic conditions needed by key life-stages. Events of a 50 year return period or greater are likely to cause 'system disturbance' or a permanent change to channel morphology (Milner 1994). The impact of rare, high magnitude events on sedimentology and distribution of morphological units and biotopes may be considered both immediately after the event and subsequently during the river's transient or recovery period (Petts 1984). Floods with return periods between 25 and 100 years in January 1995 have resulted in changes in substrate size distribution, which in turn has altered biotope type, spatial distribution and extent. Figure 6 indicates a change in biotope type and distribution at Harwood Beck; a different low flow sequence was observed at a similar discharge prior to the flood (6a) compared to post-flood conditions (6b). The aim of post-flood biotope mapping is to quantify this change, determine the implicatons for biotope distribution and extent under the same flow regime and relate magnitude of impacts to flood return period and reach stability.

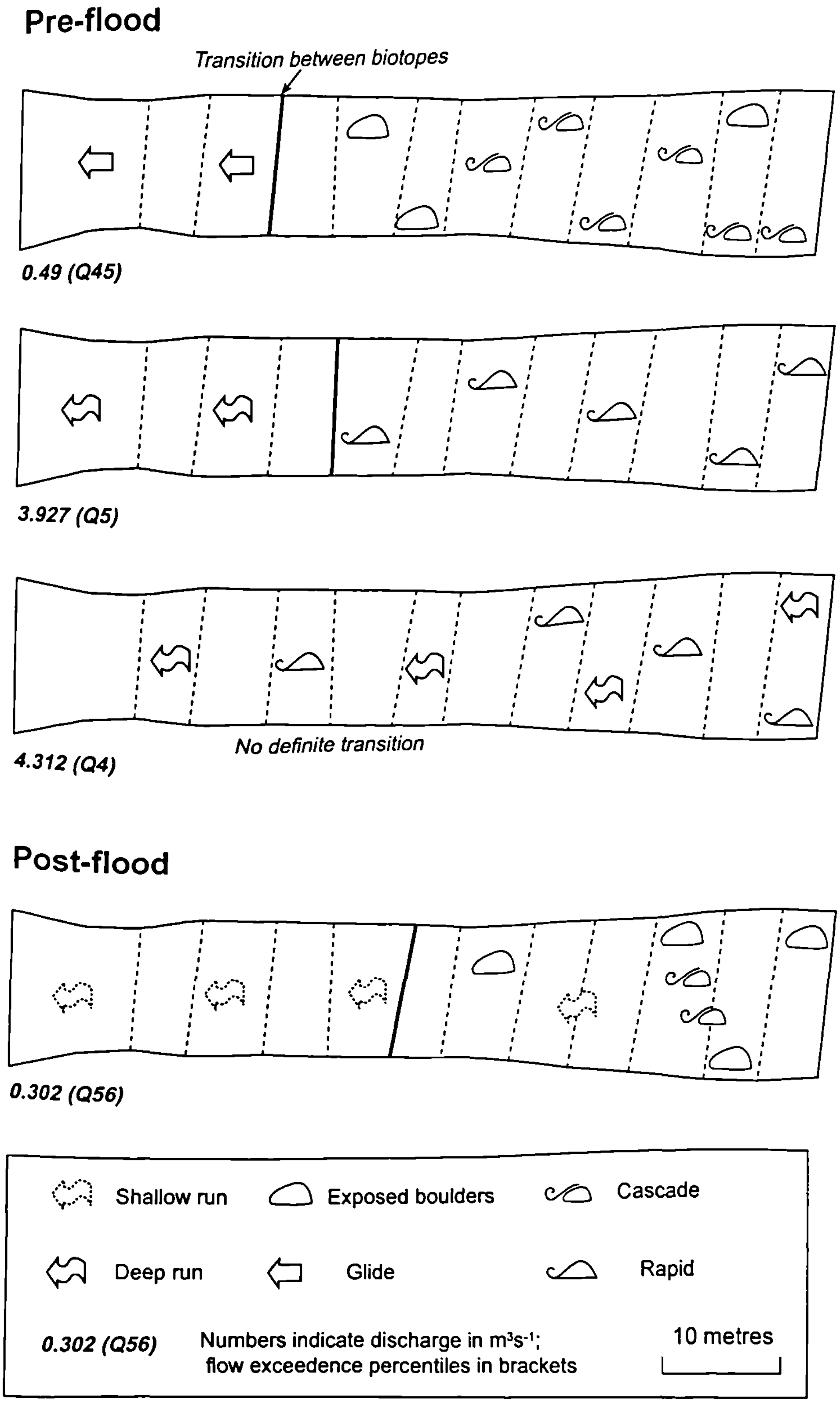


Figure 6: Biotope mapping at Harwood Beck

4. Conclusions and extrapolation

The work reported here was embarked upon with the triple aims of characterizing biotopes by dominant flow type, assessing the hydraulics of the units thus delimited and testing the applicability of the outcomes to river habitat surveying, and eventually, the setting of ecologically acceptable flows. Biotopes have been identified in the field by their characteristic morphology and flow type. By statistical analysis the subjective classification has been verified to identify units with discrete hydraulics over a range of discharges. We are, as yet, far from establishing environmentally acceptable flows, but identification of biotopes by dominant flow type has proven to be a standard, repeatable methodology, which is now incorporated into national habitat inventories within the UK. In future, all the main predictive techniques including PHABSIM, can utilise the growing convergence of view on the characterisation of channels (Jowett 1993, Wadeson 1994).

Optimism must be tempered in relation to field measurements in very rough channels, reaches where secondary flow occurs and those with seasonal growth of aquatic macrophytes and those with in relation to the ability of statistical tests to use all the field information and accurately characterise some flow types (eg. those dominated by secondary cells). Future manipulation of the SAS database will attempt to accurately reflect the hydraulics of such reaches and quantify the implications of ignoring them.

Biotope mapping and quantification of the hydraulics of low flow sequences allows characterisation of both 'representative' and 'critical' reaches. The former includes the sequence of biotopes which are repeated in a given sector (Maddock 1994) or reach (Wadeson and Rowntree 1994). Critical reaches include those units which occur infrequently in the catchment overall (Petts and Maddock 1994). Biological populations will be adapted to the hydraulics of representative reaches at low to mid flows, as these discharges occur at key life stages. King and Tharme (1993) give critical units special ecological significance, considering them *"absolutely essential for the completion of one or more life stages of the selected target species, but which are poorly represented in the reach"*. It is clear that conservation of habitat hydraulics by appropriate flow manipulation is paramount for such reaches.

Biotope mapping over a range of flows will provide information on biotope sequences over a timescale relevant to water resource managers and seek to determine threshold flows at which biotope sequences alter. It has already been used as a means of evaluating the 'habitat value' of different reaches by comparing biotopes diversity. More generally it provides a means of quantifying the impacts of alterations to the physical environment at a scale which has relevance to ecologists, fluvial geomorphologists and engineers. Post-flood biotope mapping will provide an assessment of the role of high-magnitude events in maintaining the structure and distribution of morphological units, and thus habitat hydraulics at the reach scale.

In terms of guiding habitat surveys this study clearly indicates the value of both inventories of morphological units ***and*** flow types (with the latter as a guide to the former). Channel typologies are being developed with the specific aim of describing the gross habitat hydraulics of river networks and their flow sensitivity. If such inventories and typologies are successful they provide a major alternative means of characterizing the impacts of, for example. river regulation or climate change to the more demanding and expensive instream methdologies such as PHABSIM.

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B2: Summary statistics for distributions of substrate size and hydraulic variables by flow types

Flow Type	Index	Minimum	Mean	Standard deviation	Maximum
Scarcely perceptible flow	Substrate size	0.06	90	87	Bedrock
	Froude number	-0.22	0.002	0.04	0.19
	Shear velocity	-0.21	0	0.01	0.08
	Turbulence index	-156	7.56	59.8	680
	Relative roughness	0.08	1.08	2.18	23.5
	Shelter index	0.85	6.56	9.91	59.5
	Relative exposure	0.77	393	2243	31667
Shallow rippled	Substrate size	23	117	73	375
	Froude number	-0.14	0.18	0.19	0.81
	Shear velocity	-0.12	0.01	0.03	0.21
	Turbulence index	-0.07	0.09	0.16	0.91
	Relative roughness	0.29	3.05	3.45	18.8
	Shelter index	0.04	1.4	2.45	11.6
	Relative exposure	0.28	0.73	0.78	3.48
Smooth boundary turbulent	Substrate size	0.06	95	89	610
	Froude number	-0.15	0.09	0.07	0.37
	Shear velocity	-0.01	0.009	0.001	0.05
	Turbulence index	-420	339	963	6403
	Relative roughness	0.001	0.35	0.78	10.3
	Shelter index	0.001	1.56	2.93	36.3
	Relative exposure	0.09	1339	3551	18833
Rippled	Substrate size	0.06	104	82	800
	Froude number	-0.04	0.24	0.13	0.62
	Shear velocity	-0.12	0.02	0.02	0.36
	Turbulence index	-300	90.5	525	6815
	Relative roughness	0.001	0.38	1.07	17.5
	Shelter index	0.001	1.29	5.13	77.5
	Relative exposure	0.33	227	1174	6815
Unbroken standing waves	Substrate size	2	82	65	680
	Froude number	-0.03	0.39	0.14	0.87
	Shear velocity	-0.001	0.03	0.03	0.42
	Turbulence index	-0.01	9.9	25.8	214
	Relative roughness	0.05	0.36	0.65	6.2
	Shelter index	0.003	0.24	0.21	1.53
	Relative exposure	0.16	11.1	25.1	200

Appendix B2 continued.

Flow Type	Index	Minimum	Mean	Standard deviation	Maximum
Broken standing waves	Substrate size	2	157	114	620
	Froude number	-0.11	0.52	0.24	1.5
	Shear velocity	-0.01	0.05	0.03	0.18
	Turbulence index	-1.39	11.7	29.5	148
	Relative roughness	0.002	0.51	0.48	3.1
	Shelter index	0.005	0.31	0.25	1.38
	Relative exposure	0.32	15.8	48.5	360
Chute	Substrate size	2	149	94	415
	Froude number	0.17	0.73	0.27	1.43
	Shear velocity	0.02	0.08	0.07	0.62
	Turbulence index	0.03	4.73	19.3	148
	Relative roughness	0.08	1.54	2.02	10.5
	Shelter index	0.003	0.23	0.18	0.89
	Relative exposure	0.09	4.22	15.8	125
Upwelling	Substrate size	2	48	55	250
	Froude number	-0.11	0.37	0.15	0.64
	Shear velocity	-0.01	0.03	0.01	0.09
	Turbulence index	-0.15	30.9	38.1	236
	Relative roughness	0.003	0.19	0.45	3.13
	Shelter index	0.004	0.14	0.21	1.14
	Relative exposure	0.32	34.1	42.9	275

B3: Application of biotope mapping to monitor and audit the success of restoration works

Biotopes in the 2km restoration reaches of the Cole and Skerne were mapped both before and after restoration, and additionally in pre-restoration vegetated conditions to determine the influence of macrophyte vegetation on biotope distribution. The areas of each biotope type were calculated by walking the reach and pacing their length. Channel widths were measured at several locations along the length of the restoration reach where a significant difference was evident (e.g. at bends). Biotope mapping was carried out to address the following issues:

1. How does seasonal macrophyte vegetation growth alter the type and distribution of biotopes?
2. How do biotope types and distribution alter following the restoration works?
3. Can/should restoration designs and audits use physical biotopes and their diversity as a formal tool?

The first of these issues was investigated as a rapid method of determining instream hydraulic conditions. As biotopes have been shown to be associated with discrete hydraulic conditions, biotope mapping will provide an indicator of the range and distribution of instream hydraulics. The distribution of biotopes in the River Cole restoration reach on 22 June 1995 (pre-works) was influenced primarily by the seasonal growth of macrophyte vegetation. Details of macrophyte species present are provided in the Restoration Project Interim Report (River Restoration Project, 1994a), which divides the restoration reach into three sections. This study divides the reach on the basis of physical biotopes and vegetation type (see Table 7.1). In the upstream section (1) (in unvegetated conditions) *glide* dominates the channel, with *deadwaters* being present at channel margins where sediment is deposited on ‘flams’. In section 2 the dominant biotope was a *pool* (due to ponding of water from the mill weir), and the channel was dominated by *Nuphar lutea*, characteristic of “still to sluggish” water (Spencer-Jones and Wade, 1986). This lower section was traditionally maintained by winter dredging, resulting in the removal of ‘flams’ and their associated macrophyte vegetation. Sections 3 and 4 are downstream of a roadbrige - their characteristic biotopes and dominant vegetation species in the four sections of the restoration reach are listed in Table 7.1.

Section and associated BIOTOPES	1: GLIDE	2: POOL	3: RUN, GLIDE, DEADWATER	4: GLIDE, RUN, DEADWATER
Floating	-	*Nuphar lutea*	-	*Ranunculus penicillatus*
Submerged	-	-	-	*Myriophyllum spicatum*
Emergent	*Sparganium erectum* *Schoenoplectus lacustris*	*Sparganium erectum* *Typha latifolia*	*Phalaris arundinacea* *Typha latifolia*	*Butomus umbellatus* *Phalaris arundinacea*

Table 7.1: Dominant vegetation in four sections of the restoration reach, pre-works.

The influence of vegetation on the extent of physical biotopes was calculated by walking the length of the restoration reach and annotating 1:2500 maps. The proportion of each biotope was calculated, together with areas of vegetation which led to a change in biotope type. Percentages of biotopes in March 1995 (unvegetated) are compared with June 1995 (vegetated); these are summarised in Table 7.2.

Physical biotope	Unvegetated	Vegetated
Deadwater	0.2	32.3
Pool	4.5	31.7
Glide	91.0	30.7
Run	3.7	5.1
Chute/boil	0.6	0.2

Table 7.2: Percentage distribution of physical biotopes in the River Cole restoration reach.

The overall effect of vegetation in the restoration reach is to increase the proportion of *pool* and marginal *deadwater* areas. In section 1 a stand of *Schoenoplectus lacustris* approximately 50m in length caused ponding for 213m upstream, which, in unvegetated conditions was uniform *glide*. Section 2 was totally ponded due to the dominance of *Nuphar lutea* in the channel. In unvegetated conditions there was a transition from *pool* to *glide* where the influence of the mill weir on ponding ended; in vegetated conditions the presence of *Nuphar lutea* causes the pool to extend a further 100m upstream.

Very little floating vegetation exists in the section 2, due to shading from emergent vegetation and trees. The effect of emergents is to reduce the channel width to approximately 2m in this section and increase the proportion of deadwater. Without seasonal vegetation this section is a series of *"shallow riffles and runs"* with much aquatic vegetation (River Restoration Project, 1994a). The inflow of water from the mill leat results in channel width increasing to approximately 3-4m, and a return to *glide* between this point and Coleshill Bridge. Where sediment has deposited on 'flams', dense stands of *Phalaris arundinacea* and *Typha latifolia* were present causing localised marginal deadwaters.

Sections 3 and 4 below Coleshill Bridge were dominated by *glide*, with runs and deadwaters present in smaller proportions. Emergent vegetation at channel margins caused an increase in marginal deadwater, coupled with a reduction in channel width as marginal vegetation reaches peak biomass. Vegetation appears to be reducing velocities in *runs*, especially at the deeper channel margins, increasing the area of marginal *glide* at the expense of *runs*. However, despite the slight reduction in velocities and surface flow type, runs and glides are likely to be more biologically diverse when vegetated. Boulton and Lloyd (1991) described how low invertebrate densities and taxonomic richness correlated with a depauperate aqautic macrophyte flora and absence of riparian cover. A study of 76 sites on 30 lowland rivers showed that areas with macrophytes present had a significantly higher species' abundance than macrophyte-free gravel and silt areas. Emergent macrophytes had the greatest family richness, followed by submerged and floating vegetation, then non-macrophtye areas (Wright *et al.*, 1994).

In summary, vegetation in the River Cole increased the proportion of *deadwaters* and *pools* at the expense of *glides*. The implications for invertebrates are uncertain; what is clear is that biotope diversity has increased, which is likely to be beneficial to some invertebrate communities. The role of vegetation in maintaining seasonal biotope diversity in lowland channels is essential, and should be considerd in any restoration works. In the River Skerne uninterupted *glide* dominated the length of the restoration reach prior to work commencing. Local hydraulic and biotope diversity was associated with a man-made weir in the channel. Compared to the River Cole, vegetation had less influence on biotope types and distribution, as the emergent species were submerged and did not cause ponding of flow. Species present in the Skerne include *Sparganium erectum* and *Schoenoplectus lacustris*; their influence on hydraulic conditions in the pre-restoration reach was to create upwelling flow (*boils*) within the vegetated *glide*. Emergent macrophyte vegetation has been replanted as part of the Cole restoration project with the aim of increasing the diversity of flow types and physical habitats.

To address points 2 and 3 (page XXX) post-restoration biotope mapping was carried out as a means of assessing the success of the restoration scheme, in terms of physical habitat diversity. Biotope mapping offers an alternative post-project monitoring technique which may be employed without the use of PHABSIM (cf Elliot *et al.*, 1996) with significantly less time and financial investment. In the pre-restoration Cole, attempts to increase hydraulic diversity by the addition of fisheries weirs (gravel substrate within a wire basket) had limited success. The substrate retained in them was mainly fine gravels, but a large amount of sand and silt had been trapped in the structure. Additionally, velocities in pre-restoration (man-made) *riffles* were low compared to both natural *runs* and *glides*.

Biotope mapping after the restoration works were completed was carried out at similar discharges to the pre-restoration conditions to allow a legitimate comparison of the biotope distributions. These are summarised in Table 7.3. The restoration works appear to have a greater influence on biotope diversity in the Cole; increasing the proportion of *riffles*, *runs* and *deadwaters*. The deadwaters represent additional habitat, created by retaining backwaters in the position of the original channel, adjoining the new main channel. The increase in *run* and *riffle* resulted from the increased channel length and formation of tight meander bends. In the River Welland dredging eliminated all riffles and pools, leaving uniform sandy runs as the dominant channel feature and reduced invertebrate family richness and biomass (Smith *et al.*, 1990). A re-introduction of riffles and runs should therefore encourage a more diverse invertebrate community. By comparison, the main additional biotope in the Skerne was the constructed *deadwaters* i.e. backwaters, in the position of the original channel. Six months after the restoration works were completed, lateral 'berms' constructed from logs and coarse gravel have failed to increase the diversity of flow types and initiate sediment deposition, a meandering thalweg and riffle formation. Due to a design failure, the low flow level in the upper part of the restoration reach is too low for them to be effective in this way. Once macrophyte vegetation becomes re-established, its ponding effect (as described in section 6.2.2) may well bring these berms into effect at low flows - they will certainly increase local flow diversity at high flows.

	Cole		Skerne	
Biotope	Pre-restoration *	Post-restoration	Pre-restoration	Post-restoration
Glide	91	65	90.9	88
Deadwater	0.2	7	0	9
Pool	4.5	0	-	-
Run	3.7	12	6.	3
Riffle	0	12	3	0
Mixed (boil/rapid)	0.6	4	0.1	0

Table 7.3: Comparison of biotope percentages in pre-and post- restoration reaches of the Rivers Cole and Skerne (* unvegetated).