AN EVALUATION OF THE SPATIAL CONFIGURATION AND TEMPORAL DYNAMICS OF HYDRAULIC PATCHES IN THREE UK LOWLAND RIVERS

C. E. Wallis

A thesis submitted in partial fulfilment of the University's requirements for the Degree of Doctor of Philosophy

University of Worcester

2013

ABSTRACT

Accurate characterisation of the hydraulic environment is a key step in describing hydromorphology at an ecologically relevant scale which has relevance to several aspects of river management, including monitoring river health, designing environmental flows and evaluating river rehabilitation measures. However, current hydraulic habitat quantification methods oversimplify the spatial heterogeneity of the hydraulic environment and do not explain or interpret the spatial arrangement of different habitat units sufficiently or define the dynamics of these shifting patterns. This research applied a novel numerical classification method and a landscape ecology framework to quantify the composition and configuration hydraulic patches in three UK lowland river reaches at five different flows. Five spatially coherent hydraulic patches, defined by the joint distribution of depth-velocity, were optimally delineated from hydraulic point data at each reach using the Gustafson-Kessel fuzzy clustering algorithm. Transitional zones between hydraulic patches occupied between 18-30% and represent an application of the ecotone concept to the instream environment. Hydraulic patch diversity increased with discharge, peaking at high flow (Q38-Q22), suggesting that the provision of high flows is important for maximising hydraulic heterogeneity. The dominance of shallow, slow patches at low flow was gradually replaced by faster, deeper hydraulic patches at high flow illustrating the effect of discharge on the availability of different hydraulic patch types. The spatial arrangement of patches, quantified using a range of spatial metrics from the field of landscape ecology at two spatial scales (class and reachscape), was relatively invariant to changes in discharge suggesting that the configuration of the hydraulic patch mosaic is determined by channel morphology and remains stable between channel forming discharges. The majority of hydraulic patch types occurred in relatively fixed locations in the channel, moving relatively small distances as discharge increased, associated with the gradual expansion or contraction of patch area. The results suggest that sub-bankfull flow variations will primarily affect the composition rather than the configuration of hydraulic patches, however large fluctuations are likely to result in high rates of patch turnover (change in location), with potential implications for instream biota. The hydraulic patch/transition zone model of the hydraulic environment provides a new approach for exploring the link between physical and biological heterogeneity in the instream environment, including the role of instream ecotones. Whilst the application of numerical classification is currently limited by the large hydraulic data requirement, future advances in remote-sensing technology and hydrodynamic modelling are likely to widen its applicability at a range of spatial scales. The results highlight the need for further research on the ecological significance of hydraulic patches and transition zones and ecological sensitivity to changes in hydraulic patch configuration. Wider application of the landscape ecology approach to hydraulic habitat assessment in different reach types is recommended to improve understanding of the links between geomorphic and hydraulic diversity.

ACKNOWLEDGEMENTS

There are many people without whom it would not have been possible to complete this research and who deserve my deepest thanks. I am particularly grateful for the unfailing support, encouragement and patience of my supervisors Dr. Ian Maddock, Dr. Fleur Visser and Prof. Mike Acreman. Heartfelt thanks go to the army of volunteers who willingly spent long hours in cold rivers helping to collect data, particularly to James Atkins for expert technical and fieldwork support. A special thank you is reserved for Martin Wilkes for countless fruitful discussions, much-needed encouragement in the final stages and help getting this thesis onto paper. The support from long-suffering friends and loved ones throughout this long and stressful journey has been invaluable; thank you Helen Diss, Nell Brigham, Katie Milburn, Michael Pluess, Tanya Ellis-Lawes, Jo Cavan and Matt Shaw for providing shoulders to cry on, ears to bend, understanding hearts and welcome distractions through the good times and the bad.

I dedicate this thesis in loving memory to my parents who would have been so proud.

CONTENTS

LIST OF FIGURES

LIST OF TABLES

CHAPTER 5 HYDRAULIC HETEROGENEITY: PATCH RICHNESS, FREQUENCY AND DIVERSITY

"When we try to pick out anything by itself, we find it hitched to everything else in the Universe."

John Muir, My First Summer in the Sierra, 1911

1

Introduction

Chapter overview

This chapter explains the context of the doctoral research project, introduces the project's themes and sets out the specific objectives and research questions it seeks to address. Section 1.1 explains why classifying the hydraulic environment and evaluating its heterogeneity is relevant to river science and river management. The following two sections (1.2-1.3) explain the current approaches to classifying the in-stream environment and identify their shortcomings in terms of quantifying hydraulic heterogeneity. Section 1.4 reviews innovative quantitative methods for delineating hydraulic patches and outlines the potential advantages of the method used in this project. This is followed by a short review of the ecological significance of hydraulic heterogeneity (Section 1.5) and a discussion of how hydraulic heterogeneity can and should be quantified (Section 1.6). The final section (1.7) sets out the overall aim of the research project, identifies the objectives of each chapter and illustrates the structure of the thesis.

1.1 River management, river science and water resources legislation

In their natural state rivers are inherently dynamic, heterogeneous systems that provide valuable ecosystem and human services (Townsend, 1989; Ward, 1989; Pearce, 1998). However, centuries of inappropriate channelization, flow regulation, and pollution have degraded the world's river systems, causing a decline in physical heterogeneity and biodiversity (Schoof, 1980; Ward, 1998; Negishi et al., 2002; Bunn & Arthington, 2002; Downs & Gregory, 2004; Søndergaard & Jeppesen, 2007; Poff & Zimmerman, 2010). In 2005, over 50% of the world's large river systems had been impacted by dams (Nilsson et al., 2005) and the decline in freshwater biodiversity was evident (Moyle & Leidy, 1992; Harrison & Stiassny, 1999). With climate change and population growth predicted to exacerbate existing pressures on water resources (Postel et al., 1996; Millenium Ecosystem Assessment, 2005; Wilby, 2006), there is an urgent need for holistic, sustainable management of freshwater ecosystems (Baron et al., 2002; Gerten, 2008).

This has driven the development of a more integrated approach to river science that draws on a wide range of physical sciences, engineering and computing technology to assess the structure and functioning of freshwater ecosystems (Thoms & Parsons, 2002; Rice et al., 2010). Understanding river ecosystems often requires an integrated or collaborative approach, interdisciplinary tools and frameworks. In particular, three interdisciplinary subject areas – ecohydrology/ecohydraulics, ecogeomorphology and hydromorphology - have emerged to address research questions spanning the domain, methods and scales of the traditional pillars of river science (Figure 1.1). Ecohydrology (Zalewski et al., 1997), or hydroecology (Hannah et al., 2004), and its subdiscipline ecohydraulics (Leclerc 1994; Nestler et al., 2008; Lancaster & Downes, 2010) refer to research linking hydrology (flow, hydraulics) and biological patterns and processes. Ecogeomorphology refers to the relationship and feedback between biological and geomorphic features and processes (Frothingham et al., 2002; Wheaton et al., 2011). The term has also been used to describe research linking ecology, fluvial geomorphology and hydrology, with particular emphasis on bridging the scales of features and processes pertinent to each discipline (e.g. Thoms & Parsons, 2002). Hydromorphology, a term first introduced by the EU Water Framework Directive in 2000, is now an established interdisciplinary research area aimed at investigating the interaction between hydrology and fluvial geomorphology across multiple nested scales (Table 1.1), although the term usually refers to the interaction between the hydrological regime and channel morphology as this is the typical scale of management (European Commission, 2000; Newson & Large, 2006; Orr et al., 2008; Sear, 2009; Vogel, 2011).

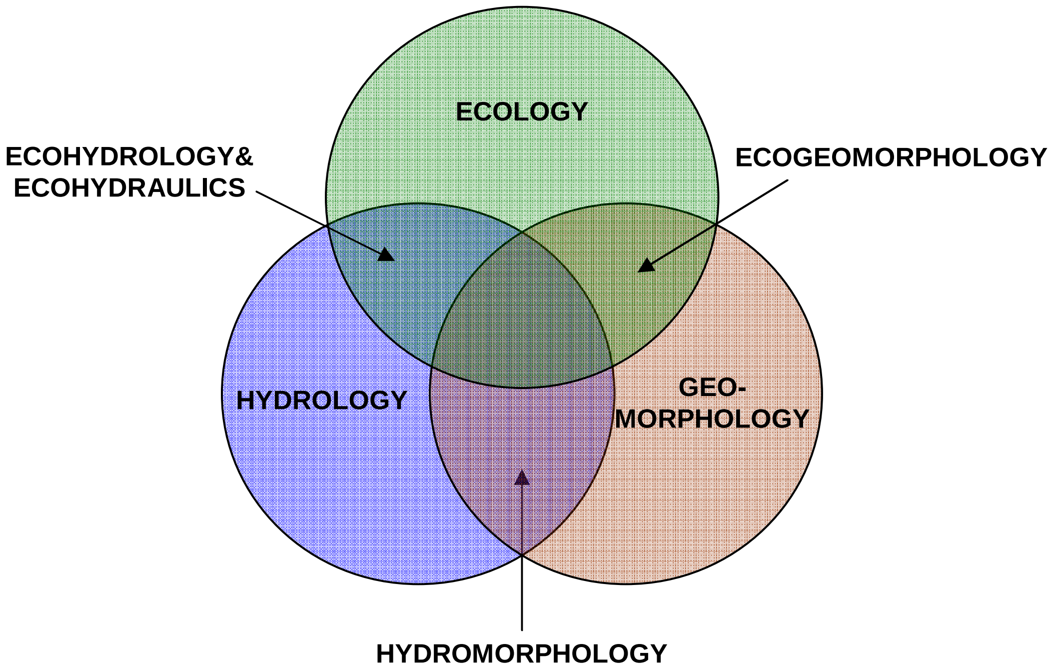


Figure 1.1. The interdisciplinary scope of river science.

Table 1.1. Spatio-temporal scale hierarchy of stream hydrology and geomorphology (adapted from Frissell et al. (1986) and Dollar et al. (2007)).

Scale	Hydrology	Timescale (years)	Geomorphology	Timescale (years)	Spatial Scale (metres)
Micro	Local fluid mechanics	10^{-6}-10^{-3}	Individual particles	10^{-6}-10^{-3}	$<10^{0}$
	Channel hydraulics	10^{-3}-10^{0}	Substrate patch	10^{-1}-10^{0}	10^{-1}-10^{0}
Meso			Bedform topography	10^{0}-10^{1}	10^{0}-10^{1}
	Hydrological regime	10^{0}-10^{2}	Reach type/channel planform	10^{0}-10^{2}	10^{2}-10^{3}
Macro	Historical climate	10^{2}-10^{4}	Segment	10^{2}-10^{4}	10^{3}
			Drainage basin	10^{4}-10^{6}	10^{5}-10^{6}
	Paleoclimate	$>10^{4}$	Geomorphic province	$>10^{6}$	10^{7}-10^{9}

Evidence from multidisciplinary research linking ecological, hydrological and geomorphological processes has engendered a gradual shift in river channel management and policy over the last few decades (Mitsch et al., 2002; Zalewski, 2002; Giller, 2005; Palmer et al., 2005; Wohl et al., 2005; Moss, 2007). For example, a growing body of research highlighting the significant role of natural flow dynamics (e.g. Junk et al., 1989; Poff et al., 1997) and physical heterogeneity (e.g. Ward, 1998; Thorp et al., 2006) in maintaining biodiversity and ecosystem integrity has provided the impetus to introduce environmental flow management (Richter et al., 2003; Tharme, 2003; Poff et al., 2010), to restore river channels (Brooks & Shields, 1996) and improve floodplain connectivity (e.g. Buijse et al., 2002; Hulea et al., 2009). Hydrological and geomorphological features shown to have ecological value have been increasingly integrated into water resource policies, offering legal protection to physical habitats (e.g. EU Habitats Directive, 1992) and water quantity (e.g. UK Water Act, 2003) in addition to water quality (Gleick, 1998). The most significant shift in water resources policy in Europe was introduced by the EU Water Framework Directive (2000/60/EC) (WFD) which introduced a statutory requirement to prevent further deterioration to EU freshwaters and, where feasible and economically reasonable, to restore river channels with the aim of achieving “good ecological status” by 2015 (Chave, 2001; Logan & Furse, 2002). The WFD explicitly recognised the link between hydromorphological conditions (defined as the quantity and dynamics of flow, river depth and width variation, connection to groundwater bodies, structure and substrate of the river bed, the structure of the riparian zone and river continuity (UK TAG, 2008)) and ecological status. Meeting WFD objectives has encouraged an eco-hydromorphic approach to river management (Raven et al., 2002; Clarke et al., 2003; Wharton & Gilvear, 2006; Large & Newson, 2002; Newson & Large, 2006; Vaughan et al., 2009).

Current UK standards for hydromorphology, based on expert judgement, suggest 'good ecological status' can be maintained in rivers where the natural flow is reduced by between 10-30% and/or morphological alterations (to bed and banks) use ≤25% of the system's capacity to absorb change (UK TAG, 2008; Acreman & Ferguson, 2010). However additional empirical testing of these standards and the development of standard assessment methods is needed (UK TAG, 2008; CEN, 2008; Boon et al., 2010). The interaction between channel morphology and discharge produces a spatially heterogeneous and temporally dynamic hydraulic environment which provides the living space, or 'hydraulic habitat' for biota (Maddock, 1999). By measuring and characterising the hydraulic environment, it is possible to evaluate hydromorphological conditions at an ecologically-relevant scale, which has the potential to improve our understanding of the relationship between ecology and hydromorphology and strengthen the scientific basis for hydromorphological standards and river restoration (Petts et al., 2006; Renschler et al., 2007; Sear, 2009; Vaughan et al., 2009).

1.2 Characterising the in-stream (hydraulic) environment

Characterising the hydraulic environment, particularly classifying hydraulic units and quantifying hydraulic heterogeneity, is not a straightforward task given the complex and dynamic nature of the flow field. At the meso scale, lateral (across channel), longitudinal (upstream-downstream) and vertical (channel bed to water surface) variations in water depth, substrate and velocity occur over the scale of a few centimetres to tens of metres and temporally over the scale of minutes to a year (Ward, 1989). Nested within these are similar hydraulic variations at the micro scale (e.g. Buffin-Belanger & Roy, 1998; Harvey & Clifford, 2009) however these are outside the scope of this study and will not be discussed further. Hitherto, the meso scale hydraulic environment has been characterised in one of two ways; either according to its hydraulic and/or geomorphic characteristics (hydromorphological approach), or in terms of its biological function (ecohydraulic approach). These two approaches are described in the following sections.

1.2.1 Hydromorphological approach

Characterising the in-stream environment in terms of channel form is an intuitive and appealing approach since it provides the spatial template for hydraulic conditions. Viewed from a catchment-scale perspective, many geomorphic variables follow a continuous gradient from headwaters to estuary (Figure 1.2). Leopold and Maddock (1953) described the downstream increase in depth, velocity and bankfull width from source to mouth. This gradual change in physical conditions was later linked to the structural and functional change in stream communities in the River Continuum Concept (Vannote et al., 1980). Disruptions to longitudinal variations were subsequently accounted for in Ward & Stanford's (1983) Serial Discontinuity Concept. Frissell et al. (1986) recognised that when the stream system is viewed at different spatial scales, a discontinuous structure emerges, superimposed on the continuous downstream gradient. Frissell et al. (1986) proposed a hierarchical classification of the geomorphic patterns evident at a range of discrete spatio-temporal scales nested within the catchment (Figure 1.3, p.8). Since then several geomorphological typologies of valley segments, channel reaches and channel units (e.g. pools and riffles) have emerged (e.g. Bisson et al., 1982; Rosgen, 1994; Montgomery & Buffington, 1997; Bisson et al., 2006; Orr et al., 2008).

Attention has focused on characterising geomorphic units at the meso (pool-riffle) scale on the assumption of its ecological relevance (e.g. Bisson et al., 1982; Hawkins et al., 1993). Meso scale classifications reflect variations in bed topography and differentiate areas with different mean depth, velocity and substrate (Bisson et al., 2006). Early classifications simply differentiated between pools (topographic lows) and riffles (topographic highs) (O'Neill & Abrahams, 1984), but later incorporated intermediate units (runs and glides) (Jowett, 1993). Hawkins et al. (1993) proposed a hierarchical classification of 19 channel geomorphic units (CGUs) which could be applied at three levels of resolution according to the needs of the study (Table 1.2).

For example, pools can be differentiated according to their formative process - scour or damming, and then by more detailed characteristics such as location in the channel, cross-sectional and longitudinal profile, substrate characteristics and constraining formative feature.

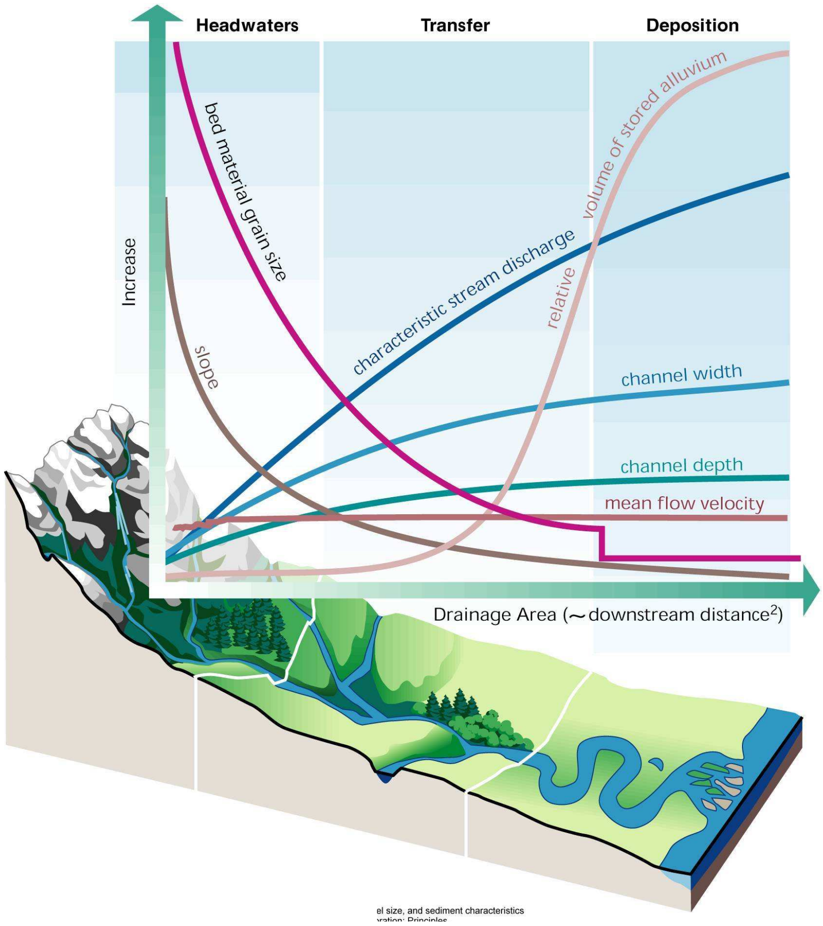


Figure 1.2 Catchment-wide continuous physical gradients (Source: FISWRG, (1998), after Church (1992) and Schumm (1977)).

CGUs are useful for describing meso scale *spatial* hydraulic variations at a given flow, but are less useful for describing *temporal* hydraulic variations because their hydraulic function is discharge-dependent (Wadeson & Rowntree, 1998; Clifford et al., 2002; Bisson et al., 2006). For example, according to the velocity reversal hypothesis (Keller, 1971; Thompson et al., 1999) near-bed velocities become higher in pools than riffles at flows approaching bankfull. Emery et al. (2003) also showed that as discharge increases, velocity increases in pools and riffles, decreases in backwaters and remains stable in channel margins and steep riffle crests. However, the magnitude and rate of change in velocity varies between sites (Emery et al., 2003). Furthermore, the point at which the influence of bedforms on the hydraulic environment is drowned out altogether depends on the bedform amplitude (Emery et al., 2003). Recent research has also highlighted considerable hydraulic overlap between different CGUs (e.g. Principe et al., 2007) as well as inconsistencies in the hydraulic character of CGU types between sites and along a reach (Pedersen & Friberg, 2006), casting doubt on their ability to differentiate or predict hydraulic conditions very precisely. Some researchers have bypassed the hydraulic environment, linking geomorphic condition directly with biodiversity (e.g. Chessman et al., 2006), however restoring structural (geomorphic) heterogeneity does not necessarily increase biodiversity (Lepori et al., 2005; Sundermann et al., 2011). Further research is needed to elucidate the relationship between geomorphic condition and hydraulic habitat (Clark et al., 2008). In summary, predicting hydraulic conditions from channel morphology is riddled with difficulties and a useful methodology remains elusive (Clifford et al., 2006). As a result, CGUs have gradually been superseded by *hydro*morphic units which reflect discharge-related changes more effectively, thus enabling an appraisal of the dynamic character of the in-stream environment (Padmore, 1997; Rowntree & Wadeson, 1999).

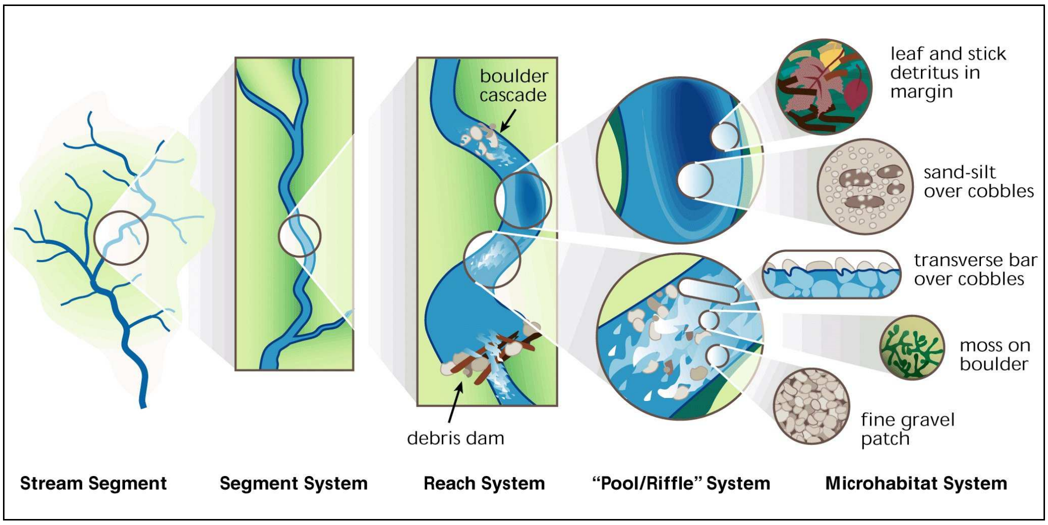


Figure 1.3. Nested hierarchy of stream habitats (Source: adapted from FISWRG (1998), after Frissell et al., 1986).

Table 1.2. Comparison of meso scale classifications of geomorphic units and hydromorphic units.

GEOMORPHIC Hawkins et al. (1993)	HYDROMORPHIC Rowntree (1996)		Padmore (1997)	
Channel geomorphic units (CGU)	Hydraulic Biotopes	Associated surface flow type (SFT)	Physical Biotopes	Associated surface flow type (SFT)
Fall	Waterfall	Free fall	Waterfall	Fall
Cascade Chute	Cascade	Free fall	Cascade	Chute
Rapid	Rapid	Unbroken standing wave/Broken standing wave	Cascade/rapid	Broken standing waves
Riffle	Riffle		Riffle	Unbroken standing waves
Sheet Run	Run	Rippled	Run	Rippled
-	Chute	Smooth boundary turbulent	-	-
-	Glide		Glide	Smooth boundary turbulent
Pool(s)	Pool	Barely perceptible flow	Pool	Scarcely perceptible flow
-	Slackwater	No perceptible flow	Marginal deadwater	
Backwater Abandoned channel	Backwater	Upstream eddies	-	-
-	Boil	Boil (vertical flow)	Boil	Upwelling
-	-	-	Multiple biotopes	Chaotic

Hydromorphic units include physical/hydraulic biotopes which were proposed as hydraulically discrete units, reflecting variations in depth, velocity and substrate, that a trained observer could visually differentiate by surface flow (SFT) characteristics (Table 1.2) (Rowntree, 1996; Padmore, 1998). Biotopes reflect hydraulically homogeneous areas nested within CGUs and so typically occur at a smaller spatial scale (Wadeson & Rowntree, 1998; Thomson et al., 2001). Results of biotope mapping over a range of flows indicate that the hydraulic environment becomes increasingly homogeneous at high discharges as the influence of substrate roughness and bedform topography on channel hydraulics is 'drowned out' (*sensu* Padmore, 1997). It has been suggested that mapping biotopes provides a cost-effective bridge between micro scale hydraulic variations pertinent to biota and catchment-wide mapping required for river management plans (Raven et al., 2000; Newson et al, 1998b).

Biotopes are currently used as the standard unit of physical habitat in the UK River Habitat Survey (Raven et al., 1997). Biotope mapping is rapid assessment method and a trained operator can map several kilometres of river per day which is advantageous in the move towards catchment-scale management. The disadvantage of this approach is the inevitable simplification of hydraulic complexity resulting from subjective, visual assessment from the river bank. Furthermore, the classification was used for many years on the assumption that physical biotopes were ecologically significant; only relatively recently has this assumption been tested and evidence is inconclusive (e.g. Rabeni et al., 2002; Heino et al., 2004; Principe et al., 2007). A full discussion of the limitations of physical biotopes as a classification of physical habitat is included in Section 1.3. However, it can be appreciated that a more detailed, quantitative approach to physical habitat assessment is necessary to represent the complexity of the hydraulic environment more faithfully. One such alternative is ecohydraulic modelling which is explained in the next section.

1.2.2 Ecohydraulic approach

Ecohydraulics emerged as a new sub-discipline of river science during the last decade and is specifically concerned with examining the interface between ecology and hydraulics/hydraulic engineering (Nestler et al., 2007; Rice et al., 2010). The ecohydraulic approach to physical assessment is driven by an organism-centred perspective whereby the hydraulic environment is characterised in terms of its biological function or suitability for a target species. At the heart of this approach is the concept of habitat; the physical resources necessary to sustain a species, community or ecological process (Southwood, 1977, 1988; Poff & Ward, 1990). Ecologists have demonstrated the significance of the separate and interactive effects of depth, velocity and substrate to the basic life processes, abundance and distribution of freshwater biota (Hynes, 1970; Statzner et al., 1986; Statzner & Higler, 1988; Lancaster et al., 1990; Kershner & Snider, 1992; Quinn & Hickey, 1994; Hart & Finelli, 1999; Malmqvist, 2002; Armstrong et al., 2003; Jowett, 2003; Riis & Bigs, 2003; Gordon et al., 2004; Gillette et al., 2006; Post et al., 2007; Schwartz & Herricks, 2008; Mérigoux et al., 2009; Lobón-Cerviá et al., 2011). This has established hydraulic variables as ecologically-meaningful habitat descriptors and laid the foundation for a ‘habitat hydraulics’ (NIT, 1994) or ecohydraulics (Leclerc et al., 1996) approach to physical habitat assessment.

Ecohydraulic modelling was specifically designed as a quantitative tool to predict how flow regulation would affect the availability of suitable habitat for target fish species and, by inference, biomass (Milhous, 1979; Bovee, 1982). In this approach habitat is defined by the depth, velocity and substrate preferences of the target species, identified by the hydraulic conditions at locations where the species has been observed. Preferences are converted to habitat suitability curves for each hydraulic variable. These indicate the suitability of a particular depth, velocity or substrate class for the target species on a scale of 0-1, where 1 is ideal (Figure 1.4). The second component of ecohydraulic modelling is the simulation of depth and velocity at a range of different discharges using a hydrodynamic model. Depth and velocity can be predicted throughout the channel under different flow scenarios from a detailed survey of channel topography and specification of boundary conditions using the Navier-Stokes equations describing the motion of fluids and the principles of

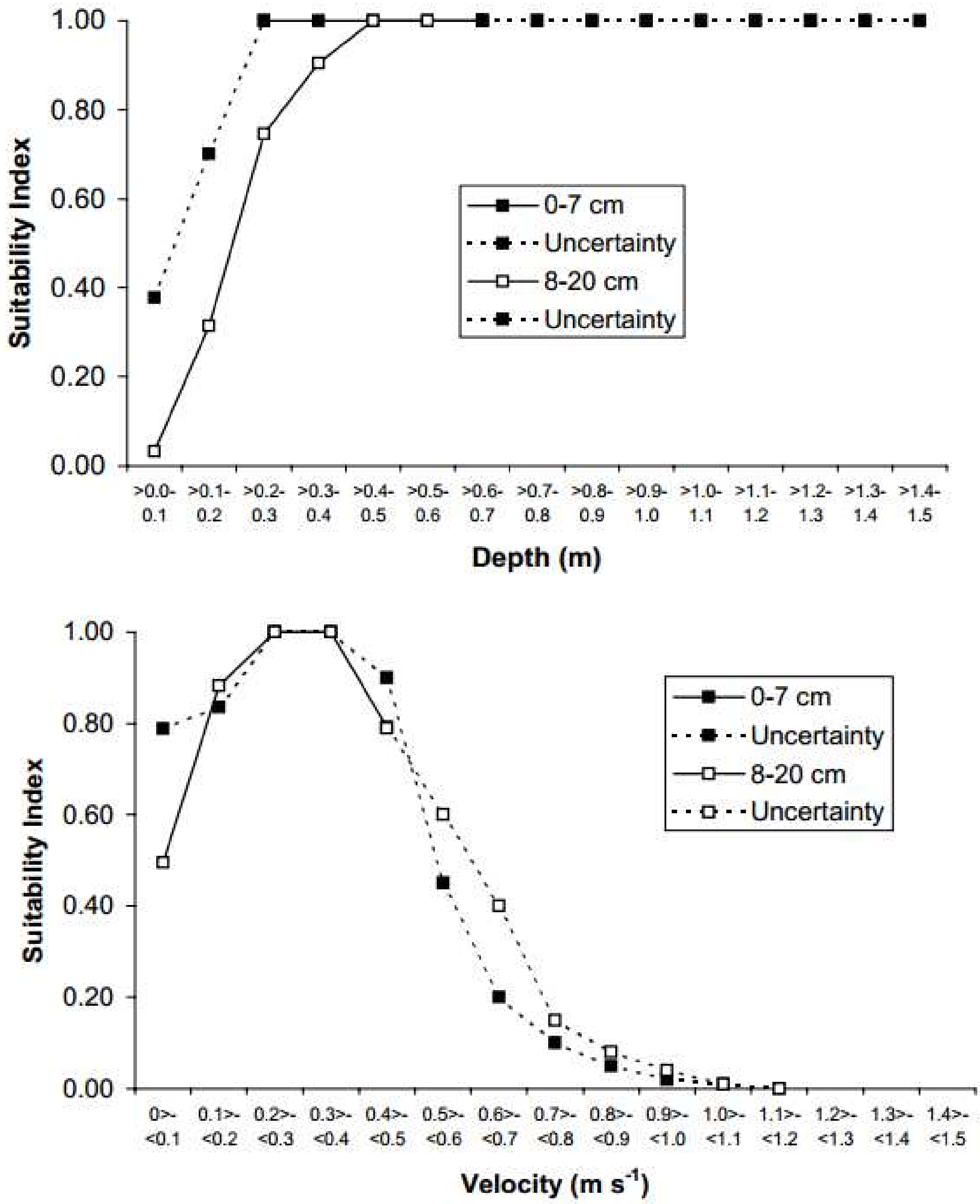


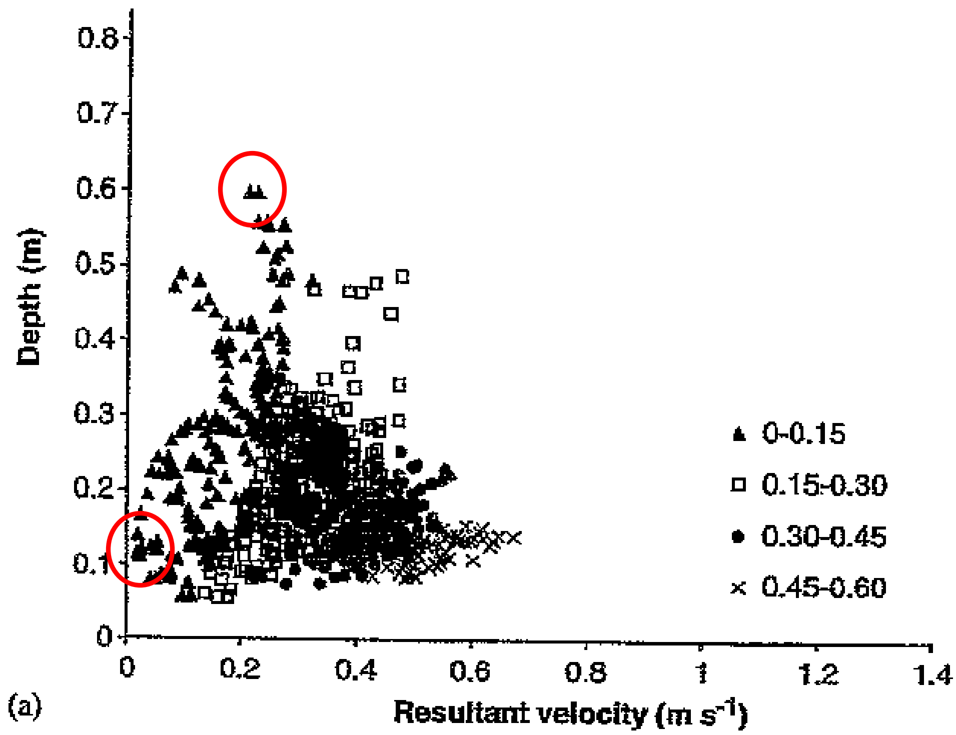
Figure 1.4. Example of generic habitat suitability curves for depth (m) and velocity (ms^{-1}) preferences of juvenile brown trout (Centre for Ecology and Hydrology, 2001).

conservation of momentum and mass. Depending on the sophistication of the model, velocity can be predicted in one (e.g. PHABSIM (Milhous et al., 1984)), two (e.g. River 2D (University of Alberta, 2002)) or three (e.g SSIM (Olsen, 1996)) dimensions. Spot-check field measurements of depth and velocity are used to calibrate and validate the model. Habitat is delineated by combining hydraulic preferences with simulated hydraulic data using an ecohydraulic model, such as PHABSIM, MesoHABSIM and CASiMiR (Milhous et al., 1984; Jorde, 1996; Parasiewicz, 2001). Model output is expressed as the quantity (m^2) of suitable habitat (weighted usable area (WUA)) in a given reach at a range of different discharges. This approach is typically used to predict fish habitat but has been extended to habitat for a range of flora and faunas as hydraulic preference data becomes available (e.g. Mérigoux et al., 2009). Despite advances to modelling techniques ecohydraulic modelling as an approach to classifying the hydraulic environment, has several weaknesses which are outlined in the following section.

1.3 The limitations of current physical habitat classifications and assessment methods for characterising the hydraulic environment

1.3.1 Visual assessment of physical biotopes

The likelihood of reflecting the true hydraulic conditions using the physical biotope classification has been questioned on several grounds. The original tests of the hydraulic distinctiveness of SFTs and biotopes used a range of hydraulic indices including Froude number, Reynolds number, shear velocity and roughness Reynolds number (Padmore, 1997; Wadeson & Rowntree, 1998). All SFTs/biotopes were shown to be significantly different according to Froude number with the exception of riffle–run types and rapid–cascade types (Padmore, 1997; Wadeson & Rowntree, 1998). More recent work also showed that SFTs reflect depth variability and cross-section geometry (Froude number) well and so provide a rapid indicator of channel morphology (Zavadil et al., 2012). However Clifford et al.'s (2006) re-analysis of the biotope concept showed that whilst SFTs may reflect distinct Froude number classes, they do not differentiate between distinct depth-velocity combinations (Figure 1.5). Clifford et al. (2006) concluded that SFTs are crude indicators of hydraulic conditions and are not a reliable substitute for detailed measurement.



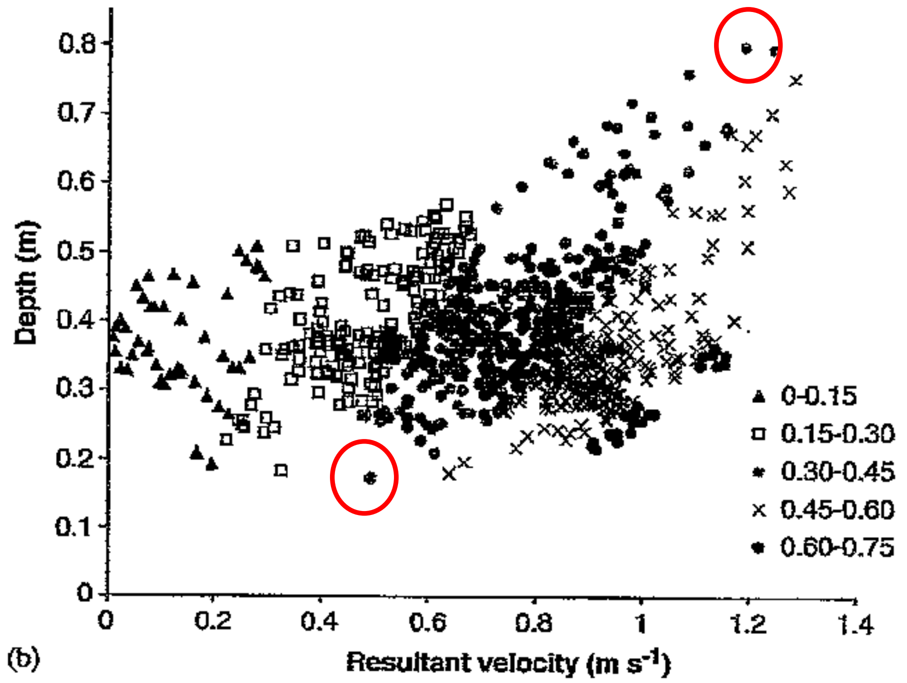


Figure 1.5. Scatterplot of depth-velocity data collected at the River Cole at (a) low flow and (b) high flow illustrating the failure of Froude number class to differentiate very different depth-velocity combinations (Source: Clifford et al., 2006). [Annotations added].

Moir & Pasternack (2008) also found a high degree of overlap in Froude number between different hydromorphological units and proposed the joint distribution of depth and velocity as a more reliable habitat descriptor. In addition to doubts about the hydraulic distinction between *different* biotope types, the hydraulic character of a *single* biotope type can vary considerably between reaches and rivers (le Coarer, 2007; Pedersen & Friberg, 2007). Furthermore, Wadeson & Rowntree's (1998) test of the consistency of biotope hydraulic characteristics between sites and between flows showed that pools and riffles show a significant degree of inter-site hydraulic variability and that pools have significantly different hydraulic characteristics (according to Froude number, Reynolds number, shear velocity and roughness Reynolds number) at low, intermediate and high flows.

The ability to reflect the true complexity of the in-stream environment is not only affected by the hydraulic distinctiveness of the classification itself, but also by the method of assessment, namely, visual surveys. Whilst this method is rapid, practical and cost-effective, it is also subjective and suffers from operator variability, particularly when observers have insufficient training or the habitat classification system contains a large number of habitat types (Hawkins et al., 1993; Roper & Scarnecchia, 1995; Poole et al., 1997). Roper et al. (2002) showed operators accounted for 1-56% of the variation in stream evaluation results. Variability worsened where conditions were close to classification boundaries (Eisner et al., 2005). Furthermore, anthropogenic scale bias is unavoidable in visual surveys, with the result that small-scale variations that cannot be distinguished easily tend to be misclassified or underrepresented, leading to oversimplification of hydraulic complexity. This may be a particular problem where the observer maps SFTs from one bank and has an oblique view of features on the far side. Human error aside, Padmore (1997) acknowledged that SFTs do not identify physical biotopes correctly in all cases – 80% accuracy was reported at high flow but this figure decreased at low flows due to the greater influence of substrate-roughness (as opposed to bedform topography) on hydraulics. In recognition of observer error the standard protocol for assessing stream-flow character in the UK River Habitat Survey is to identify dominant SFT at spot check locations and record whether SFTs are absent, present (occupies 1-33% of the total reach length) or extensive (occupies >33% reach) which provides a relatively coarse reflection of in-stream hydraulic diversity. Poole et al. (1997) argued that the subjectivity and lack of reproducibility associated with visual assessment of physical biotopes render this approach inappropriate for monitoring temporal change. Recent research has demonstrated the potential for assessing surface flow conditions more objectively and extensively using remotely-sensed images or terrestrial laser scanning (e.g. Gilvear et al., 2008; Legleiter et al., 2004; Large & Heritage, 2007; Marcus & Fonstad, 2008; Marcus & Fonstad, 2010) which might reduce error associated with visual assessment, however it is not yet possible to measure velocity directly using these techniques.

1.3.2 Ecohydraulic modelling

The success of hydraulic habitat modelling predictions depends on the quality of input data (habitat suitability curves and simulated hydraulic conditions) and the realism of the model itself (i.e. whether biomass can be predicted from physical habitat alone and/or complex real-world flow fields can be predicted from theoretical equations). Although ecohydraulic modelling is widely used, it has had mixed success predicting biomass from WUA (Shirvell, 1989; Milhous, 1999; Kondolf et al., 2000). Critics of the method have challenged the biological validity of habitat suitability curves, suggesting they routinely fail to reflect the complex biological interactions influencing community structure (Mathur et al., 1985; Lancaster & Downes, 2010). Studies have shown that habitat preferences are very flexible, changing in response to life-cycle stage (e.g. Schiemer et al., 2001; Riley et al., 2006; Harby et al., 2007; Remshardt & Fisher, 2009), sex (e.g. Greenberg & Giller, 2001), the presence of predators or competitors (e.g. Werner et al., 1983; Degerman et al., 2000), diel cycle (e.g. Davey et al., 2011), discharge (e.g. Gillette et al., 2006; Lamouroux et al., 2006), site (e.g. Leftwich et al., 1997) and spatial scale (Lee & Suen, 2011). However in practice generic curves based on data averaged across life-cycle stage and site are often used which can overestimate WUA (Waite & Barnhart, 1992; Williams, 2010). Moir et al. (2005) found that selecting an inappropriate habitat suitability curve for local environmental conditions critically affected the accuracy of PHABSIM predictions. Furthermore, habitat suitability curves are often developed separately for each hydraulic variable so fail to incorporate significant interactive effects (Mathur et al., 1985), which can lead to unrealistic habitat predictions (Parasiewicz & Walker, 2007). Gore & Nestler (1988) recommend using site-specific suitability curves based on joint depth-velocity preferences.

The bias caused by the 'observer effect' during habitat preference/use surveys is not taken into account when constructing habitat suitability curves, despite its potential impact on fish behaviour (Bain et al., 1985). Fuzzy rule-based models (e.g. CASiMiR) developed to allow a more flexible definition of habitat suitability (Jorde, 1996; Schneider et al., 2001) and have proved more successful in predicting fish densities (Schneider & Jorde, 2003). To improve the biological validity of ecohydraulic modelling further it would be necessary to incorporate biological factors affecting habitat use into the model, such as such as primary activity (feeding, resting etc.), competition and predator-prey interactions (e.g. Baker & Coon, 1997). However given the difficulty of collecting such data this might not prove time- or cost-effective (Orth, 1987; Gore & Nestler, 1988; Milhous, 1999).

The accuracy of hydraulic modelling has also been questioned (Kondolf et al., 2000; Ho et al., 2003). Accuracy of simulated hydraulic data is affected by the type of model used, the quality and spatio-temporal resolution of the input data, how well the model is calibrated and the complexity of the flow field being modelled. Criticisms of the crude, 1D approach of early models (Shirvell, 1989; Crowder & Diplas, 2000; Brown & Pasternack, 2009) have been addressed to some extent by the introduction of sophisticated 2D and 3D computational fluid dynamics models capable of modelling spatially complex flows (Leclerc et al., 1995; Ghanem et al., 1996; Hardy et al., 2000; Crowder & Diplas, 2000; Rodriguez et al., 2004; MacWillams et al., 2006; Nestler et al., 2007). However, as model complexity increases, so too does the input data requirement (e.g. steady-state 3D velocity calibration data, high resolution channel bathymetry, increased quantity and quality of boundary condition specification) (Rodriguez et al., 2004). Whilst technological advances have made this possible (Lane et al., 1998; Rhoads & Sukhodolov, 2001; Fonstad & Marcus, 2005; Lane & Carbonneau, 2007; Lejot et al., 2007), the time, cost, resources and degree of expertise required to collect and process such data increase considerably, limiting the appeal of complex ecohydraulic modelling to river managers (Nestler et al., 2007). In most cases 1D PHABSIM models are typically applied.

In a review of PHABSIM studies, Williams (2010) draws attention to spatial sampling issues, urging users to adopt a rigorous random sampling approach to avoid biased WUA predictions. Ecohydraulic modelling involves a trade-off between length of reach modelled and the level of detail achieved; overcoming scale limitations is necessary to meet catchment-scale management objectives (Parasiewicz, 2003). Complex 3D modelling is limited to sub-reach scales (Wheaton, 2008) but models designed for longer reaches (e.g. MesoHABSIM (Parasiewicz, 2001)) relax the spatial resolution of input data, potentially failing to capture the full complexity of the flow field at the habitat scale (Scruton et al., 1998; Crowder & Diplas, 2002). Recent research has highlighted the significant influence of submerged vegetation on velocity distributions (e.g. Chen & Kao, 2011); hydrodynamic models would need to incorporate equations capable of reflecting these altered patterns to ensure accurate velocity predictions.

At best, ecohydraulic models provide a simplified estimate of reality and will undoubtedly contain errors (Kondolf et al., 2000). Schweizer et al. (2007) caution that modelling should not become a substitute for empirical studies; field methods must be progressed equally alongside modelling (Clifford et al., 2005). Ecohydraulic modelling is a potentially useful tool for river management *if* it can deliver accurate, quantitative predictions of habitat efficiently and cost-effectively. However, much more biological data about habitat preferences are required to improve modelling success; indeed greater inclusion of ecological theory is needed to progress ecohydraulics (Lancaster & Downes, 2010). Whilst the integration of biological and simulated hydraulic data is ultimately desirable, at the present level of understanding attempts appear premature. Kondolf et al. (2000) caution that ecohydraulic modelling predictions should not be used as a substitute for actual biological information. Furthermore, unless it is possible to incorporate species-*interactions* into ecohydraulic models, their use will be limited to single species-based management. As the responsibility of river managers shifts from managing target species for “sport” or conservation to improving overall community structure, a more comprehensive approach to habitat assessment is increasingly required (Downs & Gregory, 2004; Wohl et al., 2005; Clifford et al., 2005; Dudgeon et al., 2005).

In summary, it has been suggested that there is considerable potential to obscure or misrepresent hydraulic differences by visually assessing physical biotopes. The alternative ecohydraulic approach of characterising the hydraulic environment in terms of its biological suitability provides a limited, species-specific view of hydraulic differences. Scale differences between channel morphology, fish habitat and hydraulic patches complicate defining the hydraulic environment in terms of other variables (underlying morphology or biological suitability) (Post et al., 2007; Renschler et al., 2007). Instead it might be more appropriate to quantitatively characterise the hydraulic environment *in and of itself* before making the links with hydromorphology and biology (Kondolf et al., 2000).

1.3.3 Further limitations common to both approaches

Assessment of the hydraulic environment based on visual surveys of physical biotopes or ecohydraulic modelling focuses on measuring the quantity and quality of hydromorphic units/hydraulic habitat at different discharges (e.g. Dyer & Thoms, 2006; Shoffner & Royall, 2008) but pays little attention to the spatial structure of the hydraulic environment. Biotope mapping assesses longitudinal variation (sequences of units) but provides little detail about cross-sectional variations and does not incorporate any specific analysis of the spatial structure of habitat units (le Coarer, 2007). Padmore (1997) recognised that the physical biotope concept would be strengthened by identifying typical biotope sequences and developing patchiness and diversity indices to quantify biotope configuration. Ecohydraulic habitat models calculate the total usable area for a target species over a range of discharges but do not "…define its locations or spatial dynamics" (Emery et al., 2003, p.534). Guisan and Zimmermann (2000) criticise the implicit assumption of ecohydraulic models for, "statistically relat[ing] the geographical distribution of species or communities to their present environment" (p.147) and failing to reflect the spatial context of suitable habitat (Jorde et al., 2001; Crowder & Diplas, 2002), despite the suitability of this approach for spatial analysis (Bovee, 1996). Clark et al.'s (2008) study of the spatial distribution of WUA of fish habitat in Vermont streams is a notable exception. Clark et al. (2008) demonstrated that the spatial pattern of WUA differed in all 6 streams surveyed and showed that fish biotic integrity was linked to the spatial distribution of WUA. Such studies highlight the importance of integrating spatial analysis into habitat assessment.

1.4 Towards a quantitative classification of the hydraulic environment

Classifying the hydraulic environment is not a straightforward task due to its complex, heterogeneous and dynamic nature. The hydraulic environment is a continuum; a gradient between deep-slow, deep-fast, shallow-fast and shallow-slow hydraulic extremes. Like many environmental variables, depth and velocity vary through space and continually change over time, making hydraulic measurements very stage-dependent (Clifford et al., 2002; Emery et al., 2003) and resulting in spatially ambiguous and dynamic boundaries between zones of different hydraulic character. This section of the chapter explores methods of measuring the hydraulic environment and delineating relatively homogeneous hydraulic patches.

1.4.1 Selecting appropriate hydraulic variables

The hydraulic environment can be characterised by numerous simple and complex hydraulic variables, either separately or in combination. Current velocity (at various depths in the water column) and water depth are first order variables whose influence on biotic processes and distributions is well established (Hynes, 1970; Statzner & Higler, 1986; Vogel, 1994; Allan, 1995). These variables are commonly used separately to define fish habitat preferences (e.g. Vismara et al., 2001; Jowett, 2002), however it has been suggested that interactive effects, defined by the joint distribution of depth and velocity, have greater ecological relevance (Statzner et al., 1988; Kemp et al., 1999; Stewardson & McMahon, 2002; Crowder & Diplas, 2006; Schweizer et al., 2007; Moir & Pasternack, 2008; Ayllon et al., 2009). Velocity, measured at 0.6 x water depth below the water surface, is widely used as a measure of mean column velocity (e.g. Wadeson & Rowntree, 1998; Emery et al., 2003; Moir & Pasternack, 2008), and has relevance to fish (e.g. Holm et al., 2001) and also benthic invertebrates (e.g. Jowett, 2003). The computational difficulty of calculating joint preference factors in the early 1980s led to complex hydraulic variables such as Froude number[1], which were easier to calculate, being used to characterise the hydraulic environment (Statzner et al., 1988, p.344). Froude number is a dimensionless hydraulic index that reflects the ratio of inertial to gravitational forces on flow and differentiates between

[1] Froude number evaluates the ratio of inertial to gravitational forces and is defined as $U/(gD)^{0.5}$ where U=mean column velocity (ms^{-1}), g=acceleration due to gravity (ms^{-2}) and D=water depth (m).

tranquil (Fr<1.0) and rapid (Fr>1.0) flow (Carling, 1992). Although studies have indicated the ecological relevance of Froude number (e.g. Kemp et al., 2000), it does not reliably differentiate between combinations of depth and velocity (Figure 1.5) (Clifford et al., 2006). Other complex hydraulic variables, such as Reynolds, roughness Reynolds and shear velocity are more useful for quantifying micro scale flow turbulence resulting from the interaction between substrate, depth and velocity and the shear force acting on near-bed hydraulic environment (e.g. Davis & Barmuta, 1989; Marchildon et al., 2011).

Substrate class has often been used to define hydraulic patches/habitat (e.g. Beisel et al., 1998; Inoue & Nakano, 1999). Substrate influences the hydraulic environment by creating turbulent flow structures (Lawless & Robert, 2001; Lacey & Roy, 2008), however the spatial distribution of substrate size classes reflects previous high flows rather than present flow conditions (Lamberti & Resh, 1979). Substrate also provides foraging ground, spawning sites and flow refugia in addition to creating physical heterogeneity (Downes et al., 1997; Lancaster, 2000; Yarnell et al., 2006) and therefore influences biotic distributions. A recent study of Chinook salmon spawning habitat (Moir & Pasternack, 2008; 2010) found that whilst a particular substrate type is a pre-requisite for successful spawning, salmon select an area of substrate with the right flow to support the spawning process, suggesting the overriding significance of hydraulic variables. As substrate is a morphological variable and this study aims to quantify the hydraulic environment, substrate will not be used as a hydraulic patch descriptor. Instead hydraulic patches will be defined by the joint distribution of depth and streamwise mean column velocity (measured at 0.6 depth). However, its relevance as a descriptor of physical habitat is acknowledged.

Hydraulic conditions can be inferred from remotely-sensed images (e.g. Lane & Carbonneau, 2007), simulated using hydrodynamic modelling (e.g. Clifford et al., 2010) or measured directly in the field (e.g. Malcolm et al., 2008). Each method has advantages and drawbacks and no one method is suitable in all environmental conditions. For example, the accuracy of depth and substrate measurements inferred from remote-sensed images can be affected by turbidity or turbulence in the water column, reflections from the water surface or atmospheric distortion, or simply obscured by overhanging vegetation (Legleiter et al., 2004; Jordan & Fonstad, 2005;

Lane & Carbonneau, 2007). As already discussed, the accuracy of simulated hydraulic data is constrained by the assumptions of the theoretical equations used to model flow; in practice representing heterogeneous real-world conditions and ecological dynamics mathematically is very challenging (Chen et al., 2006; Nestler et al., 2007). As hydrodynamic modelling and classification of remotely-sensed data both rely on high quality field measurements for calibration and validation, it is essential that field measurements are advanced alongside newer technologies.

Until relatively recently field measurements have been limited to wadeable streams and flows, hence it was necessary to rely on hydrodynamic modelling to predict hydraulic conditions at high flows. However, recent advances in stream measurement instruments (e.g. Acoustic Doppler Current Profilers (ADCPs)) made high resolution, multi-discharge hydraulic surveys a possibility (Malcolm et al., 2008). Many river management organisations (e.g Environment Agency in England) now use ADCPs as standard for discharge measurements. In theory, extending their use to characterise the hydraulic environment could be a practical, affordable alternative to hydrodynamic modelling. However the application of this technology in small UK rivers still has several environmental limitations (Malcolm et al., 2008). The results of a pilot study investigating the suitability of using a shallow-water StreamPro ADCP for collecting hydraulic data in lowland rivers is included in Appendix A.

1.4.2 Delineating hydraulic patches

Reducing the complexity of the hydraulic environment into relatively homogeneous units is a useful way of summarising the range of hydraulic conditions present, as a basis for comparing hydraulic conditions between flows or sites and as a foundation for quantifying heterogeneity. The weaknesses of qualitative visual surveys for accurately delineating hydraulic differences have already been discussed in Section 1.3 above. Identifying a quantitative method capable of delineating hydraulic patches from depth-velocity data would provide a more robust way of classifying the hydraulic environment. Whilst it is intuitive to assume that patches of different hydraulic character (e.g. deep and slow or shallow and fast) do exist, and studies have testified to the concept of coherent flow structures (Clifford et al., 2002; Emery et al., 2003; Thoms et al., 2006), the spatial and temporal boundaries between patches are not clear-cut. Spatial variations in depth and velocity throughout the channel result in fuzzy, transitional areas between patches of different hydraulic character. Temporal changes in discharge influence the location and characteristics of hydraulic patches and their boundaries. Classification methods capable of reflecting the fuzzy patchy structure of the hydraulic continuum would be advantageous (Legleiter & Goodchild, 2005).

Statistical delineation of relatively homogeneous patches can be approached in one of two ways; either by grouping objects (which in this context are point measurements of depth and velocity) with similar attributes using cluster analysis or by detecting boundaries – defined as zones of rapid change - between homogeneous patches using edge-detection software (e.g. BoundarySeer). Emery et al.'s (2003) research on the hydraulic functioning of meso scale bedforms suggested that coherent flow patches, defined by streamwise depth-averaged velocity, can be delineated using hierarchical cluster analysis. Inoue & Nakano (1999) also successfully applied this method to delineate hydraulic units according to a combination of water depth, velocity, velocity variability, substrate coarseness and substrate heterogeneity. However, a key limitation of this clustering method is that hydraulic units are represented with crisp, linear boundaries.

A more recent study by Legleiter and Goodchild (2005) suggested that the realism of hydraulic patch representation can be improved through the use of fuzzy cluster analysis, a method that allows objects to belong to more than one group and can be used to identify areas of classification ambiguity. The degree to which an object belongs to each group is reflected by a partial membership function whose value varies between 0 and 1, where 0 indicates no membership and 1 indicates complete membership (Zadeh, 1965; Bezdek et al., 1984). Hence objects which clearly belong to a single group can be differentiated from those that belong equally to several groups. Objects whose membership is split between two or more groups may be thought of as being in a boundary/transitional zone between those groups. Fuzzy cluster analysis has proved useful for classifying continuous environmental data, such as topography and soil (Burrough, 1996; Arrell et al., 2007), but as yet remains relatively unexplored for classifying hydraulic data, Legleiter & Goodchild's (2005) study being a notable exception. The application of fuzzy cluster analysis for delineating hydraulic patches and the transition zones between them is explored in Chapter 3.

1.5 The relevance of in-stream heterogeneity to freshwater organisms

Physical heterogeneity has long been identified a precursor of biodiversity (Figure 1.6) (e.g. Ricklef & Schluter, 1993; Poff et al., 1997; Ward, 1998; Tews et al., 2004). At a basic level, species diversity is positively correlated with habitat (niche) diversity because biota adapt morphologically and behaviourally to take advantage of the full range of resources and minimise competition (e.g. Gorman & Karr, 1978; Beisel et al., 1998; Minshall & Robinson, 1998; Brown, 2003). Secondly, the availability of different habitat conditions is necessary for many freshwater species to successfully complete each life-cycle stage (e.g. Armstrong et al., 2003) which is reflected in the habitat use of different age classes (Figure 1.7) (Bisson et al., 1982). As such, streams with diverse habitats are more likely to support a more diverse population structure. A variety of habitat conditions is also needed for different functional activities. For example, flow refugia (be they shallow, slow-flowing zones near channel margins or slackwater zones behind boulders during spates) provide suitable habitat for the reproduction, dispersal or persistence of macroinvertebrates (e.g. Lancaster & Hildrew, 1993; Rempel et al., 1999; Ning et al., 2009) whereas fast-flowing, well-oxygenated zones provide suitable feeding habitat for filter-feeders and slow-flowing depositional zones provide feeding habitat for shredders (Hynes, 1970; Cummins et al., 1989). Furthermore, stream reaches with complex hydraulics (i.e. hydraulic habitat diversity) can also increase the resilience of the community structure to disturbances (e.g. Pearsons et al., 1992).

The spatial structure of habitats has equal importance. For example, Freeman & Grossman (1993) showed that the availability of suitable habitat was not sufficient to explain cyprinid fish distribution; the spatial distribution of complementary habitats was also an important factor. Freshwater biota depend on the connectivity between suitable patches of habitat to disperse and migrate as well as the proximity between different habitats to successfully progress from one life-cycle stage to another (Robinson et al., 2002; Harby et al., 2007). For example, the spatial arrangement (proximity and connectivity) of juvenile, adult and spawning Atlantic salmon (*Salmo* *salar*) habitats has been shown to directly influence salmon production and is more effective to predict the size of the salmon run than simple estimates of total habitat area (Kim & Lapointe, 2011). Isaak et al. (2007) found that connectivity between suitable spawning patches was of greater importance to the occurrence of Chinook salmon (*Oncorhynchus tschawytscha*) nests than habitat size or quality. Martelo et al. (2014) demonstrated that chub *(Squalius torgalensis*) responded to the patchy dsitribution of physical resources including velocity, substrate and aquatic vegetation.

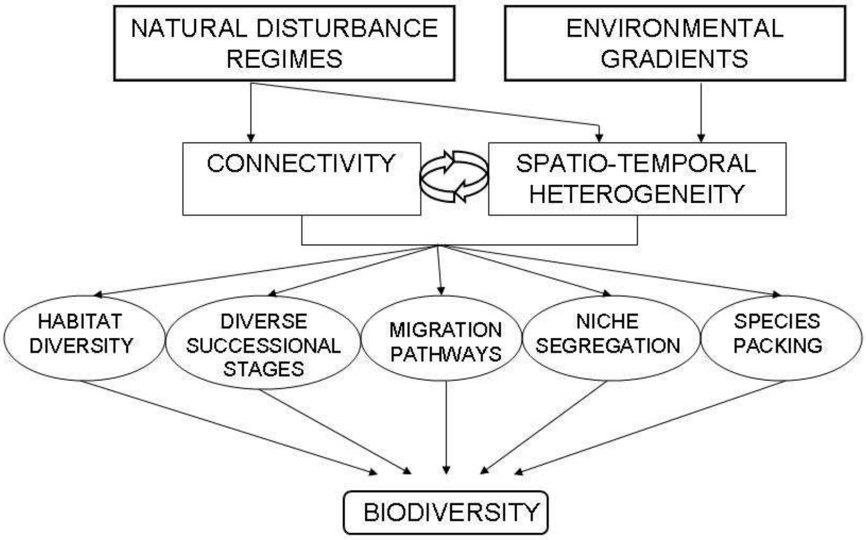


Figure 1.6. Role of spatio-temporal heterogeneity in maintaining biodiversity (Ward, 1998).

The theory of bioenergetics states that particular spatial combinations of habitat types that enable biota to maximise energy input and minimise energy expenditure are preferentially selected to maximise growth potential (e.g. Hayes et al., 2000; Guensch et al., 2001). For example, invertebrates dwell in flow refugia to minimise energy expenditure but preferentially select refugia in close proximity to resource-rich areas suitable for feeding (Vogel, 1994; Hildrew & Giller, 1994; Dolédec & Gayraud, 2004). Similar behaviour has been observed in fish: spawning redhorse suckers rest in pools adjacent to gravel riffles suitable for egg-laying (Kwak & Skelly, 1992); smallmouth bass feed in runs next to pools with structural cover (Rabeni & Jacobson, 1993); young-of-year fish occupy shallow, sheltered marginal areas adjacent to high velocity areas suitable for foraging (Bovee et al., 1994), and the distribution of dace has been linked to the availability of eddies adjacent to rapid currents (Freeman &

Grossman, 1993). Kocik & Ferrari (1998) showed that highly interspersed and juxtaposed spawning and rearing habitat increased the dispersal range of Atlantic

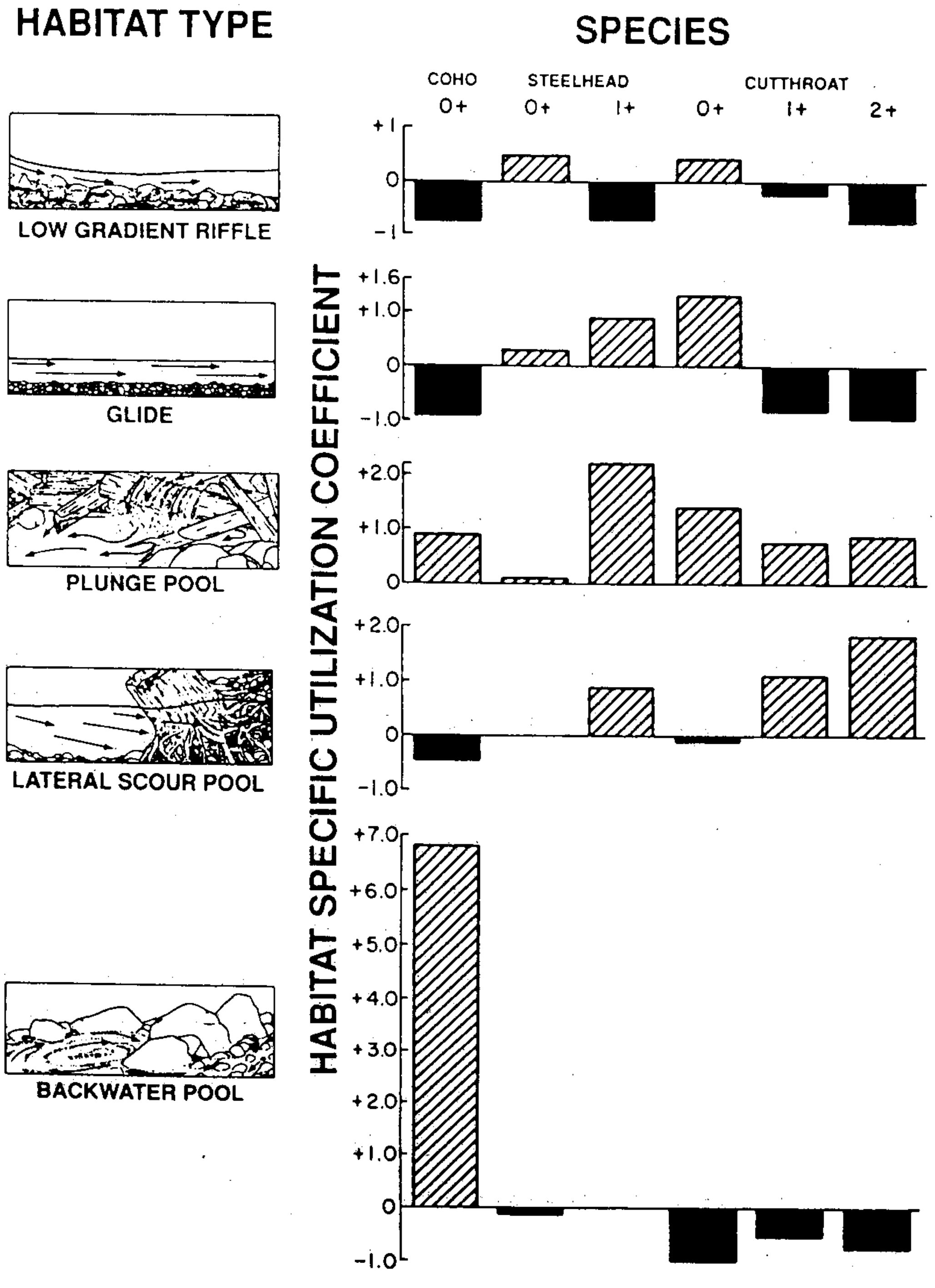


Figure 1.7. Habitat use by different age classes of salmonid species in headwaters streams, western Washington. (Source: Schlosser, 1992, adapted from Bisson et al., 1982).

salmon parr. Artificial barriers, such as dams and weirs, which fragment hydraulic habitat and limit movement up- and downstream have been shown to alter fish distributions patterns (Hoagstrom et al., 2008; Kemp & Williams, 2008; Lucas et al., 2009). For these reasons, quantifying the spatial structure of hydraulic patches and/or species-specific habitat can improve understanding of population dynamics and community structure (Dunning et al., 1992; Kocik & Ferreri, 1998; Cadenasso et al., 2006; Clark et al., 2008).

Temporal disturbances to the availability and arrangement of habitats/hydraulic patches resulting from variations in discharge also have an important ecological function. Pickett & Thompson (1978) first introduced the concept of patch dynamics to describe the change in the spatial pattern of terrestrial habitat patches over time as a result of disturbances. This concept has been successfully used to explain how the dynamic in-stream environment influences aquatic community ecology (Pringle et al., 1988; Townsend, 1989). For example, Lancaster (2000) showed that changes in patch quality resulting from variations in discharge redistributed caddis fly larvae into low-flow refugia. The so called ‘shifting mosaic’ (*sensu* Stanford et al., 2005) of aquatic habitats also plays a vital role in maintaining biodiversity (Ward et al., 2002b). Temporal disturbances to the patch mosaic affect the growth, reproduction and dispersal of individuals and the diversity, dominance and structure of the community (Pickett & White, 1985). For example, the continual movement, destruction and creation of hydraulic patches caused by discharge fluctuations limit competitive exclusion and help to maintain a diverse community structure (Pringle et al., 1988; Poff & Allan, 1995; Robinson et al., 2002). In addition, many biota adapt to take advantage of seasonal habitat/hydraulic patch turnover and in some cases their life-cycle stages are triggered by temporal changes to the habitat mosaic (Robinson et al., 2002).

Large, infrequent disturbances (i.e. floods) also play an important part in maintaining biodiversity and ecosystem integrity (Resh et al., 1988), as described by the Flood Pulse Concept (Junk et al., 1989). Flood events ‘refresh’ the ecosystem in three ways: by reconnecting the main channel with the floodplain and slackwater areas which increases nutrient and organism exchange (Amoros & Roux, 1988; Junk et al., 1989; Ward et al., 1999; Pringle, 2003); by displacing biota, thus allowing new species to colonise the area (Gibbins et al., 2004; Hein et al., 2005) and by reshaping the bed, banks and planform of the channel which establishes a new spatial template for hydraulic heterogeneity (Costa & O'Connor, 1995). Resh et al. (1988) concluded that disturbance is "the dominant organising factor in stream ecology" (p. 450). On this basis it is important that methods to assess and define the spatial and temporal dynamics of the hydraulic environment are devised and applied.

Poff et al. (2006) also put forward a conceptual model specific to river ecosystems that illustrates how the interaction between hydrology and geomorphology determines the spatial structure and temporal dynamics of physical habitat which in turn influences ecological patterns and processes (Figure 1.8). Quantifying and defining patterns of hydraulic heterogeneity is fundamental to understanding how in-stream environments function and how they influence biotic community structures and distributions. In addition, it can provide information on the implications of modifying hydromorphology on the hydraulic environment (physical habitat).

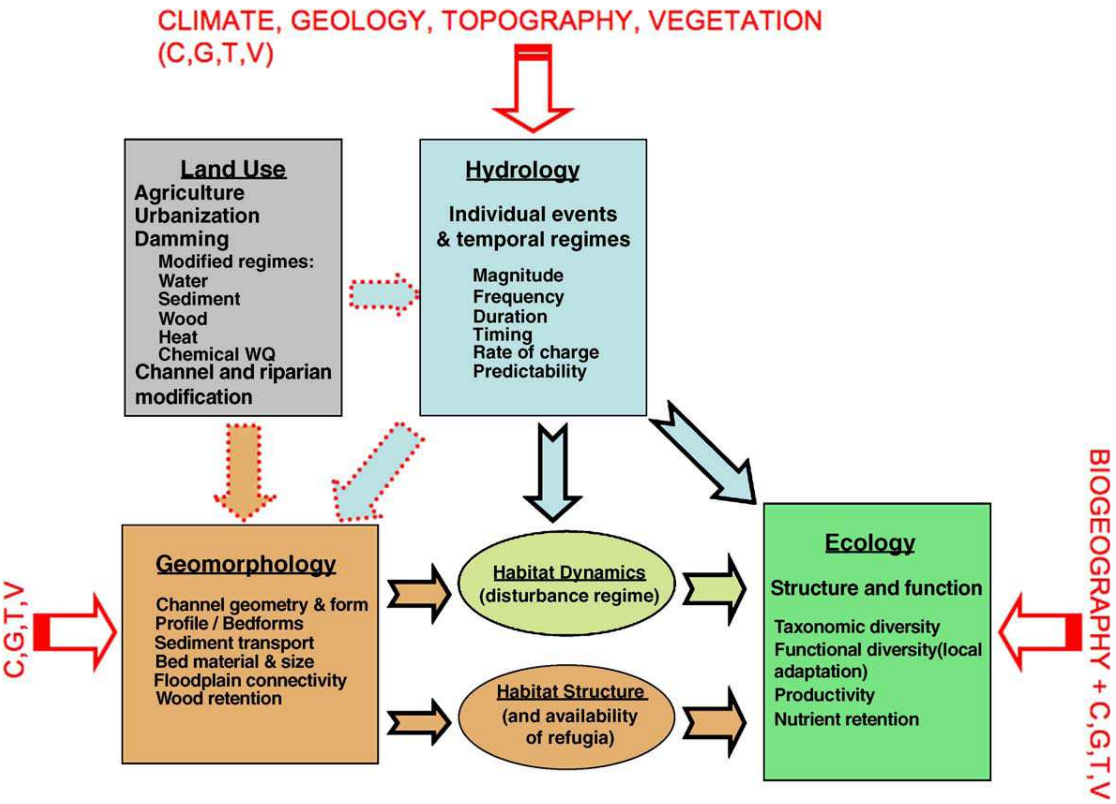


Figure 1.8. Conceptual model illustrating the ecological relevance of the spatial structure and temporal dynamics of hydraulic habitat (Source: Poff et al., 2006, p.266).

1.6 Quantifying hydraulic heterogeneity

Cadenasso et al. (2006) defined heterogeneity as one of the three dimensions of biocomplexity (Figure 1.9). They described five increasingly complex levels of heterogeneity which can be quantified to provide a complete account of spatio-temporal heterogeneity; patch richness, patch frequency (together comprising composition), patch configuration (spatial arrangement), patch change (turnover) and the shifting mosaic (the sum of temporal changes to the composition and configuration) (Figure 1.9). In landscape ecology, the basic unit for analysing spatial heterogeneity is the '**patch'** – a relatively homogeneous area (defined in terms of one or more attributes) that differs from its surroundings (Forman, 1995). All patches of the same type (i.e. with the same characteristics) are collectively referred to as a **class** (McGarigal & Marks, 1995). All patches of every class make up the landscape or, in the case of rivers, the '**riverine landscape**' (Wiens, 2002), '**riverscape**' (Allan, 2004) or '**reachscape**' (Fausch et al, 2002).

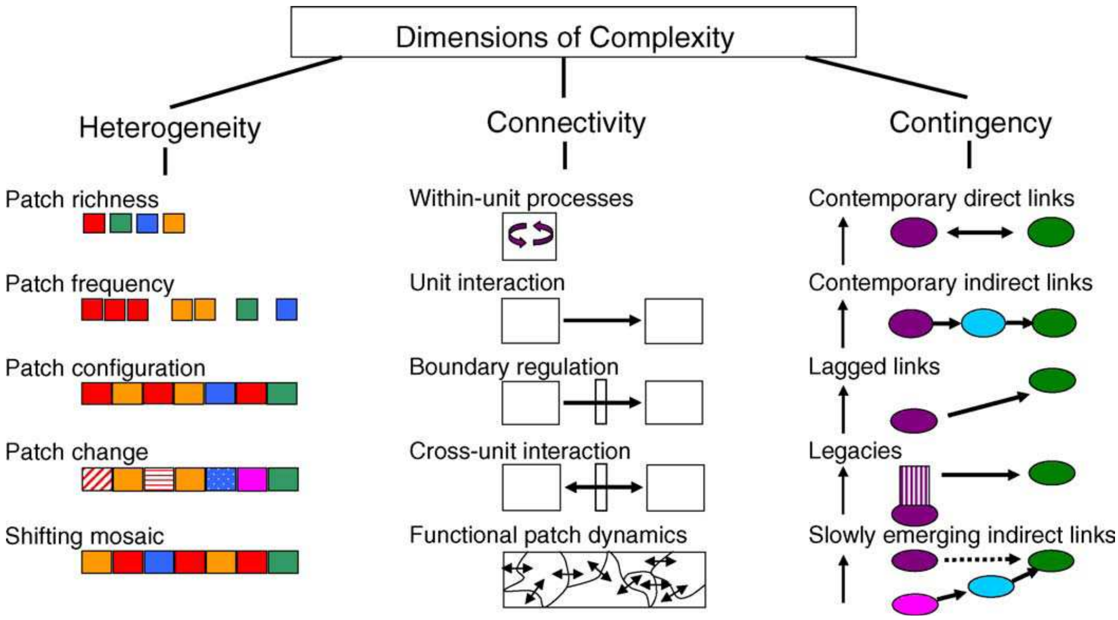


Figure 1.9. Three dimensions of biocomplexity: heterogeneity, connectivity and contingency (Cadenasso et al., 2006).

It is important to acknowledge that patch structure can be defined at multiple hierarchical spatial scales (Kotliar & Wiens, 1990). In landscape ecology studies this is usually determined by the perceptual range of the target species under investigation (Wiens, 1989). For freshwater invertebrates this could range from a few centimetres to a few metres (e.g. Hart & Resh, 1980) and for fish could range from 10s to 100s of metres depending on ecosystem size and body length (Haskell et al., 2002; Woolnough et al., 2008). However as Wiens (2002, p.508) pointed out, "*the logical outcome of advocating an organismal-based approach to landscapes is that the analysis of riverine landscapes and their ecological effects will inevitably degenerate into a series of idiosyncratic, situation-specific findings with little emergent generality*". A more holistic approach might be to define patch structure in terms of the physical homogeneity emerging at physical scales. For example, in Frissell et al.'s (1986) identified 'patch' structure at multiple nested scales from the microhabitat (often referred to as the patch-scale), mesohabitat (CGUs/SFTs), reach, stream segment to the catchment (Figure 1.3, p.8). Studies suggest macroinvertebrates (e.g. Heino et al., 2004) and fish (e.g. Inoue & Nunokawa, 2002) respond most strongly to patch scale hydraulic variability. As such defining hydraulic heterogeneity at the patch scale is likely to have the greatest ecological relevance.

During the last decade, several commentators have advocated applying a landscape ecology framework to investigate heterogeneity and dynamics in fluvial ecosystems (Figure 1.10) (Fausch et al., 2002; Poole, 2002; Tockner et al., 2002; Ward et al., 2002a; Wiens, 2002; Lianyong & Eagles, 2009). Wiens (2002) defined the key themes of landscape ecology as they apply to riverine landscapes as; the difference in patch quality over space and time; the effect of patch boundaries on the flow of materials, energy and organisms; and the significance of the spatial context and connectivity between patches. Many of these themes have already, individually, been studied in the riverine environment. For example, the theory of patch dynamics, which states that landscapes are made up of a shifting mosaic structure of habitats/patches that helps maintain biodiversity (Townsend, 1989; Pringle et al., 1988; Stanford et al., 2005), has been shown to influence the distribution of biota and the transport of organic matter (Southwood, 1977; Bormann & Likens, 1979; Kotliar & Weins, 1990; Poff & Ward, 1990; Forman, 1995) and the age structure of fish communities due to the availability and connectivity of patches required at different life stages (Schlosser, 1991).

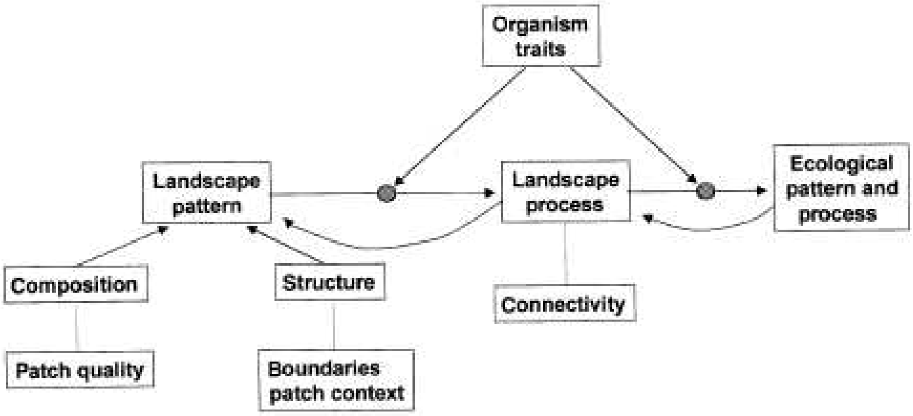


Figure 1.10. Wiens' (2002b) framework for integrating the key themes of landscape ecology.

Quantitative tools, in the form of spatial metrics have been developed to quantify the shape and spatial relationships between patches in landscapes (Li & Reynolds, 1995; Farina, 2006; Liding et al., 2008; Uuemaa et al., 2009). Advances in computing power and geographic information systems combined with the availability of open-source pattern analysis software, such as FRAGSTATS (McGarigal & Marks, 1995), have made it possible to calculate of a large number of spatial metrics describing the structure of categorical maps relatively quickly and easily. However many metrics are correlated and hence redundant whereas others may have little explanatory power for the landscape under investigation or metrics may behave erratically in response to changes in scale (Riitters et al., 1995; Li & Reynolds, 1995; Gustafson, 1998; Hargis et al., 1998; Wu et al., 2002; Lausch & Herzog, 2002; Ricotta et al., 2003; Arnot et al., 2004; Li & Wu, 2004; Kearns et al., 2005; Turner, 2005; Cushman et al., 2008). It is important to note the distinction between inherently redundant metrics – those that describe the same aspect of pattern in alternative ways so are highly correlated – and empirically redundant metrics – those that measure different aspects of spatial pattern but are correlated for the particular landscape(s) under investigation because only interpretation of the latter provides useful information (Turner, 2001). In view of these considerations metrics must be selected and applied judiciously (Li & Wu, 2004).

Until relatively recently, quantification of spatial heterogeneity has been overlooked in standard river habitat assessments (Newson & Newson, 2000). The application of landscape ecology to riverine environments has been followed up in a small number of studies to quantify the heterogeneity of floodplains, estuaries, river corridors, and catchment scale hydrological connectivity (Arscott et al., 2002; Ward et al., 2002b; Yang & Liu, 2005, van Nieuwenhuyse et al., 2011). Three recent studies have applied spatial metrics to describe the spatial distribution of biologically-defined habitat patches (e.g. Clark et al., 2008; le Pichon et al., 2009; Kim & Lapointe, 2011), although only one addressed how the spatial distribution changed over time. Thoms et al. (2006) set a precedent for using spatial metrics to quantify the configuration of velocity patches in three reaches of the Murray River, Australia. These studies are examined in more detail in Chapters 4 and 5 where a similar approach is applied. Suffice to say here that despite its relevance this type of approach is relatively rare and merits wider application (Newson & Newson, 2000; Wiens, 2002).

Summary

Understanding the relationship between a river's physical and ecological status is a primary research objective in river science and management. An integral part of this process is the characterisation of physical habitat and the assessment of its spatial and temporal variability. However, as described above, existing classifications and assessment methods have been variously criticised for lacking hydraulic distinctiveness, stage-dependency, operator variability and failing to capture spatial structure or temporal variability adequately. The potential advantages of using numerical classification methods (cluster analysis) and adopting a landscape ecology framework to characterise physical heterogeneity have been outlined and in a few cases tested, however these approaches merit wider application and further exploration, particularly in respect of quantifying hydraulic heterogeneity.

1.7 Aims, objectives and structure of the thesis

The overall aim of this research project was **to evaluate the spatial configuration and temporal dynamics of hydraulic patches in UK lowland rivers** and in so doing;

- develop a conceptual model of hydraulic patch dynamics in lowland rivers;
- recommend a framework for quantifying spatio-temporal heterogeneity, and;
- improve understanding of the effects of discharge on the hydraulic environment.

Two key **objectives** were identified;

Obj. 1 Develop a quantitative classification of the hydraulic environment, using fuzzy cluster analysis to delineate hydraulic patches and the transitional zones between them, and evaluate the relative merits of this approach;

Obj. 2 Quantify the heterogeneity (*sensu* Cadenasso et al., 2006) of the hydraulic environment using spatial metrics and GIS analysis and evaluate the effect of discharge variations on the composition, spatial configuration and location of hydraulic patches.

The thesis was structured into six chapters as shown in Figure 1.11. **Chapter 1** sets the context for the research project, justifies its relevance to the current river science research agenda and introduces its themes. **Chapter 2** provides a detailed account of the selection and hydromorphological characteristics of the three sites used in the project, as a basis for links between channel morphology and hydraulics discussed in later chapters. Chapter 3 addresses Objective 1, which is broken down into three sub-objectives outlined below, along with two hypotheses. Chapters 4 and 5 address Objective 2 and again sub-objectives and hypotheses for each are outlined below. Chapter 6 presents a discussion of the results and proposes a conceptual model of hydraulic patch dynamics to conclude the project.

Chapter 3:

Objective 1:

(a) Evaluate the performance of three fuzzy clustering algorithms

(b) Apply the optimum algorithm to generate a classification of hydraulic patches and delineate the transitional zones between them at the three study reaches

(c) Make recommendations for the application of fuzzy cluster analysis

Hypotheses:

(1) Coherent hydraulic patches, defined by the joint distribution of depth and velocity, exist and can be indexed by intra-patch homogeneity and inter-patch heterogeneity.

(2) Fuzzy cluster analysis can be used to delineate spatially coherent hydraulic patches and the transitional zones between them in a quantitative and objective manner.

Chapter 4:

Objective 2:

(a) Quantify patch richness, frequency and diversity at each site-flow combination to evaluate how reachscape composition changes in response to (seasonal) variations in discharge

Hypotheses:

(1) Shallow or slow-flowing patches dominate the reachscape at low flow and are replaced by deeper, faster-flowing patches as discharge increases, resulting in a significant difference in hydraulic patch composition at low and high flows;

(2) Maximum hydraulic patch diversity occurs at intermediate flows.

Chapter 5:

Objective 2:

(b) Quantify the configuration and geometry of same-type hydraulic patches (class level) using spatial metrics at each site-flow combination;

(c) Describe, illustrate and quantify the spatial dynamics of hydraulic patches (at the class level) over the range of discharges sampled;

(d) Quantify the reachscape configuration and geometry of all hydraulic patches (reachscape level) using spatial metrics, and;

(e) Evaluate the effects of discharge on each of the above.

Hypotheses:

(1) Patch shape is most complex at low flows and becomes more regular and linear at high flows

(2) Reachscape configuration is characterised by interspersed patch types at low flows but becomes more aggregated and connected at high flows

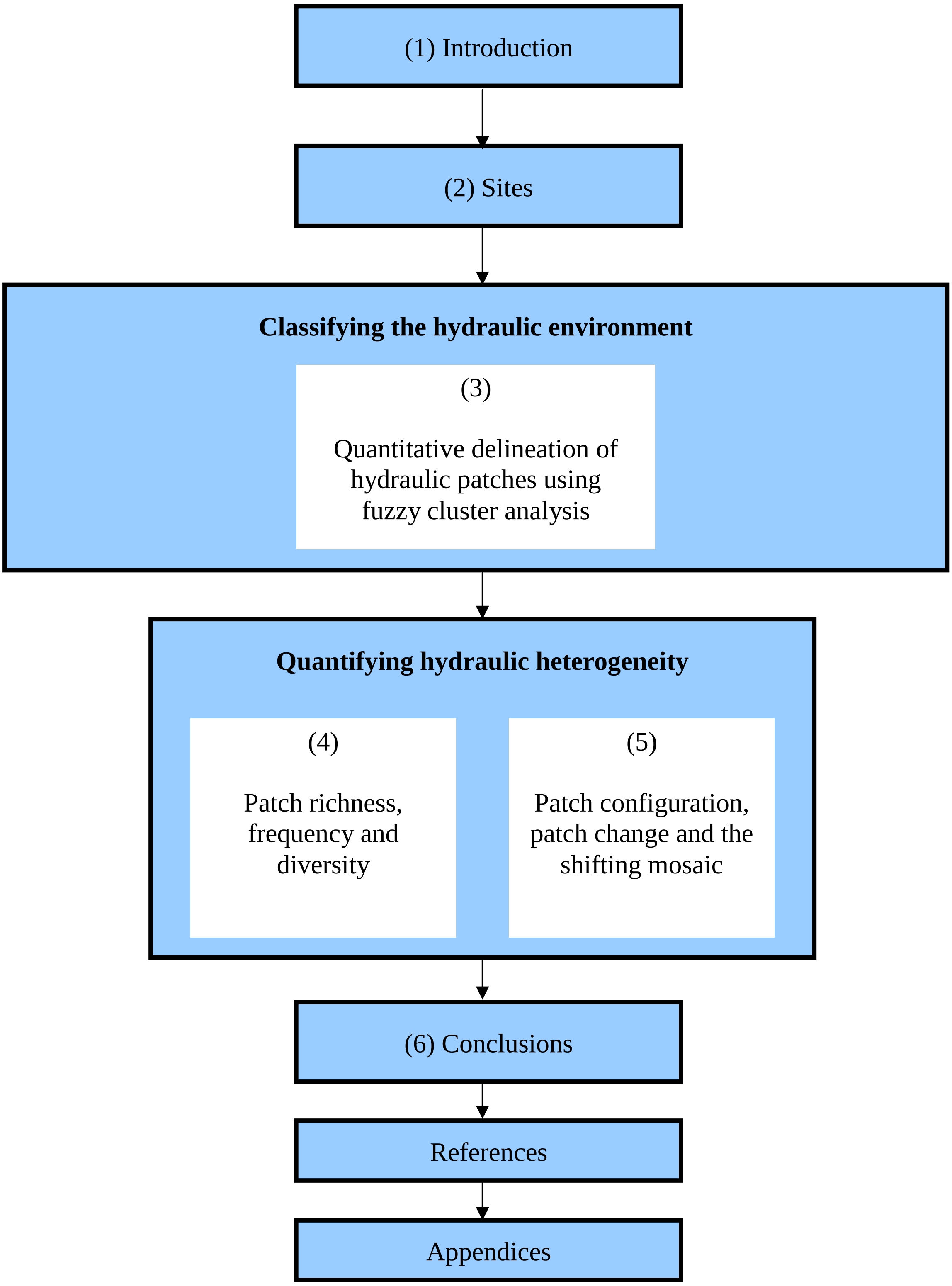


Figure 1.11. Overview of the thesis structure; chapters numbered (1) – (6).

"The biology lives in the hydrology,
and the hydrology flows over the geology."

Mattole River Restoration Council, 1995

2

Study Sites

Chapter overview

This chapter introduces the three study sites and their catchments used in this research project. The first three sections (2.1-2.3) describe the selection and location of the three rivers and reaches used, the characteristics of the rivers and the reaches. The next section (2.4) outlines the topographic survey methods and visual surveys of substrate class and key geomorphic features at each study reach. This is followed by a detailed description of study reach characteristics (2.5). The final section (2.6) of the chapter outlines the importance of geomorphic diversity and presents a quantitative summary of geomorphic diversity in each reach as a basis for identifying relationships between hydraulic heterogeneity and hydromorphological condition in Chapters 4 and 5.

2.1 River and reach selection and location

River and reach locations were selected based on their degree of modification, the level of heterogeneity present and practical sampling considerations. To evaluate hydraulic patch dynamics it was necessary to survey each site at five different discharges which, given the scope of the study, was possible at a maximum of three sites (total of 15 hydraulic surveys). Sites had to be within 1 hour's travel distance so hydraulics could be sampled at short notice when target discharges occurred. This limited site selection to lowland rivers with pool-riffle morphology in Worcestershire, Herefordshire borders or Warwickshire. Given this constraint on river type it was decided to select rivers with varying degrees of modification and physical heterogeneity to assess a range of hydraulic patch dynamics patterns in lowland rivers. Large, very heavily modified rivers such as the River Severn were excluded due to their size.

The River Arrow provided an example of a semi-natural, actively meandering, single thread lowland river with well defined pool-riffle bed morphology and gravel dominated substrate. Physical heterogeneity is largely provided by pronounced bedform topography and the river is characterised by riffle/run-pool-glide CGU sequences. Removal of much of the natural riparian vegetation to increase the area of grazed grassland has contributed to channel modification. In its middle and lower reaches the channel is incised and there is evidence of bank erosion (Lawler, 1992; Lawler, 1994). By contrast the Leigh Brook was selected as an example of an unmodified river characterised by high habitat quality, geomorphic diversity and relatively coarse substrate. The supply of coarse sediment from small tributaries, the presence of established mature trees on mid-channel bars and banks and the ready supply of large woody debris create small scale geomorphic features and hydraulic heterogeneity in addition to mesoscale pool-riffle bedforms. Lastly, the River Salwarpe was selected as an example of a more heavily modified river with a history of channel realignment. Channel modification is most prevalent in the urban areas of the catchment where channelisation and bank protection occur. In its lower reaches the river is dominated by low gradient run-glide sequences. The hydromorphological characteristics of each catchment are provided in Section 2.2.

A walkover survey of a 1-2km publicly accessible section of each river was carried out to identify potential reaches. Reaches exhibiting the types of physical heterogeneity and channel modification described above were sought. Targeting reaches with different levels of heterogeneity provided an opportunity to test of the nuances and capabilities of fuzzy cluster analysis for classifying hydraulic data, which was a primary focus of this study. At the River Arrow a 56m reach with pronounced bedforms, minimal riparian vegetation and evidence of failed bank protection was selected. At the Leigh Brook a 26m reach with no bed or bank modification, mature riparian vegetation and small-scale erosional and depositional features was selected. At the River Salwarpe a 45m laterally constrained reach in which concrete bank protection contained the river's path under a road bridge and bank reinforcement prevented the river immediately upstream of the bridge from meandering was selected. Further details are provided in Section 2.3.

Reach lengths and length:width ratios varied by site. Typically a standard reach length of 5-7 times the bankfull width is used to eliminate reach length and differences in CGU sequence as explanatory variables in comparison studies. Based on the bankfull widths of each reach in this study, a reach length of between 40-56m at the River Arrow, 45-63m at the River Salwarpe and 50-70m at the Leigh Brook would have been required. Reach lengths within these ranges were achieved at the River Arrow and River Salwarpe. At the River Arrow, where the capability of fuzzy cluster analysis to delineate hydraulic patches was first tested, a reach including several metres upstream and downstream of the main ~40m run-pool-glide CGU sequence was surveyed to avoid biasing the location of hydraulic patch boundaries by CGU boundaries. At the River Salwarpe the minimum reach length (5 x bankfull width) was used for two reasons; firstly to isolate the length of reach affected by modification and secondly as this length was more representative of the channel river upstream and downstream where the channel narrowed by approximately 3m. A 50m reach was trialled at the Leigh Brook using the 0.5m x 1m hydraulic sampling resolution used at the other two sites, however on inspection of the data it was clear that this sampling resolution was not correctly scaled to the small scale geomorphic features in the reach and the hydraulic heterogeneity was not being captured. As it was not feasible to increase the longitudinal sampling resolution without shortening the reach length, the reach length was almost halved to 26m so the longitudinal sampling resolution could be doubled to capture the small scale hydraulic heterogeneity. This shorter reach, although dominated by a glide, also contained a scour pool, areas of exposed sediment, a woody debris hydraulic control, a run and a small riffle.

It is acknowledged that using different length reaches with different CGU sequences limited the subsequent analysis in a number of ways. Firstly, it was not possible to establish a general pattern of hydraulic patch dynamics in pool-riffle reaches because the reaches were not comparable like-for-like and could not be treated as replicates in multivariate statistical tests. To facilitate a like-for-like comparison either a modelling approach or a much lower sampling resolution would have been required to evaluate longer reaches containing the same sequence of CGUs at each site. There were insufficient resources in this study to develop and calibrate a hydrodynamic model at each site. Using a lower sampling resolution would have underrepresented lateral hydraulic heterogeneity in the relatively narrow channel at the River Arrow and both lateral and longitudinal hydraulic heterogeneity associated with small scale geomorphic features at the Leigh Brook. Alternatively a variable sampling resolution could have been used to overcome this conflict; by sampling in greater density in areas with more hydraulic heterogeneity and lower density in relatively homogeneous areas of the reach the length of reach surveyed within the time and resources available for each survey could have been increased. However a detailed geostatistical pilot study would have been required to assess where and by how much to increase sampling density and test various interpolation methods to produce regularly spaced data grid from irregularly spaced measurements which was outside the scope of this study but could be explored in future research. Secondly, the effects of channel modification on hydraulic patch dynamics could not be isolated because a modified and unmodified reach of the same length and from the same river were not surveyed for direct comparison. To do this it would be important to survey each reach at exactly the same flow, making a modelling a necessity, something that was not feasible in this study as previously explained. The aim of the heterogeneous reach selection was to provide a thorough test of the methods used in this study, which have rarely been used in the instream environment before, and to describe if and how the pattern of hydraulic patch dynamics patterns differed. Additional replicate sites would be required to perform rigorous statistical comparisons.

2.2 Catchment characteristics

The location of each catchment within the context of the regional river network is shown in Figure 2.1. Table 2.1 outlines the key hydromorphological characteristics of each River Arrow, River Salwarpe and Leigh Brook catchments. All rivers have similar hydromorphology typical of lowland catchments but differ in the degree of modification; the Arrow being semi-natural, the Salwarpe being impacted, and the Leigh Brook being relatively unmodified.

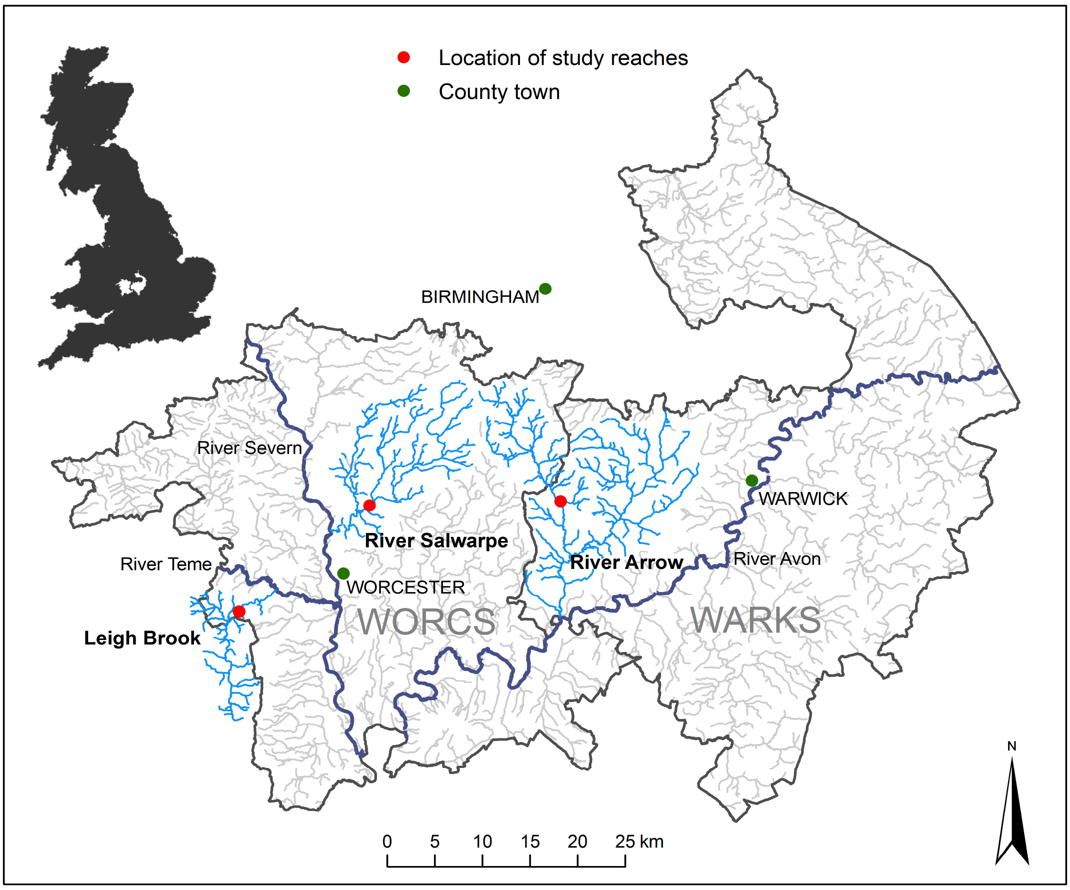


Figure 2.1. Location of study reaches and the Leigh Brook, River Salwarpe and River Arrow catchments within the Worcestershire and Warwickshire river network.

Table 2.1. Selected catchment descriptors.

Descriptor	R. Arrow	R. Salwarpe	Leigh Brook
Catchment area (km^2)§	333.5	196.8	79.7
Mean altitude (m asl) §	101	80	117
Index of Catchment steepness (m/km)§	44.7	41.9	88.1
Baseflow Index (BFI) §	0.38	0.52	0.54
Urban extent (%)§	4	5	1
RHS Habitat Modification Score (range given where available)	120-845§§	0-880§§	1*
RHS Habitat Quality Assessment score (adjusted)	33-48§§	18-44§§	45*
Q10 (m^3s^{-1}) #	1.75§§	2.50§§	1.41**
Q50 (m^3s^{-1}) #	0.55§§	0.98§§	0.55**
Q95 (m^3s^{-1}) #	0.26§§	0.47§§	0.20**
Index of Flow Variability (Q10/Q95) #	6.7	5.3	7.1

§ Flood Estimation Handbook (CEH, 2006).

§§ Data supplied by the Environment Agency; flow duration percentiles calculated from daily mean flows during 06/12/07-28/02/11, HMS and HQA(adj.) taken from RHS surveys carried out between 1994-1997.

* Maddock & Hill, 2007.

** Stage data collated from the University of Worcester monitoring station and converted to discharge.

Flow statistics are representative of conditions at or near the study reaches

Figure 2.2 shows the dimensionless flow duration curves (data period: 6th December 2007 – 28th February 2011) for each site. Although all three curves are similar, as might be expected of lowland catchments in the same region, the slightly steeper flow duration curve (FDC) for the Leigh Brook illustrates how the steeper catchment topography and smaller catchment size contributes to the more ‘flashy’ hydrological regime characterised by short duration flow events. This is confirmed by the index of flow variability. The regime is also relatively flashy at the River Arrow due to the dominance of surface run-off from the town of Redditch in the headwaters of the catchment. The River Salwarpe has the largest overall range of discharges but the least flow variability in the mid-range (Q40-Q80) of all three sites, as indicated by the slightly flatter FDC (Figure 2.2). Figure 2.3 shows the 2008-2010 discharge hydrographs for each site based on daily mean flow data from the gauging station to each study reach. Flow data from the River Salwarpe and River Arrow are gauged flows that reflect the influence of abstractions whereas gauged data from the Leigh Brook reflects the natural flow regime due to the absence of artificial influences at this site.

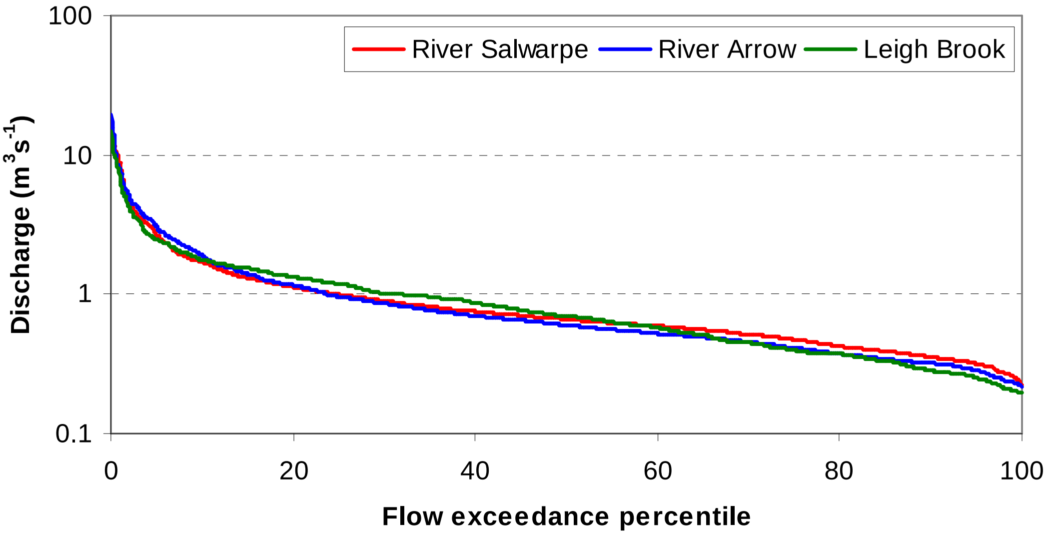
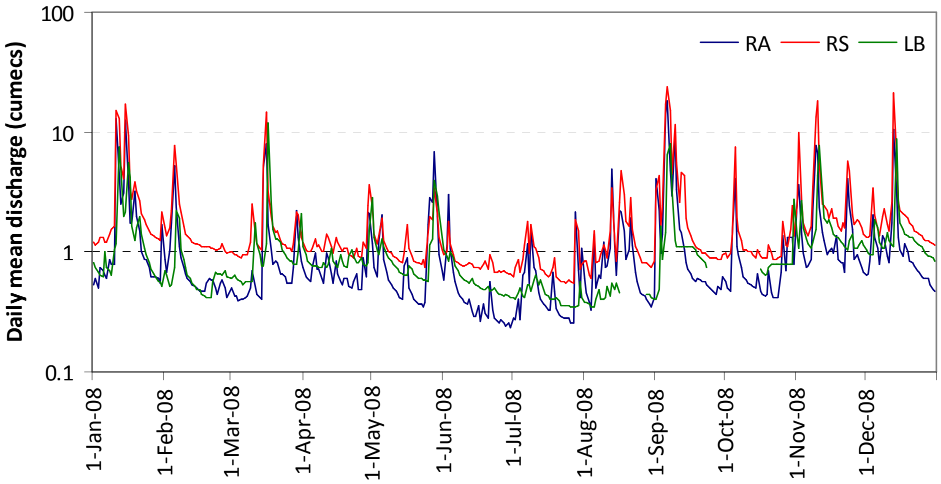
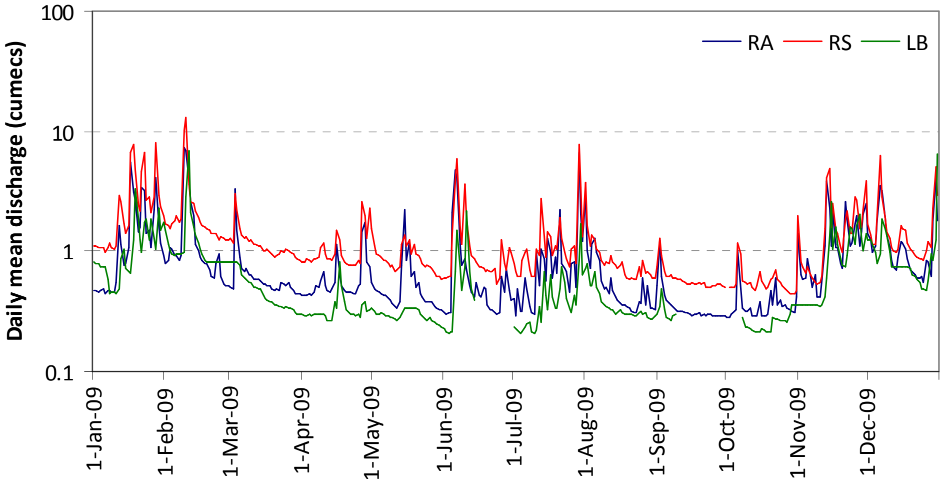


Figure 2.2. Dimensionless flow duration curves for the River Arrow, River Salwarpe and Leigh Brook study reaches (for the period 06/12/07-28/02/11)





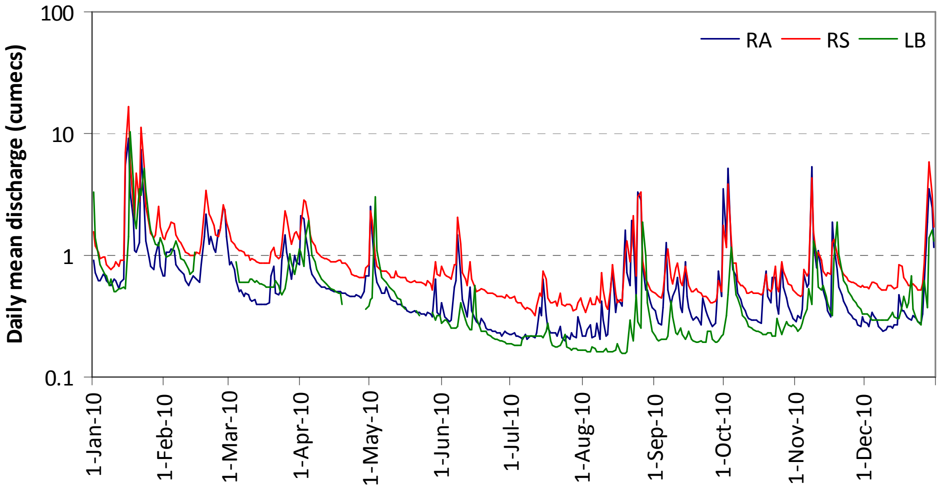


Figure 2.3. Daily mean discharge hydrograph at each study river in (a) 2008, (b) 2009 and (c) 2010.

2.2.1 River Arrow

The River Arrow is a tributary of the River Avon, rising in the Lickey Hills in north Worcestershire (SO 996 753) and flowing south east through Warwickshire. The river drains a lowland catchment underlain with Triassic sandstone and follows a meandering pattern with characteristic pool-riffle sequences. The River Alne is a major sub-catchment, joining the Arrow approximately 8km upstream of its confluence with the River Avon.

The river is predominantly fed by surface runoff (BFI = 0.38), some of which comes from urban areas around Redditch and Studley. Several abstraction licences are in place on the Arrow, however all flows <Q68 ($0.432ms^{-1}$) are subject to a "no-take" policy (Environment Agency, 2006a). The river has been subject to a moderate degree of dredging (Environment Agency, 2006a), although this is largely confined to areas of the catchment upstream of the study site. The catchment is predominantly rural and used for livestock grazing. Much of the natural riparian vegetation has been removed to increase grazing area.

2.2.2 River Salwarpe

The River Salwarpe drains the Clent and Lickey Hills, rising near Bromsgrove (SO 993694) and flowing south west to its confluence with the River Severn at Hawford. The catchment is underlain with Mercian Mudstone and Triassic saliferous beds (Woodiwiss, 1992; CEH, 2006) and current land-use is predominantly rural, with the exception of urban areas in Bromsgrove and Droitwich Spa.

The River Salwarpe has a long history of modification stretching back to the 1660s, including a failed attempt to make it navigable between Droitwich and the River Severn in 1662-75 and realignment of the channel to link the Junction and Barge canals in 1854 (Hadfield, 1985; Cumberlidge, 2009). In 2010 a section through Droitwich was canalised as part of the Droitwich canals restoration project. The River Salwarpe is classified as an over-abstracted river due to large groundwater abstractions by local water companies (Environment Agency, 2006b).

2.2.3 Leigh Brook

The Leigh Brook, a tributary of the River Teme, rises on the northern ridge of the Malvern Hills (SO 695529) and flows north east draining an 80km^2 rural catchment underlain with Triassic mudstone and Silurian limestone and sandstone. It is a high quality, 'natural' stream in terms of its biological diversity and geomorphic complexity (Environment Agency, 2005). Sections of the Leigh Brook meander through the Malvern Hills Area of Outstanding Natural Beauty and the Knapp and Papermill Nature Reserve which is designated as an SSSI. As such the stream is relatively unaffected by human modification. In 2007 it was assigned the lowest possible Habitat Modification Score and classified as having high habitat quality (Maddock & Hill, 2007).

2.3 Topographic and substrate surveys

A topographic survey of the channel bed and banks was conducted at each reach using a Nikon NPL 332 Reflectorless Total Station. Channel elevation was measured at every node on a 0.75 x 1m resolution sampling grid relative to an arbitrary datum. Additional data were collected at breaks of slope or exposed boulders not represented by a grid node. The data were interpolated to 0.05m resolution using ordinary spherical kriging then mapped in ArcGIS.

Substrate in each study reach was mapped and characterised using two methods. Firstly a visual assessment of dominant and sub-dominant substrate classes was carried out during the low flow hydraulic survey. Using the Wentworth substrate classification (Wentworth, 1922), the percentage cover of each class in a 0.25m^2 area around each hydraulic point measurement location was estimated to the nearest 10%. Secondly, the substrate particle size distribution was quantified using a Wolman pebble count procedure (Wolman, 1954). The b-axis of 200 particles, collected at random along a zig-zag path throughout the reach, was recorded to the nearest millimetre and the cumulative frequency of particle size was plotted. D_{50} particle size was used as a measure of mean particle size class and D_{10}/D_{60} was calculated as a measure of substrate size class heterogeneity.

2.4 Reach characteristics

Table 2.2 provides a comparative overview of the morphological characteristics of each study reach. All the selected reaches were low gradient 4th order streams. A detailed description of each reach including maps, photographs, field sketches and planform figures showing the topography, dominant substrate types and key geomorphic features follow in sub-sections 2.4.1-2.4.3.

Table 2.2. Characteristics of each study reach.

	River Arrow	River Salwarpe	Leigh Brook
Ordnance Survey grid reference (mid-reach)	SP 082 634	SO 881 627	SO 747 514
Strahler stream order	4th	4th	4th
Reach length (m)	56	45	26
CGUs present	Glide-run-pool-glide-run-pool	Glide-run	Run-pool-glide-riffle
Gradient (m/m)	0.016	0.002	0.011
Dom-subdom substrate class	Gravel -silt	Cobble-gravel	Cobble-gravel
Mean wetted width (at moderate flow) (m)	6.8	7.8	8.2

2.4.1 River Arrow

The 56m study reach (SP082634) (Figure 2.4) flows through open pasture, has little riparian vegetation and a low gradient of 0.016m/m. The channel is meandering with an active point bar and evidence of bank erosion. The bedform topography is pronounced with two deep pools; one in the middle of the reach and one in the downstream extent of the reach (Figure 2.5). The substrate is dominated by gravels with the exception of an area of fines/silt in a large backwater (Figure 2.5). At low flow the channel width varies between 3-10m, average depth is 0.3m and average velocity is 0.108m/s. The reach is composed of a glide-run-pool sequence of CGUs (Figure 2.6).

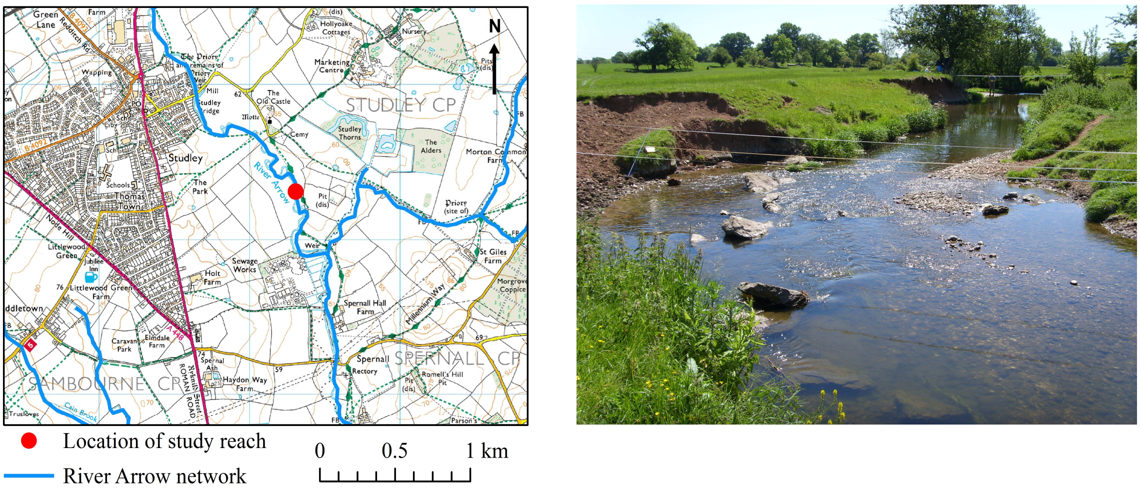


Figure 2.4. Local environs (left) and photograph (right), viewed from the upstream extent, of the River Arrow study reach. (Source: EDINA Digimap, 2010; Photo: Author's own) .

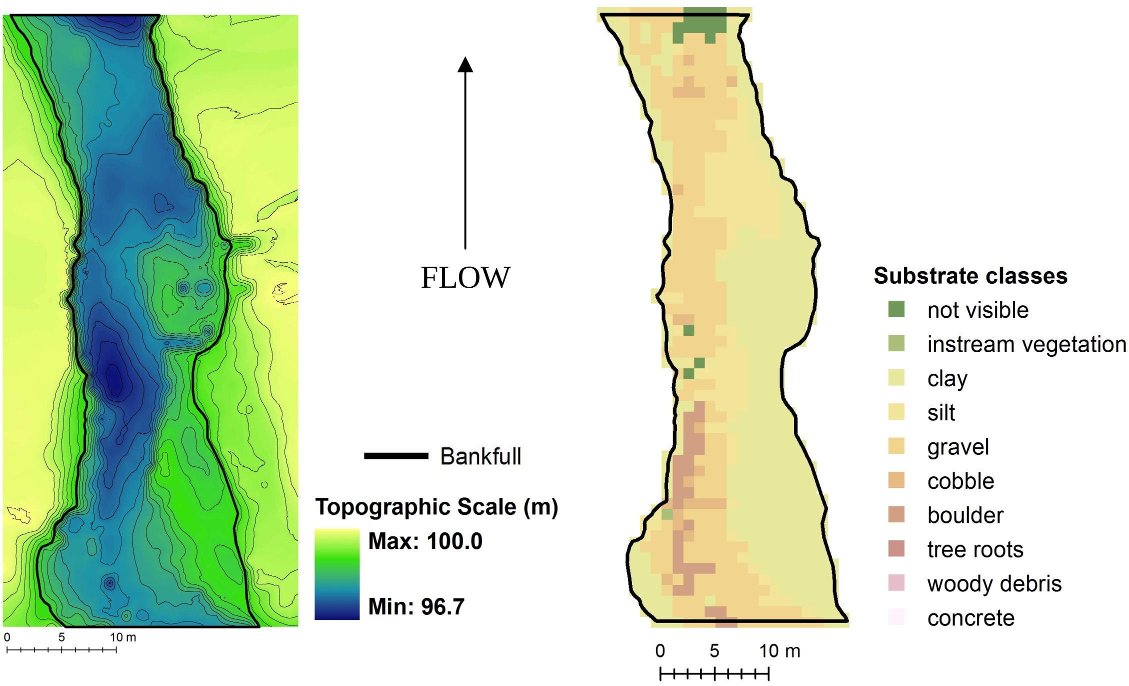


Figure 2.5. Planform diagram of the River Arrow channel and bank topography overlaid with 20cm contours (left) and dominant substrate class (right).

2.4.2 River Salwarpe

The 45m reach (SO881627) flows between public playing fields and community woodland and under a road bridge (Figure 2.7). The channel banks have been reinforced with concrete in a 12m section in the middle of the reach to support the road bridge piers. As a result of the channel width being constrained, the centre of the channel has been scoured. Immediately upstream of the bridge the river has been prevented from meandering to stop the bridge piers from being undermined. A 6m section of the left bank at the upstream extent has been reinforced with stone gabions and geotextile matting following a large flood in 2007 which eroded the river bank (Figure 2.7). On the opposite bank vegetation is encroaching in an attempt to narrow the channel. The dominant and subdominant substrate classes were cobble and gravel respectively (Figure 2.8). At low flow the channel width varies between 4.5-8m with an average velocity and average depth of 0.316m/s and 0.14m respectively. The reach has a very gentle gradient of 0.002m/m, contains a shallow glide in the first 17m of the reach (from the upstream extent) followed by a run for the remaining 29m (Figure 2.9).

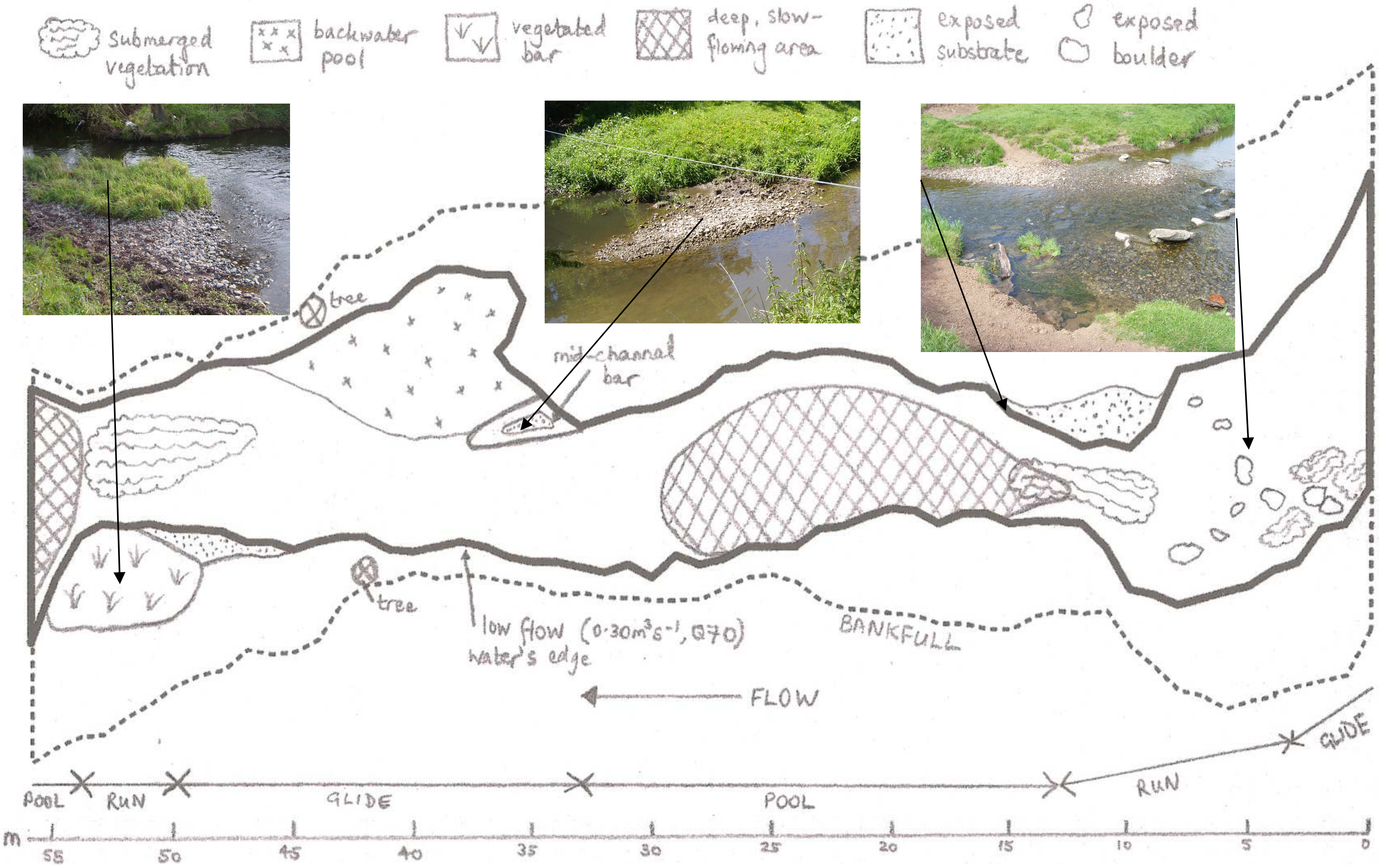


Figure 2.6. Field sketch and photographs of the key geomorphic features in the River Arrow reach.

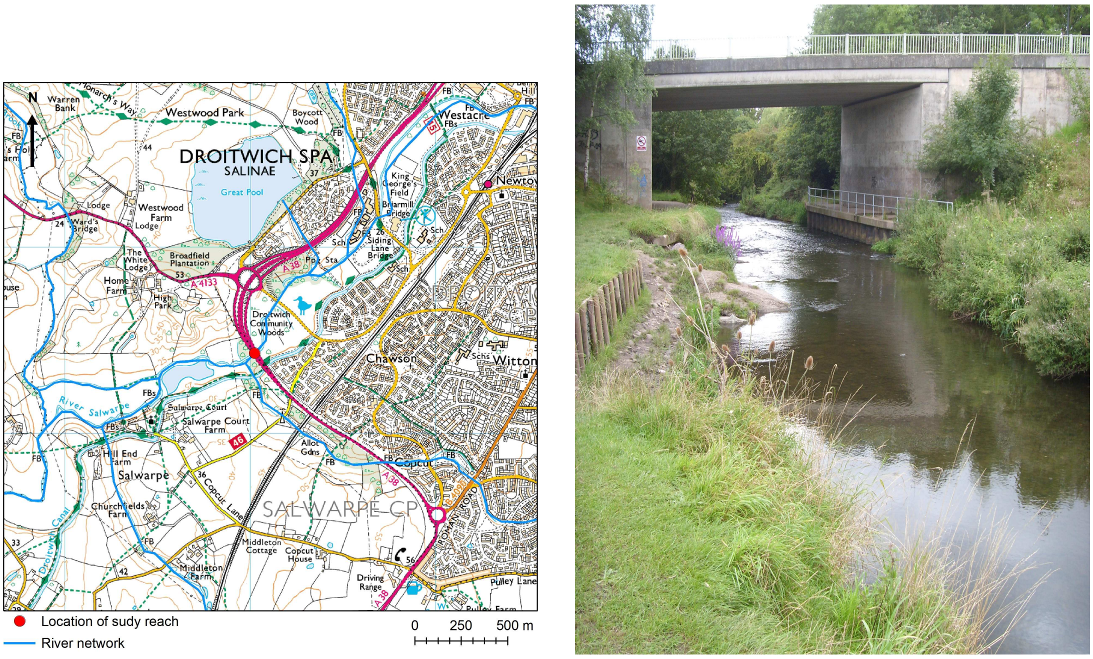


Figure 2.7. Local environs (left) and photograph, viewed from the upstream extent (right), of the River Salwarpe study reach. (Source: EDINA Digimap, 2010; Photo: Author's own).

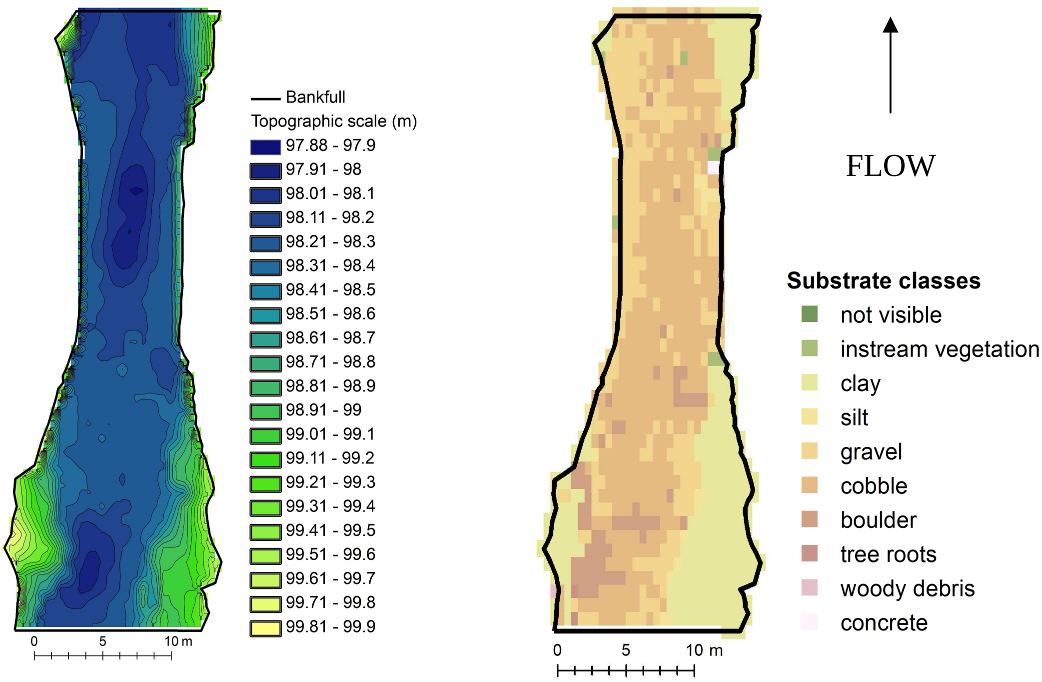


Figure 2.8. Planview of the River Salwarpe channel and river bank topography overlaid with 10cm contours (left) and dominant substrate class (right).

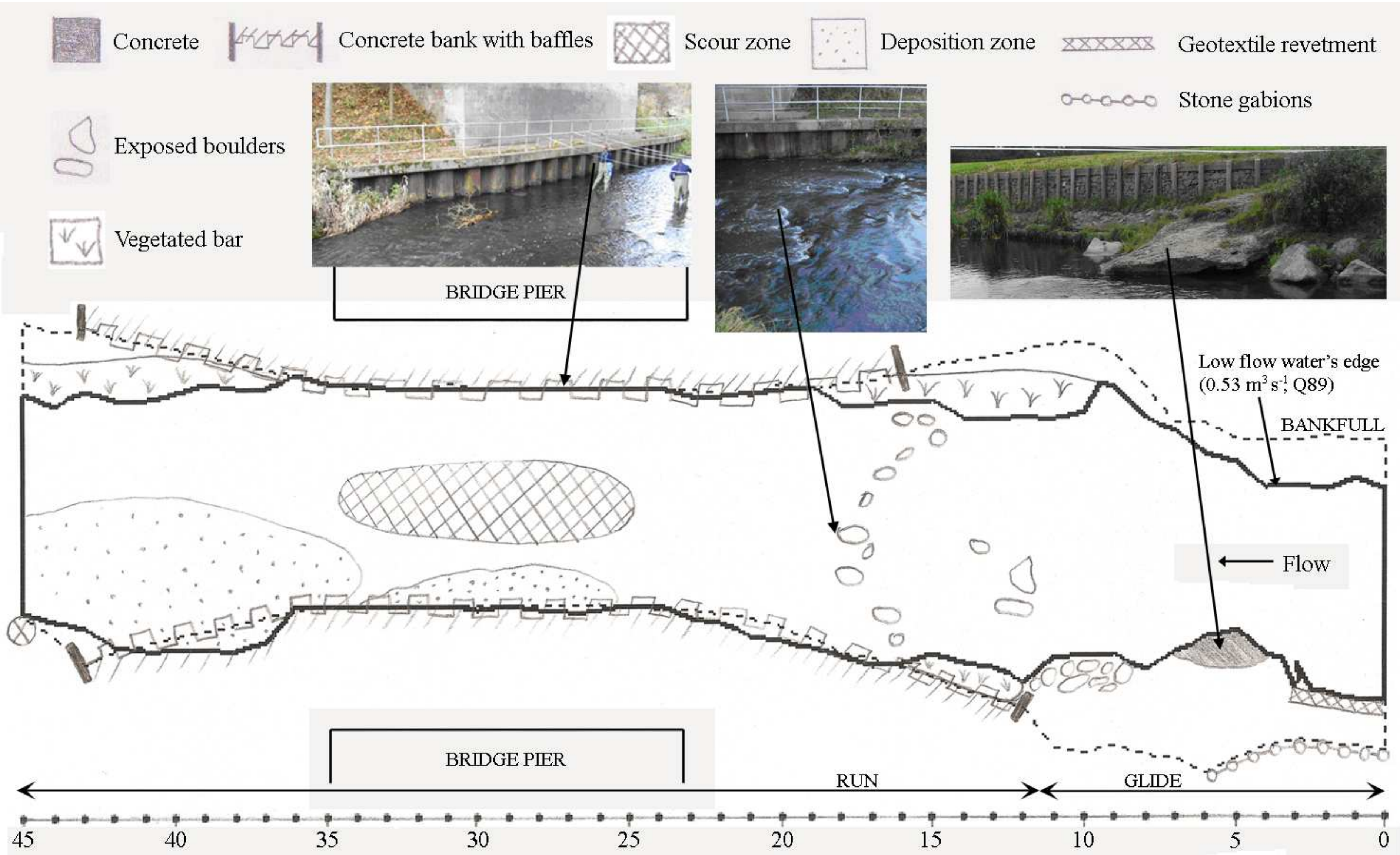


Figure 2.9. Field sketch and photographs of the key geomorphic features at the River Salwarpe.

2.5.3 Leigh Brook

The 26m study reach (SO747514) flows through a steep-sided, wooded valley and has a gradient of 0.011m/m (Figure 2.10). Although the bedform topography is relatively subdued in this reach, coarse cobble substrate, woody debris and exposed tree roots create considerable small-scale hydraulic complexity (Figure 2.11). The reach is dominated by a glide but features a short run and small scour pool in the upstream section of the reach and a riffle at the downstream extent (Figure 2.12). At low flow the channel width varies between 6-8.5m, with an average velocity of 0.154ms^{-1} and average depth of 0.11m.

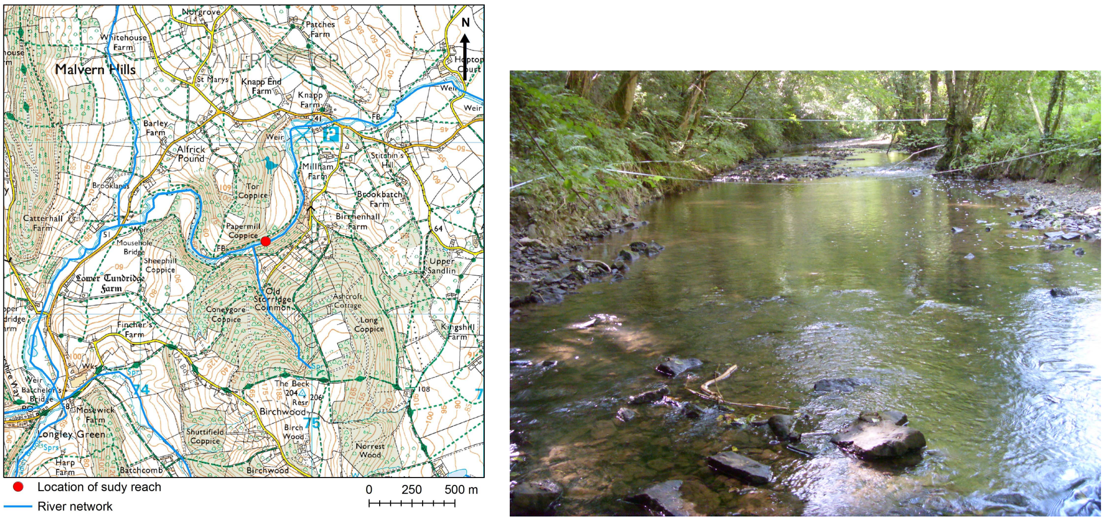


Figure 2.10. Local environs (left) and photograph (right), viewed from the upstream extent, of the Leigh Brook study reach. (Source: EDINA Digimap, 2010; Photo: Author’s own).

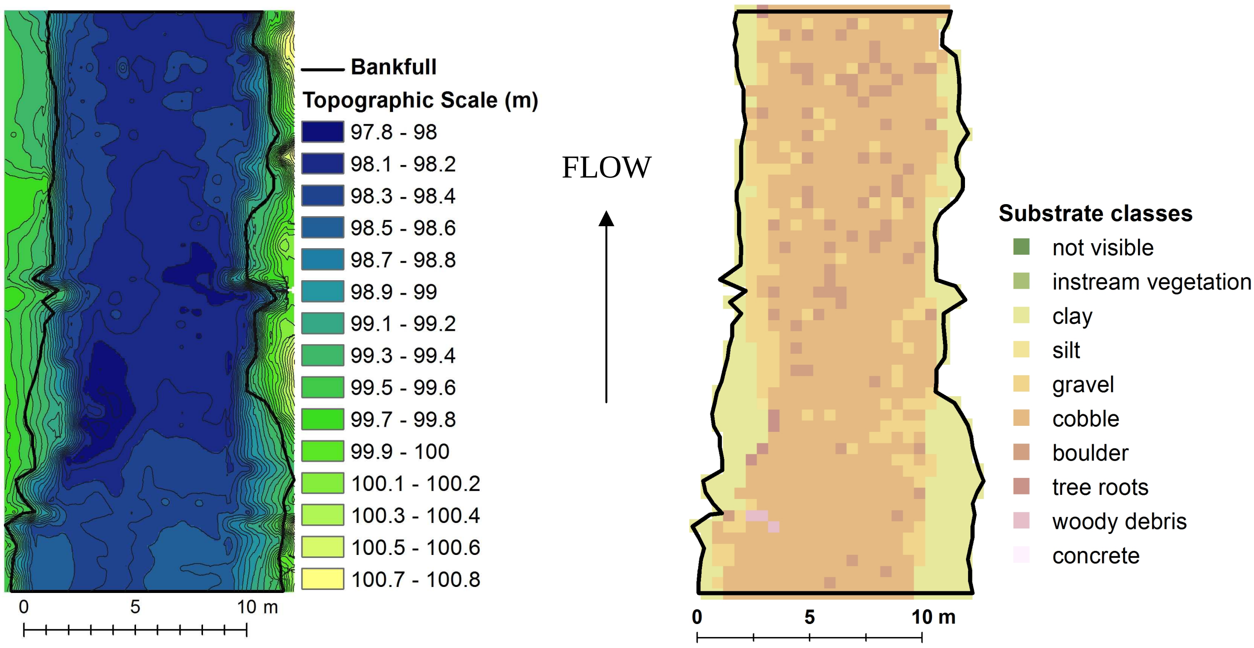


Figure 2.11. Leigh Brook site channel and river bank topography overlaid with 10cm contours (left) and dominant substrate class (right).

2.6 Geomorphic diversity

The shape of the channel bed and banks creates the spatial template for hydraulic diversity and is therefore an important determinant of the composition and configuration of hydraulic patches (e.g. Newson & Newson, 2000; Klaar et al., 2009). Variations in a channel's long-profile, cross-sectional shape and bed roughness shape the distribution of depth and velocity (Figure 2.13) (Stewardson & McMahon, 2002). Channel shape also affects the way in which changes in discharge affect the hydraulic environment. For example, in a simple, engineered channel with uniform bed elevation and vertical banks depth and velocity respond uniformly to increases in discharge and are positively correlated whereas in channels with more diverse bed and bank morphology depth and velocity respond non-uniformly to changes in discharge and do not have a simple linear relationship with each other (Figure 2.14).

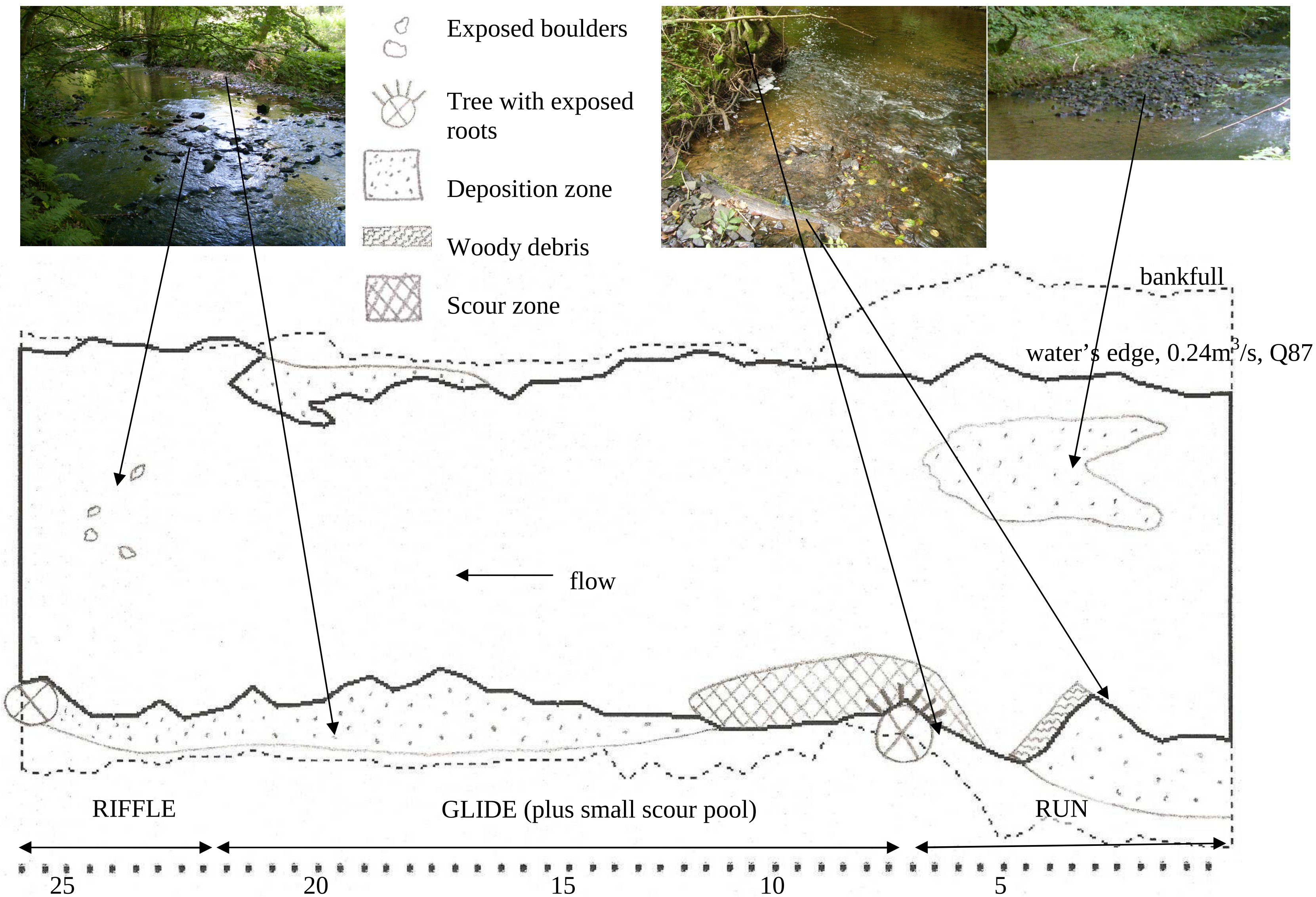


Figure 2.12. Field sketch and photographs of key geomorphic features at the Leigh Brook reach.

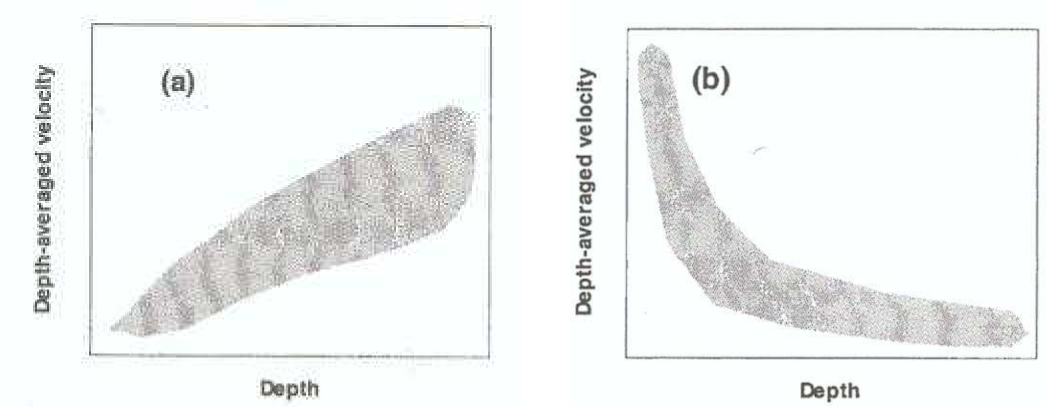


Figure 2.13. The joint distribution of depth and velocity under in an idealised (a) rectangular and (b) prismatic channel (Source: Stewardson & McMahon, 2002).

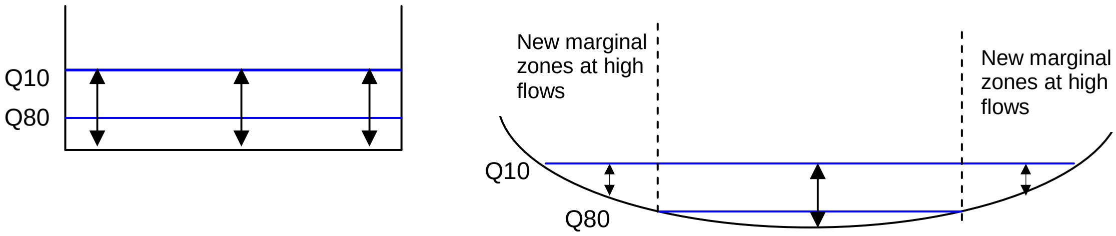


Figure 2.14. Influence of cross-section shape on depth variations under high and low flow scenarios.

Geomorphic features in each reach were mapped to guide the selection of a hydraulic patch classification in Chapter 3 (Figures 2.6, 2.9 & 2.12). In addition a range of quantitative measures of geomorphic diversity were also calculated within each reach so the relationship between channel morphology and hydraulic patch composition and configuration could be explored quantitatively in later chapters. The three elements of geomorphic diversity which have the greatest influence on the hydraulic environment at the reach scale are the variability of the long-profile, cross-section profiles and substrate characteristics (Davis & Barmuta, 1989; Emery et al., 2003; Bartley & Rutherford, 2005). Variability of the long-profile was quantified in four ways. The maximum thalweg amplitude (Emery et al., 2003) and the maximum bedform amplitude (the maximum difference in bed elevation within the inundated area at the highest survey flow) provided indicative and absolute measures of bedform amplitude respectively and represented large scale variability in the long-profile. The mean thalweg amplitude (the mean distance between the peak/trough of all bedforms and the straight line of slope joining the upstream and downstream extents of the thalweg elevation) was calculated as an indication of mean bedform height. Lastly, fractal dimension[1] was calculated to measure small scale variability of the long-profile (Bartley and Rutherford, 2005). Large scale variability reflects the influence of bedform amplitude on the shape of the long-profile whereas small scale variability reflects the influence of substrate. The variability of cross-sectional shape (between bankfull on the right and left banks) was quantified by calculating the mean sinuosity[2] of a subset of cross-section profiles representing the range of CGUs in each reach. This index differentiated between deep, steep-sided trapezoidal cross-sections (high sinuosity) and wide, shallow cross-sections with gently-shelving banks (low sinuosity). The shape and angle of channel margins and bank morphology were visually assessed from graphs of each cross-section (Figures 2.15-2.17). Median particle size and substrate heterogeneity were both calculated from the 200 particles collected during the Wolman random pebble count described in Section 2.5 above. Roughness, defined as $3.5D_{84}$ (Hey, 1979), was calculated as an indicator of flow resistance.

Table 2.3. Measures of geomorphic diversity within each reach.

Measure	Variability of:	River Arrow	River Salwarpe	Leigh Brook
Maximum thalweg amplitude (m)*	Long-profile	1.17	0.36	0.50
Maximum bedform amplitude (m)	Long-profile	2.27	1.75	1.35
Mean thalweg amplitude (m) (standard deviation)	Long-profile	0.59 (0.34)	0.16 (0.11)	0.08 (0.10)
Fractal dimension**	Long-profile	1.00132	1.000483	1.001646
Mean cross-section sinuosity (standard deviation)	Cross-section	1.10 (0.04)	1.17 (0.09)	1.09 (0.03)
D_{50} particle size (mm)	Substrate	31	64	67
Roughness (*k*) ($3.5D_{84}$)***	Substrate	221	403	450
Substrate heterogeneity (D_{10}/ D_{60}) (mm)	Substrate	0.08	0.28	0.08

* after Emery et al., 2003
** Bartley & Rutherford, 2005
*** Hey, 1979

[1] Fractal dimension (D) = log(n) / (log(n) + log(d/L) where n is the number of segments in the line, d is the straight line distance between the start and points of the line and L is the total length of the line.

[2] Sinuosity (S) = Lt / Lsf where Lt is the total length of the line and Lst is the straight line distance between the start and points of the line

Large-scale variability of the long-profile was greatest at the River Arrow according to both measures. Of the River Salwarpe and the Leigh Brook, long-profile variability was smallest at the River Salwarpe based on maximum thalweg amplitude but lowest at the Leigh Brook according to maximum bedform amplitude. The latter measure suggests the presence of coarser substrate and more elevated areas of deposition or marginal shelves that were inundated at high flows. Mean bedform height was approximately four times larger at the River Arrow than the River Salwarpe, which was twice as large as the Leigh Brook. Small-scale variability of the long-profile (fractal dimension) influenced by substrate coarseness was very small at all three sites, but largest at the Leigh Brook, slightly less at the River Arrow and very small at the River Salwarpe. Cross-section sinuosity was greatest at the River Salwarpe, owing to the deep, straight-sided modified section of the reach (Figure 2.16). Mean sinuosity was slightly less but very similar in the River Arrow and Leigh Brook reaches. Both reaches supported deep-straight-sided banks on one side of the channel, however the sinuosity of the River Arrow cross-sections was created by the relatively incised cross-sections at the pools (Figure 2.15, (c)-(f)) whereas the sinuosity of the Leigh Brook cross-sections peaked where coarse substrate introduced variability (Figure 2.17, (e)). In terms of substrate size and heterogeneity, the River Arrow supported uniformly small particles (coarse gravel), the Leigh Brook supported uniformly large particles (cobble) whereas the River Salwarpe supported a large median size class (cobble) but was more heterogeneous than the Leigh Brook. Consequently roughness was greatest at the Leigh Brook and least at the River Arrow.

Chapter Summary

This Chapter has explained how the study sites were selected and described the hydrology and geomorphic diversity of each reach in detail as a basis for exploring the relationship between hydraulic heterogeneity and hydromorphology in Chapters 4 and 5. However before hydraulic heterogeneity can be quantified effectively it is necessary to reduce the complexity of the hydraulic environment into relatively uniform units or patches. A new approach to classifying the hydraulic environment is presented in the following Chapter.

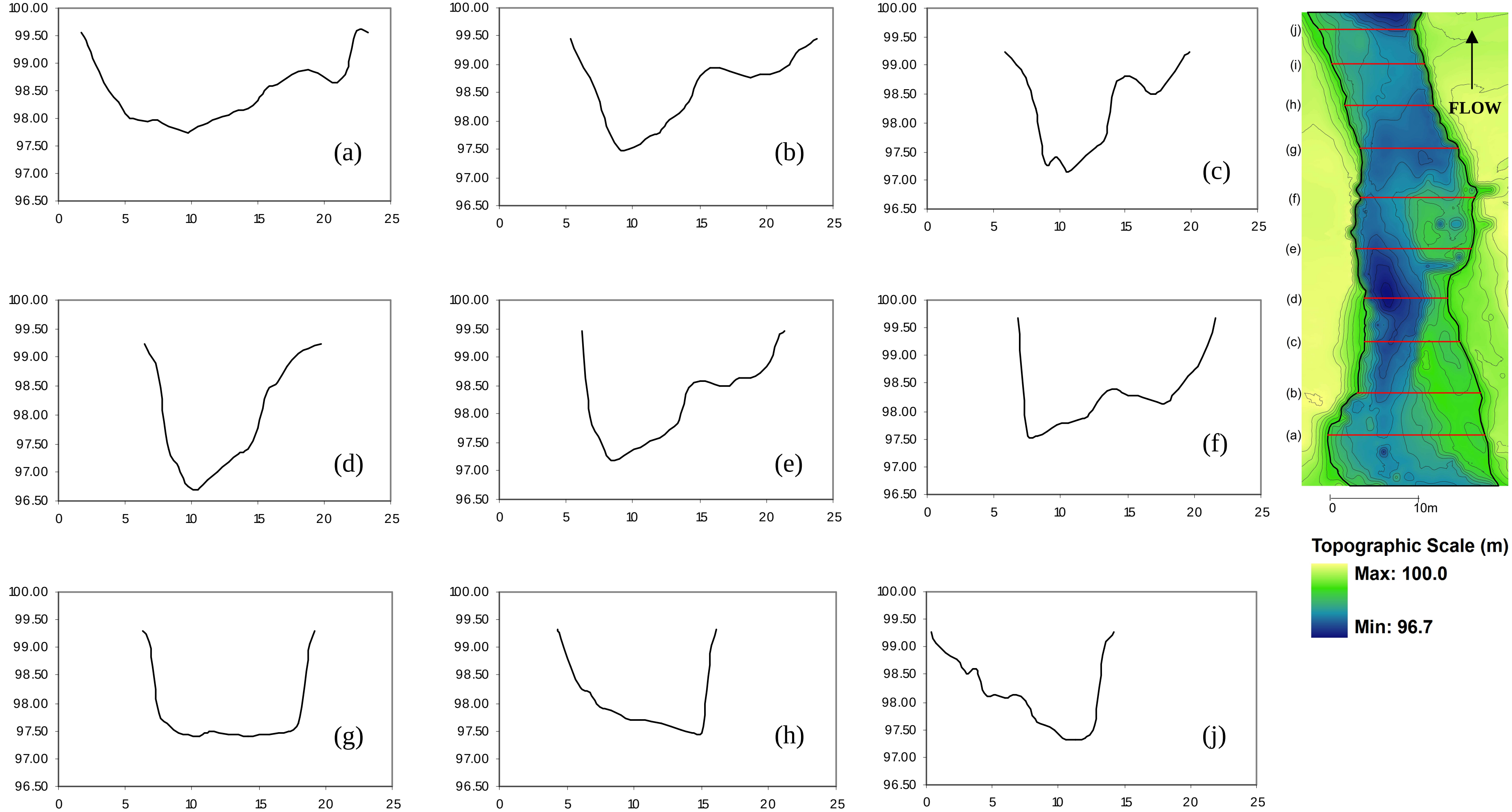


Figure 2.15. Topography of selected cross-sections used to calculate cross-sectional diversity in the River Arrow reach. Inset shows the location of each cross-section and the topography of the reach (overlaid with 20cm contours).

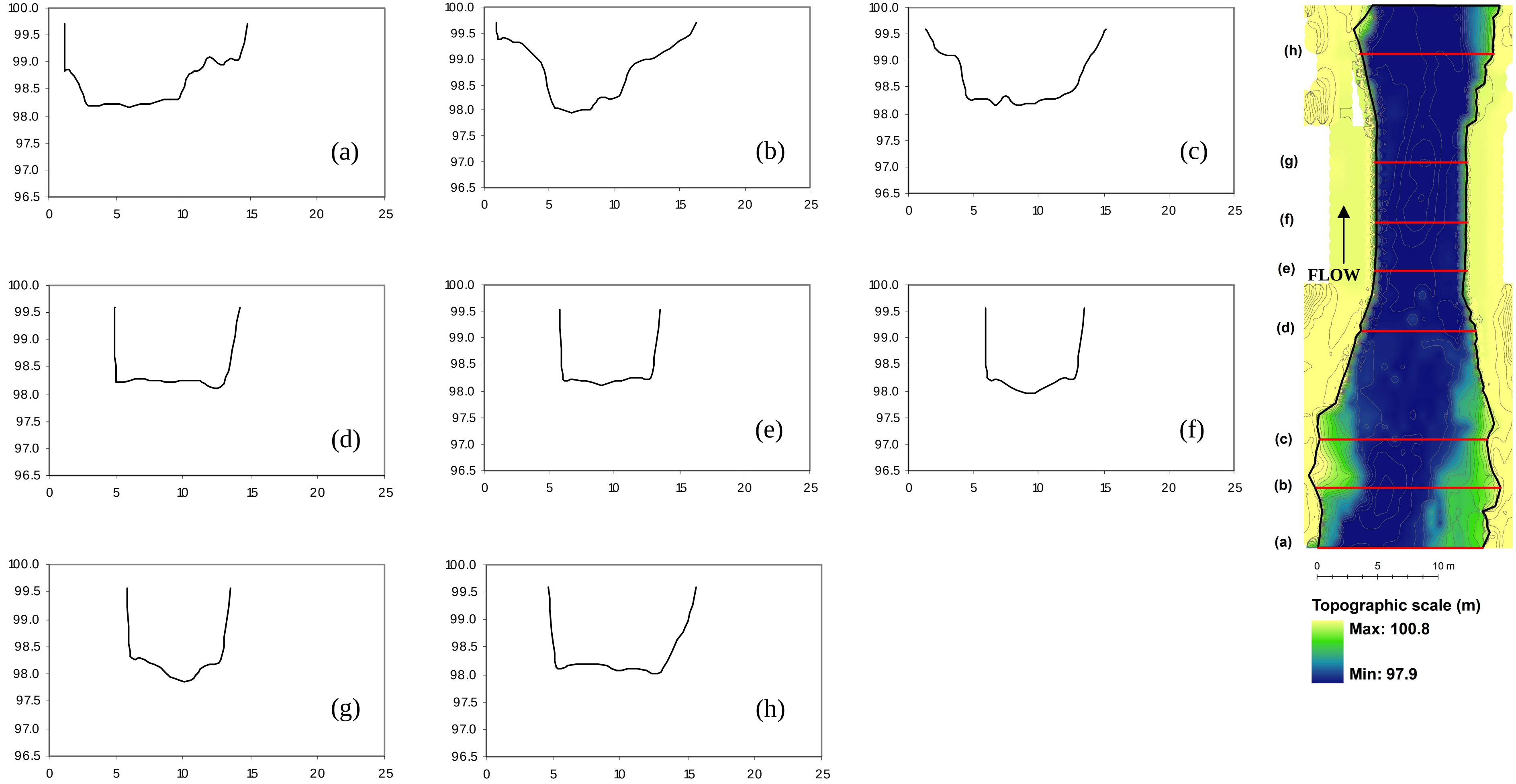


Figure 2.16. Topography of selected cross-sections used to calculate cross-sectional diversity in the River Salwarpe reach. Inset shows the location of each cross-section and the topography of the reach (overlaid with 20cm contours).

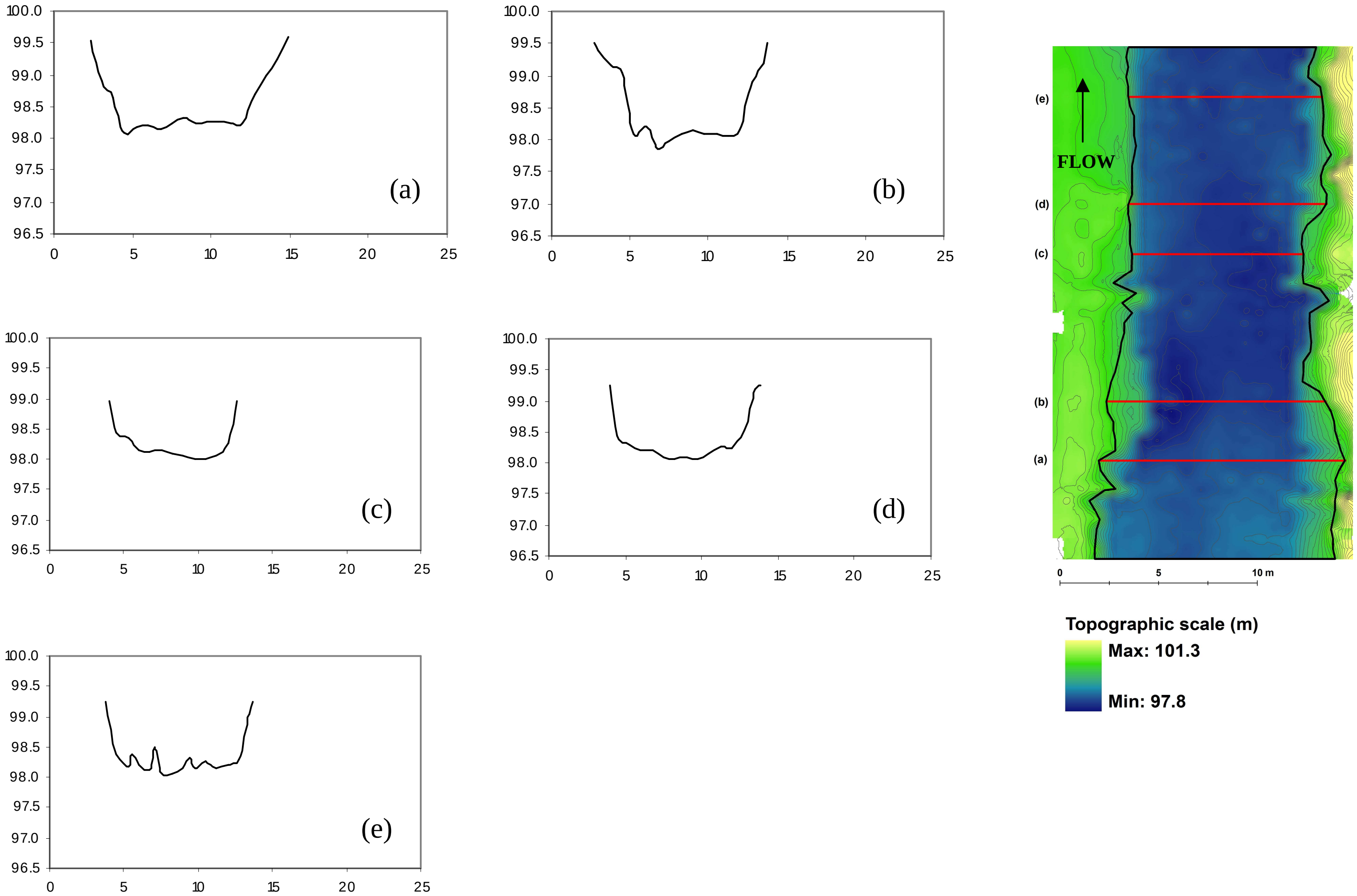


Figure 2.17. Topography of selected cross-sections used to calculate cross-sectional diversity in the Leigh Brook reach. Inset shows the location of each cross-section and the topography of the reach (overlaid with 10cm contours).

"An intelligent human being cannot treat every object it sees as a unique entity unlike anything else in the universe. It has to put objects in categories so that it may apply its hard-won knowledge about similar objects encountered in the past, to the object at hand."

Steven Pinker, How the Mind Works, 1997

3

Developing a quantitative classification of the hydraulic environment

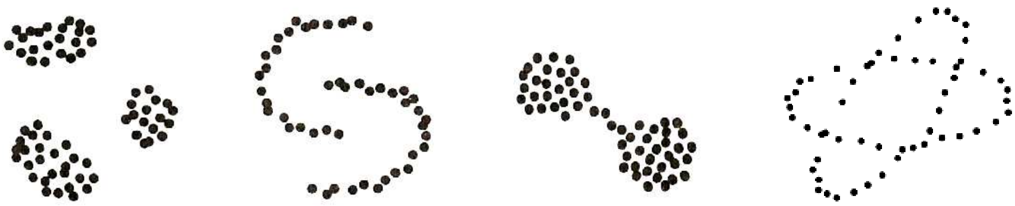
Chapter overview

This chapter addresses the first objective of the thesis – the development of a quantitative classification of the hydraulic environment using fuzzy cluster analysis. Two specific hypotheses will be tested; firstly, that hydraulic patches, defined by the joint distribution of depth and mean column velocity and characterised by within-patch homogeneity and between-patch heterogeneity exist inherently, and secondly, that fuzzy cluster analysis can be used to delineate spatially coherent hydraulic patches and the transitional zones between them in a quantitative and objective manner. The chapter begins with a general introduction to the principles and methods of cluster analysis. This is followed by a review of how cluster analysis has been previously applied in a hydraulic context and identifies how these approaches will be extended in this study. The method of hydraulic data collection and analysis used in this study are then described. Results are presented in the form of a hydraulic patch classification for each site. The chapter concludes with a discussion of whether fuzzy cluster analysis is a reliable and objective hydraulic patch classification tool that improves on current methods and recommendations for its application.

3.1 Cluster analysis

3.1.1 General principles

Cluster analysis is a heuristic tool used to explore the structure of multivariate datasets. The term ‘cluster analysis’ refers to a range of numerical classification methods used to group entities by their attribute similarity. Entities are objects or locations where several attributes/variables have been measured. Entities are normally represented as N data points (feature vectors) in a low-dimensional attribute space (R^p) in which the geometric (typically Euclidean) distance between entities reflects how similar their attributes are (Bezdek et al., 1999; Mathur, 2004). Legendre and Legendre (1998, p.303) define clustering as, “*the search for discontinuities in a continuous environment; a process which…recognises that objects are sufficiently similar to be put in the same group and to also identify distinctions or separations between groups*”. Hence a general definition of a cluster can be given as a group of entities more similar to each other than to entities outside the cluster, characterised by internal cohesion and external isolation (separation) (Cormack, 1971). Everitt (1974) also described clusters as “*…continuous regions of the [data] space containing a relatively high density of points, separated from other such regions by regions containing a relatively low density of points*” (p.44). This general principle is implicitly assumed by all clustering methods and directs the clustering process (von Luxburg & Ben-David, 2005). However it should be noted that the *precise* definition of a cluster depends on the clustering algorithm and similarity measure used to group entities (Milligan, 1996). Clusters can take different shapes, sizes and densities (Figure 3.1) and different clustering methods are needed to detect each type of cluster structure (Legendre & Legendre, 1998). Random and regularly spaced datasets containing no natural clusters are shown in Figure 3.2.



(a) (b) (c) (d)

Figure 3.1. Examples of theoretically well-defined clusters with different levels of internal cohesion (compactness) and/or external isolation (separation): a) cohesive and isolated spherical clusters, b) isolated linear clusters, c) two cohesive, linked clusters and d) overlapping hollow, linear clusters (Reproduced from Gordon, 1981, p.5; Balasko et al., 2001, p.8).

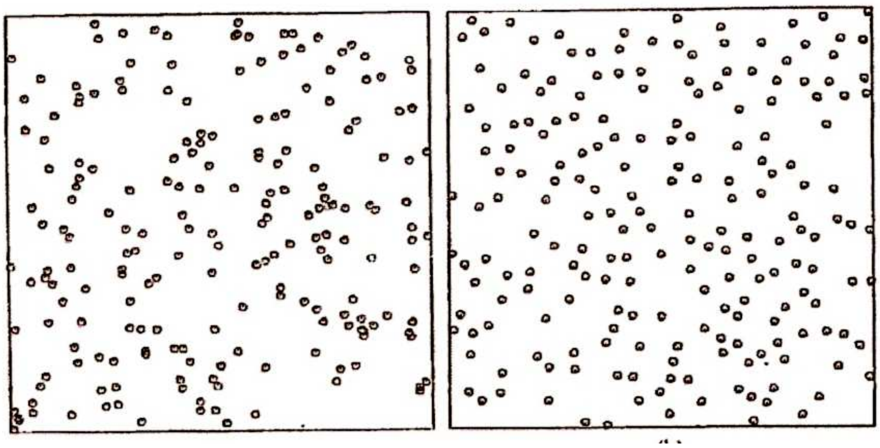


Figure 3.2. Examples of random (left) and regular (right) data containing no "natural" clusters (Source: Panayirci & Dubes, 1983)

A distinction can be made between **supervised** cluster analysis, which allocates entities to *a priori* classes (taxonomy), and **unsupervised** cluster analysis, which groups entities by virtue of their attribute similarity *without* prior knowledge of the number or characteristics of groups in the dataset (typology) (Legendre & Legendre, 1998; Bezdek et al., 1999). A further general distinction can be made between **hard** and **fuzzy** clustering; the former defines clusters within a crisp, Boolean object framework whereas the latter applies fuzzy set theory and admits overlapping clusters with indistinct, areal boundaries (Everitt et al., 2001) (Figure 3.3). This is a considerable advantage when dealing with real-world data that rarely contain neatly bounded classes. Fuzzy cluster analysis is discussed in greater detail towards the end of the following section.

3.1.2 The process of cluster analysis

Cluster analysis is a three stage process including (1) a pre-clustering assessment of cluster tendency, (2) the selection and application of a clustering algorithm to partition the data into groups (clusters), and (3) post-clustering validation of the resulting clusters (Jain & Dubes, 1988; Bezdek et al., 1999).

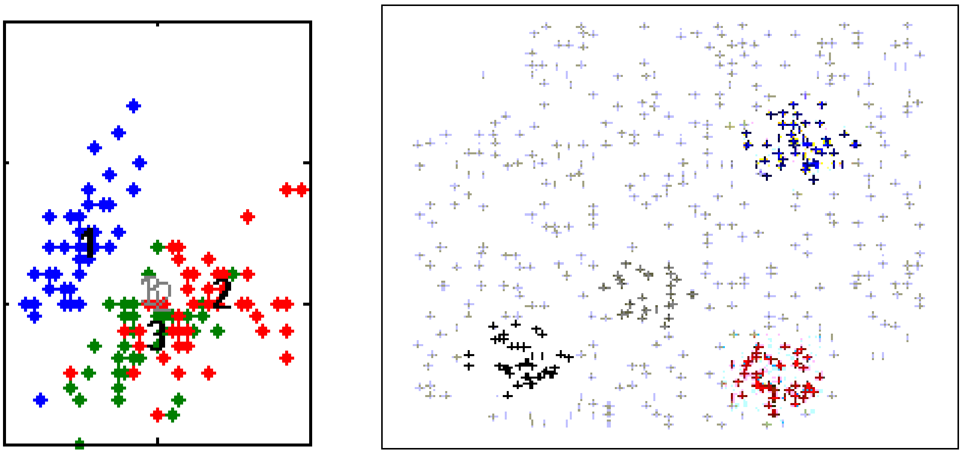


Figure 3.3. Example of fuzzy cluster structures in attribute space: (left) 3 overlapping fuzzy clusters and (right) 4 fuzzy clusters defined as relatively high density sub-regions of the data space. (Source: Fisher, 1936; Kondorf, 1997)

Stage (1) Assessment of clustering tendency

The objective of cluster tendency assessment is to establish whether natural groups are present in the data, as this has important implications for subsequent clustering processes (Jain & Dubes, 1988; Hathaway et al., 2006). Evidence of a multimodal distribution in the form of dense regions of data points interspersed with sparsely populated regions is indicative of a clustered structure (Everitt, 1974) whereas random or regularly spaced data points (Figure 3.2) suggest an unclustered structure (Arnold, 1979; Panayirci & Dubes, 1983). Cluster tendency can be assessed informally through visual inspection of a scatterplot or histogram of the data distribution (Everitt et al., 2001), or more formally using visual assessment tests based on pairwise dissimilarities (e.g. Bezdek & Hathway, 2002). Alternatively, statistical tests of cluster tendency have been developed, such as Hopkin's (1954) test of spatial randomness and Hartigan and Hartigan's (1985) dip test of unimodality (Jain & Dubes, 1988). These test the null hypothesis that the data is uniformly, randomly or unimodally distributed and conclude whether a cluster structure is present if not. However it is not clear how well these tests perform for datasets containing very fuzzy clusters, as datasets used to assess the effectiveness of the tests typically contain a well-defined cluster structure. Some commentators suggest cluster analysis should only be applied to datasets where a clear clustering tendency has been detected, to avoid the risk of partitioning data into spurious/meaningless cluster groupings (e.g. Pal & Bezdek, 1995; Jain & Dubes, 1988). However, Legendre & Legendre (1998) sanction the application of cluster analysis to data *without* natural groupings if a strong practical need for classifying the data into groups can be justified. In the case that clustering tendency is either unproved, in doubt or knowingly absent, it must be acknowledged from the outset that the group structure delineated by the clustering process is imposed on the data. As such the selection of an 'optimal' classification is a subjective decision that must be clearly justified.

Stage (2) Selecting and applying a clustering algorithm

The second stage involves selecting and applying a clustering algorithm to partition the data (*N* entities) into *c* groups (clusters). How this is achieved depends on two factors; how attribute similarity between entities is measured and which clustering method is applied. A vast array of different clustering algorithms has been developed and no single method is appropriate in all situations (Gordon, 1981; Milligan, 1996). The three most common approaches to clustering are 1) hierarchical, 2) optimisation and 3) model-based. As the type of clusters delineated strongly reflects the mathematical model that underlies the clustering method and measure of similarity used, it is important to select a method capable of detecting the type of structure believed to be present in the data or to compare the results of different methods if the structure is unknown (Gordon, 1981). It should be noted that in the majority of clustering methods it is only spatial information in attribute space (rather than real, geographic space) that is taken into consideration when grouping entities – similarity is measured using non-spatial attributes only. Therefore clusters delineated in attribute space will only produce compact clusters in geographic space if the variables being clustered are spatially correlated or regionalised (Burrough et al., 1997). A special form of clustering known as spatially-constrained clustering allows the user to specify that only entities within a specified boundary or geographic distance of each other can belong to the same cluster. This approach can be useful if administrative, regional or national boundaries are relevant to the outcome, but it was not appropriate in this study, or indeed necessary, as depth and velocity do tend to be spatially correlated.

Hierarchical clustering is a popular method that proceeds by successively combining/dividing groups of entities until either all entities are assigned to a single cluster (agglomerative clustering) or occupy their own cluster (divisive clustering) (Everitt et al., 2001). The output is viewed as a dendrogram (tree diagram), where each split/union of branches signifies a decrease/increase in the level of similarity between entities in each group (Everitt et al., 2001). The user must decide which level of similarity in the dendrogram gives the "best" partition, although this is very difficult in practice (Everitt et al., 2001). A major limitation of hierarchical clustering is that once objects are assigned to a group, they cannot be moved to another group in a subsequent step, even if it would increase within-group homogeneity (Hawkins et al., 1982; Kaufman & Rousseeuw, 1990). As a result, there is no guarantee that within-group homogeneity and between-group heterogeneity are maximised in any partition (Legendre & Legendre, 1998). Furthermore, as the name implies, hierarchical clustering is ideally suited to detecting clusters in data with a nested group structure, such as the classification of organisms in a biological taxonomy (Everitt et al., 2001). Where there is no underlying hierarchy, optimisation methods may be more appropriate.

Optimisation cluster algorithms assign N entities into a user-specified number (c) of groups such that a numerical criterion of within-group homogeneity (cluster cohesion) and between-group heterogeneity (cluster isolation) is maximised (Everitt et al., 2001). For example, the commonly used K-means optimisation algorithm minimises the within-cluster sum of squared Euclidean distances between all entities in the cluster with the cluster centroid (MacQueen, 1967). At the beginning of the process c points are selected at random to act as initial cluster centroids and each entity is assigned to the nearest cluster (Estivill-Castro & Yang, 2004). The average distance between all entities in the cluster and its centroid are calculated, the cluster centroid is moved to this point and the process is repeated until within-group variance is minimised (Estivill-Castro & Yang, 2004). The k-means algorithm tends to detect spherical, equal-sized clusters, however by altering the way distance between entities is measured and defined it is also possible to detect clusters with different shapes, orientations and densities (Bezdek, 1981; Estivill-Castro & Yang, 2004). The main problem with optimisation clustering is the need for the user to specify c groups to partition the data into (Legendre & Legendre, 1998). As this is generally unknown *a priori* multiple partitions within a user-specified $c_{\min}$ and $c_{\max}$ range are generated. The optimum value for c is subsequently determined through post-clustering validation (Jain & Dubes, 1988). Optimisation algorithms also suffer from the local minima problem, that is, the algorithm may converge and get stuck at local minima of the numerical criteria rather than find overall minima (Kaufman & Rousseeuw, 1990; Peńa et al., 1999). This is caused by the choice of initial starting position of cluster centroids, which are usually allocated randomly. Specifying initial centroids determined by a prior clustering may improve the chance of finding overall minima, however, the only way to avoid this problem altogether is to calculate all possible partitions ($2 \leq c \leq \infty$) which is computationally impossible (Legendre & Legendre, 1998).

Model-based clustering is not based on distances between entities but instead works on the assumption that the data contains entities from c populations, each characterised by its own probability distribution (Everitt et al., 2001). The aim of model-based clustering is to estimate the parameters (mean and covariance) describing each distribution and the probability of entities belonging to each distribution (Fraley & Raftery, 1998). The most common application of this approach assumes the data contains a mixture of normal distributions (Fraley & Raftery, 2002). Model-based clustering algorithms are designed to detect a particular type of distribution which each cluster in the dataset is assumed to have whereas in reality, entities may come from different shaped probability distributions. This clustering technique also requires prior classification using hierarchical cluster analysis which can be problematic for large datasets (Fraley & Raftery, 1998).

The clustering methods discussed so far are designed to detect c well-separated, mutually exclusive clusters to which entities either do or do not belong. However, a binary Boolean-object model is often inadequate for representing complex, geographic or environmental phenomena which may have indeterminate boundaries and exhibit continuous but regionalised spatial variations (Burrough, 1996; Jensen, 2005). For example, it is common to find transitional zones (ecotones) at the boundaries of different classes, in which entities bear resemblance to two or more classes simultaneously. In a river context, Moir & Pasternack (2008) identified the limitation of linear boundaries for representing hydromorphological units. Natural objects are likely to contain a degree of internal heterogeneity and have greater attribute uncertainty towards their periphery (Burrough, 1996). Zadeh (1965) developed the concept of fuzzy sets as a way to model inexact real-world data characterised by classification uncertainty/class overlap. This concept is applied in fuzzy cluster analysis whereby entities are assigned a membership function value (MFV) that quantifies the degree to which they belong to *each* cluster in the classification (Bellman et al., 1966; Ruspini, 1969; Höppner et al., 1999). Membership functions lie in the interval [0, 1] and must sum to 1 across all clusters (Zadeh, 1965). MFVs quantify how similar an entity is to each cluster centroid (i.e. class prototype); 0.99 indicating almost exact similarity and 0.01 almost no similarity (Gan et al., 2007). Membership function values are not probabilistic; instead they indicate the possibility of belonging to a class.

Figure 3.4a illustrates the membership function values of 18 entities to Group 1 of 2 in a simple 2-fuzzy cluster classification. Points to the lower left of the attribute space have strong membership to Group 1, whereas those in the upper right have no similarity to Group 1. Entities in the middle have some similarity to both groups as indicated by the intermediate membership functions (to Group 1) in the interval [0.2, 0.7]. The membership function value of each entity to Group 2 is equivalent to 1 minus the MFV to Group 1.

As it is often necessary draw a hard/crisp conclusion from a fuzzy classification, a rule for allocating entities with intermediate grades of membership to a single cluster can be applied. This process is known as **defuzzification**. In Figure 3.4a, a cut-off threshold has been applied (dotted line) whereby entities are allocated to the Group to which they have the highest membership function. This 'maximum likelihood' defuzzification rule, suggested by Zhang and Stuart (2001), is simple but makes no use of information gathered in the fuzzy clustering process. Alternative defuzzification rules, such as the Confusion Index (e.g. Burrough et al., 1997) have been designed specifically to mine

MFVs such that entities with confused class membership can be identified and allocated to a transitional boundary zone, as shown in Figure 3.4b. The Confusion Index (CI) quantifies the ratio between the second and first highest membership function values and thus reflects how confused/split membership is between the two groups to which the entity has the highest possibility of belonging (Burrough et al., 1997). When CI approaches zero there is clear membership to a single class and little confusion, but when CI approaches one, the entity belongs to two classes approximately equally and there is a high degree of classification confusion. Burrough et al. (1997) suggested entities with a CI≥0.6 should be allocated to a special intergrade (transitional) class. By mapping the value of the Confusion Index it is possible to, "*indicate parts of the landscape where spatial change in classes is clear and abrupt or diffuse and vague*" (Burrough et al., 2001, p.532). Boundaries delineated using the Confusion Index are defined as zones of class confusion (Burrough et al., 1997).

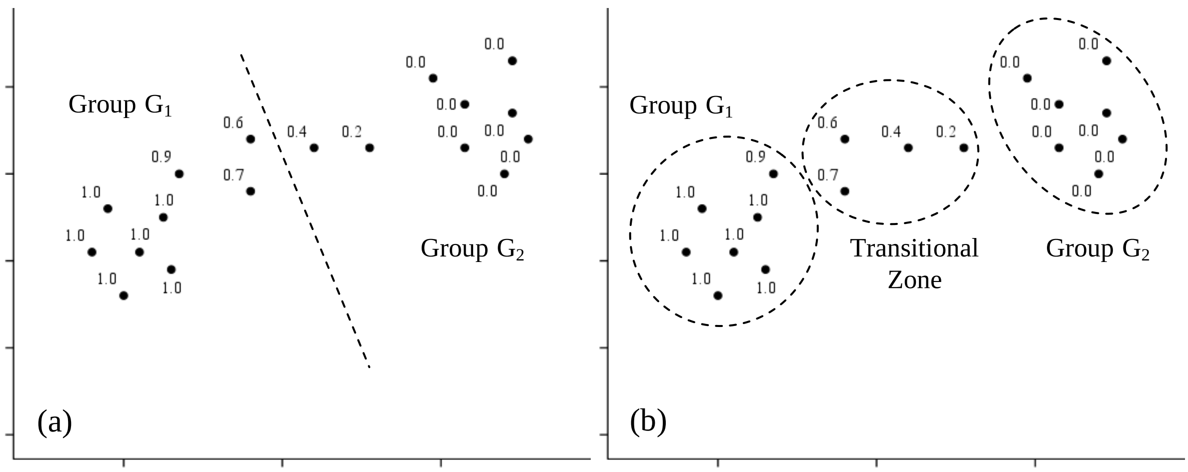


Figure 3.4. Illustration of the fuzzy membership function of each entity to Group 1 of 2 in a 2-group fuzzy classification (Source: Gordon, 1981, p.58). Entities with intermediate membership can be allocated to (a) a single cluster, or (b) a transitional zone.

Whilst the Confusion Index is very effective for identifying transitional boundary zones where classes overlap, it does not guarantee that all members allocated to a single cluster achieve a minimum level of belonging. For example, in a 5 cluster classification the membership function values of an entity to each class could be 0.45, 0.25, 0.15, 0.10 and 0.05. In this case the Confusion Index is less than 0.6 (0.25/0.45) so the entity would be assigned to the highest membership class even though it was less than 50% similar to the class centroid. As a result, clusters defuzzified using the CI>0.6 rule may contain a wide range of membership function values. To avoid this problem an alternative approach to defuzzification was proposed by Cheng et al. (2001) whereby a user-specified threshold, known as an α-cut, is applied to the maximum membership function value such that a specified degree of belonging (MFV≥α-cut) must be reached in order to be assigned to a cluster (Cheng et al., 2001). The α-cut threshold dictates how similar each entity must be the cluster centroid (class prototype) to belong the cluster. All entities with a maximum MFV<α-cut threshold (i.e. less than the prescribed level of similarity to a centroid) are assigned to epsilon bands (zones of low class similarity). This approach allows the user to control classification certainty and class homogeneity, but requires the user choose an appropriate α-cut threshold. This decision can be rather arbitrary and therefore introduces a degree of subjectivity into the defuzzification process. As the threshold increases, all members of a cluster have greater similarity to the cluster centroid but the proportion of entities reaching the threshold decreases so more entities are assigned to the epsilon bands (Cheng et al., 2001). The final classification may be easier to interpret if a high proportion of observations are assigned to clusters (i.e. by using a low α-cut threshold such as 0.5); however, this may obscure important information about membership to other classes and produce misleading results. For example, under a 0.5 α-cut threshold an entity with MFVs of 0.51, 0.40, 0.05, 0.02, 0.01, 0.01 in a 5-cluster classification would be allocated to the first class, even though its membership is relatively evenly split between the first two classes. Alternatively, under a relatively high α-cut threshold (e.g. ≥0.7) an entity which clearly belongs to one cluster more than any other (e.g. if MFVs were 0.68, 0.12, 0.10, 0.08, 0.02 in a 5-cluster classification) would be assigned to an epsilon band if its maximum MFV was less than the α-cut threshold. It is clear that the two approaches to defuzzification have a slightly different focus; the Confusion Index targets the identification of transitional boundary zones where classes overlap and entities have split membership whereas α-cut thresholds target the identification of homogeneous class "cores" in which entities have a user-specified degree of similarity to the cluster centroid (i.e. class prototype). The most appropriate rule to use depends on the aims of the study and the purpose of classification. The two rules can be used in combination to counteract the weaknesses discussed above.

Fuzzy clustering algorithms are generalisations of crisp optimisation methods such as the k-means algorithm (Höppner et al., 1999). For example, the fuzzy c-means (FCM) algorithm, developed by Bezdek et al. (1984), detects spherical fuzzy clusters by minimising the sum of squared Euclidean distances between entities and their cluster centroid but relaxes the binary membership function constraint of the k-means algorithm. Adaptations to the distance measure used in the fuzzy c-means algorithms were later introduced by Gustafson and Kessel (1979) and Gath and Geva (1989) to detect ellipsoidal clusters with unequal sizes and densities (Gan et al., 2007). Figure 3.5 illustrates the variation in cluster structure imposed by different clustering algorithms in a 4-cluster fuzzy classification. The dataset (Fig. 3.5a), which shows head acceleration of a motorcyclist through time after an impact, appears to contain four zones or clusters; constant speed, rapid deceleration, rapid acceleration and a return to constant speed. The fuzzy c-means algorithm (Fig. 3.5b) imposes 4 well-separated, spherical clusters that do not provide a very intuitive classification of the data. By contrast, the adaptive distance function used in Gustafson-Kessel (Fig. 3.5c) and Gath-Geva algorithms (Fig. 3.5d) which allows the shape, size and density of each cluster to be defined separately, detects the inherent ellipsoidal cluster structure of the data much more clearly.

In addition to specifying how many clusters (c) the data should be partitioned into, fuzzy cluster analysis requires the user to define, m, the fuzzifier, a weighting exponent applied to the distance between entities and cluster centroids (Mathur, 2004). When m=1 membership functions become completely crisp (binary). As the value of m increases, membership functions to each cluster approach equality and the classification becomes increasingly fuzzy – the distinction between clusters becomes more blurred and class overlap increases (Bezdek et al., 1984). Bezdek et al. (1984) suggested the useful range of values for m is [1, 30], although the best choice is $1.5 \leq m \leq 2.5$ (Pal & Bezdek, 1995), and often the "most valid" partition is the "least fuzzy" (Bezdek et al., 1981, p.98). The user may choose to partition the data into c groups multiple times, each time varying the value of m, however the addition of a second unknown parameter complicates the identification of the optimal classification. The implications of varying this parameter usually requires a separate study. In practice a constant value (usually m=1.5 or m=2) is often used and has proven appropriate in a range of applications (e.g. Burrough et al., 2001; Arrell et al., 2007; Legleiter & Goodchild, 2005).

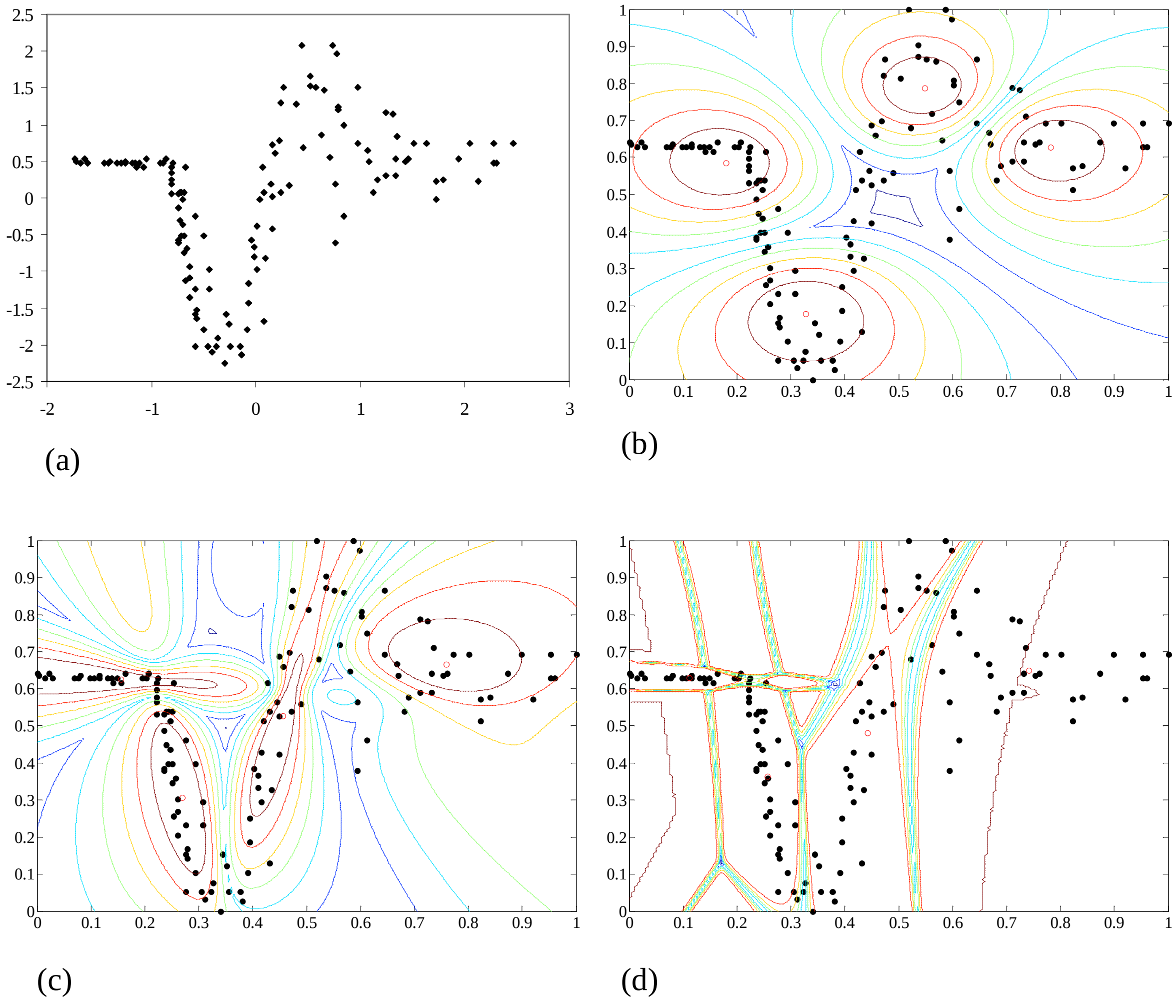


Figure 3.5. Example of the 4-cluster structures imposed by different clustering algorithms on the same dataset; (a) raw data, (b) fuzzy c-means, (c) Gustafson-Kessel and (d) Gath-Geva (Source: McNames, 2000; Balasko et al., 2001).

Stage (3) Cluster validation

Cluster analysis partitions a dataset into groups regardless of whether the data has a natural group structure or not. Therefore further evaluation is necessary to determine which partition, if any, contains meaningful clusters. Cluster validation, the third stage of cluster analysis, aims "*to evaluate the results of cluster analysis in a quantitative and objective fashion*" (Jain & Dubes, 1989, p.143) so that the optimal classification of the data can be selected. Methods of validation vary depending on whether or not the data has a natural clustering tendency.

If the dataset is believed or shown to have a natural group structure based on a prior assessment of clustering tendency, the purpose of cluster validity is to identify which classification matches the true structure of the data and detects the correct size, shape and number of clusters (Gordon, 1981; Höppner et al., 1999). This is achieved by calculating an objective measure of goodness-of-fit for each classification. As Jain and Dubes (1988) explain, "…*External criteria measure performance by matching a clustering structure to a priori information…[whereas] Internal criteria assess the fit between the structure and the data, using only the data themselves*" (p.161). External criteria judge goodness-of-fit in respect of variables not used in the clustering process, for example, judging the meaningfulness of hydraulic patches by testing whether they support different biotic communities. Often this type of data is not available or relevant external variables do not exist.

Internal criteria, often referred to as validity indices, measure the quality of individual clusters or the overall classification by quantifying cluster compactness and/or separation (Höppner et al., 1999). These work on the assumption that a valid cluster is an unusually compact (dense) and unusually isolated (separate) region of the data space (Jain & Dubes, 1989; Bensaid et al., 1996). Validity indices can be plotted for each value of *c* and visually assessed for discontinuities which may be indicative of the appropriate number of clusters (Templ et al., 2008). However this method can be misleading as large discontinuities may occur for *un*clustered data (Jongman et al., 1995) or the indices may fail to detect the optimal number of clusters for a dataset with a known cluster structure unless it is very well defined (e.g. Templ et al., 2008). Furthermore, as each index defines compactness and separation slightly differently, it is likely that predictions of optimal *c* may vary between indices (Jongman et al., 1995). Often the maximum or minimum value of a validity index is taken as indicating the best-fit partition (Höppner et al., 1999). However, as Jain and Dubes (1989, p.189) point out, *"…it is very difficult to fix thresholds on such indices that define when the index is large or small enough to be considered "unusual""*. To overcome this issue, it has become increasingly popular to attempt to define the null distribution of validity indices so significant departures from this can be assessed statistically (Jain & Dubes, 1989; von Luxburg & Ben-David, 2005). However, generating such test distributions for statistical validation is difficult, time consuming and does not guarantee that all cluster structures will be recognised (Hartigan, 1977; Gordon, 1981). As Gordon (1981) concludes, *"…it is unlikely that…any* *criteria…will find widespread acceptance in a strict hypothesis-testing sense, because of the difficulty of anticipating the behaviour of relevant statistics under the great diversity of different structures which could be present in the data*" (p.126). Furthermore there is no agreement in the literature about which criterion/validity index is best to apply (Milligan & Cooper, 1985; Pal & Bezdek, 1995; Kim et al., 2004). As each index makes a different assumption about cluster structure it is advisable not to rely on any single index but compare several (Everitt et al., 2001). After reviewing the performance of twenty three fuzzy cluster validity indices in conjunction with the fuzzy *c*-means algorithm, Wang and Zhang (2007) concluded that no validity index identifies the correct number of clusters for all datasets. As Pal and Bezdek (1995) warned, "*no matter how good your index is, there is a dataset out there waiting to trick it (and you)*" (p. 153). In view of the weaknesses of formal validation techniques Baxter (1994) concluded, "*…informal and subjective criteria, based on subject expertise, are likely to remain the most common approach*" (p.167).

Where the data does *not* have a natural group structure, all partitions impose an *artificial* group structure hence elaborate statistical validity testing to find a significantly compact and isolated partition is fruitless (Jain & Dubes, 1988; Höppner et al., 1999). Legendre and Legendre (1998) point out that where cluster analysis is used to describe the structure of a continuum, "*…it is immaterial to wonder whether these clusters are "natural" or unique"* (p.304). Instead the grouping reflects the particular clustering method and measure of similarity used; each clustering method will return a unique grouping of the same input data and the definition of "cluster" is peculiar to the method by which it was formed (Milligan, 1996). Höppner et al. (1999) warn against calculating validity indices to compare clustering results from data that lacks an underlying clustered structure, as any variation in index value is merely a reflection of innate preferences for certain parameter combinations rather than a reliable indicator of better or worse partitions; all partitions are equally "untrue". In this situation cluster validation is a more subjective process guided by the user's interpretation of the optimal classification. It is possible, even if all the classifications are artificial, that one is more useful or informative than another insofar as it fulfils the original purpose of classifying the data (Templ et al., 2008). As Everitt et al. (2001) point out, "*…it should be remembered that in general a classification of a set of objects is not like a scientific theory and should be judged largely on its usefulness, rather than in terms of whether it is "true" or "false"*" (p.4). Examples of subjective criterion used in clustering studies to select the optimal classification have included the spatial coherence of clusters when mapped in geographic space (Emery et al., 2003; Templ et al., 2008), the intuitive meaningfulness and distribution of classes based on subject expertise and knowledge of the study area (e.g. Burrough et al., 2001; Arrell et al., 2007) and how well environmental variations are differentiated (Emery et al., 2003).

3.2 From qualitative to quantitative classifications of the hydraulic environment

In Chapter 1 it was suggested that to understand how hydromorphology influences the ecological health of rivers, it is first necessary to assess its influence on the hydraulic environment which provides the physical living space for freshwater biota. It was argued that hydraulic patches, defined by the joint distribution of depth and velocity, could provide a useful means of reflecting meso scale hydromorphology. The limitations of existing physical habitat assessment methods for delineating such hydraulic patches effectively were discussed. Qualitative visual surveys of surface flow types involve a high potential for misclassifying or under-representing hydraulic differences which, "*…compromises repeatability, precision and transferability*" (Poole et al., 1997, p.816). Ecohydraulic modelling, whilst quantitative, only provides a species-specific view of the hydraulic environment based on modelled hydraulic data. Furthermore, both methods impose crisp, linear boundaries on hydraulic patches/habitats which misrepresent the spatial ambiguity associated with natural boundaries in a continuous environment.

Several studies (Inoue & Nakano, 1999; Emery et al., 2003; Legleiter & Goodchild, 2005) have highlighted the potential for using cluster analysis to develop a quantitative, data-driven classification of the hydraulic environment using single or multiple hydraulic variables. Inoue & Nakano (1999) used hierarchical cluster analysis (Ward's method) to define microhabitat units using five variables: mean depth, mean velocity, velocity variability, substrate heterogeneity and substrate coarseness. An eight unit classification was selected on the basis of clusters being "interpretable" (p.601) and significantly different in at least one variable. Of these, five units had significantly different mean depth and six had significantly different mean velocity. All eight units supported significantly different habitat use patterns by juvenile masu salmon (*Oncorhynchus masou*).

Emery et al. (2003) successfully used hierarchical cluster analysis to assess the hydraulic performance (defined by mean column velocity) of pool-riffle bedforms at low, intermediate and high flows in two physically contrasting rivers. Four different agglomerative hierarchical clustering algorithms were evaluated, of which Ward's method was selected as the best on the basis that it provided *"…the most appealing overall results in terms of cluster size, shape (compactness), density and internal homogeneity"* (p.543). Velocity data were clustered on a site-by-site and flow-by-flow basis. Multiple partitions containing two through to ten clusters were generated for comparison. At each site a 6 cluster solution was deemed to provide, "*…the most informative and well-defined classification of hydraulic patches*" (Emery et al., 2003, p.543). Solutions with fewer clusters failed to distinguish areas judged to have different hydraulic character whereas solutions with larger numbers of clusters resulted in spatial noise when clusters were mapped in geographic space. Subsequent analyses of variance showed that the number of *significantly* different velocity classes varied between three and six at each flow. Intermediate flows produced the highest number of velocity classes in the unmodified reach whereas the number of significantly different classes decreased with discharge in the semi-engineered reach where bedform amplitude was much lower. The results clearly illustrate the interactive effect of channel morphology and discharge on the hydraulic environment. It is debatable whether the use of hierarchical cluster analysis was appropriate in Emery et al.'s (2003) and Inoue & Nakano's (1990) studies as data were collected and analysed at a single spatial scale. As Hawkins et al. (1982) advised, "*…users should be wary of using hierarchic methods if they are not clearly necessary*" (p.317). Nevertheless, the studies successfully showed that hydraulic patches delineated by cluster analysis reflect hydromorphology and have relevance to biota.

Both Emery et al. (2003) and Inoue and Nakano (1999) used hard clustering which imposes crisp boundaries around hydraulic patches. Legleiter & Goodchild (2005) recognised that, "*…an innovative fuzzy approach could circumvent the subjectivity of conventional habitat classification and provide a richer representation that more faithfully honors the complexity of the fluvial environment*" (p.30). Legleiter and Goodchild (2005) used the fuzzy *c*-means algorithm to classify 1m^2 resolution hydraulic data extracted from a 2D hydrodynamic model using four variables; flow depth, velocity magnitude, Froude number and shear velocity. The model was run at baseflow conditions in a 625m reach of the Kananaskis River in Canada. 135 unique classifications were generated using (*c*, *m*) combinations in the range $2 \leq c \leq 10$ and $1.5 \leq m \leq 2.5$ (0.1 increments). Although eight validity indices were calculated to aid selection of the optimal (*c*, *m*) combination, the majority of indices increased or decreased monotonically with *c* and *m* and were unreliable indicators, probably due to a lack of clustering tendency in the input data. Therefore the classification had to be chosen subjectively based on knowledge of conditions at the study site. A prior visual survey had identified the presence of four habitat types (eddy drop zone, riffle, run/glide and pool). The four-cluster fuzzy classification was found to produce, "*spatially continuous, compact and hydraulically reasonable*" classes (p.37) and was selected as the optimal classification.

Legleiter and Goodchild (2005) went on to demonstrate how indices of classification uncertainty can be calculated from fuzzy membership functions and mapped in geographic space to explore the spatial variability of classification uncertainty. Legleiter and Goodchild (2005) suggest zones of high classification uncertainty might, "*contain several habitat types, features unlike those found elsewhere in the channel, or variability at a scale finer than the spatial resolution of the [data]*" and ventured that, "*the heterogeneity, uniqueness, and/or complexity of these zones of ambiguity make them, in a sense, the most interesting portion of the stream*" (p.15). Variations in the width of the transitional zone distinguish between relatively crisp boundaries separating distinct habitats and gradual areal boundaries that may contain a variety of habitat conditions. The authors highlight that the proportion of the channel assigned to transition zones depends not only on the defuzzification threshold chosen by the user, but also the nature and quality of the data. Whilst fuzzy cluster analysis enables the user to explore and represent the hydraulic continuum in much greater detail, the final classification is site-, data-, and potentially user-specific (Legleiter and Goodchild, 2005).

3.3 Methods

3.3.1 Data collection and preparation

Hydraulic surveys were repeated at five discharges at each site. Where possible, very low (Q80-Q99), low (Q60-Q80), moderate (Q40-Q60), high (Q20-Q40) and very high (<Q20)

flows were targeted at each site. Surveyed discharges were exceeded 88%, 70%, 53%, 22% and 13% of the time at the River Arrow site, 89%, 67%, 38%, 21% and 17% of the time at the River Salwarpe site and 87%, 67%, 45%, 23% and 14% of the time at the Leigh Brook site. Flow exceedence percentiles for the River Arrow and River Salwarpe sites were calculated from best available data from Environment Agency gauging stations. For the Leigh Brook, a flow duration curve was constructed from mean daily flow data (converted from mean daily stage data using a discharge rating curve) collected at the University of Worcester hydrological monitoring station between December 2007–February 2011.

A grid sampling strategy was adopted to ensure all areas of the channel would be represented evenly (Inoue & Nakano, 1999; Rivas-Casado *et al.*, 2005). Although a sub-metre sampling resolution (e.g. 0.5.m x 0.5m) was preferred, this approach severely limits the length of reach that can be surveyed in the field during a relatively stable period of discharge (Inoue & Nakano, 1999; Legleiter & Goodchild, 2005). To overcome this problem streamwise sampling resolution was reduced to 1m to maximise the length of reach surveyed at the River Arrow and River Salwarpe sites. Data were subsequently interpolated to 0.5m x 0.5m resolution using an ordinary spherical kriging model with 2m variable search distance in ArcGIS v9.3.1 (ESRI, 2009). This created regularly spaced points that could be converted to raster format as required for spatial analyses (Thoms et al., 2006). All original measurements were preserved. The more complex geomorphology at the Leigh Brook produced hydraulic variability at a smaller spatial scale and therefore a 0.5m x 0.5m sampling resolution was necessary. This limited the length of reach surveyed to 26m.

At each discharge surveyed, point measurements of water depth (m) and streamwise mean column velocity (at 0.6 depth) (ms^{-1}) were collected at each node in the sampling grid. Velocity at 0.6 depth is commonly used as a measure of mean column velocity (e.g. Wadeson & Rowntree, 1998; Emery *et al*., 2003; Moir & Pasternack, 2008) with relevance to fish (e.g. Holm et al., 2001) and benthic invertebrate (e.g. Jowett, 2003) habitat. Velocity was measured with a Valeport 801 electromagnetic current meter (±0.5%, ±0.5cm/s) in wadeable conditions and a Teledyne RDi StreamPro ADCP (±1.0% ,±0.2cm/s) elsewhere. In practice, only the high-flow dataset at the River Arrow was collected using an ADCP.

3.3.2 Data analysis

Prior to analysis, data from all hydraulic surveys at each site were combined, plotted as bivariate scatterplots and visually inspected for any extreme outliers. Where present, these were removed to prevent undue influence on the cluster analysis. Depth and velocity point data at each flow were also converted to raster format for mapping purposes and stored as layers in ArcGIS. Each point measurement was represented as a $0.25m^2$ pixel.

Assessment of clustering tendency

The combined hydraulic datasets collected at each site were plotted in a 2D histogram and scatterplot to examine the joint distribution of depth and velocity and facilitate an informal visual assessment of cluster tendency. The Hopkins statistic was calculated for the standardised hydraulic data at each site using MATLAB code developed by Wester and Steinberg (2008) to test the null hypothesis that the data were randomly or uniformly distributed (i.e. unclustered). In addition the scatterplots of hydraulic data collected at each individual discharge (Figures 3.7, 3.23 and 3.38) were visually inspected for evidence of natural fuzzy clusters, that is, regions with a higher density of points separated by regions with a lower density of points.

Application of fuzzy cluster analysis

At each site, data from all hydraulic surveys were combined and standardised (z-scores) using SPSS v.14 (SPSS Inc, 2005) to account for different scales of measurement. Hydraulic data were clustered on a site-by-site basis for the range $2<c<8$and a fixed value of $m=2$ using Balasko et al.'s (2001) Fuzzy Clustering and Data Analysis Toolbox for MATLAB (The Mathworks, 2009) and the Euclidean distance metric. This was repeated with three fuzzy clustering algorithms for comparison; fuzzy *c*-means (Bezdek et al., 1984), Gustafson-Kessel fuzzy covariance (Gustafson & Kessel, 1979) and Gath-Geva unsupervised optimisation algorithm (Gath & Geva, 1989), generating a total of 21 partitions at each site.

Selecting the optimal classification

The decision to pursue statistical validity testing or use subjective criteria to select the optimal classification was dependent on the outcome of the cluster tendency assessment. Where a natural cluster tendency in the pooled hydraulic data was revealed by the Hopkin's statistic, four validity indices including Xie-Beni, separation index, scaled partition coefficient and scaled classification entropy were calculated to quantify cluster compactness and separation to aid *ob*jective comparison between classifications (Xie & Beni, 1991; Pal & Bezdek, 1995; Davé, 1996; Balasko et al., 2001; Burrough et al., 2001). To aid *sub*jective selection of an optimal classification, each classification was visually represented using a scatterplot of the hydraulic data which was overlaid with cluster centroids and contours illustrating the shape, size and extent of clusters. In addition a crisp classification (defuzzified using the simple maximum likelihood defuzzification rule) was mapped in geographic space so the location and spatial coherence of clusters in each partition could be evaluated. The purpose of classifying the hydraulic environment was to assess how it is influenced by variations in discharge. Therefore the merits of each classification were judged according to three criteria. Firstly the ability of the classification to reflect discharge dependent changes to the hydraulic patch structure was considered. This was done by inspecting each classification of the pooled data to see how well any natural fuzzy clusters detected in the data collected at individual discharges were reflected. This checked that patterns in the distribution of data at each individual discharge were captured by the classification. Secondly the location of mapped clusters at low to moderate flows were compared with a detailed field sketch of geomorphic and topographic features in the reach to confirm links between hydraulic patches and channel morphology. Clusters present at low flows that did not have clear links to morphological features were considered to be spurious artefacts of the clustering process, rendering the classification sub-optimal. Thirdly classifications were judged by the spatial coherence of its clusters when mapped in geographic space. Classifications with lots of spatial noise and high proportion of single pixel patches were not considered useful for assessing the location and configuration of hydraulic patches.

Mapping fuzzy clusters and classification confusion

The location and extent of each *fuzzy* cluster in the optimal classification was illustrated by mapping fuzzy membership function values. The Confusion Index was mapped to illustrate where patch types overlapped and assess whether patch boundaries were crisp or gradual.

Delineating transitional zones (hydraulic patch boundaries)

The transition zone (TZ) was defined using a combination of two rules. Firstly entities with a Confusion Index > 0.6 were allocated to the transition zone to represent areas of the channel where class membership was confused and hydraulic patch types overlapped. Secondly, a 0.5 α-cut threshold was applied to all remaining entities to identify those that were not strongly characteristic of any cluster (i.e. max MFV<0.5) despite not having confused class membership. This extra threshold rule ensured that all entities assigned to clusters were at least 50% similar to their cluster centroid (prototype). After applying both defuzzification rules a classification containing a transitional zone and crisp clusters (hereafter referred to as a hydraulic patch types) was produced. Each hydraulic patch type was labelled with a site code (RA - River Arrow, LB - Leigh Brook, or RS - River Salwarpe) and a unique number. This was done to avoid the subjectivity associated with descriptive labels such as pool, glide and run and to avoid pre-determined notions about the hydraulic nature of such features that in reality may vary between individual instances and/or sites. The location and extent of each hydraulic patch type and the transition zone were then mapped at each discharge. Descriptive statistics were computed for each hydraulic patch type to illustrate their hydraulic characteristics.

3.4 Results

3.4.1 River Arrow hydraulic patch classification

Hydraulic survey data

Table 3.1 shows the maximum, minimum, range and spread of depths and velocities recorded at each hydraulic survey at the River Arrow site. Maximum depth was higher at the two lowest flows than the intermediate flows due to high flows scouring the channel between data collection periods. Mean depth increased with discharge, although the range and variance of depths varied very little. Mean velocity increased gradually with discharge, however the variance also increased indicating a greater spread of values at higher flows. The largest range of velocity values occurred at intermediate and high flows (Q53 and Q22).

Figure 3.6 illustrates the spatial variability of depth and velocity throughout the reach. Two very deep, slow-flowing areas in the middle and downstream extent of the reach were clearly evident at all flows. At low flows areas of high velocity were limited to the topographic high points in the channel (Figure 3.6). As discharge increased areas of fast-flow extended longitudinally, covering the entire length of the reach at very high flow. A large backwater zone, characterised by deep water and upstream eddies, was evident in the downstream half of the reach, adjacent to the right bank.

Table 3.1. Descriptive statistics of hydraulic data collected at each discharge, River Arrow.

Hydraulic parameter	$0.21 m^3/s$, Q87	$0.30 m^3/s$, Q70	$0.42 m^3/s$, Q53	$0.87 m^3/s$, Q22	$1.41 m^3/s$, Q13
$Depth_{max}$	1.33	1.26	1.22	1.28	1.41
$Depth_{min}$	0.01	0.01	0.01	0.01	0.02
$Depth_{mean}$	0.31	0.30	0.35	0.41	0.51
$Depth_{variance}$	0.07	0.06	0.06	0.06	0.06
$Depth_{range}$	1.32	1.25	1.21	1.27	1.39
$Velocity_{max}$	0.884	0.940	1.192	1.109	1.223
$Velocity_{min}$	-0.080	-0.199	-0.634	-0.610	-0.253
$Velocity_{mean}$	0.096	0.107	0.157	0.243	0.448
$Velocity_{variance}$	0.012	0.021	0.036	0.070	0.110
$Velocity_{range}$	0.964	1.139	1.826	1.719	1.476

A scatterplot of hydraulic data at each individual flow and all flows combined were produced to illustrate the data distribution in attribute space (Figure 3.7). Two potential outliers were identified at -0.634m/s (Q53) and -0.610m/s (Q22). However these values occurred in zones of turbulence at the boundary between the thalweg and slow-flowing areas so were not considered unreasonable and were retained in the dataset. Figure 3.7 illustrates the greater spread and shift of hydraulic data towards faster, deeper water as flow increases. At Q13 (Figure 3.7e), a clear distinction between fast and slow-flowing areas emerged. However the combined discharge data appeared to have a continuous distribution (Figure 3.7f).

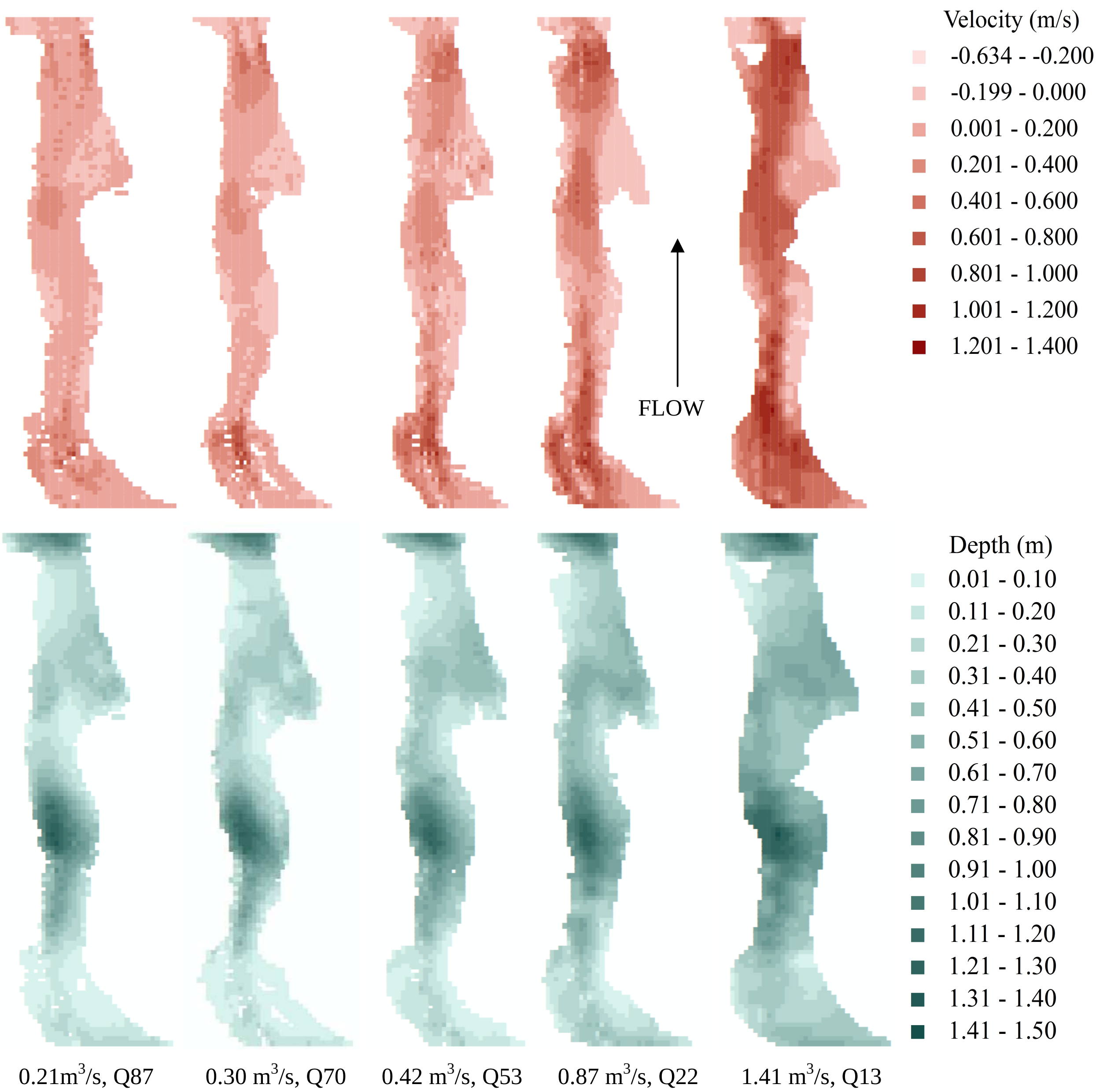


Figure 3.6. Variation in the mean column velocity (top) and depth (bottom) measured at each discharge throughout the reach, River Arrow (discharge increases from left to right).

Assessment of cluster tendency

The Hopkins statistic confirmed that the hydraulic data pooled from all discharge surveys were neither randomly nor uniformly distributed (H=1). The alternative hypotheses – unimodality (i.e. unclustered) or multimodality (i.e. clustered) were assessed visually by inspecting a scatterplot of the bivariate data distribution (Figure 3.7f). Natural clusters, if present, are evident as unusually dense and unusually isolated sets of data points (Jain & Dubes, 1988; Banjeree & Davé, 2004). The data appeared to have a continuous unimodal distribution with the greatest density of data points concentrated in the lower left region of the data space at depths ≤0.5m and velocities ≤0.250ms^{-1}. Data density decreased gradually towards the limits of the data range, and was particularly sparse in the deep-fast region. A 2D histogram of the bivariate data distribution (Figure 3.8) suggested the possible presence of three small local distribution peaks at [-0.025-0.025ms^{-1}, 0.3-0.35m], [-0.025-0.025ms^{-1}, 0-0.05m] and [0.175-0.225ms^{-1}, 0.05-0.1m] and a fourth, much smaller peak at [0.625-0.675ms^{-1}, 0.35-0.40m]. However, these local distribution peaks were not isolated or separated by regions of relatively low density and were therefore not indicative of a well-defined cluster structure.

Visual inspection of the distribution of hydraulic data collected at individual discharges (Figure 3.9) did suggest the presence of some natural fuzzy clusters. At very low to moderate flow an elongated cluster of points with a velocity of 0ms^{-1} and depth of 0-0.45m was evident (Figure 3.9 a-c). At very high flow, and to a lesser extent at high flow, the data split into two main clusters, distinguishing the fast and slower hydraulic conditions (Figure 3.9 d)

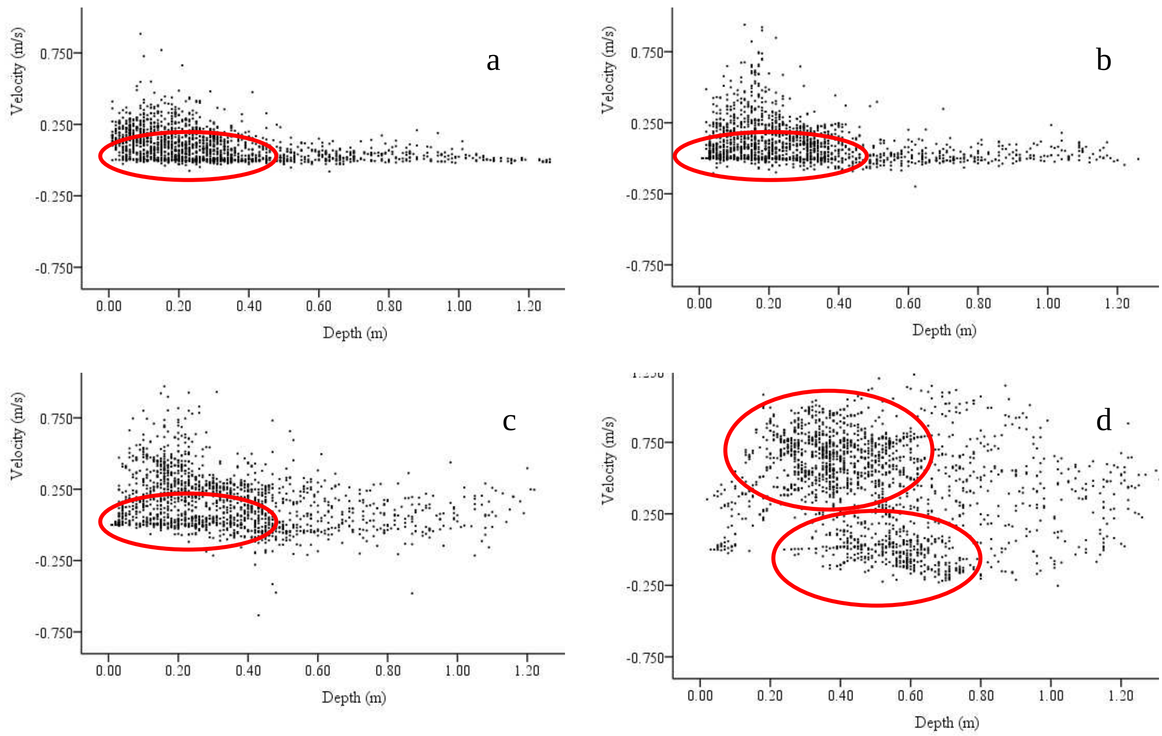


Figure 3.9. Natural fuzzy clusters evident in the data distribution at (a) very low, (b) low (c) moderate and (d) very high flow, River Arrow.

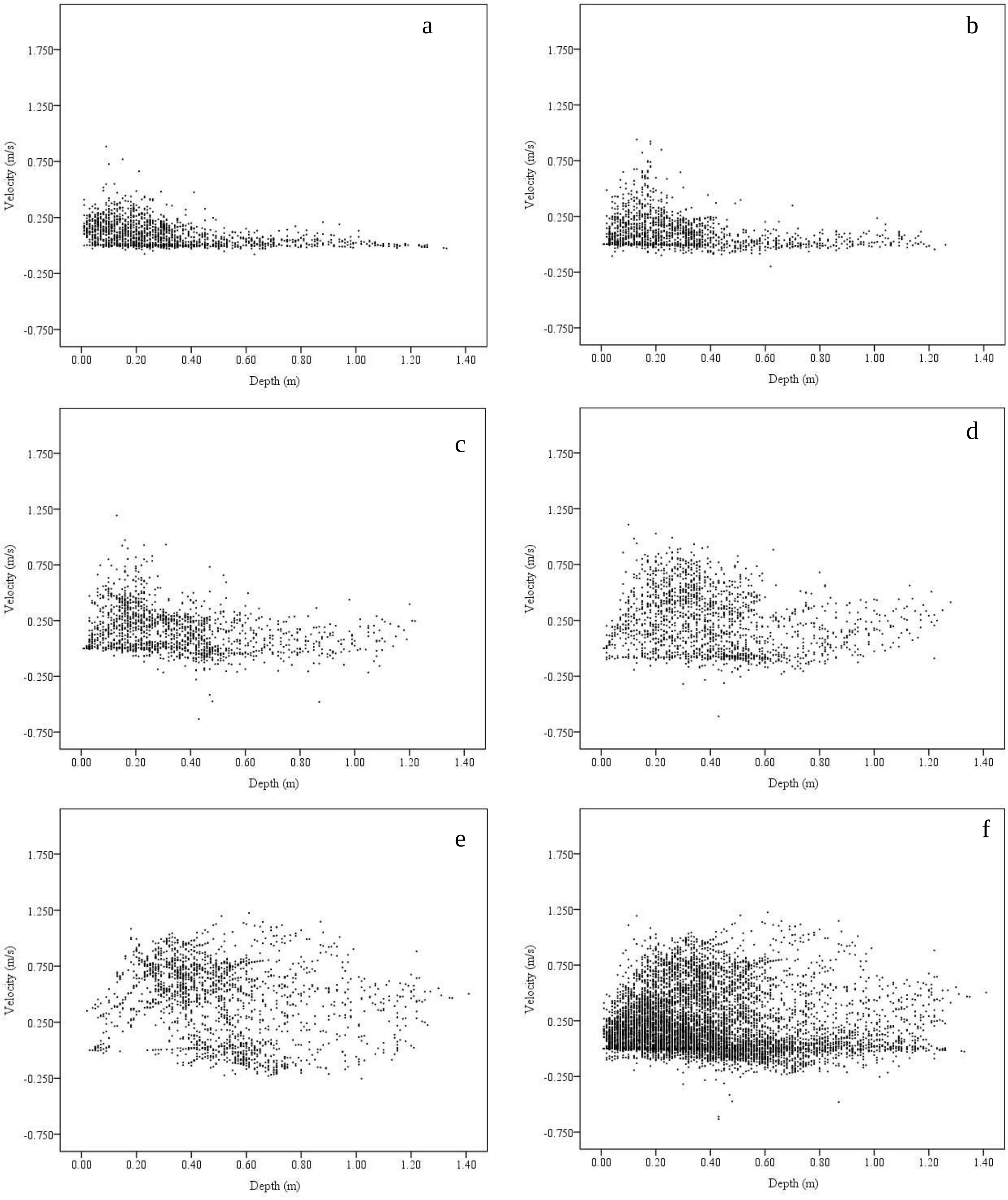


Figure 3.7. Bivariate distribution of depth-velocity measurements collected at (a) very low flow (0.21m^3/s, Q89), (b) low flow (0.30 m^3/s, Q70), (c) moderate flow (m^3/s, Q56), (d) high flow (m^3/s, Q22), (e) very high flow (m^3/s, Q13) and (f) all flows, River Arrow.

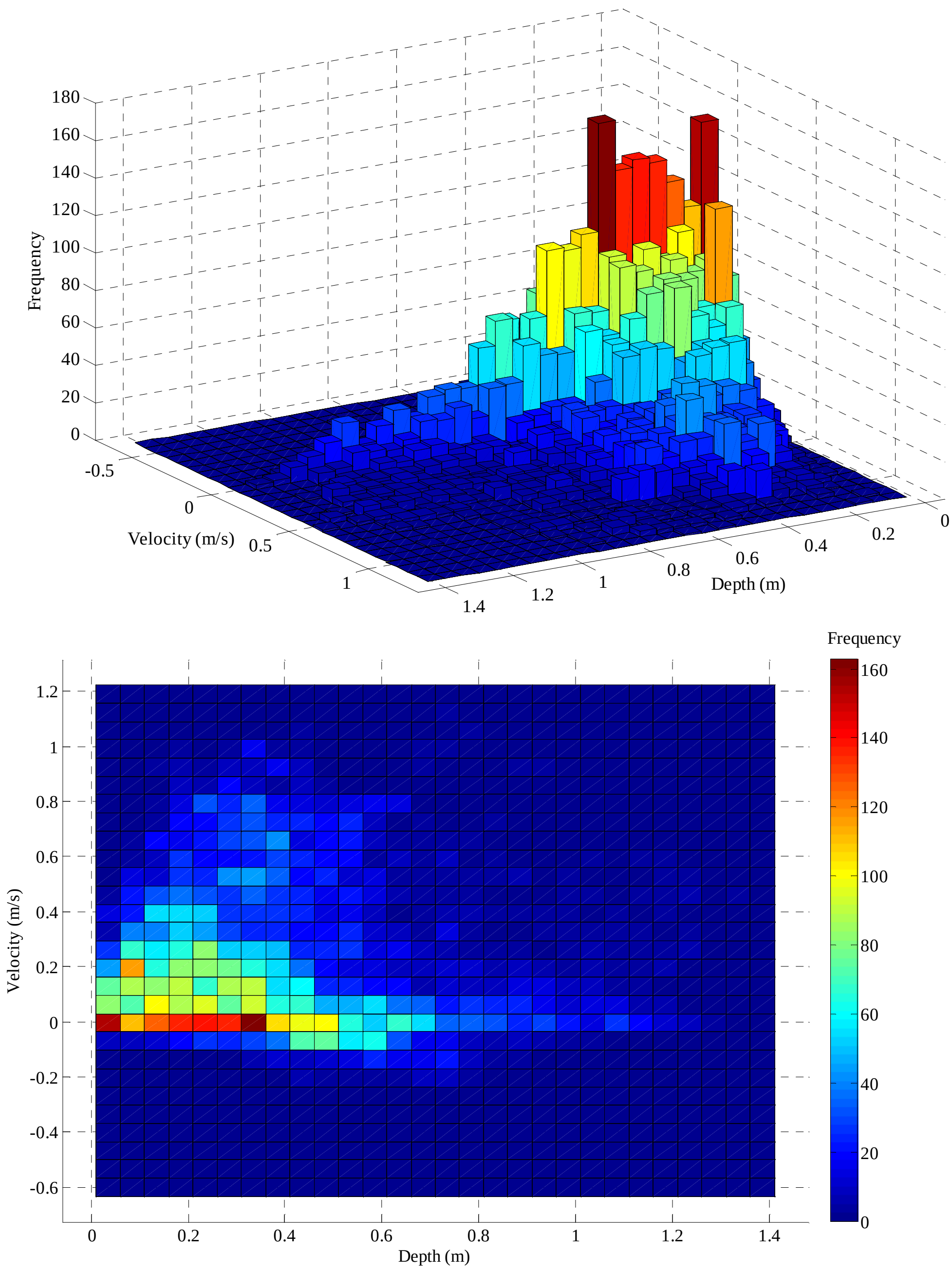


Figure 3.8. 2D histogram (top) and intensity plot (bottom) showing the bivariate distribution and frequency of all depth-velocity measurements collected at the River Arrow.

Selecting the optimal classification

As the pooled data did not exhibit a clear cluster tendency all the cluster partitions generated in the clustering process *imposed* a group structure on the data rather than reflecting an inherent group structure. Hence the use of validity indices to select the optimal classification was inappropriate. Instead the optimal classification was selected subjectively. To aid this process all classifications were represented graphically in attribute space, mapped in geographic space and visually compared (Appendix B). The classification was assessed by how well it reflected the natural fuzzy clusters evident in data collected at individual flows, as a measure of its ability to reflect the influence of discharge. The location of mapped clusters at low flows was assessed in relation to channel topography (Figure 3.10) and geomorphic features (Figure 3.11) that could have provided an underlying cause for hydraulic variations. This section includes a full explanation of how the optimal classification was selected, with reference to Figures 3.9-3.11 and Appendix B. The same decision-making process was used to select the optimal classification for the River Salwarpe and Leigh Brook.

The Gath-Geva algorithm, which optimises the fuzzy *c*-means partition by detecting clusters of different sizes, shapes and densities, proved computationally unstable for the River Arrow hydraulic dataset. The algorithm only converged for the 2, 3 and 6 cluster solutions and on this basis was not examined any further. Each classification produced by the fuzzy c-means and Gustafson-Kessel algorithms was then examined in turn.

The 2-cluster **fuzzy c-means** classification provided little hydraulic differentiation, merely delineating the shallow-fast (topographic high) and deep-slow (topographic low) areas of the channel (Appendix B, Table 1, Figures 1-2). The influence of discharge variations on the hydraulic environment was not reflected effectively. The addition of a third cluster (moderate-fast) addressed this limitation thus providing a basic representation of the hydromorphology (Appendix B, Table 2, Figures 3-4). The 3-cluster classification delineated the main CGUs (pool, run, glide) in the channel (Figure 3.11). The spatial extent of each cluster and the degree of cross-stream hydraulic variation were defined more precisely than would be typical of a visual bankside survey. Nevertheless, the classification was relatively coarse and provided minimal hydraulic differentiation, particularly at the two lowest flows where 76% of data points were assigned to a single

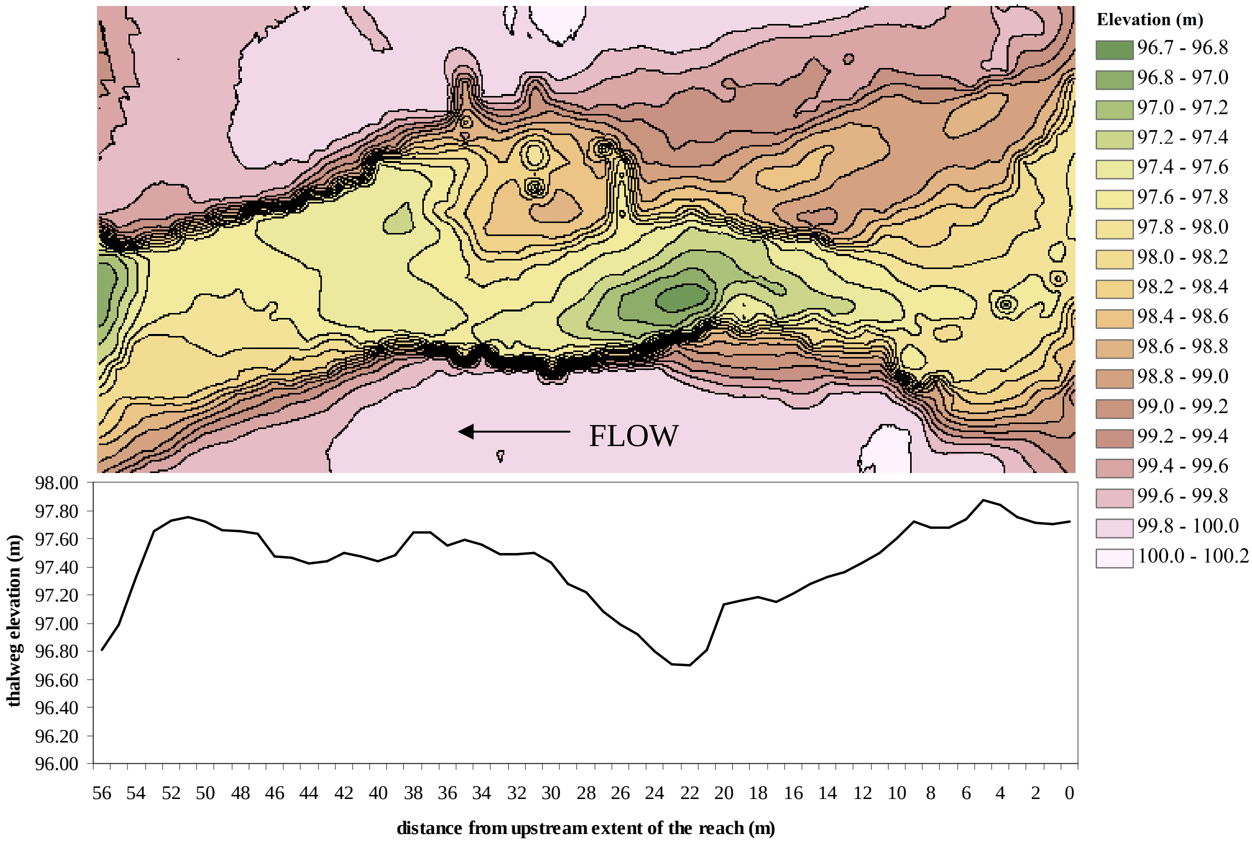


Figure 3.10. Channel topography (top) and variation of the thalweg elevation (bottom) in the River Arrow study reach.

patch type (Appendix B, Table 2, Figures 3-4). The 4-cluster classification (Appendix B, Table 3, Figures 5-6) delineated an additional slow-flowing patch type with moderate depth. It identified areas characterised by upstream eddies/recirculation and was judged to be a useful addition to the classification. This patch type reflected two hydromorphic controls/features; firstly the sudden decrease in depth at the pool margins and secondly, the narrowing of the channel immediately downstream of the main pool which, in combination with the presence of a submerged mid-channel bar, created a large backwater adjacent to the right bank (Figure 3.11). The latter, which was characterised by standing water, decaying organic matter and deposition of fines, provided very different habitat conditions to the pool margins. The addition of a fifth cluster was also useful as it differentiated between zones of moderate velocity along the channel centreline and shallow, slow-flowing zones near channel margins (Appendix B, Table 4, Figures 7-8). The 6-cluster classification (Appendix B, Table 5, Figures 9-10) introduced a fourth slow-flowing patch type with moderate depth. Had the distinction between the moderate, slow-flowing patch type and deep, slow-flowing patch type reflected the different formative processes of the backwater and pool margins (mentioned previously), the addition of a sixth patch type may have been justifiable. However this was not the case hence the sixth cluster complicated the classification unnecessarily and was regarded as representing a transitional zone between shallow, slow-flowing margins and recirculation zones, rather than a patch type in its own right. The addition of a seventh and eighth cluster also introduced spurious patch types which did not provide further insight into the relationship between hydromorphology and in-stream hydraulics. Hence the 5-cluster classification was judged most appropriate. Although the 5-FCM classification delineated hydraulically reasonable patch types that reflected hydromorphic processes/features, hydraulic differentiation and spatial coherence of cluster 3 at the two lowest flows were relatively poor. Hence the classifications generated by the Gustafson-Kessel clustering algorithm were examined to assess whether ellipsoidal clusters provided better results.

The 2-cluster Gustafson-Kessel classification was considerably different, differentiating between fast and slow areas of the channel rather than deep and shallow areas (Appendix B, Table 8, Figures 15-16). The 3-cluster GK classification was very similar to the FCM classification and identified the topographic extremes and the discharge-dependent moderate-fast patch type (Appendix B, Table 9, Figures 17-18). Greater differences were evident in the 4-cluster classification (Appendix B, Table 10, Figures 19-20) in which all the moderately deep/deep and slow-flowing areas were classified as a single patch type. Shallow to moderately deep areas of the channel were classified into three patch types with slow, moderate and fast velocity. The 4-GK classification favoured differentiation by velocity whereas the 4-FCM classification favoured differentiation by depth. However neither classification provided maximal hydraulic differentiation. By contrast the 5-cluster Gustafson-Kessel classification (Appendix B, Table 11, Figures 21-22) did provide an improvement over the 5-cluster FCM classification. In addition to the moderate-fast and deep-slow patch types that 5-FCM also distinguished, the 5-GK classification delineated shallow-slow, moderate-very slow and moderate-slow patch types. The shallow-slow patch type identified the three topographic high points in the channel at low flows and the newly inundated shallow areas at high flows. The moderate-very slow patch type described the natural fuzzy cluster evident in the individual data distributions at very low to high flow. This patch type delineated the slow-flowing recirculation zones at the margins of the pools and in the backwater. The moderate-slow patch distinguished the relatively faster flow of the thalweg at very low to high flow. This classification improved the spatial coherence and longitudinal sequencing of hydraulic patches at low flows and produced the most intuitive representation of the hydromorphology. As was the case with the fuzzy c-means classification, using a higher number of clusters did not reflect new hydromorphological features or processes and merely divided the hydraulic continuum into more classes. On this basis the 5-GK classification was selected as the optimal classification for the River Arrow reach.

Fuzzy cluster membership distributions and classification uncertainty

Table 3.2, which shows the hydraulic characteristics of each cluster centroid (prototype), confirms that the 5-GK classification identified five fuzzy clusters with distinct hydraulic character. Although the mean depth of clusters 1, 3 and 4 were similar, the velocities were very different. The location and extent of each fuzzy cluster at each discharge were illustrated by mapping the spatial distribution of membership function values for each cluster (Figures 3.12-3.17). Pixels with high membership to fuzzy cluster 1 (moderate-moderate) formed a linear-shaped patch following the channel centreline/thalweg, downstream of the pool at all but the highest flow (Figure 3.12). Pixels with partial membership were mainly distributed at the boundary of this main patch but also occurred in other places throughout the channel suggesting the high degree of class overlap between this and other hydraulic patch types. Pixels with high membership to fuzzy cluster 2 (shallow-moderate) showed strong geographic zoning at the topographic high points in the reach during low flows (Figure 3.13). The number of pixels with partial membership increased at high flows as velocity increased, causing considerable overlap with fuzzy cluster 4. Pixels with high membership to fuzzy cluster 3 (moderate-slow) were located around at the margins of the two deep, slow-flowing areas and in the backwater pool at all flows, although the strength of class membership and spatial extent both decreased with discharge (Figure 3.14). Partial membership occurred in many locations throughout the reach, demonstrating the overlap with every other fuzzy cluster in the classification. At low and intermediate flow, only pixels with partial membership to fuzzy cluster 4 (moderate-fast) were evident and these occurred in a very restricted area along the channel centreline at the topographic high point near the upstream extent of the reach (Figure 3.15). At high flow a higher percentage of pixels with high membership

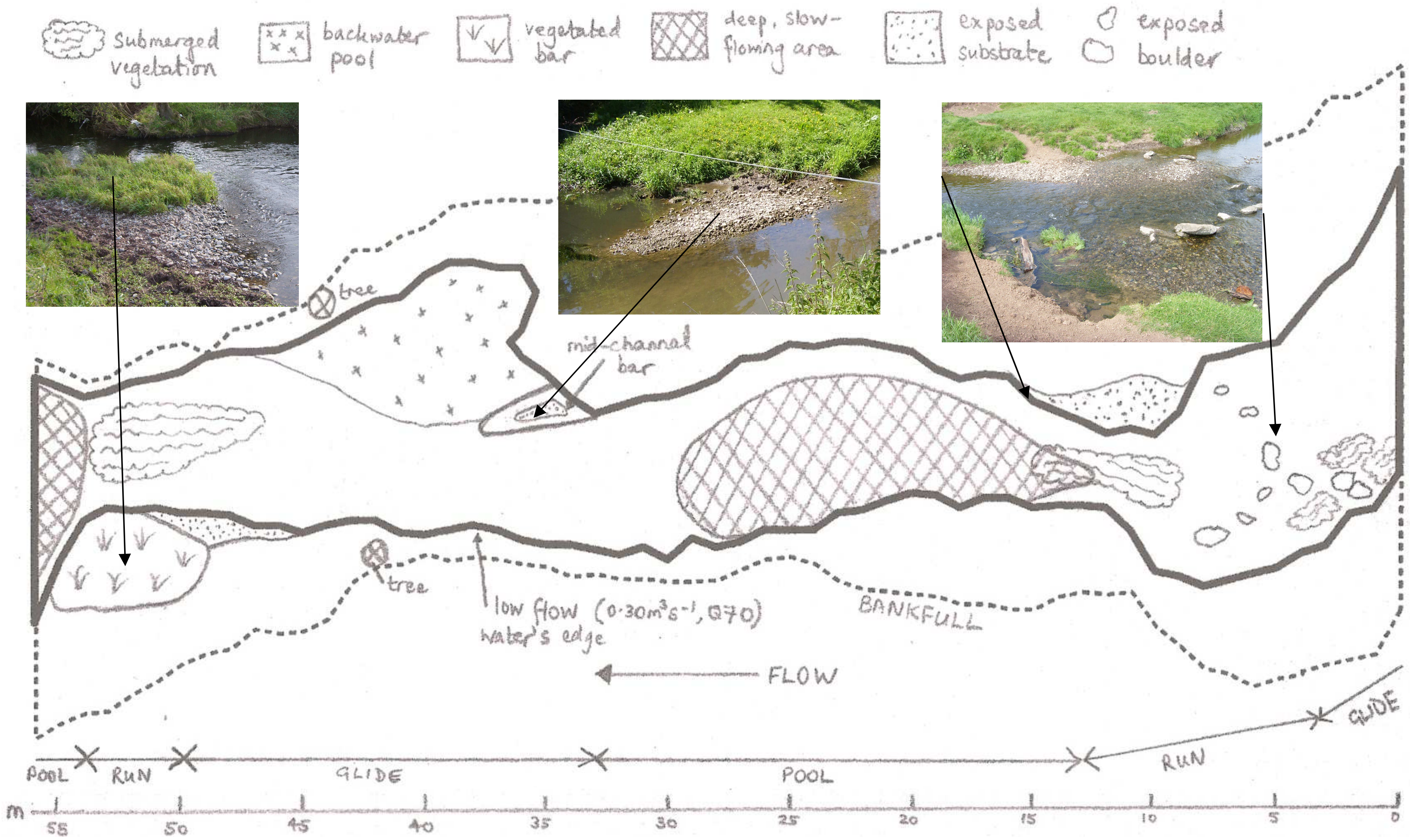


Figure 3.11. Annotated field sketch of the main geomorphic features in the River Arrow study reach. The extent of channel geomorphic units identified during a bankside rapid habitat assessment are also shown.

appeared at all three topographic high points. At very high flow high membership pixels covered approximately two thirds of the reach length along the channel centreline. A degree of class overlap with fuzzy cluster 5 (deep-slow) was also evident at very high flow. Membership to fuzzy cluster 5 (deep-slow) showed strong geographic zoning indicative of a spatially-correlated hydraulic patch type (Figure 3.16). At low to moderate flows membership to this cluster was almost binary due to very few cells having partial membership. At the two highest flows there was a degree of class overlap with fuzzy cluster 3 (moderate–slow) as depth in the recirculation zone increased and membership values to fuzzy cluster 5 fell, caused by the increase in velocity weakening the similarity of many pixels to the cluster centroid.

Table 3.2. Cluster centroids for the 5–cluster Gustafson-Kessel classification.

Cluster	Hydraulic description (depth-velocity)	Depth (m)	Velocity (ms^{-1})
1	Moderate-slow	0.37	0.217
2	Shallow-slow	0.14	0.272
3	Moderate-very slow	0.38	0.001
4	Moderate-fast	0.39	0.696
5	Deep-slow	0.86	0.062

Spatial variation of the Confusion Index was mapped to illustrate where zones of class overlap/confusion occurred (Figure 3.17). The variation in width of the dark areas reflects whether class boundaries were relatively crisp (linear) or “diffuse and vague” (Burrough et al., 1997). At the two lowest flows, areas with a high degree of class overlap occurred, in the most part, as relatively crisp linear boundaries between different clusters. Larger patches of high class confusion were evident in the areas immediately upstream of each pool, suggesting that this was an area of transition between shallow-fast and deep-slow conditions. These also coincided with patches of submerged vegetation (Figure 3.11) which may have created more variable hydraulic conditions and increased class confusion. The Confusion Index was spatially noisy in the upstream extent of the reach, possibly due to the presence of exposed boulders and patches of submerged vegetation (Figure 3.11) which created many flow refugia with deeper, slower water in an otherwise shallow, moderate to fast-flowing area.

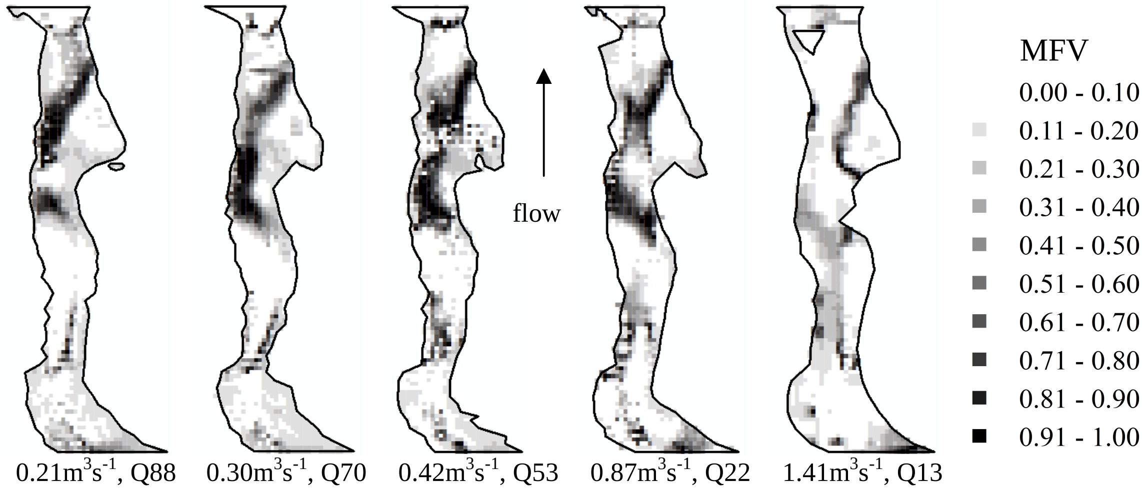


Figure 3.12. Spatial variation of membership function values to fuzzy cluster 1 in the 5-cluster Gustafson-Kessel classification, River Arrow.

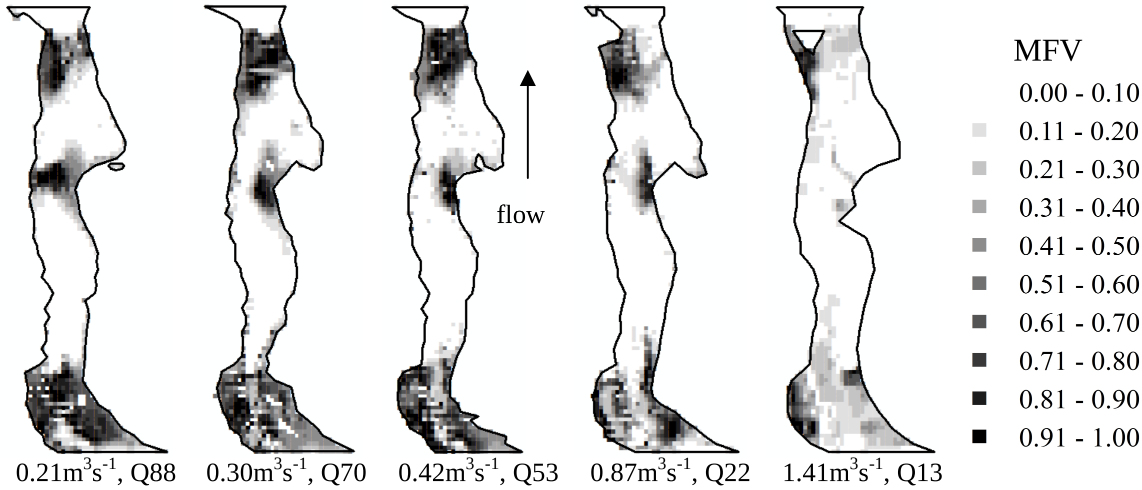


Figure 3.13. Spatial variation of membership function values to fuzzy cluster 2 in the 5-cluster Gustafson-Kessel classification, River Arrow.

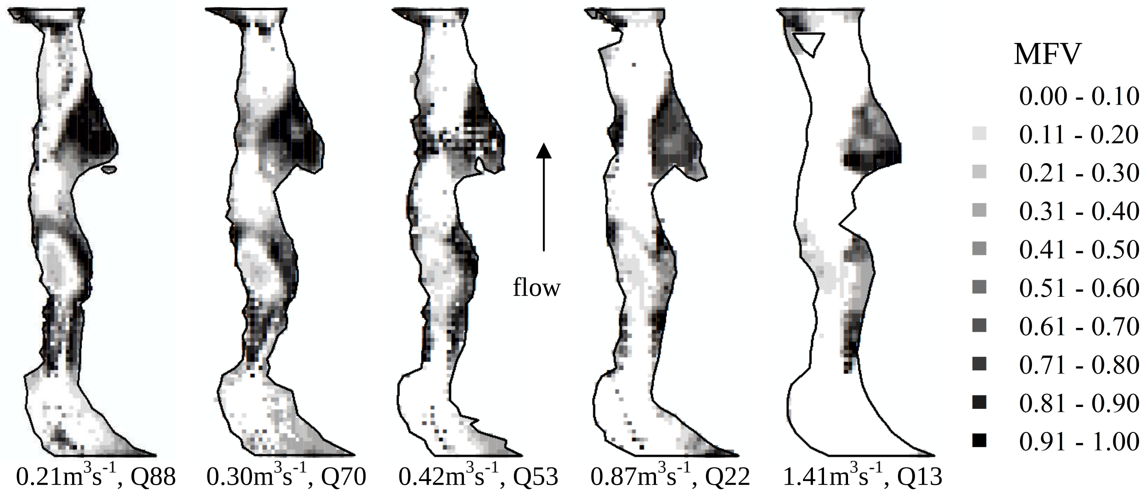


Figure 3.14. Spatial variation of membership function values to fuzzy cluster 3 in the 5-cluster Gustafson-Kessel classification, River Arrow.

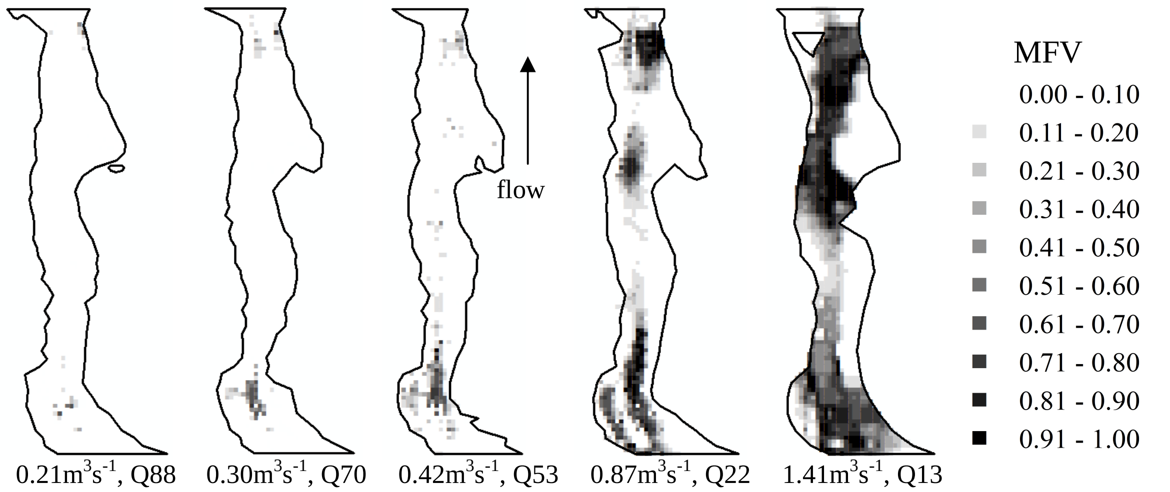


Figure 3.15. Spatial variation of membership function values to fuzzy cluster 4 in the 5-cluster Gustafson-Kessel classification, River Arrow.

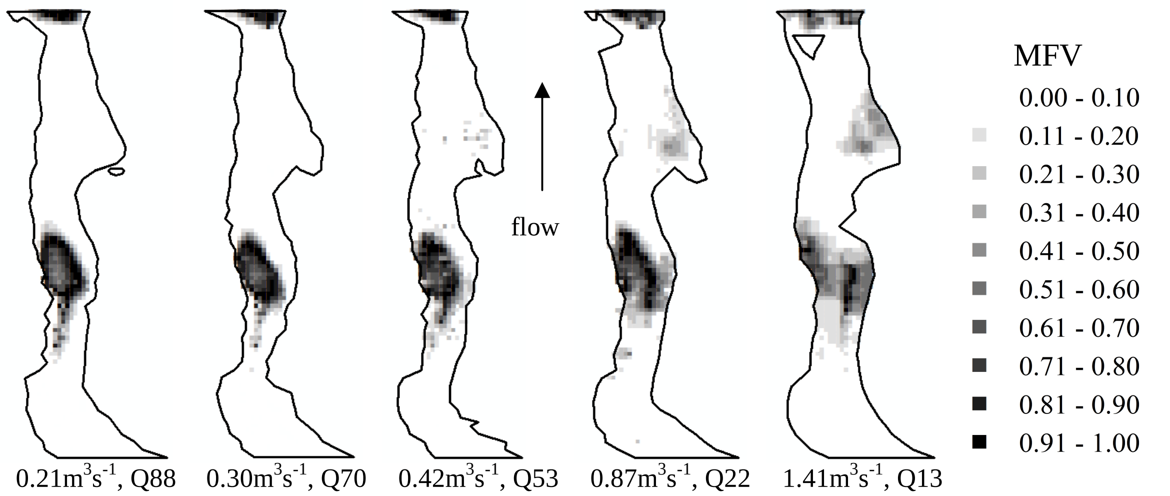


Figure 3.16. Spatial variation of membership function values to fuzzy cluster 5 in the 5-cluster Gustafson-Kessel classification, River Arrow.

The concentration of high class confusion in crisp, linear zones suggests depth and velocity data were strongly spatially correlated at the two lowest flows. Intermediate flow ($0.42ms^{-1}$, Q53) presented a very different picture, with widespread but low level class confusion throughout the reach. Class boundaries were diffuse and indistinct and within-cluster heterogeneity increased. Widespread class confusion continued at high flow although some crisp boundaries were evident around the deep, slow-flowing areas (fuzzy cluster 5). Within-cluster heterogeneity increased noticeably in fuzzy cluster 3 due to an increase in depth and decrease in velocity. It is interesting to note the increased class confusion at the core of deep, slow-flowing areas (fuzzy cluster 5) at all flows, suggesting that these areas (topographic lows) are associated with a high degree of hydraulic heterogeneity. At very high flow (1.41ms^{-1}, Q13), when discharge became the dominant influence on the hydraulic environment, class confusion was lowest in the fastest flowing areas (fuzzy cluster 4). By contrast, the deepest areas of the channel were associated with the most class confusion, probably due to the uncharacteristic increase in velocity in these areas.

Classification uncertainty can also be measured in terms of the proportion of entities that are not assigned to single clusters under different α-cut defuzzification thresholds (Figure 3.18). Figure 3.18 illustrates the relative stringency of using α-cut thresholds to defuzzify a classification of continuous data. For example, under a 0.7 α-cut, where entities must have ≥70% membership to a single cluster, only 48.2% entities would be assigned to clusters. At 0.8 α-cut this falls to 31.6%. The small proportion of the channel that can be assigned to clusters under a high level of classification certainty (i.e. a large α-cut threshold) is indicative of several factors. Firstly, the inherently fuzzy nature of the hydraulic environment means membership functions are likely to be split between classes. Secondly, as the number of classes in the classification increases and the membership function values are spread between more classes, the likelihood of achieving a high membership value to a single cluster decreases. It is unlikely for membership functions to reach 0.8 or more, except for those entities in the central core of each hydraulic patch type. In order to assign 70% of entities to single clusters in the 5-GK classification, it would be necessary to apply a relatively low α-cut threshold of 0.56.

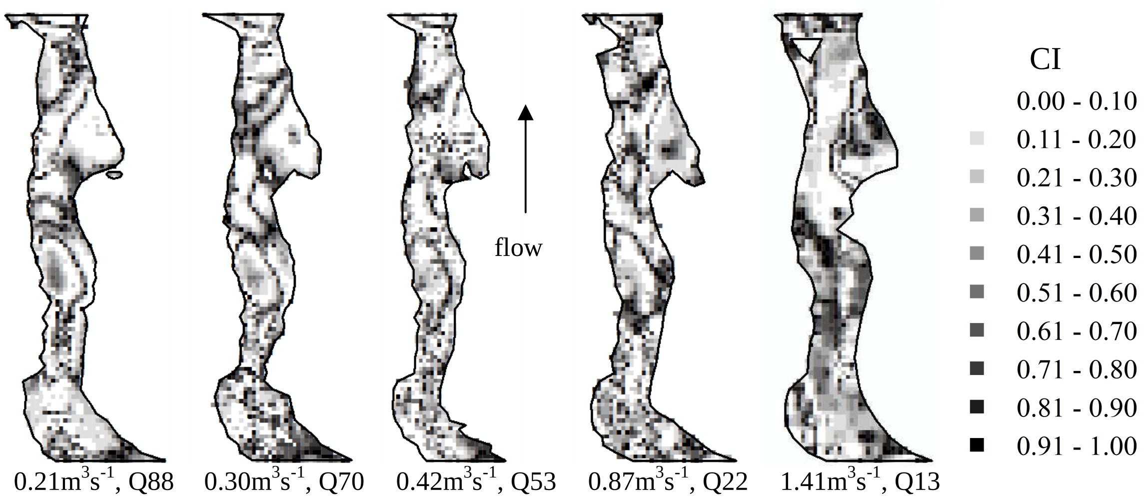


Figure 3.17. Spatial variation of the Confusion Index for the 5 fuzzy cluster Gustafson-Kessel classification (*m*=2).

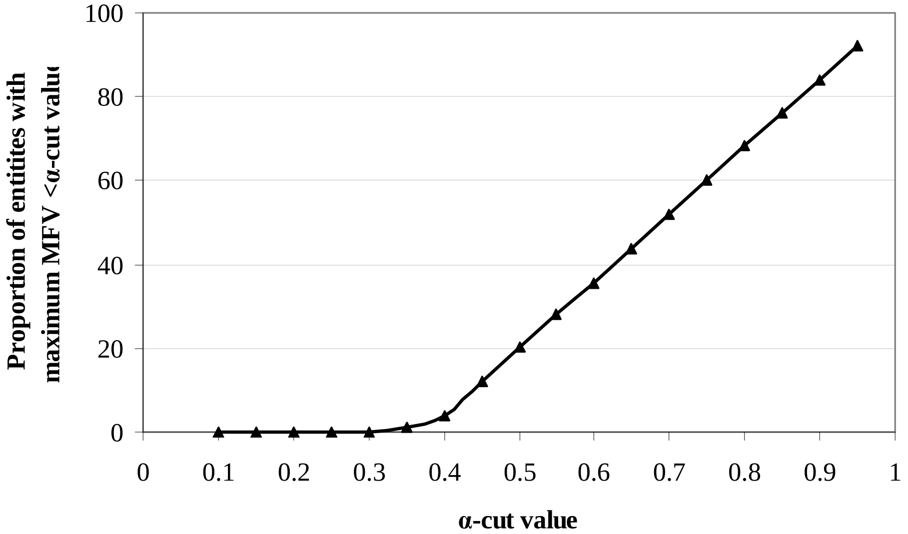


Figure 3.18. Proportion of entities assigned to the Transition Zone (TZ) under various α-cut thresholds.

Hydraulic patches and the transition zone

Fuzzy clusters from the optimal 5 Gustafson-Kessel classification were defuzzified using the combination of rules described in Section 3.3 to create five crisp hydraulic patches and a transition zone which represented areas of patch type overlap/patch type confusion (Figure 3.19). The median and spread of depth (m) and velocity (ms^{-1}) values in each hydraulic patch are shown in Figure 3.20. Figure 3.21 shows the change in location and extent of each hydraulic patch type with each increase in discharge. A summary of the hydraulic character of the patch types at each flow is shown in Table 3.3. RA1 was characterised by moderately deep and slow-flowing conditions and showed a gradual and steady increase in each variable as discharge increased. This patch type occupied the channel centreline between the two deep, slow-flowing areas at all but the highest flow, where its extent was significantly reduced and marginalised by RA4. RA2 delineated the shallow but moderately-fast conditions found at the topographic high points in the reach at all but the highest discharge when it too was replaced and marginalised by RA4. Both depth and velocity in RA4 showed a gradual increase with discharge. RA3 identified the zones of recirculating flow characterised by upstream eddies and moderately deep water. These occurred at the margins of deep, slow-flowing areas and in the backwater pool (Figure 3.11). Mean velocity in this patch type decreased slightly with discharge whilst depth increased at the two highest flows. The moderately deep and fast-flowing areas in the reach that appeared only at high discharges were delineated by RA4. This patch type occurred along the channel centreline, replacing RA1 and RA2. Mean depth increased with discharge although mean velocity remained relatively stable. The fifth hydraulic patch type occurred in the deepest areas of the channel and was characterised by deep, slow-flowing water. This patch type was very stable, both in terms of its location and hydraulic conditions at all but the highest flow, when it was bisected by the Transition Zone.

The Transition Zone incorporated the full range of depths and velocities sampled (Figures 3.18 and 3.19) and occupied between 18-28% of the reach (Figure 3.21). Boundaries between patch types where *either* depth *or* velocity were very different (e.g. between RA1 & RA2, RA3 & RA5 and RA1 & RA3) tended to occur as narrow bands. Wider, more diffuse boundaries occurred where the depth *and* velocity of adjacent patch types were both very different (e.g. RA2 & RA3 and RA4 & RA5). The topographic high point near the upstream extent of the reach had a very patchy distribution of transitional pixels, reflecting the variable conditions in this area. At the highest flow a very large transitional zone appeared where the thalweg bisected RA5. Here conditions were too fast-flowing to be classified as RA5 and too deep to be classified as RA4.

All patch types were associated with small degree of internal heterogeneity as indicated by the standard deviation (Table 3.3). In most cases within-patch heterogeneity increased marginally with discharge. The degree to which membership function values had been exaggerated (i.e. 1-MFV) to allocate fuzzy entities to a single hydraulic patch at each discharge is illustrated in Figure 3.21. The average exaggeration index across all hydraulic patches (but excluding the Transition Zone) increased slightly with discharge from 0.23-0.25. At the two lowest flows membership exaggeration was mainly limited to the outer edges of hydraulic patches with the exception of RA5 which contained more exaggeration in the centre. Membership exaggeration increased at intermediate and high flow and was more evenly spread throughout the extent of all hydraulic patch types. At very high flow membership exaggeration was noticeably lower in RA4 in the downstream extent of the reach. Both class confusion and membership exaggeration peak at intermediate to high flows (Q53 and Q22) when the hydraulic environment is equally influenced by channel topography and discharge.

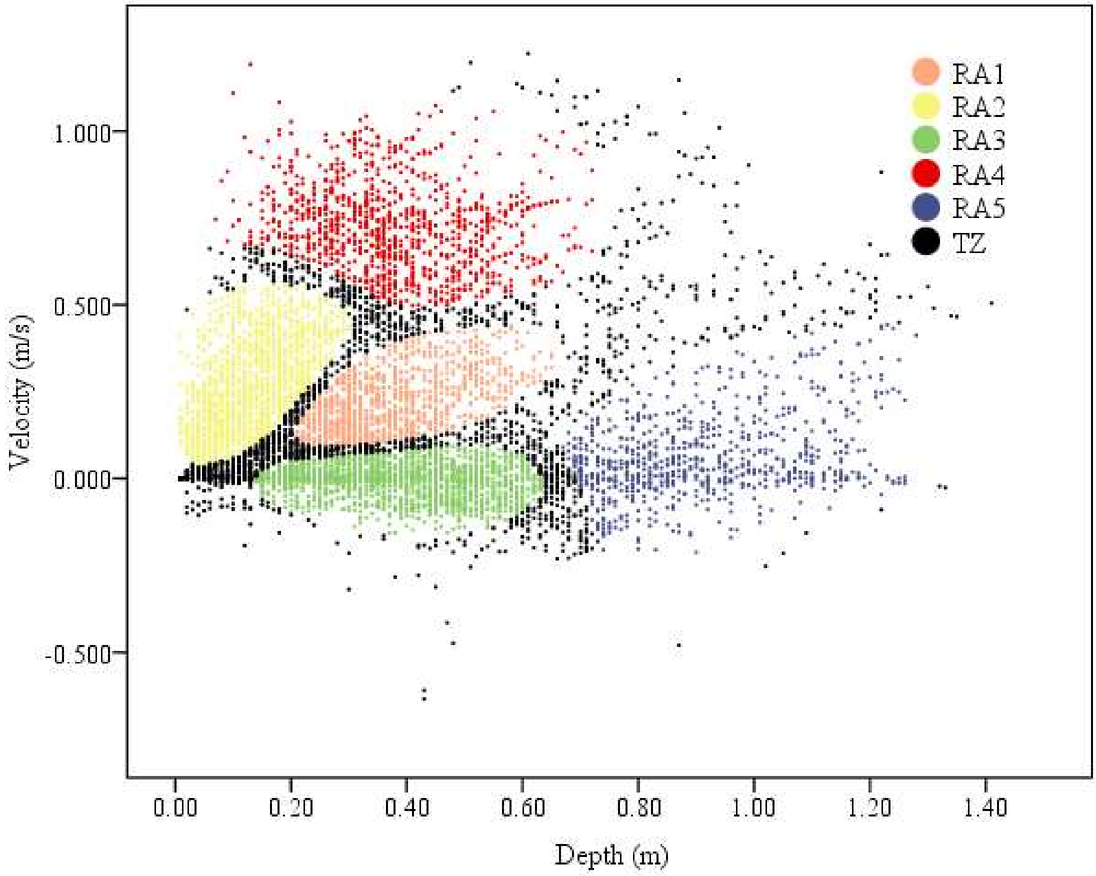


Figure 3.19. Scatterplot showing the bivariate distribution of all depth-velocity data collected at the River Arrow. Colours indicate hydraulic patch membership for the defuzzified 5-cluster Gustafson-Kessel classification of the data.

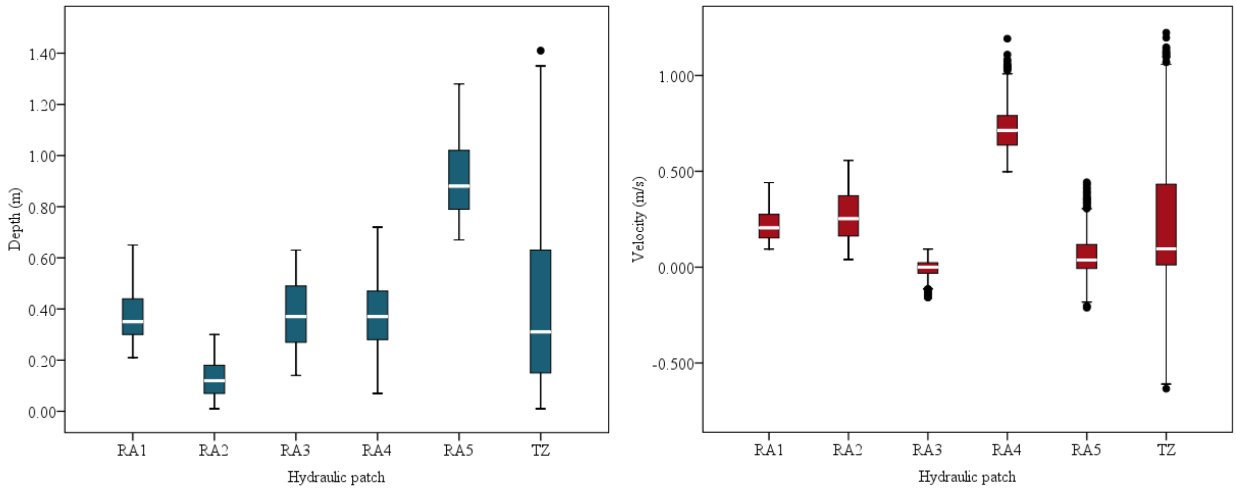


Figure 3.20. Boxplots showing the median and spread of depth (m) (left) and velocity (ms^{-1}) (right) in each hydraulic patch across all flows.

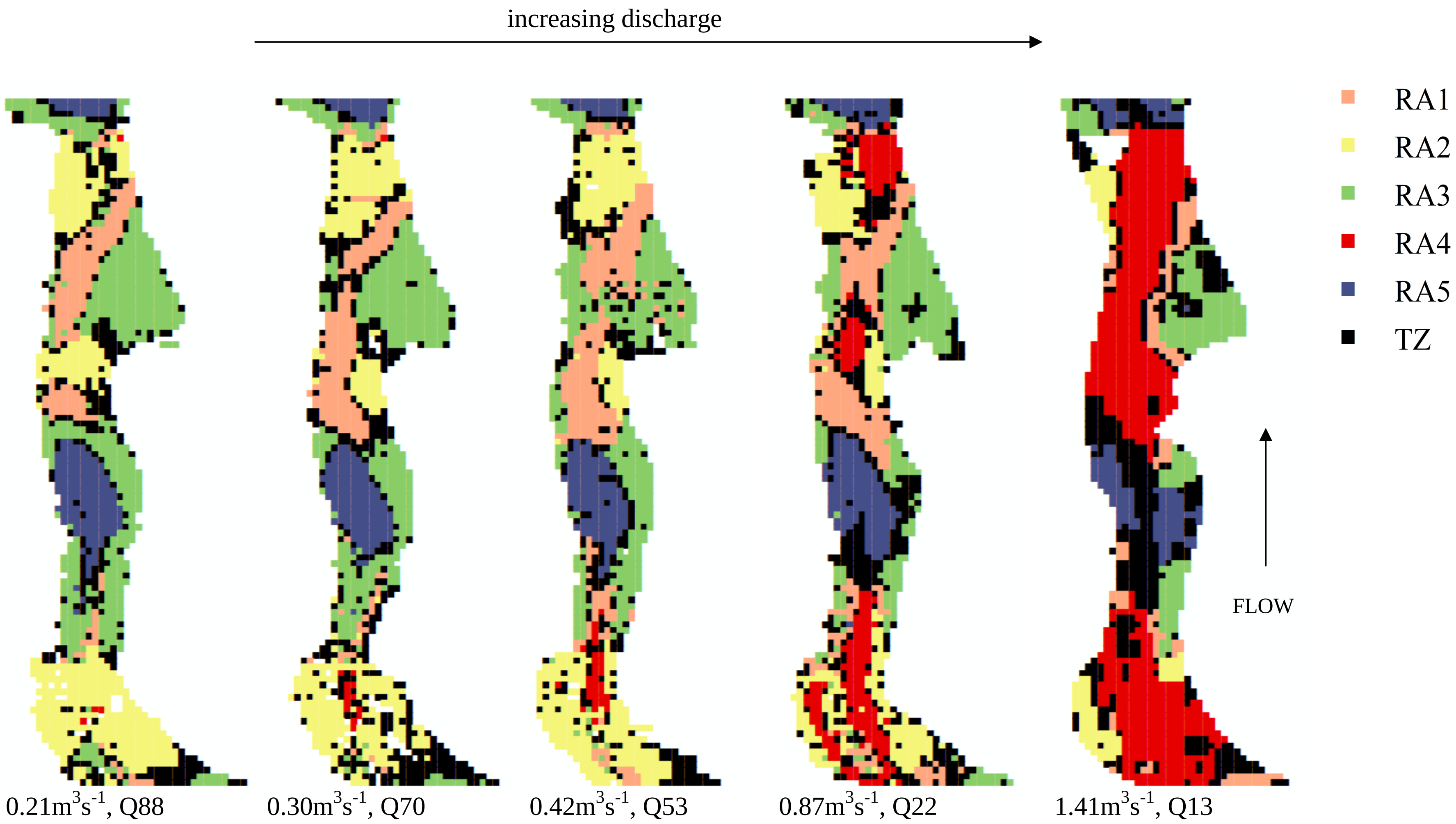
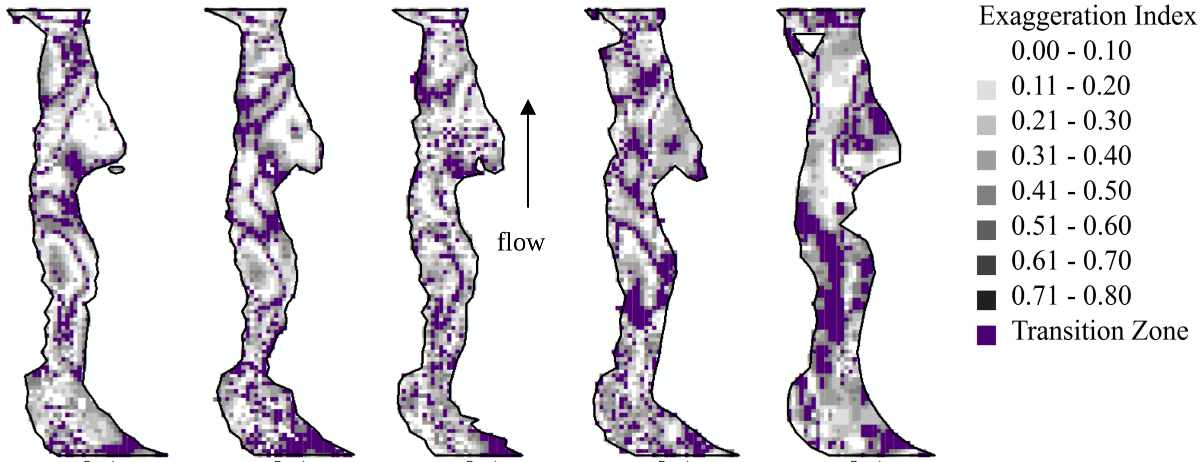


Figure 3.21. Location and extent of hydraulic patches (RA1-RA5) and transition zones (TZ) delineated at each flow at the River Arrow site.



$0.21m^3s^{-1}$, Q88 $0.30m^3s^{-1}$, Q70 $0.42m^3s^{-1}$, Q53 $0.87m^3s^{-1}$, Q22 $1.41m^3s^{-1}$, Q13

Figure 3.22. Spatial variation of the Exaggeration Index for the defuzzified 5-GK classification (*m*=2).

Table 3.3. Change in mean depth (m) (D_{mean}) and mean velocity (ms^{-1}) (V_{mean}) in each hydraulic patch with each increase in discharge. Standard deviation is shown in brackets.

Hydraulic Patch	Variable	Q88 ($0.21m^3s^{-1}$)	Q70 ($0.30m^3s^{-1}$)	Q53 ($0.42m^3s^{-1}$)	Q22 ($0.87m^3s^{-1}$)	Q13 ($1.41m^3s^{-1}$)
RA1	D_{mean}	0.30 (.06)	0.31 (.06)	0.37 (.09)	0.41 (.09)	0.47 (.09)
	V_{mean}	0.164 (.04)	0.168 (.05)	0.225 (.06)	0.257 (.09)	0.277 (.08)
RA2	D_{mean}	0.10 (.06)	0.11 (.06)	0.14 (.06)	0.17 (.07)	0.17 (.07)
	V_{mean}	0.211 (.09)	0.231 (.12)	0.300 (.13)	0.333 (.13)	0.368 (.11)
RA3	D_{mean}	0.34 (.12)	0.34 (.13)	0.37 (.12)	0.42 (.12)	0.50 (.09)
	V_{mean}	0.013 (.03)	0.002 (.03)	-0.007 (.05)	-0.040 (.05)	-0.014 (.06)
RA4	D_{mean}	0.14 (.06)	0.18 (.04)	0.23 (.12)	0.33 (.11)	0.41 (.12)
	V_{mean}	0.761 (.09)	0.765 (.10)	0.753 (.14)	0.685 (.11)	0.731 (.13)
RA5	D_{mean}	0.91 (.16)	0.90 (.15)	0.90 (.14)	0.91 (.14)	0.94 (.17)
	V_{mean}	0.029 (.04)	0.026 (.05)	0.065 (.10)	0.104 (.14)	0.083 (.18)

3.4.2 River Salwarpe hydraulic patch classification

Hydraulic survey data

Table 3.4 shows the maximum, minimum, mean, range and spread of depths and velocities recorded at each hydraulic survey. As expected, mean depth and velocity increased with discharge, as did the spread of values around the mean. Figure 3.24 illustrates the spatial variability of depth and velocity throughout the reach. In the most natural section of the reach (0-21m) a typical longitudinal sequence of relatively deep-slow followed by shallow-fast flow patterns was observed. However, in the most heavily modified section of the reach under the road bridge (22m-35m), and to a lesser degree in the downstream extent of the reach immediately below the bridge (36-45m), lateral variation dominated and depth and velocity appeared strongly positively correlated. The deepest areas of the reach were associated with local scour zones; one adjacent to a large concrete boulder on the left bank near the upstream extent, the other where the flow was directed towards the centre of the channel under the road bridge by baffles on the banks. The highest velocities occurred at the two most significant breaks of slope at 19-27m and 39-45m.

Table 3.4. Descriptive statistics of hydraulic data collected at each discharge, River Salwarpe.

Hydraulic parameter	$0.53m^3/s$, Q88	$0.79m^3/s$, Q67	$1.14\ m^3/s$, Q38	$1.63m^3/s$, Q21	$1.84m^3/s$, Q17
$Depth_{max}$	0.45	0.50	0.58	0.65	0.62
$Depth_{min}$	0.01	0.01	0.01	0.01	0.01
$Depth_{mean}$	0.14	0.17	0.19	0.25	0.27
$Depth_{variance}$	0.008	0.009	0.011	0.010	0.012
$Depth_{range}$	0.44	0.49	0.57	0.64	0.61
$Velocity_{max}$	1.140	1.299	1.252	1.346	1.999
$Velocity_{min}$	-0.082	-0.036	-0.083	-0.160	-0.107
$Velocity_{mean}$	0.316	0.371	0.511	0.608	0.736
$Velocity_{variance}$	0.039	0.045	0.048	0.064	0.090
$Velocity_{range}$	1.222	1.258	1.335	1.506	2.106

A scatterplot of hydraulic data at each individual flow and all flows combined illustrate the data distribution in attribute space (Figure 3.24). As the reach did not contain a pool or riffle the data range is considerably smaller than at the River Arrow and tends to show a positive correlation between depth and velocity indicative of predominantly lateral, rather than longitudinal, covariations (Figure 3.24f). At very low flow, the data is densely clustered at low depths and a small range of velocities. As discharge increases the spread of data increases and shifts to reflect deeper, faster conditions. At very high flow ($1.84m^3/s$, Q17), two distinct depth-velocity bands are evident in the bivariate distribution, suggesting that in different areas of the channel depth and velocity were positively or negatively correlated (Figure 3.24e).

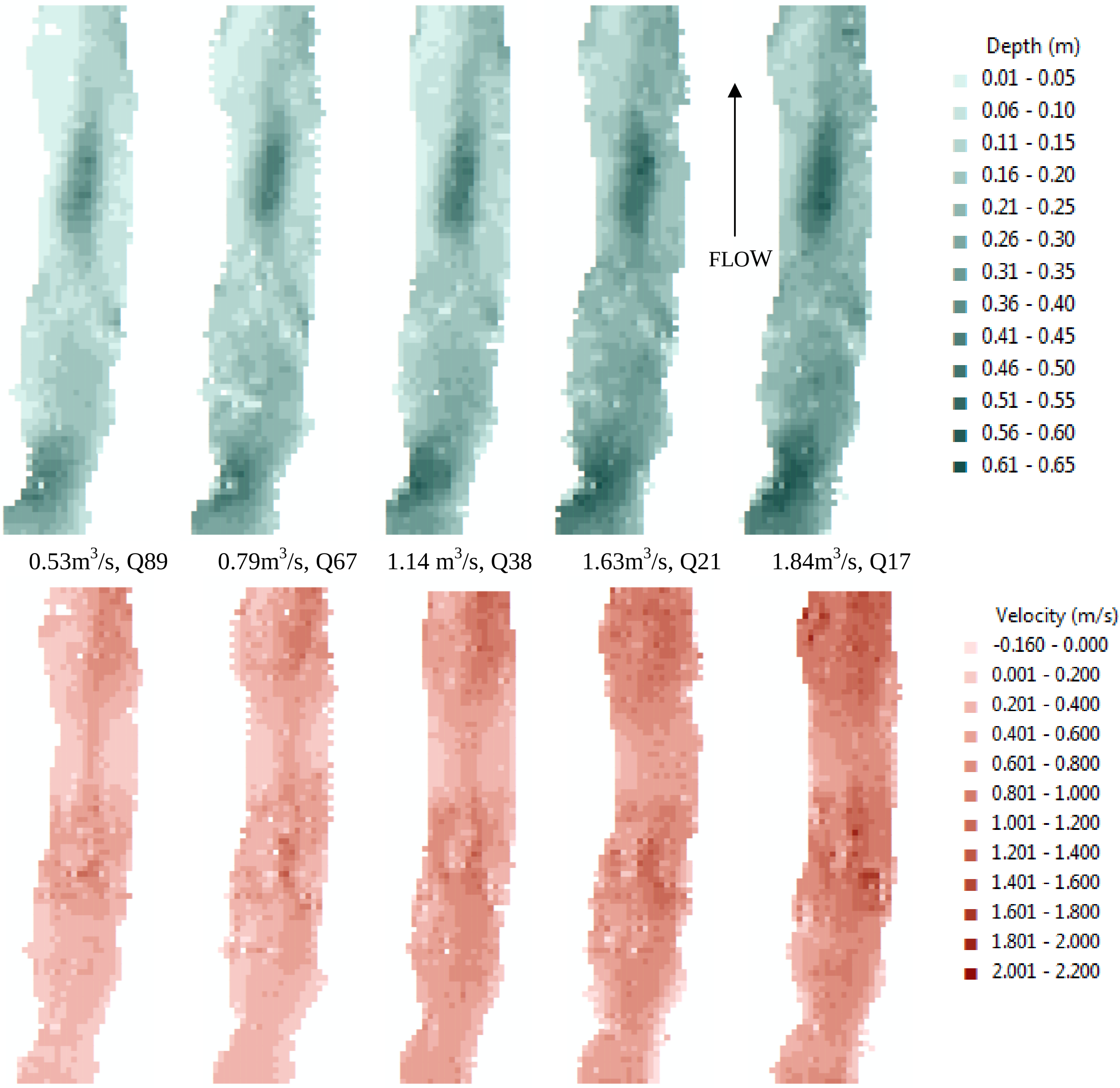


Figure 3.23. Variation in the mean column velocity (top) and depth (bottom) measured at each discharge throughout the reach, River Salwarpe.

Assessment of cluster tendency

The Hopkins statistic confirmed that the data were neither randomly or uniformly distributed (H=1). Inspection of the intensity plot (Figure 3.25) suggested local peaks in the depth-velocity distribution occurred at [~0.02m, ~0.15m/s}] [~0.08m, ~0.35m/s], [0.18m, ~0.4m/s], [0.25m, ~0.65m/s] and [~0.41m, ~0.5m/s]. However only the smallest of these was separated by a relatively low density region. As such there was no convincing evidence of a naturally clustered structure in the data from all discharges

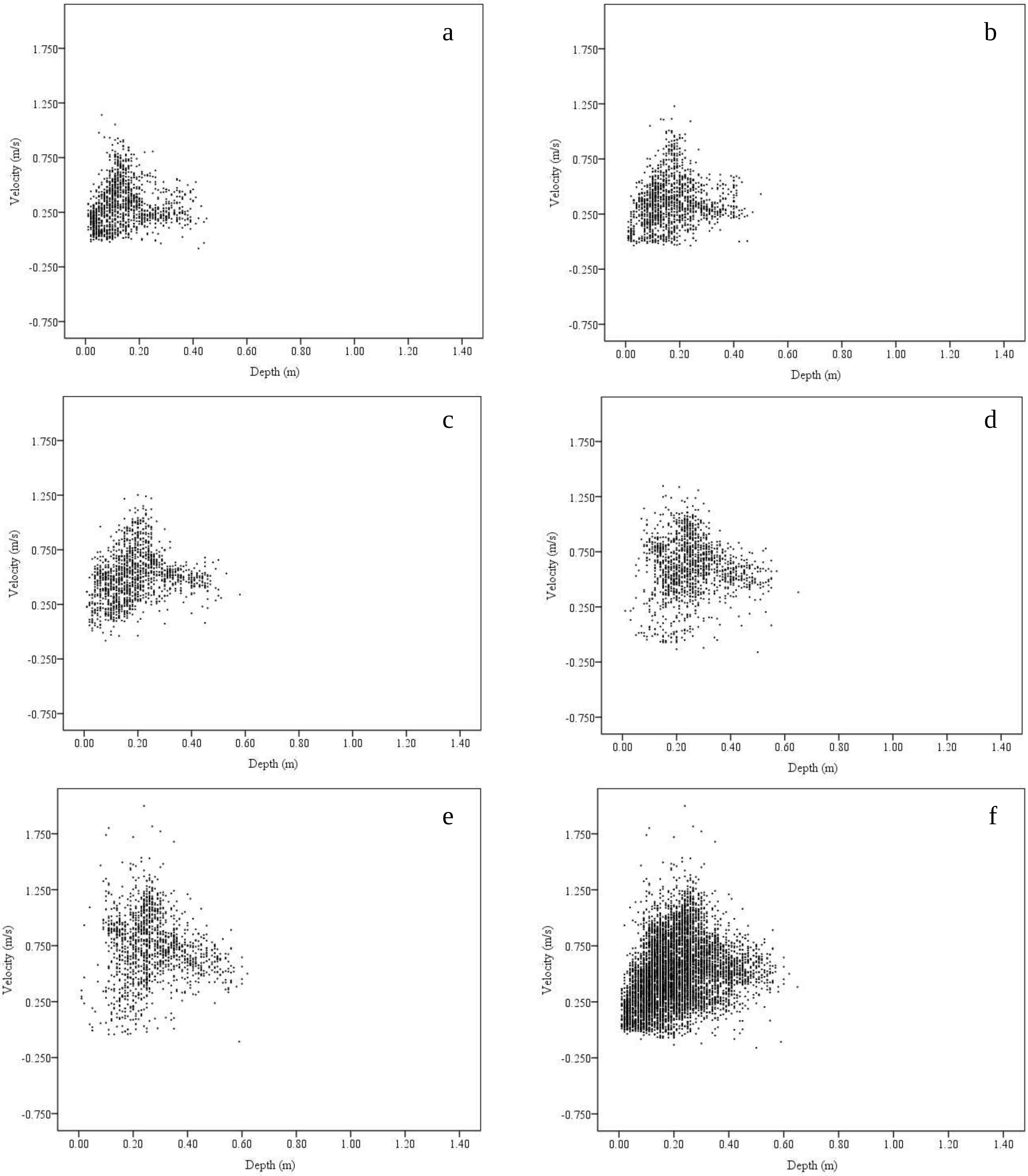


Figure 3.24. Bivariate scatterplots of River Salwarpe hydraulic survey data at (a) very low flow, (b) low flow, (c) moderate flow, (d) high flow, (e) very high flow and (f) combined flows.

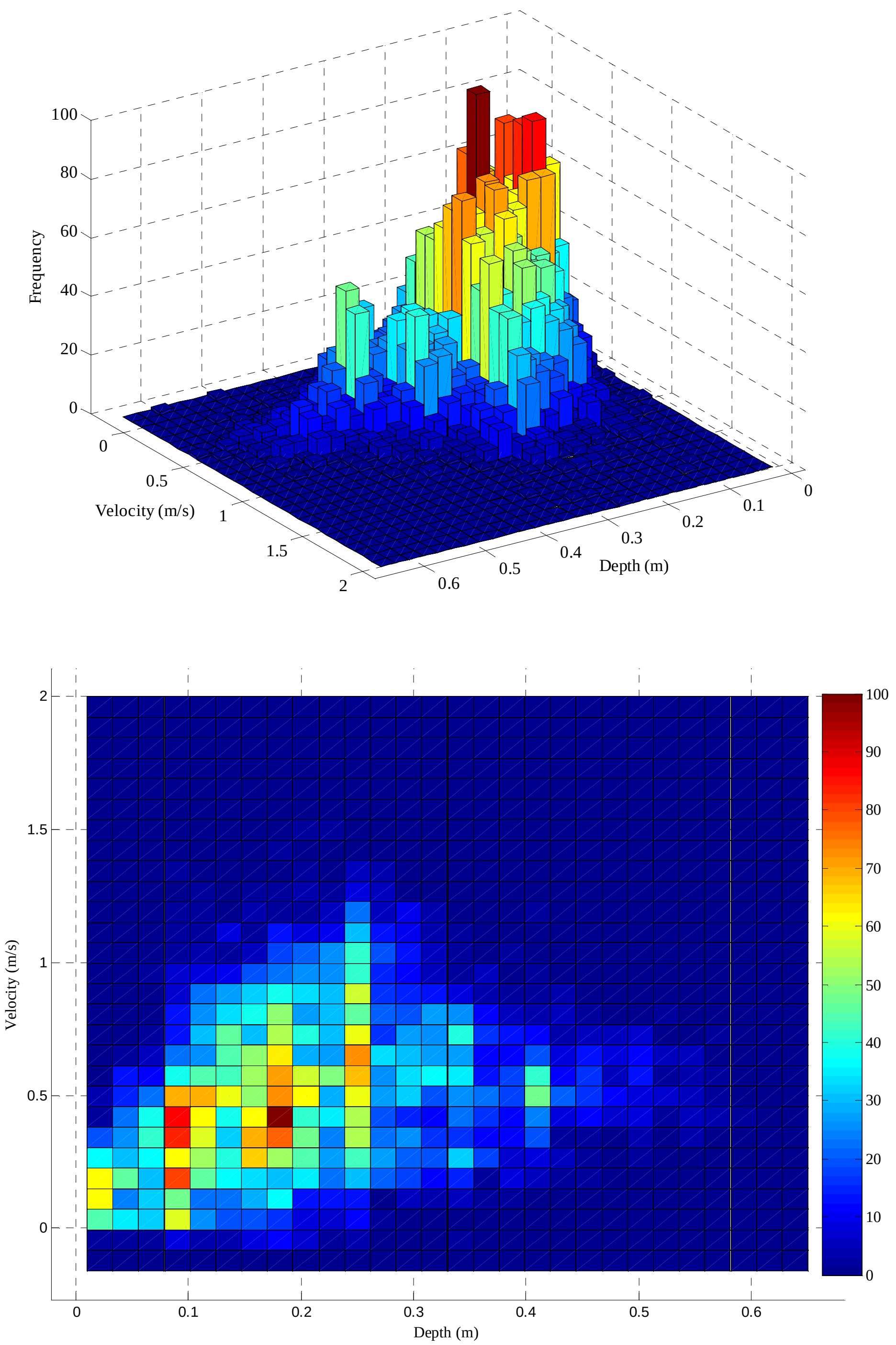


Figure 3.25. 2D histogram (top) and intensity plot (bottom) showing the bivariate distribution and frequency of all depth-velocity measurements collected at the River Salwarpe.

combined. The 2D histogram suggested the distribution was unimodal (Figure 3.25). Although the distribution of data combined from all flows was too densely populated to clearly show any natural fuzzy clusters, inspection of the data distribution at individual flows (Figure 3.26) did reveal some natural fuzzy clusters. At very low flow (Figure 3.26 a) four natural fuzzy clusters were evident; a small L-shaped cluster near the origin, a long vertical cluster of points with depths less than 0.2m and velocities greater than $0.25ms^{-1}$, a small spherical cluster centred around 0.3 ms^{-1} and 0.2m, and a horizontal cluster of points with depths greater than 0.2m and velocities less than 0.5m/s (Figure 3.26). At moderate flows the data distribution was more continuous with no clear evidence of high density areas. At high flows two different high density areas were evident in the fastest part of the data distribution; as indicated in Figure 3.26.

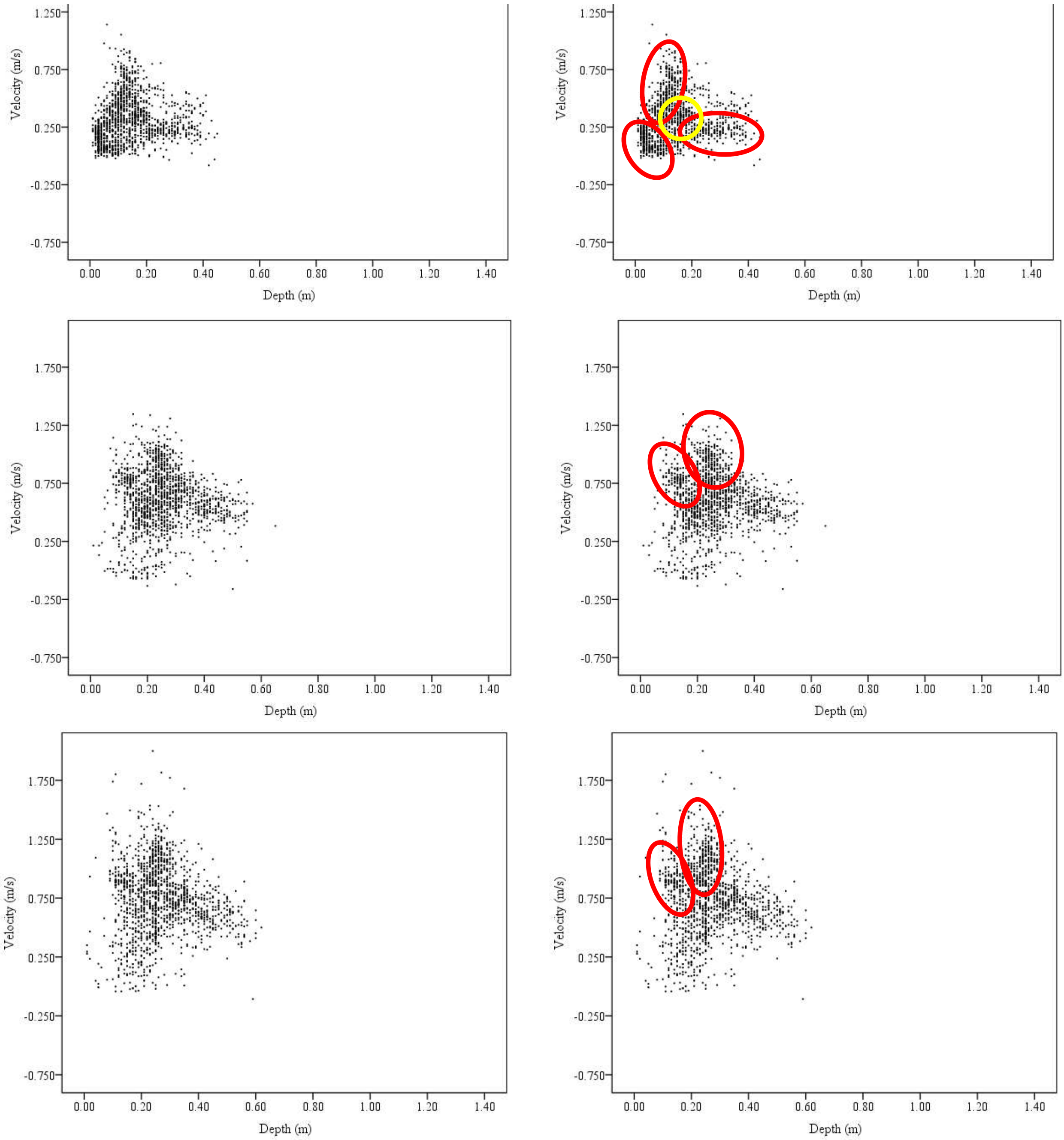


Figure 3.26. The hydraulic data distribution (left) at very low flow (top row), high flow (middle row) and very high flow (bottom row) at the River Salwarpe with markers identifying regions of high density (right)

Selecting the optimal classification

As the greater flexibility of the Gustasfon-Kessel algorithm proved most suitable for delineating hydraulic patches at the River Arrow, classifications of the River Salwarpe data generated with this algorithm were examined first. The full results of partitioning the data for 2≤c≤8 using both Gustafson-Kessel and fuzzy c-means algorithms are shown in in Appendix C and will be referred to throughout this section. As the data combined from all discharges did not exhibit a natural clustering tendency the subjective decision-making process for selecting the optimal classification at the River Arrow data was also applied here. That is, the location of mapped clusters was assessed in relation to topographic (Figure 3.27) and geomorphic features (Figure 3.28) affecting local flow patterns. The ability of the classification to reflect discharge-dependent variations in depth and velocity and natural fuzzy clusters evident in the data distribution from individual discharges (Figure 3.26) was also taken into account, as was the spatial coherence of mapped clusters.

The 5 G-K partition (Appendix C, Table 11, Figures 21-22) provided the optimal classification of hydraulic patches, identifying both lateral and longitudinal hydraulic variation at all flows. For example, lateral variation in the modified section of the reach under the road bridge where depth and velocity were positively correlated was clearly delineated by patch types 1 (moderately deep, slow), 3 (shallow, slow) and 5 (deep-moderately fast). The increase in the extent of deep flow (patch type 5) associated with the scour zone at higher discharges was also adequately reflected. The classification also revealed lateral variation in the downstream extent of the reach (36-45m) between the shallow-slow flow associated with the gravel deposition zone (Figure 3.28) and the deeper, faster flow of the channel thalweg towards the right bank. The increase in depth and velocity in both these zones as discharge increased was amply demonstrated by the transition between patch types 3 to 2 and 2 to 4. Importantly the classification preserved the lateral hydraulic variation in these two zones at all flows. Longitudinal hydraulic variations at low flows were reflected by the sequence of patch types 5 (deep-moderate), 1 (moderate-slow) and 2 (shallow-fast). The tendency towards hydraulic homogenisation as discharge increased was shown by the expansion and dominance of patch types 4 (moderate-fast) and 5 (deep-moderate). The advantage of the five cluster classification over the four cluster classification was the improved differentiation of shallow-fast and moderate-fast flow which appeared as natural fuzzy clusters at high flows. The five cluster classification using the fuzzy c-means algorithm (Appendix C, p.4) was similar, but did not provide the same level of differentiation between shallow-fast and moderate-fast conditions. The addition of a sixth cluster provided some further differentiation between moderately fast flow around the exposed boulders and faster flow at the break in slope at low flows, but decreased the spatial coherence of hydraulic patches at moderate-very high flows and so was not judged to be a useful addition to the classification.

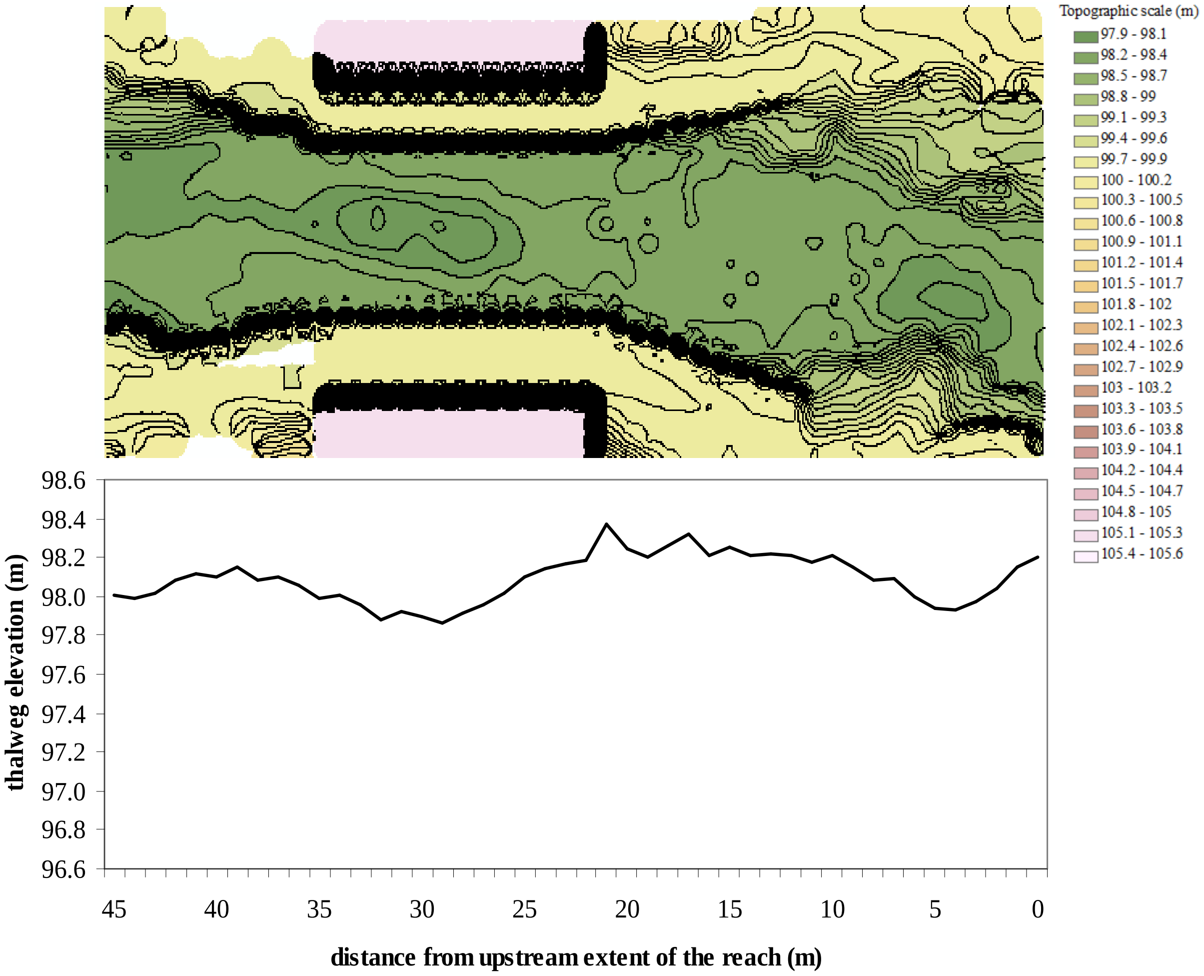


Figure 3.27. Channel topography (top) and variation of the thalweg elevation (bottom) in the River Salwarpe study reach.

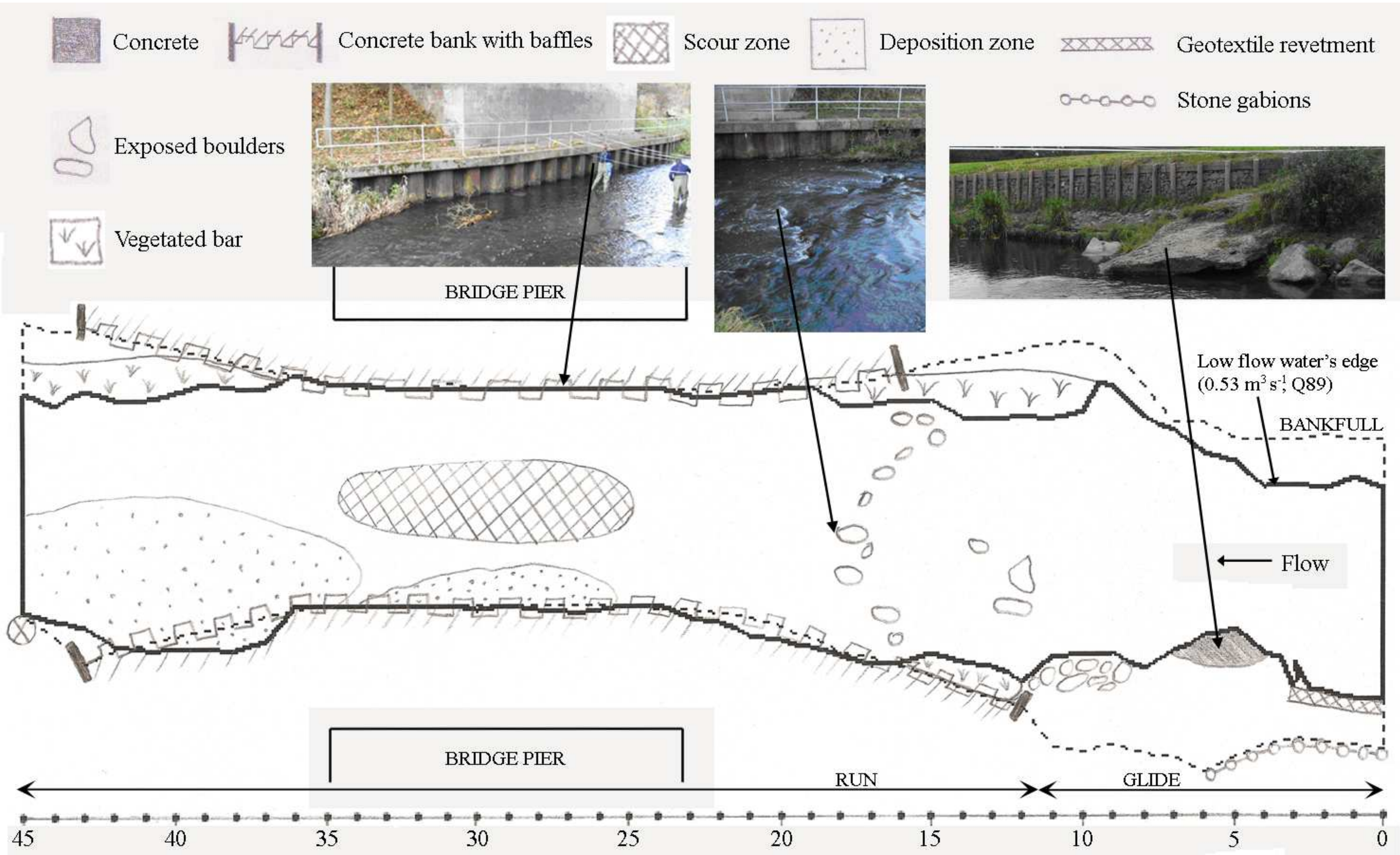


Figure 3.28. Annotated field sketch of the main geomorphic features in the River Salwarpe study reach. The extent of channel geomorphic units identified during a bankside rapid habitat assessment are also shown.

Fuzzy cluster membership distributions and classification uncertainty

Table 3.5 shows the hydraulic characteristics of each cluster centroid (prototype) in the 5 GK classification and confirms that each has distinct depth-velocity characteristics. The location and extent of the five fuzzy clusters at each discharge were illustrated by mapping the spatial distribution of membership function values for each cluster (Figures 3.29 to 3.33).

Table 3.5. Cluster centroids for the 5-cluster Gustafson-Kessel hydraulic patch classification, River Salwarpe

Cluster	Hydraulic description (depth-velocity)	Depth (m)	Velocity (ms^{-1})
1	Moderate-slow	0.22	0.319
2	Shallow-fast	0.14	0.672
3	Shallow-slow	0.08	0.226
4	Moderate-fast	0.26	0.863
5	Moderately deep-moderate	0.40	0.534

High membership values to fuzzy cluster 1 (moderate-slow) were predominantly located in the upstream third of the reach at low flows where the channel slope was marginal and the bed topography flat and uniform. An additional narrow linear band occurred between the zones of scour and marginal deposition in the central modified section of the reach. As discharge increased the location shifted towards the channel margins and the average MFV to fuzzy cluster 1 decreased from 0.20 at moderate flow to 0.16 at very high flow. Large areas of the reach had moderate MFVs to this cluster, indicating a high degree of class overlap with clusters 2 and 4. High membership function values to fuzzy cluster 2 (shallow-fast) were located at exposed boulders (~15-18m) and the two main breaks in slope (~20-25m, ~39-45m) at low flows. At moderate to high flows these conditions moved to the large deposition zone in the downstream extent of the reach and became less, frequent, patchy and marginalised at the break of slope immediately upstream of the bridge. Moderate MFVs occurred immediately upstream and adjacent to high MFVs indicating class overlap with fuzzy cluster 4 (moderate-fast). Fuzzy cluster 3 (shallow-slow) occurred as large lateral bands in areas of deposition and shallow-slow flow at channel margins at low flow. The bands shrank as discharge increased and very few high membership functions values occurred at high flows. High MFVs to fuzzy cluster 4 appeared at moderate flow but occurred predominantly at high flows, replacing fuzzy cluster 2 when its location shifted. Both the width and length of the high MFV extent increased with discharge. Fuzzy cluster 5 was present at all flows and showed strong geographic zoning. High MFVs occurred in the two zones of scour. The location of high MFVs to fuzzy cluster 5 remained stable at all flows however the width and length of both areas increased with discharge. A degree of class overlap with fuzzy cluster 4 (moderate-fast) was evident.

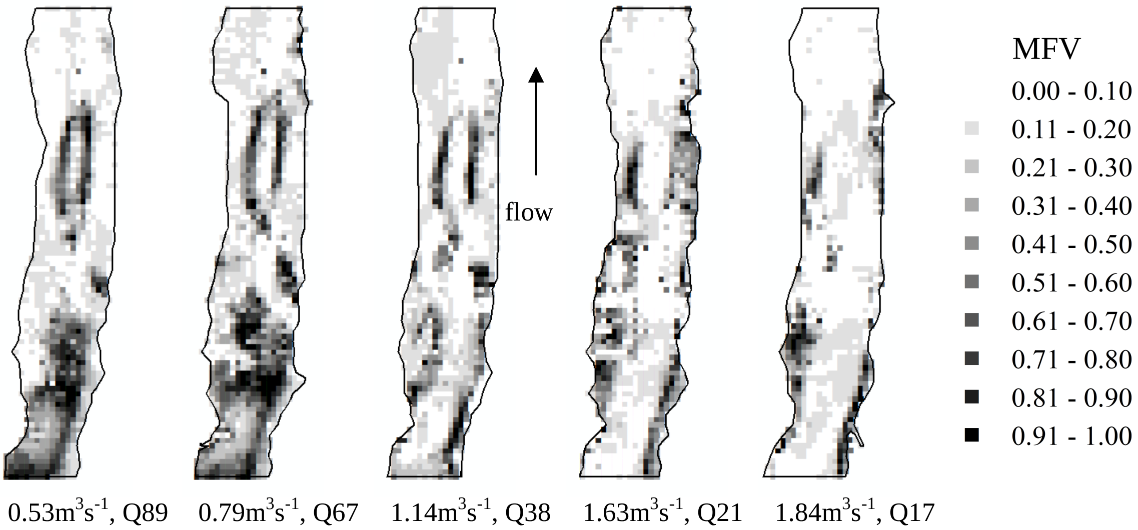


Figure 3.29. Spatial variation of membership function values to fuzzy cluster 1 in the 5-cluster Gustafson-Kessel classification, River Arrow.

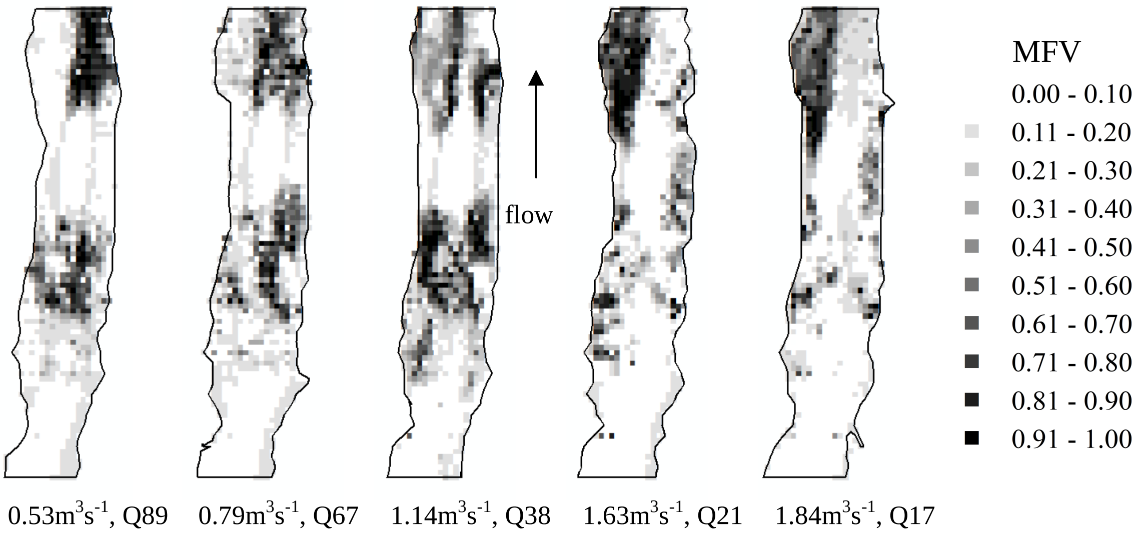


Figure 3.30. Spatial variation of membership function values to fuzzy cluster 2 in the 5-cluster Gustafson-Kessel classification, River Salwarpe.

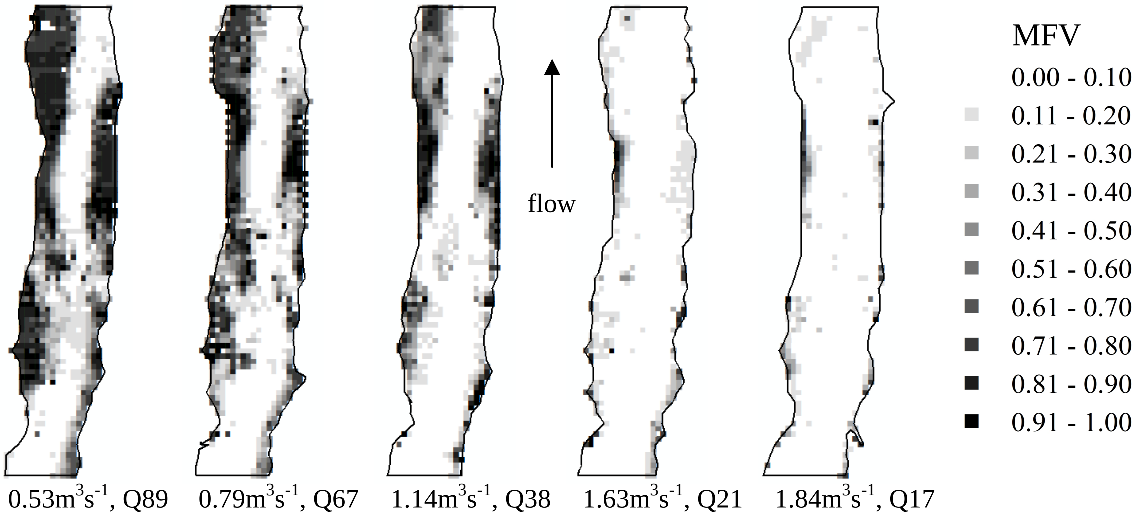


Figure 3.31. Spatial variation of membership function values to fuzzy cluster 3 in the 5-cluster Gustafson-Kessel classification, River Salwarpe.

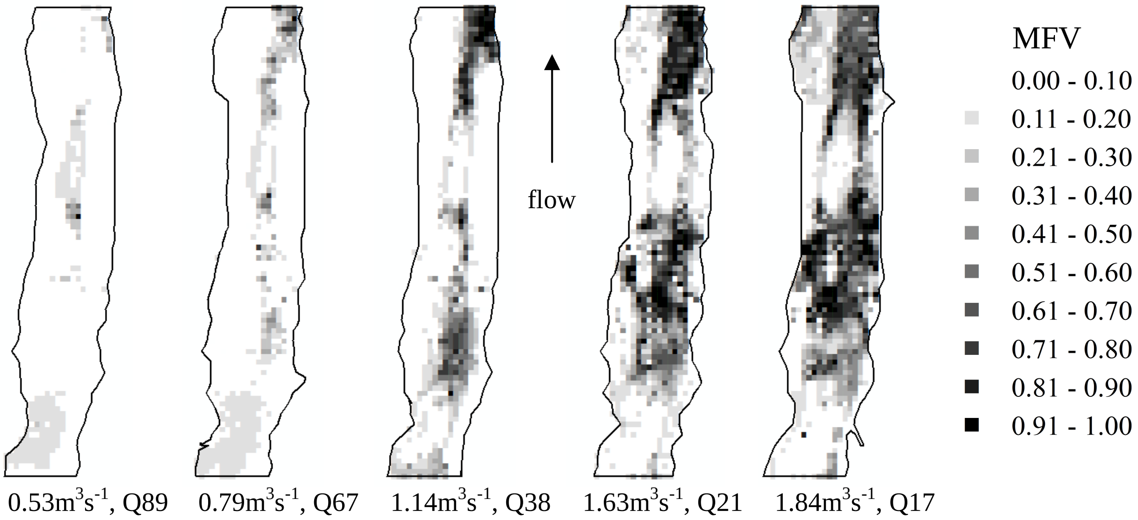


Figure 3.32. Spatial variation of membership function values to fuzzy cluster 4 in the 5-cluster Gustafson-Kessel classification, River Salwarpe.

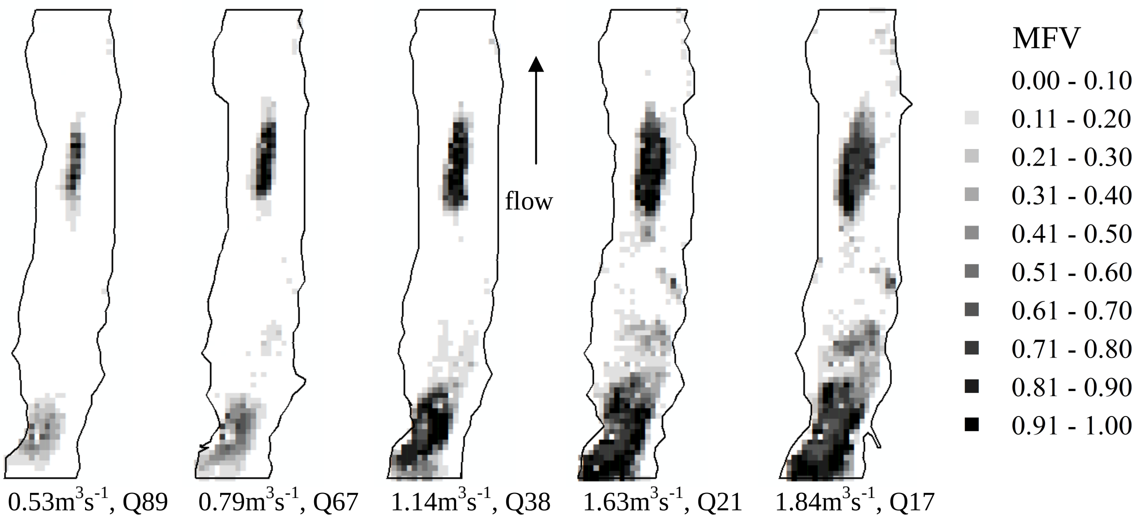


Figure 3.33. Spatial variation of membership function values to fuzzy cluster 5 in the 5-cluster Gustafson-Kessel classification, River Salwarpe.

Spatial variation of the Confusion Index was mapped in geographic space at each flow to illustrate where zones of confusion (class overlap) occurred and to what extent (Figure 3.34). Zones of confusion were widespread at every flow indicating a high degree of class overlap. The least class confusion occurred at very low and high flow when the data distribution was concentrated at or near cluster centroids. For example at very low flow a high density of data fell with in the bounds of clusters 1 and to a lesser extent clusters 2 and 3 hence these areas were least confused. Similarly at high flow depth-velocity measurements were concentrated near the centroids of clusters 2, 4 and 5 so these areas were associated with very little class confusion. At low, moderate and very high flow, more depth-velocity measurements were further from a cluster centroid and had characteristics of more than one cluster. Consequently there was a high degree of class overlap throughout the reach, however the area between 10-25m had consistently high class confusion at every discharge. With the exception of distinct zones of class confusion in the scour zone in the upstream extent of the reach at very low flow and the deposition zone at the downstream extent of the reach at moderate flow, class confusion was spatially noisy.

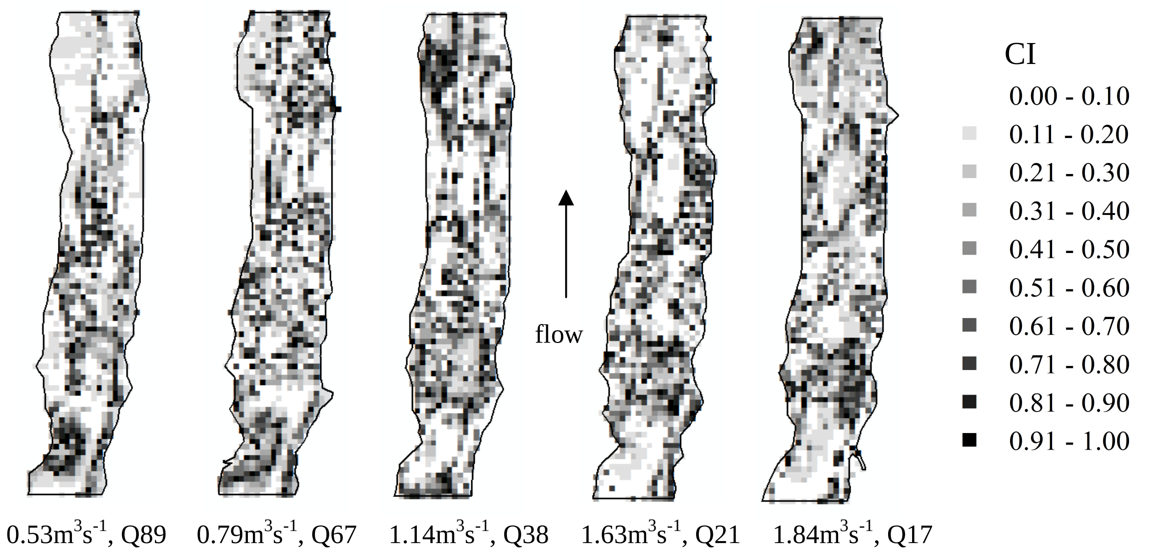


Figure 3.34. Spatial variation of the Confusion Index for the 5 fuzzy cluster Gustafson-Kessel classification (*m*=2), River Salwarpe.

Hydraulic patches and the transition zone

Figure 3.35 shows the extent of the five hydraulic patches and the transition zone in attribute space following defuzzification. The median and spread of depth and velocity values in each hydraulic patch type are illustrated in Figure 3.36. The change in location and extent of each hydraulic patch with every increase in discharge is shown in Figure 3.37 and a summary of their hydraulic character is provided in Table 3.6. RS1 was characterised by moderately deep and slow-flowing conditions. Mean depth remained relatively stable at all flows whereas mean velocity increased steadily with every increase in discharge, peaking at high flow and decreasing slightly at very high flow. This patch type occupied the upstream third of the reach, with the exception of the scour zone, and the margins of the scour zone in the central part of the reach at low flows. The extent decreased markedly as discharge increased and was constrained to the channel margins. RS2 identified the shallow, fast-flowing conditions found near the two main breaks of slope at low flows. In the central part of the reach the extent of RS2 was significantly reduced at high flows, being replaced by RS4. In the downstream extent of the reach RS2 was also replaced by RS4 however its location shifted laterally to the zone of deposition, previously occupied by RS3. Mean depth remained stable as discharge increased but mean velocity showed a steady increase. RA3 identified the shallow, slow-flowing conditions found near channel margins and deposition zones at low flow. The patch type was highly discharge-dependent, the extent falling from 41% at very low flow to 2% at very high flow. Mean depth remained stable at all flows however mean velocity peaked at moderate flow. RS4 delineated the moderately deep, fast flowing conditions that followed the channel thalweg in all but the deepest areas of the channel at moderate to very high flows. This patch type was also highly discharge dependent and occupied less than 1% of the reach at very low flow. Mean velocity showed a gradual increase with every increase in discharge whereas mean depth only increased at higher flows. RS5 identified the moderately deep, moderately fast conditions found in the two scour zones; one in the centre of the channel where flow had been deflected and the other adjacent to a concrete boulder in the upstream extent of the reach. Both mean depth and mean velocity increased steadily as discharge increased. The extent of this patch type increased markedly in the upstream extent of the reach but also to a small degree in the centre of the reach, in each case replacing RS1.

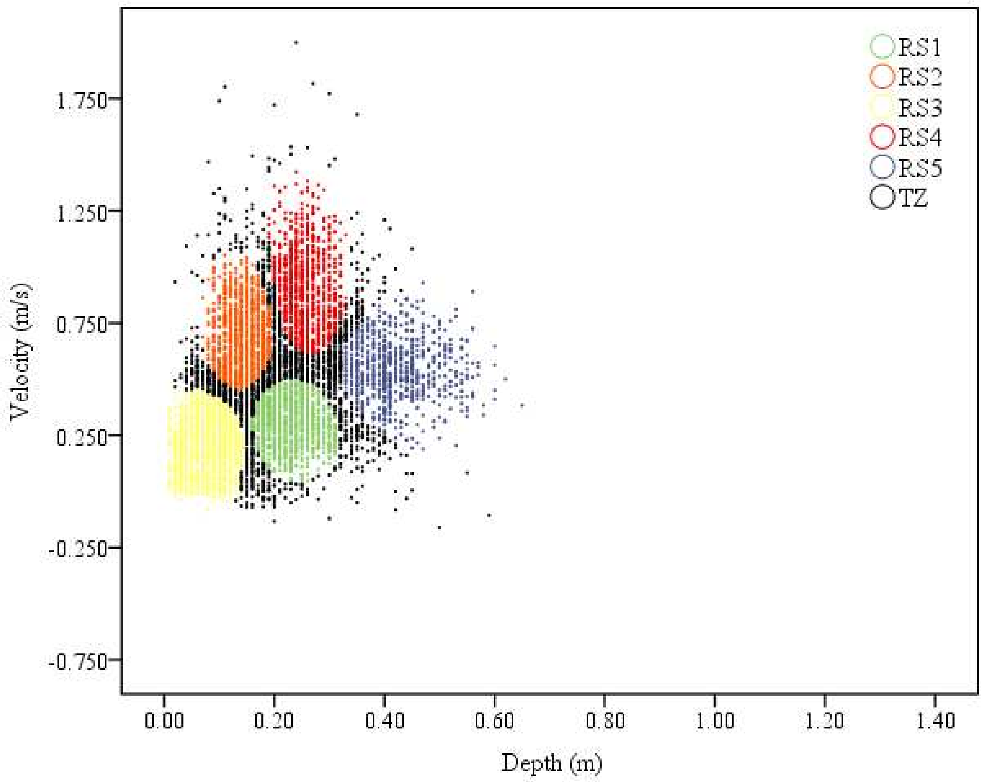


Figure 3.35. Scatterplot showing the bivariate distribution of all depth-velocity data collected at the River Salwarpe. Colours indicate hydraulic patch membership for the defuzzified 5 cluster Gustafson-Kessel classification of the data.

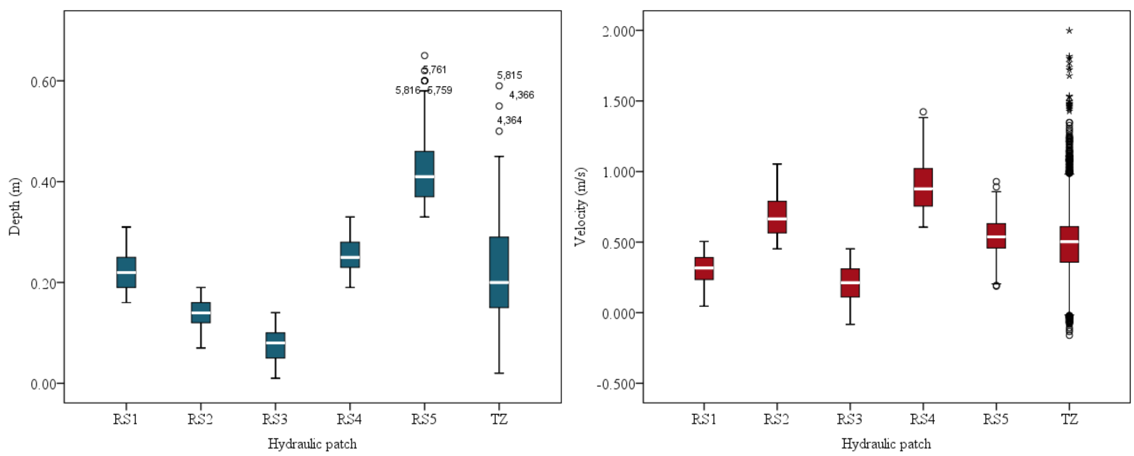


Figure 3.36. Boxplots showing the median and spread of depth (m) (left) and velocity (ms^{-1}) (right) in each hydraulic patch across all flows.

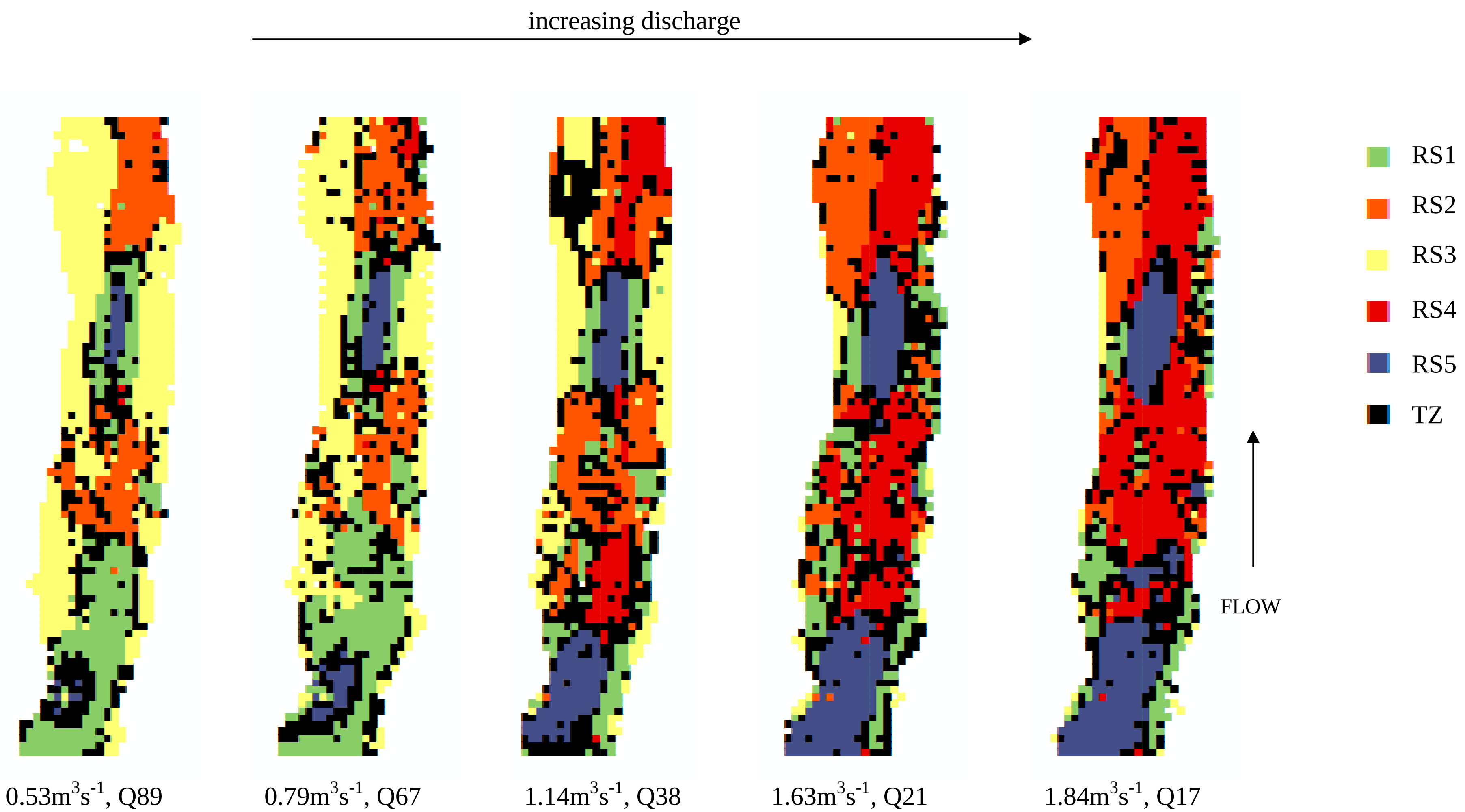


Figure 3.37. Location and extent of hydraulic patches (RS1-RS5) and transition zones (TZ) delineated at each flow at the River Salwarpe site.

The Transition Zone incorporated the full range of velocities and the vast majority of depths sampled (Figures 3.33 and 3.34). It occupied between 21-30% of the reach, peaking at moderate flow. Boundaries between patch types with a small difference in *either* depth *or* velocity (e.g. between RS1 & RS3) occurred as narrow bands. The greater the difference, the wider the transition zone (e.g. between RS1 and RS4). Where the depth *and* velocity of adjacent patch types were both very different (e.g. RS2 & RS3, RS1 & RS2, RS5 and all other patch types), the transition zone formed larger patches. These typically occurred at the boundaries between the longitudinal sequence of patch types (e.g. between RS1 and RS2). The scour zone in the upstream extent of the reach had a high proportion of transitional pixels at very low flow, indicating that it had characteristics of RS1 and RS5 at this discharge. At moderate-high flow deposition zones became transitional as depth and velocity increased. At very high flow the main transitional zones were located between patches of RS4 and RS5. At all flows the section of the reach between 8-25m contained a high proportion of scattered transitional pixels, suggesting it contained variable hydraulic conditions characteristic of several hydraulic patch types.

All patch types were associated with a small degree of internal heterogeneity as indicated by the standard deviation of depth and velocity in Table 3.6. Velocity was more heterogeneous than depth and tended to show greater variation with discharge, typically increasing as discharge rose. RS5 had the most discharge-dependent characteristics, showing a gradual increase in mean depth and mean velocity with every increase in discharge. RS1 and RS3 had stable mean depth across all flows but mean velocity peaked at moderate flow. Whereas RS2 and RS4 had stable depth but showed a gradual increase in velocity as discharge increased.

The degree to which membership function values had been exaggerated in the crisped classification at each discharge is shown in Figure 3.36. The average Exaggeration Index varied between 0.24-0.27 but did not show a clear trend with discharge. As expected, pixels immediately adjacent to a Transition Zone typically had the highest Exaggeration Index. With the exception of RS5 at moderate–very high flow, MFVs had been exaggerated throughout the reach, indicating the high degree of class confusion and the narrow range of depths and velocities associated with hydraulic patches at this site.

Table 3.6. Change in mean depth (m) (D_{mean}) and mean velocity (ms^{-1}) (V_{mean}) in each hydraulic patch with each increase in discharge. Standard deviation is shown in brackets.

Hydraulic Patch	Variable	Q89 ($0.53m^3s^{-1}$)	Q67 ($0.79m^3s^{-1}$)	Q38 ($1.14m^3s^{-1}$)	Q21 ($1.63m^3s^{-1}$)	Q17 ($1.84m^3s^{-1}$)
RS1	D_{mean}	0.23 (.04)	0.23 (.04)	0.21 (.04)	0.23 (.04)	0.22 (.03)
	V_{mean}	0.268 (.09)	0.306 (.10)	0.344 (.08)	0.349 (.11)	0.334 (.11)
RS2	D_{mean}	0.13 (.02)	0.14 (.03)	0.15 (.03)	0.14 (.03)	0.15 (.03)
	V_{mean}	0.627 (.12)	0.640 (.13)	0.661 (.13)	0.707 (.13)	0.793 (.14)
RS3	D_{mean}	0.07 (.03)	0.08 (.03)	0.08 (.03)	0.10 (.03)	0.09 (.04)
	V_{mean}	0.191 (.12)	0.215 (.12)	0.252 (.11)	0.180 (.14)	0.199 (.13)
RS4	D_{mean}	0.24 (.02)	0.23 (.02)	0.24 (.03)	0.26 (.03)	0.26 (.03)
	V_{mean}	0.746 (.10)	0.823 (.14)	0.819 (.16)	0.856 (.14)	0.957 (.18)
RS5	D_{mean}	0.38 (.03)	0.39 (.03)	0.40 (.05)	0.42 (.06)	0.44 (.09)
	V_{mean}	0.417 (.08)	0.413 (.11)	0.484 (.08)	0.529 (.27)	0.612 (.12)

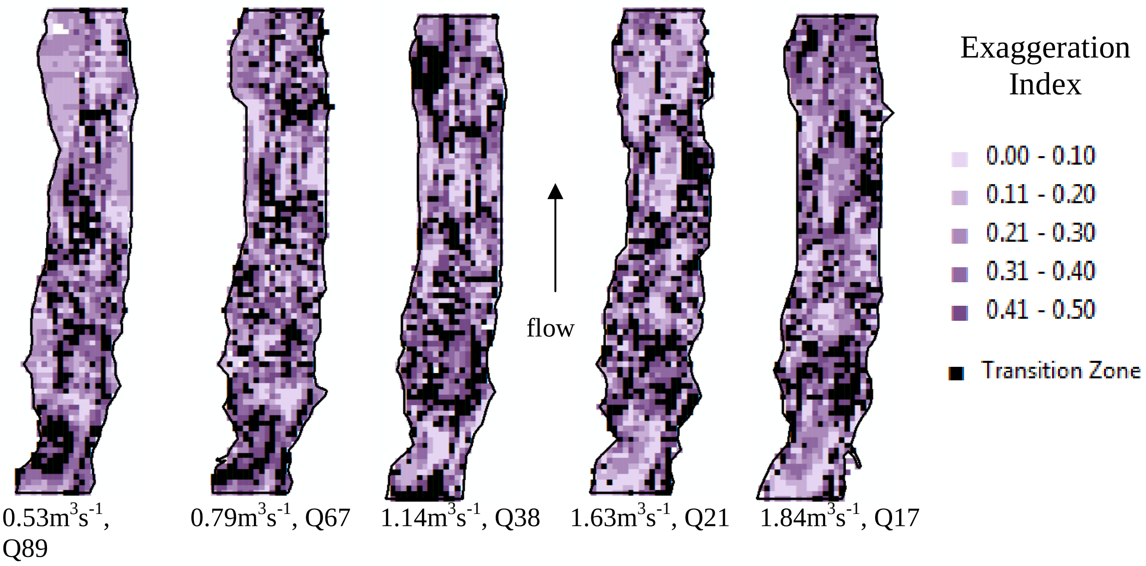


Figure 3.38. Spatial variation of the Exaggeration Index for the defuzzified 5-GK classification (*m*=2).

3.4.3 Leigh Brook hydraulic patch classification

Hydraulic survey data

Table 3.7 shows the maximum, minimum, mean, range and spread of depths and velocities recorded during each hydraulic survey. Mean depth increased steadily with discharge. The variance and range of values also increased but to a slighter degree. Mean velocity also increased with discharge but in steps, remaining similar at low-moderate flows and high-very high flows. The spread of values around the mean increased steadily with discharge until high flow then decreased slightly at very high flow. Maximum velocity occurred at the break in slope immediately upstream of the scour zone at every flow but did not increase uniformly with discharge.

Table 3.7. Descriptive statistics of hydraulic data collected at the Leigh Brook.

Hydraulic parameter	$0.26m^3/s$, Q87	$0.38m^3/s$, Q67	$0.61m^3/s$, Q45	$0.99m^3/s$, Q23	$1.30m^3/s$, Q14
$Depth_{max}$	0.51	0.52	0.63	0.63	0.72
$Depth_{min}$	0.01	0.01	0.01	0.01	0.01
$Depth_{mean}$	0.11	0.14	0.18	0.23	0.26
$Depth_{variance}$	0.007	0.009	0.011	0.013	0.014
$Depth_{range}$	0.50	0.51	0.62	0.62	0.71
$Velocity_{max}$	0.923	1.607	1.424	1.639	1.509
$Velocity_{min}$	-0.084	-0.668	-0.177	-0.350	-0.114
$Velocity_{mean}$	0.154	0.251	0.296	0.547	0.567
$Velocity_{variance}$	0.026	0.050	0.071	0.098	0.092
$Velocity_{range}$	1.007	2.275	1.601	1.989	1.623

Figure 3.39 illustrates the spatial variability of depth and velocity throughout the reach. A wide range of velocities were sampled at all but the lowest discharge surveyed. Relatively little longitudinal variation was evident in the central section of the reach (8-20m), however the lateral distinction increased with discharge as velocities at the thalweg increased. The highest velocities occurred immediately upstream of the scour zone at a break in slope where the channel thalweg flowed between two areas of deposition. The magnitude and extent of high velocities in this area increased with discharge and a new area at the downstream extent of the reach became evident. The deepest water occurred in the scour zone and the central section of the reach, spreading to the downstream extent at higher discharges.

The bivariate data distribution at each flow was plotted in attribute space (Figure 3.38). At very low flow an exponential distribution was evident. The greatest density of data points occurred in the lower left of the distribution where conditions were slow and shallow, but the distribution also had two tails indicating the fast and the deep extremes. This was similar to the River Arrow distribution at very low flow, albeit with much shorter tails, indicative of the more subdued bedforms and smaller scale patch structure at this site. At discharge increased data, the distribution shifted towards deeper, faster conditions and the data were more evenly distributed. At very high flow the distribution was more patchy; velocities <0.4m/s occurred in shallow or deep water but were not associated with moderate depths. Two diagonal bands were evident in the distribution, suggesting that in some cases depth and velocity were positively correlated and in others negatively correlated. However there was a significant degree of scatter.

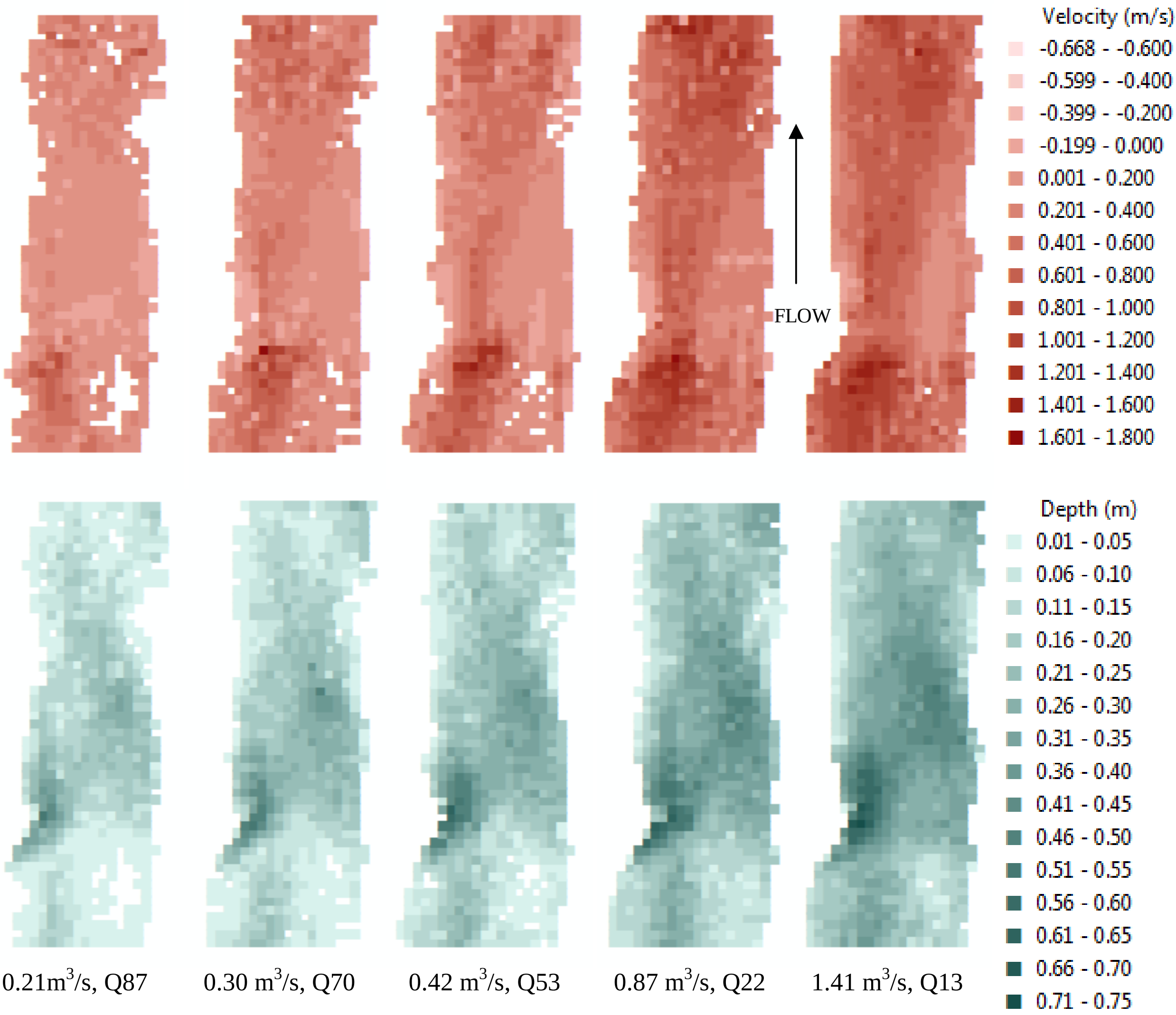


Figure 3.39. Variation in the mean column velocity (top) and depth (bottom) measured at each discharge throughout the reach, River Arrow (discharge increases from left to right).

Assessment of cluster tendency

The Hopkins statistic confirmed that the data combined across all flows were neither randomly nor uniformly distributed (H=1). The data appeared to have a continuous unimodal distribution with the greatest density of data points in the lower left region of the attribute space at depths <0.1m and velocities <0.250ms^{-1}. Data density decreased towards the limits of the data range and was particularly sparse at depths >0.4m. A 2D

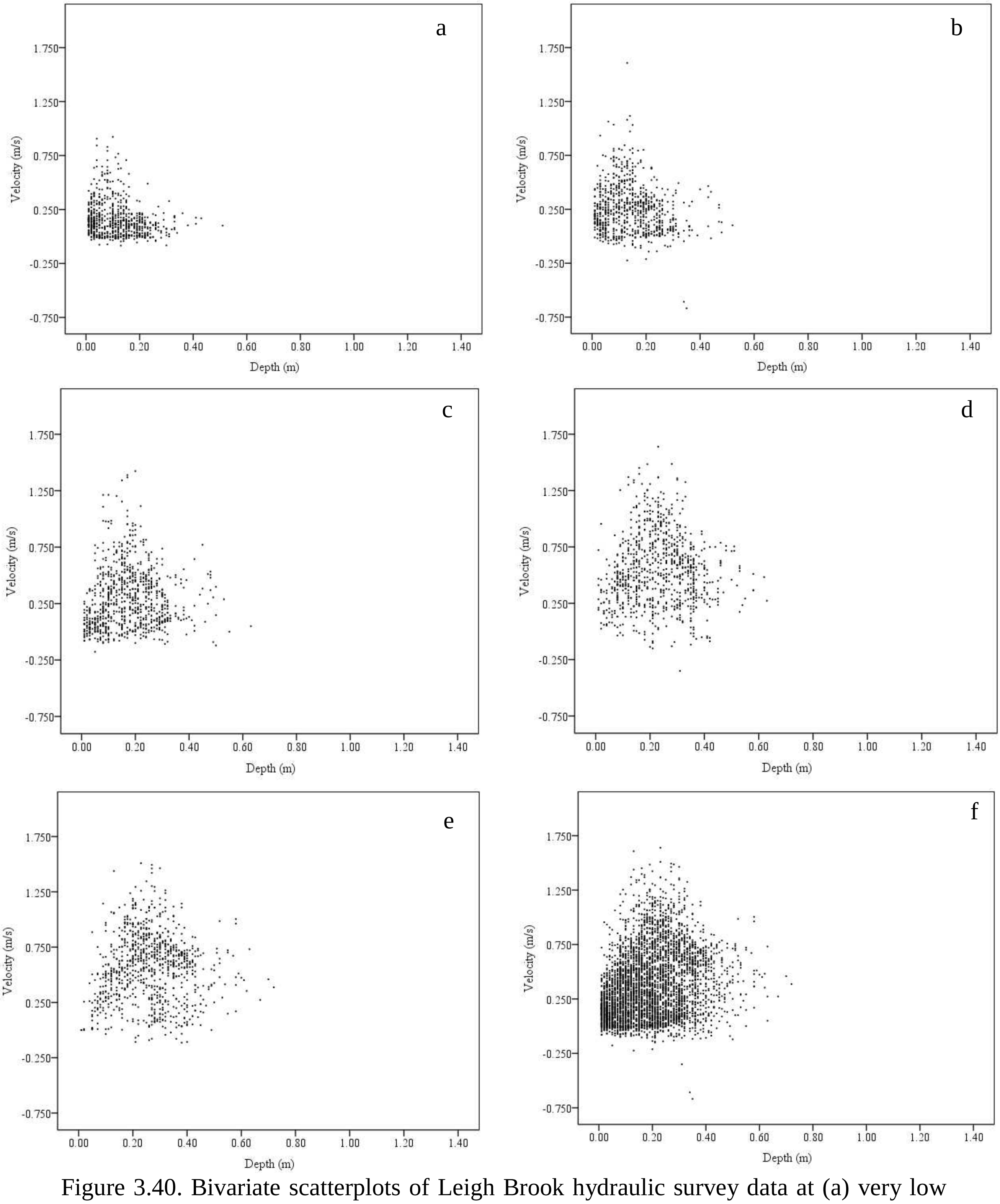


Figure 3.40. Bivariate scatterplots of Leigh Brook hydraulic survey data at (a) very low flow, (b) low flow, (c) moderate flow, (d) high flow, (e) very high flow and (f) combined flows.

histogram of the depth-velocity distribution also suggested a unimodal distribution, with a high frequency of points evident at [0-0.05m, 0-0.25ms^{-1}] (Figure 3.41). The distribution of data combined from all discharges did not appear to have a natural cluster grouping.

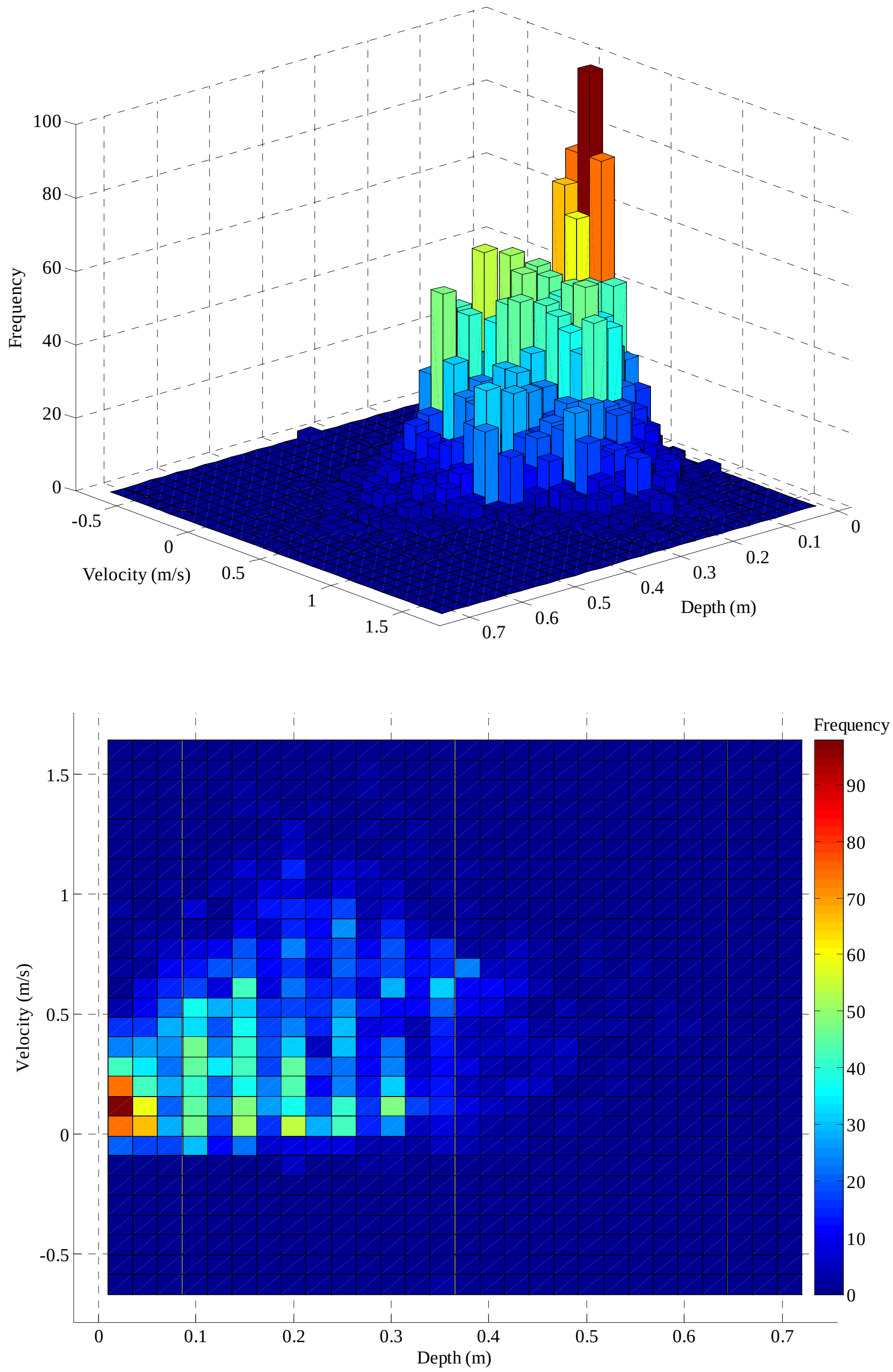


Figure 3.41. 2D histogram (top) and intensity plot (bottom) showing the bivariate distribution and frequency of all depth-velocity measurements collected at the Leigh Brook.

Visual inspection of the data distribution from individual flows (Figure 3.40) did reveal some evidence of natural fuzzy clusters. At very low flow to moderate flow a region of higher density at velocities <0.25ms^{-1} and depths <0.1m was separated by a region of lower density from the rest of the data distribution (Figure 3.42). At high and very high flow a different region of high density with slightly deeper, faster conditions was evident and a region of low density was evident in the bulk of the data distribution which separated the fastest velocities from the slightly deeper, slower conditions (Figure 3.42).

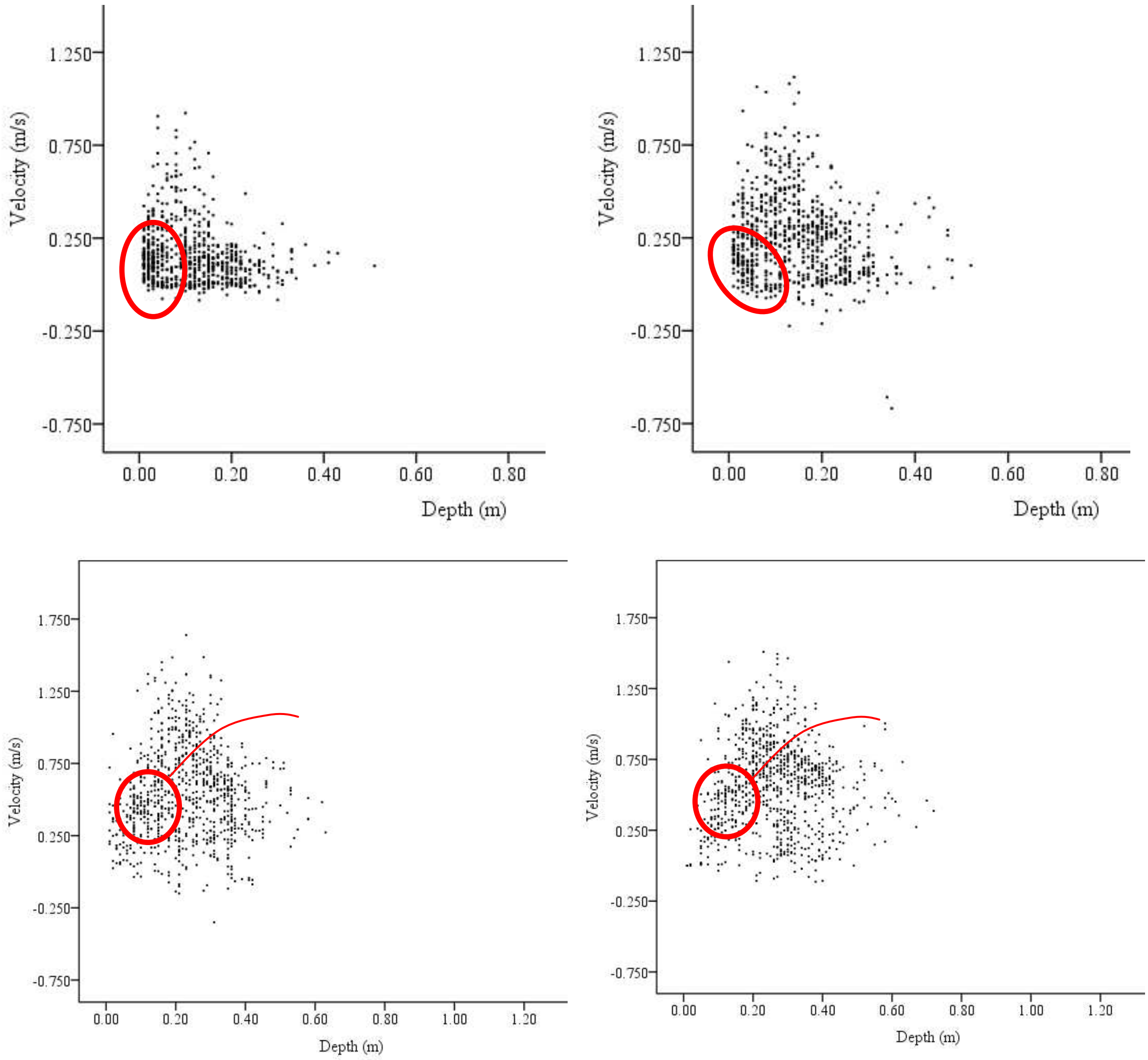


Figure 3.42. Leigh Brook hydraulic data distribution at very low (top left), low (top right), high (bottom left) and very high (bottom right) with markers encircling regions of high density and lines indicating regions of lower density.

Selecting the optimal classification

The same decision-making process was used to select the optimal classification here as at the previous two sites, aided by reference to the channel topography (Figure 3.43) and a field sketch of geomorphic features (Figure 3.44). All classifications produced at this site are shown in Appendix D. The 5 Gustasfon-Kessel classification was selected as the optimal classification at the Leigh Brook because it reflected the natural fuzzy clusters evident in the data distribution at individual discharges, it produced spatially coherent hydraulic patches that reflected the key geomorphic features shaping the hydraulic environment (Figure 3.43) and showed how their hydraulic performance changed with discharge. The shallow-slow conditions associated with zones of deposition that were evident as natural fuzzy clusters in data from individual discharges were identified as hydraulic patch type 2 at low to moderate flows and patch type 3 at high flows. At low flows hydraulic patch type 3 identified the shallow, moderately fast flow associated with exposed boulders and breaks of slope. The scour zone adjacent to the tree on the left bank in the upstream extent of the reach and the deepest part of the low gradient central section of the reach were delineated as patch type 5. This was characterised by low velocities and a range of depths (0.15-0.46m) indicative of pool type environments. The ellipsoidal shape of this cluster in the 5-GK classification reflected the larger range of depths associated with the scour pool better than the equivalent spherical shaped cluster in the 5-FCM classification. At high flows the 5-GK classification delineated two clusters with relatively high velocities which was an improvement over the 4_GK classification. These two clusters – patch types 4 and 5 - reflected the two areas of the high and very high flow data distribution described in Figure 3.42. Patch type 4 (moderate-fast) identified areas of the channel previously occupied by patch types 2 and 3 where the increase in discharge primarily increased velocity whereas patch type 1 reflected areas of the channel where increases in discharge primarily increased depth. As such this classification made the effect of discharge on the data distribution very clear. There was very little difference between the 5-GK and the 5-FCM classification however in recognition of the advantage of the ellipsoidal shaped cluster for improving the depth range associated with the scour pool the Gustafson-Kessel classification was selected as optimal.

Classifications with more clusters reflected depth and velocity gradients in more detail but resulted in more spatial noise and many more single pixel patches. The 6-GK classification reflected additional variations in velocity, identifying a moderately deep and moderately fast hydraulic patch reflecting the path of the thalweg at low to moderate flows, whereas the 6-FCM classification differentiated lateral variations in depth in the low gradient central section of the reach in greater detail. The 7-GK classification incorporated both these differences but did not reflect the influence of distinct geomorphic features. Conversely, the 2, 3 and 4 cluster classifications over-simplified hydraulic differences, failing to reflect distinct spatial or temporal variations.

Fuzzy cluster membership distributions and classification uncertainty

Table 3.8, which shows the hydraulic characteristics of each cluster centroid (prototype), confirms that the 5-GK classification identified five fuzzy clusters with different hydraulic character, relative to the range of depths and velocities in the dataset. By mapping the membership function values (MFVs) of all entities to each cluster, the location and extent of each fuzzy cluster at each discharge could be illustrated (Figures 3.45 to 3.49).

Table 3.8. Cluster centroids for the 5 Gustafson-Kessel classification

Cluster	Hydraulic description (depth-velocity)	Depth (m)	Velocity (ms^{-1})
1	Moderately deep-moderately fast	0.35	0.596
2	Shallow-slow	0.06	0.135
3	Shallow-moderate	0.14	0.448
4	Moderate-fast	0.22	0.891
5	Moderate-slow	0.26	0.104

High MFVs to fuzzy cluster 1 (moderately deep-moderately fast) were relatively infrequent until moderate flow when they appeared in a longitudinal band through the scour zone. As discharge increased this location spread to include the thalweg through the central section of the reach (8-20m). High MFVs to fuzzy cluster 2 (shallow-slow) were widespread at very low flow, particularly in and around depositions zones and channel margins. In contrast to fuzzy cluster 2, as discharge increased MFVs decreased and the extent of the cluster was noticeably reduced, becoming particularly limited at high and very high flows. High MFVs to fuzzy cluster 3 (shallow-moderate) were, on average, lower than those to fuzzy clusters 1 and 2 and shifted location more as discharge increased. At very low flow, high MFVs were located in a small area at the upstream extent of the reach where the thalweg was channelled between two deposition zones. At low flow the extent increased significantly, indicating the path of shallow-moderate flow throughout the central and downstream sections of the reach. At moderate flow the area at the upstream extent of the reach was bisected by faster, deeper flow into two bands either side of the thalweg. This trend continued at high flow with all high MFVs being constrained to the channel margins or deposition zones. Only a very small number of high MFVs to this cluster were present at very high flow. High MFVs to fuzzy cluster 4

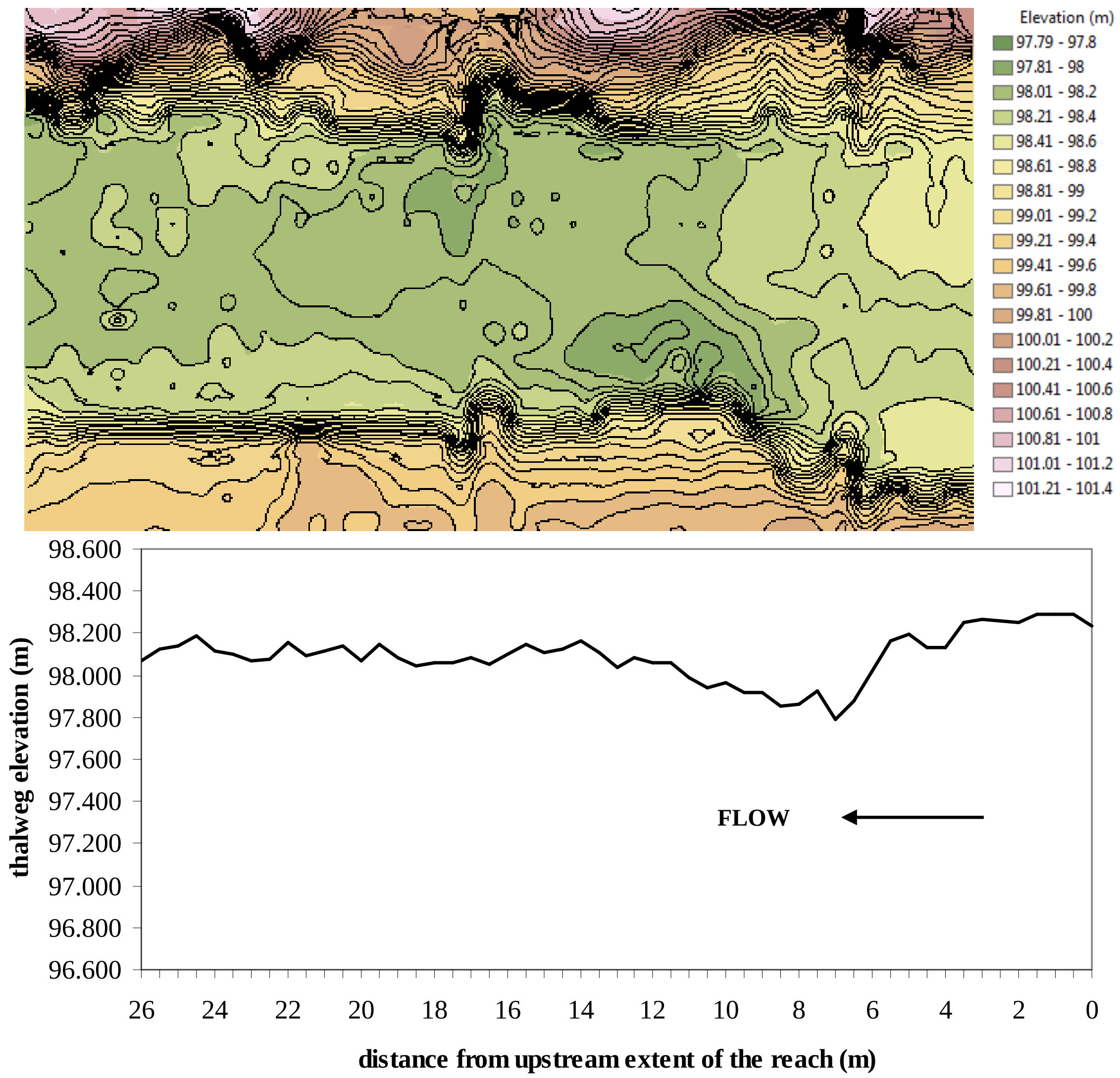


Figure 3.43. Channel topography (top) and variation of thalweg elevation (bottom) in the Leigh Brook study reach.

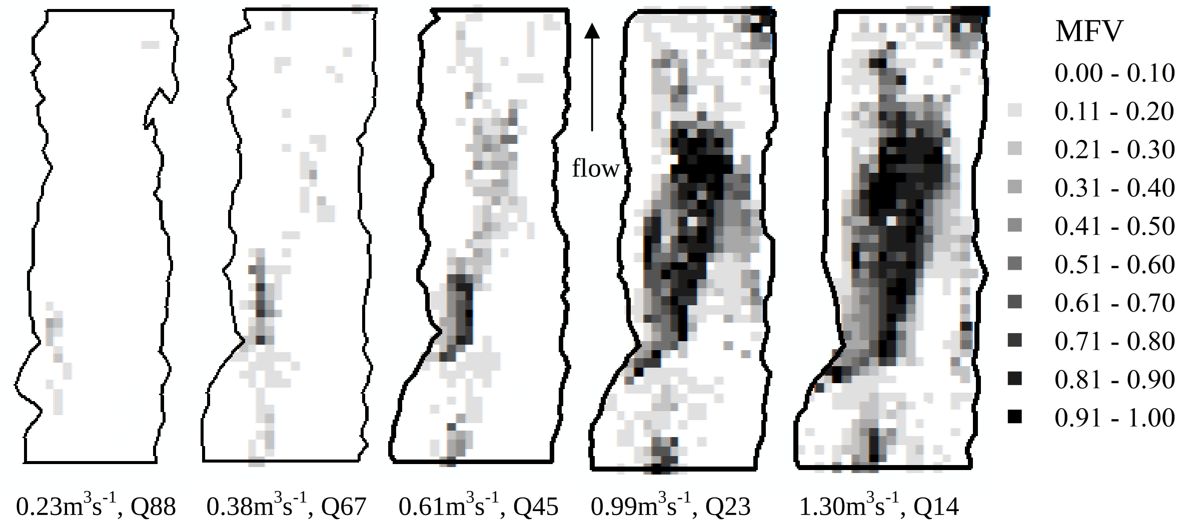


Figure 3.45. Spatial variation of membership function values to fuzzy cluster 1 in the 5-cluster Gustafson-Kessel classification, Leigh Brook.

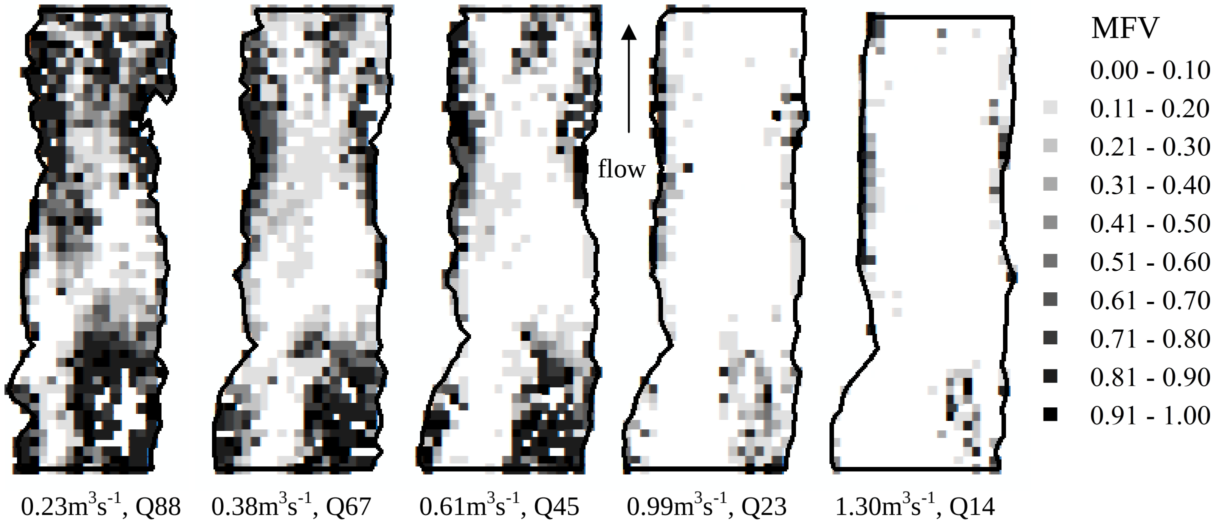


Figure 3.46. Spatial variation of membership function values to fuzzy cluster 2 in the 5-cluster Gustafson-Kessel classification, Leigh Brook.

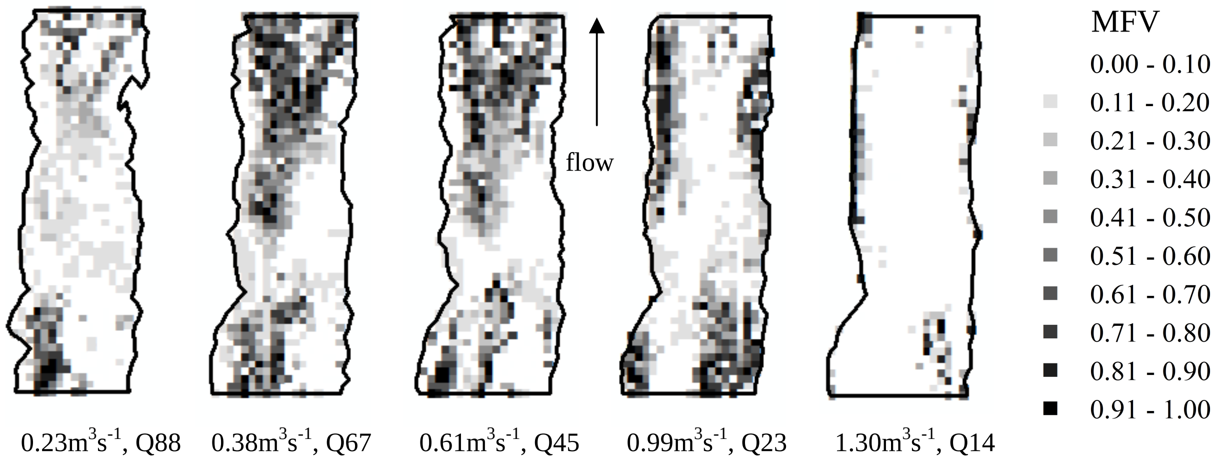


Figure 3.47. Spatial variation of membership function values to fuzzy cluster 3 in the 5-cluster Gustafson-Kessel classification, Leigh Brook.

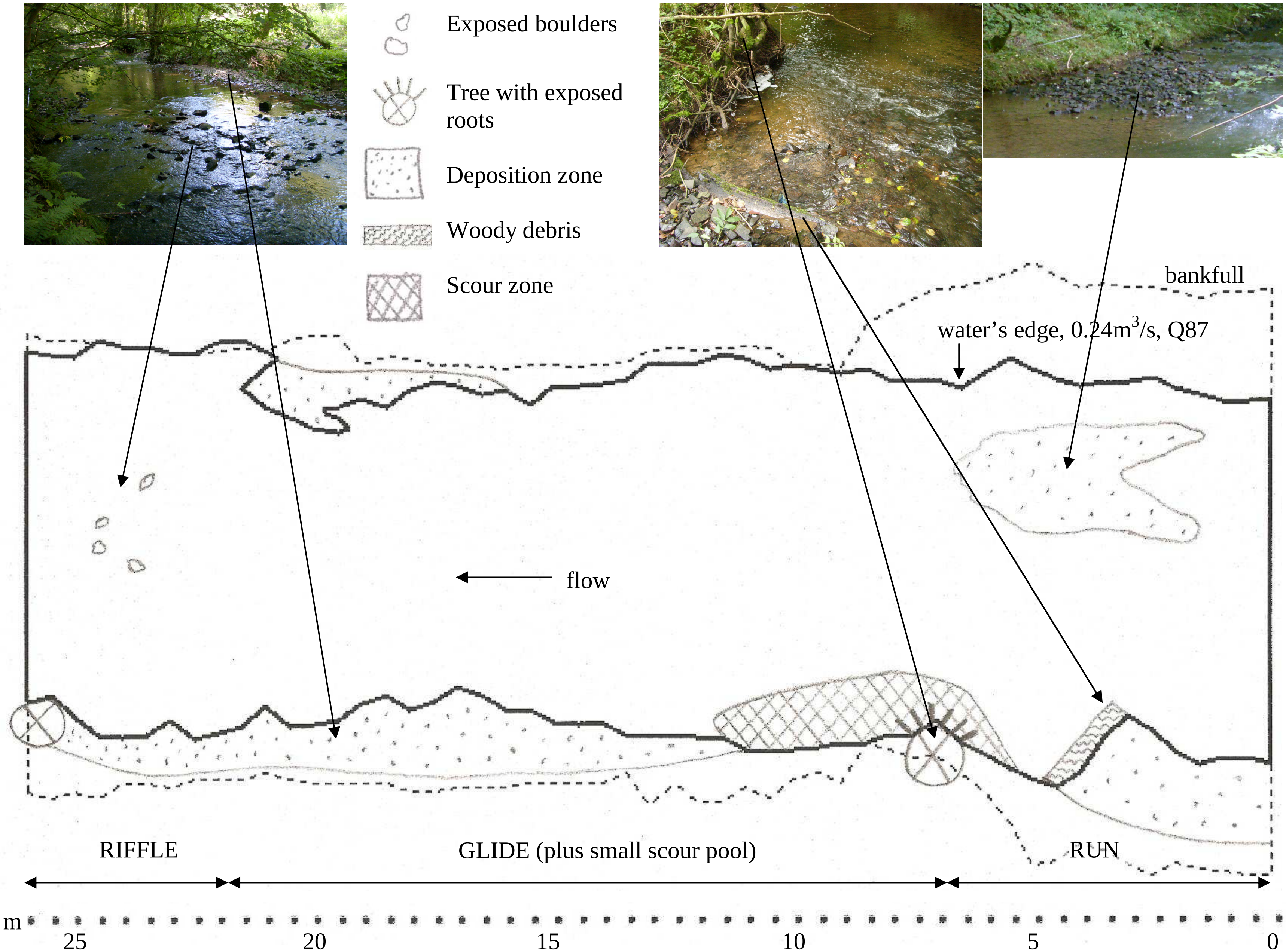
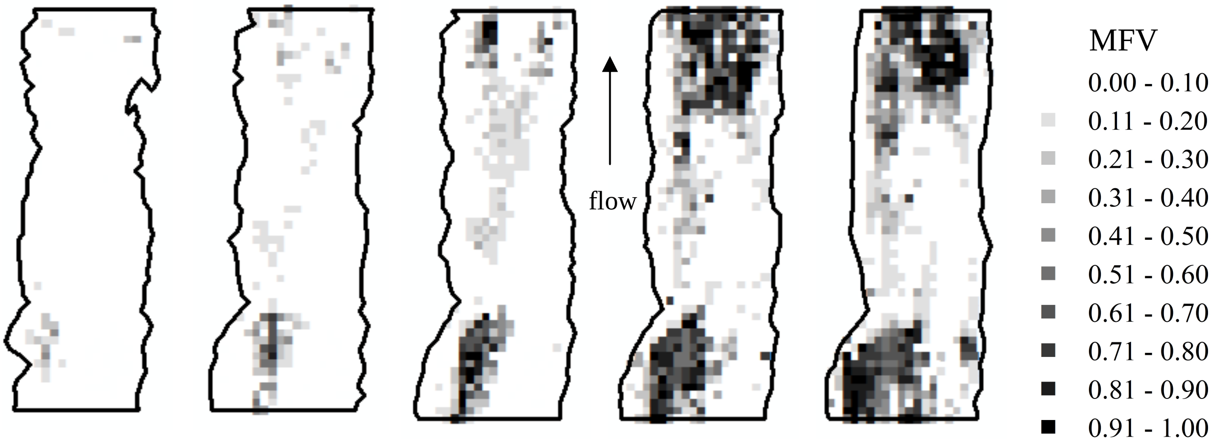
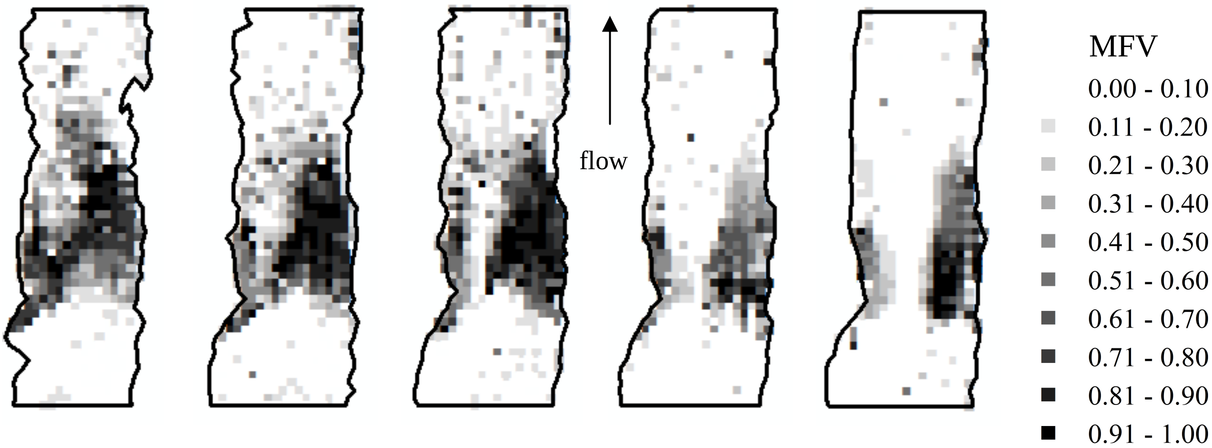


Figure 3.44. Annotated field sketch of the main geomorphic features in the Leigh Brook study reach. The extent of channel geomorphic units identified during a bankside rapid habitat assessment are also shown.



0.23m^3s^{-1}, Q88 0.38m^3s^{-1}, Q67 0.61m^3s^{-1}, Q45 0.99m^3s^{-1}, Q23 1.30m^3s^{-1}, Q14

Figure 3.48. Spatial variation of membership function values to fuzzy cluster 4 in the 5-cluster Gustafson-Kessel classification, Leigh Brook.



0.23m^3s^{-1}, Q88 0.38m^3s^{-1}, Q67 0.61m^3s^{-1}, Q45 0.99m^3s^{-1}, Q23 1.30m^3s^{-1}, Q14

Figure 3.49. Spatial variation of membership function values to fuzzy cluster 5 in the 5-cluster Gustafson-Kessel classification, Leigh Brook.

(moderate-fast) were most prevalent at high flows, appearing in two patches spanning one to two thirds of the channel width at the upstream and downstream extent of the reach. At low to moderate flows partial membership to this cluster occurred at the thalweg, indicating class overlap with fuzzy cluster 3. High MFVs to fuzzy cluster 5 were present at every discharge, appearing in the central section of the reach adjacent to both banks in the deepest section of the channel at low-moderate flows but constrained to the right bank at very high flow as the zone of fast flow expanded. Pixels with partial membership to this cluster were relatively rare, suggesting the distinct character of this cluster.

Spatial variation of the Confusion Index was mapped to illustrate the degree of class confusion/overlap throughout the reach (Figure 3.50). The Confusion Index was spatially noisy and widespread at all flows indicating a high degree of overlap between fuzzy clusters. At very low to moderate flows the areas with greatest class certainty were the deposition zone at the upstream extent of the reach and the moderately deep, slow flowing region adjacent to the right bank. The zone of confusion between these two areas became increasingly crisp at moderate flow. At high flow a very high proportion of the channel had confused class membership. The areas associated with least class confusion at lower flows were associated with a high degree of class confusion, suggestive of a discharge-related change in hydraulic character. High class certainty was evident in a larger proportion of the channel at very high flow. The thalweg was classified with a high degree of certainty along the whole reach except at two locations where there was a change in depth. Here classes overlapped increasing the Confusion Index. The moderately deep, slow-flowing area adjacent to the right bank was also classified with certainty and was separated from the thalweg by a narrow zone of class confusion indicating a relatively crisp boundary between these two areas.

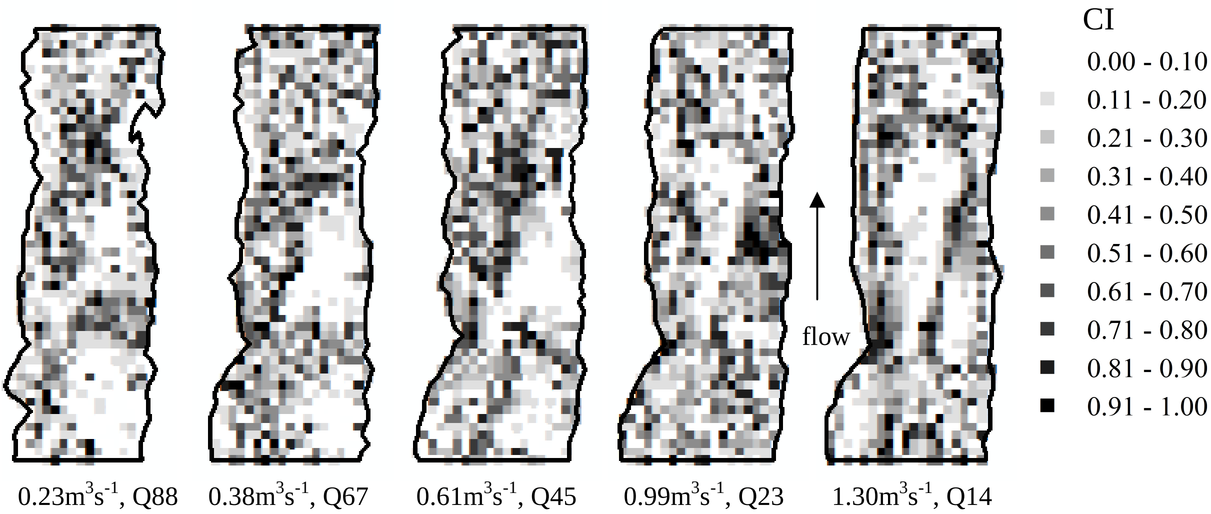


Figure 3.50. Spatial variation of the Confusion Index for the 5-cluster Gustafson-Kessel classification, Leigh Brook.

Hydraulic patches and the transition zone

The fuzzy classification was defuzzified to create five crisp hydraulic patches and a transition zone representing areas of patch type overlap/patch type confusion (Figure 3.51). The median and spread of depths (m) and velocities (ms^{-1}) in each hydraulic patch type are shown in Figure 3.52. Figure 3.53 shows the change in location and extent of each hydraulic patch type with every increase in discharge. A summary of the hydraulic character of hydraulic patches at each flow is shown in Table 3.9. LB1 was characterised by moderately deep, moderately fast conditions and showed a steady increase in velocity as discharge increased. Mean depth remained relatively stable at all flows. This patch type was not present at all at very low flow and its extent increased significantly with discharge, illustrating its discharge-dependent nature. At low flow this LB1 appeared as a narrow linear band through the scour zone adjacent to the large tree on the left bank. The area expanded laterally and longitudinally as discharge increased, indicating the path of the thalweg at higher flows. LB2 delineated the shallow, slow-flowing areas of the channel, predominantly located at zones of deposition and channel margins at low flows. As depth and velocity increased with discharge, so the extent of LB2 decreased as it became constrained to the very edge of the channel at high flows. LB3 identified areas of the channel with moderate depth and moderate velocity. At very low flow this was the path of the thalweg through the topographic high points at the upstream and downstream extent of the reach. The extent of LB3 increased with discharge until moderate flow, above which the increased depths and velocities in these areas were better represented by LB4 (moderate-fast). At high flows the extent of LB3 decreased and shifted, replacing LB2 at deposition zones and near channel margins. Mean depth and velocity varied with discharge to a small degree but did not show a uniform increase or decrease. LB4 delineated the moderately deep, fast-flowing conditions at the topographic high points at the upstream and downstream extent of the reach at high flows. This patch type was not present at very low flow and in two very small patches at low flow. The extent spread noticeably at moderate flow and high flow, but decreased slightly at very high flow in areas where depths increased and were better characterised by LB1. The final patch type in this classification, LB5, delineated the deepest parts of the channel with slow-flowing water. At low flows these conditions were found in the central section of the reach including the scour zone. As discharge and the velocity of the thalweg increased through this area, so the extent of LB5 was marginalised to an area between the thalweg and the right bank. Both depth and velocity showed a small increase with discharge.

The Transition Zone incorporated the full range of depths and the vast majority of velocities sampled (Figure 3.52). Its extent was relatively stable at each discharge, occupying between 18-23% of the reach. At very low flow the Transition Zone mainly occurred at the boundary between hydraulic patch types and was 1-2 pixels wide (0.25-0.5m). A larger transitional area was located at the boundary between LB2, LB3 and LB5 at 17-20m, signifying an area of class overlap. This area, plus the path of the thalweg through the central section of the reach, was also transitional at low and moderate flows. At high flows noticeable transitional patches occurred where LB1 and LB5 overlapped. The scour zone was classified as transitional at all but very low flow. Turbulent, variable hydraulic conditions at this location made it uncharacteristic of any single cluster; depth being too high to be classified as LB5 or velocities too slow to be LB1.

The degree to which MFVs had been exaggerated to allocated fuzzy entities to crisp clusters (i.e. 1-MFV) at each discharge was mapped (Figure 3.54). As might be expected, the greatest exaggeration occurred at patch boundaries where class confusion was more likely. The core of patch types LB1, LB2 and LB5 was exaggerated the least. LB3 and LB4 were associated with a higher degree of class exaggeration, possibly because they occurred in areas with a high degree of turbulence and variability, such as exposed or newly inundated boulders and breaks in slope.

Table 3.9. Change in mean depth (m) (D_{mean}) and mean velocity (ms^{-1}) (V_{mean}) in each hydraulic patch with each increase in discharge. Standard deviation is shown in brackets.

Hydraulic Patch	Variable	Q87 ($0.23m^3s^{-1}$)	Q67 ($0.37m^3s^{-1}$)	Q45 ($0.61m^3s^{-1}$)	Q23 ($0.99m^3s^{-1}$)	Q14 ($1.30m^3s^{-1}$)
LB1	D_{mean}	-	0.39 (.06)	0.36 (.07)	0.35 (.06)	0.37 (.06)
	V_{mean}	-	0.432 (.05)	0.529 (.10)	0.591 (.12)	0.635 (.11)
LB2	D_{mean}	0.05 (.04)	0.06 (.03)	0.06 (.03)	0.06 (.03)	0.07 (.03)
	V_{mean}	0.118 (.09)	0.135 (.10)	0.083 (.10)	0.171 (.10)	0.153 (.11)
LB3	D_{mean}	0.10 (.04)	0.13 (.05)	0.14 (.04)	0.13 (.04)	0.14 (.04)
	V_{mean}	0.474 (.12)	0.446 (.12)	0.452 (.10)	0.477 (.10)	0.471 (.10)
LB4	D_{mean}	-	0.15 (.02)	0.18 (.03)	0.22 (.05)	0.22 (.04)
	V_{mean}	-	0.932 (.26)	0.911 (.19)	0.973 (.19)	0.939 (.18)
LB5	D_{mean}	0.22 (.06)	0.25 (.05)	0.26 (.06)	0.29 (.06)	0.32 (.07)
	V_{mean}	0.068 (.06)	0.074 (.07)	0.094 (.08)	0.126 (.11)	0.116 (.09)

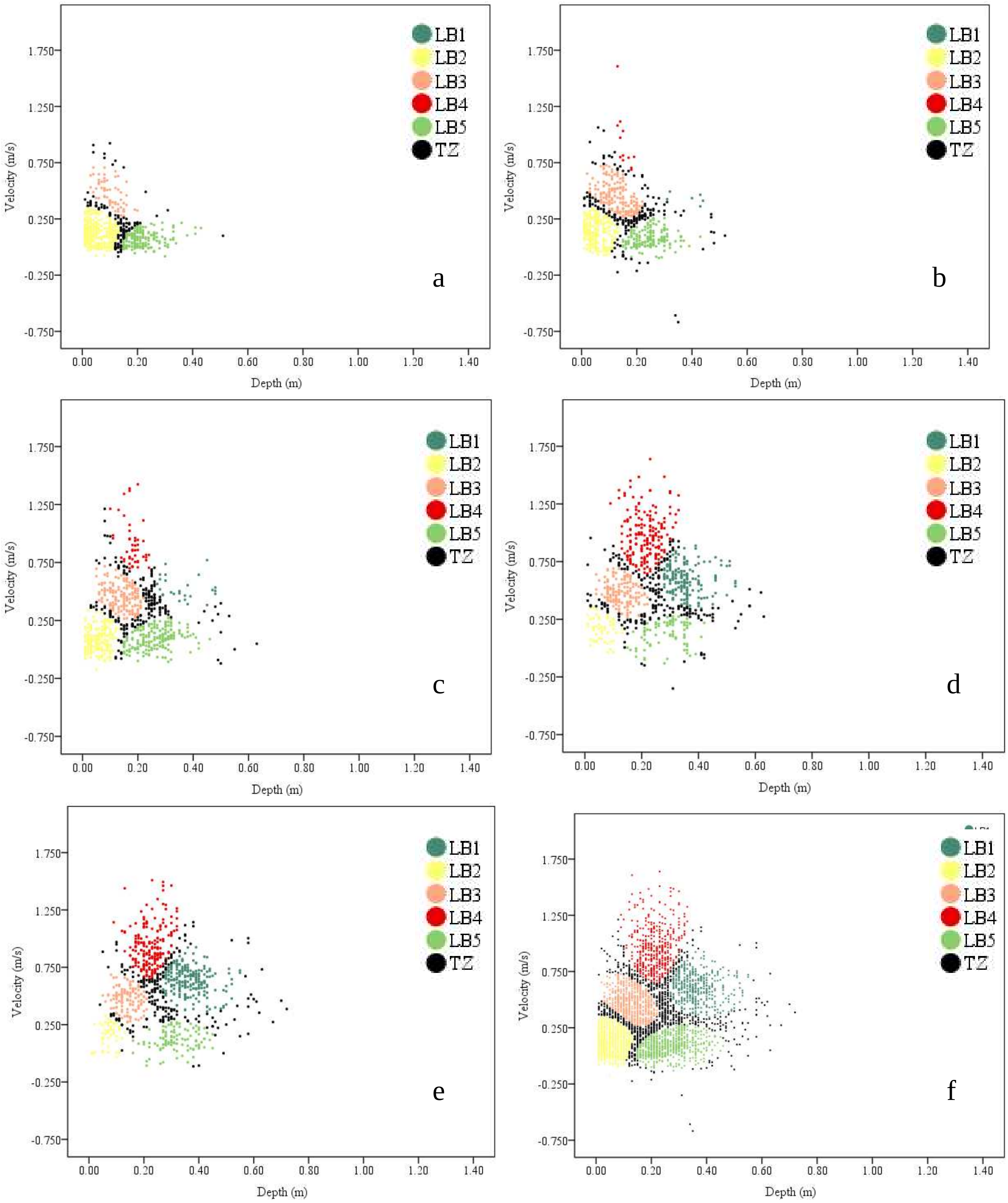


Figure 3.51. Scatterplot showing the bivariate distribution of depth-velocity data from the Leigh Brook at (a) very low, (B) low, (c), moderate, (d) high, (e) very high and (f) all flows. Colours indicate hydraulic patch membership for the defuzzified 5-cluster Gustafson-Kessel classification.

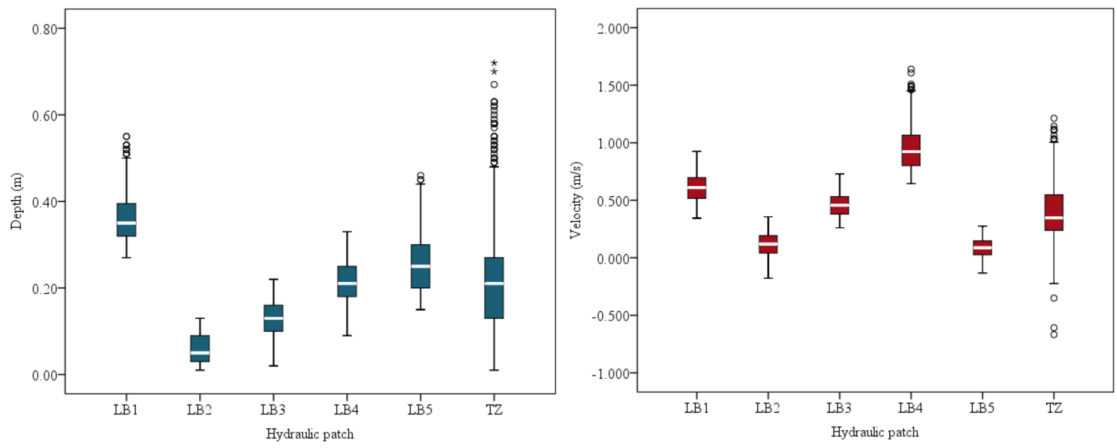
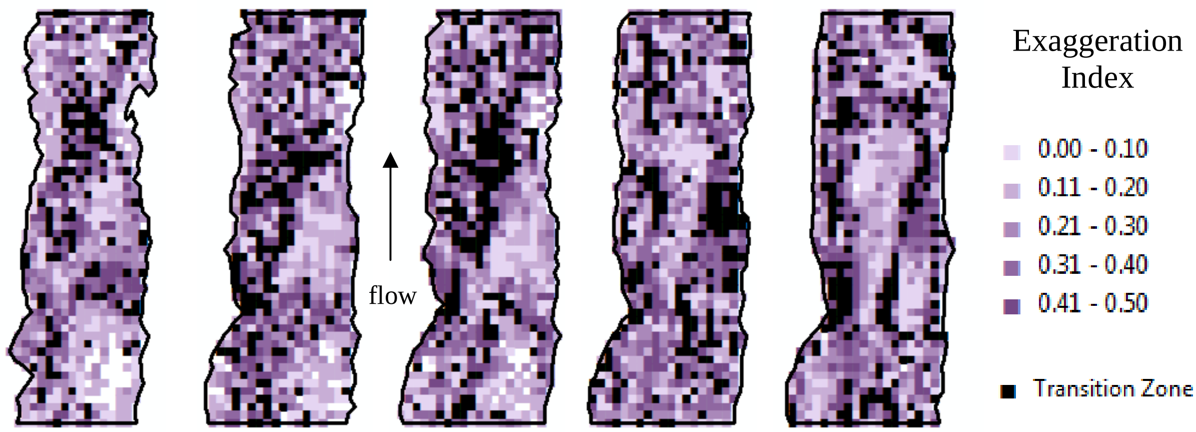


Figure 3.52. Boxplots showing the median and spread of depth (m) (left) and velocity (ms^{-1}) (right) in each hydraulic patch across all flows.



$0.23m^3s^{-1}$, Q87 $0.37m^3s^{-1}$, Q67 $0.61m^3s^{-1}$, Q45 $0.99m^3s^{-1}$, Q23 $1.30m^3s^{-1}$, Q14

Figure 3.55. Spatial variation of the Exaggeration Index for the defuzzified 5-GK classification (*m*=2)

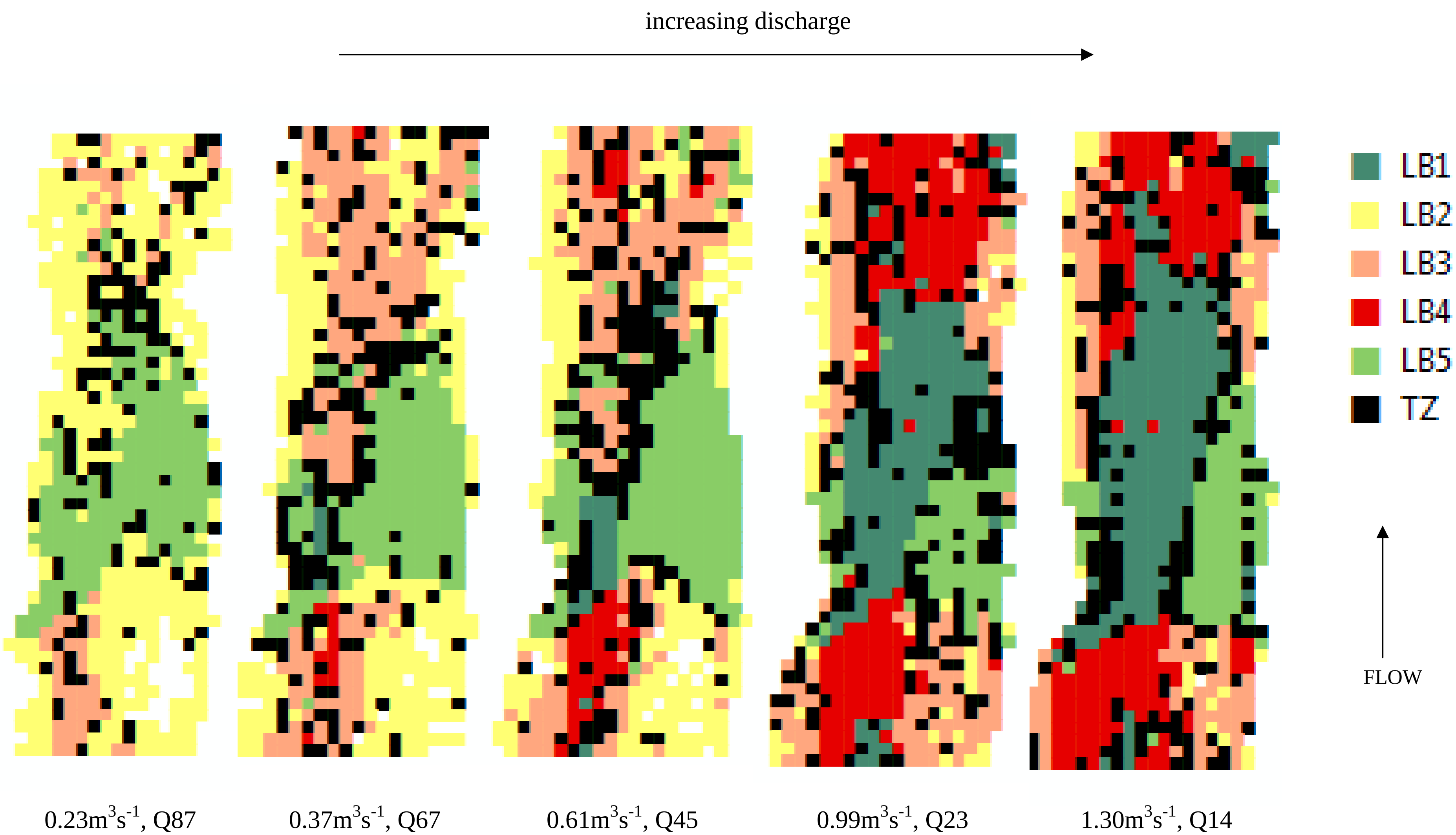


Figure 3.53. Location and extent of hydraulic patches (LB1-LB5) and transition zones (TZ) delineated at each flow at the Leigh Brook site.

3.5 Discussion

At the beginning of the chapter it was hypothesised that hydraulic patches, defined by the joint distribution of depth and velocity, indexed by within-patch homogeneity and between-patch heterogeneity, exist inherently – i.e. the hydraulic environment has a natural patch structure determined by the interaction between discharge and channel morphology. It was further hypothesised that spatially coherent hydraulic patches and the transitional zones between them could be delineated quantitatively and objectively using fuzzy cluster analysis. Assessment of clustering tendency showed that in attribute space hydraulic data combined from surveys at multiple discharges has a continuous unimodal distribution rather than a well-defined cluster/group structure. However visual inspection of the data distribution collected at individual discharges did show the presence of natural fuzzy clusters, defined as regions with a high density of points separated by regions with a relatively lower density of points. The optimal classification of the combined hydraulic data was that which best reflected the natural cluster structure of its constituent data sets. As depth and velocity are spatially correlated variables, clusters delineated in attribute space from the combined dataset produced spatially coherent hydraulic patches with strong geographic zoning when mapped in geographic space. In this sense, **meaningful hydraulic patches can be delineated using cluster analysis.** Furthermore the application of **fuzzy cluster analysis enabled transitional boundary zones, areas where hydraulic observations were characteristic of more than one hydraulic patch type, to be delineated quantitatively**. As Legleiter and Goodchild (2005) suggested this produces a more faithful representation of the fuzzy, complex in-stream environment and improves on existing classifications that represent hydraulic patches with crisp, linear boundaries (e.g. Padmore, 1997; Emery et al., 2003; Thoms et al., 2006).

Legleiter and Goodchild (2005) discussed how partial membership function values could be mined to illustrate the varying width (crisp or gradual) of transitional zones between hydraulic patches and to indicate areas of the channel with high habitat diversity (i.e. confused class membership), thus improving the spatial classification of the hydraulic environment. The results of this study showed that the **defuzzification rule used has a significant impact on the final classification** but must be chosen by the user. Previous studies have applied one of two rules; CI>0.6 (Burrough et al., 1997) or a user-specified α-cut threshold (Cheng et al., 2001). This study applied a **unique combination of**

defuzzification rules to ensure that entities with confused class membership and/or <50% membership to a single class were assigned to the transition zone. This approach is recommended to increase confidence in the homogeneity of hydraulic patches.

Under this rule the transition zone occupied a similar and significant proportion of each reach ranging from 18-23% at the Leigh Brook, 18-28% at the River Arrow and 21-30% at the River Salwarpe. The **width and shape of transitional boundary zones reflected the degree of difference in hydraulic conditions** of the patches either side; thin (one or two pixels wide) linear transitional zones occurred between patches with *either* depth *or* velocity differences whilst thicker, patchy transitional zones occurred (a) at the junction of more than two hydraulic patch types, (b) between two patch types with different depth *and* velocity characteristics or (c) between two patch types with large differences in depth or velocity (e.g. slow/fast or shallow/deep). One exception to this occurred at the Leigh Brook where a transitional zone patch appeared in the small scour pool adjacent to the left bank at low and very high flow. Whilst it appeared that this area fell on the boundary between LB1 (mod-mod) and LB5 (slow-mod) patches, closer examination revealed these hydraulic observations occurred at depth-velocity combinations beneath LB5 in attribute space, rather than between LB1 and LB5.

The study and function of boundaries forms a key theme of landscape ecology as applied to riverine landscapes (Wiens, 2002). Boundaries or ecotones often function as biodiversity hotspots so the ability to delineate such zones plays an important part in understanding the link between physical and biological diversity (Amoros et al., 1993; Malanson, 1993**;** Ward & Wiens, 2001; Ward & Tockner, 2001). To date boundary research in riverine landscapes has focussed on riparian and hyporheic zones (Naiman & Décamps, 1997; Boulton et al., 2010). The technique presented in this study for delineating in-stream boundaries between hydraulic patches provides a **new avenue for boundary research at the sub-reach scale**.

Whilst it is acknowledged that choice of an ‘appropriate’ defuzzification rule introduces a degree of subjectivity, an important feature of the quantitative approach to hydraulic patch classification presented here is that it is possible to explicitly quantify uncertainty associated with the chosen classification. The degree to which class membership is exaggerated in a crisp classification of a fuzzy environment can be quantified and mapped, illustrating the degree of heterogeneity within meso scale hydraulic patches and where and to what extent hydraulic complexity has been simplified. This is an improvement over standard meso habitat assessment methods in which the classifications are variously defined. Further work would be needed to test the ecological significance of numerically delineated hydraulic patches and transitional boundary zones.

3.5.1 Linking hydromorphology and the hydraulic environment

A current theme in river research is the development of a reliable method for linking morphology and hydraulics (hydromorphology) to understand or predict the hydraulic performance of bedforms over a range of flows (e.g. Clifford et al., 2002). Examination of the various classifications produced at each site in this study illustrated the influence of reach and meso scale hydromorphology on the hydraulic environment. The 2 cluster classifications reflected the influence of reach scale channel geometry on the depth-velocity distribution. Stewardson & McMahon (2002) described, in theoretical terms, how a prismatic channel with lateral but no longitudinal variation would produce a depth-velocity distribution similar to Figure 3.55a and a rectangular channel with longitudinal but no lateral variation would produce a depth-velocity distribution similar to Figure 3.55b. Whilst the channel geometry of the rivers used in this study had a shape somewhere in between these two theoretical extremes, it was the case that the study reaches were dominated by either lateral or longitudinal variation. Longitudinal variation dominated in the River Arrow reach due to pronounced pool-riffle bedforms. This was particularly evident in the shape of the depth-velocity distribution at low to moderate flows (Figure 3.7a-c, p.17) and was successfully reflected in the 2 cluster classification which partitioned the dataset based on longitudinal differences in depth (2-FCM) or velocity (2-GK) (Appendix C, Figures 1-3, p.1. and Figures 14-16, p. 8). By contrast, the channel geometry of the River Salwarpe and Leigh Brook reaches were characterised more by lateral than longitudinal variation. Consequently the depth-velocity distributions were more like Figure 3.55a, with depth and velocity being positively correlated in some places, such as the modified central section of the River Salwarpe reach. At both sites the 2 cluster classification bisected the depth-velocity distribution diagonally into shallow-slow and deep-fast which, when mapped, illustrated the dominance of lateral variability (Appendix D & E, Figures 1-3, p.1 and Figures 14-16, p.8).

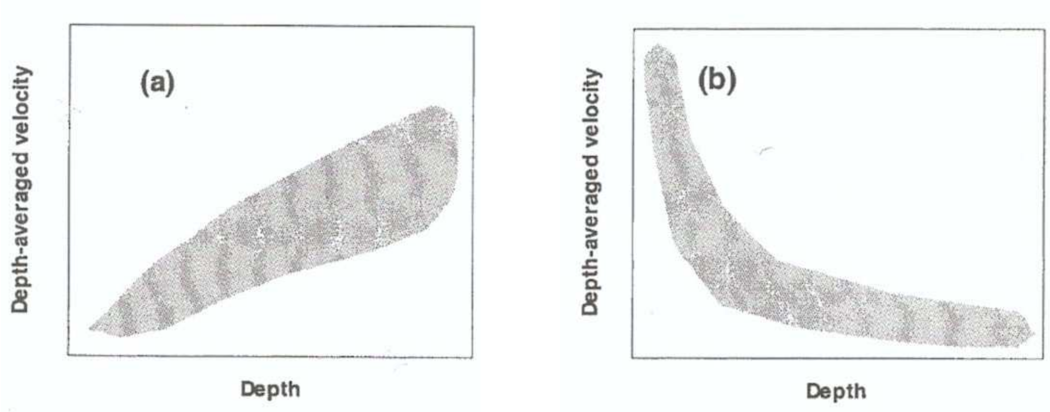


Figure 3.55. Example of the depth-velocity distribution produced by (a) a prismatic channel lacking longitudinal variation, and (b) a rectangular channel lacking lateral variation

The influence of meso scale bedforms on the hydraulic environment was evident from the spatial structure of hydraulic patches delineated from the depth-velocity distributions. In some cases hydraulic patches were nested within meso scale bedforms, reflecting additional lateral and/or longitudinal hydraulic variations. For example, faster, deeper conditions along the channel thalweg were typically distinguished from shallow, slower conditions at channel margins in glide and pool CGUs. In other cases, visually distinguishable bedforms, such as the small scour pool and the riffle in the Leigh Brook reach, were not identified as distinct hydraulic patches, but as transition zones or combinations of hydraulic patch types. This may be a reflection of the fact that these bedforms were small in area so the number of hydraulic data points from them would have formed a very small proportion of the dataset from which the classification was derived. However it may also suggest that some meso scale bedforms may be more appropriately characterised by heterogeneity than homogeneity. Riffles, for example, contain a high degree of spatial heterogeneity due to small scale variations in velocity (and to a lesser extent depth) around exposed substrate. Hydraulic homogeneity may have occurred at smaller spatial scales than could be detected by the sampling resolution in this study. The hydraulics associated with other bedforms may be more appropriately characterised by temporal heterogeneity. In the case of the scour pool, mean velocity (averaged over 10 seconds) may have disguised the turbulent properties that distinguished this unit. These findings support recent ecohydraulic research exploring patterns of microscale heterogeneity associated with meso scale bedforms (e.g. Harvey & Clifford,

2009) and suggest this approach merits further application, alongside studies focused on homogeneity.

Using the method of hydraulic patch classification presented here the combined influence of discharge and channel morphology (hydromorphology) on the hydraulic environment can be reflected. In this study, data from all hydraulic surveys were collated prior to clustering to produce a single fixed classification for each site. The shift in the depth-velocity distribution as discharge increased is reflected by the change in proportion of patch types present at each flow. Under this approach one or two patch types are likely to dominate the reach at very low and very high flows, with a bias towards higher patch diversity at moderate flows due to the position of discharge-specific depth-velocity distributions within the combined-discharge classification. Alternatively hydraulic data collected at each discharge can be clustered separately. This represents relative hydraulic differences at a particular flow in more detail but complicates inter-flow comparisons as the number and hydraulic characteristics of patch types may vary at each flow (e.g. Emery et al., 2003)

3.5.2 Process, decisions and considerations when applying FCA to classify hydraulic patches

Figure 3.56 shows the process that was followed in this study to delineate hydraulic patches and transitional zones. As was discussed earlier, it is essential to define the objectives of applying cluster analysis as this affects several decisions in the process. To identify homogenous hydraulic patches nested within meso scale bedforms it is necessary to select and sample a reach with a representative range of CGUs. Alternatively cluster analysis can be used to evaluate the hydraulic performance of a particular area, for example a modified or restored reach, over a particular range of flows. Sampling resolution of hydraulic data collection should be tailored to the objectives of the study and reflect the scale of the river. Here the method was applied to high-resolution field data from multiple flows and successfully used to explore hydraulic patch dynamics. Point measurements of water depth and streamwise velocity (at 0.6 depth) spaced every 0.5m across the channel and every 1m up/downstream were sufficient to identify hydraulic patches nested within meso scale bedforms at the River Arrow reach, which varied between 3-10m. However this sampling strategy was time consuming and labour intensive, each survey taking a minimum of three people four to six hours to complete. Reducing the sampling resolution would reduce the field effort but is likely to underrepresent hydraulic complexity. Alternatively cluster analysis could be applied to modelled data. Whilst modelling is not an ‘easy option’ and is typically limited to 100-200m reaches, once calibrated it would be possible to assess changes in the hydraulic environment over a larger range of flows without further sampling effort. Additional variables, such as shear stress, could also be derived and included. In this study hydraulic data was collected at a range of flows to evaluate the relative influence of discharge and morphology on the hydraulic environment. Results from this study suggest it is unlikely that hydraulic data combined from a range of flows will have a naturally clustered structure however it is recommended to visually inspect the structure of data distribution from individual discharge surveys for natural fuzzy clusters as this provides useful guidance in the selection of the optimal classification of the combined data.

It is important to note that classifications produced by cluster analysis are data-sensitive. Prior to clustering the data, unrealistic outliers that might skew the classification should be removed and data should be standardised to account for different scales of measurement. The results of this study also show how the classification is limited by the **data range**. Cluster analysis searches for *c* well-separated centroids within the data range. At the River Arrow, the data range incorporated deep-slow and shallow-fast hydraulic extremes representative of the wider bed morphology. Therefore cluster centroids were located in these extremes and the resulting hydraulic patches reflected meso scale bedforms effectively. The Leigh Brook and River Salwarpe reaches did not contain a representative sequence of CGUs as the objectives at these sites were to evaluate how cluster analysis would classify a reach with more subtle bedforms and to evaluate the impact of channel modification on the hydraulic environment. Despite the hydraulic range being smaller, a 5-cluster classification was judged to be optimal because it reflected the scale of morphological features/variations contained in the reach and the influence of discharge variations on the hydraulic distribution. Had longer reaches incorporating a wider range of bedforms and associated hydraulic conditions been sampled, it is less likely these subtle variations would have been distinguished. The influence of the data range on the classification raises two important considerations. Firstly it highlights the importance of having clear objectives and collecting appropriate data before applying cluster analysis. Secondly, classifications produced by cluster analysis cannot be extrapolated with a high degree of confidence to other reaches unless the bedform dimensions and hydraulic range sampled in the study reach are representative of the river at large. Even so, research suggests that the hydraulic performance of bedforms can vary significantly, even at the same site (Pedersen & Friberg, 2007), and for this reason extrapolation of hydraulic patch classifications should be approached with caution.

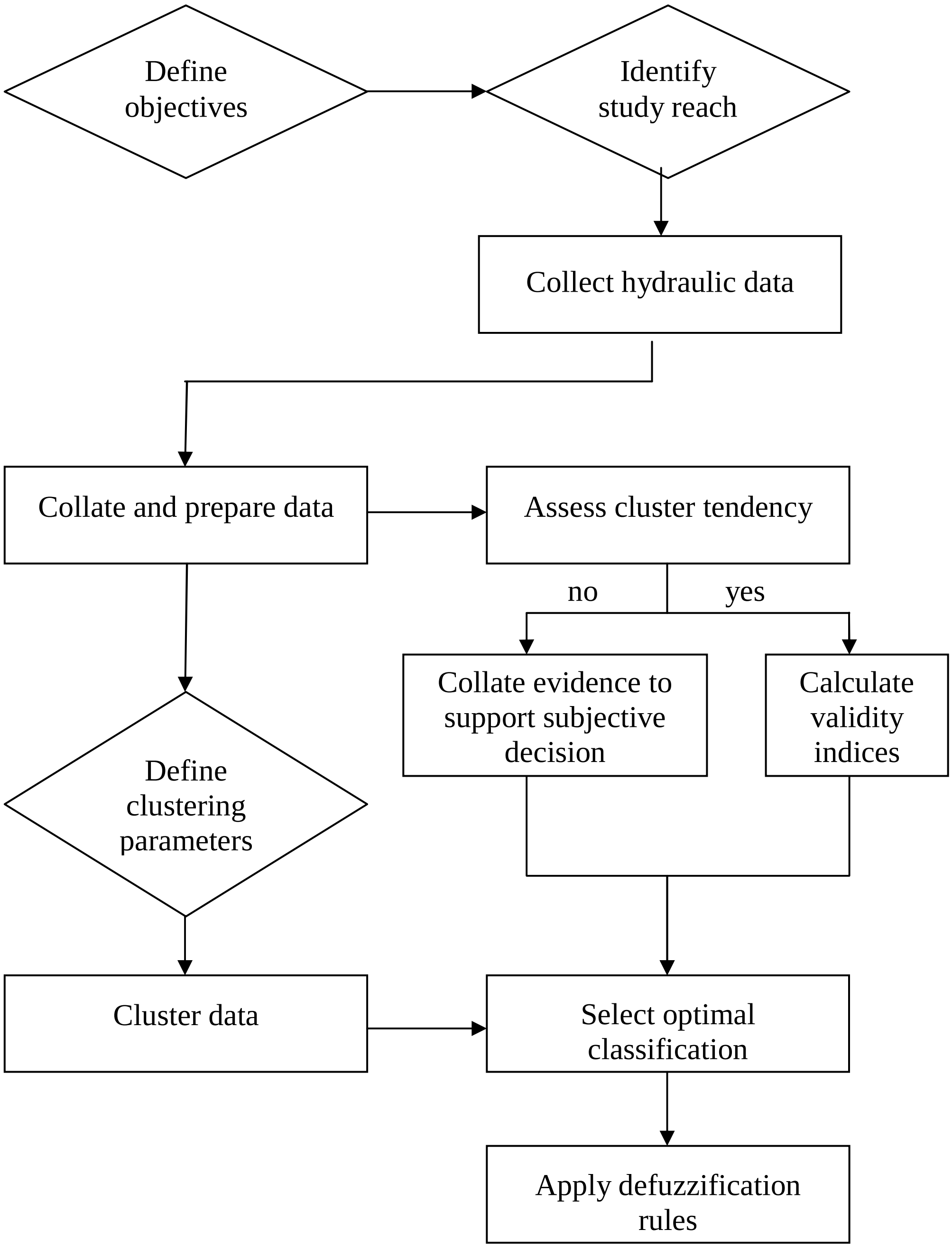


Figure 3.56. Flow chart showing processes and decisions for classifying hydraulic patches and transitional zones using fuzzy cluster analysis.

Classifications produced by fuzzy cluster analysis are also algorithm-sensitive. Of the three algorithms tested in this study, fuzzy *c*-means (MacQueen, 1967; Bezdek et al., 1984) and fuzzy covariance (Gustasfon & Kessel, 1979) both produced good results. Fuzzy *c*-means delineated equal-sized, compact, spherical clusters with low internal variance. Fuzzy covariance delineated equal-sized but *ellipsoidal* clusters with slightly higher internal variance. The greater flexibility of this algorithm made it possible to delineate patches with a large range of one variable, such as depth variations in a pool environment, which arguably provided a more intuitive representative of the influence of hydromorphology on the hydraulic environment. The fact that different clustering algorithms produce different classifications of the same data highlights the necessity of subject expertise and knowledge of the study site when evaluating clustering results and selecting the optimal classification.

One of the difficulties of using cluster analysis is selecting the value for *c* so typically a range of values are compared. Where the input data lacks a naturally clustered structure, objective measures such as validity indices are not helpful in selecting the optimal classification and subjective judgement is required. On clustering hydraulic data, Legleiter and Goodchild (2005) found that validity indices increased or decreased monotonically as *c* was increased so opted to search for four clusters based on a visual survey of meso habitat types (pool, riffle, run/glide, eddy drop zone) in the reach. Although the resulting clusters were found to be "spatially continuous, compact and hydraulically reasonable" (Legleiter & Goodchild, 2005, p.37), the spatial and hydraulic correspondence between hydraulic clusters and field-mapped habitat units was unclear; hydraulic clusters were not constrained to channel-spanning polygons like the field-mapped meso habitat units so revealed more lateral variability, and cluster analysis identified the shallow-slow 'extreme' in the depth-velocity distribution as one of the four clusters but this did not correspond with any of the field-mapped mesohabitats as described by Legleiter & Goodchild (2005). For these reasons, guiding *c* by the number of mesohabitats may not be very effective. In this study, a five cluster classification, partitioned with the Gustafson-Kessel algorithm, was chosen as optimal at each site. This classification was judged to provide the best reflection of underlying geomorphic variations *and* showed how discharge influenced the hydraulic environment at high flows. That is, the optimal value of *c* was a function of longitudinal topographic variation, lateral topographic variation and temporal hydraulic variation caused by changes in discharge.

Reference to the scatterplots of hydraulic data distribution from each individual discharge survey, along with knowledge of the study-site, in this case a field sketch of geomorphic features, photos of variations in surface flow type and the spatial variability of depth and velocity, were used to select the optimal classification.

3.6 Conclusion

In his description of land mosaics Forman (1995, p.4) argues that, "*Spatial heterogeneity occurs in two flavours. A gradient or series of gradients has gradual variation over space in the objects present. Thus a gradient has no boundaries, no patches and no corridors, but is still heterogeneous. A portion of a moist tropical rainforest is an example where the assemblage of tree species changes gradually over the land. But gradient landscapes are rare. The alternative form of spatial heterogeneity is a mosaic, where objects are aggregated, forming distinct boundaries"*. The hydraulic environment of lowland rivers is a rare gradient landscape – depth and velocity vary gradually in space in response to changes in underlying substrate and bed morphology (exceptions do sometimes occur). We impose classifications onto this hydraulic gradient to reduce its complexity for mapping, so a classification method that is data-driven and can be mined to distinguish the patch:boundary ratio, as fuzzy cluster analysis can, is ideal for hydraulic environments.

Fuzzy cluster analysis can be used to generate a quantitative summary of the complex and continuous hydraulic environment. Where other assessment methods rely on visual assessment of pre-defined generic classes, such as channel geomorphic units (Hawkins *et al*., 1993) or hydraulic biotopes (Padmore, 1997), cluster analysis delineates hydraulic patches numerically from actual data, thereby providing an accurate reflection of site-specific conditions. In this study, point measurements were grouped to define relatively homogeneous, spatially coherent hydraulic patches defined by the joint distribution of depth and velocity. The optimisation cluster algorithms used in this study performed well because they maximise within-patch homogeneity and between-patch heterogeneity for any given combination of clustering parameters, thus ensuring a non-arbitrary classification, even when the input data is continuous. Whilst the numerical, data-driven approach of cluster analysis ensured quantitative delineation, user-decisions regarding the defuzzification rule and selection of the optimal classification introduced a degree of subjectivity into the process. However this allows the user to compare different classifications and select the one most fit for purpose.

This study classified the hydraulic environment of three reaches from lowland rivers with pool-riffle / run-glide morphology with different levels of physical heterogeneity and found that all three reaches were optimally classified by five hydraulic patch types. This can be explained by a combination of three factors; the broad similarity in the shape of the combined hydraulic data distribution at each site despite differences in the hydraulic range, the similarity in the degree of shift in the data distribution between very low and very high flow and the fact that optimisation clustering algorithms maximise inter-patch heterogeneity which distributes cluster centroids across the full range of the data distribution. On this basis it is reasonable to expect that a differently shaped data distribution would be optimally classified by a different number of clusters. Further research would be needed to establish the optimal number of clusters in different reach morphologies.

It is acknowledged that further research is needed to confirm the ecological significance of hydraulic patches and transition zones delineated by numerical classification for this approach to be used to strengthen the link between hydromorphology and ecology. Do these hydraulic patches function as hydraulic habitats? It is possible that transition zones might function as in-stream ecotones and support higher levels of biodiversity (Naiman *et al*., 1988; Kark & van Rensburg, 2006). Sampling biological communities along gradients spanning several hydraulic patches and transition zones may help to determine how wide transition zones are in ecological terms and provide further guidance for establishing a suitable defuzzification rule/threshold.

"Variety's the very spice of life…"

William Cowper, The Task, 1785

4

Hydraulic heterogeneity: patch richness, frequency and diversity

Chapter overview

Chapter 4 is the first of two chapters quantifying the spatio-temporal heterogeneity of the hydraulic environment. In this Chapter the first two 'levels' of heterogeneity identified in Cadenasso et al.'s (2006) framework of biocomplexity (Figure 1.9, Chapter 1, p.29) are quantified, i.e. ***patch richness*** *and* ***patch frequency****, typically described as composition. These two aspects can be converted to an index of* ***patch diversity*** *which summarises the proportional abundance of patch types in the reachscape. A comparison and evaluation of flow-related changes in the composition of hydraulic patch types and a discussion of the influence of reach scale geomorphic diversity at each site is also presented. The third, fourth and fifth levels of heterogeneity (sensu Cadenasso et al., 2006) – patch configuration (spatial arrangement), patch change and the shifting mosaic – are examined in Chapter 4.*

4.1 Current understanding and outstanding research questions

The composition of hydraulic patches in a reachscape is an indicator of the range of hydraulic conditions available to biota at a given time. Reachscape composition varies temporally as a result of changes in flow (Wiens, 2002). Evaluating the change in composition under different flow conditions indicates how stable the available range of hydraulic conditions is. Of all the elements of in-stream heterogeneity, composition has been studied most extensively. This section summarises the results of research into the effects of flow, and its interaction with morphology, on a wide range of mesohabitat types, beginning with the overview presented in Table 4.1, and then establishes objectives and hypotheses for the chapter.

In an investigation of the effect of flow regulation in the River Murray Thoms et al. (2006) reported that the composition of different velocity classes at the reachscape varied more between sites than between flows, suggesting that site-specific morphological factors were more important than flow stage, although inter-site differences were reduced at high flows. Maddock et al. (2005) mapped the composition and distribution of CGUs in an unregulated and regulated reach in the Soča River, Slovenia, to ascertain the impact of greatly reduced flow downstream of a dam. Not surprisingly, the regulated reaches had a higher proportion of slow-flowing, non-turbulent CGUs (44-76% pools) whereas the unregulated reach contained 55% glides, the remainder being composed of runs, riffles and rapids. Further work carried out at different discharges revealed two key changes in CGU composition; the replacement of riffles with runs as discharge increased and the tendency for CGU diversity and evenness to peak at intermediate flows (Maddock et al., 2008).

Table 4.1. Effects of flow on physical habitat composition.

Flow-related change	**Impact on composition**	**Author(s)**
Regulated reaches	Dominated by slow-flowing, non-turbulent CGUs	Maddock et al., 2008
Baseflow to spring spate	Change of dominance from riffles to pools	Hilderbrand et al., 1999
Floods	Habitat composition relatively stable across flows	Arscott et al., 2002
Floods	Increased uniformity	Padmore, 1997; 1998
Intermediate flows	Biotope diversity maximised at a range of intermediate flows (Q25-90) at 11 sites	Newson & Newson, 2000
Low-high flow	Change in dominant SFT from rippled to unbroken standing waves	Principe et al., 2007
Low flows	Loss of ‘races’ (runs); decline in habitat diversity	Reuter et al., 2003

Newson & Newson (2000) showed that physical biotope diversity generally increased at low flows in rivers in north east England, with the exception of lowland silt-clay channels which had uniformly low diversity across all flows. Observations of biotope sequencing before, during and after a flood indicate a similar trend, with uniformity and dominance of rapids merged with deep runs during the flood giving way to a more diverse, distinct range of biotopes at moderate flows (Padmore, 1997). Further temporal research identified threshold discharges associated with biotope sequence changes (Padmore, 1998). Principe et al. (2007) also reported strong seasonal variation in proportional abundance of hydraulic biotopes in Argentinean mountain streams.

A comparison of hydraulic biotope assemblages in natural, rural streams and modified urban streams in North Carolina, US, showed urban streams were more homogeneous and dominated by pools whereas rural reaches were dominated by runs and glides (Shoffner & Royall, 2008). Clifford et al. (2002; 2006) described the hydraulic patch structure in a semi-engineered river at low flow as diverse and heterogeneous, becoming increasingly homogeneous and linear at high flows. Dyer and Thoms (2006) conducted a thorough analysis of the proportion, diversity and distribution of surface flow types (SFTs) in the Cotter River, Australia across 18 discharges at three spatial scales. Different patterns emerged at different scales and compositional change appeared to be triggered by discharge-related thresholds. The most distinct patterns occurred at the reach scale. Here, smooth boundary turbulent flows and ripples were the dominant surface flow types at all discharges, but the relative proportion of each became much more even when discharge exceeded 60ML/day. Diversity of SFTs gradually increased over the range 50-130ML/day, as a result of the proportional increase of minor flow types. Although discharge-thresholds for compositional change were identified, the changes did not follow a predictable relationship (Dyer & Thoms, 2006; Thoms et al., 2006), and the work negated to include flow exceedance percentiles to put the flows into context.

Hilderbrand et al. (1999) undertook a basic spatio-temporal survey of pool-riffle sequences at two discharges in a small Virginian stream. They found 50% more pools covering 33% more surface area at base flow conditions than at high discharge, however on average pools were 23% smaller. Riffles covered 56% less surface area but were greater in number because they were interspersed with pools. Although indicative of temporal changes the study is relatively crude, giving little spatially explicit information and using visually assessed data at a single spatial scale (determined by the choice of habitat classification) over only two discharges. Harby et al. (2007) assessed mesohabitat composition at low ($10m^3/s$) and high ($70m^3/s$) discharge in a bypassed section of the Rhone River, France. Habitat diversity increased at high flow, although this may have reflected the increase in wetted area. Deep, slow pools dominated at each discharge, although to a lesser degree at high flow, as would be expected resulting from discharge-induced velocity increases. Contagion index decreased at high flow, suggesting a reduction in clumping or aggregation of habitat types. Arscott et al.'s (2002) assessment of the temporal dynamics of floodplain habitat structure in a braided river system in Italy revealed a change in location of habitats after each flood event, but very little variation in habitat composition.

Concluding a review of spatio-temporal heterogeneity in river corridors, Ward et al. highlight the "lack of fundamental knowledge of their natural complexity and dynamics" (2001, p.321). Since then a modest number of studies have attempted to address this gap, most notably Arscott et al.'s (2002) study of floodplain dynamics in the Tagliamento River. Despite the proliferation of new tools and technologies for spatio-temporal analysis (GIS, spatial statistics, boundary statistics, landscape ecology metrics), application to the hydraulic environment remains relatively unexplored. Clearly further research is needed quantify the dynamics of the in-stream hydraulic mosaic (Newson & Newson, 2000).

Based on the knowledge gaps identified above, the objectives of chapter are to: (a) investigate how the composition of hydraulic patches in each study reach changes in response to (seasonal) variations in discharge; and (b) to what extent the site-specific morphology influences/controls compositional change? It was hypothesised that hydraulic patch composition would differ significantly between low and high flows.

4.2 Methods

The hydraulic patch classification at each flow was stored as a categorical raster surface in ArcGIS then exported to FRAGSTATS spatial pattern analysis program (McGarigal & Marks, 1995) for further analysis. FRAGSTATS was used to calculate three measures of reachscape composition at each flow; the proportion (% contribution) of each hydraulic patch type present, hydraulic patch diversity and patch richness density. Hydraulic patch diversity was calculated using Shannon's Diversity

Index (H')[1], substituting hydraulic patch types for species. The value of the index increases as the proportional abundance of patch types becomes more equitable. An illustration of reachscapes supporting relatively low and high values of patch diversity is shown in Figure 4.1. The maximum value of the index (natural log of the number of 'species') was 1.79 at each site. Relative differences in the index are useful for comparing each reachscape at different flows. Differences in hydraulic patch diversity (Shannon diversity index, H') between discharges were tested using pairwise permutation tests (performed in R 2.14.0). For each pairwise test the two samples (A, B) were pooled. 1000 random pairs of samples (A_i, B_i) were then taken from this pool with replacement, with replicate random pairs having the same sum of hydraulic patch type counts as in the original two samples. Diversity indices $H'(A_i)$ and $H'(B_i)$ were computed for each permuted pair. The number of times $H'(A_i)-H'(B_i)$ exceeded or equalled the observed difference in H' indicates the probability that the observed difference could have occurred at random (M Wilkes, unpublished data). The code and resulting pairwise p-values are presented in Appendix E.

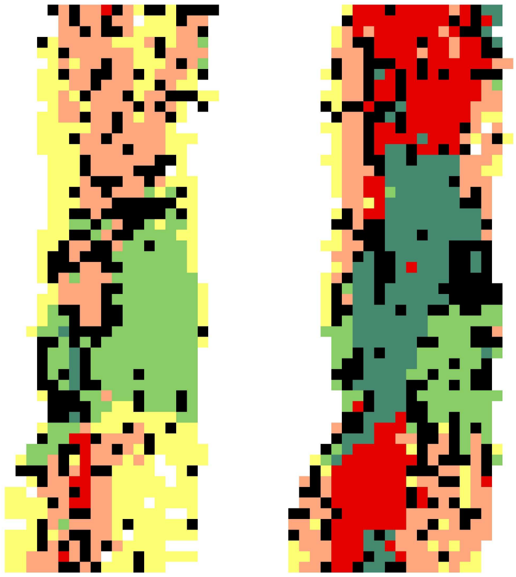


Figure 4.1 Example reachscapes with relatively low (left) and high (right) hydraulic patch diversity (H').

To explore hydraulic patch diversity at a local scale and evaluate its spatial variability, patch richness density (PRD) (number of patch types per unit area) was calculated at

---

[1] $H' = -\sum_{i=1}^{m}(P_i \ln P_i)$, where P_i is the proportion of the reachscape occupied by patch type i.

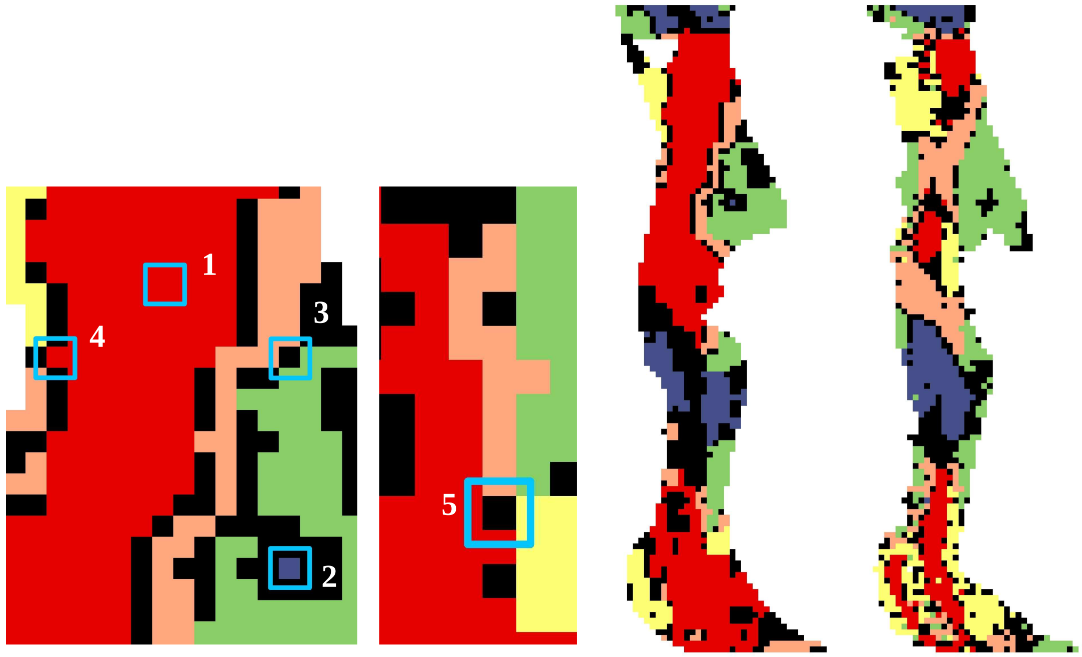
each measurement location. This was performed by passing a 1m x 1m window (actual size, including the focal cell, was $2.25m^2$) over each cell in the reachscape and calculating the number of patch types in the window and converting to the number of patch types per m^2. Areas of the reachscape where the window was not completed filled with cells (e.g. near the reach boundary or exposed substrate), were excluded from the analysis to prevent the patch types present a part-filled window biasing PRD values. Hence PRD was not calculated in a small proportion of the reach (near water's edge and around exposed substrate). As there were six hydraulic patch types (including the Transition Zone) at each site and the moving window was $2.25m^2$, in theory PRD could take one of six values (0.44, 0.89, 1.33, 1.78, 2.22 and 2.67), with a value of 0.44 indicating local homogeneity and a value of 2.67 indicating maximum local heterogeneity. In practice the maximum value did not occur at any site. Examples of the patch structure resulting in the first five values and examples of reachscapes supporting relatively low and high mean values of PRD are illustrated in Figure 4.2. The spatial variation of PRD values at each site-flow combination was then mapped in ArcGIS, using the value of PRD calculated for each focal cell. From this information additional maps were created using weighted sum overlays to show areas of the channel where PRD increased, decreased or stayed the same with each increase in discharge.

In all FRAGSTATS analyses patch neighbours were defined by the 8-cell rule which, in a 3 x 3 cell matrix, treats all 8 cells around the central cell as neighbours, whether orthogonally or diagonally adjacent to it. The transition zone was included as a patch type in all analyses.

4.3 Results

4.3.1 Proportional abundance of hydraulic patch types

Flow increases were associated with changes in the proportional abundance of hydraulic patch types at the reachscape scale at every site (Figures 4.3 to 4.5). At the River Arrow all five hydraulic patch types were present at each flow but in varying proportions. At very low ($0.21m^3/s$, Q87), low ($0.30m^3/s$, Q70) and intermediate ($0.42m^3/s$, Q53) flows, the reach was dominated by RA3 (moderate–very slow) (26-



mean PRD=0.79 mean PRD=1.03

Figure 4.2 Illustration of different patch structures in the moving window analysis (left) where the value of PRD is (1) 0.44 HP types/m^2 (homogeneous), (2) 0.89 HP types/m^2, (3) 1.33 HP types/m^2, (4) 1.78 HP types/m^2 and (5) 2.22 HP types/m^2. In practice there were no instances of 2.79 HP types/m^2 (maximum heterogeneity). To the right are two example reachscapes supporting relatively low and high mean PRD.

32%) and RA2 (shallow-slow) (24-27%) (Figure 4.3). As discharge increased and the distribution of depth-velocity measurements shifted towards deeper, faster conditions (Chapter 3), so the proportion of shallow-slow hydraulic patches was gradually replaced, firstly by RA1 (moderate-slow) and then by RA4 (moderate-fast) patch types. The proportion of RA1 (moderate-slow) peaked at 19% at intermediate flow (0.42m^3/s, Q53) but decreased thereafter as velocity increased. The proportion of RA4 (moderate-fast) increased exponentially with discharge from 0.2% to 40%, reflecting the widespread increase in velocity in the reach at very high flow (1.41m^3/s, Q13). By contrast, RA5 (very deep-very slow) occupied a small but stable proportion of the reach (8-10%) at all flows, whereas the Transition Zone occupied a significant but fluctuating proportion (19-28%) of the reach. The smallest difference between proportional abundance of all the patch types occurred at high flow (Q22) where percentage contribution ranged between 10-20% for each hydraulic patch type.

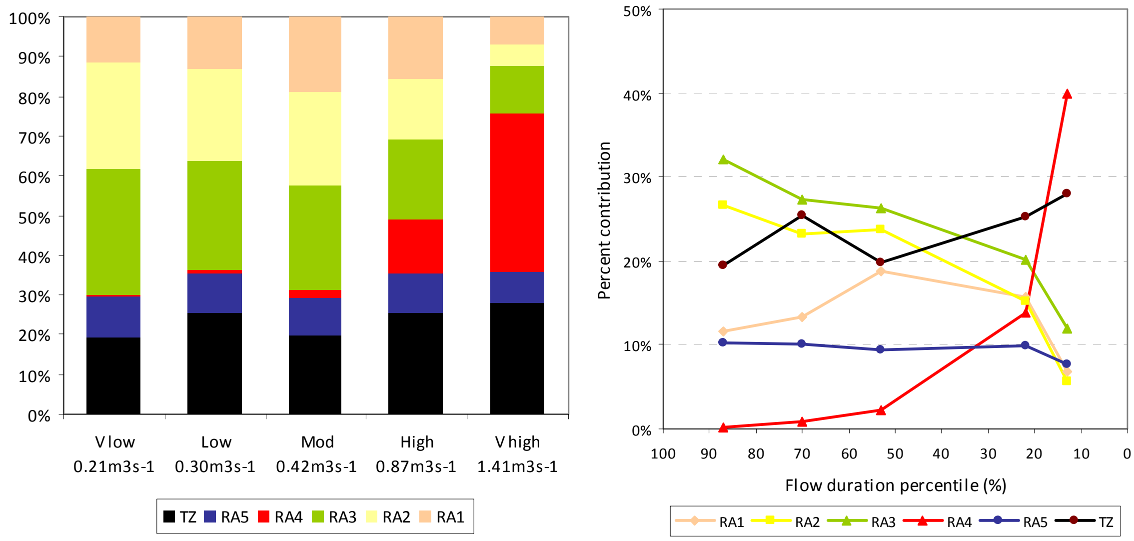


Figure 4.3. Discharge-related variation in reachscape composition, represented by the proportion of hydraulic patch types (left) and the change in percentage contribution of each hydraulic patch type (right) at the River Arrow study reach.

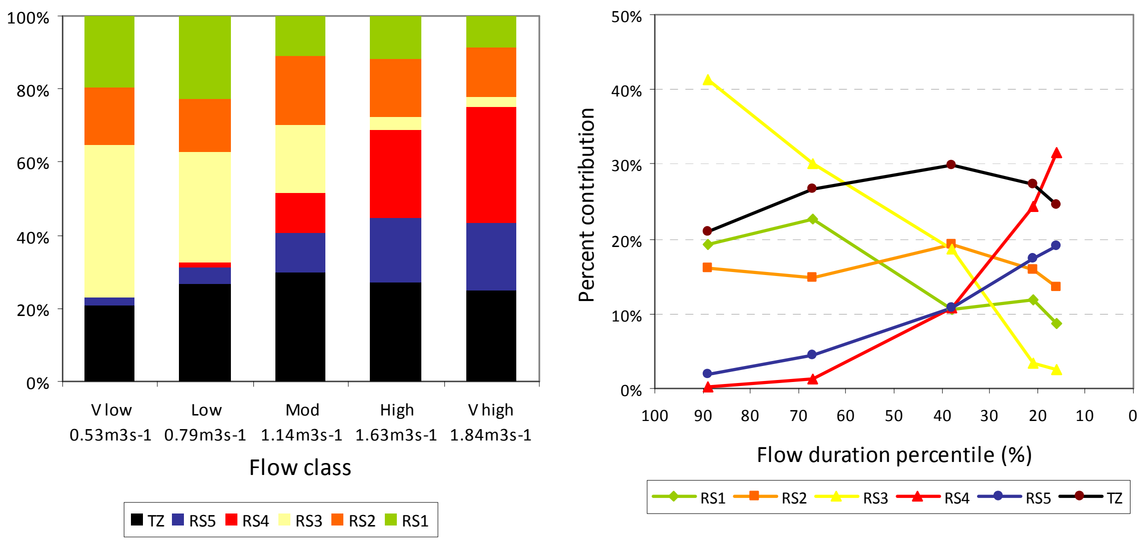


Figure 4.4. Discharge-related variation in reachscape composition, represented by the proportion of hydraulic patch types (left) and the change in percentage contribution of each hydraulic patch type (right) at the River Salwarpe study reach.

At the River Salwarpe all five hydraulic patch types were also present at every flow and the hypothesised change in proportional abundance of these hydraulic patch types occurred (Figure 4.4). For example, RA3 (shallow-slow) dominated the reach at very low flow (41% at $0.53m^3s^{-1}$, Q89) but its proportion declined sharply with every increase in discharge, occupying just 3% at very high flow ($1.84m^3s^{-1}$, Q16). As expected the proportion of deeper, faster patch types (RS4 and RS5) increased with flow; RS4 (moderate-fast) increased exponentially with flow from 0.2-31.5% whereas RS5 (deep-moderate) increased more steadily with flow from 2-19%. The proportion of RS2 (shallow-fast) fluctuated between 14%-19% and was relatively insensitive to changes in flow. The proportional abundance of the Transition Zone increased with discharge from 21% to its peak of 30% at moderate flow ($1.14m^3s^{-1}$, Q38) but decreased with every increase in discharge thereafter.

Variations in flow, particularly between very low (0.24 m^3s^{-1}, Q87) and moderate (0.52 m^3s^{-1}, Q54), had a large effect on the proportional abundance of hydraulic patches at the Leigh Brook site (Figure 4.5). Only three patch types were present at very low flow; LB2 (shallow-slow) (48%), LB5 (moderate-slow) (26%) and LB3 (shallow-moderate) (9%). The proportion of LB2 (shallow-slow) decreased sharply with flow from 48% to 6% at very high flow (1.32 m^3s^{-1}, Q12). LB5 (moderate-slow) remained relatively stable (21-26%) at low to moderate flows but decreased at high flows to between 9-11%. The proportion of LB3 (shallow-moderate) tripled between very low and low flow to 27%, replacing LB2 (shallow-slow), but decreased moderately with each subsequent increase in flow. LB1 (moderate-moderate) and LB4 (moderate-fast) first appeared at low flow, occupying <2% of the reach each. Abundance of both patch types was sensitive to flow and by very high flow LB1 and LB2 dominated the reach, occupying 26% and 22% respectively.

4.3.2 Hydraulic patch diversity

The Shannon diversity index (H') was calculated to summarise the proportional abundance of all hydraulic patch types at the reach scale. Figure 4.6 shows that reachscale hydraulic patch diversity tended to increase steadily from its value at very low flow, peak at moderate or high flow, then decrease at high flow and or very high flow. The sensitivity of hydraulic patch diversity to increases in flow, represented by the gradient of the line, differed between sites. The smallest range of H' (0.23) but also the highest value (1.75) occurred at the River Arrow. Here hydraulic patch diversity was relatively insensitive to flow increases between very low ($0.21m^3s^{-1}$, Q87) and moderate flows (0.42 m^3s^{-1}, Q53) but it increased significantly at high flow

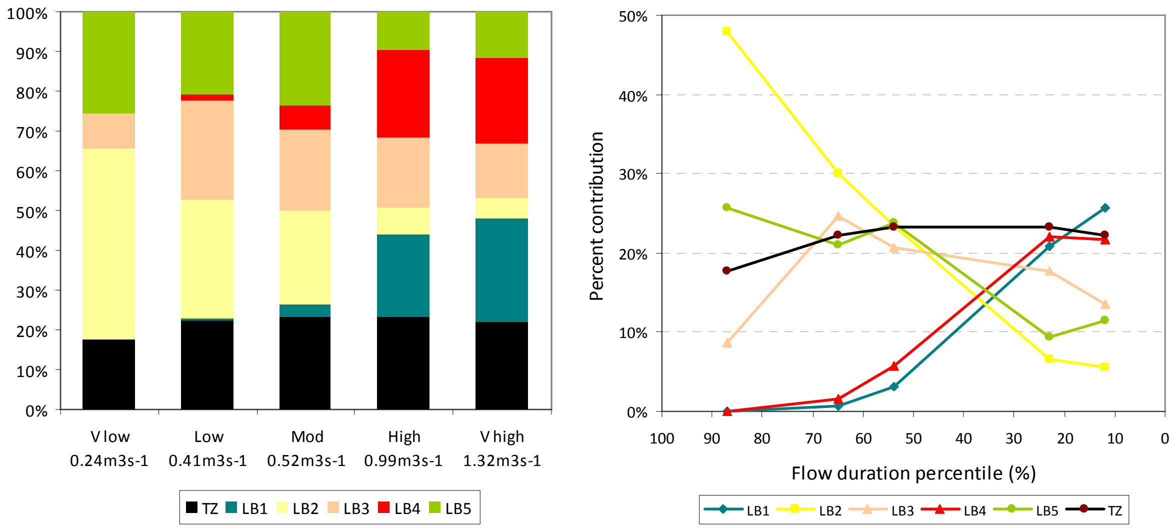


Figure 4.4. Discharge-related variation in reachscape composition, represented by the proportion of hydraulic patch types (left) and the change in percentage contribution of each hydraulic patch type (right) at the Leigh Brook study reach.

(0.87m^3s^{-1}, Q22) very sensitive to the increase between high and very high (1.41m^3s^{-1}, Q13) flow, when H' decreased significantly ($p<0.05$) by 13% to its lowest level at this site. Hydraulic patch diversity (H') at the River Salwarpe had a higher range (0.31) and showed greater sensitivity to increases in flow than the River Arrow, rising by 18% between very low (0.53 m^3s^{-1}, Q89) and moderate (1.14 m^3s^{-1}, Q38) flow, with the increase in patch diversity between low and moderate flow being statistically significant ($p<0.05$). Each increase in flow thereafter resulted in a 3-4% decrease in H', neither of which was statistically significant ($p>0.05$). The Leigh Brook had highest range of H' (0.49) and also the lowest hydraulic patch diversity of all sites at very low flow due to the absence of two hydraulic patch types. Hydraulic patch diversity increased significantly by 20% with the first increase in flow and by a lesser but still significant degree with each subsequent increase in flow until H' peaked (1.71) at high flow. Hydraulic patch diversity remained relatively stable at very high flow. The results of all tests of significant differences in hydraulic patch diversity at different flows are included in Appendix E.

4.3.3 Patch richness density

Patch richness density (PRD) provides a measure of patch diversity (richness only) at a local scale, in this case the number of patch types in a 1.25m^2 area around each focal cell in the reachscape (subject to the boundary limitations outlined in Section 4.3). Figures 4.7, 4.9 and 4.11 illustrate the spatial variation of PRD at each site-flow combination and Figures 4.8, 4.10 and 4.12 illustrate where PRD increased, decreased or stayed the same with each increase in flow at each site. Figure 4.13 shows the mean PRD values to aid inter-site comparison.

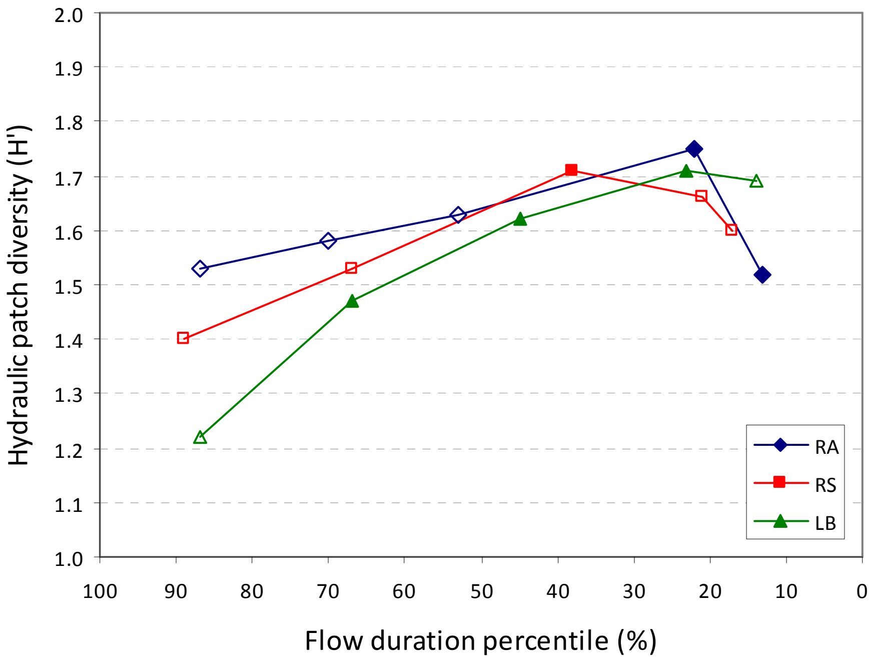
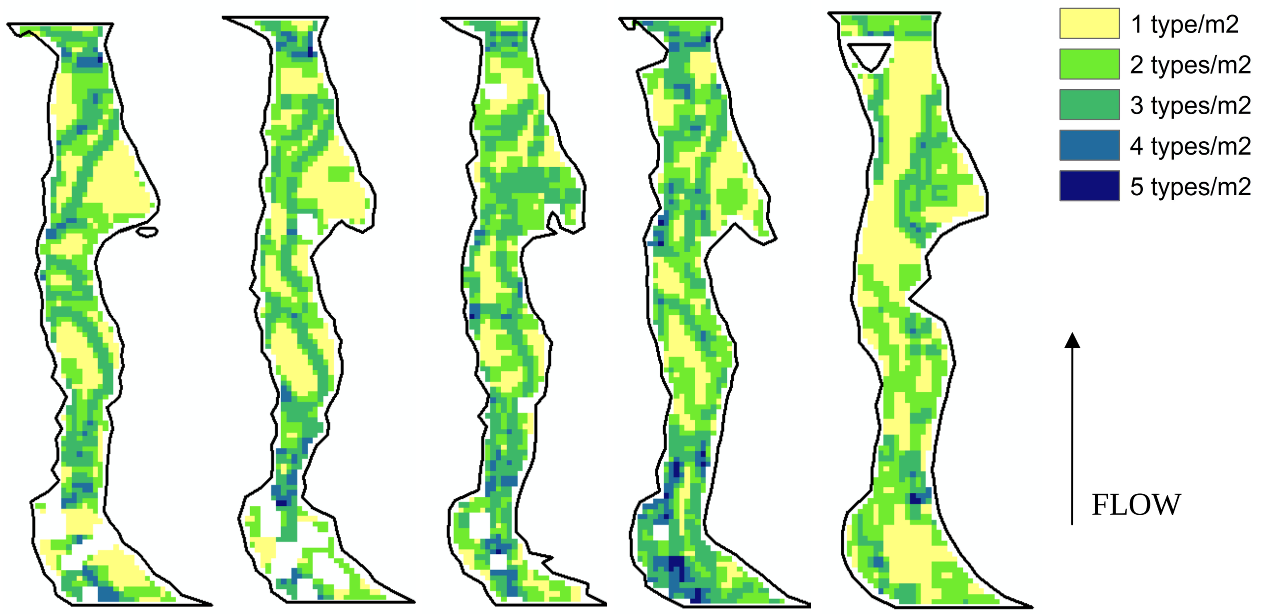


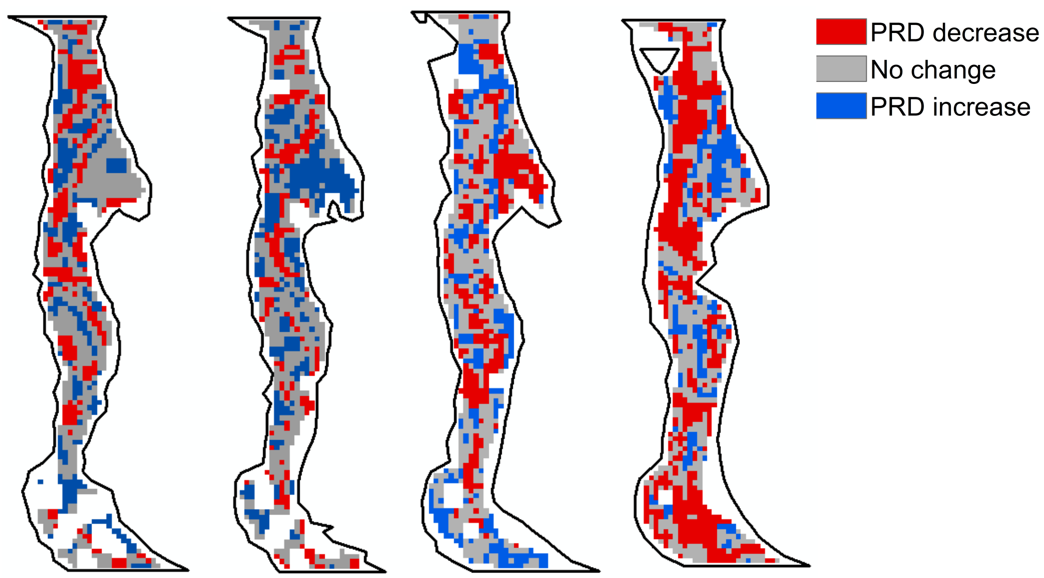
Figure 4.6. Variation of hydraulic patch diversity across all sites and flows. Solid markers indicate a significant change in H’ ($p<0.05$) from the previous flow.

At the River Arrow PRD was low-moderate (≤3 HP types/m^2) throughout the majority of the reach at all flows (Figure 4.7). At very low and low flow minimum PRD (1 HP types/m^2) occurred in the deepest, slowest areas (topographic lows/deadwater) whereas the highest PRD values were associated with areas immediately upstream of pools with steep gradients and submerged vegetation. At moderate flow PRD increased in 30% of the reach, most noticeably in the recirculation zone and margins adjacent to increased PRD in the thalweg (Figure 4.8). At high and very high flow the areas associated with low PRD were those associated with high PRD at low flows and vice versa. That is, the thalweg supported 1-2 different patch types whereas channel margins and deeper areas of the reach were associated with a high density of patch types. The biggest change in PRD occurred between high and very high flow when PRD increased in 44% and decreased in 17% of the reach respectively (Figure 4.8).



0.21m^{3}s^{-1}, Q87 | 0.30m^{3}s^{-1}, Q70 | 0.42m^{3}s^{-1}, Q53 | 0.87m^{3}s^{-1}, Q22 | 1.41m^{3}s^{-1}, Q13

Figure 4.7. Spatial variation of patch richness density at each surveyed flow, River Arrow.



0.30m^{3}s^{-1}, Q70 | 0.42m^{3}s^{-1}, Q53 | 0.87m^{3}s^{-1}, Q22 | 1.41m^{3}s^{-1}, Q13

Figure 4.8. Change in patch richness density with each increase in discharge, River Arrow.

The Leigh Brook reach supported a different spatial and temporal pattern of PRD (Figure 4.9). Here high PRD (≥3 HP types/m^2) occurred in the majority of the reach at very low flow. In contrast to the River Arrow, the deepest part of the Leigh Brook reach was a small scour pool associated with very variable velocity which therefore supported a high density of patch types (1-4 HP types/m^2). Lowest PRD values occurred in a shallow pool adjacent to the right bank. As discharge increased from low to moderate PRD decreased in 19% of the reach (Figure 4.10), increasing the area of low PRD values in the shallow pool adjacent to the right bank and along the thalweg. A marked pattern of change in PRD occurred at high flow (Figure 4.10), with PRD decreasing along the thalweg and increasing in areas hitherto associated with low PRD. This trend continued at very high flow. However overall the Leigh Brook reach supported the highest mean PRD values of all three sites at each flow (Figure 4.13).

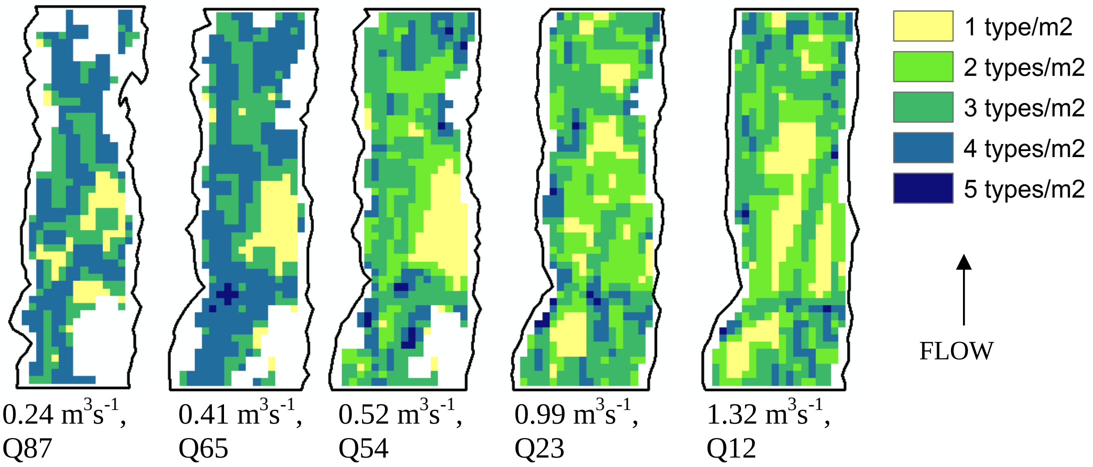


Figure 4.9. Spatial variation of patch richness density at each surveyed flow, Leigh Brook.

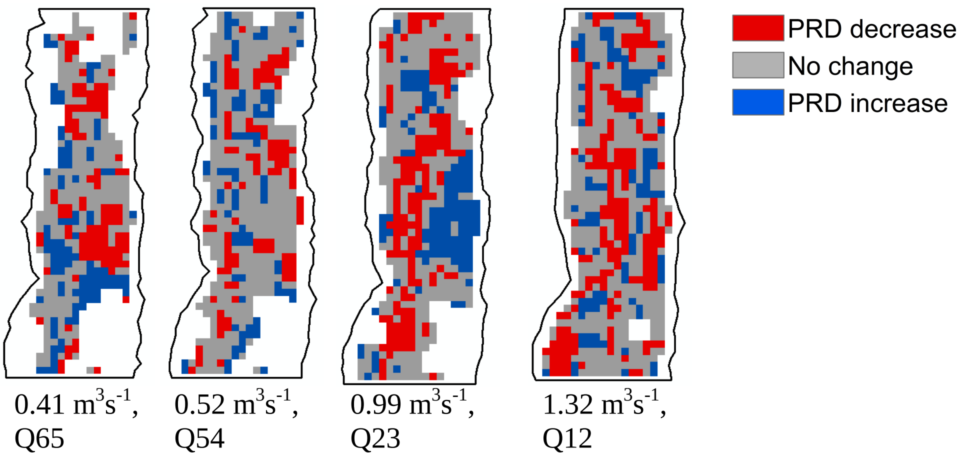


Figure 4.10. Change in patch richness density with each increase in discharge, Leigh Brook.

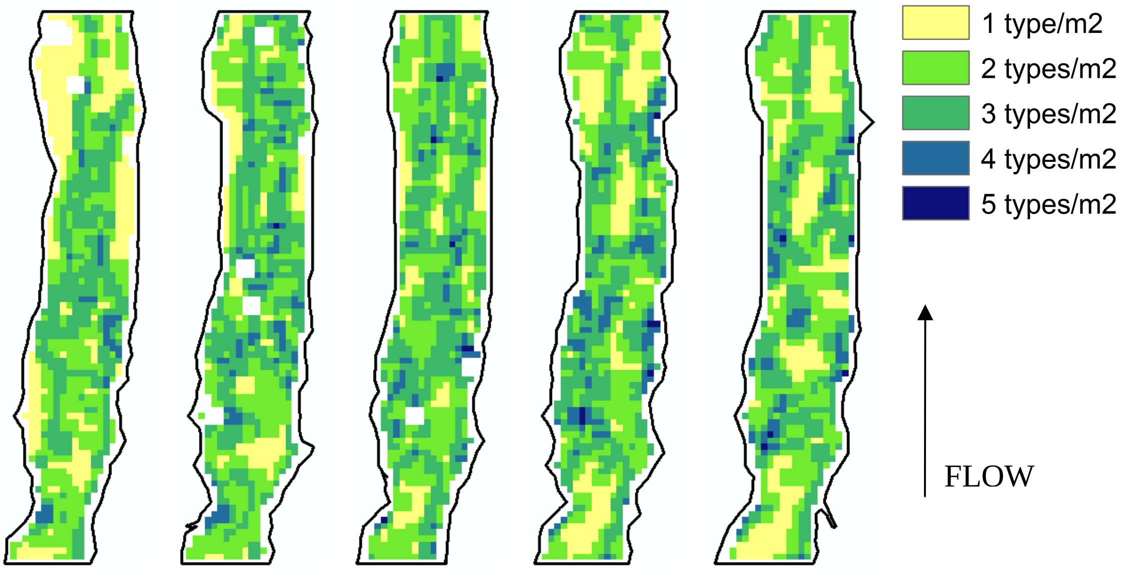


Figure 4.11. Spatial variation of patch richness density (PRD) at each surveyed flow, River Salwarpe.

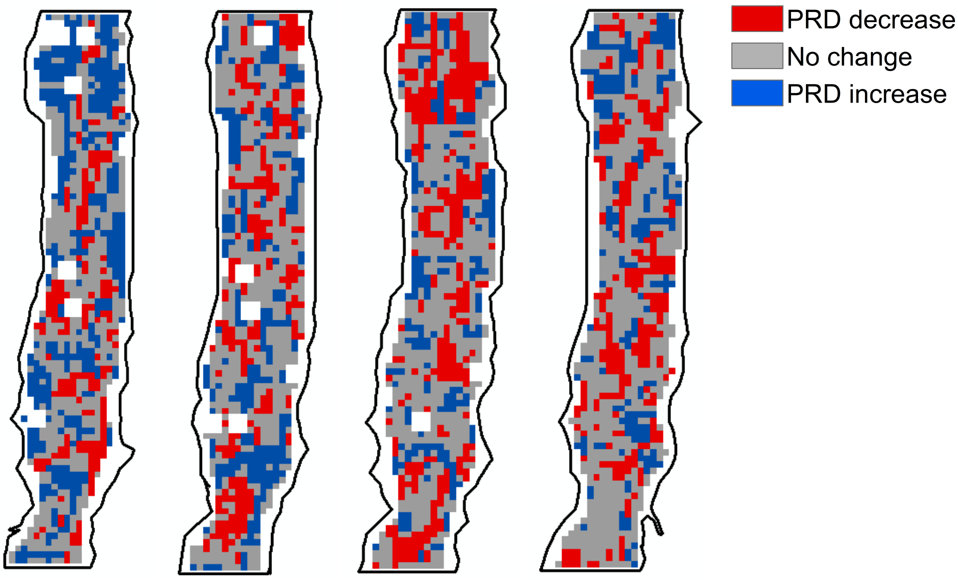


Figure 4.12. Change in patch richness density with each increase in discharge, River Salwarpe.

The River Salwarpe supported the most equitable proportion of PRD values at each flow of all three sites, indicating that there were homogeneous and heterogeneous areas of the reach (Figure 4.11). At very low flow minimum PRD values (1 HP type/m^2) occurred mainly in barely-inundated areas of deposition or at channel margins, although PRD increased in these areas as discharge increased (Figure 4.12). At low and moderate flows PRD was spatially heterogeneous throughout the reach;

lower values (≤3 HP types/m^2) typically occurred in the upstream third of the reach (Glide) and higher values (≥3 HP types/m^2) occurred in the remainder of the reach (Run). At high and very high flow PRD values decreased in the thalweg (Figure 4.12), similar to the pattern observed in the River Arrow and Leigh Brook reaches.

Figure 4.13 shows how the mean PRD (local scale richness) varied with flow and differed between sites. PRD was relatively insensitive to changes in flow at all three sites, varying by 0.1 HP types/m^2 at the Leigh Brook and by 0.23 HP types/m^2 at the River Arrow. All mean PRD values fell within the range 0.79-1.16 HP types/m^2 but differences between sites were evident. The Leigh Brook and the River Arrow supported the highest and lowest mean PRD of all three sites at every flow respectively, with the River Salwarpe supporting values within this range. Maximum values of PRD occurred at either low, moderate or high flow whilst the flow extremes supported the two lowest values of PRD.

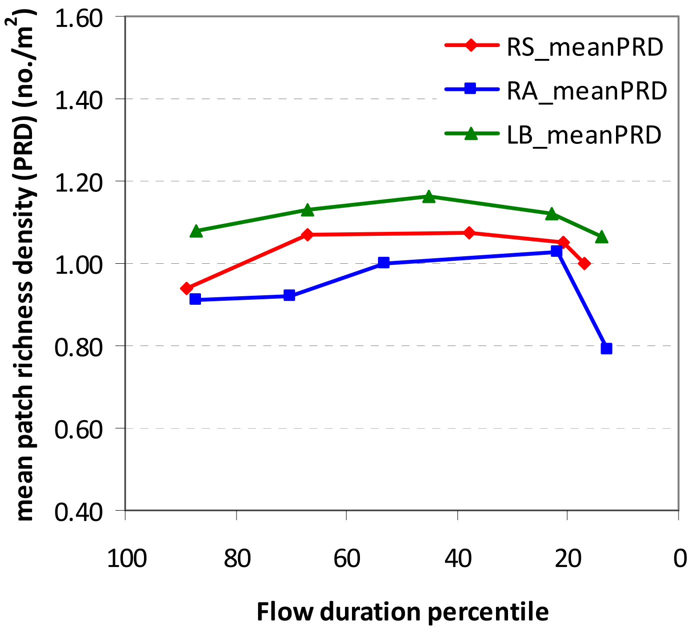


Figure 4.13. Mean patch richness density at the reach scale at each site.

4.4 Discussion

In Chapter 1 it was hypothesised that shallow or slow-flowing patches would dominate at low flow but be replaced by deeper, faster-flowing patches as discharge increased, subject to the influence of channel morphology. This was true of all three study sites (Figures 4.7 to 4.12). Figure 4.14 summarises the flow conditions (very low (≤Q80), low (Q81-Q60), moderate (Q61-Q40), high (Q41-Q20), very high (<Q20)) associated with the dominant presence of each hydraulic patch type and indicates the sensitivity of each type to changes in flow conditions. Most hydraulic patches were dependent on certain flow conditions to exist and their degree of temporal stability was related to how specific that flow condition was. For example, patch types characterised by moderate and fast velocities were dependent specifically on high flow conditions whilst shallow-slow and moderate-very slow depended specifically on low flow conditions meaning all types were very sensitive to changes in flow. Moderate-slow and deep-moderate patches were associated with a wider range of flow conditions (low to moderate and moderate to high respectively) so were less sensitive to changes in flow except the extreme flows most different to their associated flow range. Other patch types (shallow-moderate and shallow-fast) were also prevalent at a range of flow conditions and could persist even at high flows conditions providing the channel morphology supported shallow areas as discharge increased. These were provided by areas of deposition at the River Salwarpe and Leigh Brook (Chapter 2) respectively. Lastly, one hydraulic patch type (very deep-very slow) was dependent on bedform amplitude (deep topographic lows) and was temporally stable under the full range of flow conditions.

Hydraulic patch diversity was highest at the River Arrow and lowest at the Leigh Brook at low flows but at high flows this order reversed, suggesting that different factor(s) influenced diversity at different flows. At low flow and at the reach scale diversity was highest at the River Arrow, where the bedform amplitude supported the largest range of depths, and lowest at the Leigh Brook where bedform amplitude was the lowest of all three sites. The River Salwarpe had an intermediate value of patch diversity but was more similar to the Leigh Brook than the River Arrow, having a bedform amplitude most like the Leigh Brook. However at the local scale patch diversity, as measured by mean PRD, was influenced by mean substrate size, with the largest mean substrate size being associated the highest mean PRD at the Leigh Brook and vice versa at the River Arrow. In this case the River Salwarpe had a mean PRD value closer to the River Arrow, despite its mean substrate size being very similar to that at the Leigh Brook, suggesting that the high substrate heterogeneity impacted on local diversity.

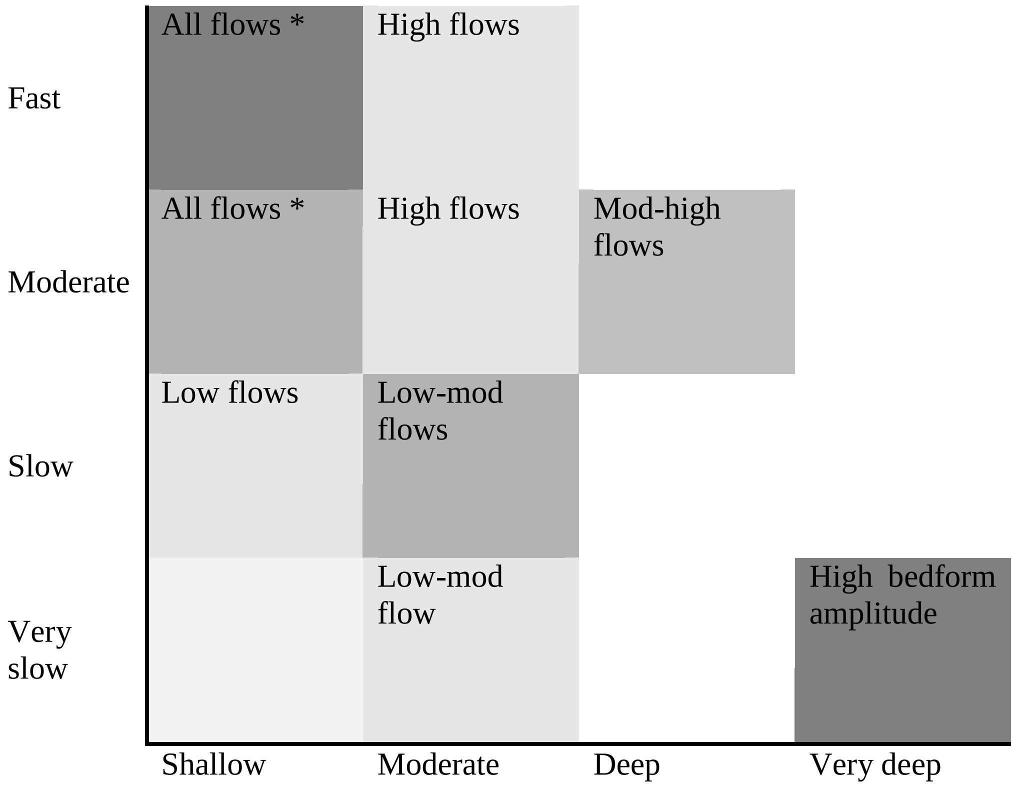


Figure 4.14. Hydromorphological conditions (flow or morphology) associated with the occurrence of each hydraulic patch type. Shading indicates the temporal resilience of HP abundance under different flow conditions, the darker the shading the more persistent the hydraulic patch. The asterisk denotes where temporal persistence at higher flows dependent on the availability of newly-inundated shallow areas elsewhere in the channel.

As discharge increased the range of velocities also increased, expanding the depth-velocity distribution to include faster-flowing hydraulic patch types, and as a result diversity increased. The increase was particularly large and statistically significant at the Leigh Brook where there were more hydraulic patch types that depended on higher flow ranges (Figure 4.14). To summarise, hydraulic diversity fundamentally depends on hydromorphological conditions that support a range of depths and velocities. Depth variation created by morphological diversity was the main predictor of hydraulic patch diversity at low flow. Meso scale bedform amplitude had a strong influence on hydraulic patch diversity at the reach scale, but where the amplitude was not pronounced, diversity was created locally by coarse substrate.

To understand patterns of spatial and temporal hydraulic diversity it is necessary to consider several spatial dimensions of geomorphic variability (cross-section, long-profile and the height of bed roughness) in combination with the temporal variability introduced by flow variations. Patch richness density (local scale hydraulic patch heterogeneity) appeared to be a function of bedform amplitude and mean substrate size. The combination of low bedform amplitude and large substrate created high local hydraulic heterogeneity at the Leigh Brook whereas the high bedform amplitude and smaller mean particle size at the River Arrow resulted in the lowest local heterogeneity, the hydraulic environment being shaped by the meso scale bedforms. Bank angle determines rate of increase in wetted area (i.e. low bank angles enable larger increase, vertical bank angles restrict increases in area to ~0%). Further research is needed to identify the influence of morphology at a range of scales on mesohabitat composition. A measure of channel or bank complexity and gradient may be a useful predictor of the temporal persistence of shallow hydraulic patches at high flows. Unlike bedform amplitude and substrate, the diversity provided by cross-sectional shape varies with flow depending on how much of the channel bed and banks are inundated. As such any indices of cross-sectional variability should be flow-specific. Furthermore, the longitudinal variability of cross-sectional shape should also be accounted for, by reflecting the proportion of the reach occupied by cross-sections of each shape. In theory it would be possible to develop a typology of cross-sectional shape based on ranges of associated shape indices with which to record the proportion of each type in a given reach.

Management Implications

Current standards for hydromorphological condition are based on the allowable levels of deviation from ‘natural’ that are thought not to affect ecological status. These standards were based on expert opinion and further research is needed to understand how hydromorphology influences the hydraulic environment. Figure 4.14 goes some way towards describing how abstraction or channel modification might affect the availability of hydraulic patches. And the results could inform predictions of how hydromorphological alteration (to flow regime or channel structure) might affect habitat availability (i.e. the proportional abundance of hydraulic patches). However, further work is needed to develop a model predicting hydraulic patch diversity to hydromorphological factors as it is not a simple linear one. Using mean measure of cross-sectional variability at bankfull is not satisfactory as cross-sectional variability changes with flow and it is better to reflect the diversity of cross-sectional shape along the reach rather than the mean (in addition to the measure of shape variability of each cross-section itself as measured by the index).

4.5 Conclusion

The composition of hydraulic patches in each study reach responded to variations in discharge. The abundance of hydraulic patch types followed one of four patterns: (1) dependent on a specific flow and highly sensitive to flow variations; (2) occur in a range of flow conditions and less sensitive to variations; (3) dependent on particular bedforms and insensitive to flow variations; and (4) occur at all flows if bank profile is steeper. These results lend themselves well to the development of new predictive models for understanding and managing hydraulic diversity in the context of flow management (e.g. abstraction) and hydromorphological assessment under the WFD.

"Without impermanence, life is not possible"

Thich Nhat Hanh, The Heart of Understanding, 1988

5

Hydraulic patch configuration, patch change and the shifting mosaic

Chapter Overview

Building on the results presented in the previous chapter, this chapter quantifies the third, fourth and fifth levels of heterogeneity described by Cadenasso et al. (2006), i.e., patch configuration (the spatial arrangement of hydraulic patches at the class level), patch change (the spatial dynamics of patches), and the shifting mosaic (the dynamic configuration of all patches in the reachscape). This completes the quantification of hydraulic heterogeneity in the study reaches and forms the basis of the conceptual model of hydraulic patch dynamics presented in Chapter 6

5.1 Current knowledge and research gaps

Standard physical habitat assessment methods typically measure the composition of hydraulic/hydromorphic units (e.g. Dyer & Thoms, 2006; Shoffner & Royall, 2008), but rarely reflect their spatial structure. It has been suggested that this poses a significant limitation to our understanding of the in-stream environment (Newson & Newson, 2000). A summary of research on the configuration of physical habitat is presented in Table 5.1.

Table 5.1. Effects of flow on the geometry and configuration of physical habitats.

Flow-related change	Impact on geometry/ configuration	Author(s)
Regulated reaches	Short, narrow CGUs	Maddock et al., 2005
Low flows	Contraction of patch area	Reuter et al., 2003
Intermediate flows	Biotope patchiness maximised at a range of intermediate flows (Q25-90) at 11 sites	Newson & Newson, 2000
High flow	Longitudinal ribboning of mesohabitats	Clifford et al., 2002; 2006
Flows approaching bankfull	Increased reachscape connectivity	Bertoldi et al., 2009
Floods	Habitat configuration relatively stable across flows (but large changes in composition)	Arscott et al., 2002
Low vs high flow	Significant changes in aggregation and shape of velocity patches	Thoms et al., 2006

The existing literature highlights the trend for a fragmented patchy structure at low flows and a more aggregated, connected structure at high flows. The literature does does not address changes in configuration across a wide range of flows or the configuration of hydraulic patches defined by the joint distribution of depth and velocity. Addressing this research gap will help to further our understanding of the relationship between hydromorphology and hydraulic heterogeneity, including the implications of modifying hydromorphology on the availability of a range of hydraulic conditions.

5.2 Methods

5.2.1 Selection of spatial metrics

The configuration, or spatial pattern, of features in a categorical map, can be defined in terms of the area, shape and density of patches, their relative isolation, proximity or connectivity, their aggregation, interspersion or fragmentation and the contrast between them (Turner et al., 2001). At the class level six spatial metrics were selected to answer specific questions about the impact of flow on patch configuration listed in Table 5.2. A full description of each metric is provided in Table 5.3.

Table 5.2. Metrics selected to quantify patch configuration (class level)

Questions	**Metric**
Do patches shrink/grow?	AREA_mean
Does patch length increase/decrease?	GYRATE_mean
Does patch shape complexity increase/decrease?	FRAC_mean
Do patches move closer together/further apart?	ENN_mean
	PROX_mean
Does patch distribution become more or less aggregated?	CLUMPY

At the reach scale an initial selection of twelve configuration metrics describing aspects of the area, density, shape, isolation/proximity, contagion/interspersion and connectivity of hydraulic patches was made, which was reduced to five uncorrelated metrics (Table 5.3 & Section 5.2.2). Of these metrics, the Connectance Index (Table 5.3, p. 166) required further user-specified information prior to calculation; the user must choose a threshold distance within which same-type patches are identified as being connected. In a typical landscape ecology study a distance relevant to the species under investigation would be used, for example the distance an organism is able to travel (e.g. Lindsay et al., 2008; Ziółkowska et al., 2012). In this (non species-specific) study, the threshold distance was scaled to physical factors instead. The Connectance Index was calculated for a range of threshold distance values between the minimum distance over which two separate patches could be connected (1m) and half the reach length at each very low, moderate and very high flow at each site. Scree plots were assessed for break in slope to guide the threshold distance used at each site. However the scree plots did not show a consistent break in slope at all flows (Figures 5.1-5.3). To avoid inconsistency and aid direct comparison between flows half the reach length was used as the threshold distance for all flows at each site. This was 28m at the River Arrow, 22.5m at the River Salwarpe and 13m at the Leigh Brook.

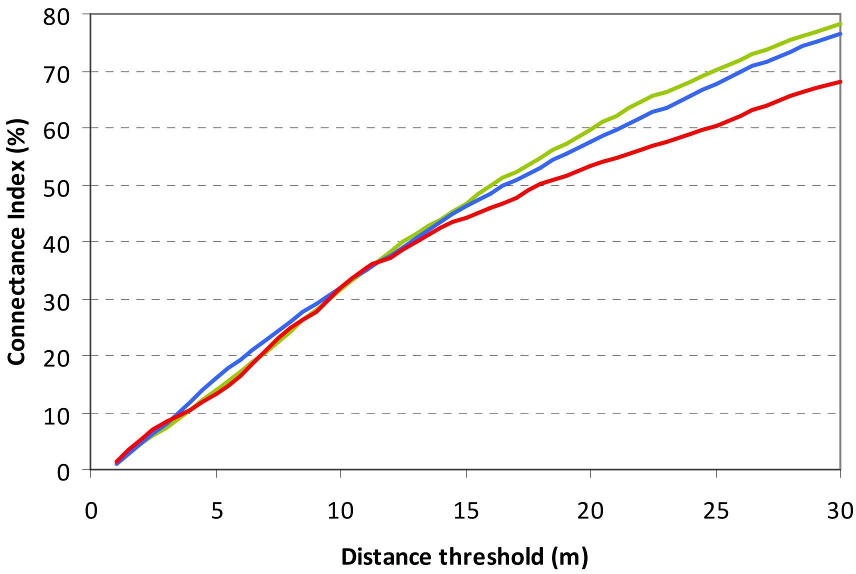


Figure 5.1. Connectance Index value at metre interval threshold distances at very low (green), moderate (blue) and very high (red) flow, River Arrow.

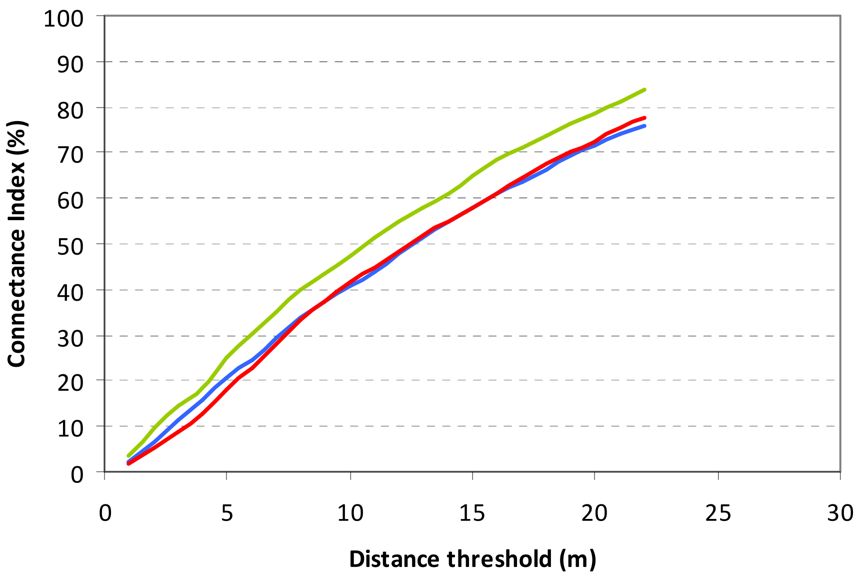


Figure 5.2. The value of the Connectance Index at metre interval threshold distances at very low (green), moderate (blue) and very high (red) flow, River Salwarpe.

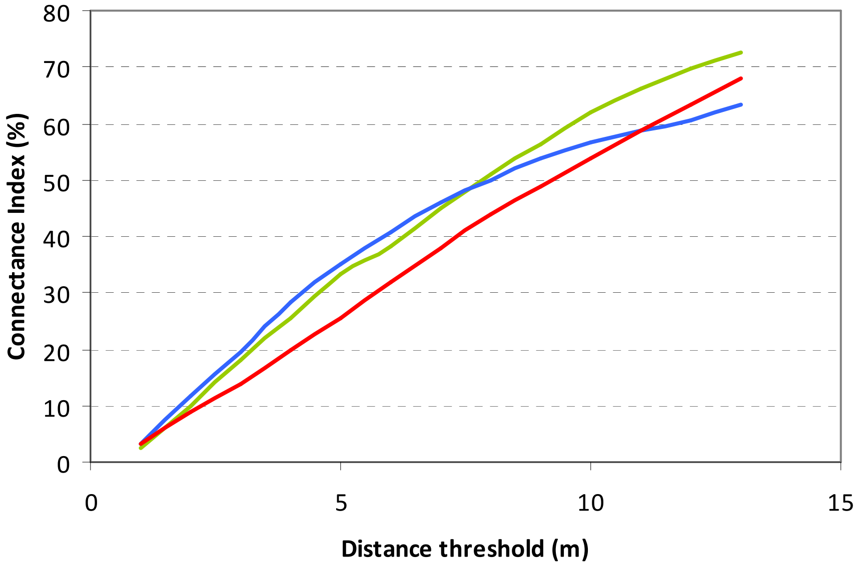


Figure 5.3. The value of the Connectance Index at metre interval threshold distances at very low (green), moderate (blue) and very high (red) flow, Leigh Brook.

5.2.2 Calculation of spatial metrics

Hydraulic patch classifications, delineated at each site-flow combination using FCA (Chapter 3), were exported as ASCII categorical maps from ArcGIS (ESRI, 2008) and imported into FRAGSTATS v3.3 spatial pattern analysis software (McGarigal & Marks, 1995). The selected metrics were then calculated at the appropriate scale (class or reachscape). In each case the 8 cell rule was used to define patch neighbours. That is, in a 3 x 3 cell matrix, all 8 cells around the central cell were treated as neighbours, whether immediately or diagonally adjacent to it. The Transition Zone was included as a patch type in all reachscape level analyses. Correlations between all pairs of metrics at the reachscape level were calculated so that inherently redundant metrics could be removed prior to statistical analysis. A subset of five uncorrelated configuration metrics that represented the full range of pattern aspects (Landscape Shape Index, Shape, Proximity, Interspersion and Juxtaposition Index and Connectance Index) were retained. These were taken forward for multivariate analysis to investigate the effects of flow variations on the configuration of hydraulic patches at the reachscape scale.

Table 5.3. Spatial metrics used to quantify patch geometry and patch configuration at the class and reachscape levels (McGarigal & Marks, 1995)

Metric	**Aspect of pattern**	**Measure of:**	**Definition**	**Range**	**Ecological Relevance***	**Reach (R) or class (C) scale**
Mean Area (m^2) AREA_MN	Area	Patch size	Equals the sum, across all patches of the corresponding patch type, of patch area (m^2), divided by the number of patches of the same type.	>0 – total area of reachscape	Patch size affects its usability as a habitat by different species and age classes. Patch occupation may be prohibited if too small whereas a large patch may support multiple individuals, larger individuals and/or multiple species. According to the species-area relationship, species richness tends to increase with patch area.	R & C
Radius of gyration (m) GYRATE_MN	Shape	Patch length	Equals the sum, across all patches of the corresponding patch type, of mean distance between each cell in the patch and the patch centroid, divided by the number of patches of the same type.	≥0 – max value when patch covers entire reach =0 when patch is a single cell	Reflects the average distance an organism, dropped at random in a landscape, could travel before leaving the patch. This may affect predator-prey dynamics; as patch length increases, so too might an an organism's ability to move away from predators in a neighbouring patch type without leaving its own preferred habitat. Patch length may also influence the number of different patch-type neighbours, with longer patches being more likely to neighbour more patch types. This may increase occupation by species requiring different habitats for different ecological functions, e.g. resting and feeding.	R & C
Fractal dimension FRAC	Shape	Patch shape	Equals 2 times the logarithm of patch perimeter (m) divided by the logarithm of patch area (m); the perimeter is adjusted to correct for the raster bias in perimeter.	0 to 2	Patch shape irregularity (high fractal dimension) is associated with a high edge:patch ratio and concomitant edge effects. Edge effects may limit the abundance of species that require uniform conditions but increase the abundance of more generalist species or those that require proximity to two or more different patch types.	C
Mean Shape SHAPE_MN	Shape	Patch shape	Equals the sum, across all patches of the corresponding patch type, of patch perimeter (m), divided by the square root of patch area (m^2), adjusted by a constant	>1 = 1 when the patches are square. Increases as patch shape becomes more irregular.	Patch shape irregularity is associated with a high edge:patch ratio and concomitant edge effects. Edge effects may limit the abundance of species that require uniform	R

			to adjust for a square standard, divided by the number of patches of the same type.		conditions but increase the abundance of more generalist species or those that require proximity to two or more different patch types.	
Mean Proximity index PROX_MN	Proximity / isolation	Patch type connectivity / fragmentation	Equals the sum, across all patches of the corresponding patch type, of patch area (m^2) divided by the nearest edge-to-edge distance squared (m^2) between the patch and the focal patch of all patches of the corresponding patch type, divided by the number of patches of the same type. Indicates the relative proximity between all same-type patches.	≥0 =0 if the focal patch has no same type patches within the reachscape. Index increases as the reachscape is increasingly occupied by patches of the same type and those patches become closer and more contiguous (i.e. less fragmented) in distribution.	This metric relates to the theory of island biogeography, which states that as the area and proximity of suitable habitat patches to the focal patch ("island") increases the net immigration rate increases and the net extinction rate decreases resulting in higher species richness. This assumes that the "islands" or habitat patches in question are heterogeneous. In the case of homogeneous patches increased proximity may be associated with an increase in abundance rather than richness.	R & C
Mean Euclidean Nearest- Neighbour (m) ENN_MN	Proximity / isolation	Patch isolation	Equals the sum, across all patches of the corresponding patch type, of the distance (m) to the single nearest patch of the same type, based on shortest edge-to-edge distance, divided by the number of patches of the same type.	>0 Approaches 0 as the distance to the nearest neighbour decreases.	The distance between nearest same-type patches affects dispersal of organisms between suitable habitat and colonisation throughout the reachscape. Isolated patches may be occupied less frequently than those which are closer to similar patches. Species in isolated patches may have reduced persistence and resilience to disturbance.	C
Interspersion and Juxtaposition (%) IJI	Inter-patch type interspersion	Interspersion of patch types	Equals minus the sum of the length (m) of each unique edge type, multiplied by the logarithm of the same quantity, summed over each unique edge type; divided by the logarithm of the number of patch types times the number of patch types minus 1 divided by 2, multiplied by 100.	>0, ≤100 Approaches 0 when the distribution of adjacencies among unique patch types becomes increasingly uneven. =100 when all patch types are equally adjacent to all other patch types (i.e. maximum interspersion and juxtaposition).	High IJI is indicative of high spatial heterogeneity which may support high biodiversity. Reaches with equally adjacent patch types (high IJI) are likely to support an abundance of multi-habitat species that require adjacency of different habitats. Lower values of IJI may be associated with more gradual spatial variations in biotic communities.	R
Clumpiness Index CLUMPY	Contagion	Patch distribution	Equals the proportional deviation of the proportion of like adjacencies involving the corresponding class from that expected under a spatially random distribution. If the proportion of like adjacencies (G_i) is greater than or equal to the proportion of the landscape comprised of the focal class (P_i), then CLUMPY equals G_i, minus P_i, divided	-1 to 1 =-1 when the focal patch type is maximally disaggregated. =0 when the focal patch type is distributed randomly. Approaches 1 when the patch type is maximally aggregated.	Clumpiness provides a measure of patch type fragmentation, independent of patch type area, i.e. patch isolation and the increase in edge:interior ratio. Patch isolation can limit dispersal to escape disturbances or predators. An increase in edge: interior ratio limits usable area which may reduce abundance and increases edge effects which may favour more generalist	C

			by 1 minus P_i. Likewise, if $G_i < P_i$, and $P_i \geq 0.5$, then CLUMPY equals G_i , minus P_i, divided by 1 minus P_i. However, if $G_i < P_i$, and $P_i < 0.5$, then CLUMPY equals P_i, minus G_i, divided by negative P_i.		species.	
Landscape Shape Index LSI	Patch aggregation	Edge density	Equals 0.25 (adjustment for raster format) times the sum of the entire landscape Boundary and all edge segments (m) within the landscape boundary involving the corresponding patch type, divided by the square root of the total landscape area (m).	≥1 =1 when the landscape consists of a single square patch of the corresponding type. Increases without limit as landscape shape becomes more irregular and/or as the length of edge within the landscape of the corresponding patch type increases.	As the density of edges increases so too do edge effects. Edges and boundary zones are associated with higher biodiversity and may favour generalist species over specialist species, or species adapted to edge/boundary environments. Edges may also affect the movement of organisms and the flow of materials through the reachscape.	R
Connectance Index (%) CONNECT	Patch Isolation	Functional connectivity	Equals the number of functional joins between all patches of the corresponding patch type, divided by the total number of possible joins between all patches of the corresponding patch type, multiplied by 100.	0 to 100 = 0 when either the focal class consists of a single patch or none of the patches of the focal class are connected (i.e. within a threshold distance of another patch of the same type). = 100 when every patch of the focal class is connected.	As the number of patches within an organism's range of perception and dispersal (threshold distance) increases, as indicated by an increase in the connectance index, movement throughout the reachscape becomes easier. This may increase the abundance of species wih a preference for the patch type and improve resilience to disturbance events.	R

* It should be noted that the precise ecological relevance of spatial metrics is species-specific; different metrics and different values of metrics will have relevance to different species. Very little research has directly tested the ecological relevance of spatial metrics in the instream environment. For this reason only a general guide to the type of ecological patterns or processes that could be affected by spatial metrics, drawn from landscape ecology studies, is given here. Further research is needed to test whether these relationships hold true in a linear, hydrologically connected environment.

5.2.3 Analysis of reachscape configuration

Spatial metrics were standardised (z-scores) in SPSS to account for the different units of measurement then imported to PRIMER-E v.6 (Plymouth Routines in Multivariate Ecological Research, 2001) software for permutation based multivariate analysis (Clarke & Warwick, 2001), with samples represented by each site-flow combination and variables represented by the five configuration metrics. Two factors were defined for each sample; site (River Arrow, River Salwarpe and Leigh Brook) and flow category (Very High (≤Q20), High (Q20-Q40), Moderate (Q40-Q60), Low (Q60-Q80) and Very Low (Q80-Q100). The data were converted into a similarity matrix using the Euclidean distance measure then represented graphically in a non-metric multidimensional scaling plot. 2-way crossed ANOSIM without replicates was used to test the null hypothesis that there were no significant differences in reachscape configuration between flow categories. In the absence of replicates within-group differences cannot be derived from multiple samples (replicates) of each factor combination – in this case site and flow. Instead within-group differences are derived from differences between one factor pooled across each level of the second factor. That is, significant differences are predicated on the assumption that all sites will respond in a similar way to changes in flow, or, that if site has a significant effect on reachscape configuration all flows categories will respond to site differences in a similar way.

5.2.4 Mapping and quantifying patch change

The location and extent of each hydraulic patch type delineated in Chapter 3 was mapped at every flow in ArcGIS. A weighted sum overlay of hydraulic patch type maps from every pair of consecutive flows (e.g. very low and low flow, low and moderate flow, moderate and high flow, high and very high flow) was performed in ArcGIS. This produced new maps combining the area occupied by the hydraulic patch type at the lower and higher flow. This was classified into to one of three categories; area occupied by the hydraulic patch type only at the lower flow, area occupied by the hydraulic patch type at both flows (i.e. spatially and temporally stable area) and area newly occupied at the higher flow. The newly occupied area, expressed as a percentage of the total area occupied at the higher flow, was used to quantify hydraulic patch turnover (Arscott et al., 2002).

5.3 Results

5.3.1 Patch configuration

Mean patch area and length

Mean patch area and mean patch length showed very similar responses to variations in flow for all patch types at all sites (Figures 5.4-5.6). At the River Arrow mean patch area of RA1 (moderate-slow) and RA3 (moderate-very slow) were relatively invariant to changes in flow with the exception of an increase in mean area of RA3 (moderate-very slow) patches at very high flow. Mean patch area of RA2 (shallow-slow) responded erratically to changes in flow but the overall trend was a decline in mean patch area as discharge increased. The fastest patch type (RA4) was most responsive to the increase in discharge between high and very high flow. Mean area of RA5 patches responded by peaking at intermediate flow. A similar but more subdued pattern of responses was evident at the River Salwarpe, but again, mean patch area and mean patch length showed a similar pattern of responses to changes in flow. The mean area of shallow and/or slow patch types (RS1 and RS3) had the smallest response to changes in flow of all patch types at this site but showed a gradual decline as discharge increased. Mean area of RS2 (shallow-fast) patches responded more erratically to increases in discharge but also showed a decreasing trend. Mean patch area of the deepest patch type (RS5 (deep-moderate)), had a u-shaped response to changes in flow, peaking at moderate flows like those at the River Arrow. Mean patch area of the fastest patch type (RS4 (moderate-fast)) increased with flow but the magnitude of the increase was smaller than that of the fastest patch type at the River Arrow.

Mean patch area was least responsive to flow at the Leigh Brook for all patch types and the pattern of responses to flow was slightly different to that at the River Arrow and River Salwarpe. The mean patch area of the deepest patch type at the Leigh Brook (LB1 (moderate-moderate)) showed a more linear response to flow, increasing as flow increased. Similarly the mean patch area of the fastest patch type (LB4 (moderate-fast)) showed a very different response, increasing gradually with discharge but decreasing at very high flow in contrast to the huge increase in mean area of similar patch types at the River Arrow and River Salwarpe. Mean patch area of LB5 (moderate-slow) increased gradually with discharge. Mean patch area of LB 2 (shallow-slow) increased gradually with flow. Mean patch area of LB3 remained relatively invariant to changes in flow.

Mean patch shape complexity

Mean patch complexity varied little with flow for all patch types at every site (Figures 5.4-5.6). The majority of patch types had a mean shape complexity value between 1 and 1.5, indicating moderate shape complexity. Patch shape complexity of RS2 (shallow-fast) patches neared the maximum possible value at very high flow although this was affected by the lack of single pixel (and hence perfectly regular) patches at this flow which lowered the average patch shape complexity at all other flows. Mean patch shape complexity was smallest (i.e. patches were most regular in shape) for patch types LB2 (shallow-slow), RS3 (shallow-slow) and RA2 (shallow-slow) at high and very high flows. This was due to the predominance of small often single pixel patches.

Patch distribution

Patch distribution was highly aggregated for most patch types at every site (Figures 5.4-5.6). The most aggregated patches were types RS5 (deep-moderate), RA5 (very deep-very slow), LB1 (moderate-moderate) and LB5 (moderate-slow). The most disaggregated patches occurred at very low flow in patch types RS2 (moderate-fast) and RA4 (moderate-fast) as they appeared as highly dispersed single pixel patches.

Patch proximity and nearest-neighbour distances

At the Leigh Brook patches of all types with the exception of LB4 (moderate-fast) were very close to their nearest neighbours at all flows (Figure 5.6). The nearest-neighbour distance between LB4 (moderate-fast) patches decreased as discharge increased, due to the increase in its proportional abundance. Very similar patterns in mean distance to nearest-neighbour were evident at the River Arrow and River Salwarpe for all patch types. Nearest-neighbour distances between RS1 (moderate-slow), RS2 (shallow-fast) and RS3 (shallow-slow) and between RA1 (moderate-slow), RA2 (shallow-slow) and RA3 (moderate-very slow) were were relatively invariant to changes in flow. Nearest neighbour distance for both RS5 (deep-moderate) and RA5 (very deep-very slow) spiked at moderate flow, however this was due to the absence of single pixel patches occurring at these flows. The nearest neighbour distance between RA4 (moderate-fast) and RS4 (moderate fast) tended to increase at extreme flows.

Mean proximity was the most responsive patch configuration metric to changes in flow (Figures 5.4-5.6), however its interpretation was the least straightforward as it reflected area and distance. At all sites the general trend was for mean proximity between patches of types LB2 (shallow-slow), LB3 (shallow-moderate) and LB5 (moderate-slow), RS3 (shallow-low), RS2 (shallow-fast) and RS1 (moderate-slow) and RA2 (shallow-slow) and RA3 (moderate-very slow) to decrease as discharge increased. Mean proximity between patches of remaining types LB1, LB4, RS4 and RS5 increased with discharge but by varying degrees. Anomalies in the mean proximity between patches of type RA5 (very deep-very slow) and RA4 (moderate-fast) occurred at the River Arrow where both took a near-zero value in spite of very different configurations.

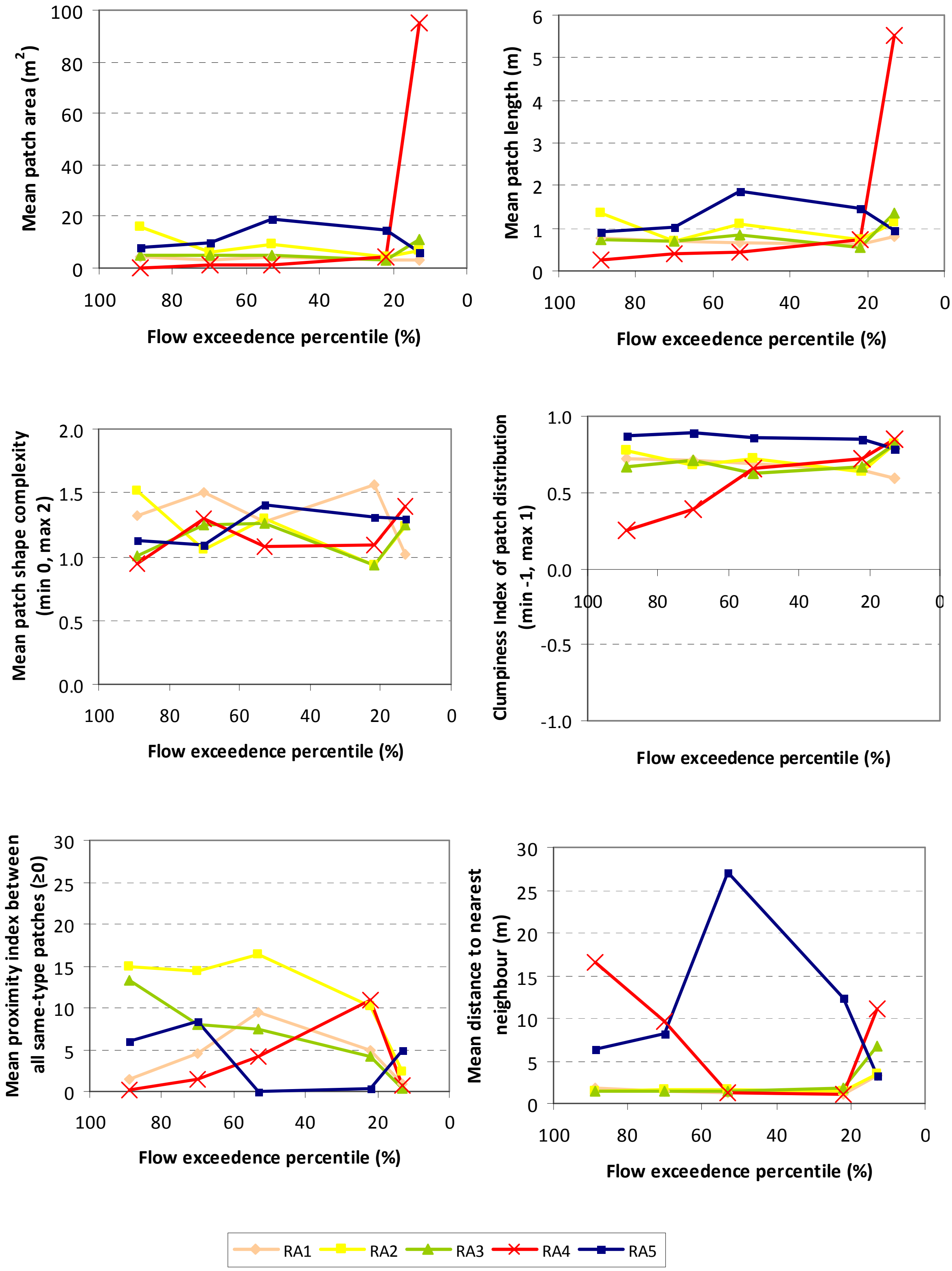


Figure 5.4. Flow-related changes to patch configuration metrics for all hydraulic patch types at the River Arrow.

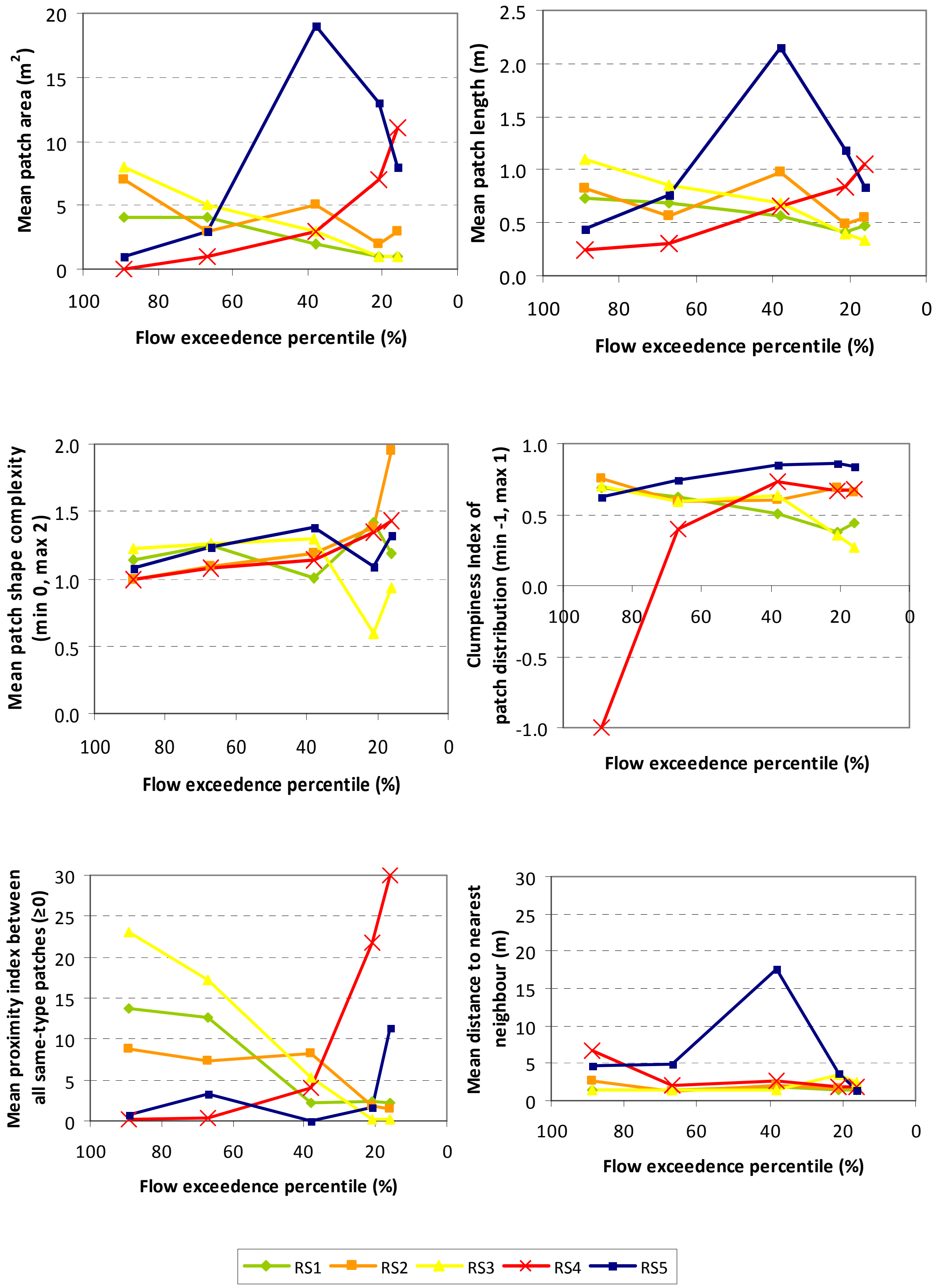


Figure 5.5. Flow-related changes to patch configuration metrics for all hydraulic patch types at the River Salwarpe.

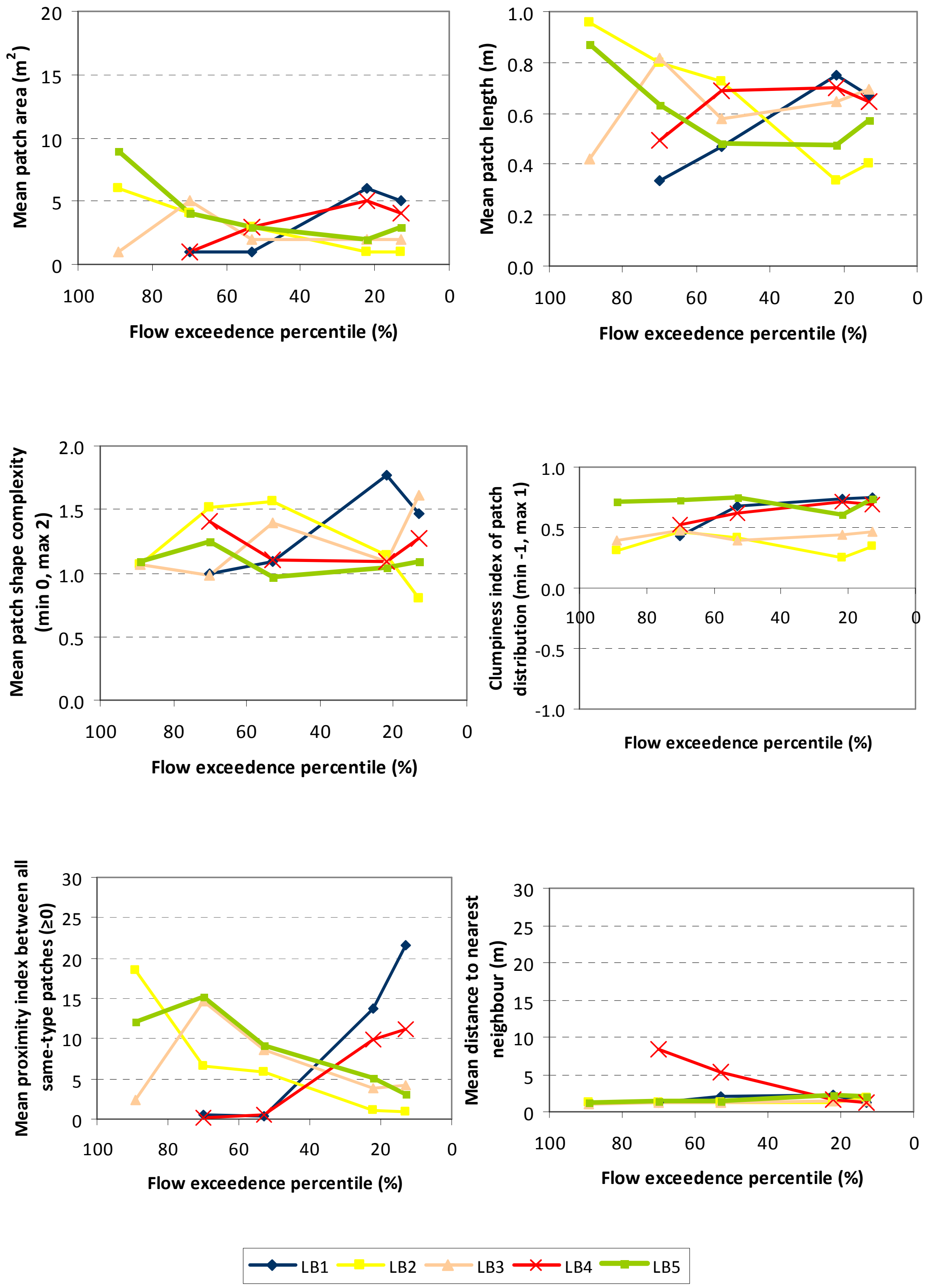


Figure 5.6. Flow-related changes of patch configuration metrics for all hydraulic patch types at the Leigh Brook.

5.3.2 Patch change

This sub-section examines where hydraulic patches moved and quantifies their spatial turnover with each increase in discharge. For ease of cross-reference the figures showing the location of hydraulic patches at each flow for the River Arrow, River Salwarpe and Leigh Brook are presented in Appendices F, G and H respectively. Spatial turnover statistics are presented in this chapter in Tables 5.5-5.6.

River Arrow

All patch types with the exception of RA4 (moderate-fast) occurred in relatively fixed locations in the channel and showed a gradual movement as discharge increased, associated with the gradual expansion or contraction of its area. RA1 (moderate-slow) (Appendix F, Figure 1) was located along the thalweg at all flows except very high when it was marginalised and formed thin, ribbon-like patches at the boundary between the thalweg and recirculation zones. Consequently spatial turnover was relatively small and uniform with the first three increases in discharges and greatest at very high flow when 79% of the patch area was in a newly occupied location. RA2 (shallow-slow) (Appendix F, Figure 2) was located in patches spanning the channel width at the topographic high points in the reach upstream and downstream of the pools at very low flow. The general location of the patch type remained relatively invariant and spatial turnover remained less than 38% with the first two increases in discharge. The biggest change in location occurred at high and very high flow when the area of the patch type decreased and the location was limited to small areas at the channel margins (64-79%area no longer occupied, Table 5.4). RA3 (moderate-moderate) patches (Appendix F, Figure 3) were located in marginal areas adjacent to the pools at all flows however as discharge increased small areas of RA3 in the channel centreline disappeared. This was reflected by the higher percentage of area no longer occupied with each increase in discharge than newly occupied area (Table 5.4). RA5 (very deep-very slow) patches (Appendix F, Figure 5) were located in the topographic low points of the reach at all flows. Their location remained static until the increase at very high flow where, like other patch types, its location was limited to its lateral extremes due to the dominance of RA4 patches along the thalweg. The location of RA4 patches (moderate-fast) explained the movement of all other patch types. RA4 patches (moderate-fast) (Appendix F, Figure 5) occupied a very small percentage of the reach until high flow when its location along the thalweg dominated the reach and replaced all other patch types. This patch type had the most variable spatial dynamics across flows (turnover of 3-93%) and the largest turnover with any single increase in discharge (75% area no longer occupied and 93% newly occupied area, Table 5.4).

River Salwarpe

The same general pattern in patch movement occurred at the River Salwarpe, with all but two patch types (RS2 (shallow-fast) and RS4 (moderate-fast)) having relatively fixed locations in the channel and responding to flow only in terms of gradual lateral and longitudinal expansion or contraction (Appendix G, Figures 1-5). RS1 (moderate-slow) and RS3 (shallow-slow) patches gradually contracted as discharge increased and were increasingly located at the channel margins. Consequently the percentage of area no longer occupied was usually higher than the percentage of newly occupied area. The deepest patch type (RS5 (deep-moderate)) was the least spatially dynamic with between 3-14% of patch area no longer occupied with each increase in discharge (Table 5.5). The location at the deepest part of the thalweg remained fixed at all flows and the area expanded as discharge increased (21-61% newly occupied area). The most spatially dynamic patch types at the River Salwarpe were also the two fastest patch types – RS2 (shallow-fast) and RS4 (moderate-fast). As discharge increased RS4 (moderate-fast) replaced the location formerly occupied by RS2 (shallow-fast) (Appendix G, Figure 4), and RS2 (shallow-fast) moved to the location formerly occupied by RS3 (shallow-slow) (Appendix G, Figure 2).

Leigh Brook

This site also had three patch types (LB2 (shallow-slow), LB3 (shallow-moderate) and LB5 (moderate-slow)) whose location varied very little as discharge increased (Appendix H, Figures 2, 3 & 5). All of these patch types showed a gradual contraction and became more limited to marginal areas at higher flows. The patch types with the greatest spatial dynamics were those occupying the extremes of the hydraulic distribution The deepest patch type at the Leigh Brook (LB1 (moderate-moderate)) originated in the deepest and fastest part of the small scour pool adjacent to the left bank at very low flow and showed considerable spatial dynamics as discharge increased (Appendix G, Figure 1). The patch expanded into 87-100% newly occupied area with the first three increases in discharge. This contrasts with the spatial stability of the deepest patch types at the River Arrow and River Salwarpe. The fastest patch type (LB4 (moderate-fast) was the most spatially dynamic patch type at this site (Appendix G, Figure 4) occupying on average 68% new area with the first three increases in discharge (Table 5.6). This patch type was least dynamic between high and very flow and remained in a channel centreline location at the topographic high points in the channel.

Table 5.4. Spatial turnover of each patch type at the River Arrow expressed as a percentage of area no longer occupied and area newly occupied after each increase in discharge. Flow increase codes refer to very low to low (VL-L), low to moderate (L-M), moderate to high (M-H) and high to very high (H-VH).

Patch type	**Depth-velocity description**	**Flow increase**	**Area no longer occupied**	**Area newly occupied**
RA1	Moderate-slow	VL-L	41%	46%
		L-M	32%	52%
		M-H	49%	43%
		H-VH	91%	79%
RA2	Shallow-slow	VL-L	38%	30%
		L-M	25%	29%
		M-H	64%	34%
		H-VH	79%	36%
RA3	Moderate-very slow	VL-L	31%	20%
		L-M	38%	31%
		M-H	41%	24%
		H-VH	57%	24%
RA4	Moderate-fast	VL-L	75%	93%
		L-M	50%	79%
		M-H	3%	86%
		H-VH	22%	73%
RA5	Very deep-very slow	VL-L	17%	11%
		L-M	17%	11%
		M-H	15%	27%
		H-VH	52%	36%

Table 5.5. Spatial turnover of each patch type at the River Salwarpe expressed as a percentage of area no longer occupied and area newly occupied after each increase in discharge. Flow increase codes refer to very low to low (VL-L), low to moderate (L-M), moderate to high (M-H) and high to very high (H-VH).

Patch type	Depth-velocity description	Flow increase	Area no longer occupied	Area newly occupied
RS1	Moderate-slow	VL-L	35%	46%
		L-M	74%	45%
		M-H	65%	66%
		H-VH	70%	59%
RS2	Shallow-fast	VL-L	49%	45%
		L-M	42%	55%
		M-H	85%	81%
		H-VH	39%	27%
RS3	Shallow-slow	VL-L	41%	13%
		L-M	53%	20%
		M-H	90%	25%
		H-VH	89%	80%
RS4	Moderate-fast	VL-L	33%	89%
		L-M	17%	90%
		M-H	29%	70%
		H-VH	26%	42%
RS5	Deep-moderate	VL-L	7%	61%
		L-M	5%	60%
		M-H	3%	41%
		H-VH	14%	21%

Table 5.6. Spatial turnover of each patch type at the Leigh Brook expressed as a percentage of area no longer occupied and area newly occupied after each increase in discharge. Flow increase codes refer to very low to low (VL-L), low to moderate (L-M), moderate to high (M-H) and high to very high (H-VH).

Patch type	Depth-velocity description	Flow increase	Area no longer occupied	Area newly occupied
LB1	Moderate-moderate	VL-L	0%	100%
		L-M	83%	96%
		M-H	12%	87%
		H-VH	18%	36%
LB2	Shallow-slow	VL-L	59%	12%
		L-M	42%	16%
		M-H	82%	11%
		H-VH	75%	50%
LB3	Shallow-moderate	VL-L	43%	82%
		L-M	52%	45%
		M-H	87%	84%
		H-VH	60%	48%
LB4	Moderate-fast	VL-L	0%	100%
		L-M	18%	27%
		M-H	10%	77%
		H-VH	33%	35%
LB5	Moderate-slow	VL-L	34%	27%
		L-M	18%	27%
		M-H	67%	20%
		H-VH	36%	47%

5.3.3 Reachscape configuration

The configuration of all patch types at the reachscape level at each site, based on the uncorrelated subset of spatial metrics identified in Section 5.2, was represented graphically by 2D nMDS plots (Figures 5.7-5.9). Each plot had a zero Kruskal stress value indicating an excellent representation of inter-flow differences by distance between points in the plot. Reachscape configuration at each site responded differently to changes in flow (Figures 5.7-5.9). The Leigh Brook reachscape was characterised by greater patch shape complexity and proximity at very low to moderate flows and by greater connectivity and aggregation of patches at higher flows (Figure 5.7). Differences in reachscape configuration at the River Salwarpe between lower and higher flows were less clear (Figure 5.8). The biggest differences occurred along an axis identified by a higher degree of interspersion and juxtaposition of patch types at very low flow and the aggregation of same-type at moderate flow. The very high and high reachscapes were the furthest apart, with the reachscape at very high flow characterised by the highest patch shape complexity whereas the reachscape at high flow was characterised most by the greatest proximity between same-type patches.

The greatest distances between reachscape configurations at the River Arrow (Figure 5.9) also occurred between very similar flows. Very low flow was characterised by the lowest connectivity and highest proximity between patches but low flow reacshcape was characterised by the highest connectivity and the lowest proximity between patches. Likewise, high flow reachscape was characterised by the highest interspersion and juxtaposition of patches and the lowest aggregation of patches but this trend was reversed at very high flow. The MDS plots highlighted the differences between sites, even where those differences are very small, as they were in across the flows at each site. The ANOSIM test confirmed that differences in reachscape configuration between flow categories (VL, L, M, H, VH) were **not significant** (Rho=-0.176, p>0.05). The possible reasons for this are discussed in the following section.

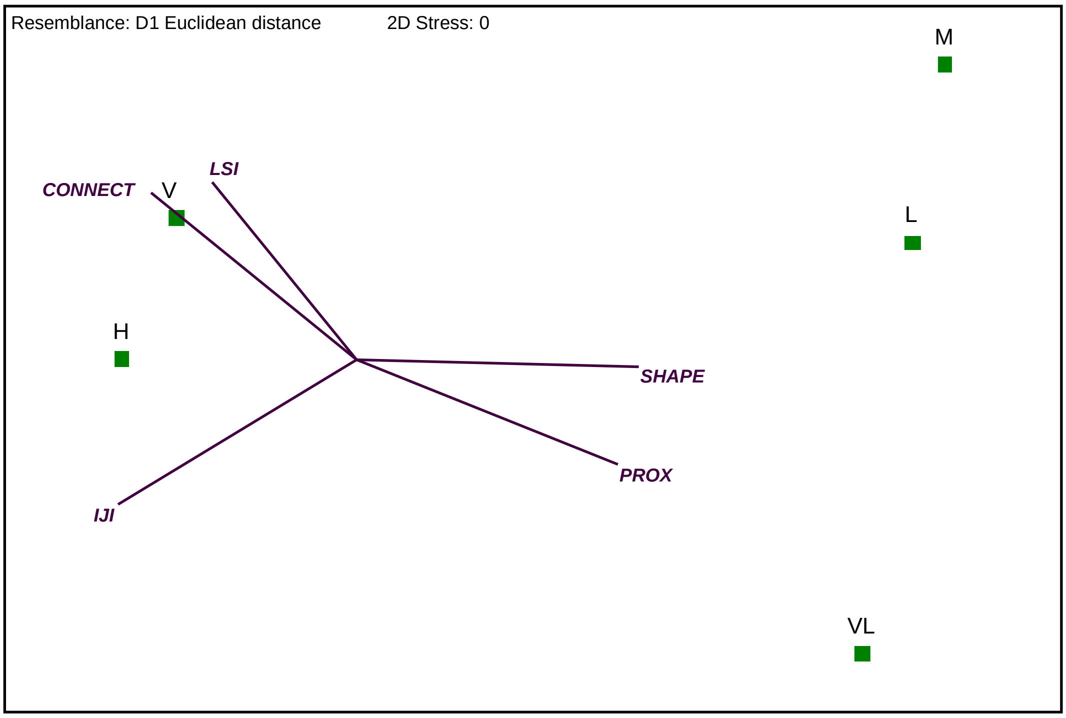


Figure 5.7. nMDS ordination plot indicating inter-flow differences in reachscape configuration at the Leigh Brook where VL=very low flow, L=low flow, M=moderate flow, H=high flow and VH=very high flow. The plot is overlaid with direction vectors of each metric of configuration.

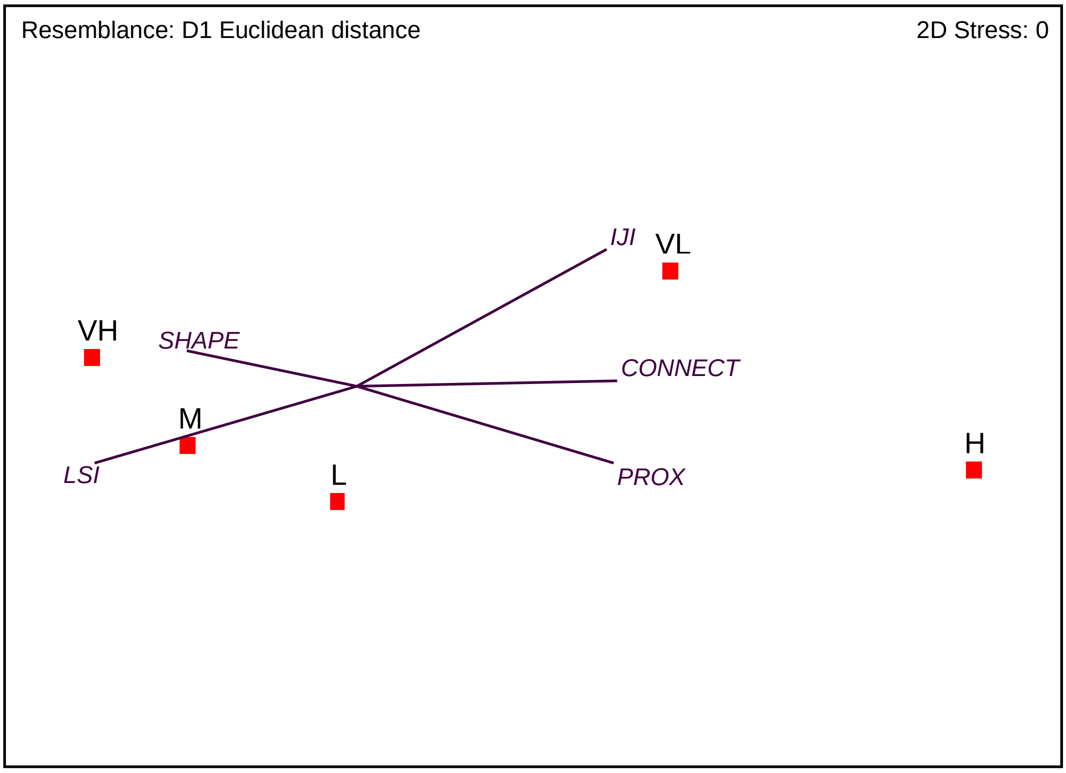


Figure 5.8. nMDS ordination plot indicating inter-flow differences in reachscape configuration at the River Salwarpe where VL=very low flow, L=low flow, M=moderate flow, H=high flow and VH=very high flow. The plot is overlaid with direction vectors of each metric of configuration.

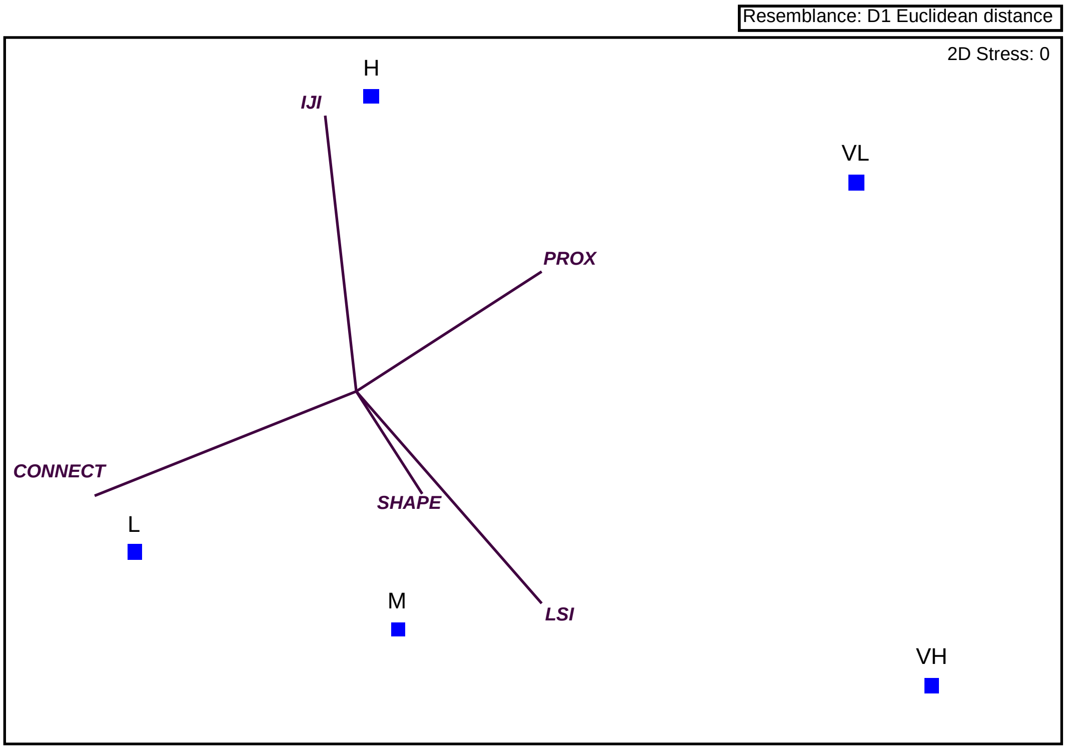


Figure 5.9. nMDS ordination plot indicating inter-flow differences in reachscape configuration at the River Arrow where VL=very low flow, L=low flow, M=moderate flow, H=high flow and VH=very high flow. The plot is overlaid with direction vectors of each metric of configuration.

Examination of the response of each individual configuration metrics to changes in discharge showed that all aspects of reachscape configuration were relatively invariant to flow (Figure 5.10). Landscape Shape Index (LSI) varied very little with discharge at all three sites, taking a value of 6-9 at all flows (Figure 5.10 a). A very slight u-shaped response curve was evident, with lower values occurring at very low and very high flows. This reflected a decrease in total edge length within the reachscape indicative of patches becoming slightly more aggregated/less fragmented at flow extremes. Mean patch shape complexity oscillated within a very small range of values (1.23-1.33) that indicated only a minor deviation from square, regular shaped patches at all sites and all flows. Mean proximity between all same-type patches in the reach was relatively invariant at the River Arrow and Leigh Brook which disguised the very different response of individual patch types discussed in Section 5.3.1. At the River Arrow mean proximity halved between very low and very high flow which reflected the underlying trend for proximity between three of the five hydraulic patches at this site to decrease with discharge but disguised the exponential increase in proximity between RS4 patches at high flows. A high level of mean patch type interspersion occurred at all sites (70-84%), with most values falling in the upper quartile of the possible range, indicating a high level of spatial hydraulic heterogeneity. Maximum interspersion occurred at high flow at all three sites although the total range of values over all flows was relatively small. The Connectance Index was also high at all sites (61-85%) and varied by ≤11% with discharge indicating that the majority of patches were within half a reach length's distance of another same-type patch at all flows.

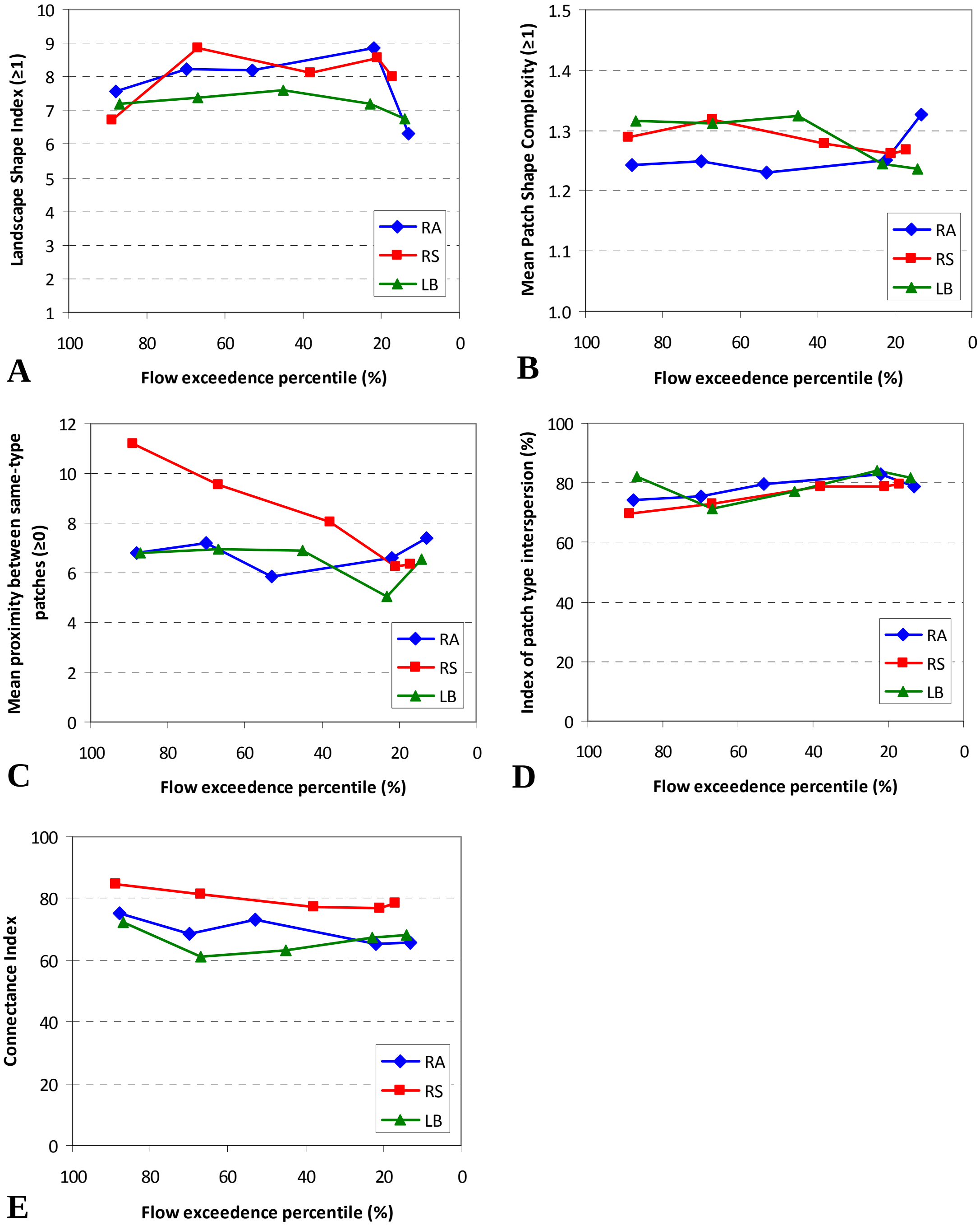


Figure 5.10 Response of individual reachscape configuration metrics to changes in discharge at each site.

5.4 Discussion

Patch configuration (class scale)

Overall flow-related changes in patch configuration were very subtle for all patch types, however several differences were identified. Patch configuration of the fastest patch type at each site was most responsive to changes in flow as might be expected for a patch type at the extremes of the depth-velocity distribution and with greater dependence on discharge than morphology. The biggest changes in patch configuration metrics occurred at intermediate flows for the deepest patch type at each site and flow extreme for the fastest patch type at each site. Area and length had very similar responses to flow; increases in patch area were associated with increases in patch length, which was not surprising in an environment dominated by the downstream movement of flow.

The results highlighted several issues when using spatial metrics to quantify patch configuration. Several metrics, including nearest-neighbour distance (ENN) and patch shape complexity (FRAC) were sensitive to the presence or absence of single pixel patches. For example, several changes to the configuration of the deepest patch types were explained by the response of the metric to the presence or absence of single pixel patches. This behaviour complicated interpretation of the metrics and caused some spikes in the general trend. It is recommended that ENN and FRAC be interpreted with care or that a minimum patch size criterion is applied before the metric is calculated to minimise this problem. Metrics which reflect a combination of changes e.g. PROXIMITY (abundance and aggregation) must be interpreted with caution as similar values can reflect very different patch configurations, as was the case for RS4 at very low and very high flows. The near-zero value reflected the influence of disaggregation at very low flow but reflected high abundance of the patch type at very high flow.

It is interesting that the deepest patch type at the River Arrow (RA5) showed greater flow-related changes in patch configuration than patches associated with relatively non-descript areas of the channel bed (e.g. RA1, RA2 and RA3). In some cases this was explained by the sensitivity of the spatial metric to single pixel patches (PROX, ENN, CLUMPY). However it may also depend on the patch type upstream of RA5. A the River Arrow RA5 patches were downstream of moderate to fast flowing patches rather than slow flowing patches which may influence the hydraulic response of RA5 to changes in flow. Another feature of FRAGSTATS (v 3.3) worth noting is theat it is not possible to analyse the spatial relationship between two specific patch types; either the configuration of patches of a single patch type can be quantified or the configuration between patches of all types can be quantified. In terms of its application in ecohydraulic studies the ability to quantify the relationship between patches with complementary functions (e.g. the proximity between resting habitat (deep-slow units) and feeding habitat (fast-flowing units)) is likely to be very important. Further development of the software to include this capability would be a useful avenue for further research.

Patch change

All sites showed that the fastest patch types, whose abundance depended on high discharges were the most spatially dynamic, whereas the deepest patch types, whose location was strongly tied to topographic lows were the least spatially dynamic. Slow patch types were moderately dynamic. At high discharges the location of all but the fastest patch type shifted to the channel margins. Lateral spatial dynamics and transitioning between RS3 (shallow-slow) RS2 (shallow-fast) to RS4 (moderate-fast) illustrated the changing hydraulic performance of some areas of the channel as discharge increased. These results show the strong influence of bed morphology on the location of all patch types with the exception of the fastest patch type at each site. This was associated with high discharges and was located along the thalweg. However the width of the fastest patch type was limited where it occurred at a topographic low, showing that topography had some influence on all patch types to a degree. Depth is an indicator of hydraulic patches with limited spatial dynamics and the velocity is an indicator of hydraulic patches that are highly spatially dynamic. This explains why LB1 (moderate-moderate), which was the deepest but also second fastest patch type in the reach, was more spatially dynamic than RA5 and RS5.

The analysis of patch change supported Clifford et al.'s (2002; 2006) suggestion that the arrangement of hydraulic patches changes from a patchy structure at low flow and to a longitudinal ribbon-like structure with clear distinctions between the channel centreline and margins as discharge approaches bankfull. Although none of the hydraulic surveys in this study approached bankful, the trend described by Clifford et al. (2002; 2006) was evident at all sites. Certainly the patches which spanned the channel width at lower flows were marginalised and became thinner and longer at higher flows.

Reachscape configuration

In Chapter 1 it was hypothesised that patch shape complexity would be greatest at low flows. Although the MDS plot appeared to support this hypothesis at the Leigh Brook, the ANOSIM results indicated that mean patch shape complexity across all patch types did not significantly differ between flows at any site. It was also hypothesised that interspersion of patches would be greatest at low flows and aggregation of patch types however no significant differences in reachscape configuration were found between flow categories (across all sites) or between sites (across all flow categories). Due to the lack of site-flow replicates ANOSIM without replicates was used to test for difference between flow categories. Detection of differences between flow categories required all sites to respond to flow changes in exactly the same way. However the different morphology at each site and the site-specific nature of the hydraulic patch classifications meant this was not the case. Had three reaches at the same site been surveyed the test could have been performed for each site independently which would have isolated the site-specific response to flow. This approach was taken by Thoms et al. (2006) where flow-related differences in configuration were significant.

The non significant result highlights that small variations in flow are not associated with significant differences in reachscape configuration. Thoms et al's (2006) study compared configuration differences at between greater flow differences. It is speculated that differences in reachscape configuration between very low and very high flows may have revealed significant differences. However the MDS plots suggest that the differences between extreme flows reflected different aspects of configuration at each site, possibly due to morphological differences. This suggests that even though lowland rivers have similar reach scale morphology (i.e. pool-riffle sequences) local variations in bedform amplitude and susbstrate can affect how the hydraulic environment changes in response to flow.

Chapter summary

In general the patterns of flow-related changes in patch configuration, patch change and reachscape configuration were very similar at every site, as might be expected at three lowland rivers, however some subtle differences relating to local differences in bed morphology and the overall depth-velocity distributions were evident. This chapter has illustrated some of the limitations and idiosyncrasies of different spatial metrics and makes useful recommendations for their future application.

6

Discussion and Conclusions

Chapter Overview

This chapter identifies the key findings of the research and discusses their implication for river science. A speculative model of hydraulic patch dynamics in morphologically contrasting reaches is presented. The relevance of the results to river habitat surveys, instream modelling and river rehabilitation is explained and directions for further research are recommended.

6.1 Key findings and implications for river science

This section summarises the key findings from each of the three results chapters in relation to the aims and objectives of the thesis, as set out in Section 1.7. The implications of the results for river science are discussed with reference to current theoretical understanding of rivers.

6.1.1 Classification of hydraulic patches and transition zones

The first aim of this study, addressed in Chapter 3, was to evaluate the merits of fuzzy cluster analysis as a method for quantitatively classifying the hydraulic environment. Three specific objectives were identified in Section 1.7 (p.32); to evaluate the performance of three different fuzzy clustering algorithms for classifying hydraulic data (Obj. 1a), to generate a classification of hydraulic patches and transitional zones to evaluate the effect of discharge on the hydraulic environment (Obj. 1b), and to make recommendations for the applications of fuzzy cluster analysis in river science (Obj. 1c).

The performance of three fuzzy clustering algorithms for delineating hydraulic patches was evaluated; fuzzy C-means, Gustafson-Kessel and Gath-Geva, only the first of which has been tested for the purpose of delineating hydraulic patches before (Obj. 1 a, p.33). The results showed that **the Gustafson-Kessel fuzzy clustering algorithm offers some advantages for delineating hydraulic patches over fuzzy C-means in that it can detect ellipsoidal shaped clusters**. For example, in the low flow environment at the River Arrow ellipsoidal shaped clusters improved the differentiation between areas of recirculating flow in pool margins and the backwater

(RA3), which were characterised by negligible, often upstream velocities but had a wide range of depths, from areas characterised by slow flow and shallow depths (RA2). At the River Salwarpe ellipsoidal shaped clusters facilitated the differentiation between shallow-fast (RS2) and moderate-fast (RS4) patch types, which appeared as two natural fuzzy clusters in the hydraulic data distribution at high and very high flow. At the Leigh Brook the ellipsoidal shape of cluster LB5 reflected the range of depths found in the scour pool better than the equivalent cluster in the 5-FCM classification. Where other studies have only used the fuzzy c-means algorithm to delineate hydraulic patches (Legleiter & Goodchild, 2005), this study shows the advantages of using the Gustafson-Kessel fuzzy covariance algorithm. It also showed that the Gath-Geva algorithm did not prove useful for classifying continuous hydraulic data. The algorithm failed to converge for most classifications of the River Arrow data so was not used on data from the remaining two sites. This finding supports prior tests of the algorithm which reported its sensitivity to the cluster centroids used to initialise the clustering process and its tendency to allow a very limited level of fuzziness in cluster membership function values (MFVs) (Höppner et al., 1999). It is best suited to detecting well separated fuzzy clusters whose shapes are all very different (Höppner et al., 1999) rather than very fuzzy clusters in continuous hydraulic data. As such it may perform better on data from step-pool reaches where the distinction between hydraulic conditions in the steps and pools is much clearer, however further research would be needed to test this theory.

The optimal classifications of hydraulic data generated using the Gustafson-Kessel fuzzy clustering algorithm delineated five hydraulic patch types, defined by the joint distribution of depth and velocity, at each site (Objective 1b). The five patches were distributed across all regions of the “heart-shaped” data distribution combined from multiple discharge surveys, i.e. the classification reflected the influence of discharge variations on the hydraulic environment whilst also producing spatially coherent patches that clearly reflected the influence of channel morphology on the hydraulic environment. Although a five patch classification was optimal at each site in this study it is not necessarily the case that a 5-cluster classification would be optimal in all pool-riffle reaches as the level of morphological and hydraulic diversity will vary on a site-by-site basis. It is important to acknowledge that in this study the optimal classification was defined relative to the hydraulic and morphological diversity sampled. Had longer pool-riffle reaches with larger hydraulic ranges been sampled at the River Salwarpe and Leigh Brook a classification with more patches or the same number of patches but with centroids in different areas of the new distribution may have been more appropriate. Likewise, if a reach containing a riffle had been sampled at the River Arrow the delineation of an additional patch type may have been useful to further differentiate velocity at shallow depths, just as the larger depth range at this site had been classified into three patch types where two were sufficient at the other sites.

It is suggested that **the optimal number of hydraulic patches may differ between reach morphology types, depending on the influence of longitudinal and lateral topographic variations on the shape of the data distribution and the influence of bedform amplitude and wavelength on the density/distribution of data points within the hydraulic range**. For example, in pool-riffle reaches longitudinal and lateral topographic variations have an approximately equal influence on the hydraulic environment which creates a heart-shaped hydraulic data distribution that is stretched along the depth-velocity axes as well as away from them (Figure 6.1 a). Pool-riffle reaches are also characterised by moderate bedform amplitude and wavelength which results in a relatively continuous and even density of data points within the hydraulic range. Thus the optimal classification delineated hydraulic patches in all regions of the data distribution. By contrast in step-pool reaches where the influence of longitudinal variations dominates the hydraulic environment, the data distribution is likely to be L-shaped and stretched along the depth-velocity axes rather than away from them (Figure 6.1 b) (Stewardson & McMahon, 2002). Here it is likely that the relatively high bedform amplitude and short wavelength will produce fewer intermediate hydraulic conditions between the topographic extremes. Instead it is probable that data points will be densely distributed in the shallow-fast (step) and deep-(relatively) slow (pool) regions of the depth-velocity space with a relatively sparse occurrence of points elsewhere (Figure 6.1 b). As such, fewer hydraulic patch types are likely to be needed to characterise the data distribution. The sparsely populated region of the data space (indicated by the dashed line in Figure 6.1 b) may be allocated to a transition zone or be delineated as a patch type in its own right, depending on the defuzzification rules used. The alternative extreme of reach morphology - plane-bed or channelised reaches - where there is very little longitudinal topographic variations and instead lateral topographic variations are the dominant influence on the hydraulic environment, the shape of the data distribution is likely to be stretched away from the depth-velocity axes more than it is stretched along them with a relatively small hydraulic range characterised by a positive correlation between depth and velocity (Figure 6.1 c) (Stewardson & McMahon, 2002). The shallow-fast and deep-slow hydraulic conditions associated with longitudinal topographic extremes are likely to be absent or scarce so the main hydraulic difference will be between the shallow-slow conditions in channel margins and the relatively deep-fast flow in the channel centreline where friction from the bed and banks is reduced. As such the optimal classification will likely contain fewer hydraulic patches than pool-riffle reaches. In addition, the relatively low bedform amplitude and large wavelength are likely to produce a highly continuous, dense distribution of data points within the relatively small hydraulic range, characterised by a high degree of overlap between hydraulic patches, which may results in a larger transition zone.

Based on these suppositions it is suggested that, as a general rule, the optimal number of hydraulic patches will be higher in reaches where both longitudinal and lateral topographic variations influence the hydraulic environment and where bedform amplitude and wavelength are moderate, as these conditions produce the largest hydraulic range and the most even density of data points within the hydraulic range. Whilst measures of bedform amplitude and wavelength are likely to reflect the influence of a wide range of morphological features on the hydraulic environment, such as the presence of large woody debris or side channels that are only connected above certain discharge thresholds, in some reach types, for example lowland chalk streams, biological factors, such as the growth of in-stream vegetation may also create seasonally-dependent hydraulic patches that are not accounted for by morphological factors (e.g. Gurnell et al., 2006). This underpins the need for good site knowledge when selecting the optimal number of hydraulic patches. Further research is needed to evaluate the effect of channel topography on the number of hydraulic patches and it is recommended to first assess reaches with contrasting morphology, such as a step-pool reach and a low gradient plane-bed reach, that traditionally have not been widely studied. Recent research has suggested ways of defining and quantifying morphological diversity (e.g. Bartley & Rutherford, 2005) which could be useful to evaluate the relationship between morphological diversity and hydraulic patch diversity. This is recommended as an area for future research.

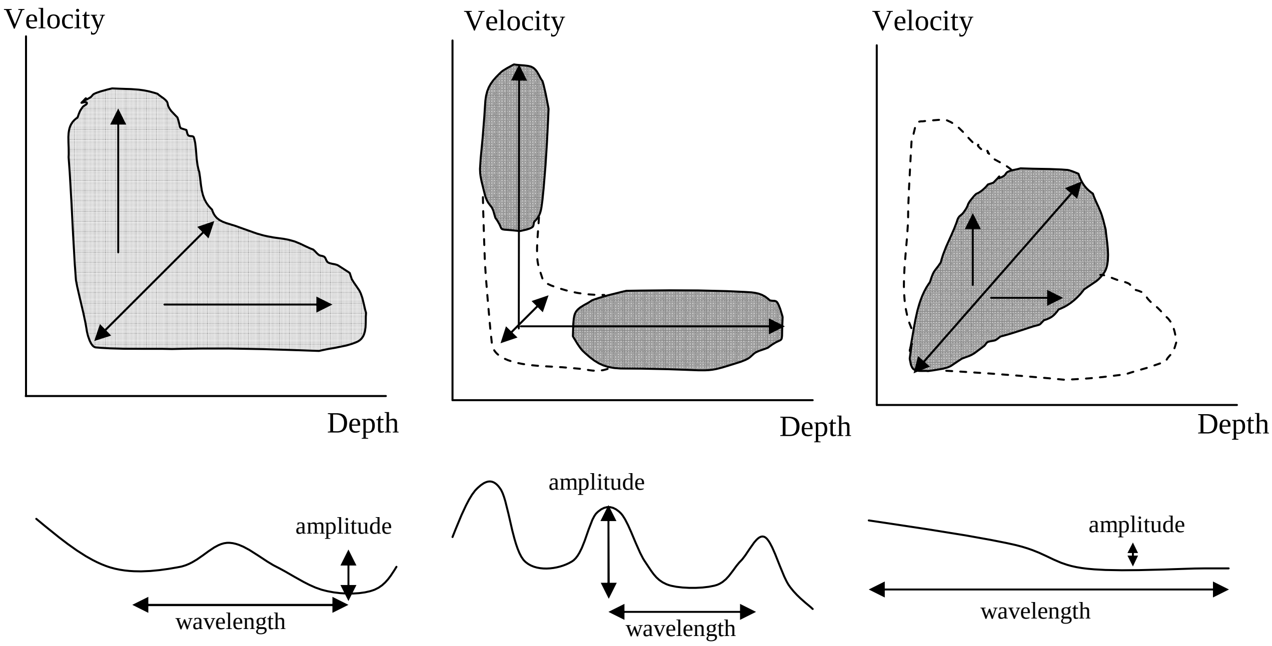


Figure 6.1 Conceptual diagram illustrating how the shape of the hydraulic data distribution, the density of points within it and the amplitude and wavelength of bedforms (long-profile) vary in (a) a pool-riffle reach, (b) a step-pool reach and (c) a channelised/plane-bed reach. The depth of shading reflects the density of points within the distribution with areas enclosed by dashed lines indicative of sparsely populated regions of the distribution. Arrows illustrate the relative influence of longitudinal and lateral topographic variations on the shape of the distribution in the upper diagrams and the relative size of bedform wavelength and amplitude in the lower diagrams.

The transition zone, which represented areas between hydraulic patches characterised by classification uncertainty occupied between 18-30% of the reach at each site-flow combination (Obj. 1b). **The delineation of the transition zone represents an application of the ecotone concept to the in-stream environment at a smaller spatio-temporal scale than has been considered before**. Ecotones are defined as transitions between relatively homogeneous patches (Ward & Wiens, 2001). Previously the concept has been applied to the longitudinal erosional/depositional ecotone between riffle and pool units (Ward & Wiens, 2001), the lateral aquatic-terrestrial “moving littoral” ecotone between the channel and its floodplain during a flood pulse (Junk et al., 1989) and the vertical hyporheic ecotone between groundwater and surface water (Williams et al., 2010). In-stream hydraulic ecotones – transitions between relatively homogeneous hydraulic patches - have never been explicitly defined, but rather incorporated into the hydraulic range of hydromorphic/hydraulic units (Figure 6.2 a). This not only increases the heterogeneity associated with each unit but represents the hydraulic continuum with spatially discrete units that have crisp, linear boundaries which exaggerate internal homogeneity and underestimate the spatial extent over which conditions change between units. Hydraulic and/or ecological sampling strategies designed to test the distinctiveness of hydraulic/hydromorphic units typically target the core of hydromorphic/hydraulic units in recognition of the uncertainty of hydraulic characteristics near boundaries. A more accurate model of the continuum represents rapid hydraulic gradients as areal zones in their own right occurring between relatively homogeneous patches (Figure 6.2 b). This type of model is made possible using fuzzy cluster analysis to classify the hydraulic environment.

The transition zones, or in-stream hydraulic ecotones, delineated in this study extended for 10^{-1} – 10^{1}m in longitudinal and lateral dimensions, although it is acknowledged that this was a function of the defuzzification rules used. They described gradients in depth and/or velocity and were bounded by two or more hydraulic patches. Hydraulic ecotones represent an integral part of the shifting habitat mosaic that persist for 10^{-5} -10^{-2} years in response to variations in discharge. In addition to segregating relatively homogeneous hydraulic patches and contributing to hydraulic diversity in-stream ecotones may be indicative of changes in community assemblages (i.e. modify the flow of organisms) (Wiens, 2002) and/or provide hot-spots for biodiversity. The availability of methods to delineate in-stream ecotones/hydraulic boundaries is a necessary pre-cursor to adopting a landscape ecology approach to riverine assessment and provides new opportunities to explore the potential ecological significance of in-stream ecotones.

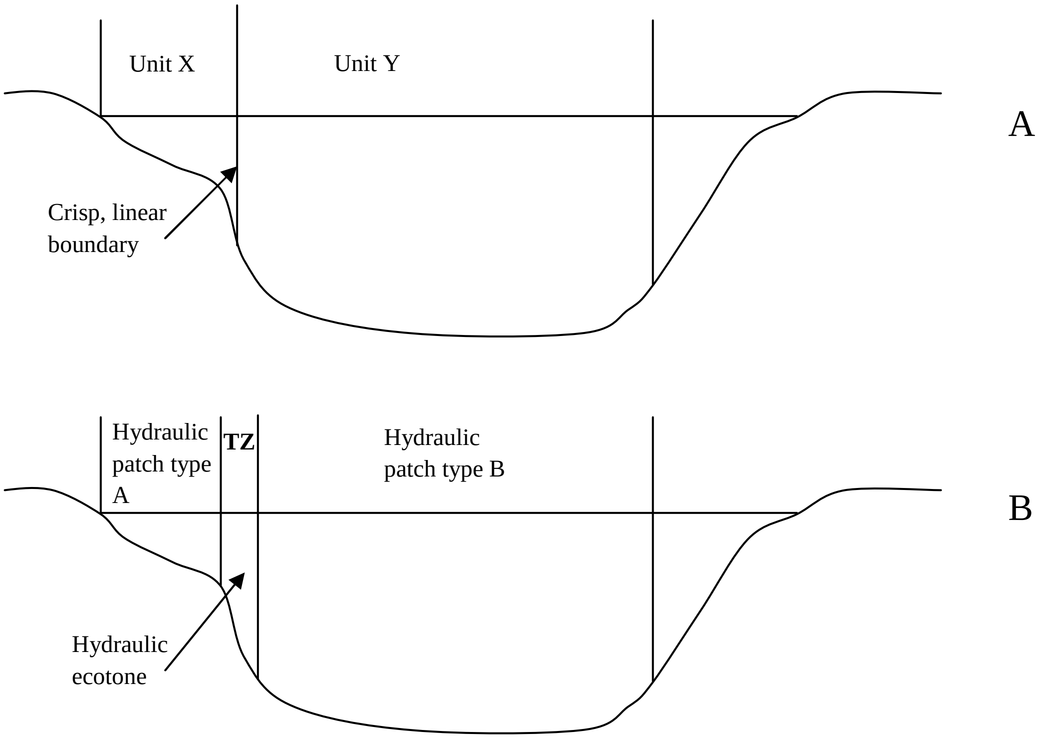


Figure 6.2 Classification of the lateral hydraulic continuum with a) hydromorphic/hydraulic units with crisp, linear boundaries (top) and b) hydraulic patches and transition zones.

The **hydraulic patch:transition zone ratio varied between 70:30 to 80:20** in all three reaches. Discharge did not have a clear effect on this ratio. Instead it is likely to vary in response to differences in reach morphology. As discussed above, the combination of high bedform amplitude and short wavelength evident in a high gradient step-pool reach type that produces a relative sharp, narrow transition between hydraulic patches may result in a 90:10 hydraulic patch: transition ratio, whereas the combination of low bedform amplitude and long wavelength in a low gradient channelised or plane-bed reach which produce very gradual changes in hydraulics may be characterised by relatively large transition zones and a 60:40 ratio. This suggested general trend however, is just that; it is likely that the number of hydraulic patches and the hydraulic patch:transition zone ratio will vary within reach types as well as among them. It is recommended to apply the same approach (and the same defuzzification rules) to hydraulic data from a selection of reaches that fall along a morphological/energy continuum to test this assumption.

The assessment of the effect of discharge on the hydraulic environment using cluster analysis can be approached in different ways. Previous studies (e.g. Emery et al., 2003) have generated a separate classification of hydraulic patches for each discharge surveyed. This has the advantage of being able to evaluate the aggregate hydraulic performance of fixed bedforms across a range of discharges. It provides a morphological, or bottom-up, perspective of hydraulic patch dynamics that is most relevant to channel/bedform design aspect of river rehabilitation but complicates tracking particular hydraulic conditions that biota may prefer. This study adopted an alternative method, combining hydraulic data collected at multiple discharges prior to clustering to generate a single classification of hydraulic patches relative to the total hydraulic range at the site. This has the advantage of being able to track the location and movement of the same hydraulic patch types (i.e. depth-velocity conditions) at every discharge. Assuming the hydraulic patches are ecologically significant, this could help predict the distribution of mobile biota that track their preferred hydraulic conditions across a range of flows. It is acknowledged that the total hydraulic range described by each patch type incorporated some of the effect of discharge on depth and velocity. For example, RA4 was characterised by a mean depth of 0.14m at very low flow but a mean depth of 0.41m at very high flow. Nevertheless each patch type still described clear hydraulic differences relative to the hydraulic range at any given discharge. **The approach of combining discharge data prior to clustering is recommended for ecohydraulic studies focussed on mobile biota or for assessing how flow release changes will alter the quantity and location of hydraulic patches**. The approach adopted by Emery et al. (2003) may have greater relevance for understanding the range of hydraulic conditions to which immobile biota are exposed or for evaluating the range of hydraulic conditions provided by a specific flow, for example, a minimum flow release.

The obvious limitation to the wider application of numerical classification of the hydraulic environment is the time and resources needed to collect hydraulic point data in the field over a range of discharges. It is precisely for this reason that rapid hydraulic assessment methods based on visual identification of mesoscale units have become widely adopted and remain the most expedient and cost-effective means to evaluate riverine health (Newson & Newson, 2000). Recent advances in remote sensing technology, such as unmanned aerial vehicles, terrestrial laser scanners and aquatic-terrestrial LiDAR, have made the collection of sub-centimetre resolution channel bathymetry data over large spatial extents (10-1000m reach length) possible and this can be used to run hydrodynamic models and produce accurate hydraulic data (Tamminga et al., 2014; Bangen et al., 2014). Such methods are not a panacea for hydraulic data collection; data accuracy can be affected by a multitude of factors such as water turbidity, in-stream vegetation, surface water turbulence and refraction within the water column, added to which the technology cannot be used in all locations, for example in channels where dense overhanging vegetation obscures the view of the channel or in deep pools where bed elevation cannot be ground-truthed (Marcus, 2012; Tamminga et al., 2014). Nor is the time and expertise required to post-process the data or costs of the equipment insubstantial (Schwendel et al., 2010; Milan et al., 2011; Bangen et al., 2014). However, it is likely that future research in this rapidly expanding area will address some of these challenges and may eventually facilitate the direct, rather than indirect, measurement of velocity (Carbonneau et al., 2012). In the interim, ADCP technology provides a less-costly alternative method of collecting channel bathymetry data for use in hydrodynamic models (Milan & Heritage, 2012). With the expectation that high resolution hydraulic data at the mesoscale will become widely available in the near future it is also anticipated that the application of numerical classification techniques will be become increasingly feasible at a range of spatial scales relevant to fish and macroinvertebrates. Until then numerical classification may be most useful at sites where hydraulic models already exist. The spatial analysis methods discussed in the following two sub-sections have wider application at the current time.

6.1.2 Hydraulic heterogeneity: the composition and diversity of hydraulic patches

The second objective of the research project referred to the quantification of hydraulic heterogeneity (composition and configuration of hydraulic patches) and examining its response to discharge variations. In Chapter 4 hydraulic patch richness, frequency and diversity were quantified at each site-flow combination to evaluate how reachscape composition changed in response to (seasonal) variations in discharge (Obj. 2a). It was hypothesised that (1) shallow or slow-flowing patches would dominate the reachscape at low flow but be replaced by deeper, faster-flowing patches as discharge increased, resulting in a significant difference between hydraulic patch composition at low and high flows, and (2) that maximum hydraulic patch diversity would occur at intermediate flows.

Hydraulic patch diversity increased with discharge in all three reaches and peaked at relatively high flow (Q22 at River Arrow, Q23 at the Leigh Brook and Q38 at River Salwarpe) (Figure 4.6). Hydraulic patch diversity was most sensitive to discharge variations at the Leigh Brook where significant increases in diversity occurred between very low-low, low-moderate and moderate-high flows. At the River Arrow significant changes in patch diversity only occurred at high and very high flows. The discharge at which maximum hydraulic diversity occurred at each site was also the discharge threshold for compositional change, below which the shallowest and slowest hydraulic patch types dominated the reach and above which the deepest and fastest patch types dominated the reach. Flows associated with low diversity were characterised by data that were densely distributed in a small region of the combined discharge data range from which the hydraulic patch classification was derived. For example, data from the River Arrow at very high flow was densely distributed in two small areas of the combined discharge data distribution (Figure 6.3). Likewise, data from the Leigh Brook at very low flow was densely distributed in the lower left region of the combined discharge data distribution (Figure 6.4). This supports Clifford et al.'s (2002) finding that it is the increase in the unevenness of the data distribution rather than a change in the hydraulic range that affects diversity.

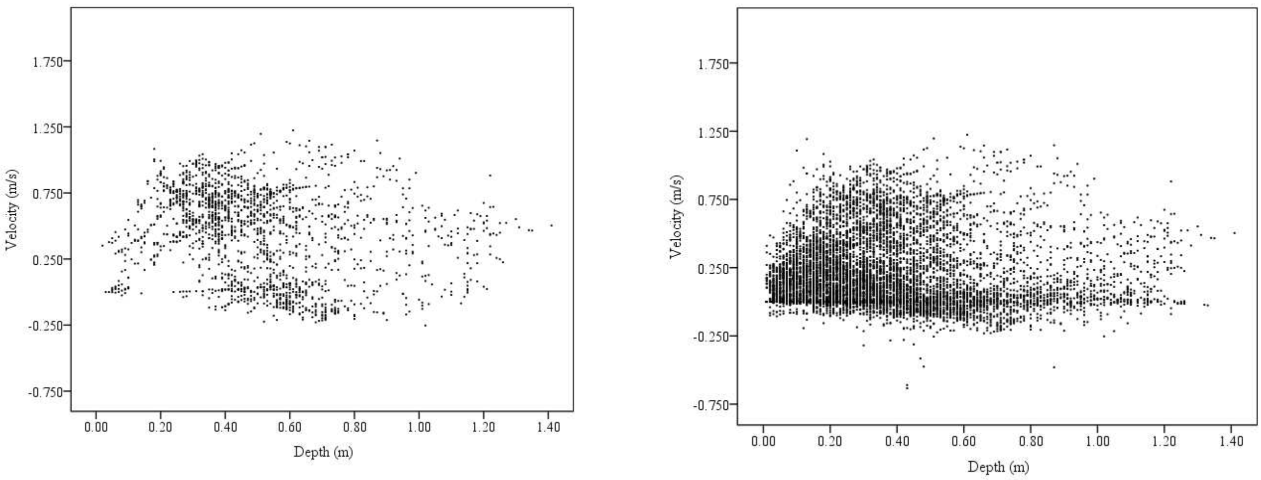


Figure 6.3. Comparison of the hydraulic data distribution at very high flow (left) relative to the data distribution of the combined discharge dataset (right) at the River Arrow

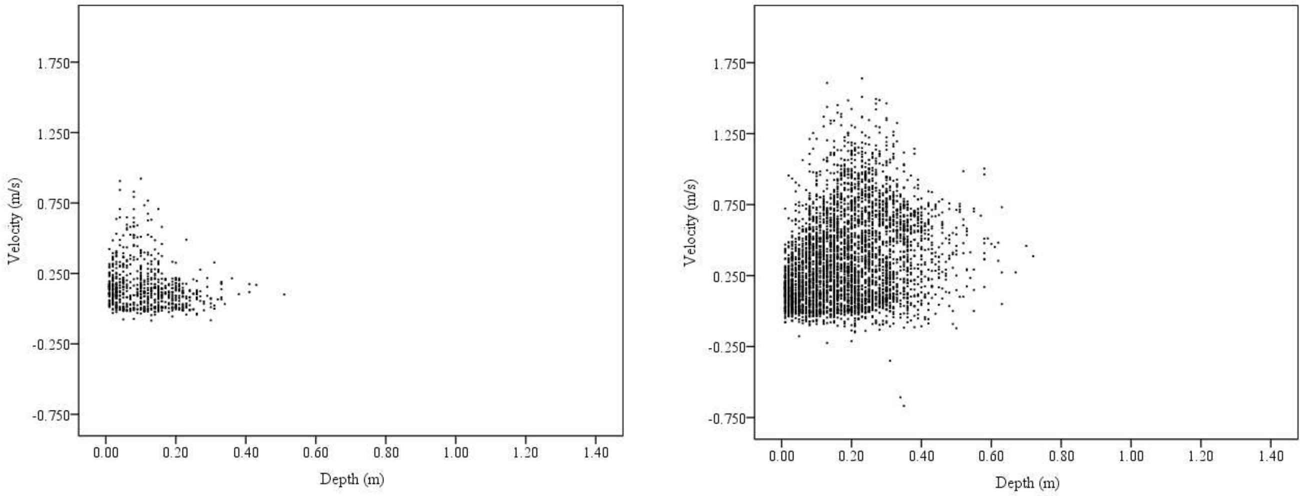


Figure 6.4. Comparison of the hydraulic data distribution at very low flow (left) relative to the data distribution of the combined discharge dataset (right) at the Leigh Brook.

The general trend for hydraulic patch diversity to increase with discharge and the occurrence of maximum hydraulic patch diversity at high flow observed in this study contrasts with the findings of hydraulic biotope studies which have reported maximum diversity at low to moderate flows (e.g. Padmore, 1998; Dyer & Thoms, 2006). Heritage et al. (2010) noted that high biotope diversity at high flows depended on the availability and inundation of morphologically diverse marginal areas. The difference in the discharge at which maximum hydraulic patch diversity occurs likely reflects the fact that the influence of relative roughness (substrate and bedform topography) on water surface characteristics, by which hydraulic biotopes are identified, is drowned out at moderate flows whereas its influence on depth and velocity within the water column, by which hydraulic patches are defined, persists at high flows. As such, **hydraulic patches provide a more robust approach for evaluating hydraulic diversity over a wider range of flows which could inform our understanding of high flow hydraulic reference conditions and help predict the impact of flow management decisions, in particular abstraction, on hydraulic diversity**. For example, these results demonstrate the importance of the magnitude component of the flow regime in delivering hydraulic diversity.

Patch richness (number of patch types) was largely invariant to discharge variations at all three sites, the only change occurring at the Leigh Brook where patch richness increased from three to five hydraulic patches between very low and low flow. The invariant trend observed at the River Arrow supports Emery et al.'s (2003) suggestion that hydraulic richness remains high across a range of flows where bedform amplitude is well-defined. However Emery et al.'s (2003) suggestion that hydraulic patch richness would decrease with discharge in reaches with subdued bedforms owing to bedforms being drowned out at lower discharge thresholds, was not shown to be the case in this study, instead the opposite trend was observed at the site with the smallest bedform amplitude (Leigh Brook). In summary, the **results support the literature on pool-riffle hydraulics that shows the hydraulic environment remains heterogeneous at flows below bankfull (Keller et al., 1971; Clifford & French, 1992) and provide further evidence that substrate and bedforms of varying amplitude exert an influence on the hydraulic environment over a wide range of flows**.

It is acknowledged that the different method for generating hydraulic patch classifications and the addition of water depth as a defining variable of hydraulic patches may in part explain these differences. Had hydraulic data from individual flows been clustered separately, as Emery et al. (2003) did, a decline in patch richness may have been observed at high flows at all sites. As an example, the very high flow data distribution at the River Arrow was characterised by two distinct regions of greater point density (Figure 6.5a). These regions distinguished between the marginal recirculation zones (RA3, shallow-very slow) and the thalweg in all but the deepest areas of the channel (RA4, moderate-fast). Had these data been clustered independently of data from all other flows is likely that a two cluster classification describing these two regions of high density would have been optimal, with the sparsely populated remainder of the hydraulic range being classified as transitional. This would have resulted in a slight decrease in patch richness, as Emery et al. (2003) suggested and a lower diversity, as Clifford et al. (2002) suggested, although this would have been true of all the sites, not just those with smaller bedform amplitude. Instead the classification was derived from data combined from multiple hydraulic surveys which meant that whilst the dominant pattern of RA3 and RA4 was reflected, a higher degree of diversity across the full hydraulic range was represented as hydraulic patches rather than transition zones (Figure 6.5b). This provided some useful insights into the hydraulic environment at high flow; it showed that the influence of the pool bedform (topographic low) on the hydraulic environment was preserved near the margins (RA5) but not at the channel centreline, where velocity increased markedly, even though depth was greatest here. The classification also differentiated between two types of marginal conditions at very high flow – the recirculation zones characterised by upstream flows eddying out from the thalweg (RA3) and the shallow-slow areas (RA2) in newly-inundated areas of the channel. This distinction is important not only because shallow-slow conditions provide high flow refugia (Lancaster & Hildrew, 1993) but because these ecologically relevant hydraulic conditions are often overlooked or underrepresented in traditional transect-level hydraulic biotope surveys (Padmore, 1998). In summary, the way in which hydraulic diversity is defined can affect the trends observed. **Further research to test the ecological relevance of hydraulic patches is needed to guide the selection of appropriate methods.**

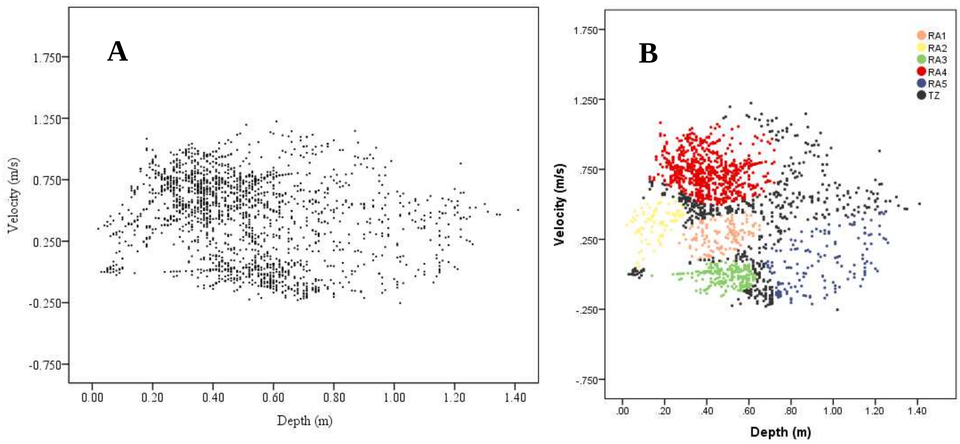


Figure 6.5 Scatterplot of the data distribution at very high flow at the River Arrow shown (a) without and (b) with the hydraulic patch classification.

Changes to composition were also quantified in terms of spatial and temporal variations in patch richness density (PRD). This provided new evidence about hydraulic diversity at the local scale. Spatial variations in PRD showed a complex response to flow and no systematic trends could be determined across the three sites although a difference in mean PRD was evident. Mean PRD was highest at the Leigh Brook and decreased at the River Salwarpe and River Arrow respectively. This trend mirrors the decrease in substrate size (D_{50}) and substrate roughness (3.5 x D_{84}) between the sites, suggesting that **mean PRD provides a measure of the influence of substrate on local hydraulic diversity**. As substrate size and heterogeneity increased, so too did local hydraulic diversity. PRD has not been used to describe hydraulic composition in any other studies so no clear ecological interpretations are available. However as the index showed a high degree of spatial and temporal variability it is suggested that it may provide useful contextual information with which to explain the patchy nature of biotic assemblages.

The observed effect of discharge on hydraulic diversity has several implications for flow management. Where river managers aim to maximise hydraulic heterogeneity, these results suggest that considerably higher flow magnitudes (<Q40) are needed to support maximum hydraulic *patch* diversity than are needed to achieve maximum hydraulic *biotope* diversity (>Q50). Current UK policy on abstraction allows a greater proportion of high flows to be abstracted on the basis that this does not have an adverse ecological effect (Acreman et al., 2008). This research suggests that large abstractions from high flow magnitudes would have the greatest impact on hydraulic diversity. Under the building block methodology for setting environmental flows a minimum, ecologically acceptable baseflow condition is supplemented with higher flows that have particular ecological or geomorphic functions at targeted times of the year, e.g. high flows during salmon migration season (King & Tharme, 2000). The results of this study show that the presence of fast-flowing hydraulic patches (RA4, RS4, LB4) was dependent on high flows. They occupied less than 5% of the reach at flows >Q70 which only increased to ≥10% when flows exceeded Q55. This suggests that the maintenance of moderate flows is necessary to provide hydraulic conditions for species with fast velocity preferences; methods other than flow management would be required to improve the provision of fast-flow refugia at low flows. Similarly the availability of shallow-slow refugia during high flows dropped below 20% of the reachscape at flows >Q40 and below 5% at flows >Q20 where the bank morphology did not provide bars that could be newly inundated as flows increased. The impact this may have on biota should be considered when releasing flushing flows or designing the magnitude of high flow pulses of a flow regime. The results showed that all discharge variations are likely to result in a change in hydraulic patch diversity, although more significant changes over a wider range of flows may occur at sites with low bedform amplitude. The hydraulic response of a river to changes in flow regime cannot be predicted from channel type alone; more detailed morphological information should be taken into consideration when designing flow regimes. Assuming the ecological relevance of hydraulic patches, even small increases in minimum flow allocations or small reductions in abstraction over the low flow range could be ecologically beneficial as hydraulic patch diversity increased between very low and moderate flow at all sites regardless of differences in bedform amplitude. However to date research on the effects of discharge on the hydraulic environment has focussed on the effects of natural flow variations on hydraulic diversity in pool-riffle reaches (e.g. Clifford et al., 2002; Emery et al., 2003; Pasternack et al., 2008; Harvey & Clifford, 2009). Additional research in regulated reaches and other reach types would provide a useful comparison of the effects of flow alteration on patterns of hydraulic diversity.

6.1.3 Hydraulic heterogeneity: patch configuration, patch change and the shifting mosaic

The second objective of the thesis which referred to quantifying hydraulic heterogeneity was also addressed in Chapter 5, but here the focus was on evaluating the configuration of hydraulic patches. The geometry and spatial arrangement of same-type patches (Obj. 2b), the change in location of same-type hydraulic patches (Obj. 2c) and the spatial arrangement of all hydraulic patches in the reach (Obj. 2d) were quantified at each site-flow combination to evaluate the effect of discharge variations on configuration. It was hypothesised that (1) patch shape would be most complex at low flows and become regular and linear at high flows and (2) that reachscape configuration would be characterised by interspersed patch types at low flows and aggregated and connected patches at high flows.

Flow-related changes to the geometry and spatial arrangement of same-type patches (at the class-level) were, in the most part, relatively small, either oscillating within a small range of values (e.g. mean patch shape complexity of all patch types at the River Arrow) or remaining relatively invariant (e.g. mean patch length of all patch types at the Leigh Brook) (Obj. 2b). A limited number of large responses to discharge were observed in the configuration of fast-flowing patches, such as the rapid increase in mean area and length of RA4 patches at very high flow, the disaggregation (i.e. fragmentation) of RA4 and RS3 patches at low flows, and the increase in mean distance to the nearest-neighbour patch for RA4, RS4 and LB4 patch types at very low flow (Obj. 2b). The other large changes to same-type patch configuration, such as the decrease in mean patch area and length of RA5 and RS5 at low and high flows were caused by very local changes in the hydraulic environment (i.e. the appearance of single pixel patches) skewing the metric and could not clearly be ascribed to the effect of discharge variations (Obj. 2b). This highlights the importance of examining a plan of the spatial distribution of the hydraulic patch mosaic when interpreting the value of each spatial metric. Mean proximity was the most variable aspect of configuration for all patch types at all sites; the shallow and/or slow patch types that dominated each reachscape at low flow became less abundant and more fragmented as discharge increased and *vice versa* for fast-flowing patch types (Obj. 2b). Reachscape configuration (the arrangement of all hydraulic patches) was also relatively invariant to discharge variations (Obj. 2d). All five reachscape configuration metrics exhibited minor variations within a relatively small range of values at all flows and no statistical differences in reachscape configuration were evident between flows (Obj. 2d). Hydraulic patch turnover (change in location) varied between patch types however two general responses were evident (Obj. 2c). The fastest hydraulic patch types at all three sites (RA4, RS4, LB1 & LB4) were very spatially dynamic (high turnover) and expanded rapidly into the thalweg as discharge increased (Obj. 2c). The exception to this rule was RS2 (shallow-fast) which was spatially dynamic but moved from the thalweg at low flow into the margins at high flow. All other patch types occurred in relatively fixed locations in the channel, exhibiting small-moderate levels of turnover and a gradual change in location as discharge increased (Obj. 2c).

The relatively small changes observed in all aspects of configuration in response to discharge variations suggest that the spatial pattern, or configuration, of the hydraulic patch mosaic is determined by channel morphology which remains stable between channel forming discharges. Minor discharge-related variations in configuration do occur, most noticeably at very low flow when the potential influence of substrate and bedform topography to create spatial hydraulic heterogeneity had not been fully realised/‘activated’ by discharge and at very high flow when bedform containment on the hydraulic environment started to decline. As such configuration will vary within a small range of values across the whole range of flows. Whilst the overall spatial pattern of hydraulic patches was relatively invariant to discharge, the hydraulic character associated with patches in the mosaic did change, reflecting the variable hydraulic performance of bedforms under different flow conditions. This was evidenced by the transitioning between patch types at particular locations in the channel. For example, a hydraulic patch was clearly associated with the area of deposition at the downstream extent of the River Salwarpe reach at all flows, however its character transitioned between RS3, TZ and RS2 as discharge increased. Similarly, the character of the hydraulic patch associated with the downslope area immediately upstream of the large pool in the River Arrow reach changed from RA3 (moderate-very slow) at low flows to RA1 (moderate-slow) at moderate flow and to RA4 (moderate-fast) at high flows.

The results support the idea that hydraulic patch configuration and composition are in dynamic equilibrium and oscillate within a small range of values bounded by the fixed template determined by channel morphology (Figure 6.6). Composition is more temporally variable than configuration because it can vary within a relatively invariant configuration (Figure 6.7). **These findings support Gostner et al.'s (2013) conclusion that geomorphic diversity creates the spatial template for hydraulic heterogeneity and discharge creates the temporal template for hydraulic heterogeneity. Without further research it is not possible to state with any certainty how the different aspects of patch configuration in other reach types would respond to discharge variations.**

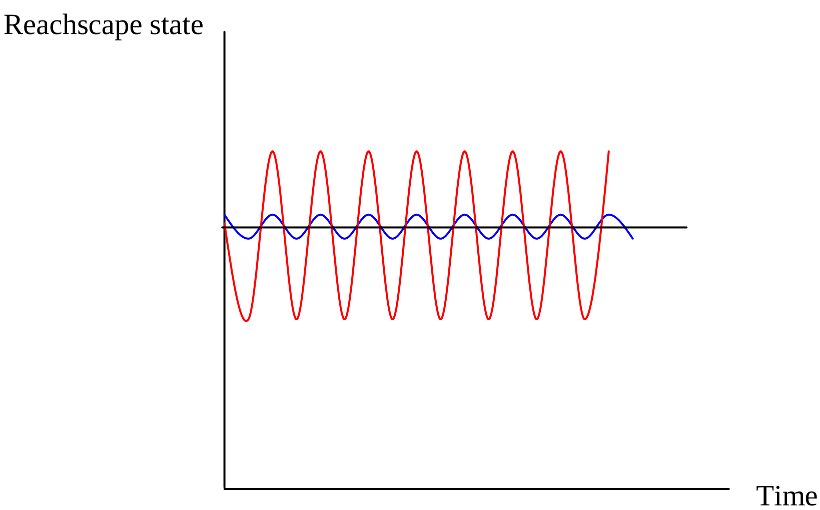


Figure 6.6. Conceptual diagram illustrating the dynamic equilibrium of hydraulic patch composition (red line) and configuration (blue line) in relation to the relative stability of bedform morphology (black line) during sub-bankfull flows. Variations in composition and configuration are explained by variations in flow.

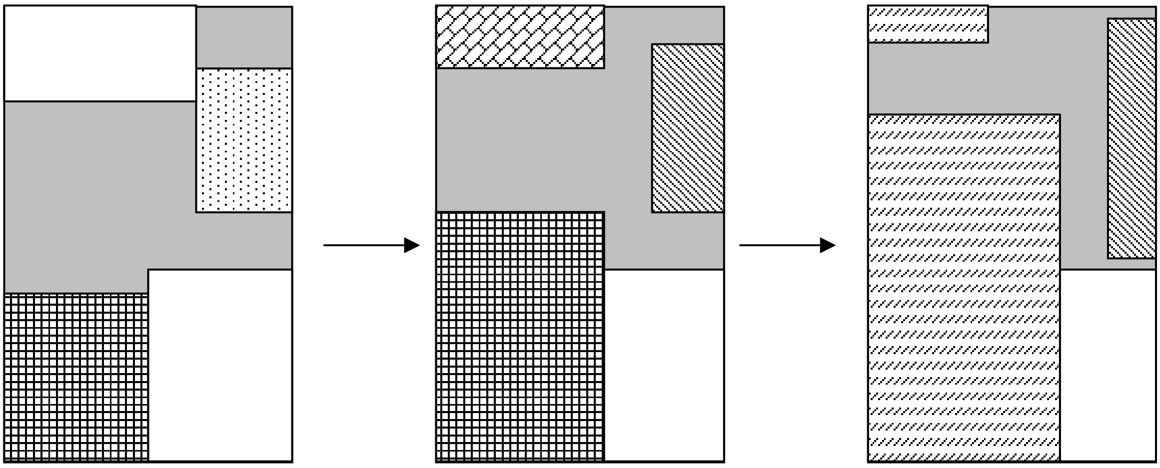


Figure 6.7 Schematic diagram illustrating how large changes in composition (different patch types are indicated by shading/cross-hatching) can occur within relatively minor variations in the configuration of a patch mosaic.

This study quantifying hydraulic patch configuration is the first of its kind in a UK river and as such there are no published results with which to make direct comparisons. Thoms et al. (2006) found that landscape shape index and patch shape complexity were significant descriptors of the configuration of velocity patches between three weir pool reaches in the Murray River (Australia). Values varied most at low flows but converged at high flow, however site explained more differences in configuration than flow. Whilst the relative importance of LSI and patch shape complexity was not evident in this study, this might be explained by the very different morphology of the reaches used in this study. In fact the difference in configuration between this and Thoms et al’s (2006) study strengthens the argument that configuration is a reflection of reach morphology. Arscott et al. (2002) investigated the spatial configuration of aquatic habitats before and after flood flows and seasonal flow pulses in a headwater braided channel in the Tagliamento River. Their results showed that at the temporal scale of flood flows habitat composition and configuration were invariant but turnover changed by 62%, leading the authors to conclude that the system provided an example of the shifting mosaic steady state model of landscape equilibrium. However at the temporal scale of sub-bankfull flow pulses aquatic habitat composition did vary and was correlated with water level, suggesting that systems are dynamic at a spatio-temporal scale nested within the “steady-state” of the shifting mosaic. The results presented in this study, whilst conducted in a very different river system, also suggest that a level of dynamism is present in the hydraulic environment. However, at the spatial scale of hydraulic patches and the temporal scale of sub-bankfull discharge variations the results do not support the shifting mosaic steady-state model because the proportion of hydraulic patch types varied with discharge.

The potential influence of the study's limitations on the results must be acknowledged. Firstly, the behaviour of configuration metrics may be more reliably inferred from examination of longer reaches as the frequency of each patch type increases and the relative influence of single-pixel patches decreases. The identification of similarities or differences in same-type patch configuration (class level) between sites was confounded by the site-specific classification of hydraulic patches. Ideally the same hydraulic patch classification would be used at each site to isolate changes in spatial configuration more effectively. Furthermore, the small number of sites used in this study, in combination with the fact that reach morphology differed between sites, meant that there was not enough statistical power to satisfactorily test for discharge-related differences in configuration. Ideally data from >3 very similar pool-riffle reaches would be required for reliable inter-flow comparisons.

Although the observed changes in configuration were small in absolute terms they may be ecologically significant. For example, small increases in patch area may make a hydraulic patch useable by biota and change its ecological status from 'potential' to 'active'. Conversely, a small decrease in the distance to the nearest same-type patch may enable migration from one patch to another, changing the status of the first patch from 'active' to 'degrading'. As such this work has relevance to the application of the patch dynamics framework to the in-stream environment (White & Pickett, 1985; Townsend, 1989, Fausch, 2002). However **ecological research on the relevance of different spatial metrics and the sensitivity of biota to changes in metric values as applied to in-stream habitats is urgently required**. It is suggested that particular metric combinations are likely to have the most ecological relevance. For example, turnover statistics become much more relevant when considered in combination with the distance the patch moves; 100% turnover within a small distance is likely to be less challenging to an organism than 100% turnover over a large distance where the distance to the new patch may exceed the organism's scale of reference (Figure 6.8). Similarly, considering the degree of contrast between patches in combination with how interspersed they are is likely to have greater ecological relevance than just a measure of interspersion as the degree of contrast is likely to affect whether an organism can cross between two interspersed patches, as illustrated in Figure 6.8. It was also clear from the results that the ecological relevance of some metrics, such as mean proximity, can *only* be interpreted meaningfully in combination with others. For example, high mean proximity at the class scale can indicate two large patches close together or many small patches (occupying the same proportion of the reachscape in total) close together. Where patch size is a limiting factor, the latter reachscape may be less ecologically favourable even though proximity is the same. It is suggested that future research is directed towards identifying **combinations of metrics** that explain the variability in biotic distributions.

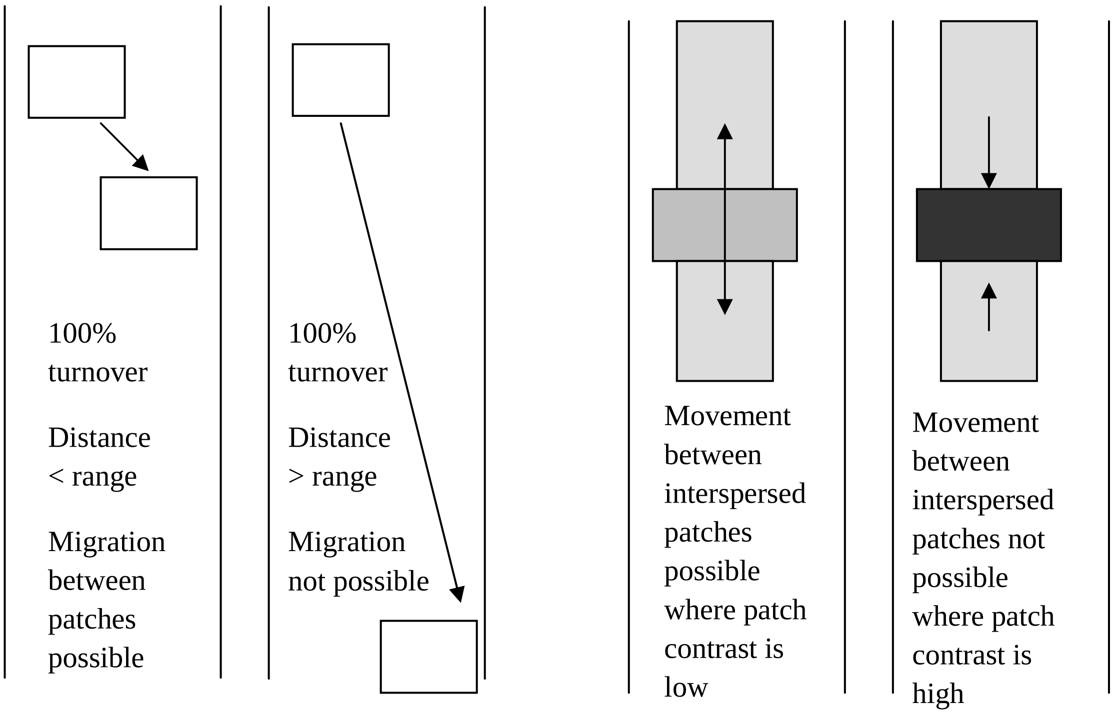


Figure 6.8. Examples of ecologically favourable (left) and unfavourable (right) patch configurations, distinguishable only in terms of combinations of patch configuration metrics.

Until further research investigating the ecological relevance of patch configuration is conducted the implications of these results for flow management are not clear.

Although configuration did not change much with discharge some small changes to aspects of configuration may prove to be ecologically significant for individual species, as discussed above. What is favourable for one species may be unfavourable for another so the management of flow to produce the ‘optimal’ patch configuration would depend on the conservation objectives of the river in question. If most biota are adapted to benefit from small, frequent variations in configuration, flow regime alterations that affect the frequency and rate of change of flow variations rather than the flow magnitude *per se*, may have the largest ecological impact. For example, a hydropeaking regime may create a very static configuration of hydraulic patches during minimum compensation releases which may favour a certain proportion of species, increasing their abundance to the detriment of others, followed by a large and rapid change in configuration during peak releases that biota cannot adjust to quickly enough as it is rarely experienced under natural flow conditions, resulting in a sharp decline in abundance.

Figure 6.7 provides a speculative model of how the number of hydraulic patches, flow-related variations in composition and the complexity of patch configuration might vary in three morphologically contrasting reach types. The upper set of diagrams illustrate the potential shape of the hydraulic data distribution at low (solid line) and high (dotted line) flow, and how this might be classified into hydraulic patch types. The lower set of diagrams illustrate where each hydraulic patch type is likely to occur within the reach (planview) at low flow. In reach type 1, a channelised trapezoidal channel with no longitudinal topographic variation and very minimal lateral variation at the margins, it is suggested that at low flow the bivariate depth-velocity distribution will be characterised by a very narrow range of depths and a relatively small range of velocities that differentiate between hydraulic conditions at the margins and everywhere else, resulting in a simple 2 patch classification. It is likely that the relative proportion of hydraulic patch 2 (slower, marginal conditions) will decrease as flows increase, resulting in a slight decrease in diversity. Patches are likely to be arranged in a simple, linear configuration parallel to the channel that is invariant with flow. Channel type 2, a steep, high energy step-pool reach, is likely to support an L-shaped hydraulic data distribution characteristic of reaches with a high degree of longitudinal topographic variation, as explained in Section 6.1.1. It is probable that this channel type supports an intermediate number of hydraulic patch types broadly corresponding with the step (HP1), the scour pool (HP2), the pool tail (HP3) and the pool margins (HP4). Consequently patch diversity will be higher than in the channelized reach. As the bed morphology is very pronounced it is likely to exert hydraulic control over a very wide range of flows. As a result there is likely to be minimal shift in the distribution at high flows and no additional high flow-specific patch types. However small changes in composition resulting from a reduction in the proportion of the reach occupied by hydraulic patches at the pool margins and pool tail are likely at high flow. The higher number of patches increases the density of edges and the complexity of the reachscape (LSI). It is also probable that the distance to nearest same-type patch (ENN) and the overall proximity between all patches (PROX) will be relatively high in this type of reach, owing to the small bedform wavelength. It is proposed that Type 3, the pool-glide-run-riffle reach, will be the most spatially diverse and temporally dynamic, for the reasons outlined in Section 6.1.1-6.1.3. Indeed, recent research has highlighted the spatial diversity of morphological units in gravel-cobble rivers (Wyrick & Pasternack, 2014). It is openly acknowledged that the model outlined in Figure 6.7 is highly speculative and based on the findings from a small number of reaches evaluated in this study. Further research is needed to test these suggestions and extend the model to other reach types.

6.2 Significance of the work to river habitat survey methods, instream flow modelling and river rehabilitation

River habitat surveys are typically carried out at a single low flow when rivers are wadeable and the channel bed and banks are more visible (Raven et al., 1997). Consequently, relatively little is known about the high flow environment. The results presented in this study show that significant changes in hydraulic composition and diversity occur at high flows which are not captured by standard methods of assessment. As a result, hydraulic diversity may be underestimated. However, as previously discussed, the lack of robustness of the hydraulic biotope classification at high flows suggests that an alternative approach would be required to capture changes

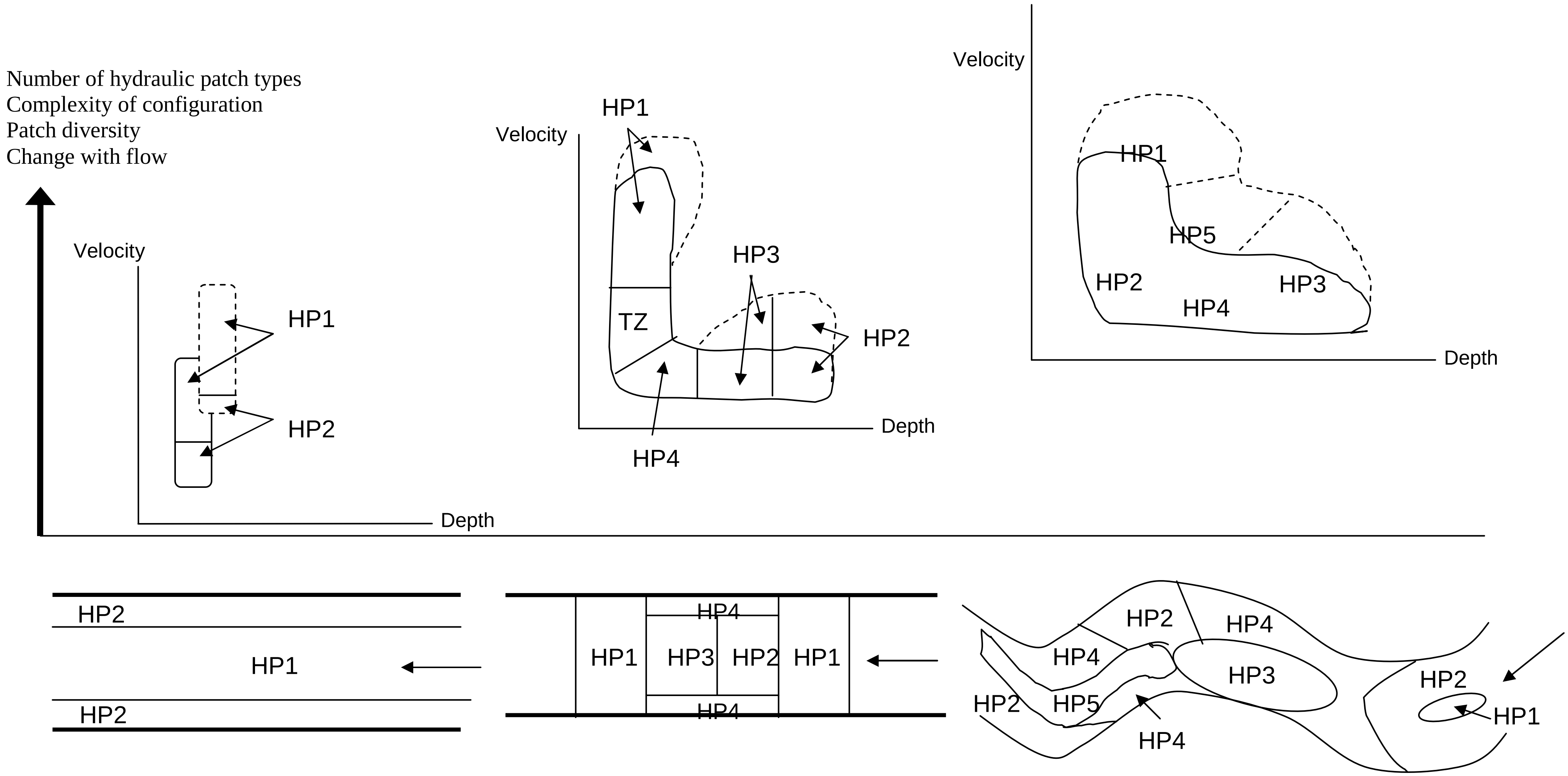


Figure 6.9. Suggested model of hydraulic patch dynamics in three morphologically contrasting channel types; (1) trapezoidal, (2) step-pool and (3) pool-glide-run-riffle. The upper diagrams indicate the shape of the hydraulic data distribution at low (solid line) and how this shifts at high (dashed line) flow. and suggests how the distribution would be classified into hydraulic patches. The lower diagrams provide a simplified planview of the spatial configuration of hydraulic patches in a theoretical reach.

to patch composition at high flows. The application of fuzzy cluster analysis to delineate hydraulic patches defined by depth-velocity characteristics and the transition zones between them provides an alternative conceptual basis for physical habitat characterisation that may provide an ecologically relevant model of spatio-temporal hydraulic diversity with which to explore biotic distributions and thus help in the reconciliation between hydromorphology and ecology.

As Wiens (2002) stated, “patch context matters”, however the spatial configuration of habitat (WUA/biotopes/functional habitat) is not routinely quantified in river habitat assessment. As a growing body of work continues to demonstrate the relevance of the spatial arrangement of habitats to biotic distributions (Lancaster, 2000; Clark et al., 2008; Kim & Lapointe, 2011; Martelo et al., 2014), the application of spatial analysis becomes increasingly important to habitat/ecohydraulic studies. Whilst some recent studies are beginning to incorporate a spatial approach (e.g. Wyrick & Pasternack, 2014) this is still the exception rather than the rule. The methods for quantifying the spatial configuration of the hydraulic patch mosaic outlined in this study could easily be applied to modelled hydraulic data generated for habitat modelling studies and would provide useful spatially explicit, supplementary information about the hydraulic environment. For example, some studies that have tried to relate weighted usable area to biomass or biotic indices to validate the IFIM/PHABSIM model but have found poor or negative correlations (e.g. Conder & Annear, 1987; Beecher et al., 2010) could, in part, be explained by differences in the spatial configuration of useable habitat patches, or by the location of useable habitat patches within the overall hydraulic patch mosaic. For example, it may be that useable spawning habitat patches must reach a certain minimum size threshold, be within a certain distance of nursery habitat and not be immediately adjacent to predator’s or competitor’s preferred habitat but near to suitable resting habitat. As such the quantification of patch area, patch contrast, proximity and interspersion would be useful. It is suggested that analysing spatial characteristics of WUA would improve the differentiation between available habitat and useable habitat, which may improve the predictive power of habitat models. The spatial configuration of hydraulic biotopes could also be evaluated if a spatially explicit map of biotopes was created at the time of a river habitat survey – this could be a useful extension of the method that would not require much additional time or effort. The results of the study suggested that spatial configuration of hydraulic patches reflects the influence of channel morphology on the hydraulic environment and is relatively invariant to changes in discharge. Therefore the fact that river habitat surveys are routinely carried out at low flows should not present too skewed an impression of biotope configuration, as it might about composition. It is suggested that quantitative information about the spatial arrangement of biotopes would strengthen the differentiation of reaches with similar biotope assemblages and provide an ecologically-relevant interpretation of the effects of channel modification on the hydraulic environment. As a first step, it would be beneficial to develop an understanding of the spatial configuration of unmodified reaches so that deviations from 'reference' conditions can be evaluated.

It is increasingly recognised that river rehabilitation needs to be planned at a catchment wide scale (Gilvear et al., 2013) however limited resources often dictate that small-scale mitigation measures are targeted at specific problems, following the principles of a 'catchment acupuncture' approach (Newson, 2010). Instream restoration measures such as barrier removal, the introduction of large woody debris or flow deflectors, adding and/or reprofiling riffles or bars, reprofiling banks and removing bank protection aim to increase hydraulic heterogeneity, guided by the principle "build it and they will come" (Palmer et al., 2005). However, post-project appraisals to evaluate whether these measures are successful are rare and often inadequate and qualitative where they are undertaken (Bernhardt et al, 2005; Jähnig et al., 2011). Where scientific, quantitative post-project appraisals have been carried out evidence of success is mixed; with some studies reporting an increase in habitat diversity and biodiversity (e.g. Gerhard & Reich, 2000) and others reporting an increase in physical heterogeneity but no ecological response (e.g. Lepori et al., 2005). Whilst it is acknowledged that increasing hydraulic heterogeneity is just one of many factors that affect a river's ecological health, having reliable and accurate tools to measure differences in the hydraulic environment before and after restoration is important. It is suggested that the spatio-temporal approach to hydraulic assessment outlined in this study provides an ideal method for pre- and post-project appraisal for several reasons. Firstly it could be applied at unmodified reaches to define the "dynamic state" (*sensu* Palmer et al., 2005) that forms the guiding image for restoration efforts. Secondly, an analysis of the spatial arrangement of hydraulic patches may help to explain why an increase in hydraulic heterogeneity does or does not improve biodiversity. Similarly an analysis of the temporal changes to the hydraulic environment might indicate over what range of flows different restoration measures are most effective. Lastly, information about the spatial configuration of the pre-restoration hydraulic environment could help guide the placement of in-stream restoration measures (e.g. large woody debris, flow deflectors) to produce the most ecologically favourable configuration of hydraulic patches post-restoration.

6.3 Further research

This study has presented a new way of representing hydraulic heterogeneity in terms of hydraulic patches and transition zones and has demonstrated how the spatial configuration and temporal dynamics of the hydraulic patch mosaic can be quantified. Whilst these methods appear promising and suggestions about their application have been made, for hydraulic patches to help integrate hydromorphology and ecology and provide useful tools for river research and management they must be related to the larger-scale context of reach morphology and be shown to describe ecologically meaningful units. Further research is needed to clarify how variations in channel morphology affect the composition and configuration of the hydraulic patch mosaic and to evaluate the ecological significance of hydraulic patches, transition zones, spatial metrics and flow-related changes to the composition of patches.

In order to better understand and quantify the relationship between channel morphology and hydraulic diversity further applications of the approach are required across a range of reach types. Additional research should aim to identify the hydraulic patch mosaic “signature” of different reach types and quantify how variations in morphological characteristics, such as bedform amplitude, bedform wavelength, cross-sectional shape and substrate size, change the composition and configuration of the hydraulic patch mosaic. This will help define reference or baseline conditions for future monitoring, help quantify, and potentially predict, the impact of channel modification on the hydraulic environment and inform river rehabilitation strategies. It is recommended that three step-pool reaches, three pool-riffle-run-glide reaches and three trapezoidal channelised reaches are sampled so that causes of variability between types can be distinguished from causes of within-type variability. Remote-sensing technology, such as UAVs, could be used to collect channel bathymetry data suitable for running a hydrodynamic model at each site. This would overcome the difficulty of collecting hydraulic data in steep headwater reaches and minimise sampling efforts. It is recommended that a wide range of spatial metrics are calculated to describe the configuration of the hydraulic patch mosaic at all sites which can be reduced to a subset that best characterises the configuration of each reach type. Developing a hydrodynamic model at each site would allow the composition and configuration at many more flows to be calculated so that flow-related changes could be understood in more detail. This would provide more information about the impact of small changes to flow which is needed for environmental flow design. Further research could specifically quantify the effect of channel modification on hydraulic patch configuration by comparing the configuration of an unmodified, partly modified and heavily modified reach from the same river.

The alternative model of hydraulic classification presented in this thesis opens up several new avenues for ecological research. Further applications of the methods described here but supplemented with concurrent collection of biological data is recommended so that the ecological relevance of hydraulic patches and transition zones can be explored. As hydraulic patches are delineated after hydraulic data has been collected it is recommended that biological samples are taken at frequent intervals along several longitudinal and lateral transects in a reach to ensure that biota from a range of hydraulic patches are collected. It is suggested that the resulting data is used to investigate the following questions. Do numerically-delineated hydraulic patches describe ecologically distinct areas of the hydraulic environment? Do transition zones act as instream ecotones and what is their ecological role? Do they act barriers to dispersal or support more diverse assemblages? Does the hydraulic patch: TZ model of the hydraulic environment help explain the patchy distribution of biota? Could it be used as a proxy for biological diversity or an indicator of ecological health? Investigation of the ecological significance of spatial metrics is also urgently required. Which metrics or combinations of metrics do biota respond to? Do small changes in configuration affect the ecological status (potential/active/degraded) of hydraulic patches? It is suggested that experimental flume work where the configuration of patches can be manipulated more easily might be the most fruitful approach for this type of analysis.

Chapter Summary

The first part of this thesis (Ch. 3) presented a novel method for classifying the hydraulic environment into relatively homogeneous hydraulic patches, defined by the joint distribution of depth and velocity, and the transition zones (boundaries) between them. The optimal classification depended on the size, shape and density of the data distribution and hence is likely to differ between reach types. The Gustafson-Kessel fuzzy clustering algorithm provided some advantages over fuzzy c-mean In this study five spatially coherent hydraulic patch types were delineated which were associated with different bedforms or geomorphic features. The transition zone occupied between 18-30% of the reach at each site-flow combination and it is suggested that these may function as instream ecotones. The hydraulic patch/transition zone model of the hydraulic environment provides a new avenue for ecological research, in terms of investigating biotic distributions and the ecological significance of hydraulic patches and transition zones.

The second part of the thesis (Ch. 4 & 5) quantified how the composition and configuration of hydraulic patches / transition zones varied in response to changes in flow. Composition varied most, with a gradual shift in dominance from shallow, slow patches to faster, deeper patches as flow increased. Hydraulic patch diversity increased with discharge, peaking at high flows (Q38-22) at all sites which, if hydraulic patches area ecologically significant will have implications for flow management strategies. Configuration varied very little with flow at both the class level and reachscape scales, suggesting that channel morphology determines the spatial template for the hydraulic patch mosaic, which is influential over a wide range of flows. Further research into the effect of channel modification on patch configuration and its stability over a range of flows is recommended.

In summary this thesis adopted a landscape ecology framework to evaluate the instream environment and demonstrated how the five elements of landscape pattern – patch quality, composition, boundaries, patch context and patch structure – can be quantified as a basis for understanding how hydraulic patterns at the reachscape scale affect ecological patterns and processes.

REFERENCES

Acreman, M.C. & Ferguson, A.J.D. (2010) Environmental flows and the European Water Framework Directive. *Freshwater Biology*, 55 (1): 32-48.

Allan, J.D. (1995) *Stream ecology: structure and function of running waters*. London, Chapman and Hall.

Allan, J.D. (2004) Landscapes and riverscapes: the influence of land use to stream ecosystems. *Annual Review of Ecology, Evolution, and Systematics*, 35: 257–284.

Amoros C., Gibert J. & Greenwood M. (1993) Interactions entre unite´s de l'hydrosyste`me fluvial. In: Amoros, C. & Petts G.E. (eds.) *Hydrosyste`mes Fluviaux*, Masson, Paris, pp. 169–199.

Amoros, C. & Roux, A.L. (1988) Interactions between water bodies within the floodplains of large rivers: function and development of connectivity. In: Schrieber, K-F. (ed.) *Connectivity in Landscape Ecology*. Proceedings of the 2nd International Seminar of the International Association of Landscape Ecology, Münster 1987, Münsterische geographische Arbeiten. pp. 125–130.

Armitage, P.D., Pardo, I. & Brown, A. (1995) Temporal constancy of faunal assemblages in mesohabitats – application to management. *Archiv für Hydrobiologie*, **133**: 367-387.

Armstrong, J.D., Kemp, P.S., Kennedy, G.J.A., Ladle, M. & Milner, N.J. (2003) Habitat requirements of Atlantic salmon and brown trout in rivers and streams. *Fisheries Research*, **62** (2): 143-170.

Arnold, S.J. (1979) A test for clusters. *Journal of Marketing Research*, **16**: 545-551.

Arnot, C. & Fisher, P. (2007) Mapping the ecotone with fuzzy sets. In: Morris, A. & Kokhan, S. (eds.) *Geographic Uncertainty in Environmental Security*, pp.19-32. New York, Springer.

Arnot, C., Fisher, P.F., Wadsworth, R. & Wellens, J. (2004) Landscape metrics with ecotones: pattern under uncertainty. *Landscape Ecology*, **19** (2): 181-195.

Arrell, K.E., Fisher, P.F., Tate, N.J. & Baston, L. (2007) A fuzzy c-means classification of elevation derivatives to extract the morphometric classifications of landforms in Snowdonia, Wales. *Computers & Geoscience*, **33**: 1366-1381.

Arscott, D.B., Tokcner, K. & Ward, J.V. (2002) Aquatic habitat diversity along the corridor of an Alpine floodplain river (Fiume Tagliamento, Italy). *Archiv für hydrobiologie*, **149**: 679-704.

Arthington, A.H., Naiman, R.J., McClain, M.E. & Nilsson, C. (2010) Preserving the biodiversity and ecological services of rivers: new challenges and research opportunities. *Freshwater Biology*, **55**: 1-16.

Ayllon, D., Almodovar, A., Nicola, G.G. & Elvira, B. (2009) Interactive effects of cover and hydraulics on brown trout habitat selection patterns. *River Research and Applications*, **25** (8): 1051-1065.

Bain, M.B., Finn, J.T. & Brooke, H.E. (1985) A quantitative method for sampling riverine microhabitats by electrofishing. *North American Journal of Fish Management*, **5**: 489-493.

Baker, E.A. & Coon, T.G. (1997) Development and Evaluation of Alternative Habitat Suitability Criteria for Brook Trout. *Transactions of the American Fisheries Society*, **126** (1): 65-76.

Balasko, B., Abonyi, J. & Feil, B. (2001) Fuzzy clustering and data analysis toolbox for MATLAB. Available from: http://www.mathworks.com/matlabcentral/fileexch ange/7486 [Accessed 12th April 2004].

Bangen, S.G., Wheaton, J.M., Bouwes, N., Bouwes, B. & Jordan, C. (2014) A methodological intercomparison of topographic survey techniques for characterizing wadeable streams and rivers. *Geomorphology* **206**: 343–361.

Banjeree, A. & Davé, R.N. (2004) *Validating clusters using the Hopkins statistic*. Proceedings of the IEEE International Conference on Fuzzy Systems, Volume 1.

Baptist, M.J. (2001) *Review on biogeomorphology in rivers: processes and scales*. CFR Project Report 3. Delft University of Technology.

Baron, J.S., Poff, N.L., Angermeier, P.L., Dahm, C.N., Gleick, P.H., Hairston Jr, N.G., Jackson, R.B, Johnston, C.A., Richter, B.G. & Steinman, A.D. (2002) Balancing human and ecological needs for freshwater: the case for equity. *Ecological Applications*, **12** (1): 1247-1260.

Bartley, R. & Rutherford, I. (2005) Measuring the reach-scale geomorphic diversity of streams: application to a stream disturbed by a sediment slug. R*iver Research and Applications*, **21**: 39-59.

Baxter, M.J. (1994) *Exploratory Multivariate Analysis in Archaeology*. Edinburgh, Edinburgh University Press.

Bayliss, A. (1999) *Flood Estimation Handbook. Volume 5: Catchment Descriptors*. Wallingford, Institute of Hydrology.

Beecher, H. A., Caldwell, B. A., DeMond, S. B., Seiler, D. & Boessow, S. N. (2010) An empirical assessment of PHABSIM using long-term monitoring of coho salmon smolt production in Bingham Creek, Washington. *North American Journal of Fisheries Management*, **30**(6), 1529-1543.

Beisel, J-N., Usseglio-Polatera, P. & Moreteau, J-C. (2000) The spatial heterogeneity of a river bottom: a key factor determining macroinvertebrate communities. *Hydrobiologia*, **422/423**: 163-171.

Beisel, J-N., Usseglio-Polatera, P.; Thomas, S.; Moreteau, J-C. (1998) Stream community structure in relation to spatial variation: the influence of mesohabitat characteristics. *Hydrobiologia*, **389** (1): 73-88.

Bellman R., Kalaba K. & Zadeh L.A. (1966) Abstraction and pattern classification. *Journal of Mathematical Analysis and Applications*, **13** (1): 1–7.

Benda, L., Poff, N.L., Miller, D., Dunne, T., Reeves, G., Pess, G. & Pollock, M. (2004) The Network Dynamics Hypothesis: how channel networks structure riverine habitats. *BioScience*, **54** (5): 413-427.

Bensaid, A.M., Hall, L.O., Bezdek, J.C., Clarke, C.P., Silbiger, M.L., Arrington, J.A. & Murtagh, R.F. (1996) Validity-guided (Re) clustering with applications to image segmentation. *IEEE Transactions on Fuzzy Systems*, **4**: 112-123.

Bernhardt, E. S., Palmer, M., Allan, J. D., Alexander, G., Barnas, K., Brooks, S., Carr, J., Clayton, S., Dahm, C., Follstad-Shah, J., Galat, D., Gloss, S., Goodwin, P., Hart, D., Hassett, B., Jenkinson, R., Katz, S., Kondolf, G.M., Lake, P.S., Lave, R., Meyer, J.L., O'Donnell, T.K., Pagano, L., Powell, B. & Sudduth, E. (2005) Synthesizing U.S. river restoration efforts. *Science*, **308**(5722): 636-637.

Bertoldi, W., Gurnell, A., Surian, N., Tockner, K., Zononi, L., Ziliani, L. & Zolezzi, G. (2009) Understanding reference processes: Linkages between river flows, sediment dynamics and vegetated landforms alon the Tagliamento River, Italy. *River Research and Applications*, **25**: 501-516.

Bezdek, J.C. & Hathaway, R.J. (2002) *VAT: a visual assessment tool of (cluster) tendency*. Proceedings of the International Joint Conference on Neural Networks. New Jersey, IEEE Press, pp2225-2230.

Bezdek, J.C. (1981) *Pattern recognition with fuzzy objective function algorithms*. New York, Plenum Press.

Bezdek, J.C., Ehrlich, R. & Full, W. (1984) FCM: the fuzzy c-means clustering algorithm. *Computers & Geosciences*, **10** (2-3): 191-203.

Bezdek, J.C., Keller, J., Krisnapuram, R. & Pal, N.R. (1999) *Fuzzy models and algorithms for pattern recognition and image processing*. Dordrecht, Kluwer Academic Publishers Inc.

Biol. 49 1230-9

Bisson, P.A., Buffington, J.M. & Montgomery, D.R. (2006) Valley segments, stream reaches and channel units. In: Hauer, F.R. & Lamberti G.A. (eds.) *Methods in stream ecology*. London Academic Press, pp.23-52.

Bisson, P.A., Nielsen, J.L., Palmason, R.A. & Grove, L.E. (1982) A system of naming habitat types in small streams, with examples of habitat utilization by salmonids during low stream flow. In: Armantrout N.B. (ed.), *Acquisition and utilization of aquatic habitat inventory information.* Proceedings of a symposium, 28-30 October 1981. Portland Oregon, American Fisheries Society.

Bockelmann, B.N., Fenrich, E.K., Lin, B. & Falconer, R.A. (2004) Development of an ecohydraulics model for stream and river restoration. *Ecological Engineering*, **22**: 227-235.

Booker, D.J. & Acreman, M.C. (2007) Generalisation of physical habitat-discharge relationships. *Hydrology and Earth System Sciences,* **11** (1): 141-157.

Boon, P.J., Homes, N.T.H. & Raven, P.J. (2010) Developing standard approaches for recording and assessing river hydromorphology: the role of the European Committee for Standardization (CEN). *Aquatic Conservation: Marine and Freshwater Ecosystems*, **20** (1): S55-S61.

Bormann, F.H. & Likens, G.E. (1979) *Pattern and process in a forested ecosystem*. New York, Springer Verlag.

Borsányi, P., Alfredson, K., Harby, A., Ugedal, O. & Kraxner, C. (2004) A meso-scale habitat classification method for production modelling of Atlantic salmon in Norway. *Hydroécologie Appliquée*, **14** (1): 119-138.

Boulton, A.J., Datry, T., Kashara, T., Mutz, M. & Standford, J.A. (2010) Ecology and management of the hyporheic zone: stream–groundwater interactions of running waters and their floodplains, *Journal of the North American Benthological Society*, **29** (1): 26-40.

Bovee, K.D. (1982) *A guide to stream habitat analysis using the Instream Flow Methodology*. Instream Flow Information Paper No. 12. FWS/OBS 82/86, Western Energy and Land Use team. Fort Collins, Colorao, U.S. Fish and Wildlife Service.

Bovee, K.D. (1996) Perspectives on two-dimensional river habitat models: the PHABSIM experience. In: Leclerc, M., Capra, H., Valentin, S., Boudreault, A. & Cote, Y. (eds.) *Proceedings of the Second International Symposium on Habitat Hydraulics, Quebec: INRS-Eau*. FQSA, IAHR/AIRH, pp. B149-162.

Bovee, K.D., Newcomb, T.J. & Coon, T.G. (1994) *Relations between habitat variability and population dynamics of bass in the Huron River, Michigan*. National Biological Service Biological Report 21. US Geological Survey.

Bradley J. Pusey and Angela H. Arthington.

British Waterways (2007) *Restoration of the Droitwich canals Non Technical Summary* [online]. Available from: http://www.britishwaterways.co.uk/media/documents/Droit_Restoration_Non_Technical_Summary.pdf [Accessed 20th October 2009].

Brookes, A. & Shields Jr., F. D. (1996) Perspectives on river channel restoration. In: Brookes, A. & Shields Jr., F. D. (eds.) *River Channel Restoration*. Chichester, Wiley.

Brooks, A.J., Haeusler, T., Reinfields, I. & Williams, S. (2005) Hydraulic microhabitats and the distribution of macroinvertebrate assemblages in riffles. *Freshwater Biology*, **50**: 331-344.

Brown, B.L. (2003) Spatial heterogeneity reduces temporal variability in stream insect communities. *Ecology Letters*, **6** (4): 316-325.

Brown, R.A. & Pasternack, G.B. (2009) Comparison of methods for analysing salmon habitat rehabilitation designs for regulated rivers. *River Research and Applications*, **25** (6): 745-772.

Buffagni, A., Crosa, G.A., Harper, D.M. & Kemp, J.L. (2000) Using macroinvertebrate species assemblages to identify river channel habitat units: an application of the functional habitats concept to a large, unpolluted Italian river (River Ticino, northern Italy). *Hydrobiologia*, **435** (1-3): 213-225.

Buffin-Bélanger, T. & Roy, A.G. (1998) Effects of a pebble cluster on the turbulent structure of a depth-limited flow in a gravel-bed river. *Geomorphology*, **25** (3-4): 249-267.

Buffin-Bélanger, T., Rice, S., Reid, I. & Lancaster, J. (2006) Spatial heterogeneity of near-bed hydraulics above a patch of river gravel. *Water Resources Research*, **42**: W04413.

Buijse, A.D., Coops, H., Staras, M., Jans, L.H., Van Geest, G.J., Grift, R.E., Ibelings, B.W., Oosterberg, W. & Roozen, F. (2002) Restoration strategies for river floodplains along large lowland rivers in Europe. *Freshwater Biology*, **47** (4): 889-907.

Bunn, S.E. & Arthington, A.H. (2002) Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental Management*, **30** (4): 492-507.

Burrough, P.A. (1989) Fuzzy mathematical methods for soil survey and land evaluation. *Journal of Soil Science*, **40**: 477-492.

Burrough, P.A. (1996) Natural objects with indeterminate boundaries. In: Burrough, P.A. & Frank, A.U. (eds.) *Geographic objects with indeterminate boundaries*. London, Taylor & Francis, pp. 3-28.

Burrough, P.A., van Gaans, P.F.M. & Hootmans, R. (1997) Continuous classification in soil survey: spatial correlation, confusion and boundaries. *Geoderma*, **77**: 115-135.

Burrough, P.A., Wilson, J.P., van Gaans, P.F.M. & Hansen, A.J. (2001) Fuzzy k-means classification of topo-climatic data as an aid to forest mapping in the Greater Yellowstone Area, USA. *Landscape Ecology*, **16**: 523-546.

Burrough, P.A., Wilson, J.P., van Gaans, P.F.M. & Hansen, A.J. (2001) Fuzzy k-means classification of topo-climatic data as an aid to forest mapping in the Greater Yellowstone Area, USA. Landscape Ecology, 16: 523-546.

Cadenasso, M.L., Pickett, S.T.A. & Grove, J.M. (2006) Dimensions of ecosystem complexity: heterogeneity, connectivity, and history. *Ecological Complexity*, **3**: 1-12.

Carbonneau, P., Fonstad, M. A., Marcus, W. A. & Dugdale, S. J. (2012) Making riverscapes real. *Geomorphology*, **137**(1), 74-86.

Carling, P. (1992) In-stream hydraulics and sediment transport. In: Calow, P. & Petts, G. (eds.) *The Rivers Handbook, Volume 1*. Oxford, Blackwell, pp.101-125.

CEN (2008) *Water quality - Guidance standard for assessing the hydromorphological features of rivers. EN 14614*. Available at: http://www.riverhabitatsurvey.org/wp-content/uploads/2012/09/CEN-2004-standard-for-assessing-hydromorphology.pdf [Accessed 2nd July 2013].

Centre for Ecology and Hydrology (CEH) (2001) *Further validation of PHABSIM for the habitat requirements of salmonid fish*. Final Report to the Environment Agency (W6-036) and CEH (C00962).

Centre for Ecology and Hydrology (CEH) (2006) *Flood Estimation Handbook CD-ROM. Version 2.0.* Wallingford, Natural Environment Research Council.

Chave, P.A. (2001) *The EU Water Framework Directive: an introduction*. London, IWA Publishing.

Chen, Q., Morales-Chaves, Y., Li, H. & Mynett, A.E. (2006) Hydroinformatics techniques in eco-environmental modeling and management. *Journal of Hydroinformatics*, **8** (4): 297-316.

Chen, Y-C. & Kao, S-P (2011) Velocity distribution in open channels with submerged aquatic plants. *Hydrological Processes*, **25** (13): 2009-2017.

Cheng, T., Molenaar, M. & Lin, H. (2001) Formalizing fuzzy objects from uncertain classification results. *International Journal of Geographical Information Science*, **15**: 27-42.

Chessman, B.C., Fryirs, K.A. & Brierley, G.J. (2006) Linking geomorphic character, behaviour and condition to fluvial biodiversity: implications for river management. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **16** (3): 267-288.

Church, M. (1992) Channel morphology and typology. In: Calow, P. & Petts, G. (eds.) *The Rivers Handbook, Volume 1*. Oxford, Blackwell, pp.126-143.

Clark, J.S., Rizzo, D.M., Watzin, M.C. & Hession, W.C. (2008) Spatial distribution and geomorphic condition of fish habitat in streams: an analysis using hydraulic modelling and geostatistics. *River Research and Applications*, **24** (7): 885-899.

Clarke, K.R. & Gorley, R.N. (2006) *PRIMER v6: User Manual/Tutorial*. PRIMER-E, Plymouth.

Clarke, K.R. (1993). Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology*, **18**: 117-143.

Clarke, S.J., Bruce-Burgess, L. & Wharton, G. (2003) Linking form and function: towards an eco-hydromorphic approach to sustainable river restoration. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **13** (5): 439–450.

Clifford, N.J., Harmar, O.P., Harvey, G. & Petts, G.E. (2006) Physical habitat, eco-hydraulics and river design: a review and re-evaluation of some popular concepts and methods. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **16**: 389-408.

Clifford, N.J., Soar, P.J., Emery, J.C., Gurnell, A.M. & Petts, G.E. (2002) Sustaining water-related ecosystems – the role of in-stream bedform design in river channel rehabilitation. In: *Proceedings of the FRIEND 2002 International Conference on Regional Hydrology: bridging the gap between research and practice*. Cape Town, 18-22 March 2002. IAHS Publication no. 274, pp. 407-416.

Clifford, N.J., Soar, P.J., Harmar, O.P., Gurnell, A.M., Petts, G.E. & Emery, J.C. (2005) Assessment of hydrodynamic simulation results for eco-hydraulic and eco-hydrological applications: a spatial semivariance approach. *Hydrological Processes*, **19**: 3631-3648.

Clifford, N.J., Wright, N.G., Harvey, G., Gurnell, A.M., Harmar, O.P. & Soar, P.J. (2010) Numerical modeling of river flow for ecohydraulic applications: some experiences with velocity characterization in field and simulated data. Journal of Hydraulic Engineering.

Conder, A. L. & Annear, T. C. (1987) Test of weighted usable area estimates derived from a PHABSIM model for instream flow studies on trout streams. *North American Journal of Fisheries Management*, **7**(3), 339-350.

Connell, J. H. 1978. Diversity in tropical rain forest and coral reefs. *Science*, **199**: 1302-1310.

Cormack, R.M. (1971) A review of classification. *Journal of the Royal Statistical Society*, **134**: 321-367.

Costa, J.E. & O'Connor, J.E. (1995) Geomorphically effective floods. In: Costa, J.E., Milner, A.J., Potter, K.W. & Wilcock, P.R. (eds.) *Natural and anthropogenic influences in fluvial geomorphology*. Washington, American Geophysical Union, pp.45-56.

Crowder, D.M. & Diplas, P. (2002) Assessing changes in watershed flow regimes with spatially explicit hydraulic models. *Journal of the American Water Resources Association*, **38** (2): 397-408.

Crowder, D.W. & Diplas, P. (2000) Using two-dimensional hydrodynamic models at scales of ecological importance. *Journal of Hydrology*, **230**: 172-191.

Crowder, D.W. & Diplas, P. (2006) Applying spatial hydraulic principles to quantify stream habitat. *River Research and Applications*, **22** (1): 79-89.

Cumberlidge, J. (2009) *Inland Waterways of Great Britain*. 8th edition. Imray, Laurie Norie & Wilson Ltd.

Cummins, K.W., Wilzbach, M.A., Gates, D.M. Perry, J.B. & Taliaferro, W.B. (1989) Shredders and riparian vegetation. *BioScience*, **39** (1): 24-30.

Cushing, C.E., McIntire, C.D., Cummins, K.W., Minshall, G.W., Petersen, R.C., Sedell, J.R. & Vannote, R.L. (1983) Relationships among chemical, physical and biological indices along river continua based on multivariate analyses. *Archiv für Hydrobiologie*, **98** (3): 317-326

Cushman, S.A., McGarigal, K. & Maile, N. (2008) Parsimony in landscape metrics: strength, universality and consistency. *Ecological Indicators*, **8** (5): 691-703.

Dauwalter, D.C., Splinter, D.K., Fisher, W.L. & Marston, R.A. (2008) Biogeography, ecoregions, and geomorphology affect fish species composition in streams of eastern Oklahoma, USA. *Environmental Biology of Fishes*, **82** (3): 237-249.

Davé, R.N. (1996) Validating fuzzy partition obtained through c-shells clustering. *Pattern Recognition Letters*, **17** (6): 613-623.

Davey, A.J.H., Boker, D.J. & Kelly, D.J. (2011) Diel variation in stream fish habitat suitability criteria: implications for instream flow assessment. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **21** (2): 132-145.

Davis, J.A. and Barmuta, L.A. (1989) An ecologically useful classification of mean and near-bed flows in streams and rivers. *Freshwater Biology*, **21**: 271-282.

Death, R. & Winterbourn, M. (1995) Diversity patterns in stream benthic invertebrate communities: the influence of habitat stability. *Ecology*, **76** (5): 1446-1460.

Degerman, E., Nasland, I. & Sers, B. (2000) Stream habitat use and diet of juvenile (0+) brown trout and grayling in sympatry. *Ecology of Freshwater Fish*, **9** (4): 191-201.

Dinehart, R.L. & Burau, J.R. (2005a) Averaged indicators of secondary flow in repeated acounstic Doppler current profiler crossings of bends. *Water Resources Research*, **41**, W09405.

Dinehart, R.L. & Burau, J.R. (2005b) Repeated surveys by Acoustic Doppler Current Profiler for flow and sediment dynamics in a tidal river. *Journal of Hydrology*, **314**: 1-21.

Dollar, E.S.J., James, C.S., Rogers, K.H. & Thoms, M.C. (2007) A framework for interdisciplinary understanding of rivers as ecosystems. *Geomorphology*, **89**: 147-162.

Downes, B., Lake, P., Schreiber, E. & Glaister, A. (1997) Habitat structure and regulation of local species diversity in a stony, upland stream. *Ecological Monographs*, **68** (2): 237-257.

Downs, P.W. & Gregory, K.J. (2004) *River channel management: towards sustainable catchment hydrosystems*. London, Arnold Publishers.

Dudgeon, D., Arthington, A.H., Gessner, M.O., Kawabata, Z.I., Knowler, D.J., Lévêque, C., Naiman, R.J., Prieur-Richard, A.H., Soto, D. & Stiassny, M.L.J. (2005) Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological Reviews*, **81** (2): 163-182.

Dunning, J.B., Damielson, B.J. & Pulliam, H.R. (1992). Ecological processes that affect populations in complex landscapes. *Oikos*, **65** (1): 169-175.

Dyer, F.J. & Thoms, M.C. (2006) Managing river flows for hydraulic diversity: an example of an upland regulated gravel-bed river. *River Research and Applications*, **22** (2): 257-267.

Eisner, A., Young, C., Schneider, M. & Kopecki, I. (2005) MesoCASIMIR – new mapping method and comparison with other current approaches. In: Harby A., Baptist M., Dunbar M.J. & Schmutz S. (eds.) *Proceedings of the 7th meeting of the EU COST Action 626 in Aquatic Habitat Modelling*, Silkeborg, Denmark, May 2005. pp. 47-56.

Elliott, C.R.N. (2005) Habitat assessment in flood risk management and the Environment Agency. In: Harby A., Baptist M., Dunbar M.J. & Schmutz S. (eds.) *Proceedings of the 7th meeting of the EU COST Action 626 in Aquatic Habitat Modelling*, Silkeborg, Denmark, May 2005. pp. 47-56.

Elliott, J.M. (2000) Pools as refugia for brown trout during two summer droughts: trout responses to thermal and oxygen stress. *Journal of Fish Biology*, **56** (4): 938-948.

Emery, J.C., Gurnell, A.M., Clifford, N.J., Petts, G.E., Morrissey, I.P. & Soar, P.J. (2003) Classifying the hydraulic performance of riffle-pool bedforms for habitat assessment and river rehabilitation design. *River Research and Applications*, **19**: 533-549.

Environment Agency (2005) *The Teme Catchment Abstraction Management Strategy*. Bristol, Environment Agency.

Environment Agency (2006a) *The Warwickshire Avon Catchment Abstraction Management Strategy*. Solihull, Environment Agency.

Environment Agency (2006b) *The Worcestershire Middle Severn Catchment Abstraction Management Strategy.* Shrewsbury, Environment Agency.

Environment Agency (2009) *River Arrow discharge data from Studley Gauging Station*. Provided by Midlands Regional Office.

Estivill-Castro, V. & Lee, I. (2000) *AMOEBA: hierarchical clustering based on spatial proximity using Delaunay diagram* [online]. Available from: http://citeseer.ist.psu.edu/cache/papers/cs/13648/ftp:zSzzSzftp.cs.newcastle.edu.auzSzpubzSztechreportszSztr99-05.pdf/estivill-castro00amoeba.pdf [Accessed 2nd April 2008].

Estivill-Castro, V. & Yang, J. (2004) Fast and robust general purpose clustering algorithms. *Data Mining and Knowledge Discovery*, **8** (2): 127-150.

European Commission (2000). Directive 2000/60/EC of the European Parliament and of the Council of 23rd October, 2000: Establishing a framework for Community action in the field of water policy. *Official Journal of the European Communities*, **43**: L327.

Everitt, B. (1974) *Cluster analysis*. London, Heinemann Educational Books Ltd.

Everitt, B.S., Landau, S. & Leese, M. (2001) *Cluster analysis*. 4th edition. London, Arnold.

Fagan, W.F., Cantrell, R.S. & Cosner, C. (1999). How habitat edges change species interactions. *The American Naturalist*, **2** (153): 165-182.

Fagan, W.F., Fortin, M-J. & Soykan, C. (2003) Integrating edge detection and dynamic modelling in quantitative analyses of ecological boundaries. *BioScience*, **53** (8): 730-738.

Farina, A. (2006) *Methods in Landscape Ecology*. Dordrecht, Springer.

Fausch, K.D. Torgersen, C.E., Baxter, C.V. & Li, H.W. (2002) Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. *BioScience*, **52** (6): 483-498.

FISWRG (Federal Interagency Stream Restoration Working Group) (1998) *Stream Corridor Restoration: Principles, Processes, and Practices* [online]. GPO Item No. 0120-A; SuDocs No. A 57.6/2:EN3/PT.653. Available at: http://www.nrcs.usda .gov/technical/stream_restoration/PDF FILES/ ALL-SCRH-08-01.pdf [Accessed 1st May 2008].

Fisher, R.A. (1936) The use of multiple measurements in taxonomic problems. *Annals of Eugenics*, **7**: 179–188.

Flather, C.H. & Bevers, M. (2002). Patchy reaction-diffusion and population abundance: the relative importance of habitat amount and arrangement. *The American Naturalist*, **159** (1): 40-56.

Fonstad, M.A. & Marcus, W.A. (2005) Remote sensing of stream depths with hydraulically assisted bathymetry (HAB) models. *Geomorphology*, **72**: 320-339.

Fonstad, M.A. (2008) Email to Caroline Wallis, 6th July.

Forman, R.T.T. (1995) *Land Mosaics: the ecology of landscapes and regions*. Cambridge, Cambridge University Press.

Fortin, M-J., Olson, R.J., Ferson, S., Iverson, L., Hunsaker, C., Edwards, G., Levine, D., Butera, K. & Klemas, V. (2000) Issues related to the detection of boundaries. *Landscape Ecology*, **15**: 453-466.

Fraley, C. & Raftery, A.E. (1998) How many clusters? Which clustering methods? Answers via model-based cluster analysis. *Computer Journal*, **41**, 578-588.

Fraley, C. & Raftery, A.E. (2002) *MCLUST: Software for Model-Based Clustering, Density Estimation and Discriminant Analysis*. Technical Report No. 415, Department of Statistics, University of Washington.

Freeman, M.C. & Grossman, G.D. (1993) Effects of habitat availability on dispersion of a stream cyprinid. *Environmental Biology of Fishes*, **37** (2): 121-130.

Frissell, C.A., Liss, W.J., Warren, C.E. & Hurley, M.D. (1986) A hierarchical framework for stream habitat classification: viewing streams in a watershed context. *Environmental Management*, **10** (2): 199-214.

Frothingham, K.M. & Rhoads, B.L. (2002) A multiscale conceptual framework for integrated ecogeomorphological research to support stream naturalization in the agricultural Midwest. *Environmental Management*, **29** (1):16-33.

Gaeumann, D. & Jacobson, R.B. (2005) *Aquatic habitat mapping with an acoustic Doppler current profiler: considerations for data quality*. USGS Report 2005-1163.

Gan, G., Ma, C. & Wu, J. (2007) *Data Clustering: theory, algorithms and application*. Philadelphia, SIAM.

Gath, I. & Geva, A.B. (1989) Unsupervised optimal fuzzy clustering. *IEEE Transactions on Pattern Recognition and Machine Intelligence*, **11** (7): 773-781.

Gerhard, M. & Reich, M. (2000) Restoration of streams with large wood: effects of accumulated and built-in wood on channel morphology, habitat diversity and aquatic fauna. *International Review of Hydrobiology*, **85**, 123–137.

Gerten, D., Rost, S., von Bloh, W. & Lucht, W. (2008) Causes of change in 20^{th} century global river discharge. *Geophysical Research Letters*, **35** (20): DOI: 10.1029/2008GL035258.

Ghanem, A., Steffler, P. & Hicks, F. (1996) Two-dimensional hydraulic simulation of physical habitat conditions in flowing streams. *Regulated Rivers: Research and Management*, **12**: 185-200.

Gibbins, C.N, Soulsby, C., Campbell L., Scott, E., McEwam I. & Malcolm, I. (2004) *Influence of channel hydraulics and sediment mobility on stream invertebrate drift*. Proceedings of the British Hydrological Society International Conference held at Imperial College London, 12-

16 July 2004. Available from: http://www.hydrology.org.uk/Publications/imperial/4_11.pdf [Accessed 10th November 2004].

Gibson, R.J. (2002) The effects of fluvial processes and habitat heterogeneity on distribution, growth and densities of juvenile Atlantic salmon (Salmo salar), with consequences on abundance of the adult fish. *Ecology of Freshwater Fish*, **11** (4): 207-222.

Giller, P.S. (2005) River restoration: seeking ecological standards. *Journal of Applied Ecology*, **42** (2): 201-207.

Gillette, D.P., Tiemann, J.S.E., Edds, D.R. & Wildhaber, M.L. (2006) Habitat use by a Mindwestern U.S.A. fish assemblage: effects of season water temperature and river discharge. *Journal of Fish Biology*, **68**: 1494-1512.

Gilvear, D. J., Spray, C. J., & Casas-Mulet, R. (2013). River rehabilitation for the delivery of multiple ecosystem services at the river network scale. *Journal of environmental management*, **126**, 30-43.

Gilvear, D.J., Sutherland, P. & Higgins, T. (2008) An assessment of the use of remote sensing to map habitat features important to sustaining lamprey populations. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **18** (6): 807-818.

Gleick, P.H. (1998) *The world's water 1998-1999: the biennial report on freshwater resource.* Washington DC, Island Press.

Gordon, A.D. (1981) *Classification*. London, Chapman & Hall.

Gordon, N.D., McMahon, T.A., Finlayson, B.L., Gippel, C.J. & Nathan, R.J. (2004) *Stream Hydrology: an introduction for ecologists*. 2nd edition. Chichester, Wiley.

Gordon, R.L. (1996) *Acoustic Doppler Current Profiler Principles of Operation: a practical primer. San Diego: RD Instruments.* Available at: http://www.ldeo.columbia.edu/~jkarsten/teach/docs/rdi_primer.pdf [Accessed 21 September 2010].

Gore, J.A. & Nestler, J.M. (1988) Instream flow studies in perspectives. *Regulated Rivers: Research and Management*, **2** (2):93-101.

Gorman, O.T. & Karr, J.R. (1978) Habitat structure and stream fish communities. *Ecology*, **59** (3): 507-515.

Gostner, W., Parasiewicz, P. & Schleiss, A.J. (2013) A case study on spatial and temporal hydraulic variability in an alpine gravel-bed stream based on the hydromorphological index of diversity. *Ecohydrology*, **6**(4): 652-667.

Gosz, J.R. & Sharpe, P.J.H. (1989) Broad-scale concepts for interactions of climate, topography, and biota at biome transitions. *Landscape Ecology*, **3**: 229-243.

Greenberg, L.A. & Giller, P.S. (2001) Individual variation in habitat use and growth of male and female brown trout. *Ecography*, **24**: 212-224.

Griffith, D.A. & Amerheim, C.G. (1997) *Multivariate Statistical Analysis for Geographers*. New York, Prentice Hall.

Grime, J. (1973) Control of species density in herbaceous vegetation. *Journal of Environmental Management*, **1**: 151-167.

Guensch, G.R., Hardy, T.B. & Addley, R.C. (2001) Examining feeding strategies and position choice of drift-feeding salmonids using an individual-based, mechanistic foraging model. *Canadian Journal of Fisheries and Aquatic Science,* **58** (3): 446-457.

Guisan, A. & Zimmermann, N.E. (2000) Predictive habitat distribution models in ecology. *Ecological Modelling*, **135**: 147-186.

Gunawan, B., Sterling, M. & Knight, D.W. (2010) Using an acoustic Doppler current profiler in a small stream. *Water and Environment Journal,* **24** (2): 147-158.

Gunawan, B., Sun, X., Sterling, M., Knight, D.W., Shiono, K., Chandler, J., Rameshwaran, P., Wright, N.G., Sellin, R.H.J., Tang, X & Fujita, I. (2008) *An integrated and novel approach for estimating the conveyance capacity of the River Blackwater*. Proceedings of the Eighth International Conference on Hydro-Science and Engineering, Nagoya, Japan.

Gurnell, A.M., van Oosterhout, M.P., de Vlieger, B. & Goodson, J.M. (2006) Reach-scale interactions between aquatic plants and physical habitat: River Frome, Dorset. *River Research and Applications*, **22**(6): 667-680.

Gustafson, E. & Kessel, W. (1979) *Fuzzy clustering with a fuzzy covariance matrix*. Proceedings of the IEEE Conference on Decision and Control, San Diego, California, pp. 761-766.

Gustafson, E.J. (1998) Quantifying landscape spatial pattern: what is the state of the art? *Ecosystems*, **1**: 143-156.

Gutzwiller, K.J. (2002) *Applying landscape ecology in biological conservation*. New York, Springer.

Hadfield, C. (1985) *The canals of The West Midlands*. 3rd edition. Devon, David and Charles.

Halleraker, J.H., Sundt, H., Alfredsen, K.T., Dangelmaier, D.G., Kitzler, C. & Schei, T. (2005) Application of multi scale habitat modelling techniques and ecohydrological analysis for optimized management of a regulated national salmon water course in Norway. In: Harby A., Baptist M., Dunbar M.J. & Schmutz S. (eds.) *Proceedings of the 7th meeting of the EU COST Action 626 in Aquatic Habitat Modelling*, Silkeborg, Denmark, May 2005. pp. 47-56.

Hannah, D.M., Wood, P.J. & Sadler, J.P. (2004) Ec\ohydrology and hydroecology: A ‘new paradigm.? *Hydrological Processes*, **18** (17): 3439-3445.

Harby, A., Olivier, J-M., Merigoux, S. & Malet, E. (2007) A mesohabitat method used to assess minimum flow changes and impacts on the invertebrate and fish fauna in the Rhône River, France. *River Research and Applications*, **23** (5): 525-543.

Hardy, R.J., Bates, P.D. & Anderson, M.G. (2000) Modelling suspended sediment deposition on a fluvial floodplain using a two-dimensional dynamic finite element model. *Journal of Hydrology*, **229**: 202-218.

Hargis, C.D., Bissonette, J.A. & David, J.L. (1998) The behaviour of landscape metrics commonly used in the study of habitat fragmentation. *Landscape Ecology*, **13** (3): 167-186.

Harper, D.M. & Everard, M. (1998) Why should the habitat-level approach underpin holistic river survey and management? *Aquatic Conservation: Marine and Freshwater Ecosystems*, **8** (4): 395-413.

Harrison, I. J. & Stiassny, M. J. (1999) The quiet crisis: a preliminary listing of the freshwater fishes of the world that are extinct or 'missing in action'. In: MacPhee, R.D.E. & Sues, H-D. (eds.) *Extinctions in Near Time – causes, contexts and consequences*. New York, Kluwer Academic/Plenum Publishers, pp. 271-331.

Hart, D.D. & Resh, V.H. (1980) Movement patterns and foraging ecology of a stream caddisfly larva, *Canadian Journal of Zoology*, **58**(6): 1174-1185.

Hart, D.D. & Finelli, C.M. (1999) Physical-biological coupling in streams: the pervasive effects of flow on benthic organisms. *Annual Review of Ecological Systems*, **30**: 363-395.

Hartigan, J.A. & Hartigan, P.M. (1985) The dip test of unimodality. *Annals of Statistics*, **13** (1): 70-84.

Hartigan, J.A. (1977) Distribution problems in clustering. In: Ryzim, J.V. (ed.) *Classification and Clustering*. New York, Academic.

Harvey, G.L. & Clifford, N.J. (2009) Microscale hydrodynamics and coherent flow structures in rivers: implications for the characterisation of physical habitat. *River Research and Applications*, **25** (2): 160-180.

Harvey, G.L., Clifford, N.J. & Gurnell, A.M. (2008) Towards an ecologically meaningful classification of the flow biotope for river inventory, rehabilitation, design and appraisal purposes. *Journal of Environmental Management*, **88** (4): 638-650.

Haskell, J.P., Ritchie, M.E. & Olff, H. (2002) Fractal geometry predicts varying body size scaling relationships for mammal and bird home ranges. *Nature*, **418**: 527-530.

Hathaway, R.J., Bezdek, J.C., Huband, J.M. (2006) Scalable visual assessment of cluster tendency for large data sets. *Pattern Recognition*, **39**: 1315-1324.

Hauer, C., Mandlburger, G. & Habersack, H. (2009) Hydraulically related hydro-morphological units: description based on a new conceptual mesohabitat evaluation model (MEM) using LIDAR data as geometric input. *River research and Applications*, **25**: 29-47.

Hauer, F.R., Lorang, M.S., Matson, P.L. & Whited, D.C. (2006) *Hyperspectral imagery acquisition and analysis of the Elwha River corridor*. FLBS Report Number 191-06. Flathead Lake Biological Station, The University of Montana, Polson, MT.

Hawkins, C.P., Hogue, J.N., Decker, L.M. & Feminella, J.W. (1997) Channel morphology, water temperature, and assemblage structure of stream insects. *Journal of the North American Benthological Society*, **16** (4): 728-749.

Hawkins, C.P., Kershner, J.L., Bisson, P.A., Bryant, M.D., Decker, L.M., Gregory, S.V., McCullough, D.A., Overton, C.K., Reeves, G.H., Steedman, R.J. & Young, M.K. (1993) A hierarchical approach to classifying stream habitat features. *Fisheries*, **18** (6): 3-12.

Hawkins, D.M., Muller, M.W. & ten Krooden, J.A. (1982) Cluster analysis. In: Hawkins, D.M. (ed.) (1982) *Topics in Applied Multivariate Analysis*. Cambridge, Cambridge University Press.

Hayes, J.W., Stark, J.D. & Shearer, K.A. (2000) Development and test of a whole-lifetime foraging and bioenergetics growth model for drift-feeding brown trout. *Transactions of the American Fisheries Society*, **129** (2): 315-332.

Hein, T., Reckendorfer, W., Thorp, J.H. & Schiemer, F. (2005) The role of slackwater areas for biogeochemical processes in rehabilitated river corridors: examples from the Danube. *Archiv für Hydrobiologie*, **15** (1-4): 425-442.

Heino, J., Louhi, P. & Muotka, T. (2004) Identifying the scales of variability in stream macroinvertebrate abundance, functional composition and assemblage structure. *Freshwater Biology*, **49** (9): 1230-1239.

Hey, R.D. (1979) River regulation and channel stability. *Proceedings of the 18th Congress of the International Association of Hydraulic Research*, **2**: 35 l-358.

Hilderbrand, R.H., Lemly, A.D. & Dolloff, C.A. (1999) Habitat sequencing and the importance of discharge in inferences. *North American Journal of Fisheries Management*, **19** (1): 198-202.

Hildrew, A.G. & Giller, P.S. (1994) Patchiness, species interactions and disturbance in the stream benthos. In: Hildrew, A.G., Giller, P.S. & Rafaelli, D.G. (eds.) (1994) *Aquatic ecology: scale, pattern and process*. London, Blackwell Science, pp. 21-62.

Ho, D.K.H., Donohoo, S.M., Boyes, K.M. & Lock, C.C. (2003) *Numerical analysis and the real world: it looks pretty but is it right?* [online]. Available from:

http://www.flow3d.com/pdfs/tp/wat_env_tp/FloSci-Bib14-03.pdf [Accessed: 16th February 2008].

Hoagstrom, C.W., Brooks, J.E. & Davenport, S.R. (2008) Recent habitat association and the historical decline of Notropis simuspecosensis. *River research and Applications*, **24** (6): 789-803.

Hoffman, A.L., Olden, J.D., Monroe, J.B., Poff, N.L.R., Wellnitz, T. & Wiens, J.A. (2006) Current velocity and habitat patchiness shape stream herbivore movement. *Oikos*, **115** (2): 358-368.

Holm, C.F., Armstrong, J.D. & Gilvear, D.J. (2001) Investigating a major assumption of predictive instream habitat models: is water velocity preference of juvenile Atlantic salmon independent of discharge? *Journal of Fish Biology*, **59** (6): 1653-1666.

Hopkins, B. (1954) A new method of determining the type of distribution of plant individuals. *Annals of Botany*, **18**: 213-226.

Höppner, F., Klawonn, F. Kruse, R. & Runkler, T. (1999) *Fuzzy Cluster Analysis*. Chichester, Wiley.

Hulea, O., Ebert, S.0. & Strobel, D. (2009) Floodplain restoration along the Lower Danube: a climate change adaptation case study. *Climate Change: Global Risks, Challenges and Decisions. IOP Conference Series: Earth and Environmental Science*, **6**: 402002.

Huston, M. (1979) A general hypothesis of species diversity. *American Naturalist*, **113**: 81-101.

Huston, M.A. (1994) *Biological Diversity: The Coexistence of Species on Changing Landscapes*. Cambridge, Cambridge University Press.

Hynes H.B.N. (1975) The stream and its valley. *Verhandlungen des Internationalen Verein Limnologie*, **19**: 1-15.

Hynes, H.B.N. (1970) The ecology of running waters. Toronto, University of Toronto Press.

Inoue, M. & Nakano, S.H.I. (1999) Habitat structure along channel-unit sequences for juvenile salmon: a subunit-based analysis of in-stream landscapes. *Freshwater Biology*, **42** (4): 597-608.

Isaak, D.J., Thurow, R.F., Rieman, B.E. & Dunham, J.B. (2007). Chinook salmon use of spawning patches: relative roles of habitat quality, size and connectivity. *Ecological Applications*, **17** (2): 352-364.

Jacquez, G.M., Maruca, S. & Fortin, M-J. (2000) From fields to objects: a review of geographic boundary analysis. *Journal of Geographical Systems*, **2** (3): 221-241.

Jahnig, S.C., Lorenz, A.W. & Hering, D. (2009) Restoration effort, habitat mosaics, and macroinvertebrates-does channel form determine community composition? *Aquatic Conservation: Marine and Freshwater Conservation*, **19** (2): 157-169.

Jähnig, S. C., Lorenz, A. W., Hering, D., Antons, C., Sundermann, A., Jedicke, E., & Haase, P. (2011) River restoration success: a question of perception. *Ecological Applications*, **21**(6), 2007-2015.

Jain, A.K. & Dubes, R.C. (1988) *Algorithms for Clustering Data*. Englewood Cliffs, NJ, Prentice-Hall Ltd.

Jensen, J.R. (2005) *Introductory Digital Image Processing: a remote-sensing perspective*. 3rd edition. Upper Saddle River, Prentice-Hall.

Johnston, C.E. (2000) Movement patterns of imperiled blue shiners (Pisces: Cyprinidae) among habitat patches. *Ecology of Freshwater Fish*, **9** (3): 170-176.

Jongman, R.H.G., ter Braak, C.J.F. & Van Tongeren, O.F.R (eds.) (1995) *Data analysis in community and landscape ecology*. Cambridge, Cambridge University Press.

Jordan, D.C. & Fonstad, M.A. (2005) Two dimensional mapping of river bathymetry and power using aerial photography and GIS on the Brazos River, Texas. *Geocarto International*, **20** (3): 13-20.

Jorde, K. (1996) Ecological Evaluation of Instream Flow Regulations based on Temporal and Spatial Variability of Bottom Shear Stress and Hydraulic Habitat Quality. Proceedings of the 2nd International Symposium on Habitat Hydraulics, Ecohydraulics 2000, Quebec City, June 1996, Volume B, pp. 163-174.

Jorde, K. (2002) *River restoration: potential, constraints and the role of ecohydraulics*. Proceedings of the Fourth International Ecohydraulics Conference, 3-8 March 2002, Cape Town, South Africa. [CD-ROM].

Jorde, K., Schneider, M., Peter, A. & Zoellner, F. (2001) *Fuzzy based models for the evaluation of fish habitat quality and instream flow assessment*. In: Proceedings of the International Symposium on Environmental Hydraulics, IAHR, Tempe, Dec. 5-8 2001.

Jowett, I.G. (1993) A method for objectively identifying pool, run, and riffle habitats from physical measurements. *New Zealand Journal of Marine and Freshwater Research*, **27**: 241-248.

Jowett, I.G. (2002) In-stream habitat suitability criteria for feeding inanga (Galaxias maculatus). *New Zealand Journal of Marine and Freshwater Research*, **36** (2): 399-407.

Jowett, I.G. (2003) Hydraulic constraints on habitat suitability for benthic invertebrates in gravel-bed rivers. *River Research and Applications*, **19**: 495-507.

Junk, W.J., Bayley, P.B.; Sparks, R.E. (1989) The flood pulse concept in river-floodplain systems. *Canadian Special Publication of Fisheries and Aquatic Sciences*, **106**: 110-127

Kaufman, L. & Rousseeuw, P.J. (1990) *Finding Groups in Data: An Introduction to Cluster Analysis*. New York, Wiley-Interscience.

Kearns, F.R., Kelly, N.M., Carter, J.L. & Resh, V.H. (2005) A method for the use of landscape metrics in freshwater research and management. *Landscape Ecology*, **20** (1): 113-125.

Keller, E.A. (1971) Areal sorting of bed-load material: the hypothesis of velocity reversal. *Geological Society of America Bulletin*, **82**: 753–756.

Kemp, J.L., Harper, D.M. & Crosa, G.A. (1999) Use of functional habitats to link ecology with morphology and hydrology in river restoration. *Aquatic Conservation: Marine and Freshwater Ecosystems,* **9**: 159-178.

Kemp, J.L., Harper, D.M. & Crosa, G.A. (2000) The habitat-scale ecohydraulics of rivers. *Ecological Engineering*, **16**: 17-29.

Kemp, P.S. & Williams, J.G. (2008) Response of migrating Chinook salmon (Oncorhynchus tshawytscha) smolts to in-stream structure associated with culverts. *River Research and Applications*, **24** (5): 571-579.

Kemp, P.S. & Williams, J.G. (2008) Response of migrating Chinook salmon (Oncorhynchus tshawytscha) smolts to in-stream structure associated with culverts. *River Research and Applications*, **24** (5): 571-579.

Kent, R. & Carmel, Y. (2011) Evaluation of five clustering algorithms for biodiversity surrogates. *Ecological Indicators*, **11**: 896-901.

Kershner, J.L. & Snider, W.M. (1992) Importance of habitat level classification to design instream flow studies. In: Boon, P., Petts, G. & Calow, P. (eds.) *River Conservation and Management*. Sussex, John Wiley & Sons, pp.179-193.

Kim, D-W., Lee, K.H. & Lee, D. (2004) On cluster validity for estimation of the optimal number of fuzzy clusters. *Pattern Recognition*, **37**: 2009-2025.

Kim, M. & Lapointe, M. (2011) Regional variability in Atlantic salmon (Salmo salar) riverscapes: a simple landscape ecology model explaining the large variability in size of salmon runs across Gaspé watersheds, Canada. *Ecology of Freshwater Fish*, **20**: 144-156.

King, J.M., Cambray, J.A. & Impson, D.N. (1998) Linked effects of dam-released floods and water temperature on spawning of the Clanwilliam yellowfish Barbus capensis. *Hydrobiologia*, **384**: 245-265.

Kitchell, J. F., O'Neill, R.V., Webb, D., Gallepp, G.W., Bartell, S.M., Koonce, J.F. & Ausmus, B.S. (1979) Consumer regulation of nutrient cycling. *BioScience*, **29**: 28–34.

Klaar, M.J., Maddock, I. & Milner, A.M. (2009) The development of hydraulic and geomorphic complexity in recently formed streams in Glacier Bay National Park, Alaska. *River Research and Applications,* **25** (10): 1331-1338.

Kocik, J.F. & Ferreri, C.P. (1998) Juvenile production variation in salmonids: population dynamics, habitat, and the role of spatial relationships. *Canadian Journal of Fisheries and Aquatic Science*, **55** (S1): 191-200.

Kondolf, G.M., Larsen, E.W. & Williams, J.G. (2000) Measuring and modeling the hydraulic environment for assessing instream flows. *North American Journal of Fisheries Management*, **20** (4): 1016-1028.

Kondorf, M. (1997) Fuzzy clustering applet [online]. Available from: http://www.cs.rpi.edu/~moorthy/Fuzzy/docs/WriteUp.html [Accessed 5th November 2010].

Kotliar, N.B. & Wiens, J.A. (1990) Multiple scales of patchiness and patch structure: a hierarchical framework for the study of heterogeneity. *Oikos*, **59**: 253-260.

Krueger, A.M. & Frothingham, K.M. (2007) Application and comparison of gemorphological and hydrological pool and riffle quantification methods. *The Geographic Bulletin*, **48**: 85-95.

Kuiper, F.K. & Fisher, L. (1975) A Monte Carlo comparison of six clustering procedures. *Biometrics*, **31**: 777-783.

Kwak, T.J. & Skelly, T.M (1992) Spawning habitat, behaviour and morphology as isolating mechanisms of the golden redhorse, Moxostoma erythurum, and the black redhorse, M. duquesnei, two syntopic fishes. *Environmental Biology of Fishes*, **34**: 127-137.

Lacey, R.W.J. & Roy, A.G. (2008) The spatial characterization of turbulence around large roughness elements in a gravel-bed river. *Geomorphology*, **102** (3-4): 542-553.

Lacey, R.W.J., Legendre, P., & Roy, A.G. (2007) Spatial-scale partitioning of in situ turbulent flow data over a pebble cluster in a gravel-bed river. *Water Resources Research*, **43** (3): W03416.

Lamberti, G.A. & Resh, V.H. (1979) Substrate Relationships, Spatial Distribution Patterns, and Sampling Variability in a Stream Caddisfly Population. *Environmental Entomology*, **8** (3): 561-567.

Lamouroux, N., Capra, H., Pouilly, M. & Souchon, Y. (1999a) Fish habitat preferences in large streams of southern France. *Freshwater Biology*, **42** (4): 673-687.

Lamouroux, N., Olivier, J.M., Persat, H., Pouilly, M.C., Souchon, Y.S., Statzner, B. (1999b) Predicting community characteristics from habitat conditions: fluvial fish hydraulics. *Freshwater Biology*, **42** (4): 275-299.

Lamouroux, N., Poff, N.L.R. & Angermeier, P.L. (2002) Intercontinental convergence of stream fish community traits along geomorphic and hydraulic gradients. Ecology, 83 (7): 1792-1807.

Lamouroux, N., Dolédec, S. & Gayraud, S. (2004) Biological traits of stream macroinvertebrate communities: effects of microhabitat, reach, and basin filters. *Journal of the North American Benthological Society*, **23** (3): 449-466.

Lamouroux, N., Olivier, J.M., Capra, H.É., Zylberblat, M., Chandesris, A. & Roger, P. (2006) Fish community changes after minimum flow increase: testing quantitative predictions in the Rhone River at Pierre-Benite, France. *Freshwater Biology*, **51** (9): 1730-1743.

Lancaster, J., Hildrew, A.G. & Townsend, C.R. (1990) Stream flow and predation effects on the spatial dynamics of benthic invertebrates. *Hydrobiologia*, **203** (3): 177-190.

Lancaster, J. & Downes, B.J. (2010) Linking the hydraulic world of individual organisms to ecological processes: putting ecology into ecohydraulics. *River Research and Applications*, **26** (4): 385-403.

Lancaster, J. & Hildrew, A.G. (1993) Flow refugia and the microdistribution of lotic invertebrates. *Journal of the North American Benthological Society*, **12** (4): 385-393.

Lancaster, J. (2000) Geometric scaling of microhabitat patches and their efficacy as refugia during disturbance. *Journal of Animal Ecology*, **69** (3): 442-457.

Lane, S.N. & Carbonneau, P.E. (2007) High Resolution Remote Sensing for Understanding Instream Habitat. In: Wood, P.J., Hannah, D.M. & Sadler, J.P. (eds.) *Hydroecology and Ecohydrology: Past, Present and Future*. London, Wiley, pp.185-204.

Lane, S.N., Biron, P.M., Bradbrook, K.F., Butler, J.B., Chandler, J.H., Crowell, M.D., McLelland, S.J., Richards, K.S. & Roy, A.G. (1998) Three-dimensional measurement of river channel flow processes using acoustic Doppler velocimetry. *Earth Surface Processes and Landforms*, **23** (13): 1247-1267.

Lane, S.N., Hardy, R.J., Elliott, L. & Ingham, D.B. (2002) High-resolution numerical modeling of three-dimensional flows over complex river topography. *Hydrological Processes*, **16** (11): 2261-2272.

Large, A.R.G. & Newson, M.D. (2006) *Hydromorphological Quality – A Policy Template for Channel Design in River Restoration*. Encyclopedia of Hydrological Sciences, Wiley.

Large, A. & Heritage, G. (2007) *Terrestrial laser scanner based instream habitat quantification using a random field approach* [online]. Available from: http://www.ceg.ncl.ac.uk/rspsoc2007/papers/113.pdf [Accessed 13th February 2008].

Lausch, A. & Herzog, F. (2002) Applicability of landscape metrics for the monitoring of landscape change: issues of scale, resolution and interpretability. *Ecological Indicators*, **2**: 3-15.

Lawler, D.M. (1992) Process dominance in bank erosion systems. In: Carling, P.A. & Petts, G.E. (eds.) (1992) *Lowland floodplain rivers: geomorphological perspectives*. Chichester, Wiley, pp.117-143.

Lawler, D.M. (1994) *Temporal variability in streambank response to individual flow events: the River Arrow, Warkwickshire, UK*. Variability in Stream Erosion and Sediment Transport Proceedings of the Canberra Symposium, December 1994. IAHS Publ. No. 224, pp. 171-180.

Lawless, M. & Robert, A. (2001) Three-dimensional flow structure around small-scale bedforms in a simulated gravel-bed environment. *Earth Surface Processes and Landforms*, **26** (5): 507-522.

Le Coarer, Y. (2007) Hydraulic signatures for ecological modelling at different scales. *Aquatic Ecology*, **42** (3): 451-459.

Le Pichon, C., Gorges, G., Baudry, J., Goreaud, F. & Boet, P. (2009) Spatial metrics and methods for riverscapes: quantifying variability in riverine fish habitat patterns. *Environmetrics*, **20**: 512-526.

Leclerc, M., Boudreau, P., Bechara, J.A., Belzile, L., and Villeneuve, D. (1994) A model of habitat dynamics applied to landlocked salmon (Salmo salar) juveniles of the Ashuapmushuan River (Québec, Canada). *Bulletin Francais de la Peche et de la Pisciculture*, **332**: 11–32.

Leclerc, M., Boudreault, A., Bechara, T.A. & Corfa, G. (1995) Two-dimensional hydrodynamic modeling: a neglected tool in the Instream Flow Incremental Methodology. *Transactions of the American Fisheries Society*, **124** (5): 645-662.

Leclerc, M., Capra, H., Valentin, S., Boudreault, A. & Cote, Y. (1996) Ecohydraulics 2000: Proceedings of the Second IAHR International Conference on Habitat Hydraulics. Quebec: IAHR.

Lee, P-Y & Suen, J-P (2011) *Niche partitioning of fish assemblages in a mountain stream with frequent natural disturbances – an examination of microhabitat in riffle areas*. Ecology of Freshwater Fish, **21** (2): 255–265.

Lefkovitch, L.P. & Fahrig, I.H. (1985) Spatial characteristics of habitat patches and population survival. *Ecological Modelling*, **30**: 297-308.

Leftwich, K.N., Angermeier, P.L. & Dolloff C.A. (1997) Factors influencing behavior and transferability of habitat models for a benthic stream fish. *Transactions of the American Fisheries Society*, **126**: 725-734.

Legendre, P. & Legendre, L. (1998) *Numerical Ecology*. 2nd edition. Amsterdam, Elsevier.

Legleiter, C.J. & Goodchild, M.F. (2005) Alternative representations of in-stream habitat: classification using remote sensing, hydraulic modelling, and fuzzy logic. *International Journal of Geographic Information Science*, **19** (1): 29-50.

Legleiter, C.J., Roverts, D.A., Marcus, W.A. & Fonstad, M.A. (2004) Passive optical remote sensing of river channel morphology and in-stream habitat: physical basis and feasibility. *Remote sensing of Environment*, **93**: 493-510.

Lejot, J., Delacourt, C., Piégay, H., Fournier, T., Trémélo, M.L. & Allemand, P. (2007) Very high spatial resolution imagery for channel bathymetry and topography from an unmanned mapping controlled platform. *Earth Surface Processes and Landforms*, **32** (11): 1705-1725.

Leopold, A. (1933) *Game management*. New York, Scribners.

Leopold, L. B. & Maddock. T. (1953) *The hydraulic geometry of stream channels and some physiographic implications*. Professional Paper 252. Washington, DC, US Geological Survey.

Lepori, F., Palm, D., Brännäs, E. & Malmqvist, B. (2005) Does restoration of structural heterogeneity in streams enhance fish and macroinvertebrate diversity? *Ecological Applications*, **15**: 2060-2071.

Levin, S.A. (1992) The problem of pattern and scale in ecology. *Ecology*, **73** (6): 1943-1967.

Li, H. & Reynolds, J.F. (1995) On definition and quantification of heterogeneity. *Oikos*, **73**: 280-284.

Li, H. & Wu, J. (2004) The use and misuse of landscape indices. *Landscape Ecology*, **19**: 389-399.

Li, X., He, H.S., Bu, R., Wen, Q., Chang, Y., Hu, Y. & Li, Y. (2005) The adequacy of different landscape metrics for various landscape patterns. *Pattern Recognition*, **38** (12): 2626-2638.

Lianyong, W. & Eagles, P.F.J. (2009) Some theoretical considerations: from landscape ecology to waterscape ecology. *Acta Ecologica Sinica*, **29** (3): 176-181.

Lobón-Cerviá, J., González, G. & Budy, P. (2011) Factors driving spatial and temporal variation in production and production/biomass ratio of stream-resident brown trout (Salmo trutta) in Cantabrian streams. *Freshwater Biology*, **56** (11): 2272–2287.

Logan, P. & Furse, M. (2002) Preparing for the European Water Framework Directive-making the links between habitat and aquatic biota. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **12** (4): 425-437.

Lu, H. & Carlin, B.P. (2005) Bayesian areal wombling for geographical boundary analysis. *Geographical Analysis*, **37**: 265-285.

Lucas, M.C., Bubb, D.H., Jang, M-H., Ha, K. & Masters, J.E.G. (2009) Availability of and access to critical habitats in regulated rivers: effects of low-head barriers on threatened lampreys. *Freshwater Biology*, **54** (3): 621-634.

MacQueen, J. (1967) Some methods for classification and analysis of multivariate observations. In: LeCam, L. & Neyman, J. (eds.) (1967) *Proceedings of the Fifth Berkeley Symposium on Mathematical Statistics and Probability, Vol. 1*. Berkeley: University of California Press. pp.281-297.

MacWilliams, M.L., Wheaton, J.M., Pasternack, G.B., Street, R.L. & Kitanidis, P.K. (2006) Flow convergence routing hypothesis for pool-riffle maintenance in alluvial rivers. *Water Resources Research*, **42** (10): W10427.

Maddock, I. (1999) The importance of physical habitat assessment for evaluating river health. *Freshwater Biology*, **41**: 373-391.

Maddock, I., Hill, G. & Smolar-Žvanut, N. (2005) The effect of flow regulation on channel geomorphic unit (CGU) composition in the Soca River, Slovenia. In: Harby A., Baptist M., Dunbar M.J. & Schmutz S. (eds.) *Proceedings of the 7th meeting of the EU COST Action 626 in Aquatic Habitat Modelling*, Silkeborg, Denmark, May 2005. pp. 205-214.

Maddock, I., Thoms, M., Jonson, K., Dyer, F. & Lintermans, M. (2004) Identifying the influence of channel morphology on physical habitat availability for native fish: application to the two-spined blackfish (Gadopsis bispinosus) in the Cotter River, Australia. *Marine and Freshwater Research*, **55** (2): 173-184.

Maddock, I.M. & Hill, G. (2007) *The rivers of Worcestershire: a baseline geomorphological survey*. Report undertaken for Herefordshire and Worcestershire Earth Heritage Trust and Worcestershire County Council.

Malanson G.P. (1993) *Riparian Landscapes*. Cambridge, Cambridge University Press.

Malcolm, I.A., Youngson, A.F. & Gibbins, C.N. (2008) *The potential use of Acoustic Doppler Current Profilers in hydraulic habitat studies: feasibility and limitations report*. Fisheries Research Services Collaborative Report No. 04/08. Available from: http://www.frs-scotland.gov.uk/FRS.Web/Uploads/Documents /Collab%2004-08.pdf [Accessed 1st July 2011].

Malmqvist, B. (2002) Aquatic invertebrates in riverine landscapes. *Freshwater Biology*, **47** (4): 679-694.

Marchildon, M.A, Annable, W.K., Imhof, J.G. & Power, M. (2011). A high-resolution hydrodynamic investigation of brown trout (Salmo trutta) and rainbow trout (Oncorhynchus mykiss) redds. *River Research and Applications*, **27** (3): 345-359.

Marcus, W.A. (2012). Remote sensing of the hydraulic environment in gravel-bed rivers. In: Church, M., Biron, P.M. & Roy, A.G. (eds) *Gravel-Bed Rivers: Processes, Tools, Environments*. Chichester: John Wiley & Sons. pp. 259–285.

Marcus, W.A. & Fonstad, M.A. (2008) Optical remote mapping of rivers at sub-metre resolutions and watershed extents. *Earth Surface Processes and Landforms*, **33** (1): 4-24.

Marcus, W.A. & Fonstad, M.A. (2010) Remote sensing of rivers: the emergence of a subdiscipline in the river sciences. *Earth Surface Processes and Landforms*, **35** (15): 1867-1872.

Marcus, W.A. (2002) Mapping of stream microhabitats with high spatial resolution hyperspectral imagery. *Journal of Geographical Systems*, **4** (1), 113-126.

Martelo, J., Grossman, G.D., Porto, M. & Filomena Magalhaes, M. (2014) Habitat patchiness affects distribution and microhabitat use of endangered Mira chub *Squalius torgalensis*(*Actinopterygii*, *Cypriniformes*). *Hydrobiologia*, **732** (1): 93-109.

Mathur, D., Bason, W.H., Purdy, E.J. & Silver, C.A. (1985) A critique of the Instream Flow Incremental Methodology. *Canadian Journal of Fisheries and Aquatic Science*s, **42**: 825-831.

Mathur, P.M. (2004) *Computer-processing of remotely-sensed images: an introduction*. 3rd edition. Chichester, John Wiley & Sons Ltd.

McGarigal, K. & Marks, B.J. (1995) *FRAGSTATS: spatial pattern analysis program for quantifying landscape structure*. General Technical Report PNW-351. Denver, US Forest Service.

McNames, J. (2000) *Motorcycle dataset* [online]. Available at: http://web.cecs.pdx.edu/~mcnames/DataSets/index.html [Accessed 1st July 2011].

Merckelbach, L.M. (2006) A model for a high-frequency Acoustic Doppler Current Profiler backscatter from suspended sediment in strong currents. *Continental Shelf Research*, **26**: 1316-1335.

Mérigoux, S., Lamouroux, N., Olivier, J.M. & Dolédec, S. (2009) Invertebrate hydraulic preferences and predicted impacts of changes in discharge in a large river. Freshwater *Biology*, **54** (6): 1343-1356.

Mertes, L.A.K. (2002) Remote sensing of riverine landscapes. *Freshwater Biology*, **47**: 799-816.

Milan, D.J., Heritage, G.L., Large, A.R.G., Fuller, I.C. (2011) Filtering spatial error from DEMs: implications for morphological change estimation. *Geomorphology* **125** (1): 160–171.

Milan, D. J. & Heritage, G. L. (2012) LiDAR and ADCP Use in Gravel-Bed Rivers: Advances Since GBR6. In: Church, M., Biron, P.M. & Roy, A.G. (eds) *Gravel-Bed Rivers: Processes, Tools, Environments*. Chichester: John Wiley & Sons.

Milhous RT, Wegner DL & Waddle T. (1984) *Users Guide to the Physical Habitat Simulation System (PHABSIM).* Instream Flow Inf. Paper. No. 11.

Milhous, R.T. (1979) *The PHABSIM system for instream flow studies*. Proceedings of the Summer Computer Simulation Conference, La Jolla, CA. The Society for Computer Simulation, Toronto, Ontario, pp. 440-446.

Milhous, R.T. (1999) History, theory, use and limitations of the physical habitat simulations system. Proceedings of the Third International Symposium on Ecohydraulics, Salt Lake City, Utah.

Millennium Ecosystem Assessment (2005) *Ecosystems and human well-being: wetlands and water synthesis*. Washington DC, World Resources Institute.

Milligan, G.W. & Cooper, M.C. (1985) An examination of procedures for determining the number of clusters in a dataset. *Psychometrika*, **50**: 159-179.

Milligan, G.W. (1996) Clustering validation: results and implications for applied analyses. In: Arabie, P., Hubert, L.J. & De Soete, G. (eds.) *Clustering and Classification*. Singapore, World Scientific, pp.341-375.

Minshall, G.W. & Robinson, C.T. (1998) Macroinvertebrate community structure in relation to measures of habitat heterogeneity. *Archiv für Hydrobiologie*, **141** (2): 129-151.

Mitsch, W. J., Lefeuvre, J.-C. & Bouchard, V. (2002) Ecological engineering applied to river and wetland restoration. *Ecological Engineering*, **18** (5):529-541.

Moir, H.J. & Pasternack, G.B. (2008) Relationships between mesoscale morphological units, stream hydraulics and Chinook salmon (Oncorhynchus tshawytscha) spawning habitat on the Lower Yuba River, California. *Geomorphology*, **100** (3-4): 527-548.

Moir, H.J. & Pasternack, G.B. (2010) Substrate requirements of spawning Chinook salmon (Oncorhynchus tshawytscha) are dependent on local channel hydraulics. *River Research and Applications*, **26** (4): 456-468.

Moir, H.J., Gibbins, C.N., Soulsby, C. & Webb, J.H. (2006) Discharge and hydraulic interactions in contrasting channel morphologies and their influence on site utilization by spawning Atlantic salmon (Salmo salar). *Canadian Journal of Fish and Aquatic Science*, **63** (11): 2567-2585.

Moir, H.J., Gibbins, C.N., Soulsby, C. & Youngson, A.F. (2005) PHABSIM modeling of Atlantic salmon spawning habitat in an upland stream: testing the significance of habitat suitability indices on model output. *River Research and Applications*, **21**: 1021-1034.

Montgomery, D.R. & Buffington, J.M. (1997) Channel reach morphology in mountain drainiage basins. *Geological Society of America Bulletin*, **109**: 596-611.

Moss, T. (2007) Institutional drivers and constraints of floodplain restoration in Europe. *International Journal of River Basin Management*, **5** (2): 121-130.

Mouton, A., Meixner, H., Goethals, P.L.M., De Pauw, N. & Mader, H. (2007) Concept and application of the usable volume for modeling the physical habitat of riverine organisms. *River Research and Applications*, **23**: 545-558.

Moyle, P.B. & Leidy, R.A. (1992) Loss of biodiversity in aquatic ecosystems: evidence from fish faunas. In: Fiedler, P.L. & Jain, S.K. (eds.) *Conservation Biology: The Theory and Practice of Nature Conservation, Preservation and Management.* New York, Chapman and Hall, pp 127-169.

Mueller, D.S. (2003) *Field evaluation of boat-mounted Acoustic Doppler instruments used to measure streamflow*. Proceedings of the IEEE/OES Seventh Working Conference on Current Measurement Technology, San Diego, CA, USA.

Muste, M., Yu, K. & Spasojevic, M. (2004a) Practical aspects of ADCP data use for quantification of mean river flow characteristics; Part I: moving-vessel measurements. *Flow Measurement and Instrumentation*, **15** (1): 1-16.

Muste, M., Yu, K., Pratt, T & Abraham, D. (2004b) Practical aspects of ADCP data use for quantification of mean river flow characteristics; Part II: fixed-vessel measurements. *Flow Measurement and Instrumentation*, **15** (1): 17-28.

Naiman, R.J. & Décamps, H. (1998) The ecology of interfaces: Riparian zones. Annual Review of Ecological Systems, **28**: 621–658.

Negishi, J.N., Inoue, M. & Nunokawa, M. (2002) Effects of channelisation on stream habitat in relation to a spate and flow refugia for macroinvertebrates in Northern Japan. *Freshwater Biology*, **47** (8): 1515 – 1529.

Nestler, J.M., Goodwin, R.A. & Loucks, D.P. (2005) Coupling of engineering and biological models for ecosystem analysis. *Journal of Water Resources Planning and Management*, **131** (2): 101-109.

Nestler, J.M., Goodwin, R.A., Smith, D.L. & Anderson, J.J. (2008) A mathematical and conceptual framework for ecohydraulics, In: Wood, P.J., Hannah, D.M. & Sadler, J.P. (eds.) *Hydroecology and Ecohydrology: Past, Present and Future.* London, Wiley.

Newson, M. (2010) Understanding 'hot - spot' problems in catchments: the need for scale - sensitive measures and mechanisms to secure effective solutions for river management and conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **20**(S1), S62-S72.

Newson, M.D. & Newson, C.L. (2000) Geomorphology, ecology and river channel habitat: mesoscale approaches to basin-scale challenges. *Progress in Physical Geography*, **24** (2): 195-217.

Newson M.D. & Large A.R.G. (2006) 'Natural' rivers, 'hydromorphological quality' and river restoration: a challenging new agenda for applied fluvial geomorphology. *Earth Surface Processes and Landforms*, **31**: 1606–1624.

Newson, M.D., Clark, M.J., Sear, D.A. & Brookes, A. (1998a). The geomorphological basis for classifying rivers. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **8**: 415-430.

Newson, M.D., Harper, D.M., Padmore, C.L., Kemp, J.L. & Vogel, B. (1998b) A cost-effective approach for linking habitats, flow types and species requirements. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **8** (4): 431-446.

Nilsson, C., Reidy, C.A., Dynesius, M. & Revenga, C. (2005) Fragmentation and flow regulation of the world's large river systems. *Science*, **308** (5270): 405-408.

Nilsson, C., R. Jansson, B. Malmqvist, and R. J. Naiman (2007) Restoring riverine landscapes: the challenge of identifying priorities, reference states, and techniques. *Ecology and Society*, **12** (1): 16.

Ning, N.S.P., Nielson, D.L., Paul, W.L., Hillman, T.J. & Suter, P.J. (2009). Macroinvertebrate dynamics in riverine slackwater and mid-channel habitats in relation to physico-chemical parameters and food availability. *River Research and Applications*, **26** (3): 279-296.

NIT (1994) *Proceedings of the First International Symposium on Habitat Hydraulics*. Trondheim, Norwegian Institute of Technology.

Nystrom, E.A., Rehmann, C.R., Asce, M. & Oberg, K.A. (2007) Evaluation of mean velocity and turbulence measurements with ADCPs. *Journal of Hydraulic Engineering*, **133** (12): 1310-1318.

Oberg, K.A. (2002) *In search of easy-to-use methods for calibrating ADCPs for velocity and discharge measurements*. Proceedings of Hydraulic Measurements & Experimental Methods, Estes Park, CO, USA.

Olsen N.R.B. (1996) *A three-dimensional numerical model for simulation of sediment movements in water intakes with multi-block option*. SSIIM Users Manual Version 1-4.

O'Neill, M.P. & Abrahams, A.D. (1984) Objective identification of pools and riffles. *Water Resources Research*, **20**: 921-926.

O'Neill, R.V., DeAngelis, D.L., Waide, J.B. & Allen, T.F.H. (1986) *A hierarchical concept of ecosystems*. Princeton, Princeton University Press.

Orr, H.G., Large, A.R.G., Newson, M.D. & Walsh, C.L. (2008) A predictive typology for characterising hydromorphology. *Geomorphology*, **100** (1-2): 32-40.

Orth, D.J. (1987) Ecological considerations in the development and application of instream flow-habitat models, *Regulated Rivers: Research & Management*, **1** (2): 171-181.

Osborne, J.W. & Costello, A.B. (2004) Sample size and subject to item ratio in principal components analysis. *Practical Assessment, Research & Evaluation*, 9 (11) [online]. Available from: http://PAREonline.net/getvn.asp?v=9&n=11 [Accessed 1st July 2011].

Padmore, C.L. (1997) Biotopes and their hydraulics: a method for defining the physical component of freshwater quality. In: Boon, P.J. & Howell, D.L. (eds.) *Freshwater quality: defining the indefinable?* Edinburgh, HMSO, pp. 251-257.

Padmore, C.L. (1998) The role of physical biotopes in determining the conservation status and flow requirements of British rivers. *Aquatic Ecosystem Health Management*, **1** (1): 25-35.

Pal, N.N. & Bezdek, J.C. (1995) On cluster validity for the fuzzy c-means model. *IEEE Transactions on Fuzzy Systems*, **3**: 370-379.

Pal, N.N. & Bezdek, J.C. (1997) Correction to 'On cluster validity for the fuzzy c-means model'. *IEEE Transactions on Fuzzy Systems*, **5** (1): 152-f153.

Palmer, M.A. & Poff, N.L.R. (1997) The influence of environmental heterogeneity on patterns and processes in streams. *Journal of the North American Benthological Society*, **16** (1): 169-173.

Palmer, M.A., Swan, C.M., Nelson, K., Silver, P. & Alvestad, R. (2000) Streambed landscapes: evidence that stream invertebrates respond to the type and spatial arrangement of patches. *Landscape Ecology*, **15**: 563-576.

Palmer, M. A., Bernhardt, E. S., Allan, J. D., Lake, P. S., Alexander, G., Brooks, S., Carr, J., Clayton, S., Dahm, C.N., Follstad Shah, J., Galat, D.L., Loss, S.G., Goodwin, P., Hart, D.D., Hassett, B., Jenkinson, R., Kondolf, G.M., Lave, R., Meyer, J.L., O'Donnell, T.K., Pagano, L. & Sudduth, E. (2005) Standards for ecologically successful river restoration. *Journal of applied ecology*, **42**(2), 208-217.

Panayirci, E. & Dubes, R.C. (1983) A test for multidimensional clustering tendency. *Pattern Recognition*, **16**: 433-444.

Parasiewicz , P. & Walker, J.D. (2007) Comparison of MesoHABSIM with two microhabitat models (PHABSIM and HARPHA). River Research and Applications, 23 (8): 904-923.

Parasiewicz, P. (2001) MesoHABSIM: a concept for application of instream flow models in river restoration planning. *Fisheries*, **26** (9): 6-13.

Parasiewicz, P. (2003) Upscaling: integrating habitat model into river management. *Canadian Water Resources*, **28** (2): 1-17.

Parsons, D.R., Best, J.L., Orfeo, O., Hardy, R.J., Kostaschuk, R. & Lane, S.N. (2005) Morphology and flow fields of three-dimensional dunes, Rio Paraná, Argentina: results from simultaneous multibeam echo sounding and acoustic Doppler current profiling. *Journal of Geophysical Research*, **110**: F04S03.

Parsons, M. & Thoms, M.C. (2007) Hierarchical patterns of physical–biological associations in river ecosystems. *Geomorphology*, **89** (1-2): 127-146.

Pasternack, G.B., Bounrisavong, M.K. & Parikh, K.K. (2008) Backwater control on riffle-pool hydraulics, fish habitat quality, and sediment transport regime in gravel-bed rivers. Journal of Hydrology, 357 (1-2): 125-139.

Pearce, D. (1998) Auditing the earth: The value of the world's ecosystem services and natural capital. *Environment*, **40** (2): 23-28.

Pearsons, T.N., Li, H.W. & Lamberti, G.A. (1992) Influence of habitat complexity on resistance to flooding and resilience of stream fish assemblages. *Transactions of the American Fisheries Society*, **121** (4): 427-436.

Pearsons, T.N., Li, H.W. & Lamberti, G.A. (1992) Influence of habitat complexity on resistance to flooding and resilience of stream fish assemblages. *Transactions of the American Fisheries Society*, **121** (4): 427-436.

Pedersen, M.L. & Friberg, N. (2007) Two lowland stream riffles – linkages between physical habitats and macroinvertebrates across multiple spatial scales. *Aquatic Ecology*, **41** (3): 475-490.

Peña, J. M., Lozano, J. A. & Larranaga, P., 1999. An empirical comparison of four initialization methods for the k-means algorithm. *Pattern Recognition*, **20** (10): 1027-1040.

Petts, G., Morales, Y. & Sadler, J. (2006) Linking hydrology and biology to assess the water needs of river ecosystems. *Hydrological Processes*, **20**: 2247-2251.

Pickett, S.T.A. & Thompson, J.N. (1978) Patch dynamics and the design of nature reserves. *Biological Conservation*, **13**: 27-37.

Pickett, S.T.A. & White, F.S. (1985) Patch dynamics: a synthesis. In: White, F.S. & Pickett, S.T.A. (eds.) *The ecology of natural disturbance and patch dynamics*. London, Academic Press, pp. 3-13.

Poff, N.L. & Ward. J.V. (1990) Physical habitat template of lotic systems: recovery in the context of historical pattern of spatio-temporal variability. *Environmental Management*, **14** (5): 629-646.

Poff, N.L. & Zimmerman, J.K.H. (2010) Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. *Freshwater Biology*, **55** (1): 194-205.

Poff, N.L., Richter, B.D., Arthington, A.H., Bunn, S.E., Naiman, R.J., Kendy, E., Acreman, M, Apse, C., Bledsoe, B.P. & Freeman, M.C. (2010) The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards. *Freshwater Biology*, **55** (1): 147-170.

Poff, N.L.R. & Allan, J.D. (1995) Functional organization of stream fish assemblages in relation to hydrological variability. *Ecology*, **76**: 606-627.

Poff, N.L.R., Allan, J.D., Bain, M.B., Karr, J.R., Prestegaard, K.L., Richter, B.D., Sparks, R.E. & Stromberg, J.C. (1997) The natural flow regime. *Bioscience*, **47** (11): 769-784.

Poole, G.C (2002) Fluvial landscape ecology: addressing uniqueness within the river discontinuum. *Freshwater Biology*, **47**: 641-660.

Poole, G.C., Frissell, C.A. & Ralph, S.C. (1997) In-stream habitat unit classification: inadequacies for monitoring and some consequences for management. *Journal of the American Water Resources Association*, **33** (4): 879-896.

Post, D.M., Doyle, M.W., Sabo, J.L. & Finlay, J.C. (2007) The problem of boundaries in defining ecosystems: a potential landmine for uniting geomorphology and ecology. *Geomorphology*, **89**: 111-126.

Postel, S.L., Daily, G.C. & Ehrlich, P.R. (1996) Human appropriation of renewable freshwater. *Science*, **271** (5250): 785-788.

Power, M.E., Sun, A., Parker, M. Dietrich, W.E. & Wootton, J.T. (1995) Hydraulic food-chain models: an approach to the study of food-web dynamics in large rivers. *BioScience*, **45**: 159-167.

Principe, R.E., Raffaini, G.B., Gualdoni, C.M., Oberto, A.M. & Corigliano, M.C. (2007) Do hydraulic units define macroinvertebrate assemblages in mountain streams of central Argentina? *Limnologica: Ecology and Management of Inland Waters*, **37** (4): 323-336.

Pringle, C. (2003) What is hydrologic connectivity and why is it ecologically important? *Hydrological Processes*, **17** (3): 2685-2689.

Pringle, C.M., Naiman, R.J., Bretschlo, G., Karr, J.R., Oswood, M.W., Webster, J.R., Welcomme, R.L. & Winterbourn, M.J. (1988) Patch dynamics in lotic systems: the stream as a mosaic. *Journal of the North American Benthological Society*, **7**: 503-524.

Pusey & Arthington 2003 Importance of the riparian zone to the conservation and management of freshwater fish: a review. *Marine and Freshwater Research*, **54** (1): 1 – 16.

Quinn, J.M. & Hickey, C.W. (1994) Hydraulic parameters and benthic invertebrate distributions in two gravel-bed New Zealand rivers. *Freshwater Biology*, **32**: 489-500.

Rabeni, C.F. & Jacobson, R.B. (1993) The importance of fluvial hydraulics to fish-habitat restoration in low-gradient alluvial streams. *Freshwater Biology*, **29**: 211-220.

Rabeni, C.F., Doisy, K.E. & Galat, D.L. (2002) Testing the biological basis of a stream habitat classification using benthic invertebrates. *Ecological Applications*, **12** (3): 782-796.

Raven, P.J., Fox, P., Everard, M., Holmes, N.T.H. & Dawson, F.H. (1997) River Habitat Survey: a new system for classifying rivers according to their habitat quality. In: Boon, P.J. & Howell, D.L. (eds.) *Freshwater quality: defining the indefinable?* Edinburgh, HMSO, pp. 215-234.

Raven, P.J., Holmes, N.T.H., Naura, M. & Dawson, F.H. (2000) Using river habitat survey for environmental assessment and catchment planning in the U.K. *Assessing the Ecological Integrity of Running Waters: Developments in Hydrobiology*, **149**: 359-367.

Raven, P.J., Holmes, N.T.H., Charrier, P., Dawson, F.H., Naura, M. & Boon, P.J. (2002) Towards a harmonized approach for hydromorphological assessment of rivers in Europe: a qualitative comparison of three survey methods. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **12** (4): 405-424.

Raven, P.J., Holmes, N.T.H., Vaughan, I.P., Dawson, F.H. & Scarlett, P. (2010) Benchmarking habitat quality: observations using River Habitat Survey on near-natural streams and rivers in northern and western Europe. Aquatic conservation: marine and freshwater ecosystems, **20** (1): S3-S30.

Reice, S.R. (1994) Nonequilibrium determinants of biological communi structure. *American Science*, **82**: 424-435.

Reinfields, I., Brooks, A.J., Haeusler, T. & Williams, S. (2006) Temporal Patterns and Effects of Surface-Water Diversions on Daily Flows and Aquatic Habitats: Bega-Bemboka River, New South Wales, Australia. *Geographical Research*, **44** (4): 401-417.

Rempel, L.L., Richardson, J.S. & Healey, M.C. (1999) Flow refugia for benthic macroinvertebrates during flooding of a large river. *Journal of the North American Benthological Society*, **18** (1): 34-48.

Remshardt, W.J. & Fisher, W.L. (2009) Effects of variation in streamflow and channel structure on smallmouth bass habitat in an alluvial stream. *River Research and Applications*, **25** (6): 661-674.

Rennie, C.S. & Church, M. (2007) *ADCP shear stress and bedload transport in a large wandering gravel-bed river*. IAHR Venice Congress, 2007.

Rennie, C.S., Millar, R.G. & Church, M. (2002) Measurement of bed load velocity using an Acoustic Doppler Current Profiler. *Journal of Hydraulic Engineering*, **128** (5): 473-483.

Renschler, C.S., Doyle, M.W. & Thoms, M. (2007) Geomorphology and ecosystems: Challenges and keys for success in bridging disciplines. *Geomorphology*, **89** (1-2): 1-8.

Resh, V.H., Brown, A.V. & Covich, A.P., Gurtz, M.E., Li, H.W., Minshall, W., Reice, S.R., Sheldon, A.L., Wallace, J.B. & Wissmar, R.C. (1988) The role of disturbance in stream ecology. *Journal of the North American Benthological Society*, **7**: 433-455.

Reuter, J.M., Jacobson, R.B. & Elliott, C.M. (2003) *Physical stream habitat dynamics in Lower Bear Creek, Northern Arkansas*. Technical Report. Geological Survey, Columbia, MO, Biological Resources Division.

Rhoads, B.L. & Suckhodolov, A.N. (2001) Field investigation of three-dimensional flow structure at stream confluences: 1. Thermal mixing and time-averaged velocities. *Water Resources Research*, **37** (9): 2393-2410.

Rhoads, B.L., Schwartz, J.S. & Porter, S. (2003) Stream geomorphology, bank vegetation, and three-dimensional habitat hydraulics for fish in midwestern agricultural streams. *Water Resources Research*, **39** (8): 1218.

Rice, S., Lancaster, J. & Kemp, P. (2010) Experimentation at the interface of fluvial geomorphology, stream ecology and hydraulic engineering and the development of an effective, interdisciplinary river science. *Earth Surface Processes and Landforms*, **35**: 64-77.

Richter, B. D., Matthews, R., Harrison, D. L. & Wigington, R. (2003). Ecologically sustainable water management: managing river flows for ecological integrity. *Ecological Applications*, **13**: 206–224.

Ricklefs, R.E. & Schluter, D. (1993) *Species diversity in ecological communities*. Chicago, Chicago University Press.

Ricotta, A., Corona, P. & Marchetti, M. (2003) Beware of contagion! *Landscape and Urban Planning*, **62** (3): 173-177.

Ries, L., Fletcher, R.J.Jr., Battin, J. & Sisk, T.D. (2004). Ecological responses to habitat adges: mechanisms, models and variability explained. *Annual Review of Ecology, Evolution and Systematics*, **35**: 491-522.

Riis, T. & Biggs, B.J.F. (2003) Hydrologic and hydraulic control of macrophyte establishment and performance in streams. *Limnology and Oceanography*, **48** (4): 1488-1497.

Riitters, K.H., O'Neill, R.V., Hunsaker, C.T., Wickham, J.D., Yankee, D.H., Timmins, S.P., Jones, K.B. & Jackson, B.L. (1995) A factor analysis of landscape pattern and structure metrics. *Landscape Ecology*, **10** (1): 23-39.

Riley, W.D., Ives, M.J., Pawson, M.G. & Maxwell, D.L. (2006) Seasonal variation in habitat use by salmon, Salmo salar, trout, Salmo trutta and grayling, Thymallus thymallus, in a chalk stream. *Fish Management and Ecology*, **13** (4): 221-236.

Rivas-Casado, M., Ballamy, P., White, S., Maddock, I., Dunbar, M. & Booker, D. (2005) Defining the spatial and temporal hydromorphological sampling strategies for the Leigh Brook river site. In: Harby, A., Baptist, M., Duel, H., Dunbar, M., Goethals, P., Huusko, A., Ibotson, A., Mader, H., Pedersen, M.L., Schmutz, S. & Schneider, M. (eds.) (2005) *Proceedings of the final meeting of the European Aquatic Modelling Network, COST 626*. Silkeborg, Denmark, 19-20th May 2005.

Robinson, C.T., Tockner, K. & Ward, J.V. (2002) The fauna of dynamic riverine landscapes. *Freshwater Biology*, **47**: 661-677.

Rodriguez, J.F., Bombardelli, F.A., Garcia, M.H., Frothingham, K., Rhoads, B.L. & Abad, J.D. (2004) High-resolution numerical simulation of flow through a highly sinuous reach. *Water Resources Management*, **18**: 177-199.

Roland, J. (1993) Large-scale forest fragmentation increases the duration of tent caterpillar outbreak. *Oecologica*, **93**: 25-30.

Roper, B.B. & Scarnecchia, D.L. (1995) Observer variability in classifying habitat types in stream surveys. *North American Journal of Fisheries Management*, **15**: 49-53.

Roper, B.B., Kershner, J.L., Archer, E., Henderson, R. & Bouwes, N. (2002) An evaluation of physical stream habitat attributes used to monitor streams. *Journal of the American Water Resources Association*, **38** (6): 1367-1646.

Rosgen, D.L. (1994) A classification of natural rivers. *Catena*, **22**:169-199.

Rowntree, K.M. & Wadeson, R.A. (1999) *A hierarchical framework for categorising the geomorphology of river systems*. Water Research Commission Report, 497/1/99. Pretoria, SA, WRC.

Rowntree, K.M. (1996) (ed.) *The hydraulics of physical biotopes – terminology, inventory and calibration*. Water Research Commission Report KV84/96.

Ruspini, E.H. (1969) A new approach to clustering. *Information and Control*, **15** (1): 22-32.

Schlosser, I.J. (1987) A conceptual framework for fish communities in small warmwater streams. In: Matthews, W.J. & Heins, D.C. (eds.) *Community and evolutionary ecology of North American stream fishes*. Oklahoma, Oklahoma University Press, pp. 17-24.

Schlosser, I.J. (1991) Stream fish ecology: a landscape perspective. *BioScience*, **41**: 704–12.

Schiemer, F., Flore, L. & Keckeis, H. (2001) 0+ fish as indicators of the ecological status of large rivers. *Archiv für Hydrobiologie*, **135**: 115-116.

Schneider M & Jorde K. (2003) *Fuzzy-rule based models for the evaluation of fish habitat quality and instream flow assessment*. Proceedings of the International IFIM Users Workshop, Fort Colins, CO, p. 22.

Schneider, M., Jorde, K., Zollner, F & Kerle, F. (2001) *Development of a user-friendly software for ecological investigations on river systems, integration of a fuzzy rule-based approach*. Proceedings of the 15th International Environmental Informatics Symposium 2001, Zurich, Switzerland.

Schneider, M., Jorde, K., Zöllner, F., Kerle, F. & Eisner, A. (2002) *Use of Habitat models for decision support in water resources management*. Proceedings of the 3rd International Conference on Water Resources and Environment Research (ICWRER), Dresden.

Schoof, R. (1980) Environmental impact of channel modification. *Water Resources Bulletin*, **16**: 697–701.

Schumm, S.A. (1977) *The Fluvial System*. New York, John Wiley & Sons.

Schwartz, J.S. & Herricks, E.E. (2008) Fish use of ecohydraulic-based mesohabitat units in a low-gradient Illinois stream; implications for stream restoration. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **18**: 852-866.

Schweizer, S., Borsuk, M.E., Jowett, I. & Reichert, P. (2007) Predicting joint frequency distributions of depth and velocity for instream habitat assessment. *River Research and Applications*, **23**: 287-302.

Schwendel, A.C., Fuller, I.C. & Death, R.G. (2010) Assessing DEM interpolation methods for effective representation of upland stream morphology for rapid appraisal of bed stability. *River Research and Applications*, **28** (5): 567–584.

Scruton, D.A., Heggenes, J., Valentin, S., Harby, A. & Bakken, T.H. (1998) Field sampling design and spatial scale in habitat-hydraulic modelling: comparison of three models. *Fisheries Management and Ecology*, **5** (3): 225–240.

Sear, D. (2009) *Characterising hydromorphology* [online]. Available from: http://www.macaulay .ac.uk /hydroworkshop/SEAR2.pdf [Accessed 1st July 2011].

Shields Jr, F.D., Knight, S.S., Testa III, S & Cooper, C.M. (2003) Use of Acoustic Doppler Current Profilers to describe velocity distributions at the reach scale. *Journal of the American Water Resources Association*, **39** (6): 1397-1408.

Shih, H.H., Payton, C., Sprenke, J. & Mero, T. (2000) *Towing basin speed calibration of Acoustic Doppler Current Profiling instruments*. Proceedings of 2000 Joint Conference on Water Resources Engineering and Water Resources Planning and Management, ASCE, Reston, VA.

Shirvell, C.S. (1989) Ability of PHABSIM to predict chinook salmon spawning habitat. *Regulated Rivers: Research and Management*, **3** (1): 277-289.

Shoffner, D. & Royall, D. (2008) Hydraulic habitat composition and diversity in rural and urban stream reaches of the North Carolina Piedmont (USA). *River Research and Applications,* **24** (8): 1082-1103.

Sime, L.C., Ferguson, R.I. & Church, M. (2007) Estimating shear stress from moving boat acoustic Doppler velocity measurements in a large gravel bed river. *Water Resources Research*, **43**: W03418.

SØndergaard, M. & Jeppesen, E. (2007) Anthropogenic impacts on lakes and stream ecosystems, and approaches to restoration. *Journal of Applied Ecology,* **44**: 1089-1094.

Souchon, Y & Capra, H. (2004) Aquatic habitat modelling: biological validations of IFIM/Phabsim methodology and new perspectives. *Hydroécologie Appliquée*, **14** (1): 9-25.

Southwood, T.R.E. (1977) Habitat, the templet for ecological strategies? *The Journal of Animal Ecology*, **46**: 337-365.

Southwood, T.R.E. (1988) Tactics, strategies and templets. Oikos, 52: 3-18.

Stanford, J.A., Lorang, M.S. & Hauer, F.R. (2005) The shifting habitat mosaic of river systems. *Verhandlungen des Internationalen Verein Limnologie,* **29**: 123-136

Statzner, B. & Higler, B. (1986) Stream hydraulics as a major determinant of benthic invertebrate zonation patterns. *Freshwater Biology*, **16** (1): 127-139.

Statzner, B., Gore, J.A. & Resh, V.H. (1988) Hydraulic stream ecology: observed patterns and potential applications. *Journal of the North American Benthological Society,* **7** (4): 307-360.

Stewardson, M.J. & McMahon, T.A. (2002) A stochastic model of hydraulic variations within stream channels. *Water Resources Research,* **38** (1): 8-1-14.

Stone, M.C. & Hotchkiss, R.H. (2007) Evaluating velocity measurement techniques in shallow streams. *Journal of Hydraulic Research,* **45** (6): 752-762.

Sundermann, A., Antons, C., Cron, N., Lorenz., A.W., Hering, D & Haase, P. (2011) Hydromorphological restoration of running waters: effects on benthic invertebrate assemblages. *Freshwater Biology,* **56** (8): 1689-1702.

Tamminga, A., Hugenholtz, C., Eaton, B. & Lapointe, M. (2014) Hypersepctral remote sensing of channel reach morphology and fish habitat using an unmanned aerial vehicle (UAV): a first assessment in the context of river research and management. River Research and Applications. Available online at: http://onlinelibrary.wiley.com/doi/10.1002/rra.2743/abstract [26/5/2014].

Taniguchi, H., Nakano, S. & Tokeshi, M. (2003) Influences of habitat complexity on the diversity and abundance of epiphytic invertebrates on plants. *Freshwater Biology,* **48** (4): 718-728.

Teledyne RDi (2005) *StreamPro Quick Start Guide*. Report No. P/N 95B 6004-00.

Teledyne RDi (2007) *WinRiverII User's Guide*. P/N 957-6231-00.

Templ, M., Filzmoser, P. & Reimann, C. (2008) Cluster analysis applied to regional geochemical data: Problems and possibilities. *Applied Geochemistry*, **23** (8): 2198-2213.

Tews, J., Brose, U., Grimm, V., Tielbörger, K., Wichmann, M.C., Schwager, M. & Jeltsch, F. (2004) Animal species diversity driven by habitat heterogeneity/diversity: the importance of keystone structures. *Journal of Biogeography*, **31** (1): 79-92.

Tharme, R.E. (2003) A global perspective on environmental flow assessment: emerging trends in the development and application of environmental flow methodologies for rivers. *River Research and Applications*, **19** (5-6): 397-441

The Mathworks (2009) *MATLAB v.7.8.0 R2009a*.

Thompson, D.M., Wohl, E.E. & Jarett, R.D. (1999). Velocity reversal and sediment sorting in pools and riffles controlled by channel constrictions. *Geomorphology*, **27**: 29–241.

Thoms, M.C. & Parsons, M. (2002). *Eco-geomorphology: an interdisciplinary approach to river science. The Structure, Function and Management Implications of Fluvial Sedimentary Systems*. Proceedings of an International Symposium held at Alice Springs, Australia, September 2002. IAHS Publication no. 276.

Thoms, M.C., Reid, M., Christianson, K. & Munro, F. (2006) *Variety is the spice of life: recognizing hydraulic diversity as a tool for managing flows in regulated rivers. In: Sediment dynamics and the hydromorphology of fluvial systems (Proceedings of a symposium held in Dundee, UK, July 2006).* IAHS Publication No. 306. Wallingford, IAHS Press.

Thomson, J.R., Taylor, M.P., Fryirs, K.A. & Brierly, G.J. (2001) A geomorphological framework for river characterization and habitat assessment. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **11** (5): 373-389.

Thorp, J.H., Thoms, M.C. & Delong, M.D. (2006) The riverine ecosystem synthesis: biocomplexity in river networks across space and time. *River Research and Applications*, **22** (2): 123-147.

Tockner, K. & Stanford, J.A. (2002) Riverine flood plains: present state and future trends. *Environmental Conservation*, **29** (3): 308-330.

Townsend, C.R. (1989) The patch dynamics concept of stream community ecology. *Journal of the North American Benthological Society*, **8**: 36-50.

Townsend, C.R., Scarsbrook, M.R. & Doledec, S. (1997) The Intermediate Disturbance Hypothesis, refugia and biodiversity in streams. *Limnology and Oceanography*, **42** (5): 938-949

Turner, M.G., Gardner, R.H. & O'Neill, R.V. (2001) Landscape Ecology in Theory and Practice: Pattern and Process. New York, Springer.

Turner, M.G. (2005) Landscape ecology: what is the state of the science? *Annual Review of Ecology, Evolution and Systematics*, **36**: 319-344.

Turner, W., Spector, S., Gardiner, N., Fladeland, M., Sterling, E. & Steininger, M. (2003) Remote sensing for biodiversity science and conservation. *Trends in Ecology and Evolution*, **18** (6): 306-314.

UKTAG (UK Technical Advisory Group on the Water Framework Directive) (2008) *UK environmental standards and conditions (Phase 1). SRI-2006* [online]. Available at: http://www.wfduk.org/stakeholder_reviews/Standards_Jan_2006/LibraryPublicDocs/UKTAG%20ReportAug%202006UKEnvironmentalStandardsandConditionsFinalReport [Accessed 1st July 2011].

University of Alberta (2002) *River 2D* [online] Available at: http://www.river2d.ualberta.ca/index.htm [Accessed 22nd August 2010].

Uuemaa, E., Antrop, M., Roosaare, J., Marja, R & Mander, U. (2009) Landscape metrics and indices: an overview of their use in research. *Living Reviews in Landscape Research*, **3**: 1-28.

Vadas, R.L.J. & Orth, D.J. (1998) Use of physical variables to discriminate visually determined mesohabitat types in North American streams. *Rivers: Studies in the Science, Environmental Policy and Law of Instream Flow*, **6** (3): 143-159.

Van Nieuwenhuyse, B.H.J, Antoine, M., Wyseure, G. & Govers, G. (2011) Pattern-process relationships in surface hydrology: hydrological connectivity expressed in landscape metrics. *Hydrological Processes*, **25** (24): 3760-3773.

Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R. & Cushing, C.E. (1980) The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences*, **37**: 130-137.

Vaughan, I.P., Diamond, M., Gurnell, A.M., Hall, K.A., Jenkins, A., Milner, N.J., Naylor, L.A., Sear, D.A., Woodward, G. & Ormerod, S.J. (2009) Integrating ecology with hydromorphology: a priority for river science and management. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **19** (1): 113-125.

Vismara, R., Azzellino, A., Bosi, R., Crosa, G. & Gentili, G. (2001) Habitat suitability curves for brown trout (Salmo trutta fario L.) in the River Adda, Northern Italy: comparing univariate and multivariate approaches. *Regulated Rivers: Research and Management*, **17** (1): 37-50.

Vogel, R. (2011) Hydromorphology. *Journal of Water Resources Planning and Management*, **137** (2): 147-149.

Vogel, S. (1994) *Life in moving fluids: the physical biology of flow*. 2nd edition. Princeton, Princeton University Press.

von Luxburg, U. & Ben-David, S. (2005) *Towards a statistical theory of clustering. PASCAL Workshop on Statistics and Optimization of Clustering* [online]. London, 6-7th July 2005. Available from: http://www.cs.waterloo.ca/~shai/LuxburgBendavid05.pdf [Accessed 25th October 2010].

Wadeson, R.A. & Rowntree, K.M. (1998) Application of the hydraulic biotope concept to the classification of instream habitats. *Aquatic Ecosystem Health Management*, **1** (2): 143-157.

Wadeson, R.A. & Rowntree, K.M. (2001) The application of a hydraulic biotope matrix to the assessment of available habitat: Potential application to IFRs and river health monitoring. *African Journal of Aquatic Science*, **26** (1): 67-73.

Wadeson, R.A. (1994) A geomorphological approach to the identification and classification of instream flow environments. *South African Journal of Aquatic Science*, **20**: 38-61.

Waite, I.A.N.R. & Barnhart, R.A. (1992) Habitat criteria for rearing steelhead: a comparison of site-specific and standard curves for use in the instream flow incremental methodology. *North American Journal of Fisheries Management*, **12** (1): 40-46.

Wang, F. & Hall, G.B. (1996) Fuzzy representation of geographical boundaries in GIS. *International Journal of Geographical Information Systems*, **10** (5): 573-590.

Wang, W. & Zhang, Y. (2007) On fuzzy cluster validity indices. *Fuzzy Sets and Systems*, **158**: 2095-2117.

Ward, J. V. & Wiens, J. A. (2001) Ecotones of riverine ecosystems: role and typology, spatio-temporal dynamics, and river regulation. *Ecohydrology and Hydrobiology*, **1**(1-2): 25-36.

Ward, J. V., Tockner, K. & Schiemer, F. (1999) Biodiversity of floodplain river ecosystems: ecotones and connectivity. *Regulated Rivers: Research and Management*, **15**: 125-139.

Ward, J.V. & Stanford, J.A. (1983) The intermediate-disturbance hypothesis: an explanation for biotic diversity patterns in lotic ecosystems. In: Fontaine, T.D. & Bartell, S.M. (eds.) *Dynamics of Lotic Ecosystems.* Ann Arbor, MI, Ann Arbor Science Publishers. pp.347-356.

Ward, J.V. (1989) The four-dimensional nature of lotic ecosystems. *Journal of the North American Benthological Society*, **8** (1): 2-8.

Ward, J.V. (1998) Riverine landscapes: biodiversity patterns, disturbance regimes, and aquatic conservation. *Biological Conservation*, **83** (3): 269-278.

Ward, J.V., Malard, F. & Tockner, K. (2002a) Landscape ecology: a framework for integrating pattern and process in river corridors. *Landscape Ecology*, **17** (S1): 35-45.

Ward, J.V., Tockner, K., Arscott, D.B. & Claret, C. (2002b) Riverine landscape diversity. *Freshwater Biology*, **47**: 517-539.

Ward, J.V., Tockner, K., Uehlinger, U. & Malard, F. (2001) Understanding natural patterns and processes in river corridors as the basis for effective river restoration. *Regulated Rivers: Research & Management*, **17**: 311-323.

Wellnitz, T.A., Poff, N.L., Cosyleón, G. & Steury, B. (2001) Current velocity and spatial scale as determinants of the distribution and abundance of two rheophilic herbivorous insects. *Landscape Ecology*, **16**: 111-120.

Wentworth, C.K. (1922) A scale of grade and class terms for clastic sediments. *Journal of Geology*, **30**: 377-392.

Werner, E.E., Gilliam, J.F., Hall, D.J. & Mittelbach, G,G, (1983) An experimental test of the effects of predation risk on habitat use in fish. *Ecology*, **64** (6): 1540-1548.

Wester, M. & Steinberg, S. (2008) *Gold code. MATLAB toolbox* [online]. Available from: http://stmc.health.unm.edu/tools_software.html [Accessed 21st September 2010].

Wharton, G. & Gilvear, D.J. (2006) River restoration in the UK: Meeting the dual needs of the European Union Water Framework Directive and flood defence? *International Journal of River Basin Management*, **5** (2): 1-12.

Wheaton, J. (2008) CFD Modelling [online]. Available at: http://www.joewheaton.org.uk/ Research/Tools/CFD.asp [Accessed 20th January 2008].

Wheaton, J. M., Gibbins, C., Wainwright, J., Larsen, L. & McElroy, B. (2011) Multiscale feedbacks in ecogeomorphology. *Geomorphology*, **126**: 265-268.

Wheaton, J. M., Pasternack, G. B., and Merz, J. E. (2004) *Use of habitat heterogeneity in salmonid spawning habitat rehabilitation design*. Fifth International Symposium on Ecohydraulics: Aquatic Habitats: Analysis and Restoration, IAHR-AIRH: Madrid, Spain. pp. 791-796.

White, F.S. & Pickett, S.T.A. (1985) Natural disturbance and patch dynamics: an introduction. In: White, F.S. & Pickett, S.T.A. (eds.) *The ecology of natural disturbance and patch dynamics*. London, Academic Press, pp. 3-13.

Wiens, J.A. (1989) Spatial scaling in ecology. *Functional Ecology*, **3**: 385-397.

Wiens J.A. (2002) Riverine landscapes: taking landscape ecology into the water. *Freshwater Biology*, **47** (4): 501-515.

Wiens, J. A. (2002) Central Concepts and Issues of Landscape Ecology. In: Gutzwiller, K.J. (ed.) *Applying landscape ecology in biological conservation*. New York, Springer, pp. 3-21.

Wilby, R. L. (2006) When and where might climate change be detectable in UK river flows? *Geophysical Research Letters*, **33**: L19407.

Williams, J.G. (2010) Lost in space, the sequel: spatial sampling issues with 1-D PHABSIM. *River Research and Applications*, **26** (3): 341-352.

Williams, D. D., Febria, C. M., & Wong, J. C. (2010) Ecotonal and other properties of the hyporheic zone. *Fundamental and Applied Limnology/Archiv für Hydrobiologie*, **176**(4): 349-364.

Wohl, E., Angermeier, P. L., Bledsoe, B., Kondolf, G.M., MacDonnell, L., Merritt, D.M., Palmer, M.A., Poff, N.L. & Tarboton, D. (2005) River restoration. *Water Resources Research*, **41** (10): W10301.

Wolman, M.G. (1954) A method of sampling coarse river-bed material. *Transactions of the American Geophysical Union*, **35** (6): 951-956.

Womble, W.H. (1951) Differential systematics. *Science*, **114**: 315-322.

Woodiwiss, S. (1992) *Iron Age and Roman salt production and the medieval town of Droitwich: excavations at The Old Bowling Green and Friar Street. Centre for British Archaeology Report No. 81* [online]. Available at: http://www.ads.ahds.ac.uk/catalogue/library/cba/rr81.cfm?CFID=3192885&CFTOKEN=85585209 [Accessed 21st October 2009].

Woolnough, D.A., Downing, J.A. & Newton, T.J. (2008) Fish movement and habitat use depends on water body size and shape. *Ecology of Freshwater Fish*, **18** (1): 83–91.

Wright, K.K. & Li, J.L. (2002) From continua to patches: examining stream community structure over large environmental gradients. *Canadian Journal of Fisheries and Aquatic Science*, **59** (8): 1404-1417.

Wu, J. & Loucks, O.L. (1995) From balance of nature to hierarchical patch dynamics: a paradigm shift in ecology. *The Quarterly Review of Biology*, **70** (4): 439-466.

Wu, J., Shen, W., Sun, W. & Tueller, P.T. (2002) Empirical patterns of the effects of changing scale on landscape metrics. *Landscape Ecology*, **17**: 761-782.

Wyrick, J.R. & Pasternack, G.B. (2014) Geospatial organisation of fluvial landforms in a gravel-cobble river: beyond the riffle-pool couplet. *Geomorphology*, **213**: 48-65

Xie, X.L. & Beni, G. (1991) A validity measure for fuzzy clustering. *IEEE Transactions on Pattern Analysis and Machine Intelligence*, **13** (8): 841-847.

Yang, X. & Liu, Z. (2005) Quantifying landscape pattern and its change in an estuarine watershed using satellite imagery and landscape metrics. *International Journal of Remote Sensing*, **26** (23): 5297-5323.

Yarnell, S.M., Mount, J.F. & Larsen, E.W. (2006) The influence of relative sediment supply on riverine habitat heterogeneity. *Geomorphology*, **80**: 310-324.

Yarnell, S.M., Viers, J.H. & Mount, J.F. (2010) Ecology and management of the spring snowmelt recession. *Bioscience*, **60** (2): 114-127.

Zadeh, L.A. (1965) Fuzzy sets. *Information and Control*, **8**: 338-353.

Zalewski, M. (2002) Ecohydrology-the use of ecological and hydrological processes for sustainable management of water resources. *Hydrological Sciences*, **47** (5): 823-832.

Zalewski, M., Janauer, G.A., Jolankai, G. (1997) *Ecohydrology. A new paradigm for the sustainable use of aquatic resources*. International Hydrological Programme UNESCO Paris, Technical Document in Hydrology 7.

Zavadil, E.A., Stewardson, M.J., Turner, M.E. & Ladson, A.R. (2012) An evaluation of surface flow types as a rapid measure of channel morphology for the geomorphic component of river condition assessments. *Geomorphology*, **139-140**: 303-312.

Zhang, J. & Stuart, N. (2001) Fuzzy methods for categorical mapping with image-based land cover data. *International Journal of Geographical Information Science*, **15** (2): 175-195.

Appendix A

ADCP pilot study

A.1 Introduction

Quantification of hydraulic patch dynamics requires high resolution depth and velocity measurements across the full range of discharges. Using conventional stream gauging methods, such as electromagnetic current meters or Acoustic Doppler Velocimeters, field data collection is time consuming, expensive and limited to wadeable flows. Consequently, most habitat studies use low resolution point data collected at low to moderate flows (e.g. Jowett, 1993; Padmore, 1997; Parasiewicz, 2001; Harvey & Clifford, 2009). Hydraulic conditions at high flows have either been overlooked, visually estimated or simulated using hydrodynamic models.

In the last 15 years the introduction of shallow water Acoustic Doppler Current Profilers (ADCPs) has revolutionised stream gauging. The instruments are designed to be towed across the water surface in a continuous steady movement between the banks (moving transect) (Gordon, 1996; Teledyne RDi, 2005). High resolution cross-profile depth and velocity distributions are collected within a few minutes. Conventionally these measurements are converted to stream discharge, however if the raw depth and velocity data could be extracted, ADCPs would be a valuable tool for ecohydraulic studies and could potentially eliminate the limitations of conventional assessment methods (Shields et al., 2003; Stone & Hotchkiss, 2007).

Investigations into the application of ADCPs for uses other than discharge measurements have focused on its suitability to measure turbulence (Muste et al., 2004a,b; Nystrom et al., 2007; Rennie & Church, 2007), secondary currents (Dinehart & Burau, 2005a; Parsons et al., 2005), bed shear stress (Rennie et al., 2002; Sime et al., 2007) and sediment transport (Dinehart & Burau, 2005b; Merckelbach, 2006). Adaptation of ADCPs to collect data suitable for hydraulic patch assessment has only recently begun to be explored (Shields et al., 2003; Stone & Hotchkiss, 2007; Malcolm et al., 2008; Gunawan et al., 2010). Several significant limitations have already been highlighted. Firstly, ADCPs cannot measure near-surface or near-bed velocities due to acoustic ringing and side lobe interference, which compromises their use for characterising benthic invertebrate habitat (Stone & Hotchkiss, 2007; Malcolm et al., 2008; Gunawan et al., 2010). Secondly, the accuracy of ADCP data can be adversely affected by turbulence, turbidity and moving beds, conditions typically associated with high flows, when use of ADCPs would be most advantageous (Stone & Hotchkiss, 2007; Malcolm et al., 2008). Thirdly, it is not recommended to extract data from a standard moving transect to characterise mean velocity distribution as the ADCP motion during the measurement can cause errors (Muste et al., 2004; Gaeuman & Jacobson, 2005). Instead ADCPs must be held at fixed locations to collect reliable estimates of mean velocity (Muste et al., 2004) which compromises the opportunity to collect full cross-profile distributions. Stone & Hotchkiss (2007) recommended collecting point data for a 3-5 minute period to reduce the effect of turbulence on the estimate of velocity. However, this would increase data collection time beyond conventional methods and would not be feasible for a high sampling-resolution hydraulic assessment. Lastly, ADCPs require a minimum depth to operate (≥10 cm for the StreamPro), which limits their use for characterising marginal and shallow water areas (Malcolm et al., 2008). Malcolm et al. (2008) recommend reverting to standard current metering in these areas. However this compromises the potential time savings of using an ADCP and raises compatibility issues between ADCP and current meter data.

ADCP data must be mined to extract depth and velocity data and post-processed to convert data to a form suitable for hydraulic patch characterisation. To maximise compatibility between current meter and ADCP data Malcolm et al. (2008) recommend exporting “Velocity Magnitude” from WinRiverII (ADCP software), extracting velocity at 0.6 depth and temporally averaging measurements collected at each fixed location. However these conclusions were based on a very small field trial using data from just ten point measurements in a small Scottish burn. On closer consideration of the operational differences between current meters and ADCPs it would seem that “Earth Projected Velocity” would be the more suitable variable to export. Current meter readings are the relatively crude product of velocity magnitude and velocity direction. The two components cannot be displayed separately. Velocities in all directions are sensed and water moving in a downstream direction ±90degrees is given a positive value and water moving in an upstream direction ±90degrees is returned as negative velocity (Figure 5). In contrast, ADCPs measure the magnitude and direction of velocity more precisely and store each component separately. “Velocity magnitude” is the average strength of velocity in all directions; no directional threshold is applied to distinguish between positive (downstream) and negative (upstream) velocities. To do so, “Earth Projected Velocity” (Ref. Bottom Tracking) must be exported from WinRiverII. The user must specify the projection angle corresponding with the streamwise direction. This can be calculated from the “Course Made Good” angle, displayed in the Navigation Section of the Composite Tabular Window in WinRiverII (Teledyne RDi, 2007, pp.24-25).

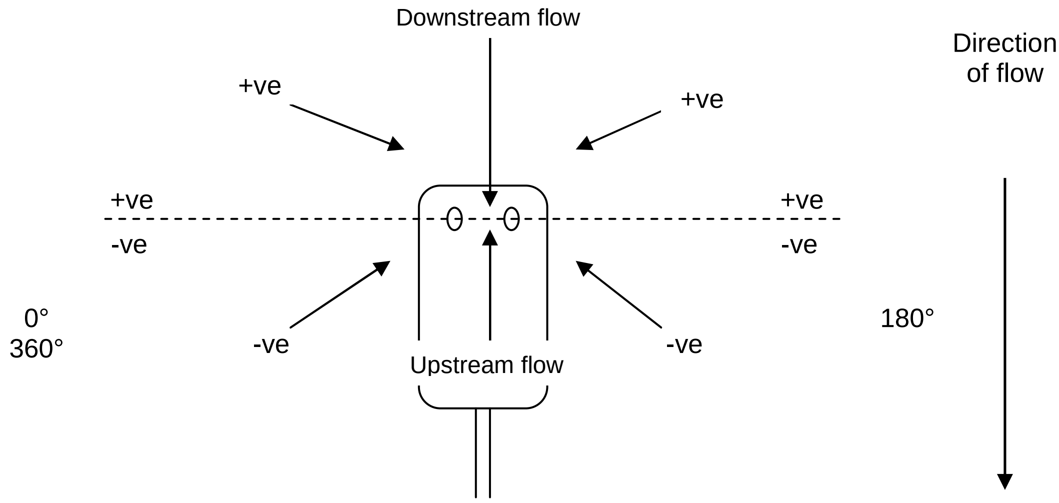


Figure 5. Illustration of how positive (downstream) and negative (upstream) velocities are differentiated by an electromagnetic current meter probe.

In view of the fact that using ADCPs for hydraulic patch assessment is a relatively new application of the technology and Malcom et al.’s (2008) recommendations on deployment and data mining/post-processing recommendations were based on a very limited trial at ten locations in a small Scottish burn, additional field trials in local conditions were deemed worthwhile.

A.2 Methods

Trials were carried out on 27-28th November 2008 at the River Arrow on the declining limb of a flood hydrograph. Discharge was 0.955m3/s (Q20) and 0.516m3/s (Q42) respectively. The objectives were to a) evaluate compatibility between ADCP and current meter data collected at fixed locations for 10 seconds and b) determine procedures for exporting and post-processing ADCP data. Eleven cross-sections were selected in run, pool and glide CGUs. Velocity was sampled for 10 seconds at 50cm intervals on each cross-section using a Valeport electromagnetic current meter (deployed at 0.6 depth) and a Teledyne RDi StreamPro ADCP. “Velocity Magnitude” and “Earth Projected Velocity” were exported from WinRiverII to Excel for post-processing and comparison. Velocity at 0.6 depth was extracted from the ten ensembles (vertical profiles) collected at each location and averaged.

A.3 Results

A total of 94 locations were sampled using each method. Compatibility between each method at three cross-profiles is shown in Figure 6. Contrary to Malcolm et al.’s (2008) findings, large discrepancies between current meter and ADCP “Velocity Magnitude” data were evident, particularly at channel margins where eddying often creates negative velocities. Figure 6 shows much better agreement between current meter data and ADCP “Earth Projected Velocity”, particularly in the glide/backwater cross-profile, where the pattern of positive and negative velocities concords. Total mean velocity at the 94 sample locations was 0.177m/s, 0.316m/s and 0.188m/s, as measured by the current meter, ADCP “Velocity Magnitude” and ADCP “Earth Projected Velocity” methods respectively. Pairwise comparisons of mean velocity measured using each method were performed using the Mann-Whitney U test. Results showed a significant difference in mean velocity measured by the current meter and ADCP “Velocity Magnitude” methods (Mann-Whitney $U=2131.5$, $n1=n2=94$, $p<0.05$) and between the two ADCP methods (Mann-Whitney $U=2175.0$, $n2=n3=94$, $p<0.05$). However mean velocity measured using the current meter and ADCP “Earth Projected Velocity” methods was not significantly different (Mann-Whitney $U=4245$, $n1=n3=94$, $p>0.05$).

A.4 Discussion

Malcolm et al. (2008) reported good compatibility between current meter data and ADCP “Velocity Magnitude” data, provided velocity at 0.6 depth was extracted. This contrasts with the significant difference found in this study. Where ADCP “Earth Projected Velocity” data were used, the mean difference in velocity magnitude recorded at the 94 point locations was reduced to -0.011m/s (6%), a non-significant difference. Small discrepancies between measurements made by each method are inevitable due to natural hydraulic variability over the sampling period and the greater directional precision of the ADCP streamwise velocity data. The findings reflect the 5% accuracy rates between current meter and ADCP velocity measurements reported in other studies (Shih et al., 2000; Oberg, 2002; Mueller, 2003; Gunawan et al., 2008).

The results suggest that the potential time savings of using an ADCP are lost by the need to deploy the ADCP at fixed locations to ensure data compatibility. In this study a 10 second sampling period was used for each measurement. Even so, the average time to sample a cross-profile at 50cm intervals where average channel width was 3.8m was 15 minutes. A standard moving transect was required to configure the ADCP at each cross-profile before point measurements could be taken. In addition, supplementary current metering is necessary in all shallow marginal areas. As such, the main advantage of ADCP use is the ability to collect data at high flows from the safety of the river bank rather than providing any significant time savings compared with using current meters.

A.5 Conclusions

The use of ADCPs for hydraulic patch assessment is a relatively new application of the technology. Fixed location deployment and careful data mining and post-processing are required to extract reliable estimates of mean velocity (Muste et al., 2004b; Malcolm et al., 2008). The results from this trial suggest that 6% accuracy rates between 10 second point velocity measurements made using a current meter and StreamPro ADCP can be achieved, where “Earth Projected Velocity” variable is exported from WinRiverII. The results of the pilot study formed the basis of data collection methods used in the main study, which are described in the following section.

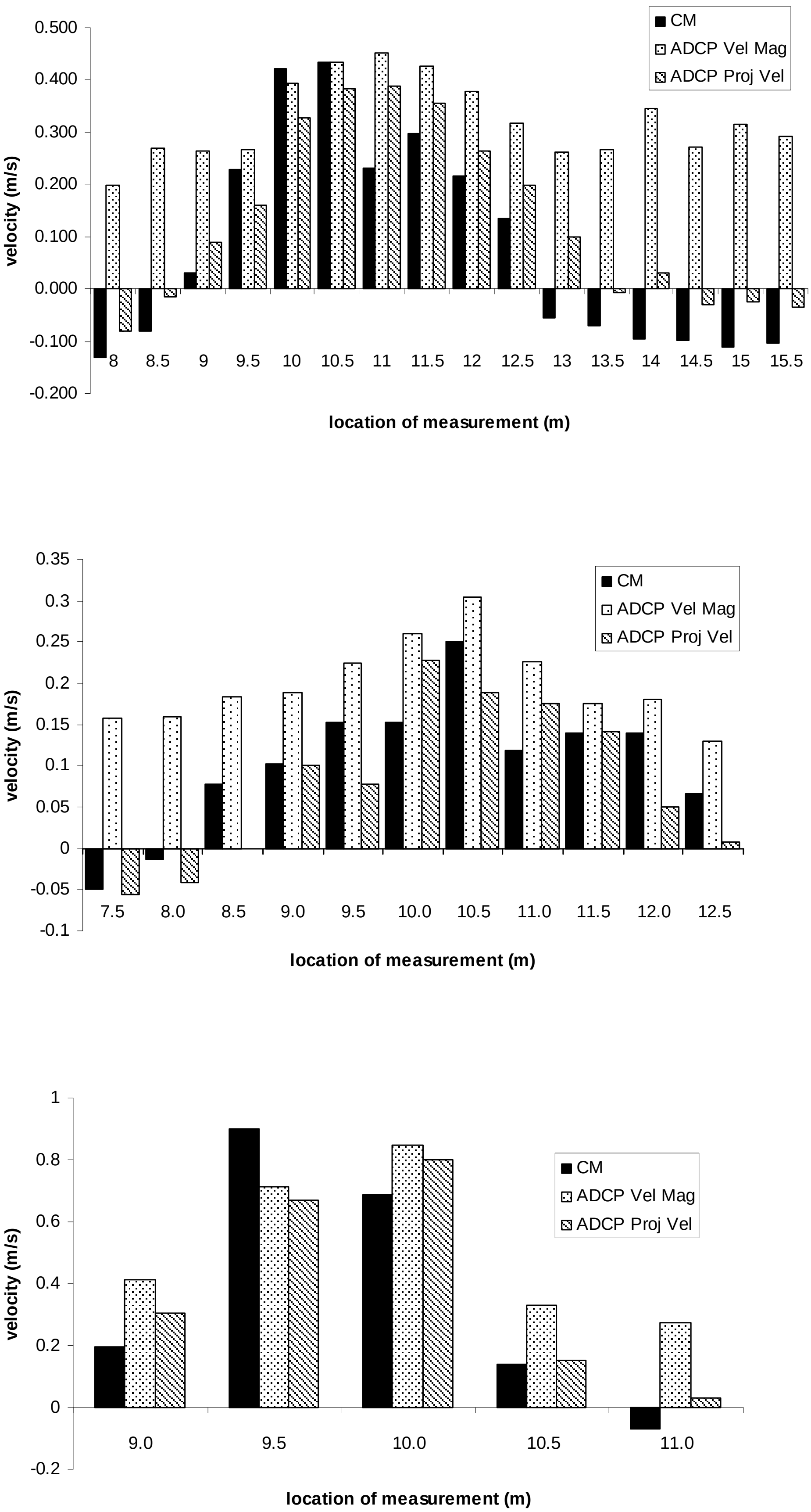


Figure 6. Compatibility between velocity measurements using a current meter, ADCP "Velocity Magnitude" and ADCP "Earth Projected Velocity". Cross-profile data from a glide/backwater (top), pool (middle) and run (bottom) are shown for comparison.

Appendix B

River Arrow hydraulic patch classifications

1. Fuzzy *c*-means classifications

Table 1. Cluster centroids for the 2 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Deep	0.62	0.142
2	Shallow	0.23	0.242

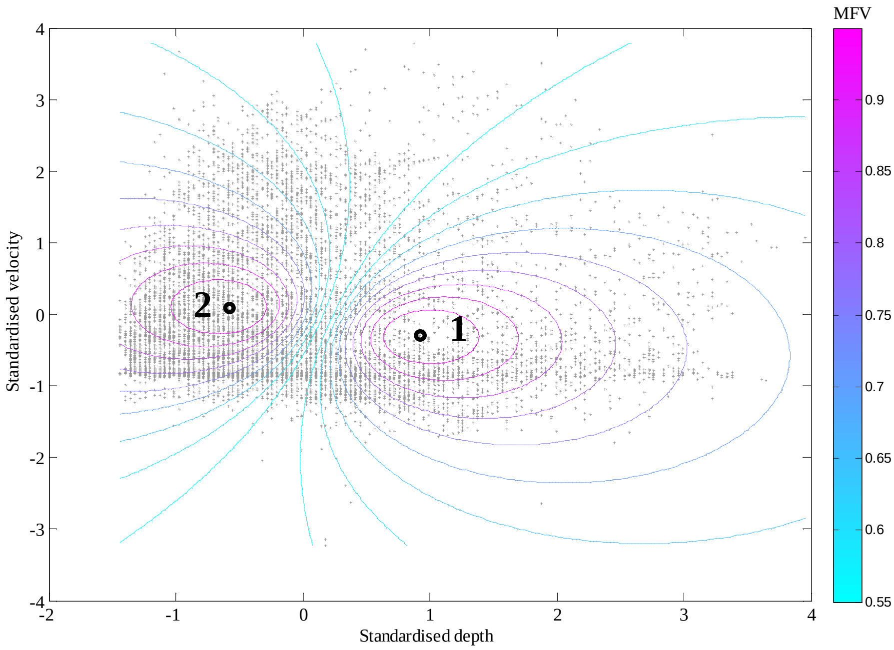


Figure 1. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 2-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 2.

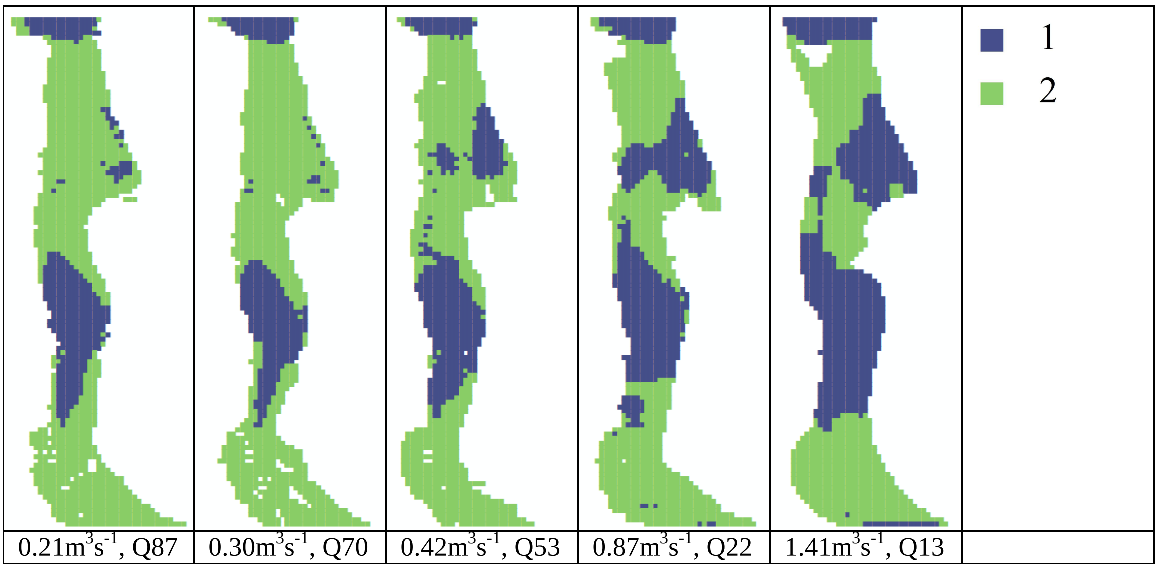


Figure 2. Location and extent of clusters in the 2-FCM maximum likelihood classification

Table 2. Cluster centroids for the 3 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Deep- slow	0.70	0.052
2	Shallow-slow	0.22	0.108
3	Moderate-fast	0.37	0.616

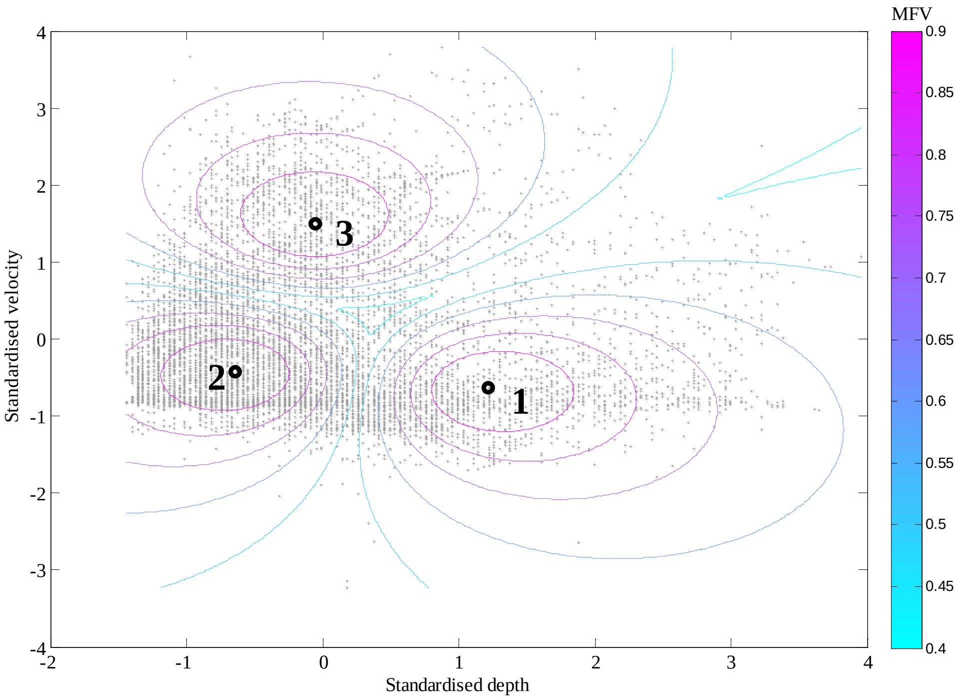
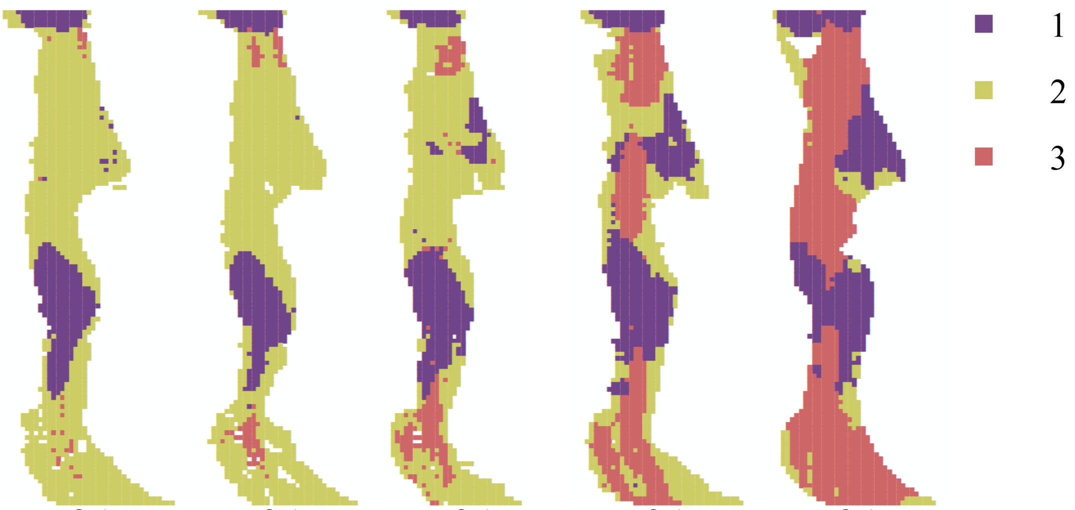


Figure 3. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 3-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 4.



$0.21m^3s^{-1}$, Q87 $0.30m^3s^{-1}$, Q70 $0.42m^3s^{-1}$, Q53 $0.87m^3s^{-1}$, Q22 $1.41m^3s^{-1}$, Q13

Figure 4. Location and extent of clusters in the 3-FCM maximum likelihood classification.

Table 3. Cluster centroids for the 4 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Deep-slow	0.87	0.092
2	Moderate-fast	0.36	0.646
3	Moderate-slow	0.43	0.046
4	Shallow-slow	0.16	0.138

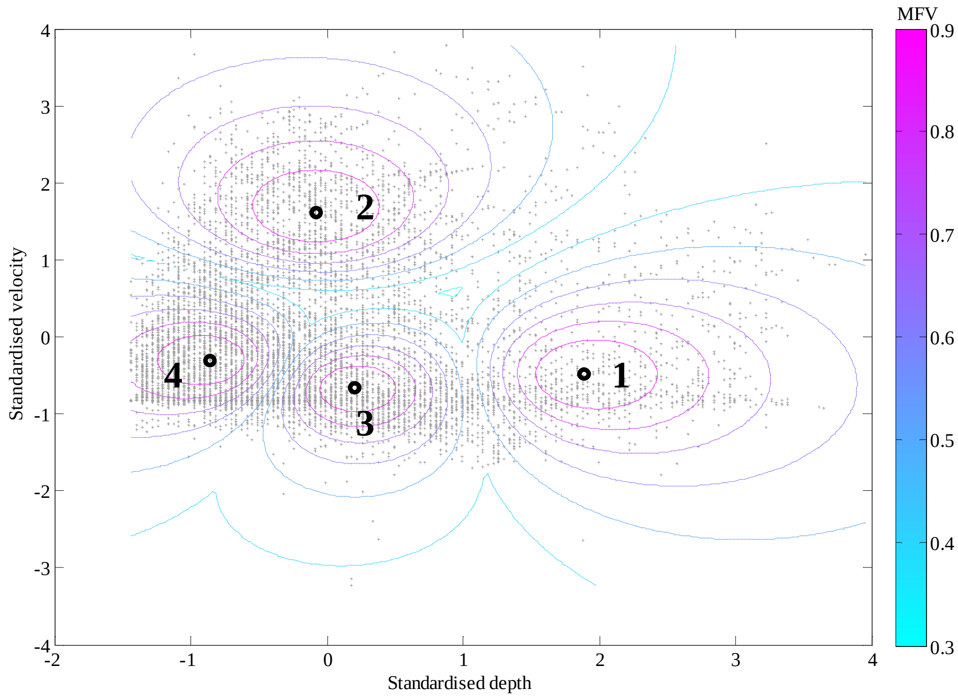
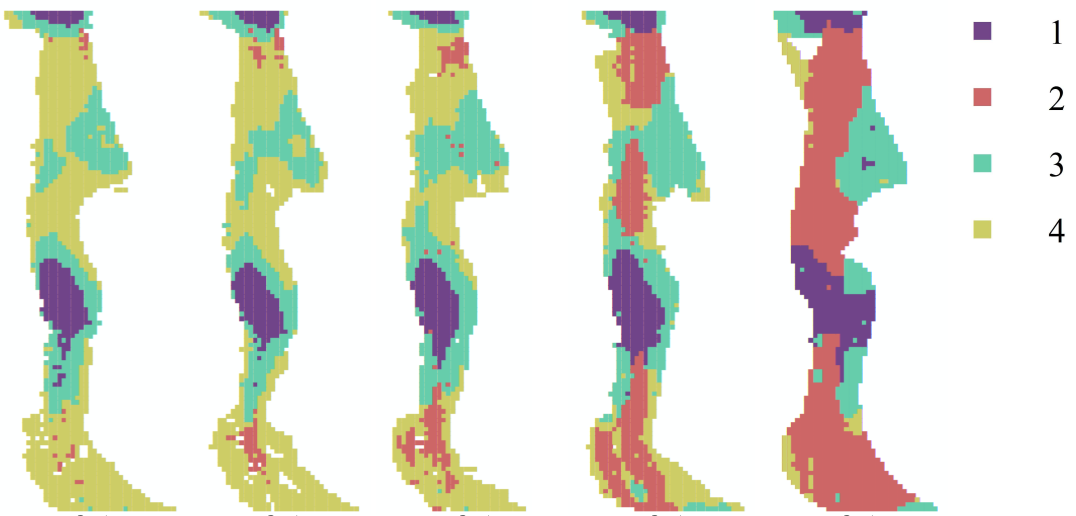


Figure 5. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 4-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 6.



0.21m³s⁻¹, Q87 0.30m³s⁻¹, Q70 0.42m³s⁻¹, Q53 0.87m³s⁻¹, Q22 1.41m³s⁻¹, Q13

Figure 6. Location and extent of clusters in the 4-FCM maximum likelihood classification

Table 4. Cluster centroids for the 5 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Shallow-moderate	0.25	0.342
2	Shallow-slow	0.16	0.070
3	Moderate-slow	0.49	0.020
4	Moderate-fast	0.39	0.706
5	Deep-slow	0.91	0.092

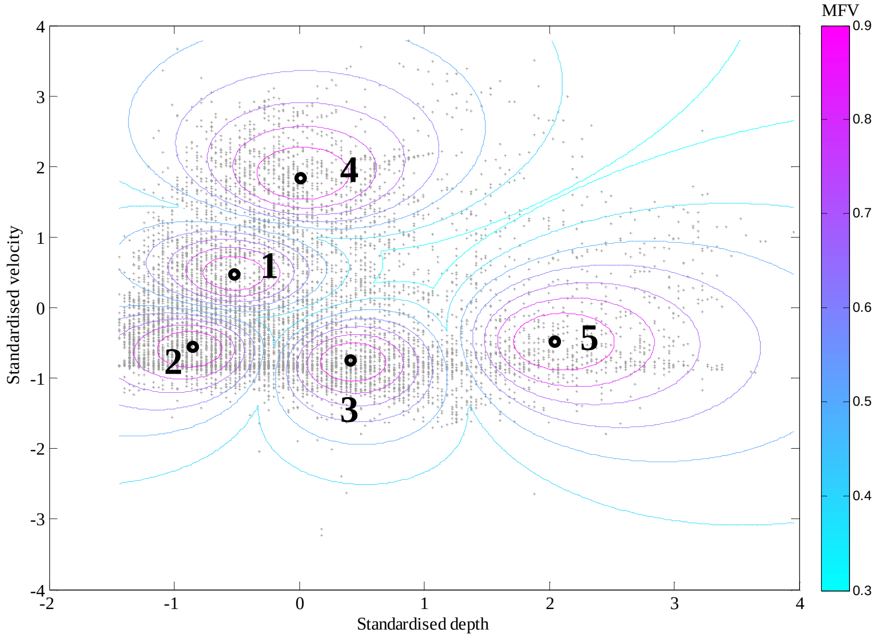
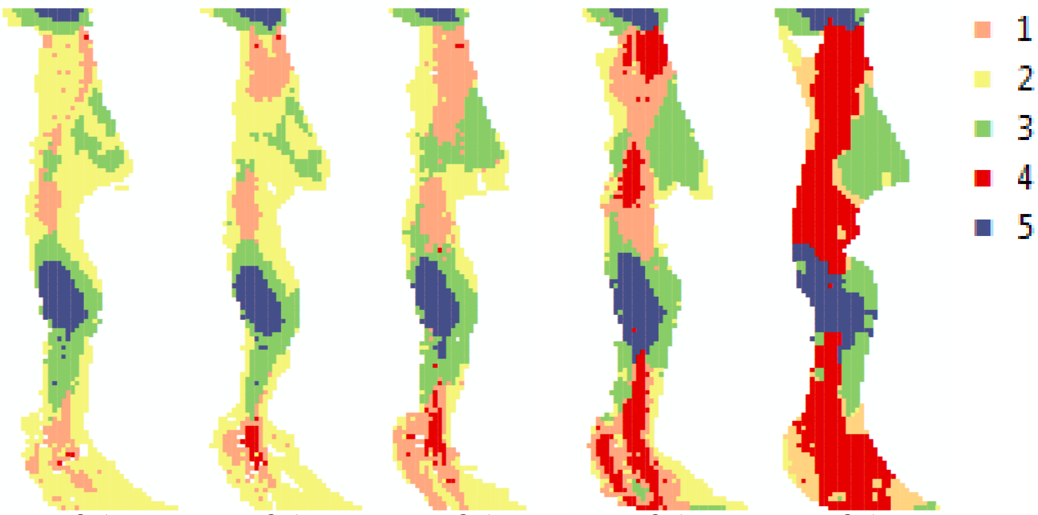


Figure 7. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 5-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 8.



$0.21m^3s^{-1}$, Q87 $0.30m^3s^{-1}$, Q70 $0.42m^3s^{-1}$, Q53 $0.87m^3s^{-1}$, Q22 $1.41m^3s^{-1}$, Q13

Figure 8. Location and extent of clusters in the 5-FCM maximum likelihood classification

Table 5. Cluster centroids for the 6 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Deep-slow	0.59	0.010
2	Shallow-slow	0.12	0.093
3	Moderate-fast	0.39	0.725
4	Moderate-slow	0.34	0.063
5	Very deep-slow	0.97	0.107
6	Shallow-moderate	0.26	0.407

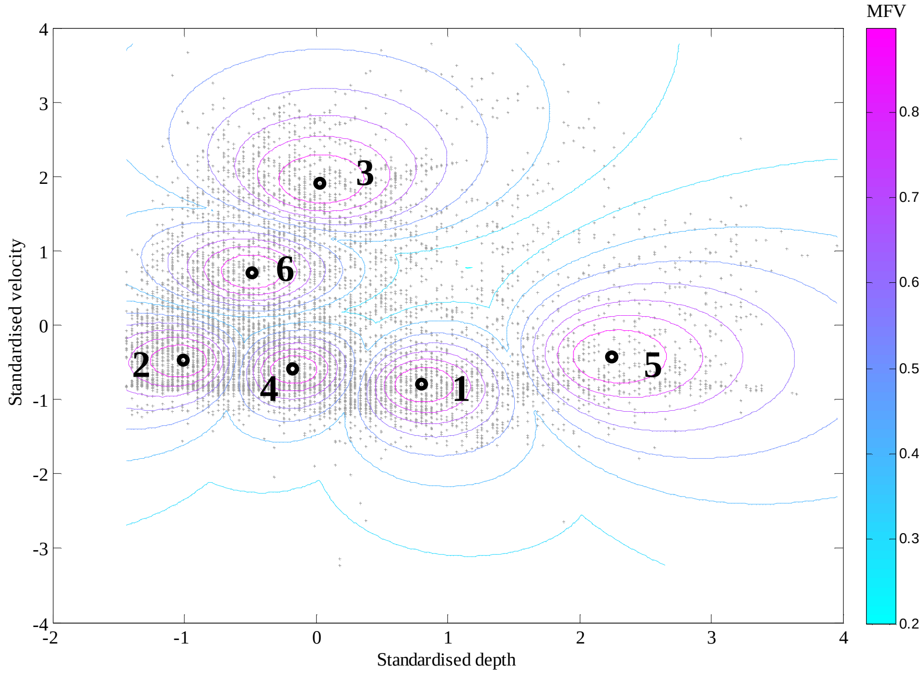
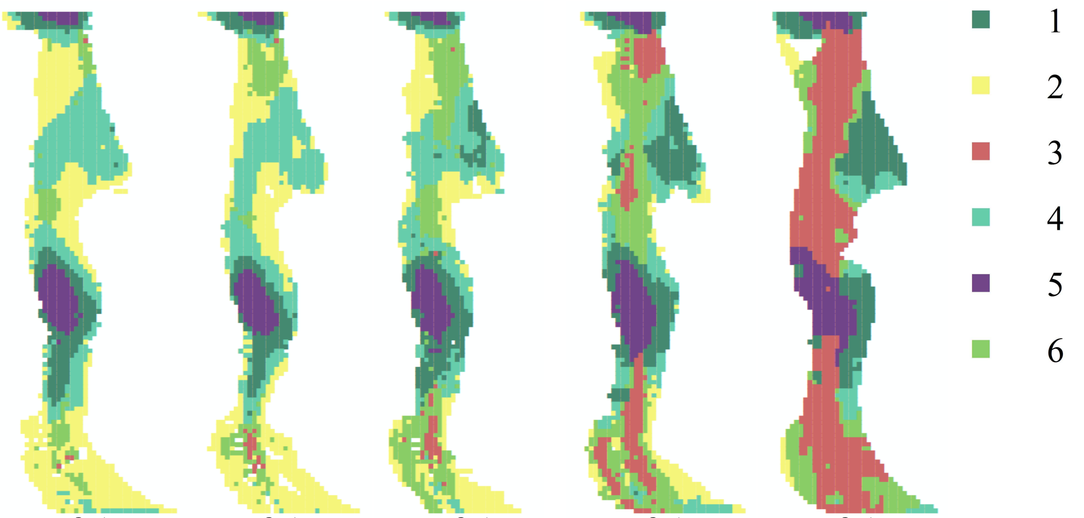


Figure 9. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 6-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 10.



$0.21m^3s^{-1}$, Q87 $0.30m^3s^{-1}$, Q70 $0.42m^3s^{-1}$, Q53 $0.87m^3s^{-1}$, Q22 $1.41m^3s^{-1}$, Q13

Figure 10. Location and extent of clusters in the 6-FCM maximum likelihood classification

Table 6. Cluster centroids for the 7 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Deep-slow	0.60	0.002
2	Shallow-slow	0.12	0.069
3	Moderate-slow	0.35	0.048
4	Shallow-moderate	0.20	0.328
5	Very deep-slow	0.97	0.089
6	Moderate-moderate	0.49	0.527
7	Moderate-fast	0.33	0.749

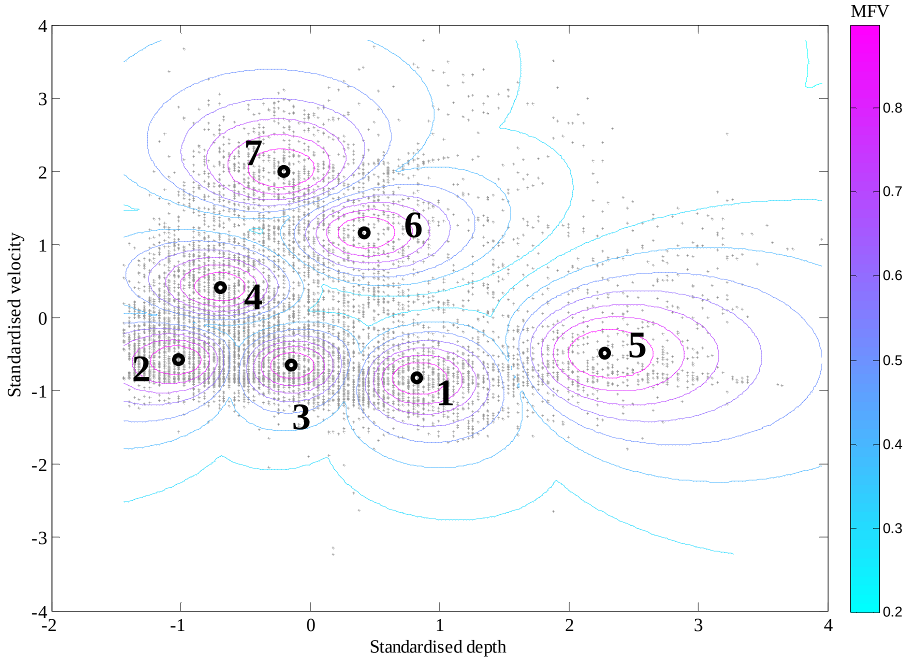
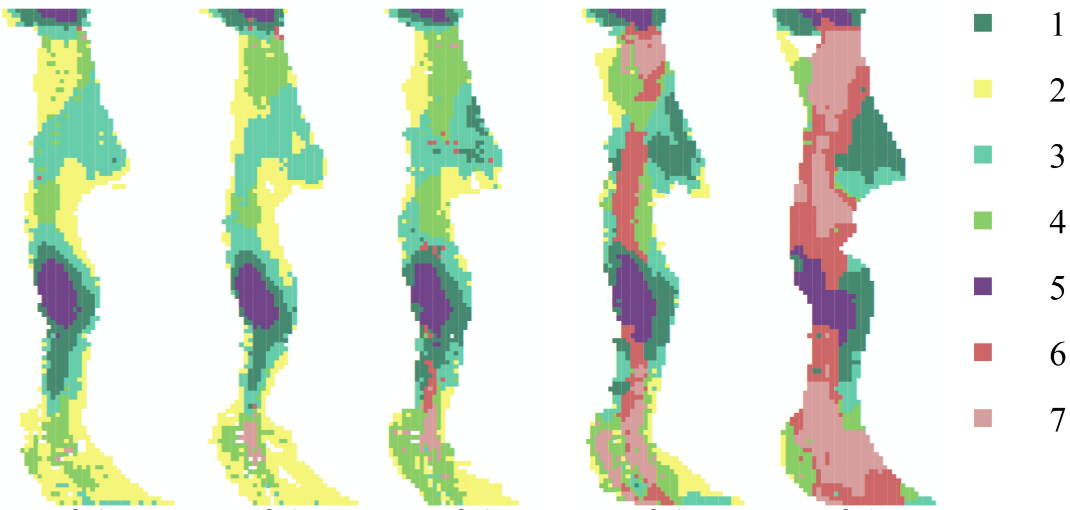


Figure 11. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 7-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 12.



0.??m^3s^{-1}, Q87 0.30m^3s^{-1}, Q70 0.42m^3s^{-1}, Q53 0.87m^3s^{-1}, Q22 1.41m^3s^{-1}, Q13

Figure 12. Location and extent of clusters in the 7-FCM maximum likelihood classification

Table 7. Cluster centroids for the 8 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Very deep-slow	0.99	0.088
2	Shallow-slow	0.11	0.061
3	Shallow-moderate	0.20	0.405
4	Moderate-fast	0.32	0.755
5	Deep-slow	0.63	0.005
6	Moderate-moderate	0.53	0.581
7	Moderate-slow	0.38	0.015
8	Shallow-slow 2	0.28	0.193

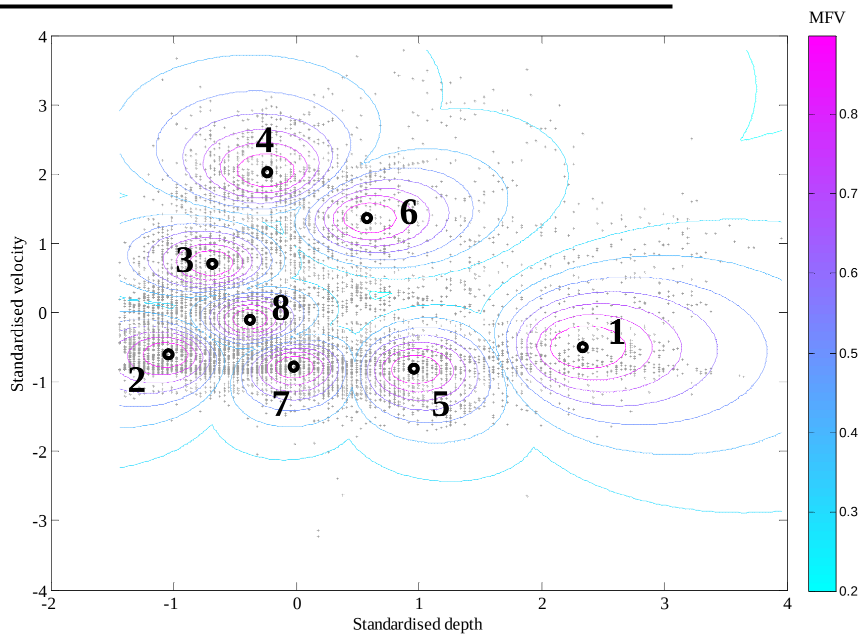
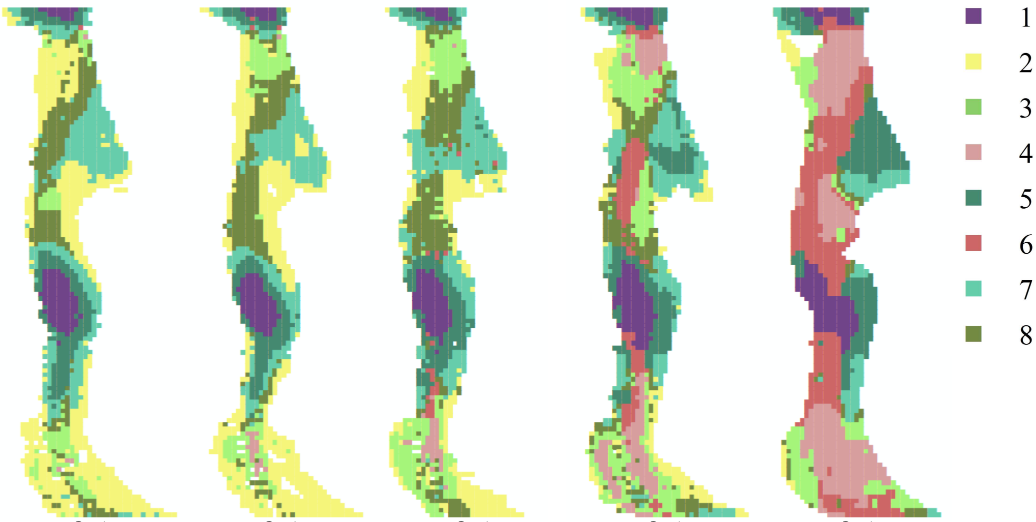


Figure 13. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 8-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 14.



0.21m^3s^{-1}, Q87   0.30m^3s^{-1}, Q70   0.42m^3s^{-1}, Q53   0.87m^3s^{-1}, Q22   1.41m^3s^{-1}, Q13

Figure 14. Location and extent of clusters in the 8-FCM maximum likelihood classification

2. Gustasfon-Kessel fuzzy covariance classifications

Table 8. Cluster centroids for the 2-cluster Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Slow	0.42	0.049
2	Fast	0.34	0.517

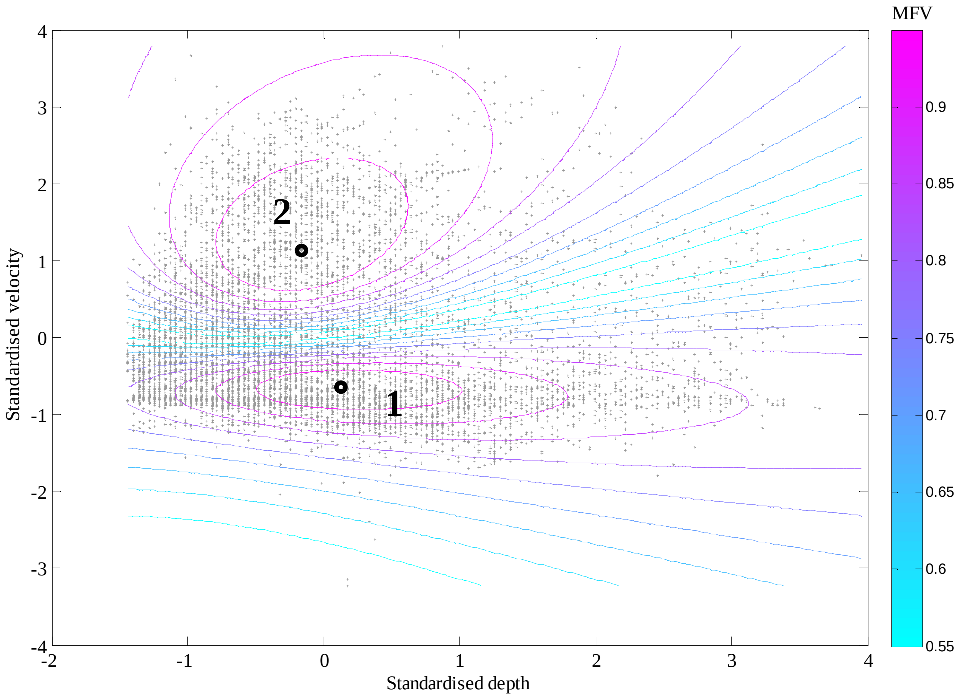
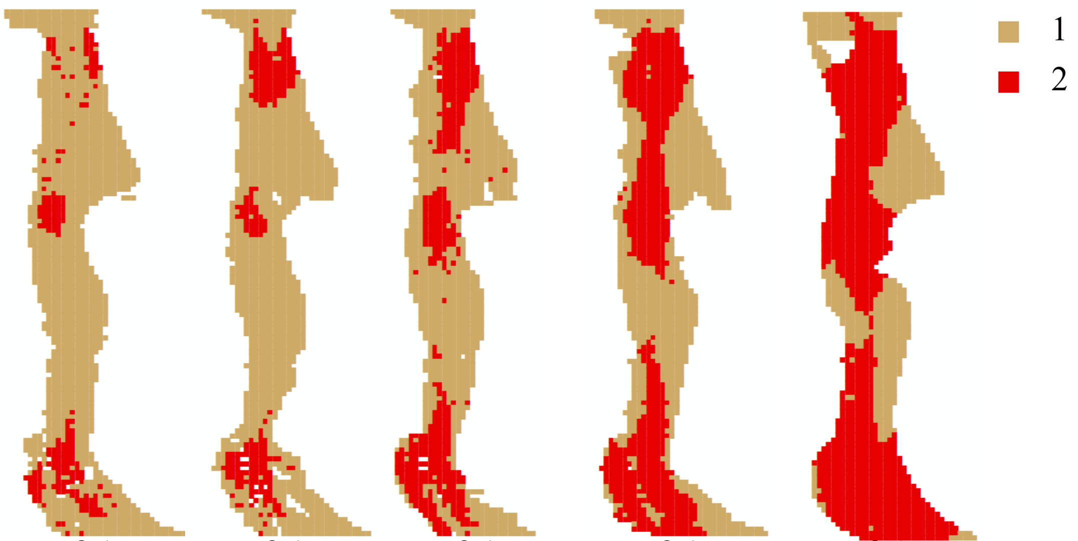


Figure 15. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 2-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 16.



$0.21m^3s^{-1}$, Q87 $0.30m^3s^{-1}$, Q70 $0.42m^3s^{-1}$, Q53 $0.87m^3s^{-1}$, Q22 $1.41m^3s^{-1}$, Q13

Figure 16. Location and extent of clusters in the 2-GK maximum likelihood classification

Table 9. Cluster centroids for the 3-cluster Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Deep- slow	0.63	0.019
2	Shallow-slow	0.21	0.145
3	Moderate-fast	0.38	0.630

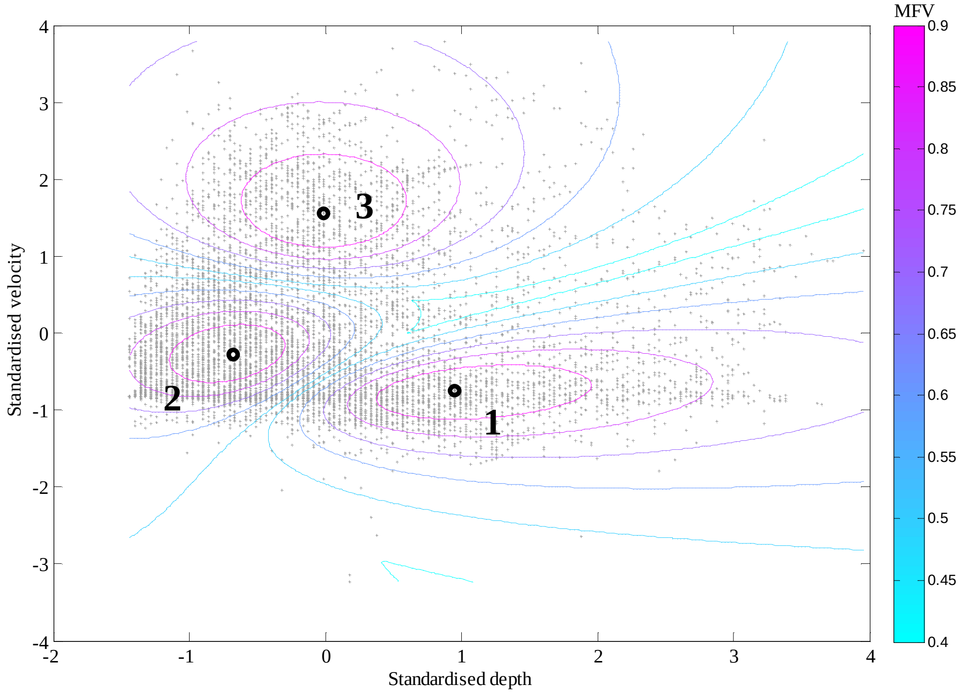
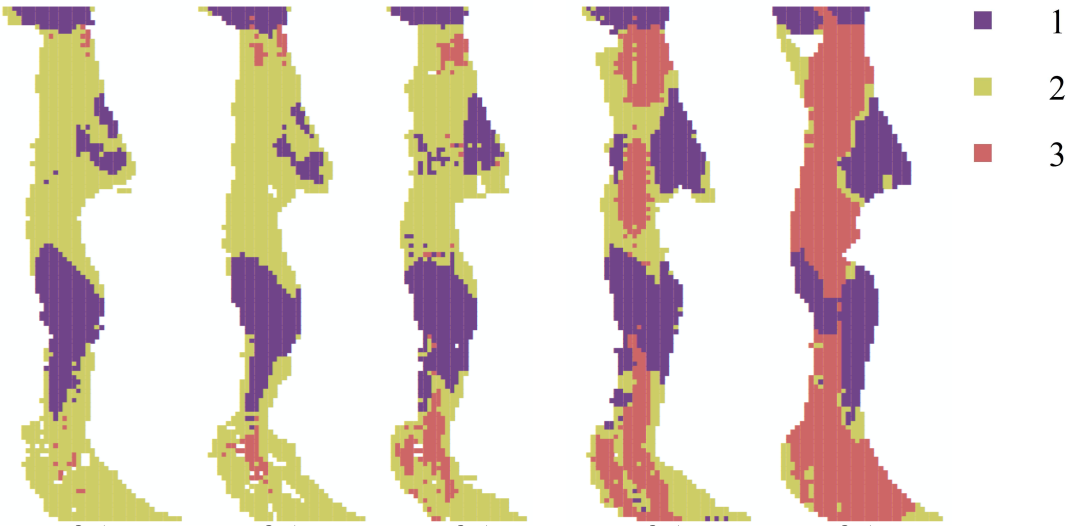


Figure 17. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 3-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 18.



$0.21m^3s^{-1}$, Q87 $0.30m^3s^{-1}$, Q70 $0.42m^3s^{-1}$, Q53 $0.87m^3s^{-1}$, Q22 $1.41m^3s^{-1}$, Q13

Figure 18. Location and extent of clusters in the 3-GK maximum likelihood classification

Table 10. Cluster centroids for the 4 Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderate-fast	0.39	0.690
2	Shallow-moderate	0.21	0.293
3	Deep-slow	0.71	0.009
4	Shallow-slow	0.27	0.054

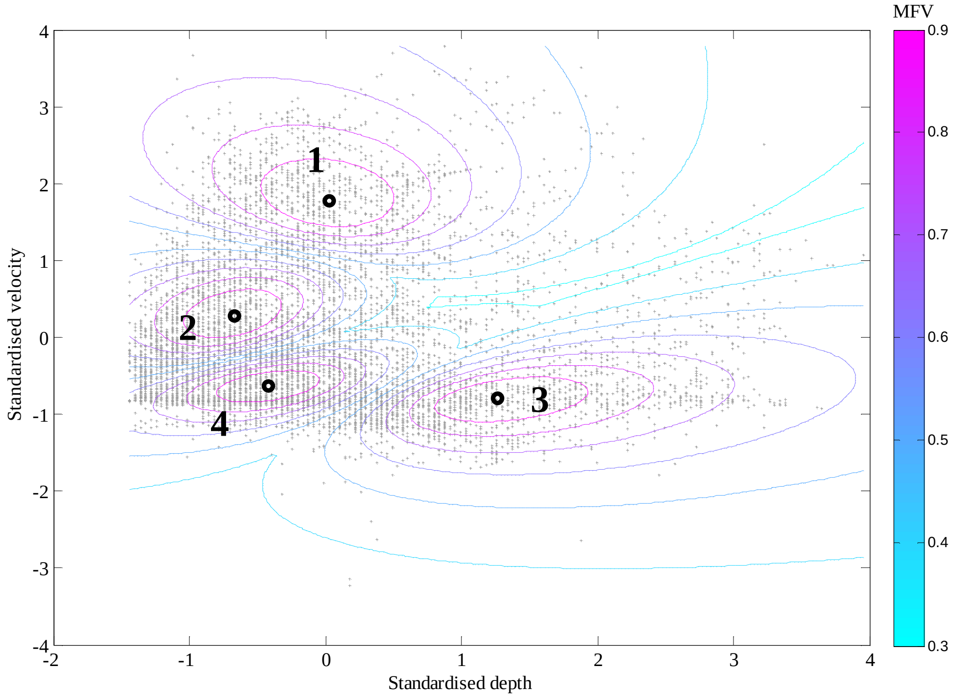


Figure 19. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 4-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 20.

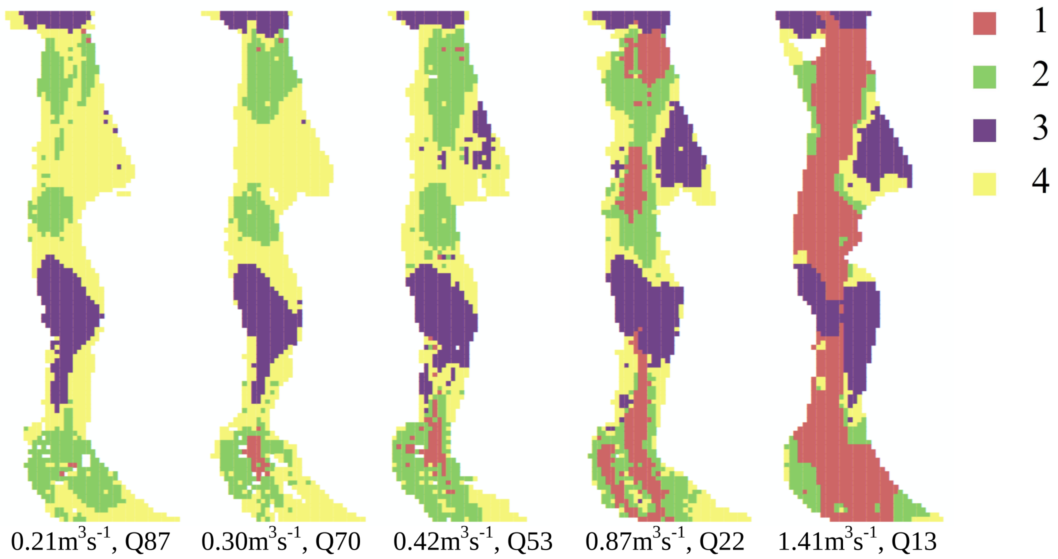


Figure 20. Location and extent of clusters in the 4-GK maximum likelihood classification.

Table 11. Cluster centroids for the 5-cluster Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderate-slow	0.37	0.217
2	Shallow-slow	0.14	0.272
3	Moderate-very slow	0.38	0.001
4	Moderate-fast	0.39	0.696
5	Very deep-very slow	0.86	0.062

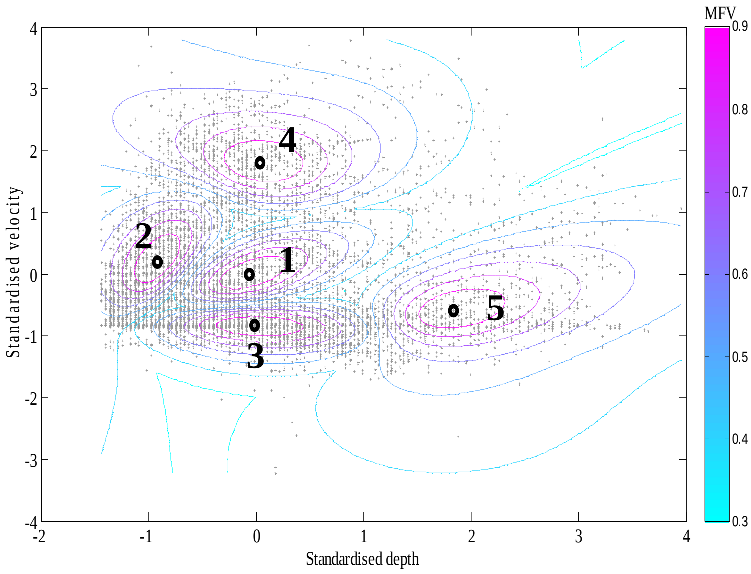
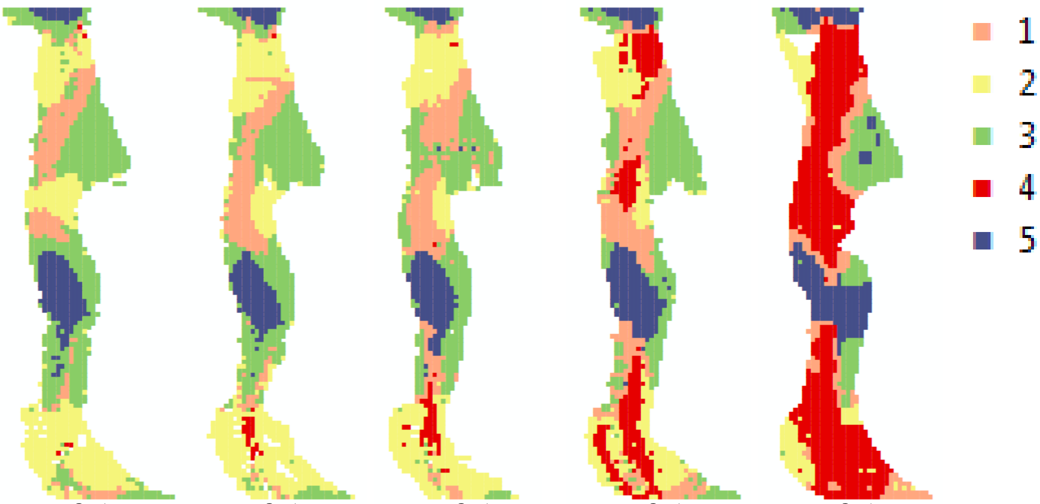


Figure 21. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 5-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 22.



$0.21m^3s^{-1}$, Q87 $0.30m^3s^{-1}$, Q70 $0.42m^3s^{-1}$, Q53 $0.87m^3s^{-1}$, Q22 $1.41m^3s^{-1}$, Q13

Figure 22. Location and extent of clusters in the crisp 5-GK maximum likelihood classification

Table 12. Cluster centroids for the 6-cluster Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Deep-slow	0.93	0.068
2	Shallow-slow	0.12	0.148
3	Moderate-fast	0.37	0.748
4	Deep-slow	0.54	-0.006
5	Moderate-slow	0.30	0.077
6	Moderate-moderate	0.31	0.453

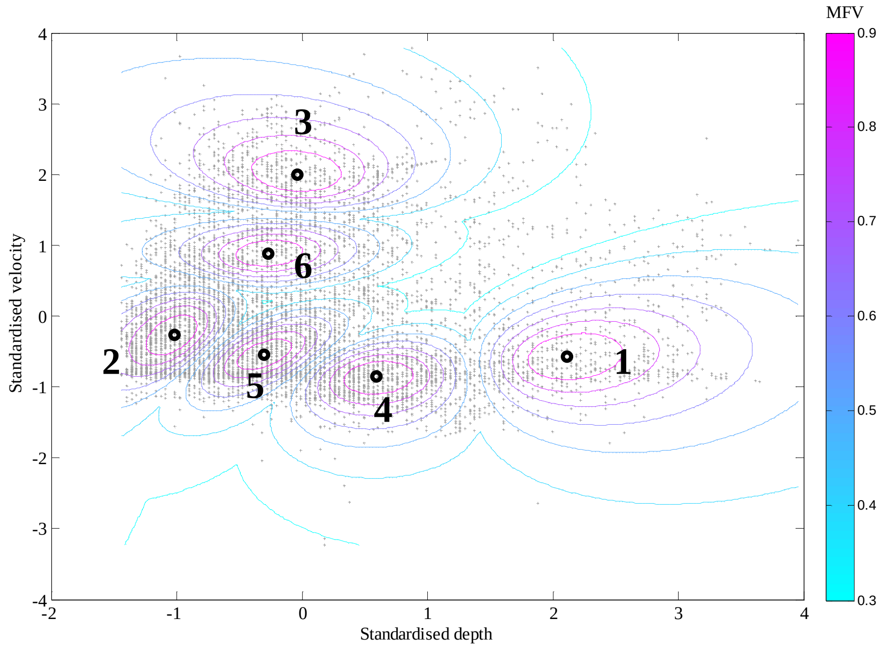
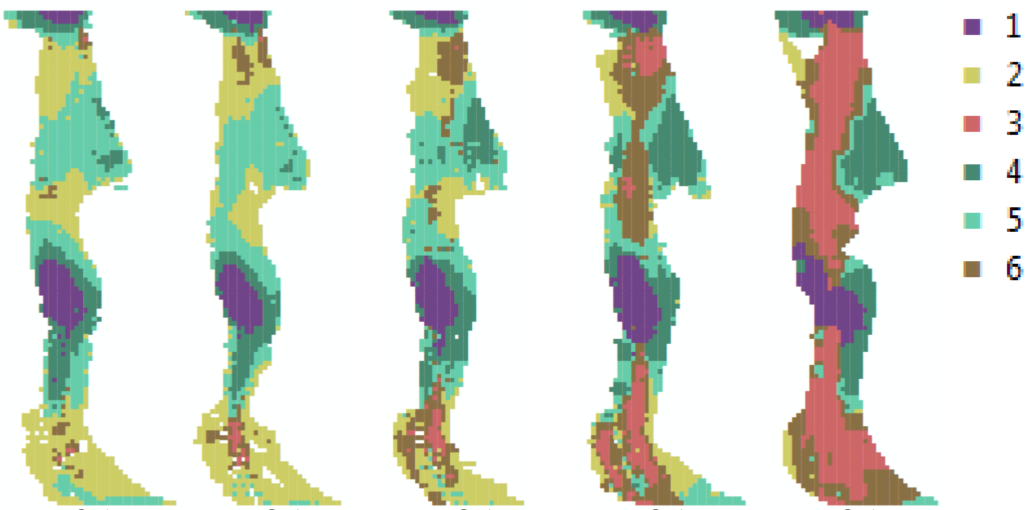


Figure 23. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 6-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 24.



$0.21m^3s^{-1}$, Q87 $0.30m^3s^{-1}$, Q70 $0.42m^3s^{-1}$, Q53 $0.87m^3s^{-1}$, Q22 $1.41m^3s^{-1}$, Q13

Figure 24. Location and extent of clusters in the crisp 6-GK maximum likelihood classification.

Table 13. Cluster centroids for the 7-cluster Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Deep-slow	0.57	-0.008
2	Shallow-moderate	0.12	0.241
3	Moderate- very slow	0.26	0.015
4	Moderate-slow	0.33	0.201
5	Very deep-slow	0.95	0.065
6	Moderate-moderate	0.39	0.510
7	Moderate-fast	0.37	0.767

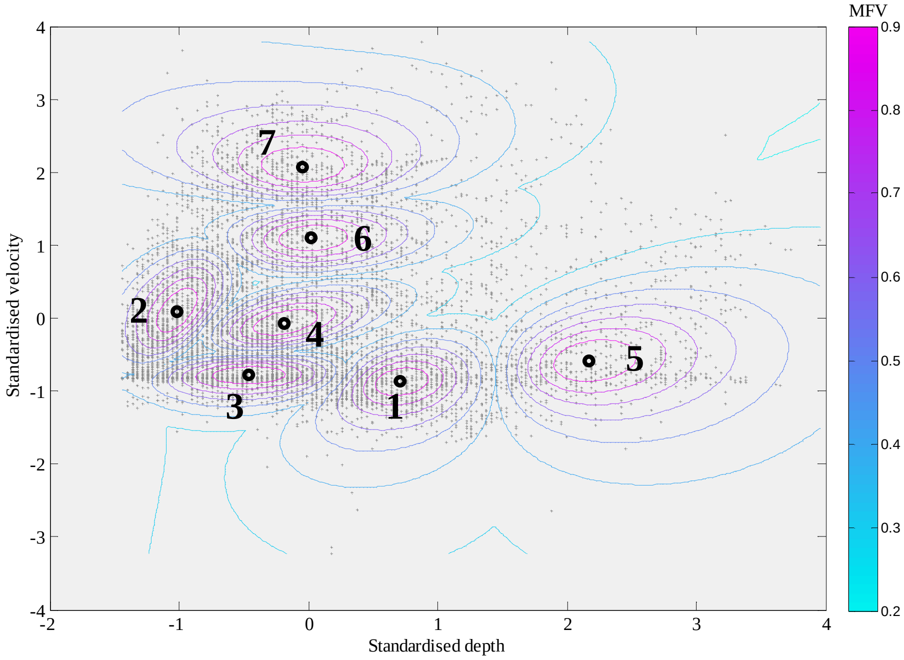
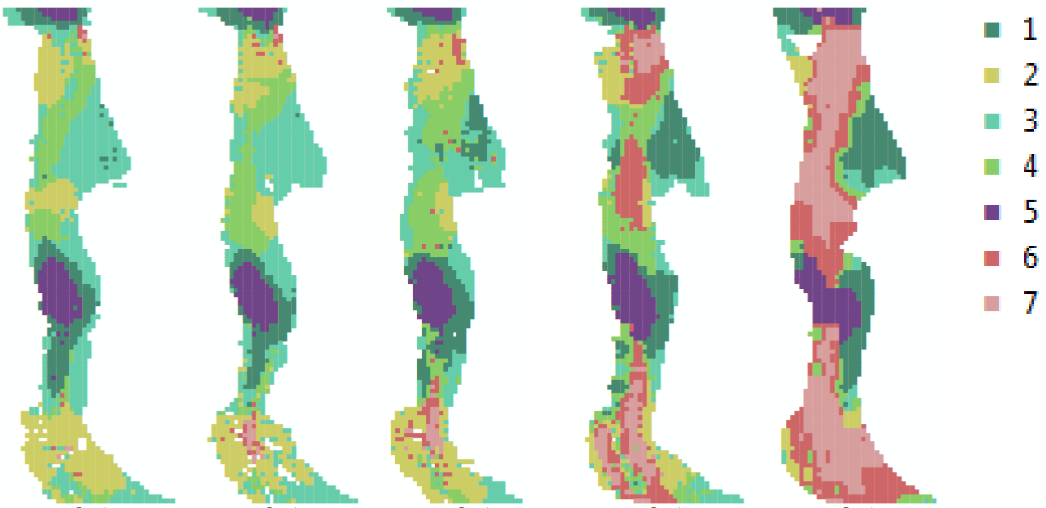


Figure 25. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 6-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 26.



0.21m^3s^{-1}, Q87 0.30m^3s^{-1}, Q70 0.42m^3s^{-1}, Q53 0.87m^3s^{-1}, Q22 1.41m^3s^{-1}, Q13

Figure 26. Location and extent of clusters in the crisp 7-GK maximum likelihood classification

Table 14. Cluster centroids for the 8-cluster Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Very deep-slow	0.95	0.061
2	Shallow-slow	0.13	0.185
3	Moderate-moderate	0.26	0.430
4	Moderate-fast	0.29	0.744
5	Deep-very slow	0.57	-0.013
6	Deep-fast	0.54	0.663
7	Shallow-very slow	0.26	0.008
8	Moderate-slow	0.36	0.174

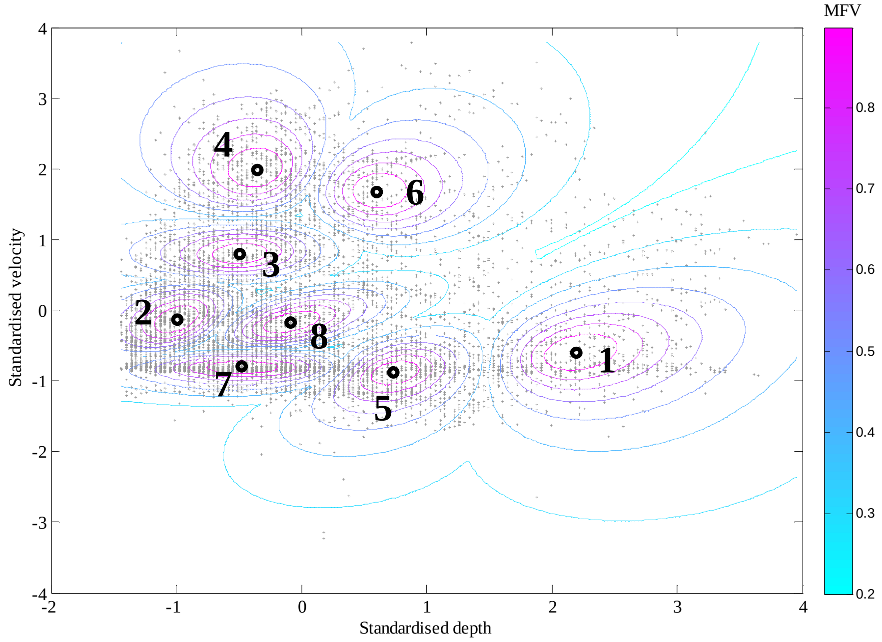
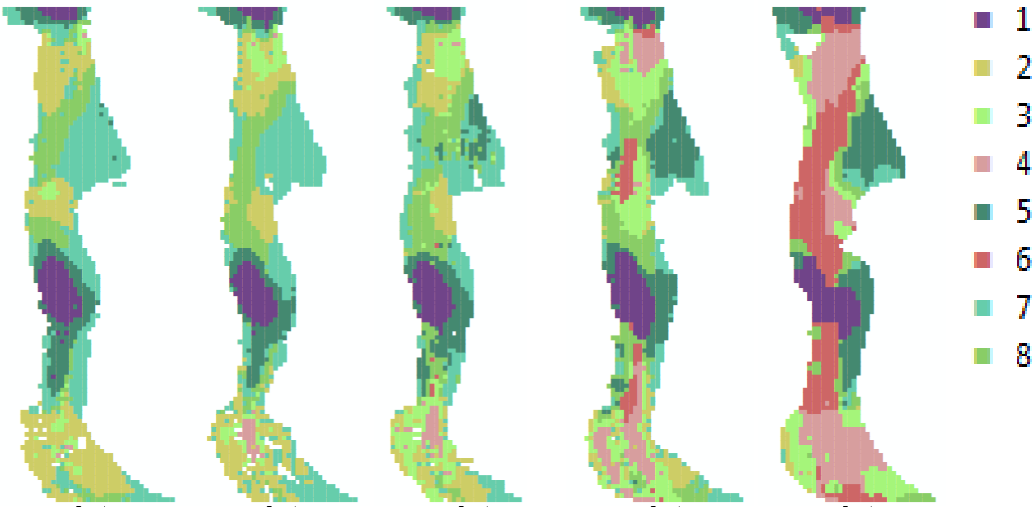


Figure 27. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 6-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 28.



$0.21m^3s^{-1}$, Q87 $0.30m^3s^{-1}$, Q70 $0.42m^3s^{-1}$, Q53 $0.87m^3s^{-1}$, Q22 $1.41m^3s^{-1}$, Q13

Figure 28. Location and extent of clusters in the crisp 8-GK maximum likelihood classification.

Appendix C

River Salwarpe hydraulic patch classifications

1. Fuzzy *c*-means classifications

Table 1. Cluster centroids for the 2 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderate-fast	0.28	0.708
2	Shallow-slow	0.14	0.315

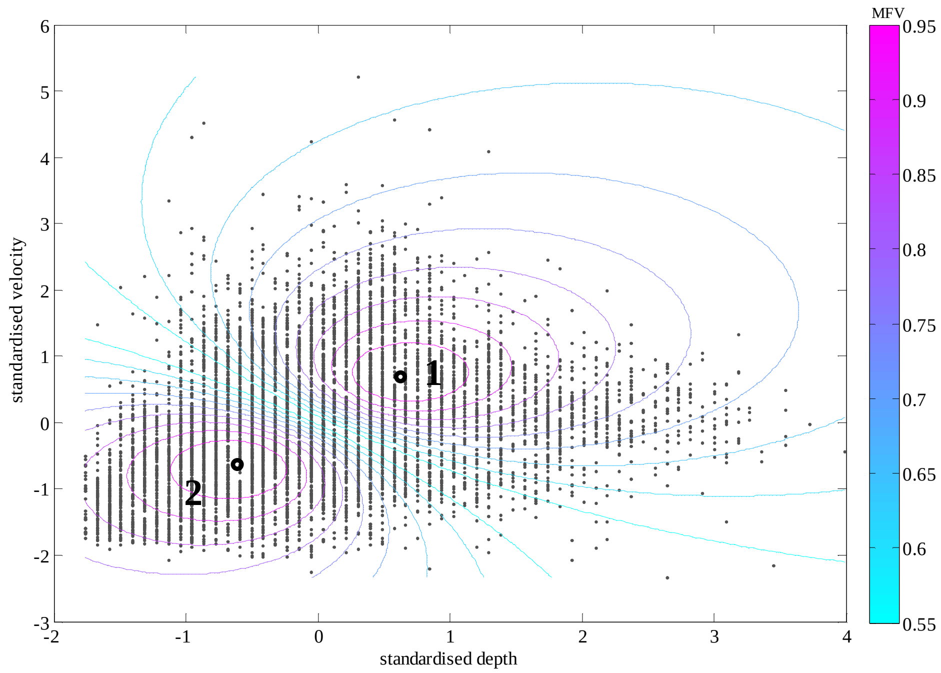
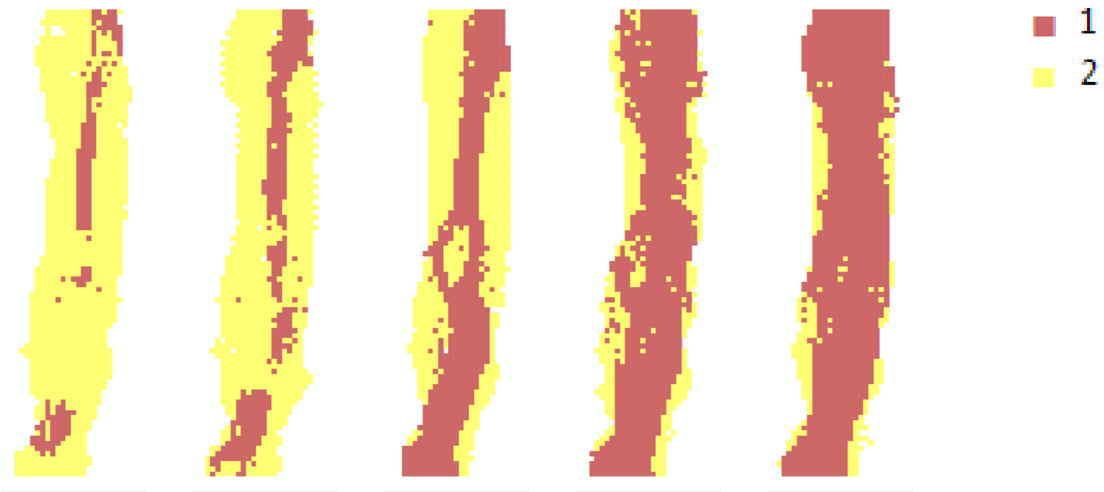


Figure 1. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 2-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 2.



$0.53m^3s^{-1}$, Q89 $0.79m^3s^{-1}$, Q67 $1.14m^3s^{-1}$, Q38 $1.63m^3s^{-1}$, Q21 $1.84m^3s^{-1}$, Q16

Figure 2. Location and extent of clusters in the 2-FCM classification (defuzzified using the maximum likelihood rule)

Table 2. Cluster centroids for the 3 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Deep-moderate	0.35	0.493
2	Moderate-fast	0.20	0.813
3	Shallow-slow	0.12	0.288

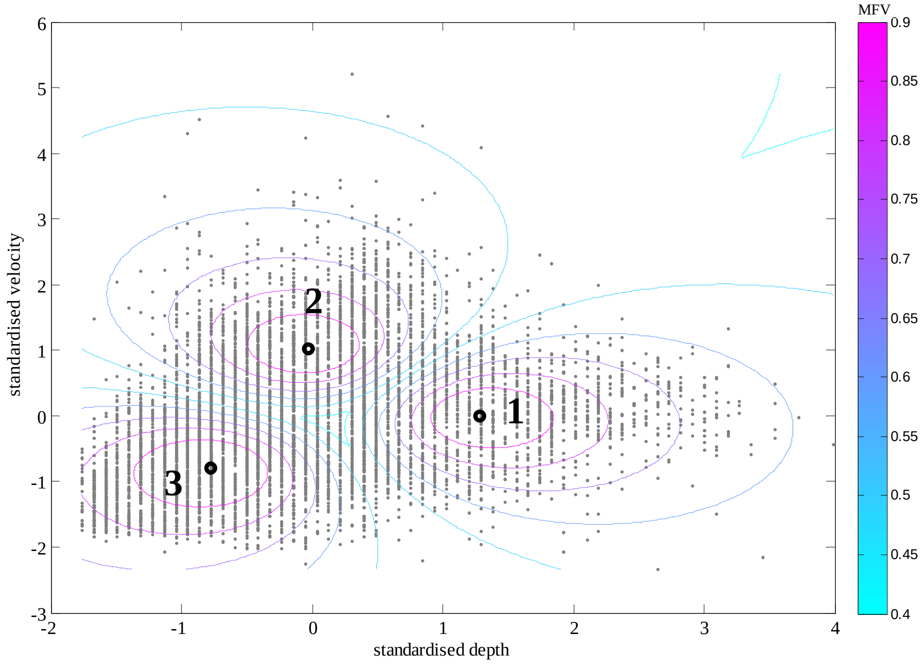
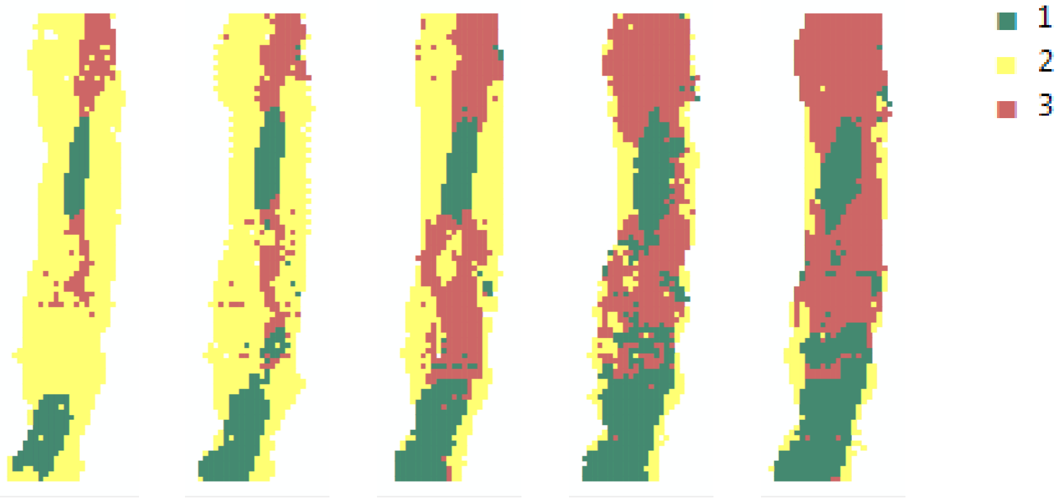


Figure 3. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 3-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 4.



$0.53m^3s^{-1}$, Q89 $0.79m^3s^{-1}$, Q67 $1.14m^3s^{-1}$, Q38 $1.63m^3s^{-1}$, Q21 $1.84m^3s^{-1}$, Q16

Figure 4. Location and extent of clusters in the 3-FCM classification (defuzzified using the maximum likelihood rule)

Table 3. Cluster centroids for the 4 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Deep-moderate	0.38	0.506
2	Moderate-fast	0.22	0.904
3	Moderate-moderate	0.18	0.477
4	Shallow-slow	0.10	0.202

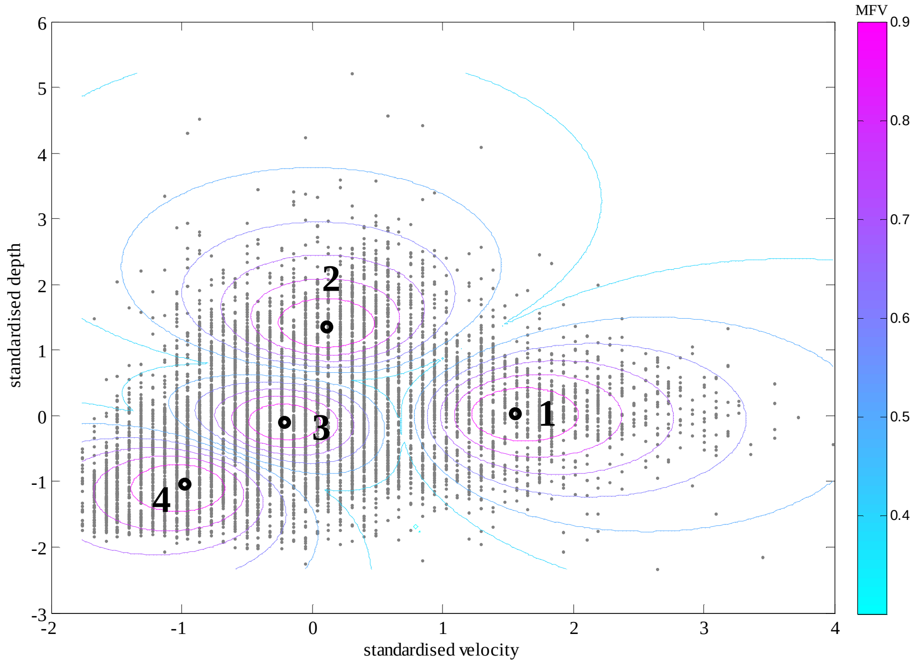
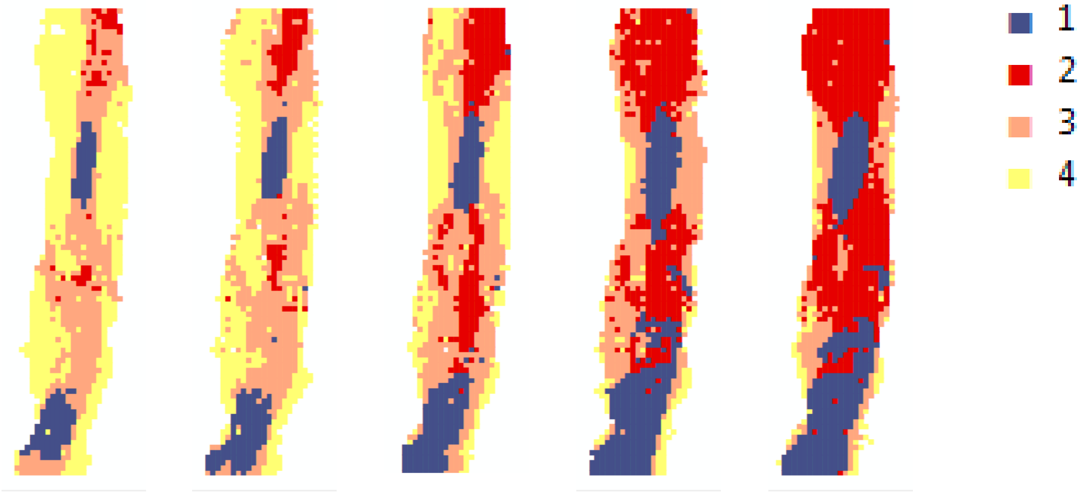


Figure 5. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 4-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 6.



$0.53m^3s^{-1}$, Q89 $0.79m^3s^{-1}$, Q67 $1.14m^3s^{-1}$, Q38 $1.63m^3s^{-1}$, Q21 $1.84m^3s^{-1}$, Q16

Figure 6. Location and extent of clusters in the 4-FCM classification (defuzzified using the maximum likelihood rule)

Table 4. Cluster centroids for the 5 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Shallow-moderate	0.15	0.592
2	Shallow-slow	0.09	0.203
3	Moderate-slow	0.23	0.337
4	Moderate-fast	0.23	0.936
5	Deep-slow	0.40	0.542

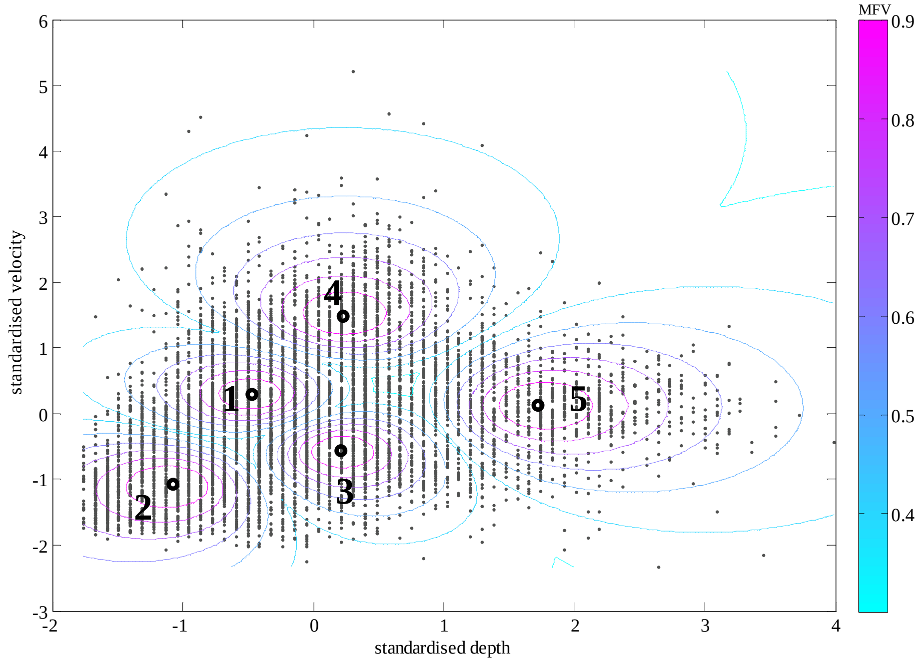
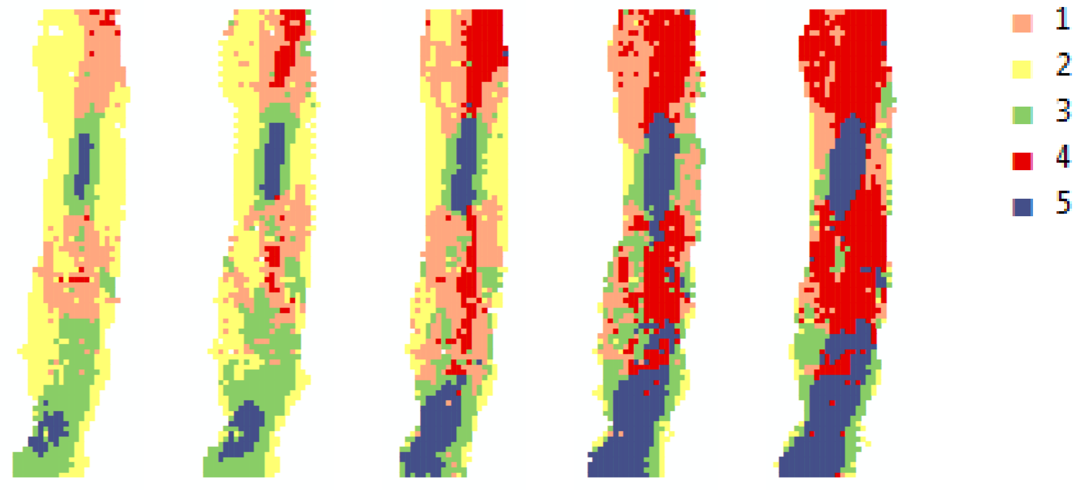


Figure 7. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 5-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 8.



$0.53m^3s^{-1}$, Q89 $0.79m^3s^{-1}$, Q67 $1.14m^3s^{-1}$, Q38 $1.63m^3s^{-1}$, Q21 $1.84m^3s^{-1}$, Q16

Figure 8. Location and extent of clusters in the 5-FCM classification (defuzzified using the maximum likelihood rule)

Table 5. Cluster centroids for the 6 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Deep-moderate	0.42	0.510
2	Shallow-slow	0.08	0.179
3	Moderate-fast	0.21	0.983
4	Moderate-slow	0.22	0.278
5	Moderate-moderate	0.26	0.643
6	Shallow-moderate	0.13	0.545

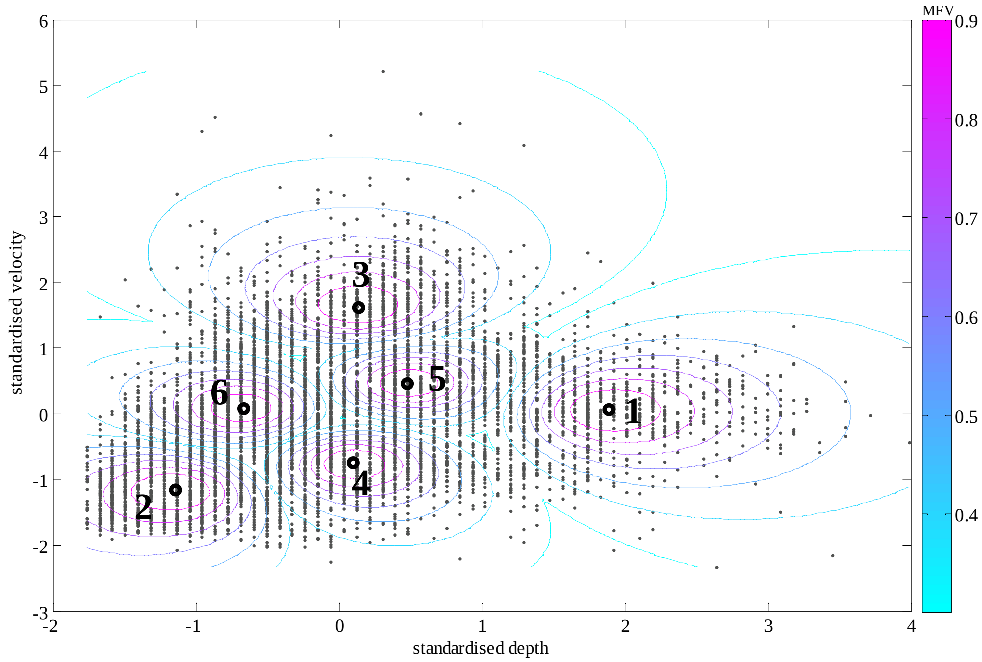
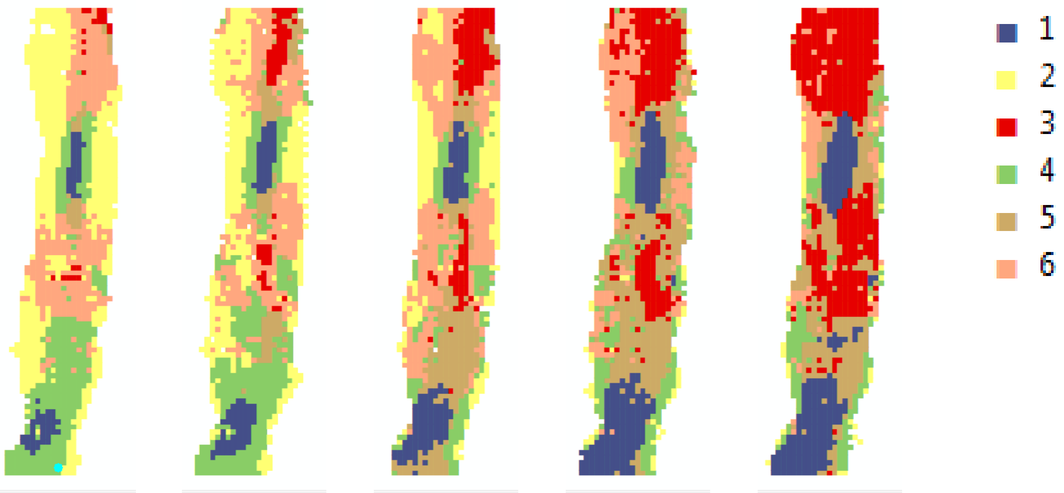


Figure 9. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 6-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 10.



0.53m^3s^{-1}, Q89 0.79m^3s^{-1}, Q67 1.14m^3s^{-1}, Q38 1.63m^3s^{-1}, Q21 1.84m^3s^{-1}, Q16

Figure 10. Location and extent of clusters in the 6-FCM classification (defuzzified using the maximum likelihood rule).

Table 6. Cluster centroids for the 7 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Shallow-slow	0.08	0.146
2	Shallow-moderate	0.12	0.427
3	Moderate-slow	0.24	0.271
4	Shallow-fast	0.16	0.731
5	Deep-moderate	0.43	0.511
6	Moderate-moderate	0.28	0.628
7	Moderate-fast	0.24	1.028

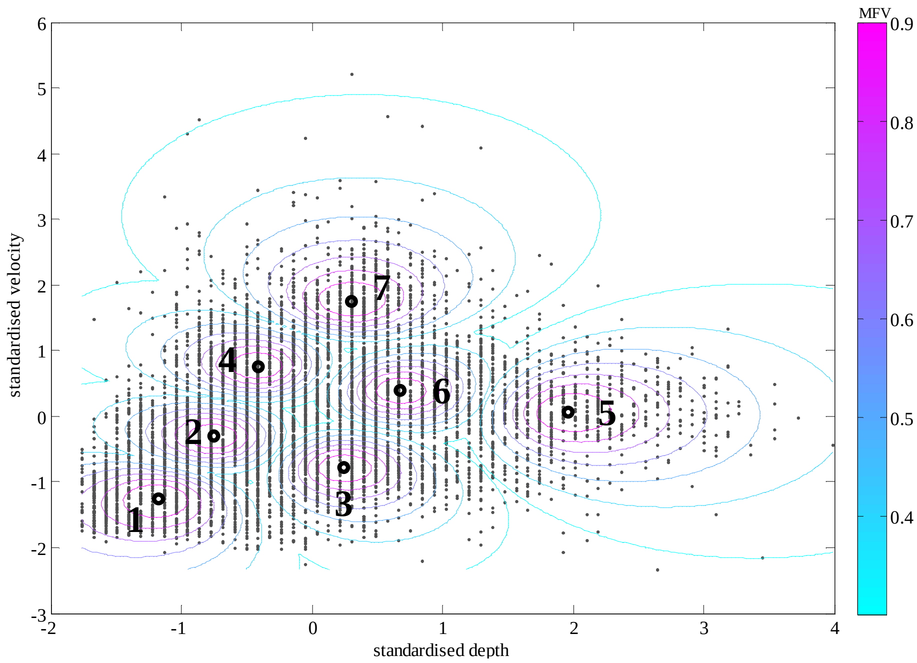
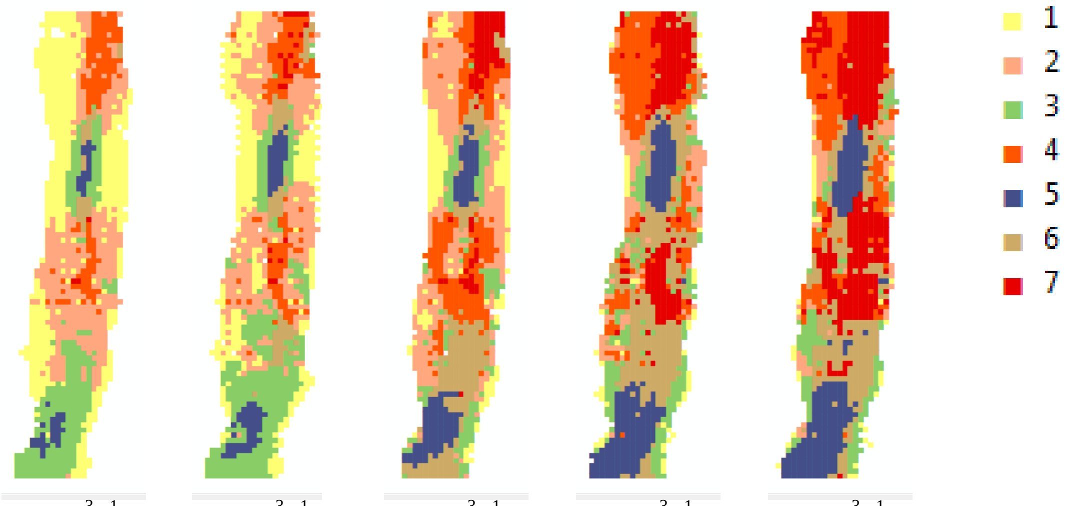


Figure 11. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 7-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 12.



$0.53m^3s^{-1}$, Q89 $0.79m^3s^{-1}$, Q67 $1.14m^3s^{-1}$, Q38 $1.63m^3s^{-1}$, Q21 $1.84m^3s^{-1}$, Q16

Figure 12. Location and extent of clusters in the 7-FCM classification (defuzzified using the maximum likelihood rule).

Table 7. Cluster centroids for the 8 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Deep-moderate	0.43	0.542
2	V shallow-v slow	0.07	0.143
3	Shallow-fast	0.16	0.758
4	Moderate-fast	0.24	1.044
5	Moderate-slow	0.30	0.294
6	Moderate-moderate	0.28	0.649
7	Shallow-slow	0.18	0.299
8	Shallow-moderate	0.11	0.466

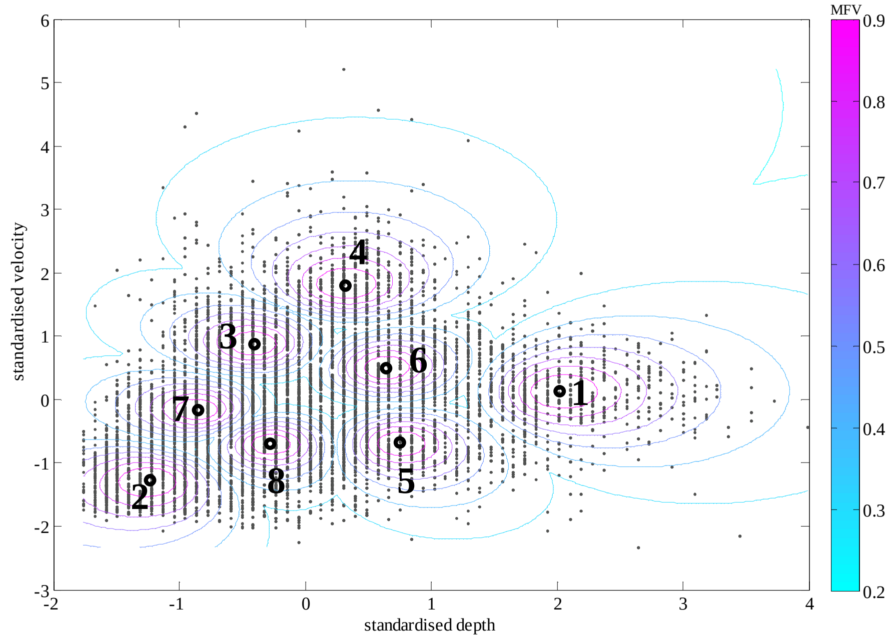
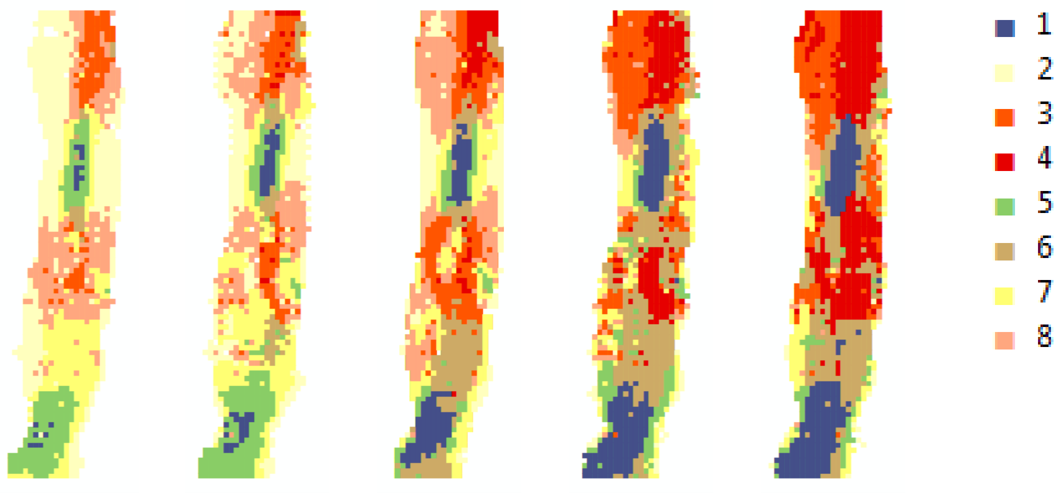


Figure 13. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 8-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 14.



$0.53m^3s^{-1}$, Q89 $0.79m^3s^{-1}$, Q67 $1.14m^3s^{-1}$, Q38 $1.63m^3s^{-1}$, Q21 $1.84m^3s^{-1}$, Q16

Figure 14. Location and extent of clusters in the 8-FCM classification (defuzzified using the maximum likelihood rule).

2. Gustafson-Kessel classifications

Table 8. Cluster centroids for the 2 Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Deep-mod/fast	0.29	0.711
2	Shallow-slow/mod	0.14	0.336

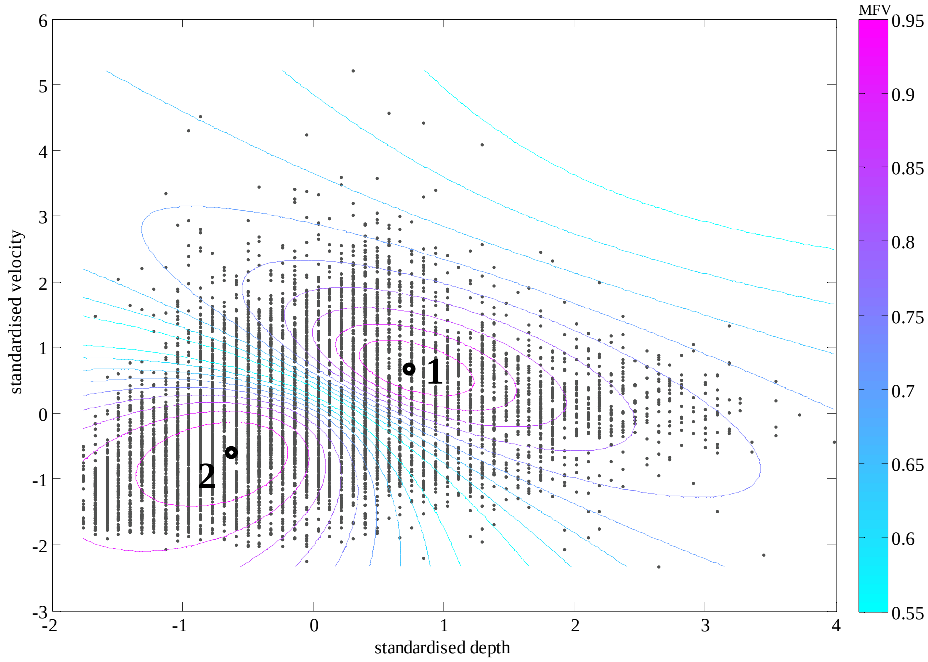
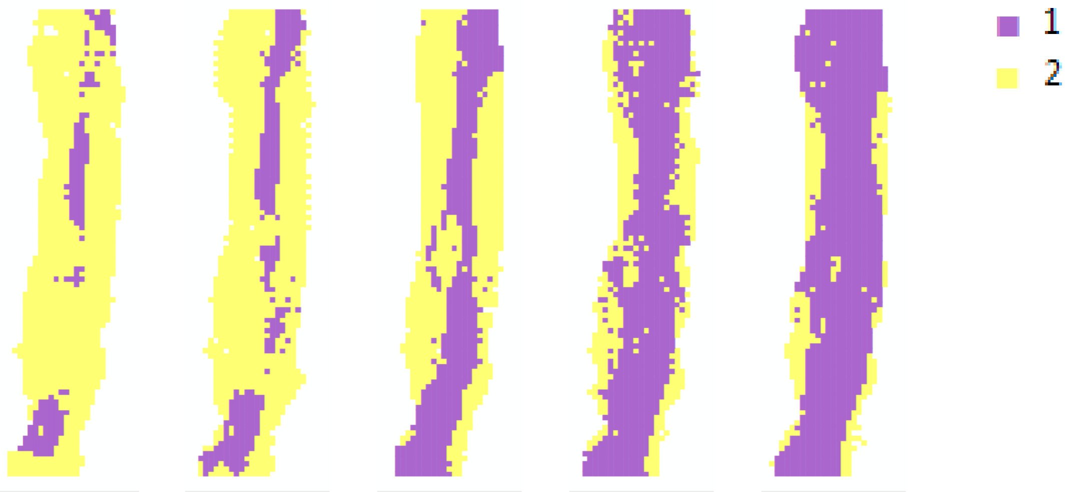


Figure 15. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 2-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 16.



$0.53 m^3 s^{-1}$, Q89 $0.79 m^3 s^{-1}$, Q67 $1.14 m^3 s^{-1}$, Q38 $1.63 m^3 s^{-1}$, Q21 $1.84 m^3 s^{-1}$, Q16

Figure 16. Location and extent of clusters in the 2-GK classification (defuzzified using the maximum likelihood rule).

Table 9. Cluster centroids for the 3 Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Deep-moderate	0.35	0.485
2	Shallow-slow	0.12	0.285
3	Moderate-fast	0.20	0.813

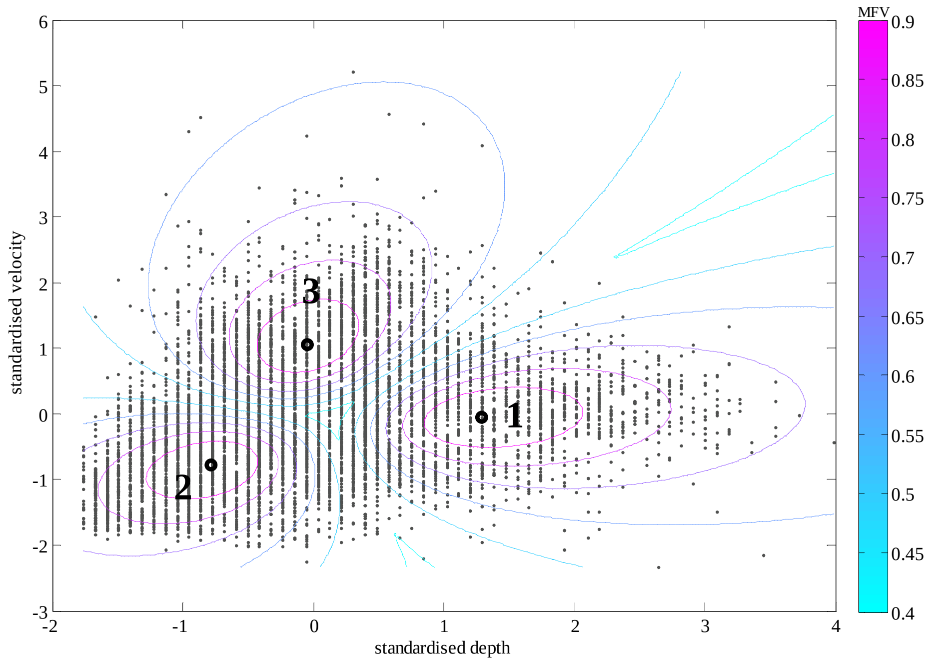
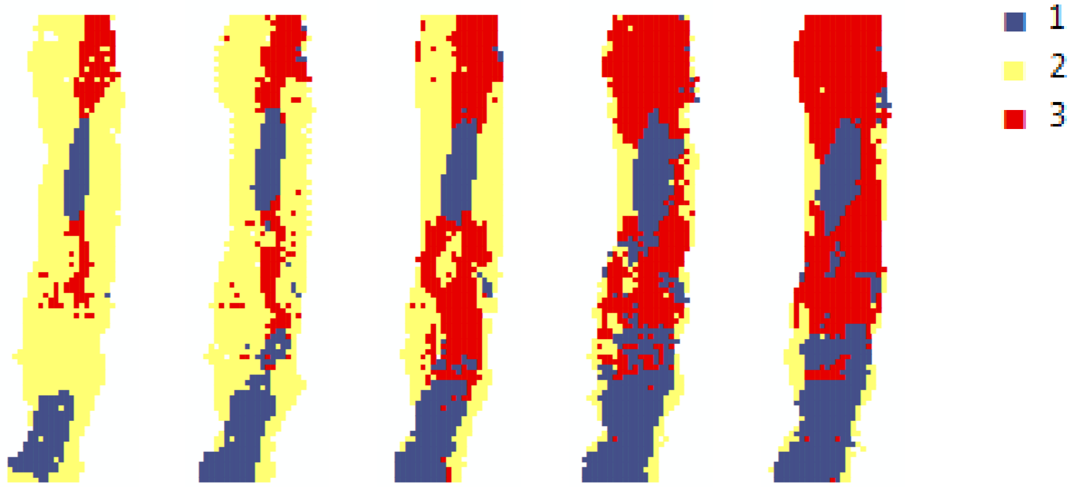


Figure 17. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 3-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 18.



$0.53m^3s^{-1}$, Q89 $0.79m^3s^{-1}$, Q67 $1.14m^3s^{-1}$, Q38 $1.63m^3s^{-1}$, Q21 $1.84m^3s^{-1}$, Q16

Figure 18. Location and extent of clusters in the 3-GK classification (defuzzified using the maximum likelihood rule).

Table 10. Cluster centroids for the 4 Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Deep-moderate	0.39	0.517
2	Moderate-fast	0.23	0.858
3	Moderate-slow	0.20	0.306
4	Shallow-slow	0.09	0.389

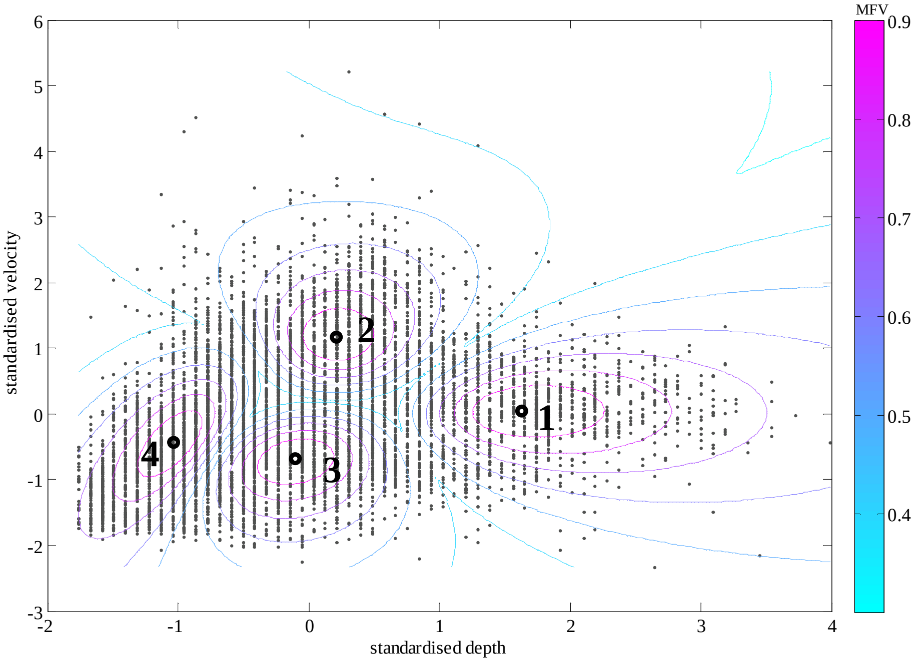
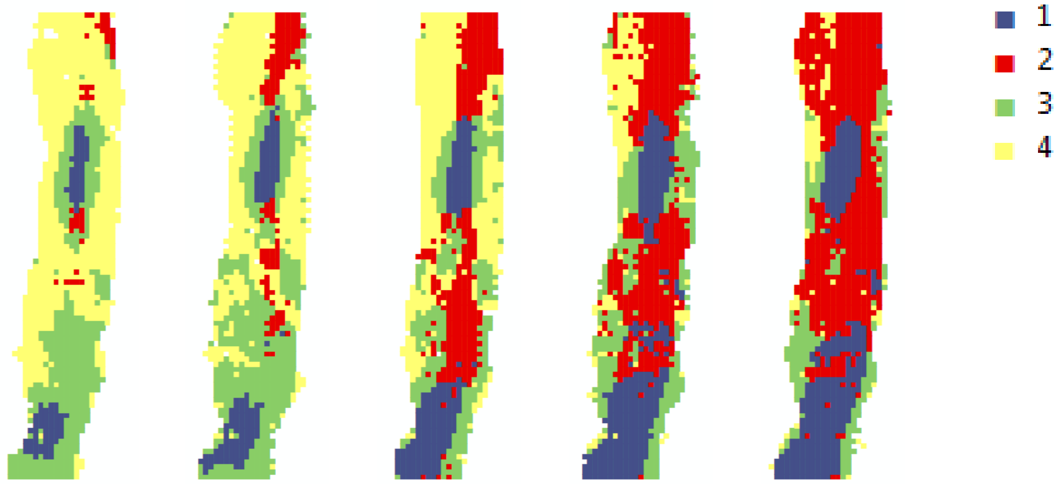


Figure 19. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 4-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 20.



$0.53m^3s^{-1}$, Q89 $0.79m^3s^{-1}$, Q67 $1.14m^3s^{-1}$, Q38 $1.63m^3s^{-1}$, Q21 $1.84m^3s^{-1}$, Q16

Figure 20. Location and extent of clusters in the 4-GK classification (defuzzified using the maximum likelihood rule).

Table 11. Cluster centroids for the 5 Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderate-slow	0.22	0.319
2	Shallow-fast	0.14	0.672
3	Shallow-slow	0.08	0.226
4	Moderate-fast	0.26	0.863
5	Deep-moderate	0.40	0.534

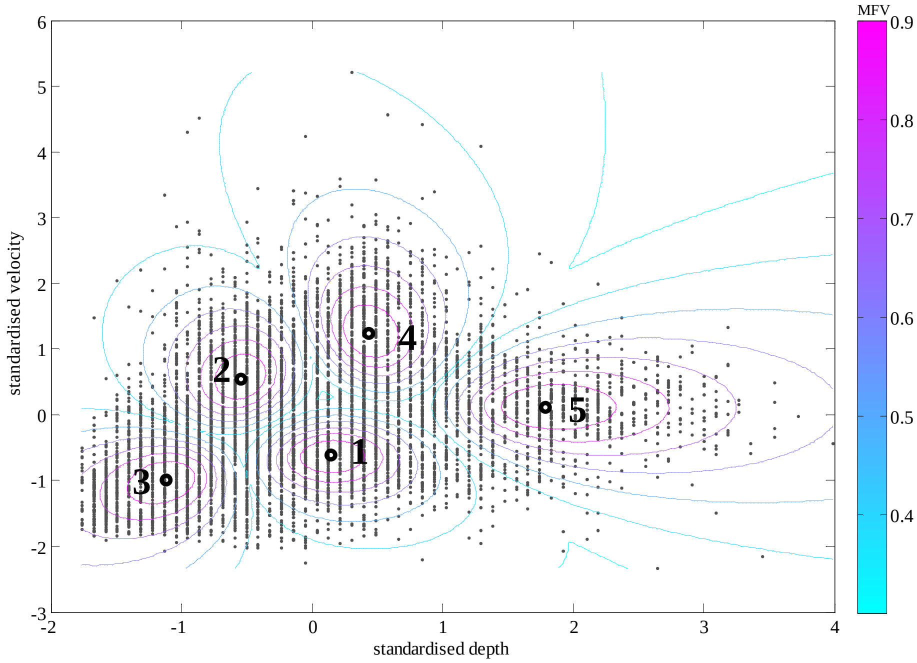
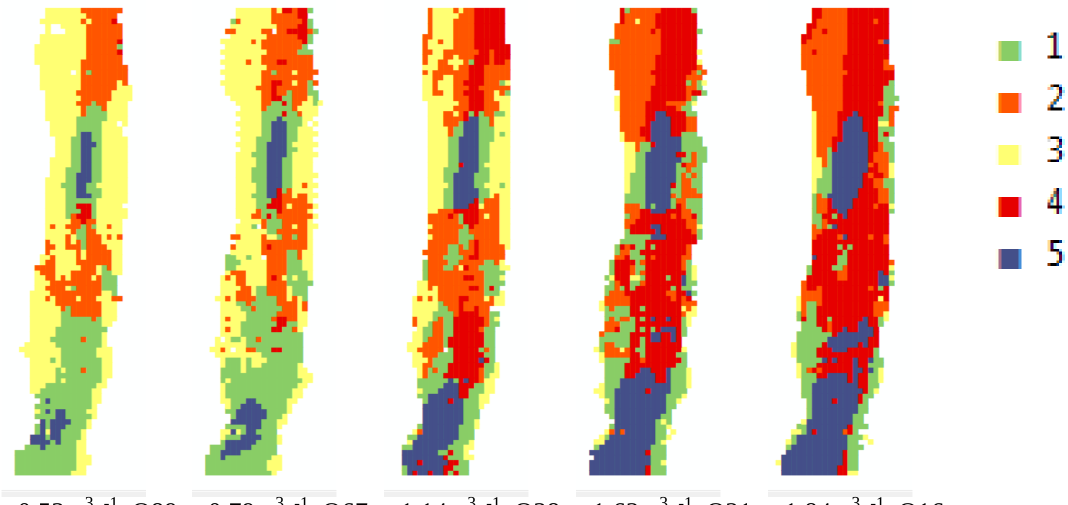


Figure 21. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 5-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 22.



0.53m^3s^{-1}, Q89 0.79m^3s^{-1}, Q67 1.14m^3s^{-1}, Q38 1.63m^3s^{-1}, Q21 1.84m^3s^{-1}, Q16

Figure 22. Location and extent of clusters in the 5-GK classification (defuzzified using the maximum likelihood rule).

Table 12. Cluster centroids for the 6 Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderate-slow	0.24	0.204
2	V shallow-slow	0.08	0.257
3	Moderate-fast	0.25	0.937
4	Shallow-mod/fast	0.14	0.716
5	Deep-moderate	0.41	0.572
6	Moderate-moderate	0.21	0.473

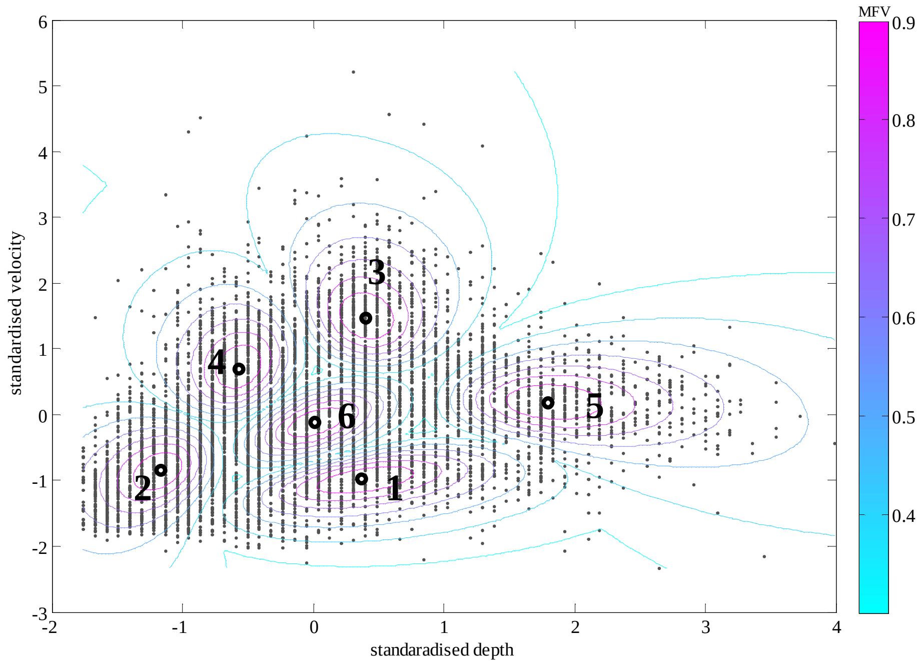
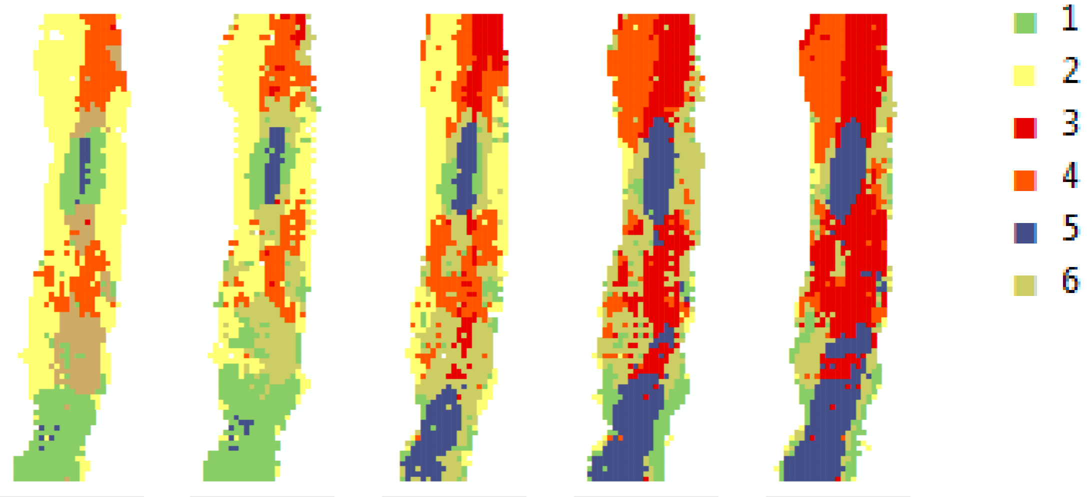


Figure 23. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 6-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 24.



$0.53m^3s^{-1}$, Q89 $0.79m^3s^{-1}$, Q67 $1.14m^3s^{-1}$, Q38 $1.63m^3s^{-1}$, Q21 $1.84m^3s^{-1}$, Q16

Figure 24. Location and extent of clusters in the 6-GK classification (defuzzified using the maximum likelihood rule).

Table 13. Cluster centroids for the 7 Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	V shallow-slow	0.09	0.133
2	Shallow-moderate	0.12	0.423
3	Moderate-slow	0.26	0.268
4	Shallow-fast	0.15	0.764
5	Deep-moderate	0.43	0.540
6	Moderate-moderate	0.27	0.615
7	Moderate-fast	0.25	0.996

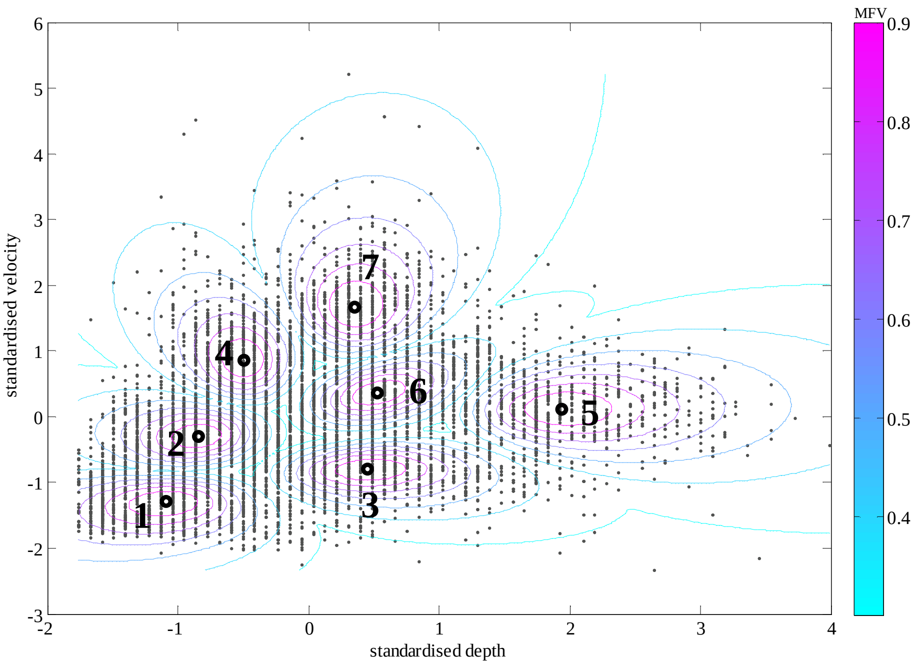
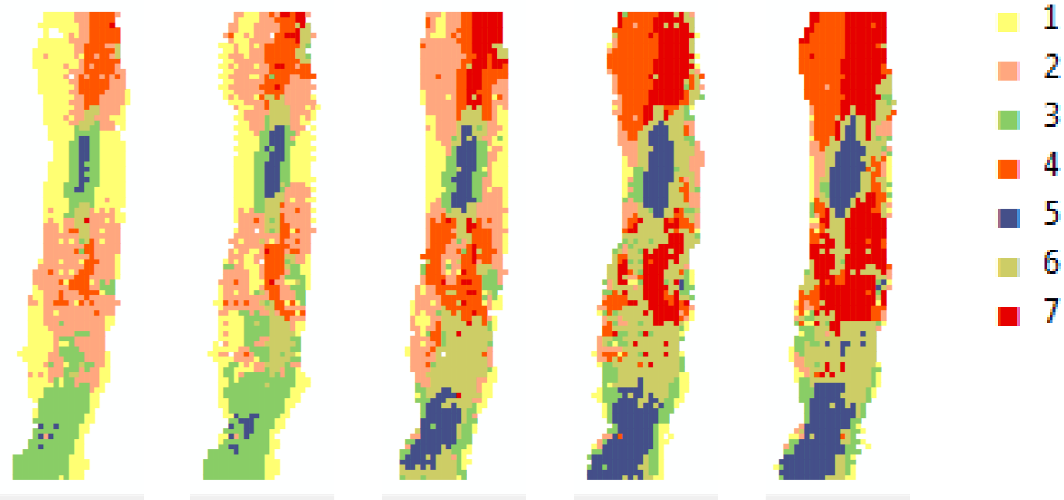


Figure 25. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 7-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 26.



$0.53m^3s^{-1}$, Q89 $0.79m^3s^{-1}$, Q67 $1.14m^3s^{-1}$, Q38 $1.63m^3s^{-1}$, Q21 $1.84m^3s^{-1}$, Q16

Figure 26. Location and extent of clusters in the 7-GK classification (defuzzified using the maximum likelihood rule).

Table 14. Cluster centroids for the 8 Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Deep-moderate	0.44	0.543
2	V shallow-moderate	0.09	0.435
3	Shallow-fast	0.15	0.789
4	Moderate-fast	0.25	0.999
5	Moderate-slow	0.29	0.252
6	Moderate-moderate	0.29	0.648
7	V shallow-slow	0.09	0.124
8	Shallow-moderate	0.18	0.413

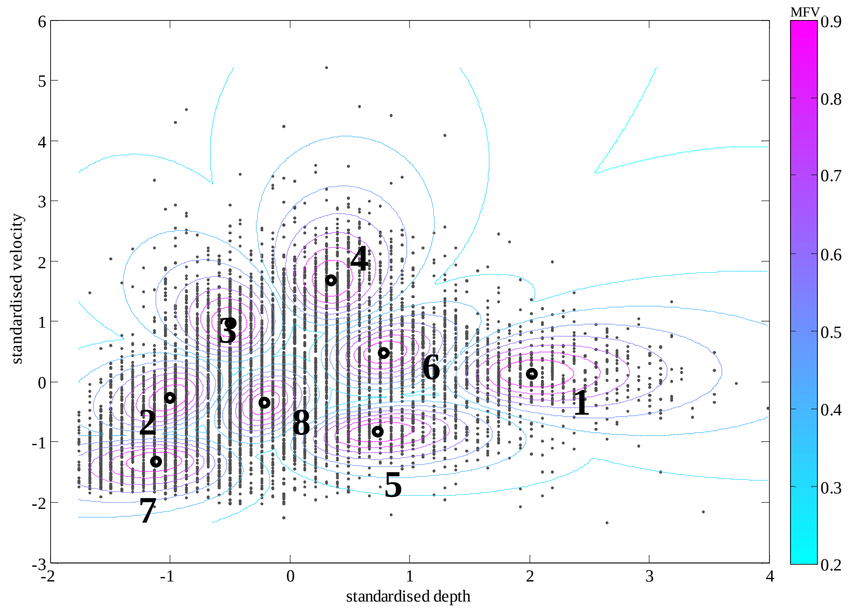
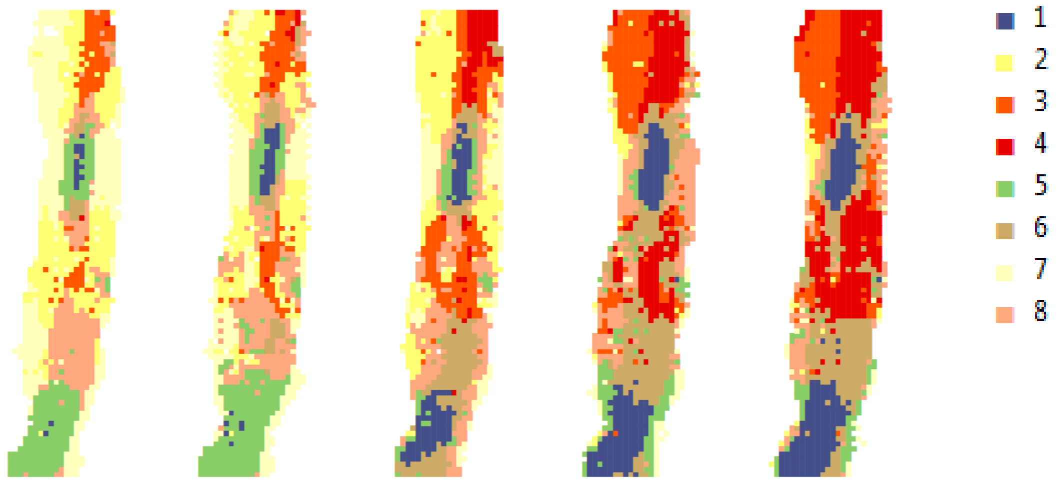


Figure 27. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 8-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 28.



$0.53m^3s^{-1}$, Q89 $0.79m^3s^{-1}$, Q67 $1.14m^3s^{-1}$, Q38 $1.63m^3s^{-1}$, Q21 $1.84m^3s^{-1}$, Q16

Figure 28. Location and extent of clusters in the 8-GK classification (defuzzified using the maximum likelihood rule).

Appendix D

Leigh Brook hydraulic patch classifications

1. Fuzzy *c*-means classifications

Table 1. Cluster centroids for the 2 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderate-moderate	0.28	0.625
2	Shallow-slow	0.13	0.198

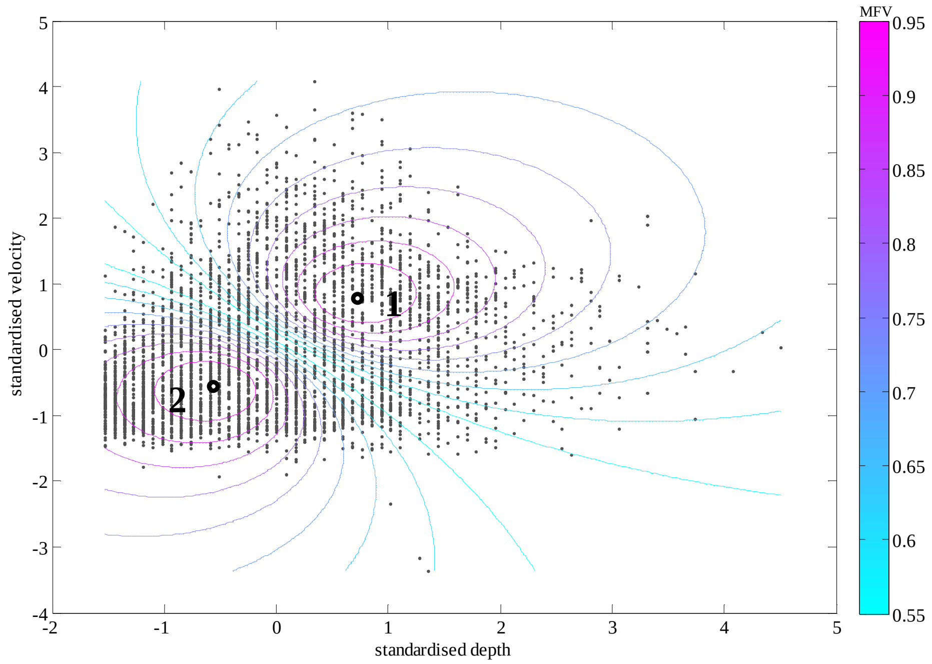
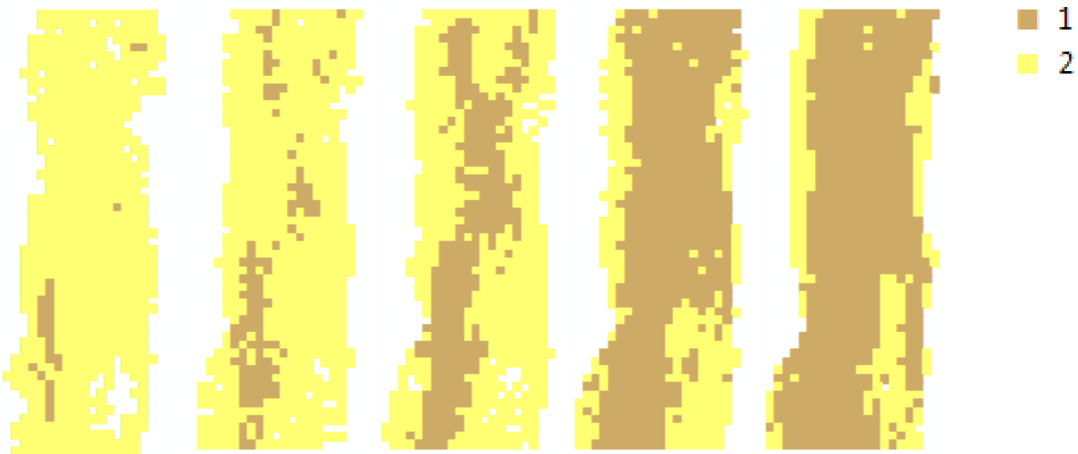


Figure 1. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 2-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 2.



$0.23m^3s^{-1}$, Q87 $0.37m^3s^{-1}$, Q65 $0.61m^3s^{-1}$, Q54 $0.99m^3s^{-1}$, Q23 $1.3m^3s^{-1}$, Q12

Figure 2. Location and extent of clusters in the 2-FCM classification (defuzzified using the maximum likelihood rule)

Table 2. Cluster centroids for the 3 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderate-slow	0.30	0.301
2	Shallow-slow	0.09	0.210
3	Moderate-fast	0.23	0.777

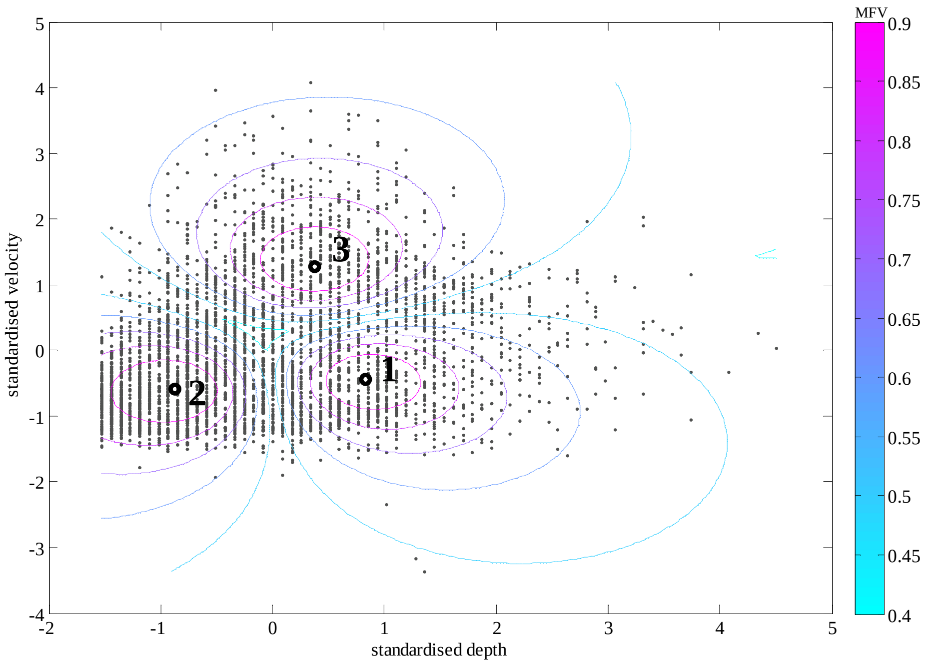
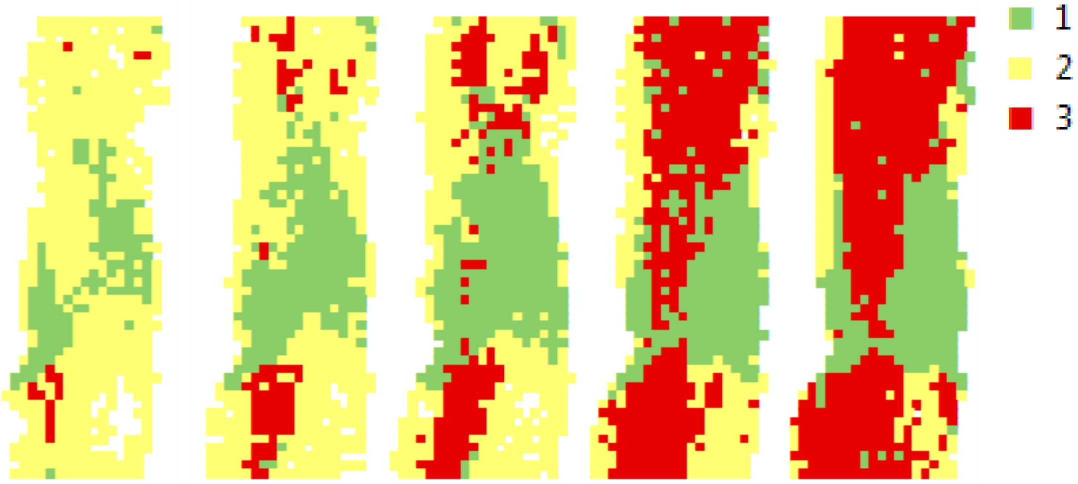


Figure 3. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 3-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 4.



$0.23m^3s^{-1}$, Q87 $0.37m^3s^{-1}$, Q65 $0.61m^3s^{-1}$, Q54 $0.99m^3s^{-1}$, Q23 $1.3m^3s^{-1}$, Q12

Figure 4. Location and extent of clusters in the 3-FCM classification (defuzzified using the maximum likelihood rule)

Table 3. Cluster centroids for the 4 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderate-fast	0.29	0.802
2	Shallow-moderate	0.14	0.512
3	Moderate-slow	0.30	0.179
4	Shallow-slow	0.08	0.130

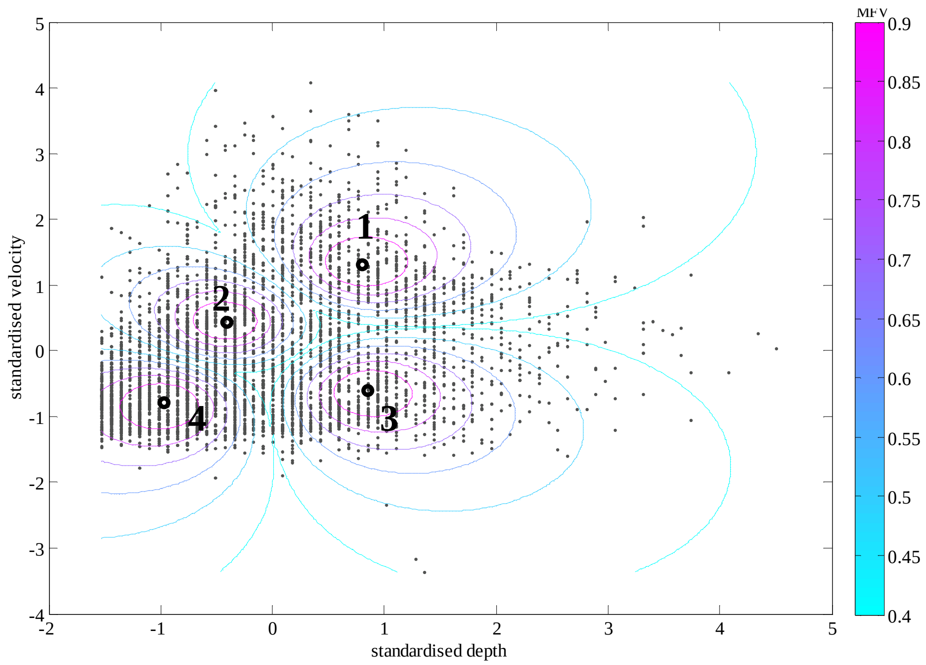
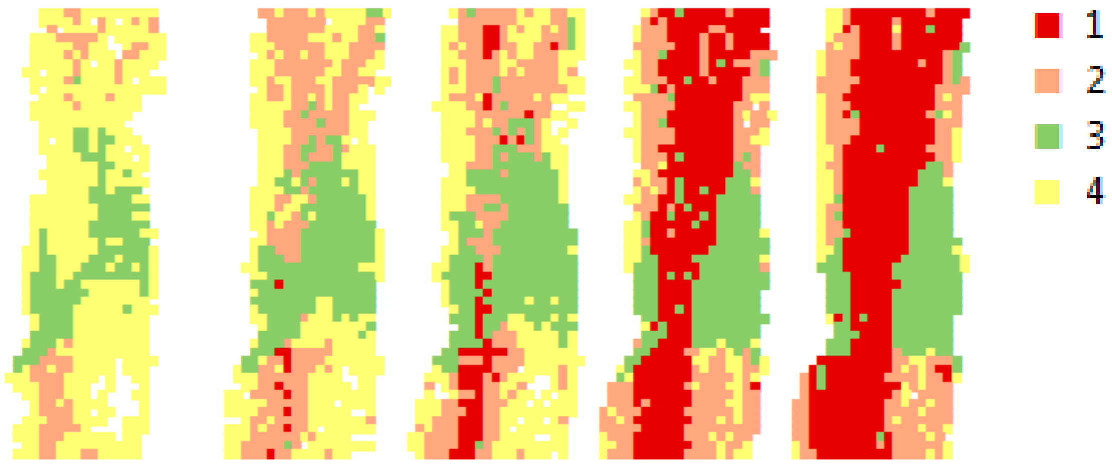


Figure 5. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 4-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 6.



$0.23m^3s^{-1}$, Q87 $0.37m^3s^{-1}$, Q65 $0.61m^3s^{-1}$, Q54 $0.99m^3s^{-1}$, Q23 $1.3m^3s^{-1}$, Q12

Figure 6. Location and extent of clusters in the 4-FCM classification (defuzzified using the maximum likelihood rule)

Table 4. Cluster centroids for the 5 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderately deep-moderate	0.37	0.534
2	Shallow-slow	0.07	0.115
3	Shallow-moderate	0.13	0.462
4	Moderate-fast	0.22	0.910
5	Moderate-slow	0.26	0.132

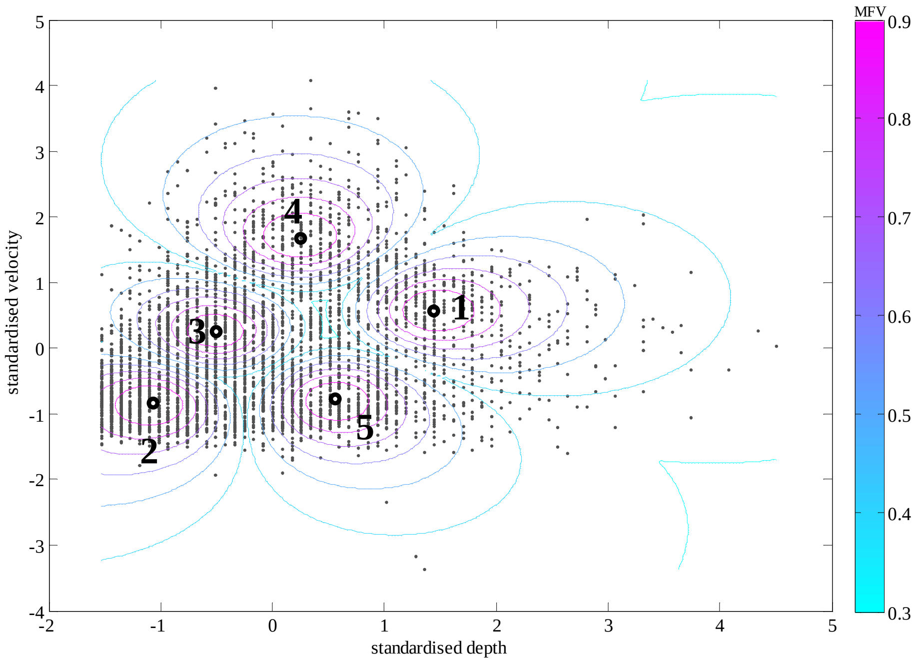
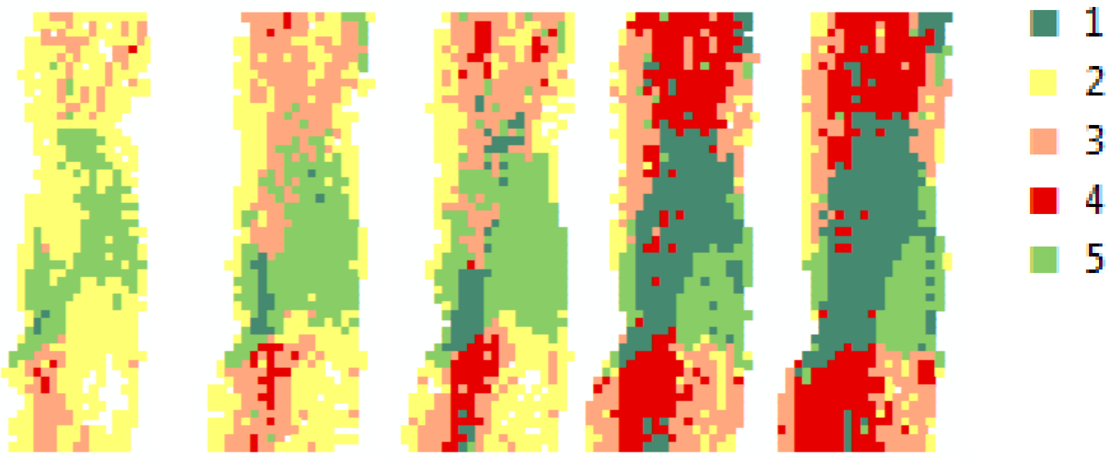


Figure 7. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 5-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 8.



0.23m^3s^{-1}, Q87 0.37m^3s^{-1}, Q65 0.61m^3s^{-1}, Q54 0.99m^3s^{-1}, Q23 1.3m^3s^{-1}, Q12

Figure 8. Location and extent of clusters in the 5-FCM classification (defuzzified using the maximum likelihood rule)

Table 5. Cluster centroids for the 6 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderately deep-slow	0.32	0.167
2	Shallow-slow	0.05	0.128
3	Moderate-fast	0.22	0.937
4	Moderate-slow	0.18	0.134
5	Moderately deep--moderate	0.36	0.597
6	Shallow-moderate	0.13	0.498

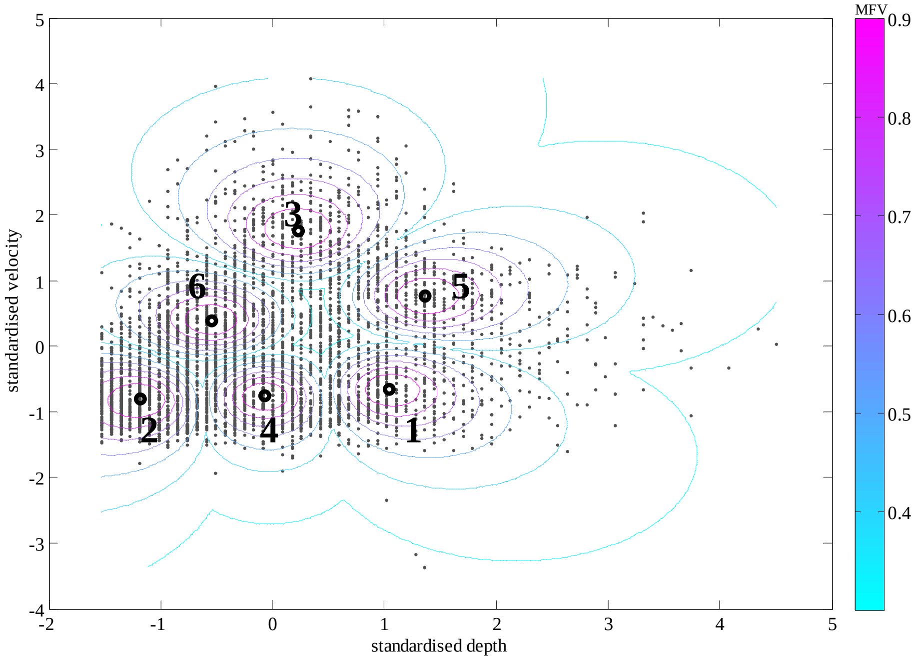
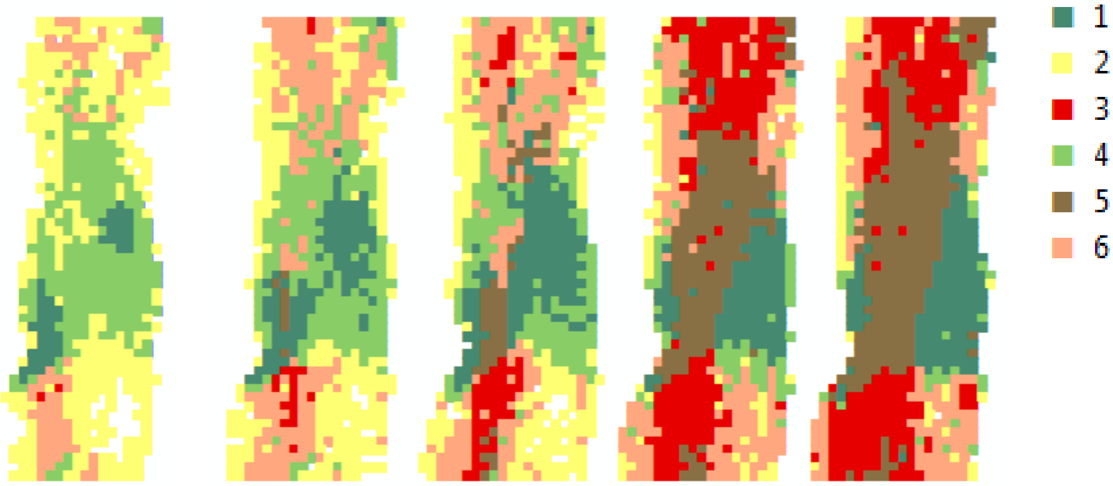


Figure 9. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 6-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 10.



$0.23m^3s^{-1}$, Q87 $0.37m^3s^{-1}$, Q65 $0.61m^3s^{-1}$, Q54 $0.99m^3s^{-1}$, Q23 $1.3m^3s^{-1}$, Q12

Figure 10. Location and extent of clusters in the 6-FCM classification (defuzzified using the maximum likelihood rule).

Table 6. Cluster centroids for the 7 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderate-slow	0.18	0.120
2	Shallow-slow	0.05	0.107
3	Moderately deep-slow	0.32	0.155
4	Shallow-moderate	0.10	0.435
5	Moderately deep-moderate	0.38	0.594
6	Moderate-moderate	0.21	0.587
7	Moderate-fast	0.22	0.997

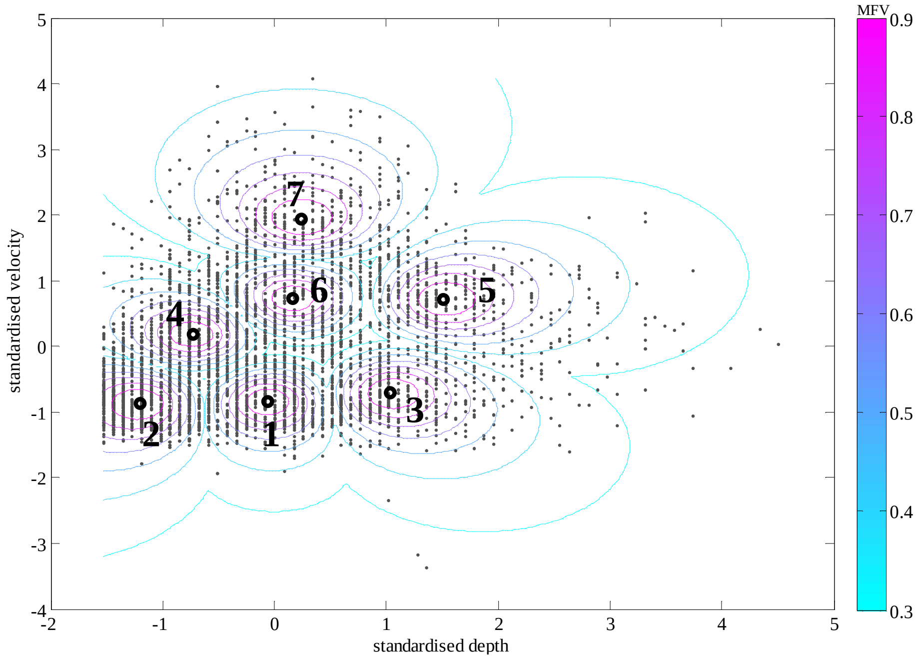


Figure 11. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 7-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 12.

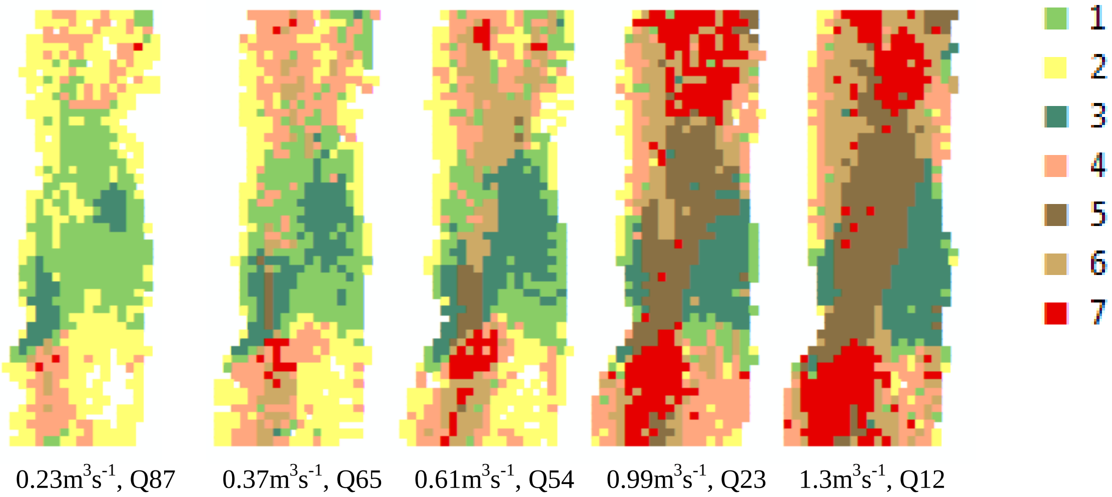


Figure 12. Location and extent of clusters in the 7-FCM classification (defuzzified using the maximum likelihood rule).

Table 7. Cluster centroids for the 8 fuzzy *c*-means classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderately deep-slow	0.33	0.131
2	Shallow-slow	0.05	0.097
3	Moderately shallow-moderate	0.15	0.670
4	Moderate-fast	0.23	0.998
5	Moderate-moderate	0.24	0.455
6	Moderately deep-moderate	0.39	0.599
7	Moderate-slow	0.18	0.098
8	Shallow-moderate	0.10	0.376

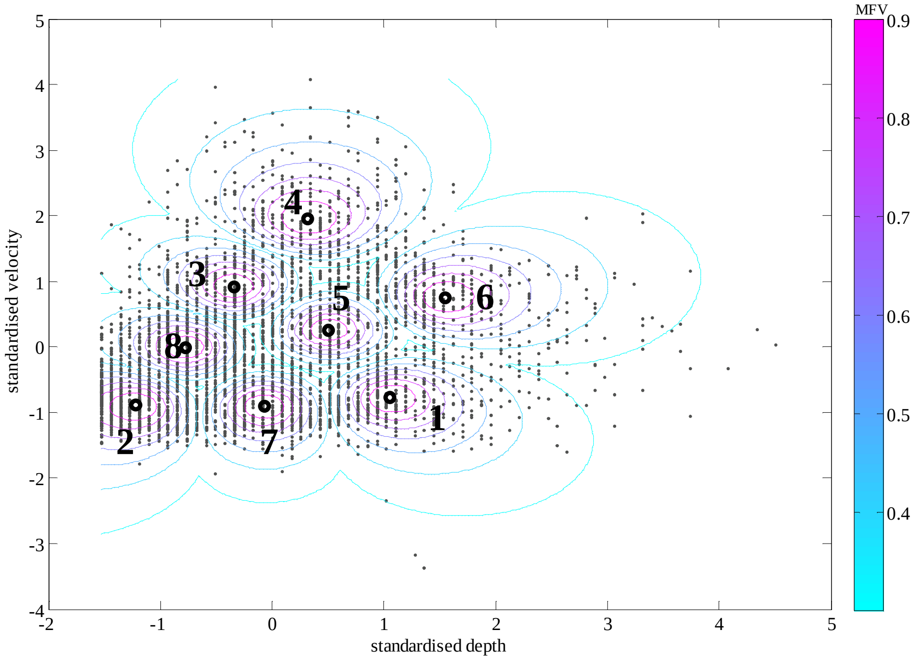
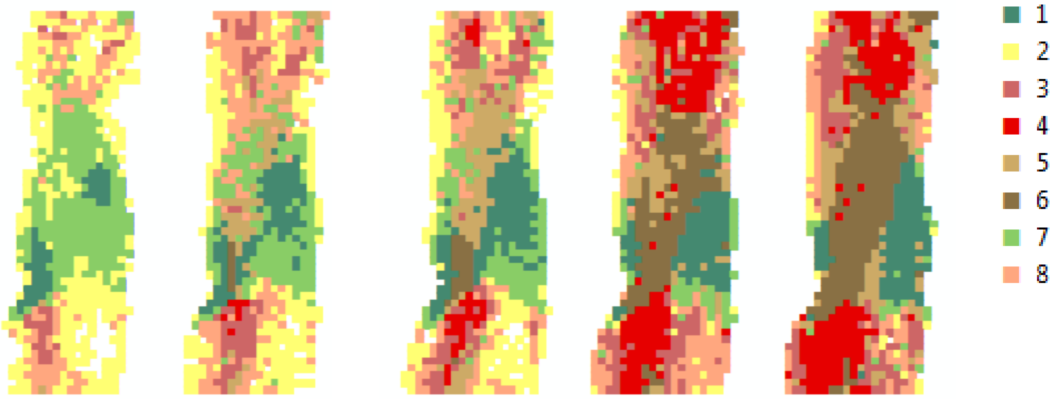


Figure 13. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 8-cluster fuzzy c-means classification. Cluster numbers correspond to the images in Figure 14.



$0.23m^3s^{-1}$, Q87 $0.37m^3s^{-1}$, Q65 $0.61m^3s^{-1}$, Q54 $0.99m^3s^{-1}$, Q23 $1.3m^3s^{-1}$, Q12

Figure 14. Location and extent of clusters in the 8-FCM classification (defuzzified using the maximum likelihood rule).

2. Gustafson-Kessel classifications

Table 8. Cluster centroids for the 2 Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderate-fast	0.26	0.681
2	Shallow-slow	0.15	0.180

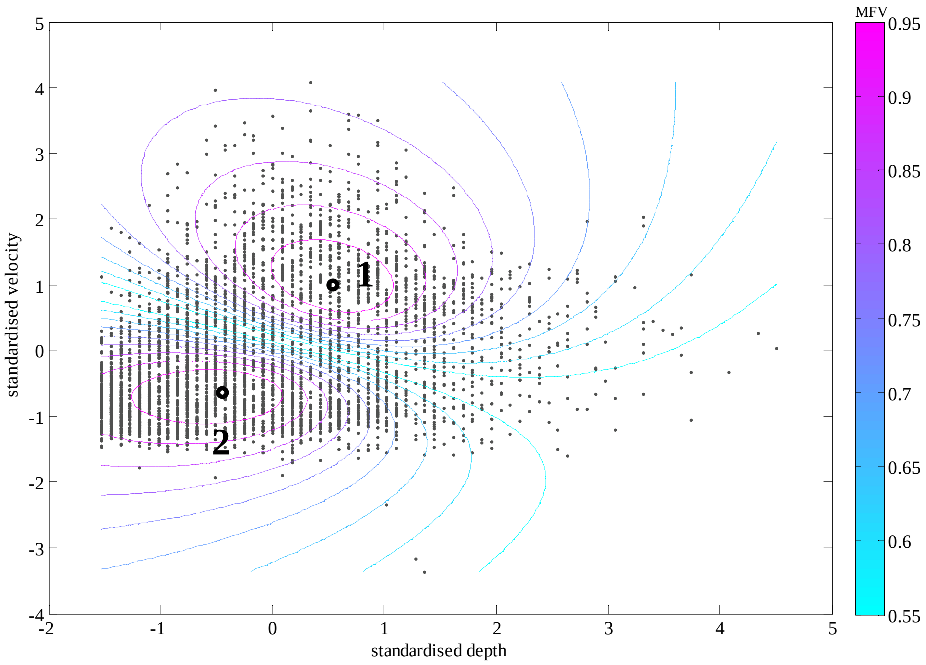
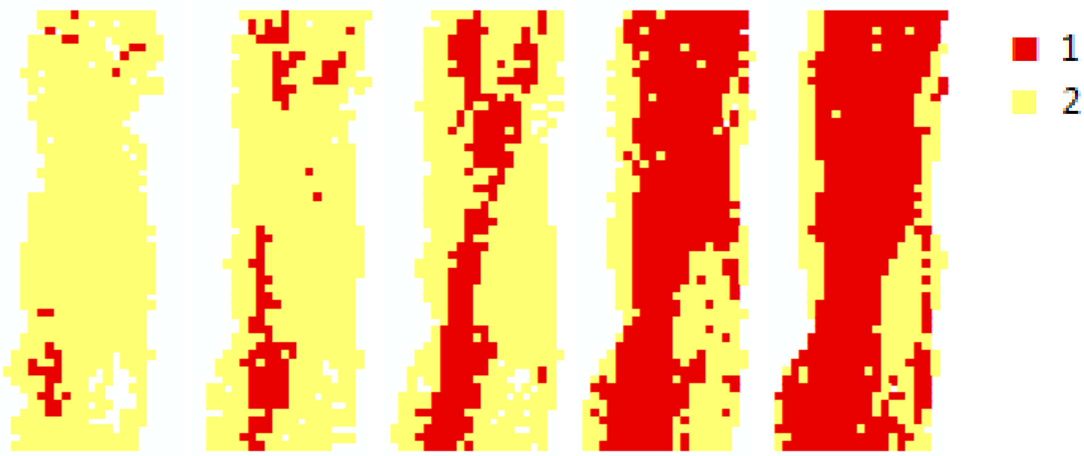


Figure 15. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 2-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 16.



$0.23m^3s^{-1}$, Q87 $0.37m^3s^{-1}$, Q65 $0.61m^3s^{-1}$, Q54 $0.99m^3s^{-1}$, Q23 $1.3m^3s^{-1}$, Q12

Figure 16. Location and extent of clusters in the 2-GK classification (defuzzified using the maximum likelihood rule).

Table 9. Cluster centroids for the 3 Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderate-slow	0.24	0.142
2	Shallow-moderate	0.09	0.323
3	Moderate-fast	0.29	0.745

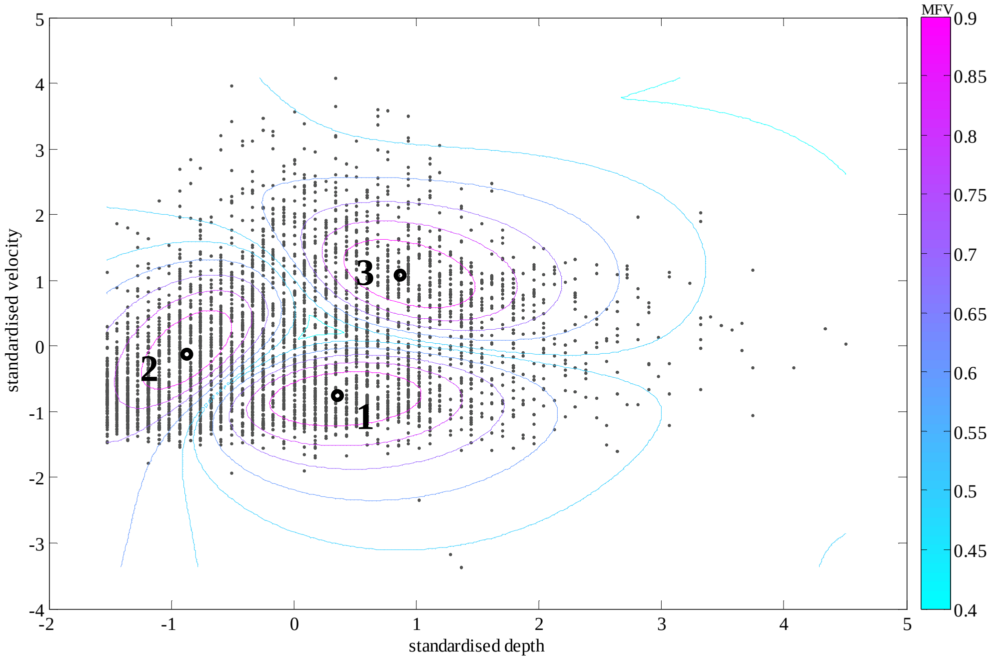
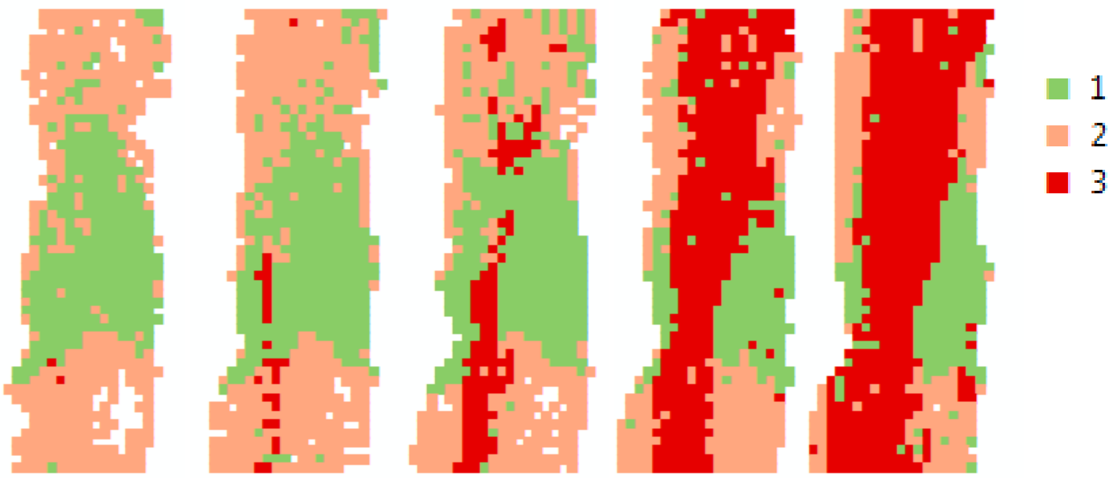


Figure 17. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 3-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 18.



$0.23m^3s^{-1}$, Q87 $0.37m^3s^{-1}$, Q65 $0.61m^3s^{-1}$, Q54 $0.99m^3s^{-1}$, Q23 $1.3m^3s^{-1}$, Q12

Figure 18. Location and extent of clusters in the 3-GK classification (defuzzified using the maximum likelihood rule).

Table 10. Cluster centroids for the 4 Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderately deep-fast	0.31	0.769
2	Shallow-moderate	0.16	0.553
3	Moderate-slow	0.26	0.132
4	Shallow-slow	0.07	0.169

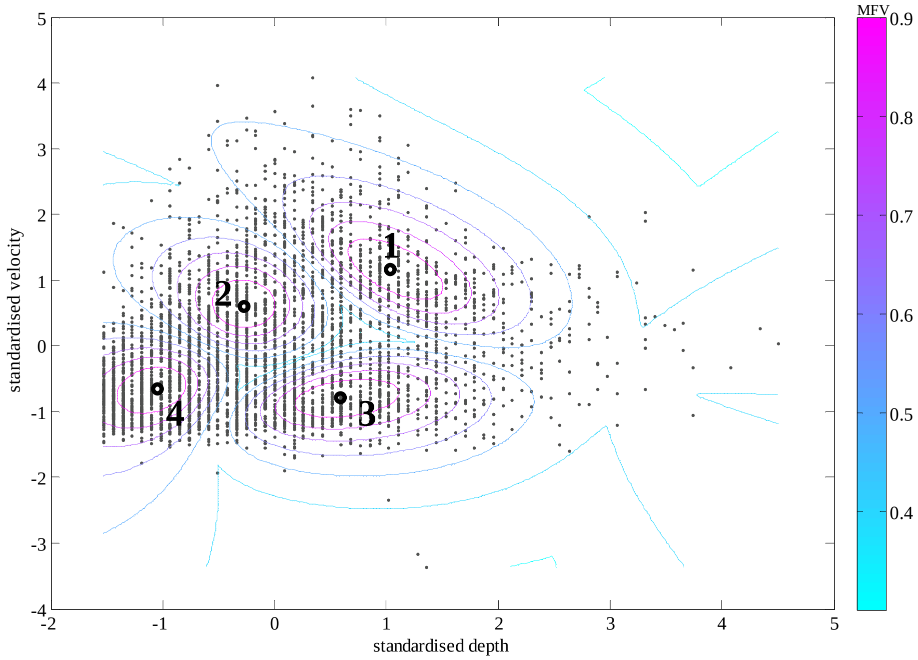
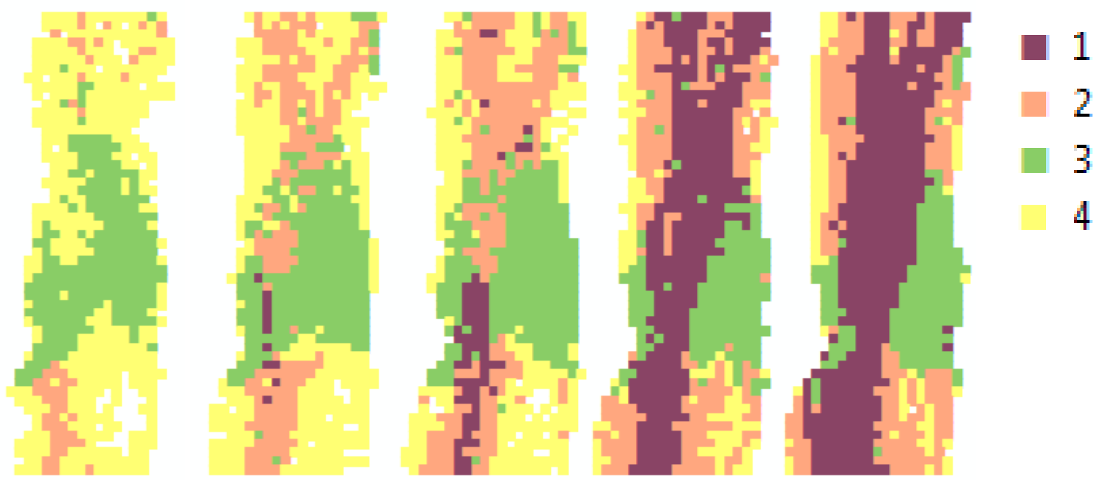


Figure 19. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 4-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 20.



$0.23m^3s^{-1}$, Q87 $0.37m^3s^{-1}$, Q65 $0.61m^3s^{-1}$, Q54 $0.99m^3s^{-1}$, Q23 $1.3m^3s^{-1}$, Q12

Figure 20. Location and extent of clusters in the 4-GK classification (defuzzified using the maximum likelihood rule).

Table 11. Cluster centroids for the 5 Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderate-moderate	0.35	0.584
2	Shallow-slow	0.06	0.135
3	Shallow-moderate	0.14	0.453
4	Moderate-fast	0.21	0.912
5	Moderate-slow	0.26	0.107

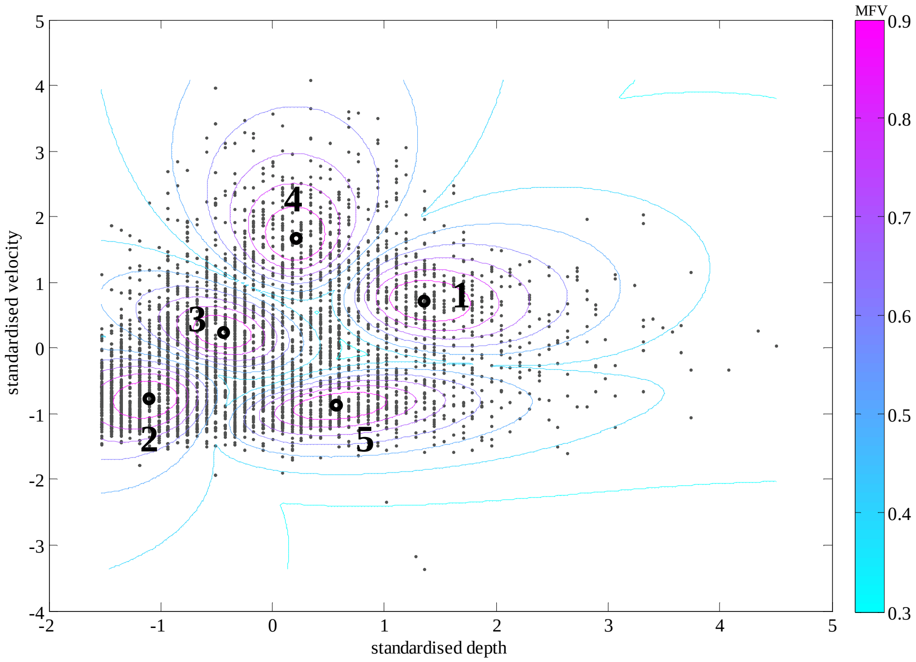


Figure 21. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 5-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 22.

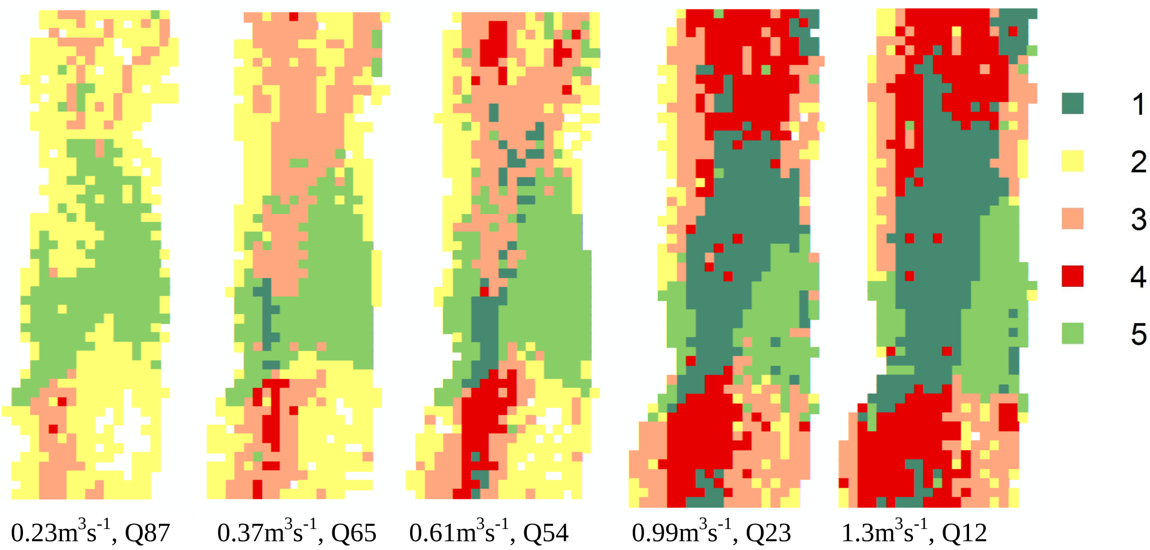


Figure 22. Location and extent of clusters in the 5-GK classification (defuzzified using the maximum likelihood rule).

Table 12. Cluster centroids for the 6 Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderate-slow	0.27	0.073
2	Shallow-slow	0.06	0.125
3	Moderate-fast	0.22	0.926
4	Shallow-moderate	0.11	0.531
5	Moderately deep-moderate	0.37	0.610
6	Moderate-moderate	0.21	0.312

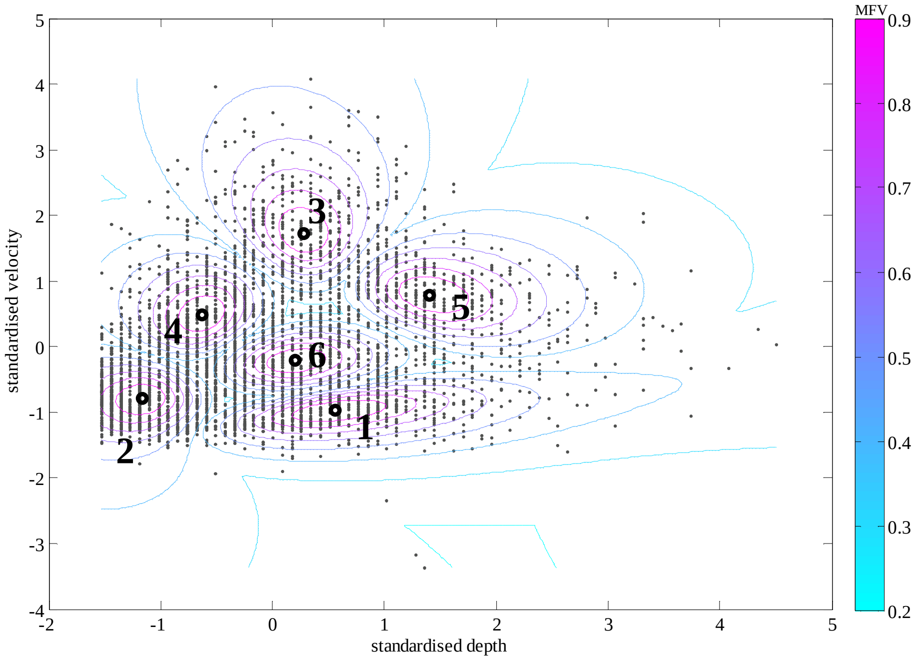
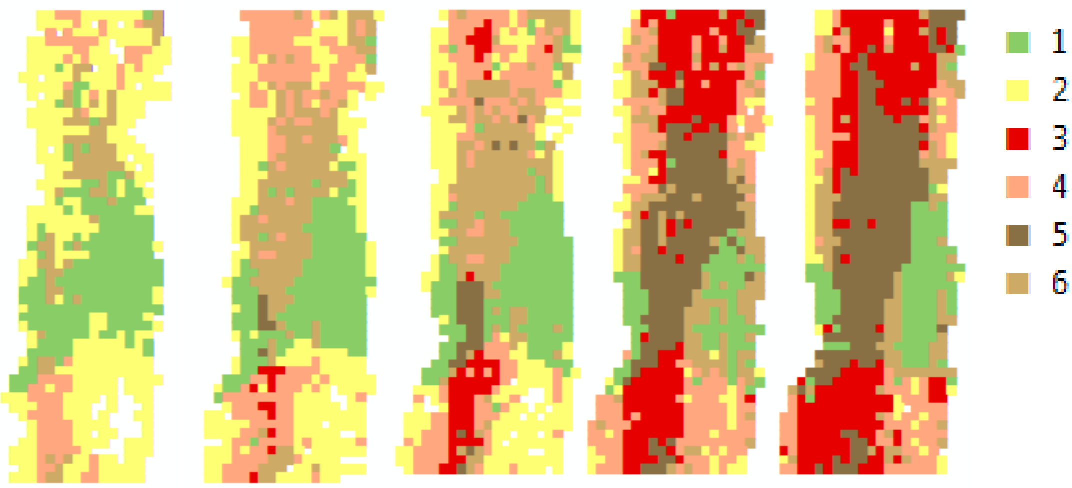


Figure 23. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 6-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 24.



$0.23m^3s^{-1}$, Q87 $0.37m^3s^{-1}$, Q65 $0.61m^3s^{-1}$, Q54 $0.99m^3s^{-1}$, Q23 $1.3m^3s^{-1}$, Q12

Figure 24. Location and extent of clusters in the 6-GK classification (defuzzified using the maximum likelihood rule).

Table 13. Cluster centroids for the 7 Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderate-slow	0.28	0.068
2	Shallow-slow	0.05	0.132
3	Moderately shallow-slow	0.16	0.204
4	Shallow-moderate	0.11	0.537
5	Moderately deep-moderate	0.37	0.628
6	Moderate-fast	0.22	0.957
7	Moderate-moderate	0.27	0.405

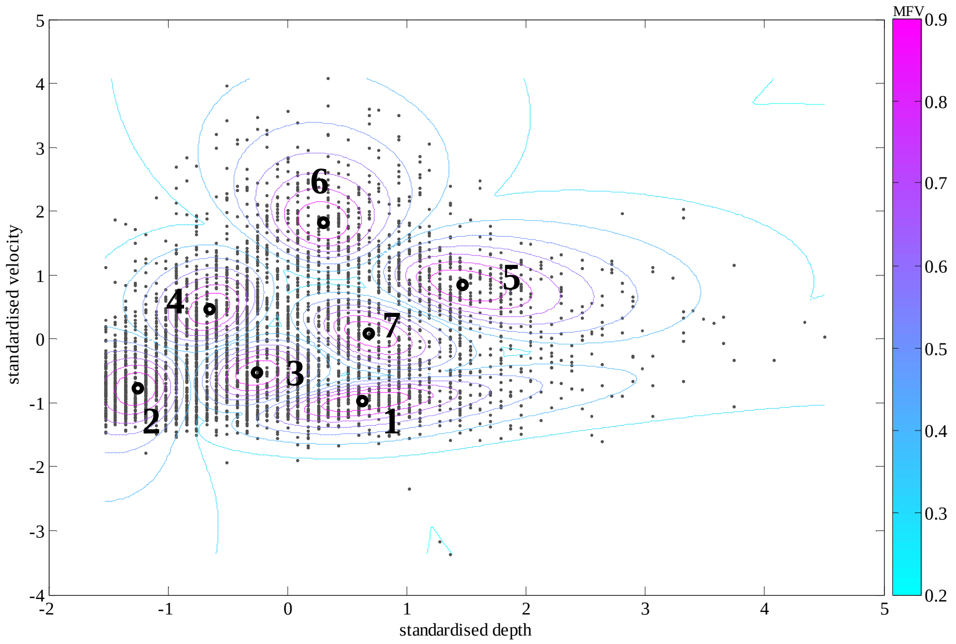
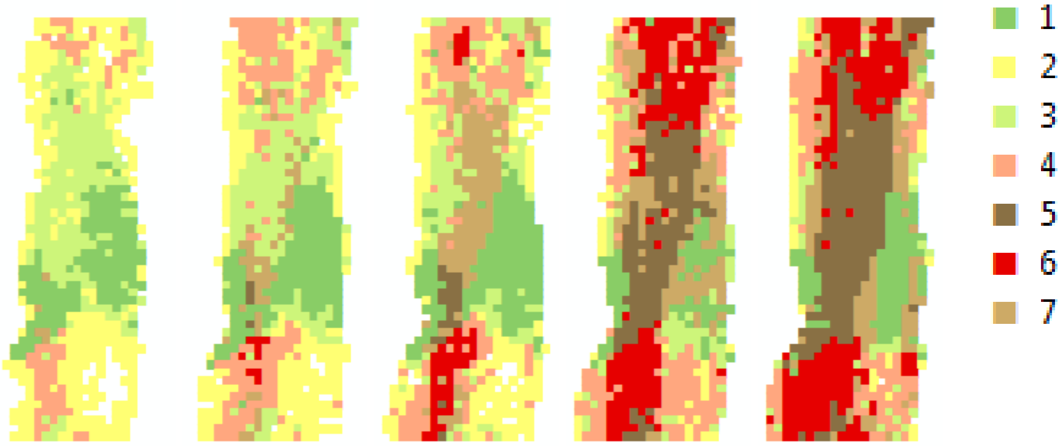


Figure 25. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 7-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 26.



$0.23m^3s^{-1}$, Q87 $0.37m^3s^{-1}$, Q65 $0.61m^3s^{-1}$, Q54 $0.99m^3s^{-1}$, Q23 $1.3m^3s^{-1}$, Q12

Figure 26. Location and extent of clusters in the 7-GK classification (defuzzified using the maximum likelihood rule).

Table 14. Cluster centroids for the 8 Gustafson-Kessel classification

Cluster	Hydraulic description	Mean depth (m)	Mean velocity (ms^{-1})
1	Moderate-slow	0.33	0.116
2	Shallow-slow	0.04	0.148
3	Moderate-fast	0.28	0.778
4	Moderately shallow-fast	0.18	0.909
5	Moderate-moderate	0.20	0.328
6	Moderately deep-moderate	0.40	0.586
7	Moderately shallow-slow	0.15	0.058
8	Shallow-moderate	0.11	0.520

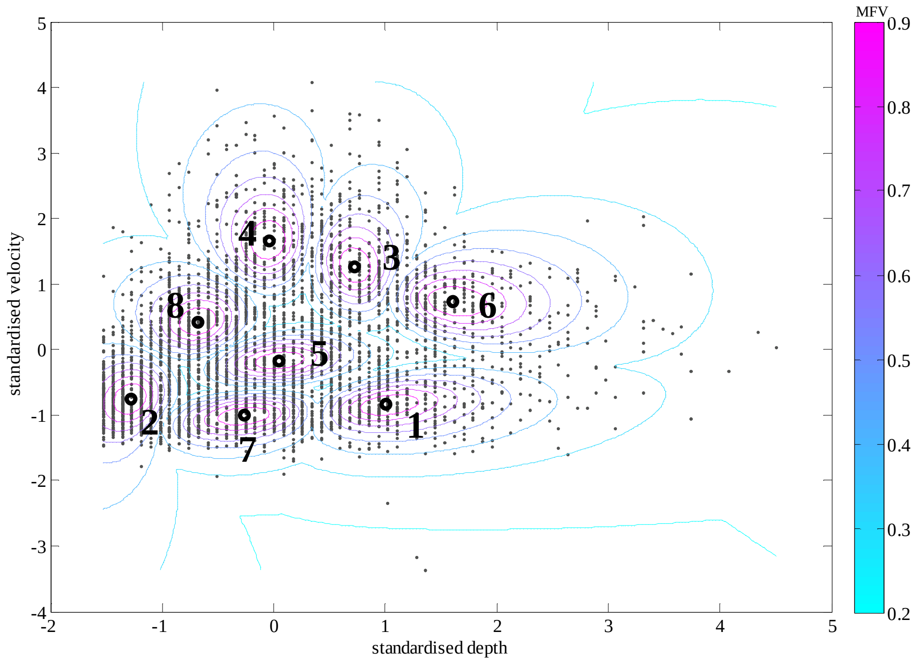


Figure 27. Scatterplot of hydraulic data overlaid with cluster centroids (black circle) and membership function value (MFV) contours for the 8-cluster Gustafson-Kessel classification. Cluster numbers correspond to the images in Figure 28.

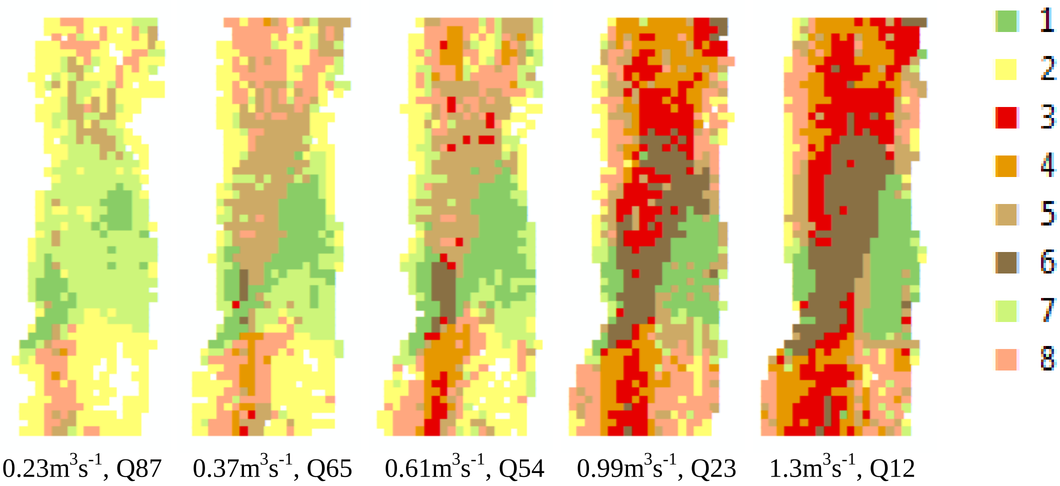


Figure 28. Location and extent of clusters in the 8-GK classification (defuzzified using the maximum likelihood rule).

Appendix E

Additional methods and results

R code for resampling the diversity (*H'*) of hydraulic patches

```
require(vegan)

### two samples with abundance data:
### 1st sample of counts (numbers representing hydraulic patches 1-n
at **FLOW 1**)
dat1 <- c(**ADD VALUES OF HYDRAULIC PATCH COUNTS SEPARATED BY
COMMA**)

### 2nd sample of counts (numbers representing hydraulic patches 1-n
at **FLOW 2**)
dat2 <- c(**ADD VALUES OF HYDRAULIC PATCH COUNTS SEPARATED BY COMMA,
MUST BE SAME NUMBER OF HYDRAULIC PATCHES AS SITE 1**)

(div1=diversity(t(dat1),"shannon"))
(div2=diversity(t(dat2),"shannon"))
(rich1=sum(dat1>0))
(rich2=sum(dat2>0))

(tr.diff.div=abs(div1-div2))          ### observed difference
(tr.diff.rich=abs(rich1-rich2))       ### ...

K=2000

pop.diff.div <- pop.diff.rich <- rep(NA,K)      ### dataframe for
                                    null population of differences
pop.diff.div[1]=tr.diff.div
pop.diff.rich[1]=tr.diff.rich

for(i in 2:K){                        ### loop to generate null
                                          pop.diff. of differences

  ind1<-sum(dat1)                     ### sum of individuals sample
                                          no.1
  ind2<-sum(dat2)                     ### sum of ... no.2
  pool<-c(rep(1:length(dat1),dat1),   ### pooled sample with numbers
                                          representing species
          rep(1:length(dat2),dat2))   ### replicated as often as no.
                                          of individuals

  temp1=sample(pool,ind1,replace=T)   ### resample no.1
  temp2=sample(pool,ind2,replace=T)   ### resample no.2

  ### calculate diversity:
  div1.temp=diversity(t(tabulate(temp1)),"shannon")
  div2.temp=diversity(t(tabulate(temp2)),"shannon")

  rich1.temp=sum(tabulate(temp1)>0)
  rich2.temp=sum(tabulate(temp2)>0)

  temp.diff.div=abs(div1.temp-div2.temp)
  pop.diff.div[i]=temp.diff.div
  temp.diff.rich=abs(rich1.temp-rich2.temp)
  pop.diff.rich[i]=temp.diff.rich}

(p.div=sum(pop.diff.div>=abs(tr.diff.div))/K)
(p.rich=sum(pop.diff.rich>=abs(tr.diff.rich))/K)

### diagramms to show null-distributions with obs. differences
pdf (file= "**FILE_NAME_HERE**.pdf", onefile = TRUE)

hist(pop.diff.div, breaks=100, xlab="Difference in Shannon diversity
index")
abline(v=tr.diff.div,lty=3,col=2,lwd=2)
dev.off()
```

Results of tests of significant differences between hydraulic patch diversity (H') significance test results

Table 1. p-values associated with differences in hydraulic patch type diversity between each flow at the Leigh Brook site. Bold italic indicates highly significant difference (p<0.001), italic indicates significant difference (p<0.05), and orange text indicates no significant difference (p>0.05).

	v low	low	mod	high	v high
	0.26m^{3}s^{-1}	0.38 m^{3}s^{-1}	0.61 m^{3}s^{-1}	0.99 m^{3}s^{-1}	1.30 m^{3}s^{-1}
	Q87	Q67	Q45	Q23	Q14
v low					
low	***0.0005***				
mod	***0.0005***	*0.043*			
high	***0.0005***	***0.0005***	*0.029*		
v high	***0.0005***	***0.0005***	*0.044*	0.765	

Table 2. p-values associated with differences in hydraulic patch type diversity between each flow at the River Salwarpe site. Bold italic indicates highly significant difference (p<0.001), italic indicates significant difference (p<0.05), and orange text indicates no sig difference (p>0.05).

	v low	low	mod	high	v high
	0.53 m^{3}s^{-1}	0.79 m^{3}s^{-1}	1.14 m^{3}s^{-1}	1.63 m^{3}s^{-1}	1.84 m^{3}s^{-1}
	Q89	Q67	Q38	Q21	Q17
v low					
low	0.105				
mod	***0.005***	*0.016*			
high	***0.0005***	*0.0105*	0.365		
v high	***0.0005***	0.1895	0.055	0.408	

Table 3. p-values associated with differences in hydraulic patch type diversity between each flow at the River Arrow site. Bold italic indicates highly significant difference (p<0.001), italic indicates significant difference (p<0.05), and orange text indicates no significant difference (p>0.05).

	v low	low	mod	high	v high
	0.21 m^{3}s^{-1}	0.30 m^{3}s^{-1}	0.42 m^{3}s^{-1}	0.87 m^{3}s^{-1}	1.41 m^{3}s^{-1}
	Q87	Q70	Q53	Q22	Q13
v low					
low	0.33				
mod	0.15	0.45			
high	***0.0005***	***0.0075***	*0.028*		
v high	0.82	0.26	*0.031*	***0.0025***	

Appendix F

River Arrow hydraulic patch dynamics

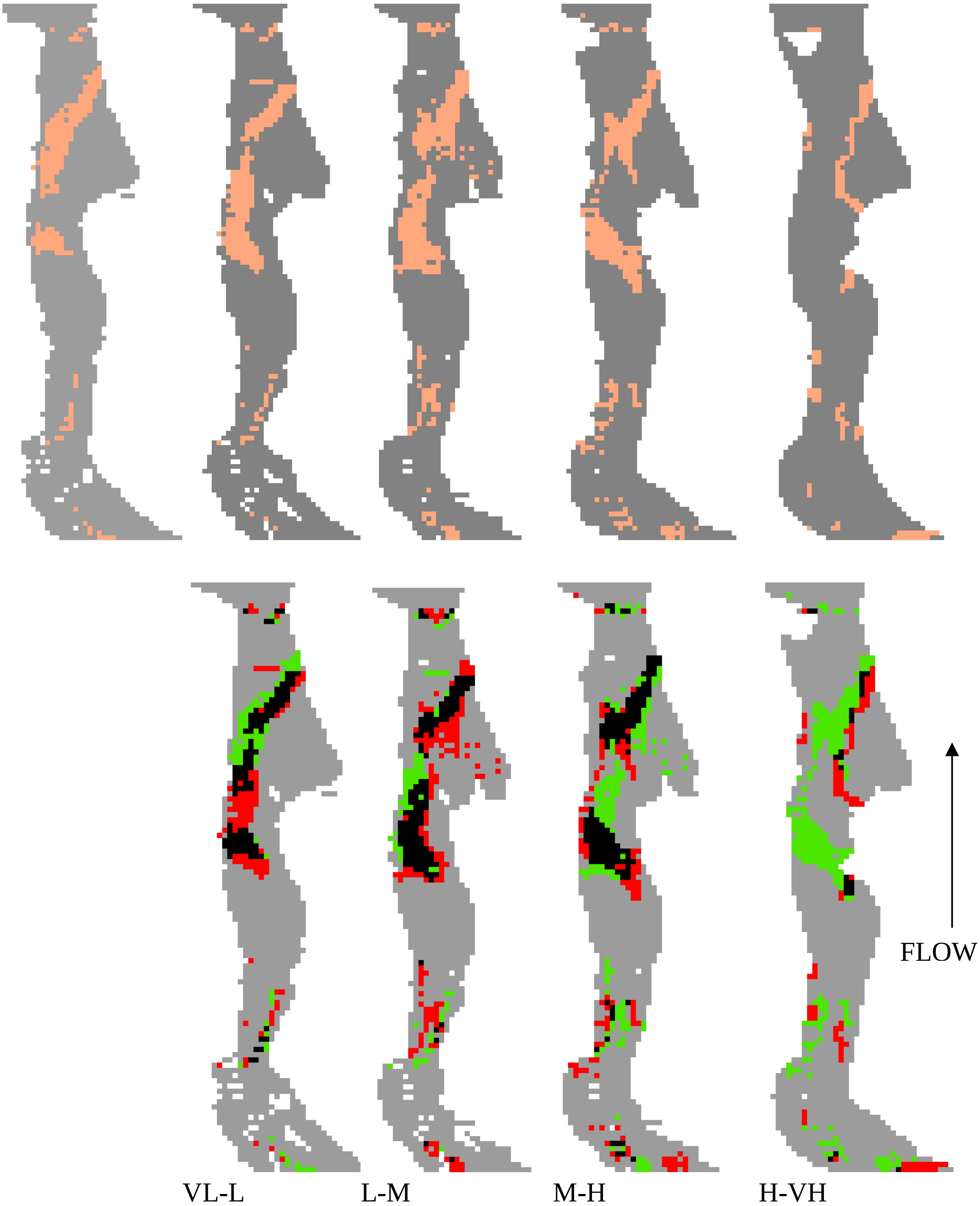
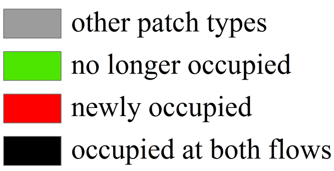


Figure 1. Change in the location and extent of hydraulic patch type RA1 (moderate-slow) between consecutive hydraulic surveys



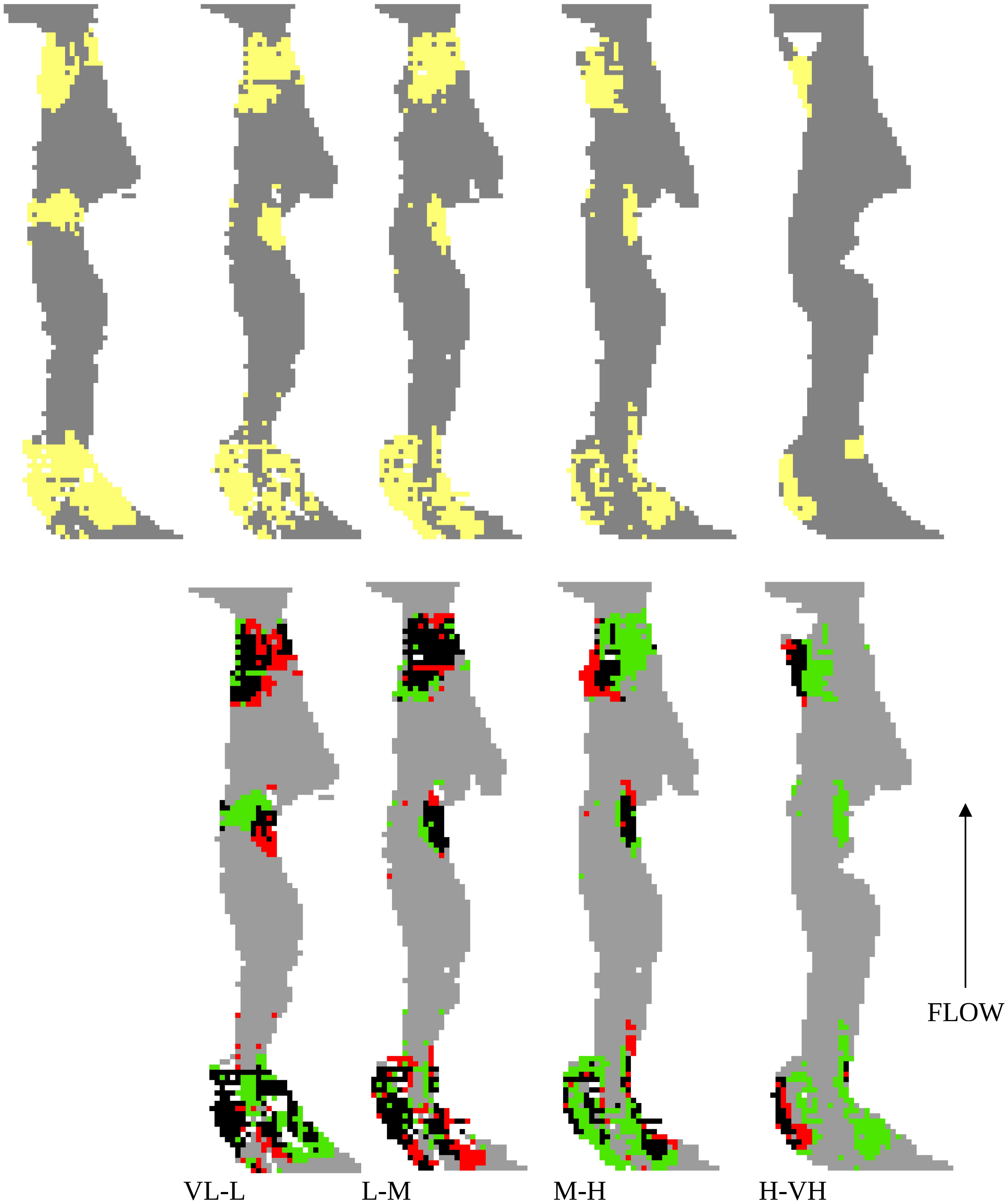


Figure 2. Change in the location and extent of hydraulic patch type RA2 (shallow-slow) between consecutive hydraulic surveys

- other patch types
- no longer occupied
- newly occupied
- occupied at both flows

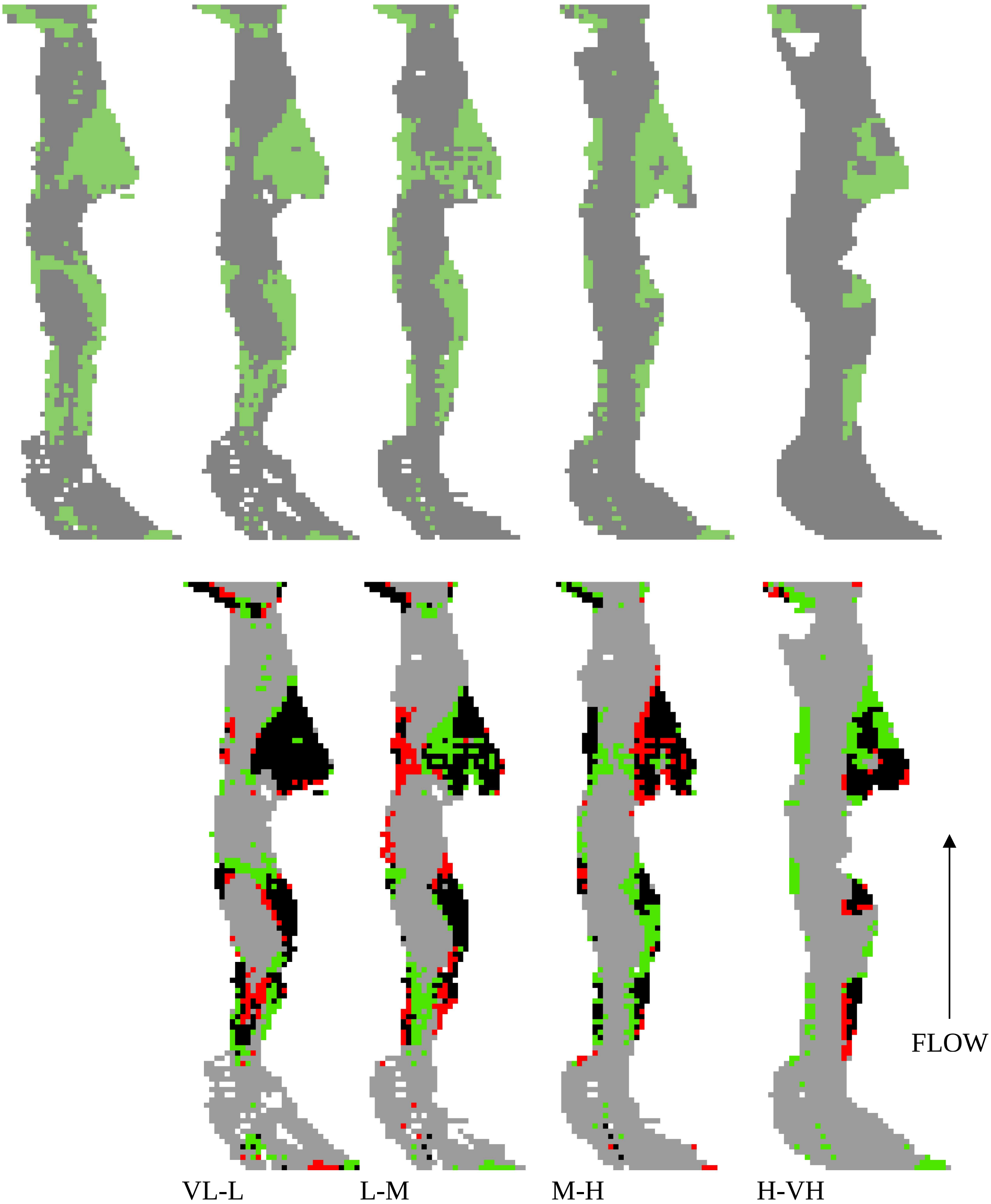


Figure 3. Change in the location and extent of hydraulic patch type RA3 (moderate-very slow) between consecutive hydraulic surveys

other patch types
no longer occupied
newly occupied
occupied at both flows

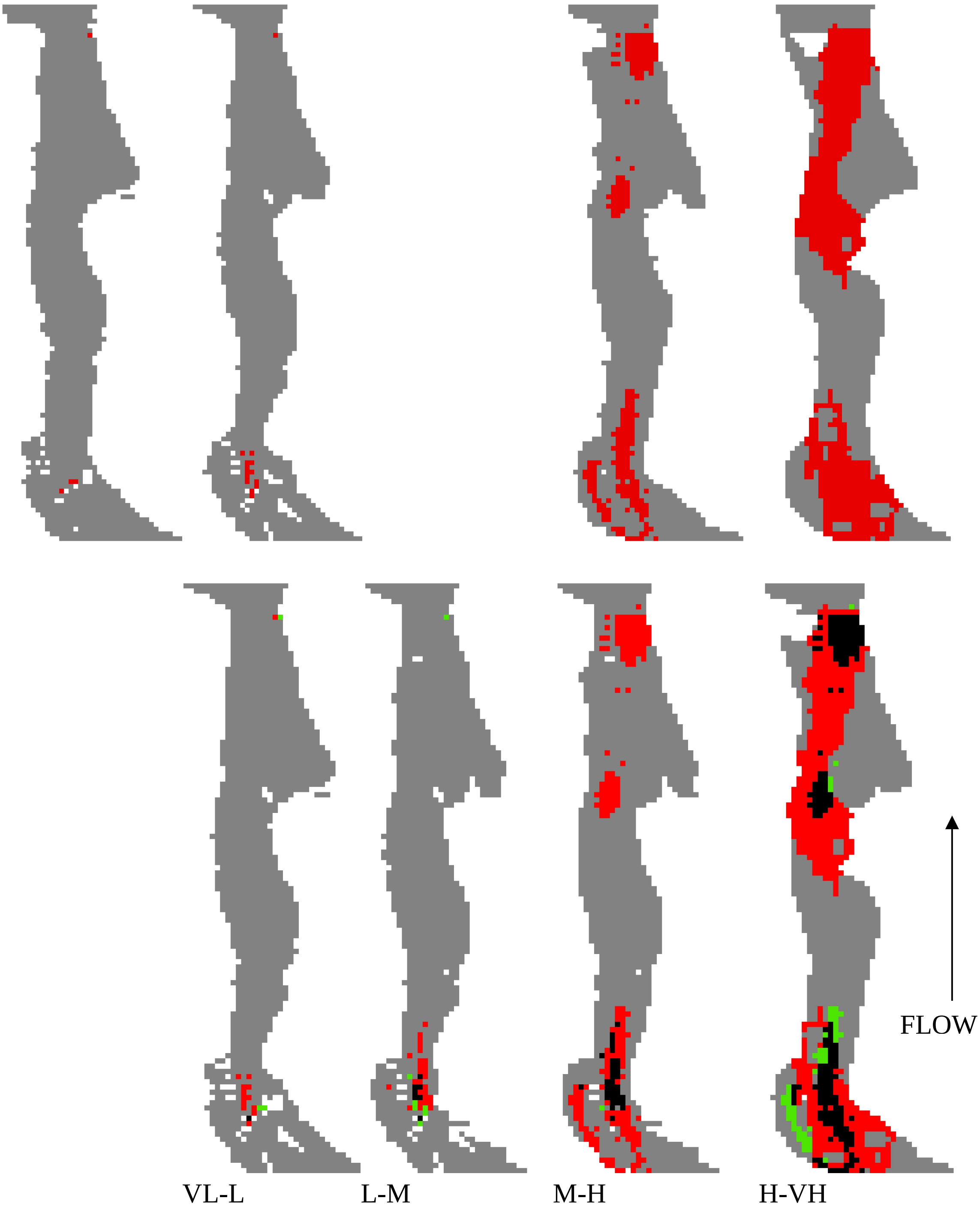
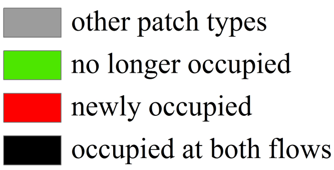


Figure 4. Change in the location and extent of hydraulic patch type RA4 (moderate-fast) between consecutive hydraulic surveys



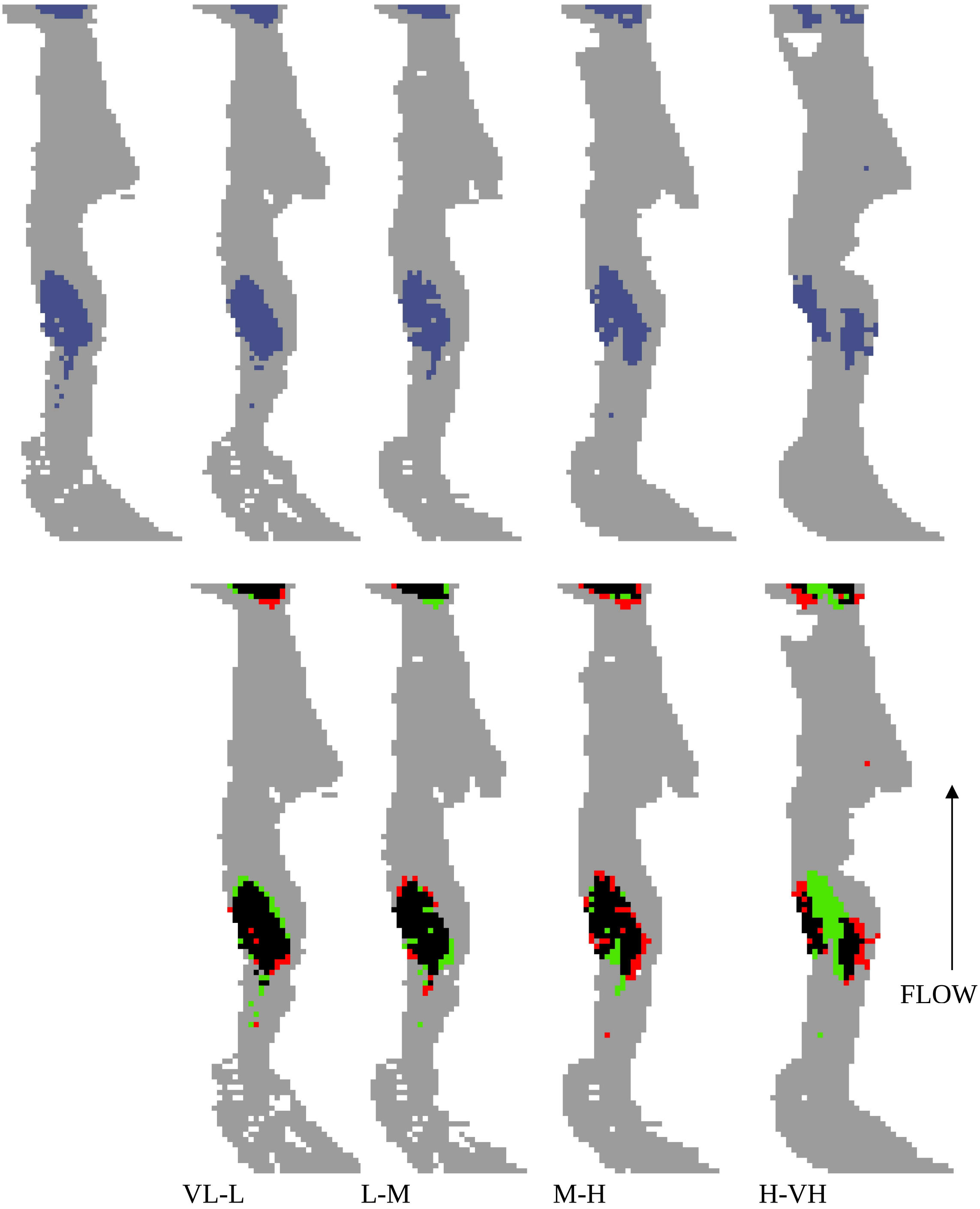
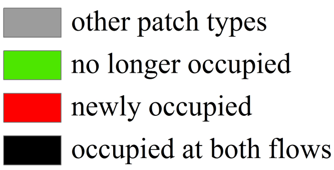


Figure 5 Change in the location and extent of hydraulic patch type RA5 (very deep-very-slow) between consecutive hydraulic surveys



Appendix G

River Salwarpe hydraulic patch dynamics

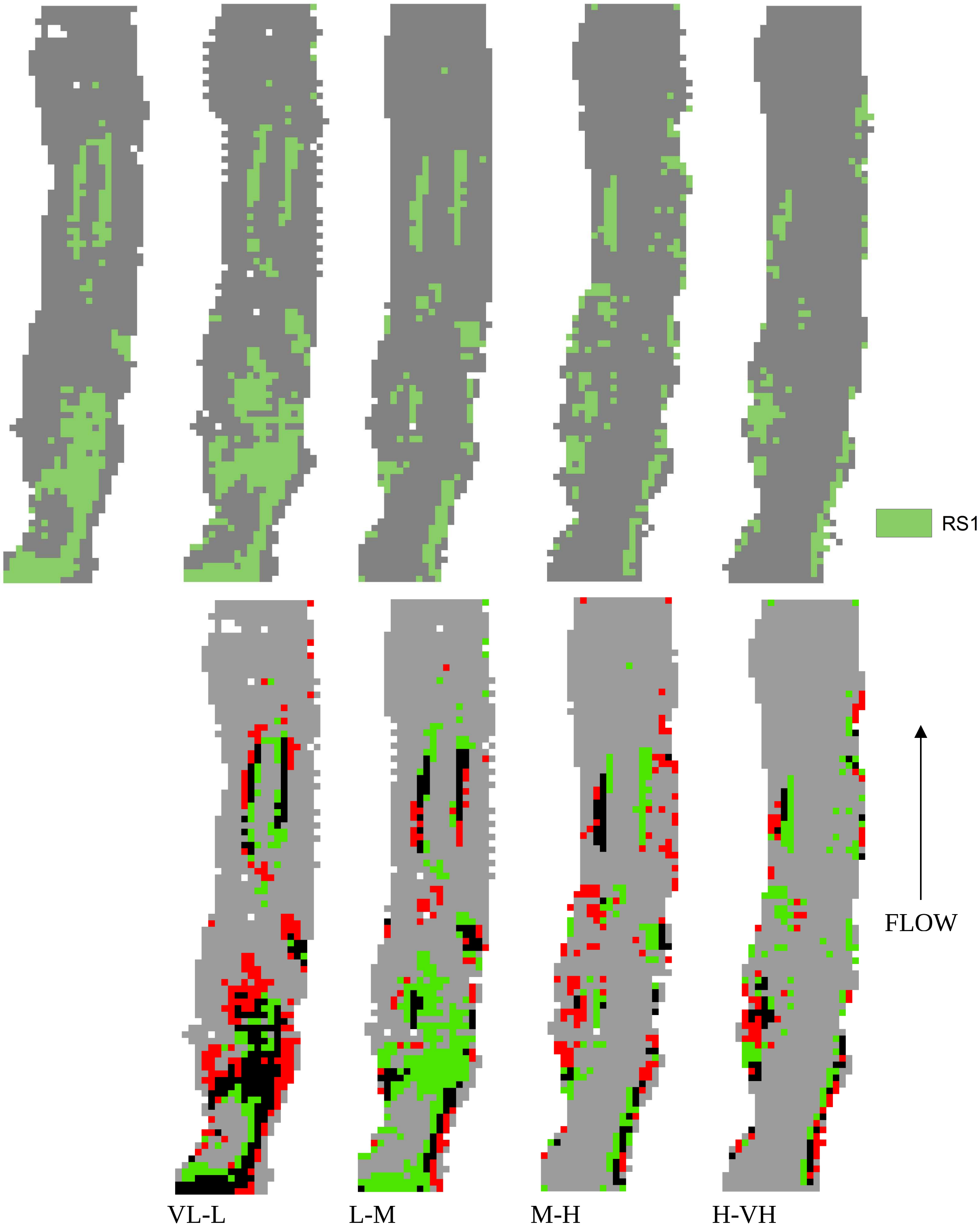


Figure 1. Location and extent of hydraulic patch type RS1 (moderate-slow) at each hydraulic survey (top row) and the change in location of RS1 between consecutive hydraulic surveys (bottom row).

- other patch types
- no longer occupied
- newly occupied
- occupied at both flows

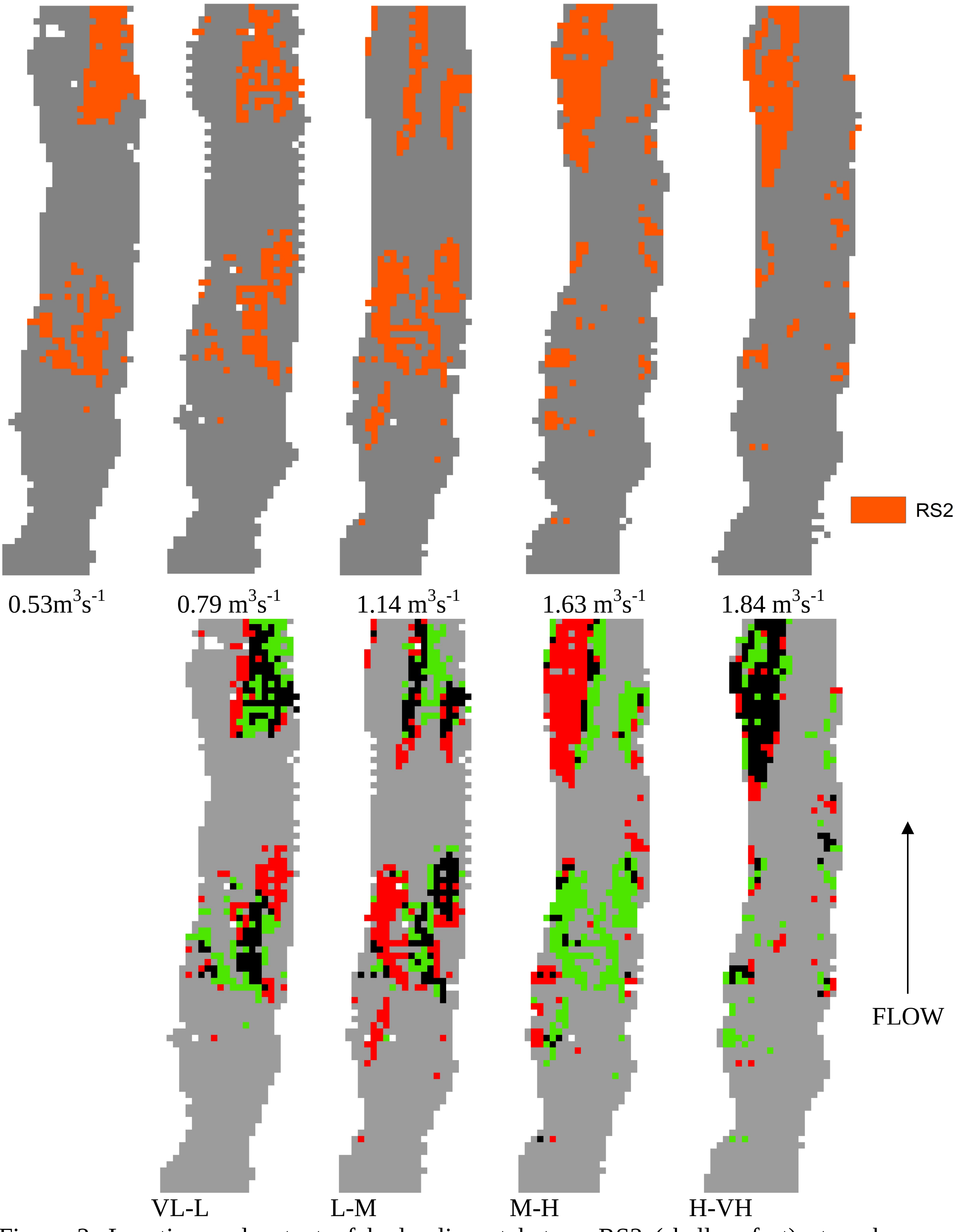


Figure 2. Location and extent of hydraulic patch type RS2 (shallow-fast) at each hydraulic survey (top row) and the change in location of RS2 between consecutive hydraulic surveys (bottom row).

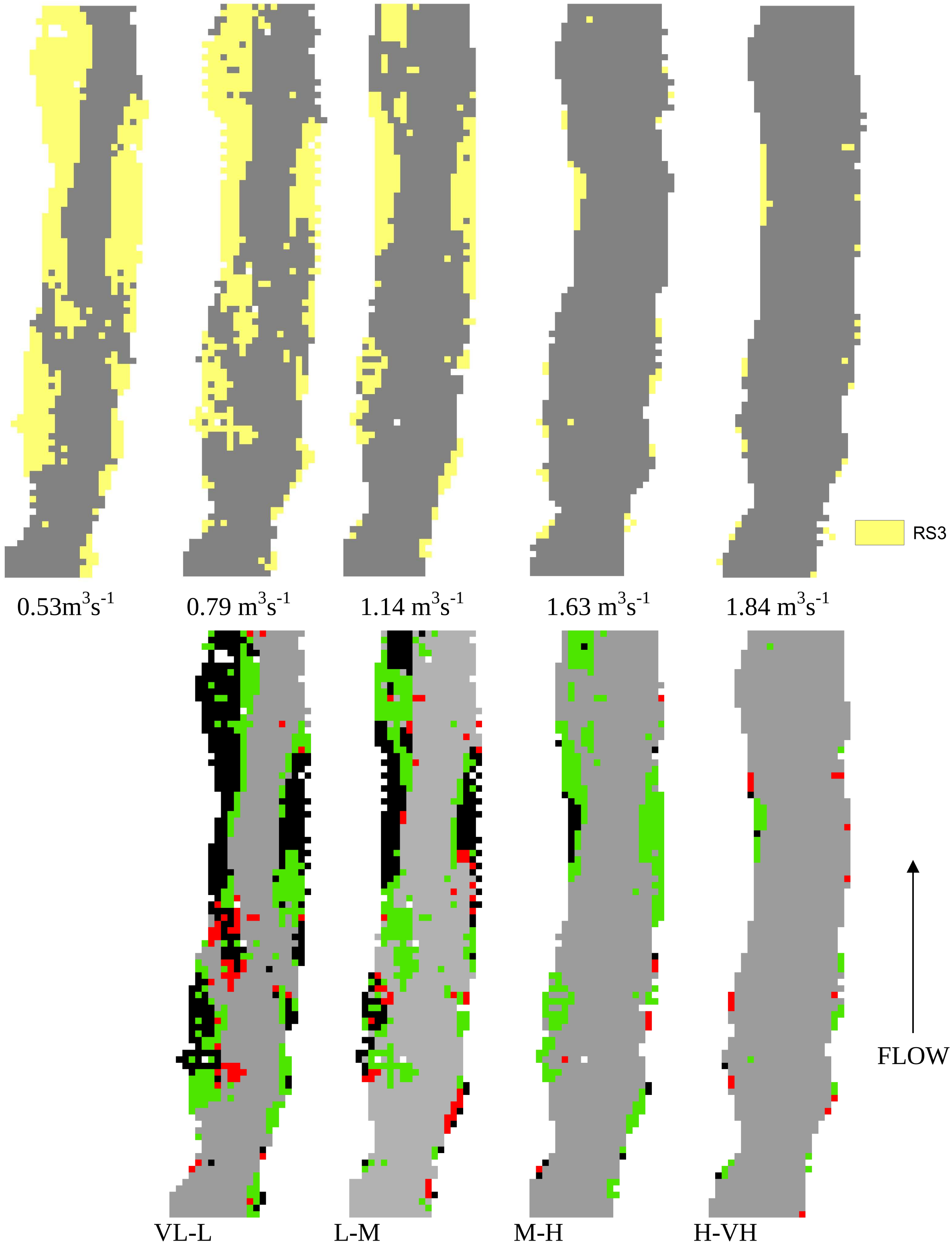


Figure 3. Location and extent of hydraulic patch type RS3 (shallow-slow) at each hydraulic survey (top row) and the change in location of RS3 between consecutive hydraulic surveys (bottom row).

other patch types
no longer occupied
newly occupied
occupied at both flows

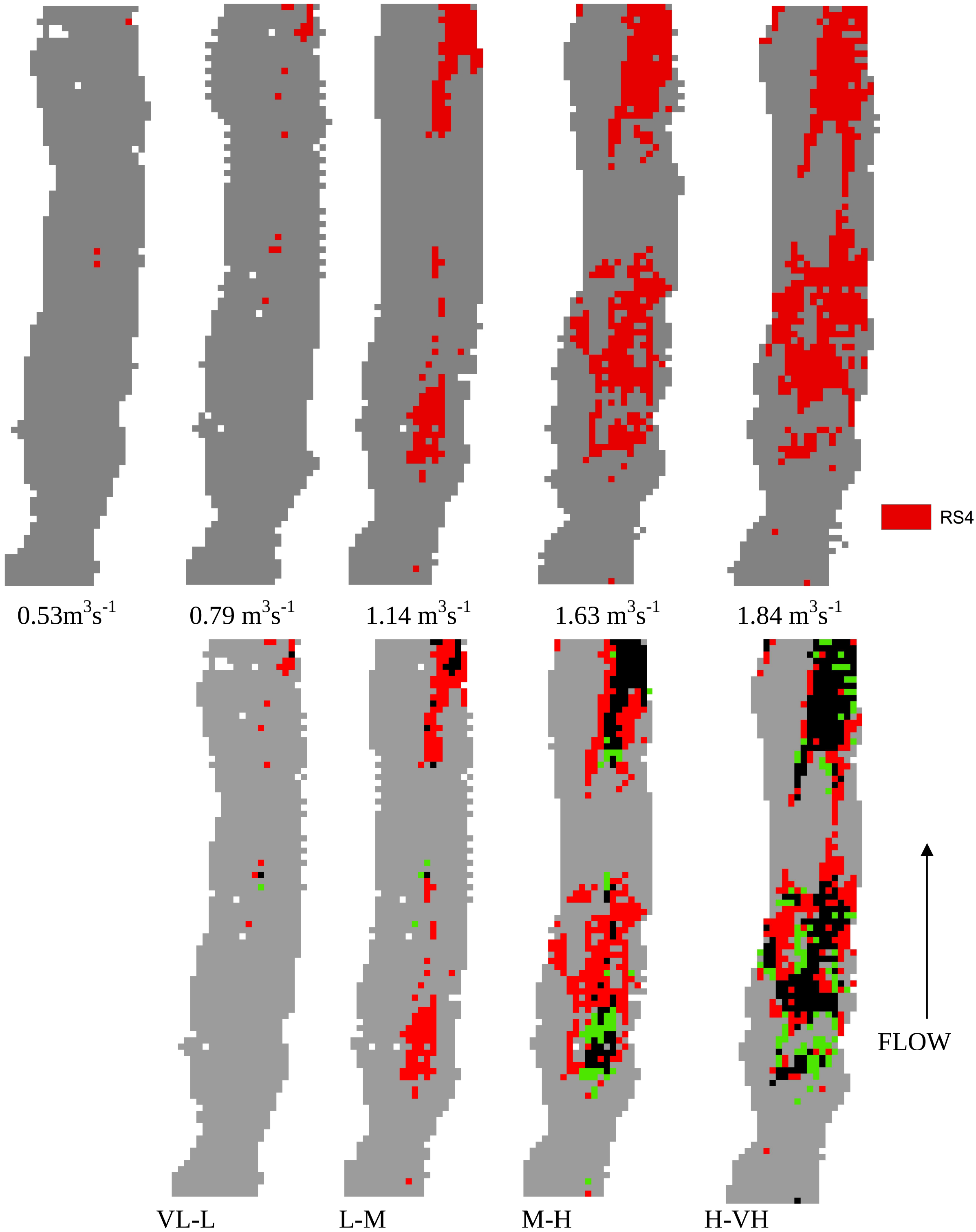


Figure 4. Location and extent of hydraulic patch type RS4 (moderate-fast) at each hydraulic survey (top row) and the change in location of RS4 between consecutive hydraulic surveys (bottom row).

other patch types
no longer occupied
newly occupied
occupied at both flows

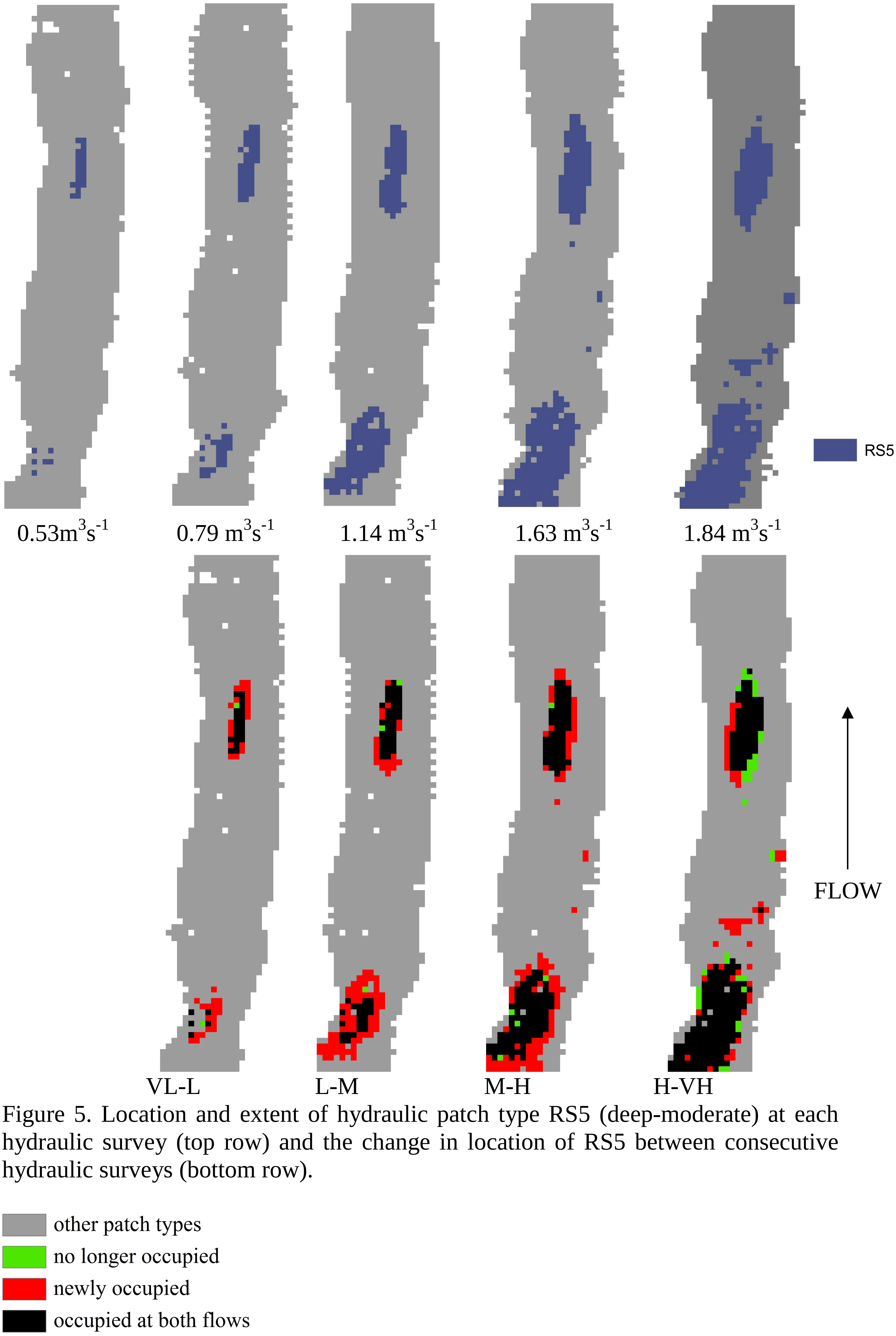


Figure 5. Location and extent of hydraulic patch type RS5 (deep-moderate) at each hydraulic survey (top row) and the change in location of RS5 between consecutive hydraulic surveys (bottom row).

Appendix H

Leigh Brook hydraulic patch dynamics

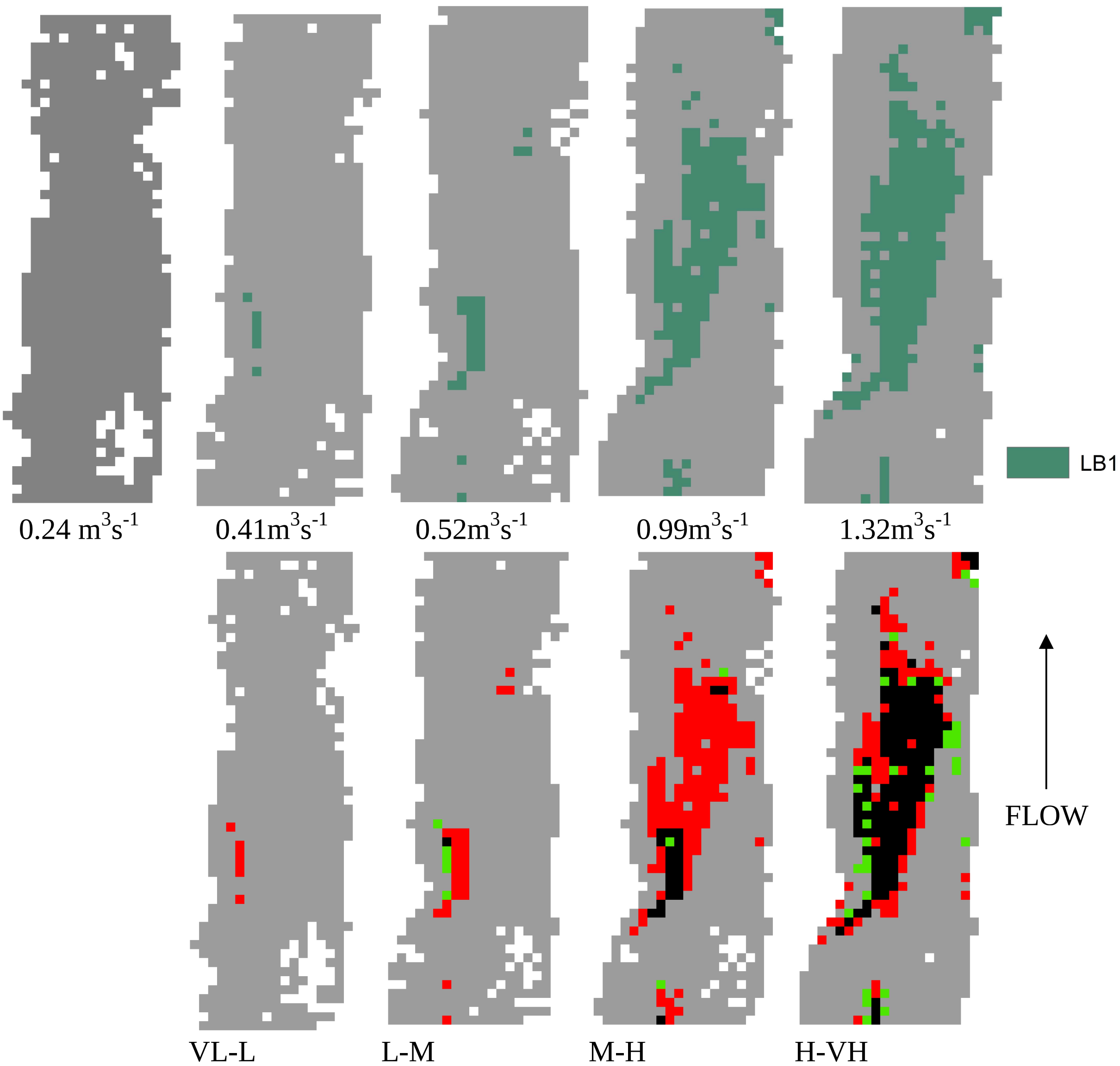


Figure 1. Location and extent of hydraulic patch type LB1 (moderate-moderate) at each hydraulic survey (top) and the spatial turnover of LB1 between consecutive hydraulic surveys (bottom).

- other patch types
- no longer occupied
- newly occupied
- occupied at both flows

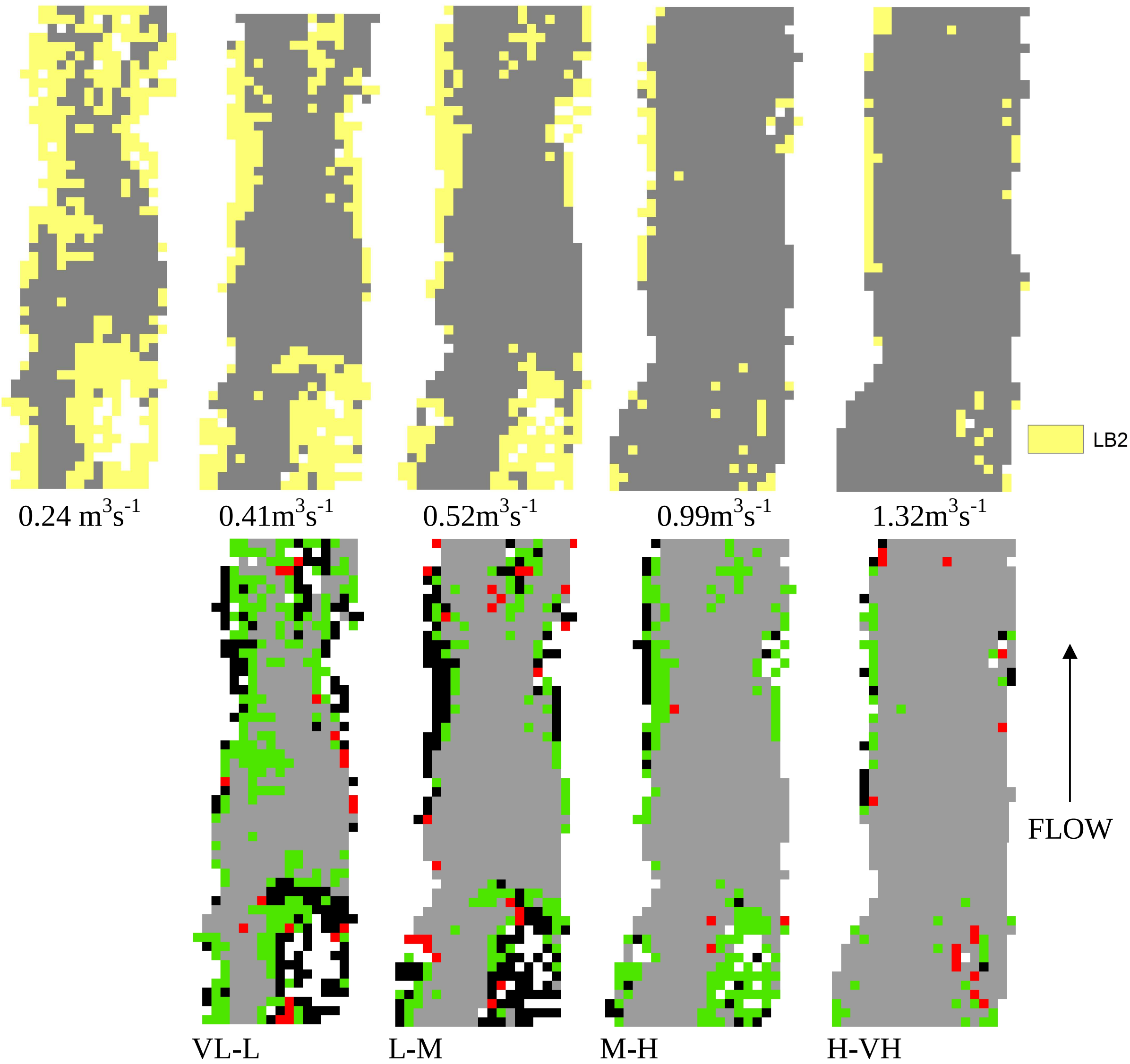


Figure 2. Location and extent of hydraulic patch type LB2 (shallow-slow) at each hydraulic survey (top) and the spatial turnover of LB2 between consecutive hydraulic surveys (bottom).

other patch types
no longer occupied
newly occupied
occupied at both flows

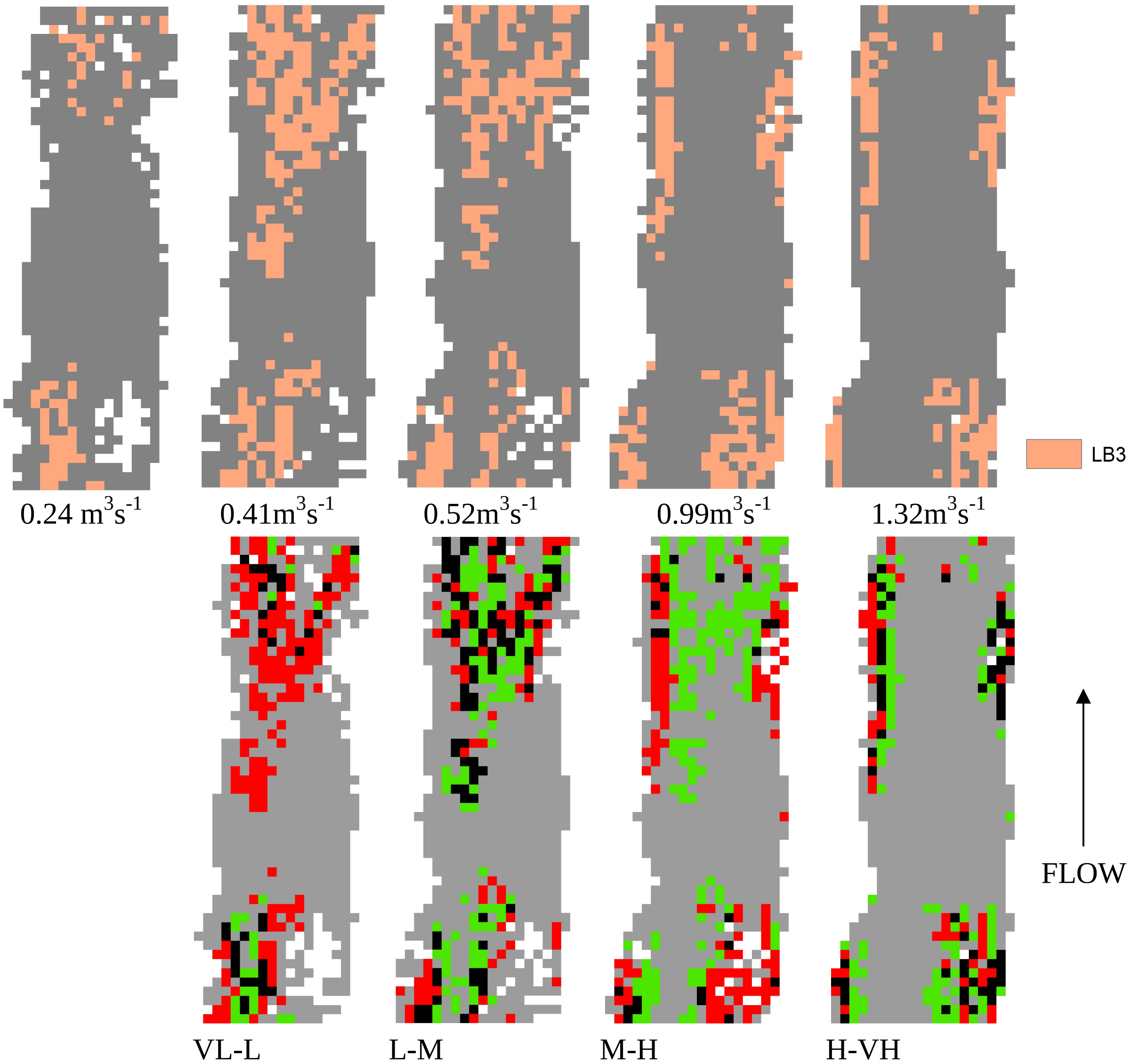
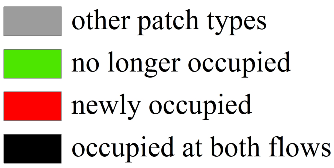


Figure 3. Location and extent of hydraulic patch type LB3 (shallow-moderate) at each hydraulic survey (top) and the spatial turnover of LB3 between consecutive hydraulic surveys (bottom).



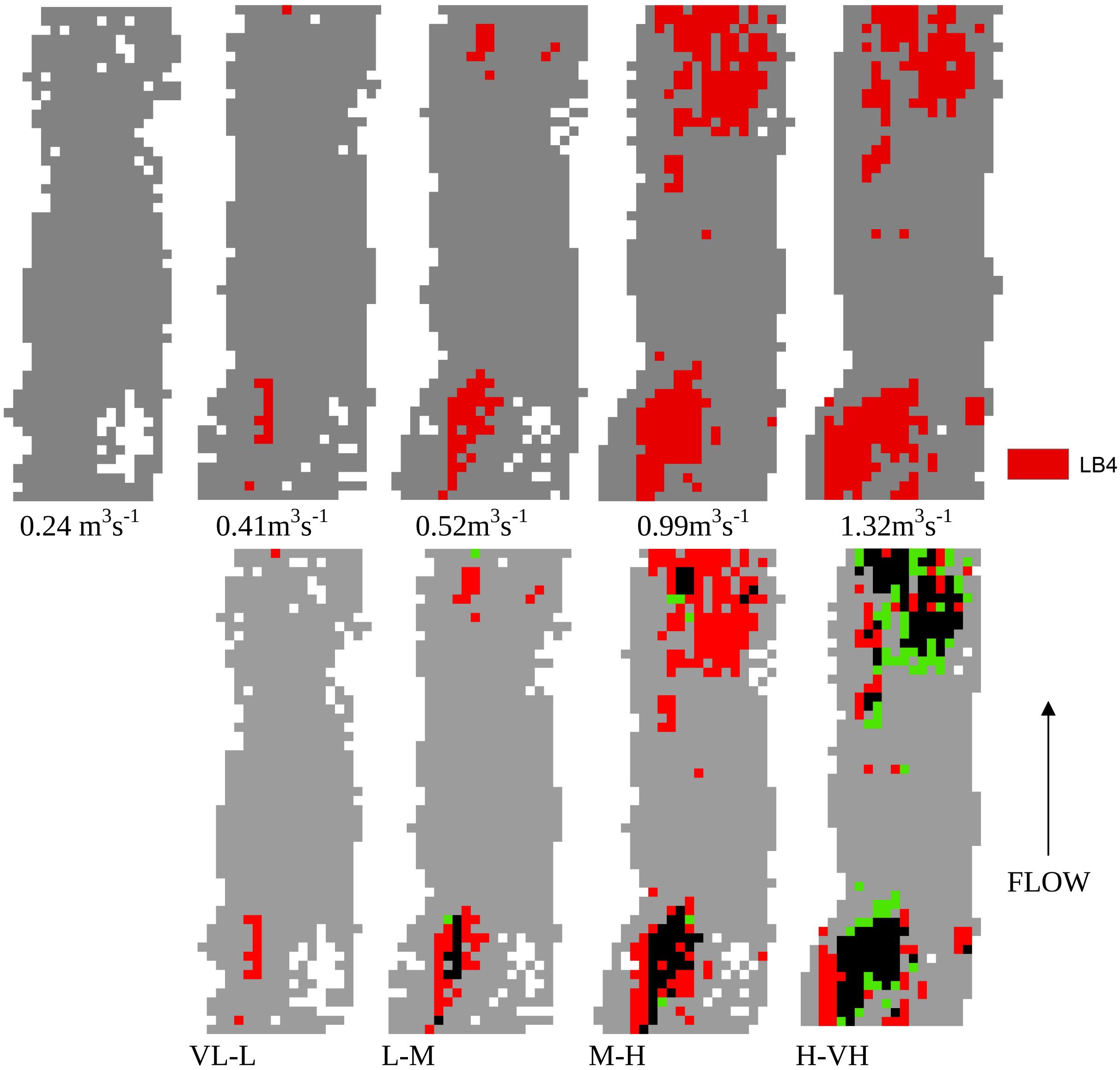
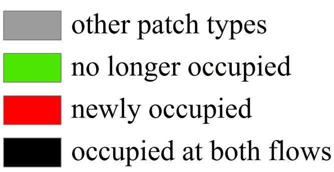


Figure 4. Location and extent of hydraulic patch type LB4 (moderate-fast) at each hydraulic survey (top) and the spatial turnover of LB4 between consecutive hydraulic surveys (bottom).



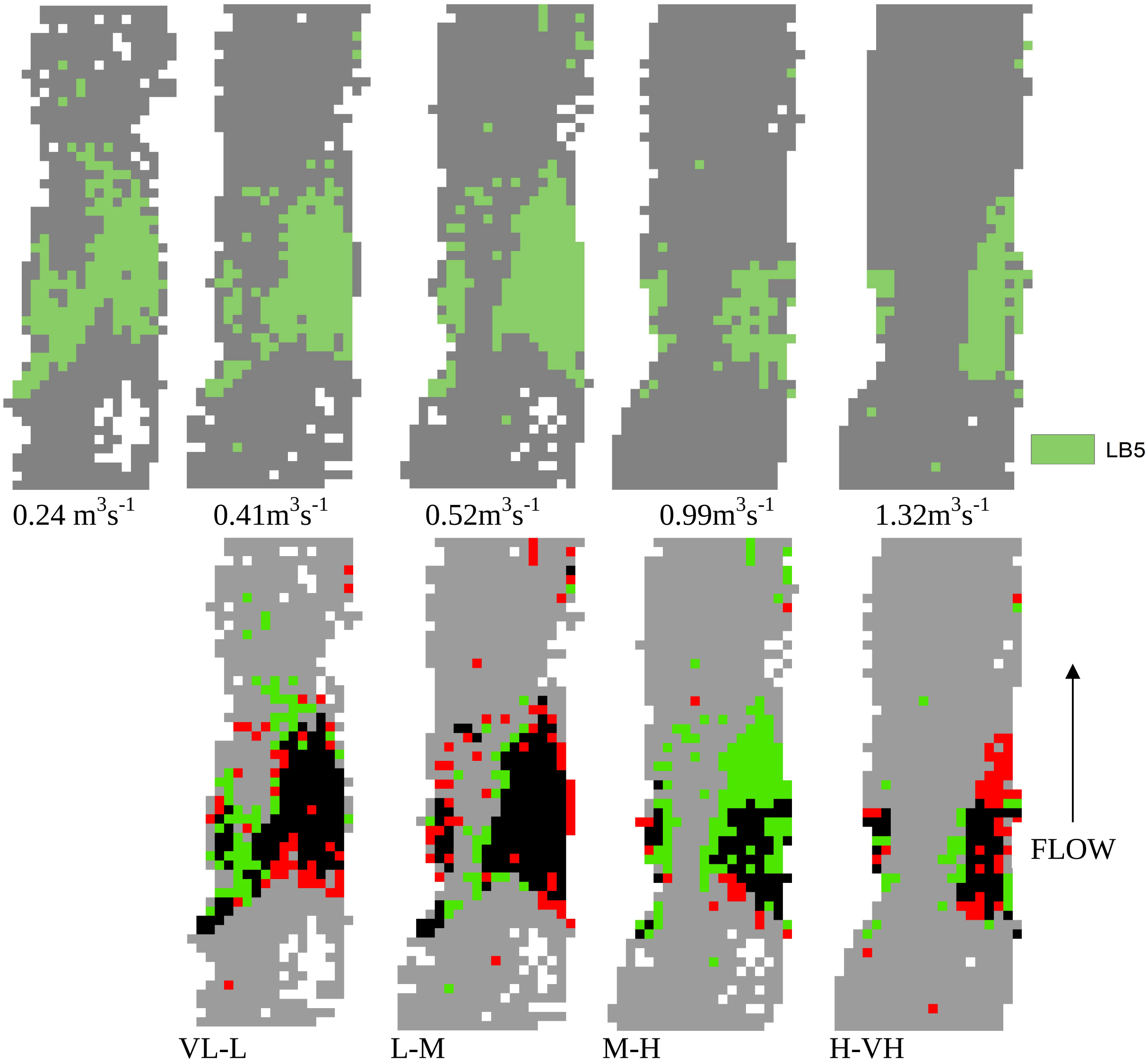


Figure 5. Location and extent of hydraulic patch type LB5 (moderate-slow) at each hydraulic survey (top) and the spatial turnover of LB5 between consecutive hydraulic surveys (bottom).

other patch types
no longer occupied
newly occupied
occupied at both flows

Appendix I

Wallis et al. (2012) paper

RIVER RESEARCH AND APPLICATIONS
River Res. Applic. (2010)
Published online in Wiley Online Library
(wileyonlinelibrary.com) DOI: 10.1002/rra.1468

A FRAMEWORK FOR EVALUATING THE SPATIAL CONFIGURATION AND TEMPORAL DYNAMICS OF HYDRAULIC PATCHES

C. WALLIS,[a]* I. MADDOCK,[a] F. VISSER[a] and M. ACREMAN[b]
[a] *Institute of Science and the Environment, University of Worcester, Henwick Grove, Worcester, UK*
[b] *Centre for Ecology and Hydrology, Crowmarsh Gifford, Wallingford, UK*

ABSTRACT

A framework for evaluating the spatial configuration and temporal dynamics of hydraulic patches was tested in a UK lowland river. Detailed hydraulic assessment was carried out at four discharges between 0.303 $m^3 s^{-1}$ and 1.410 $m^3 s^{-1}$ in a 56 m reach. Five hydraulic patches, as combinations of depth and mean column velocity, and the transitional zones between them were delineated using fuzzy cluster analysis. The composition and configuration of the hydraulic patch mosaic was quantified using spatial metrics. Results showed that the proportion, size, shape and relative location of hydraulic patches all varied with discharge, however intermediate flows appeared to support the most diverse hydraulic patch composition and configuration. This suggests the spatial influence of mesoscale bedforms on hydraulic patches is mediated by temporal variations in discharge. Transitional areas between hydraulic patches occupied a significant proportion of the reach at all flows (33–38%) and may function as in-stream ecotones. The framework addresses the need for a quantitative, spatially explicit approach to hydraulic assessment which could be used to assess the implications of flow regulation and hydromorphological alteration on hydraulic diversity. Copyright © 2010 John Wiley & Sons, Ltd.

KEY WORDS: hydraulic heterogeneity; patch dynamics; discharge; fuzzy cluster analysis

Received 7 May 2010; Revised 7 September 2010; Accepted 21 September 2010

INTRODUCTION

Understanding the links between hydromorphology and ecology is a key research objective and will improve the scientific basis for hydromorphological standards required by the EU Water Framework Directive (WFD) and inform river restoration (Petts *et al*., 2006; Renschler *et al*., 2007; Sear, 2009; Vaughan *et al*., 2009). Meso scale interactions between channel morphology and discharge create a heterogeneous hydraulic environment which provides a variety of in-stream habitats for freshwater biota (Maddock, 1999). By characterizing the hydraulic environment, it is possible to evaluate hydromorphological conditions at an ecologically relevant scale. Hydraulic heterogeneity can be characterized by the relative proportion of hydraulic units present (composition), their spatial arrangement (configuration) and their temporal dynamics (change with discharge) (Cadenasso *et al*., 2006). Each component has ecological significance and should be incorporated into hydraulic assessment methods to give a full account of hydraulic heterogeneity (Newson and Newson, 2000).

Developing an accurate classification of hydraulic units provides the foundation for effective assessment, but is not a straightforward task given the complex, dynamic and heterogeneous nature of the hydraulic environment (Poole *et al*., 1997; Legleiter and Goodchild, 2005). Physical biotopes (Padmore, 1997; Wadeson and Rowntree, 1998) are currently used as the standard meso scale unit of physical habitat in the UK (Raven *et al*., 1997). Whilst rapid and cost-effective, this approach is affected by operator variability (Roper and Scarnecchia, 1995; Eisner *et al*., 2005), can obscure hydraulic differences (Clifford *et al*., 2006) and imposes crisp boundaries on hydraulic units. Previous work has demonstrated the potential advantages of numerical classification methods (cluster analysis) for delineating hydraulic units quantitatively using a single (e.g. mean column velocity) (Emery *et al*., 2003) or combination (e.g. depth, velocity and substrate) of hydraulic variables (Inoue and Nakano, 1999). Fuzzy cluster analysis has added advantages for classifying continuous environmental data because it replaces the binary group membership (0/1) associated with crisp, linear boundaries with partial membership functions (0-1) that reflect the degree to which each observation belongs to every group in the classification, and in so doing better representing ambiguous class boundaries (Zadeh, 1965; Bezdek *et al*., 1984; Burrough, 1996; Arrell *et al*., 2007). Legleiter and Goodchild (2005) showed that by using fuzzy cluster analysis to delineate hydraulic patches it is possible to, "circumvent the subjectivity of conventional habitat classification and provide a richer representation that more faithfully honours the complexity of the fluvial environment" (p. 30).

*Correspondence to: C. Wallis, Institute of Science and the Environment, University of Worcester, Henwick Grove, Worcester WR2 6AJ, UK.
E-mail: c.wallis@worc.ac.uk