From engineered channel to functioning stream ecosystem; rates, patterns and mechanisms of development in a realigned river channel

Thesis submitted in accordance with the requirements of
the University of Stirling for the degree of Doctor in Philosophy

By
Charles Perfect

Submitted to the University of Stirling
February 5th 2010

Acknowledgements

To Dave, Nigel, Mum, Dad, Phil, Kate, Stephen, Amber, Tory, Roser, Helen, Scot, Bill, Dilley, David, Catherine, Kevin, Jon, Sandy and Ian. My sincerest apologies, considerable gratitude and heartfelt thanks.

Without the generous funding from Scottish Coal Ltd and the University of Stirling and considerable support from staff this research would not have been possible.

Declaration of Authenticity

I certify that this dissertation is my original work, gathered and utilized especially to fulfil the requirements and objectives of this research degree, and has not been previously submitted to any other University for the degree of Doctor in Philosophy. All external sources of information are properly referenced and acknowledged.

Signed:

Date:

Abstract

1. Realigning rivers is becoming common as a solution to conflicting needs of land development and ecolosystem preservation. Although an increasing number of projects are monitored, exactly how these channels develop as functional stream ecosystems is still poorly understood. Mining in the upper catchment of the River Nith (Scotland) required the realignment of 3km (approx.) of river. The engineered channel was designed around sound geomorphological principles of sediment transport and supply with a sinuous planform and pool-riffle sequences along the installed gravel-bed.

2. A comprehensive survey covering biotic and abiotic development was devised and implemented to test models and hypotheses relating to the development riverine habitats over the first three years.

2. Physical habitat development at the reach scale was investigated using fixed-point photography and differential GPS surveys of the thalweg and of cross-sectional form every 100m. This revealed the development of a relatively diverse streambed habitat in response to both the channel slope and planform. However, other than at meander bends where asymmetry developed over several years, little change was observed to the form of the engineered riverbanks.

3. Kick-net surveys of benthic invertebrate communities at 10 sites showed a negative relationship between specific measures of diversity and downstream distance during the early stages of development. (e.g. Richness with chainage at the 6 month stage) but the relationship degrades rapidly and is likely in part to appear as a result of low population densities.

4. Survey of transects through the riparian zone perpendicular to the river indicated that colonisation by vegetation is also related to distance along the realignment but physical habitat and geographical factors play a more dominant role over development (Canonical correspondence analysis of vegetation data in 2007)

5. Many of the indices of diversity for both biotic and abiotic elements of the ecosystem proved ineffective at detecting development at the reach scale. This may be because significant changes occur at a smaller scale than was detected by the surveys. It is likely that greater resolution is required to detect more ecologically meaningful relationships and patterns.

6. Overall study shows constructed realignments can rapidly develop a diverse streambed community within 24 months. Riparian communities are slower to develop because of the slow development of riverbank habitat diversity. Other ecosystem properties such as resilience and connectivity may take much longer.

Contents

Chapter 11 – Conclusions and Recommendations 229

Chapter 1

REALIGNING THE RIVER NITH

This short chapter introduces the aims of the thesis and presents general facts and figures about the River Nith and its catchment, laying out the background to the realignment project. The River Nith flows east out of the Ayrshire hills in Southern Scotland, then south to the sea at Dumfries. The realignment of a 3km section of the river diverted the course around the House of Water opencast mine site operated by Scottish to allow further excavation of coal in 2000 and again in 2004. The project aimed to mimic the natural channel as far as was practical and lessons learned following the 2000 diversion allowed a number of improvements to be made in this respect. This provided the opportunity to investigate a range of hypotheses relating to the colonisation and development of man-made channels into functioning ecosystems, testing many of the assumptions made by river engineers and restoration ecologists.

Aims

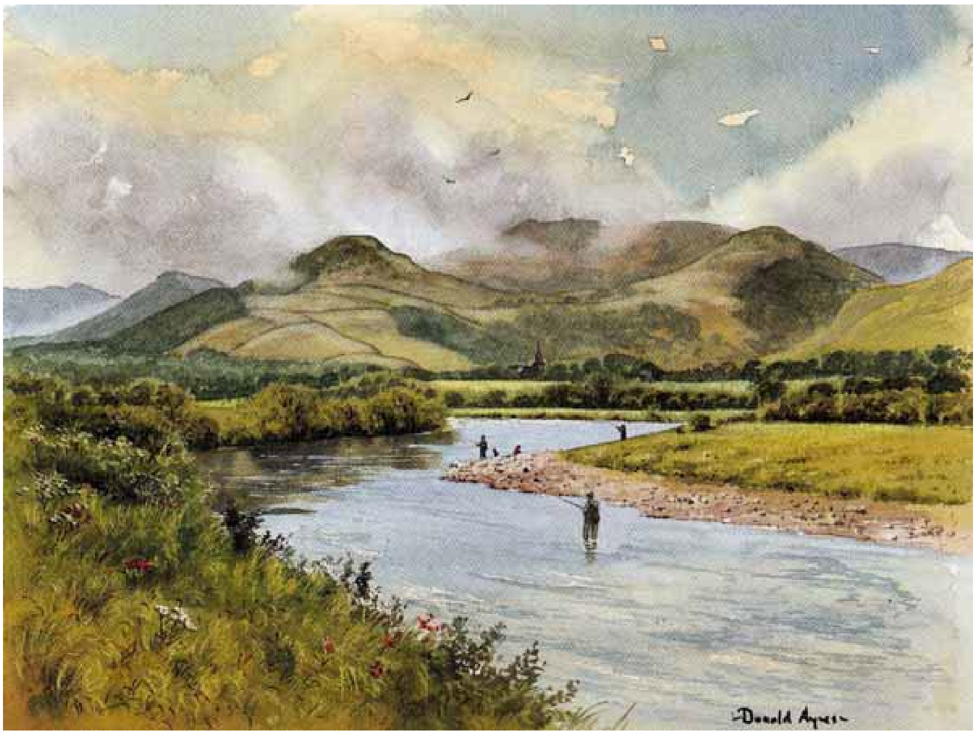
Provide an overview of the aims and objectives of the thesis
Describe the character of the River Nith including its condition prior to mining activities
Provide the background of the realignment of the Nith and Beoch at the House of Water site
Describe in detail the 2004 rerouting of the Nith and Beoch

To Thee Lov'd Nith

To thee, lov'd Nith, thy gladsome plains,
Where late wi' careless thought I rang'd,
Though prest wi' care and sunk in woe,
To thee I bring a heart unchang'd.
I love thee, Nith, thy banks and braes,
Tho' mem'ry there my bosom tear;
For there he rov*d that brake my heart.
Yet to that heart, ah ! still how dear

Robert Burns 1759-1796

River Nith



Donald Ayres 1936-

1.1 Introduction

For much of history, rivers have been revered by the arts and in folklore, although this did not prevent serious degradation to their morphology and biota over much of the 19th and 20th centuries. Despite the natural untamed image presented of Scottish rivers, 50% of upland and 72% of lowland rivers are not entirely natural. The Nith is typical in this respect with many of the same pressures that have affected many of Scotland's rivers. In the Ayrshire uplands these have been principally agriculture, silviculture and, more recently, opencast mining (SEPA, 2007). However, modern approaches to development and management are becoming more sensitive to the needs and importance of the environment, including an appreciation for the importance of scientific monitoring and discovery, providing an opportunity for original scientific research.

1.2 Overview of Aims and Objectives

This thesis presents research on the rates and patterns of development of a man-made river channel from an engineered near-trapezoidal form to a more natural geomorphologically and ecologically functional river. An overview of the aims to achieve this are set out below -

AIM1: Summarise the key events driving modern approaches to river channel design

i) Review our history of, and motivations for diverting rivers
ii) Describe the modern approaches to channel design
iii) Place in a context showing relevance to restoration ecology

AIM 2: Review theories in geomorphology and ecology relevant to the development of artificial channels

i) Relevant theory and understanding relating to fluvial hydrology, geomorphology and ecology
ii) Relevant theory and understanding relating to colonisation potential of biota

AIM 3: Test hypotheses relating to the development of channel morphology.

i) Establish the rate of physical habitat development on the bed of the new channel
ii) Investigate the ability of indices of physical habitat diversity to detect development

AIM 4: Examine the patterns of colonisation by invertebrate communities

i) Test the relationship of decreasing biological diversity with distance from upstream species pools
ii) Test the power of diversity indices to summarise the key elements of habitat colonisation and development
iii) Investigate the community structure of early colonists
iv) Investigate the role that mossy boulders could play in enhancing the restoration of rivers.

AIM 5: Examine the patterns of colonisation by riparian vegetation communities

i) Test the relationship between colonists and various pathways
ii) Investigate the importance of physical habitat development

AIM 6: Examine the role that planform plays in the development of a healthy ecosystem

i) Track the development of cross-sectional assymetry
ii) Investigate the affect that meander form has on the stream bed substrates
iii) Investigate the response of invertabrate communities to meander form

AIM 7: Draw conclusions and develop recommendations for the design of river channels and restoration projects

1.3 The Geography of the Nith

The River Nith is the largest river in the Scottish south-west. Rising in the Ayrshire uplands at 500mOD it then flows for 112km before discharging into the Solway Firth 22km south of Dumfries, draining a catchment area of over 1200km^2 (SEPA, 2007). The combined catchments of the Nith and Beoch above the House of Water site cover approximately 21km^2. When the House of Water site and associated drainage area is included the figure rises to 27.9km^2. There are areas of rough grazing but the catchment has mainly been turned over to conifer forestry plantations. The 23kms of stream and river channel feeding the R. Nith and Beoch Lane Burn upstream of the site have a predominantly gravel bed with short sections of bedrock.

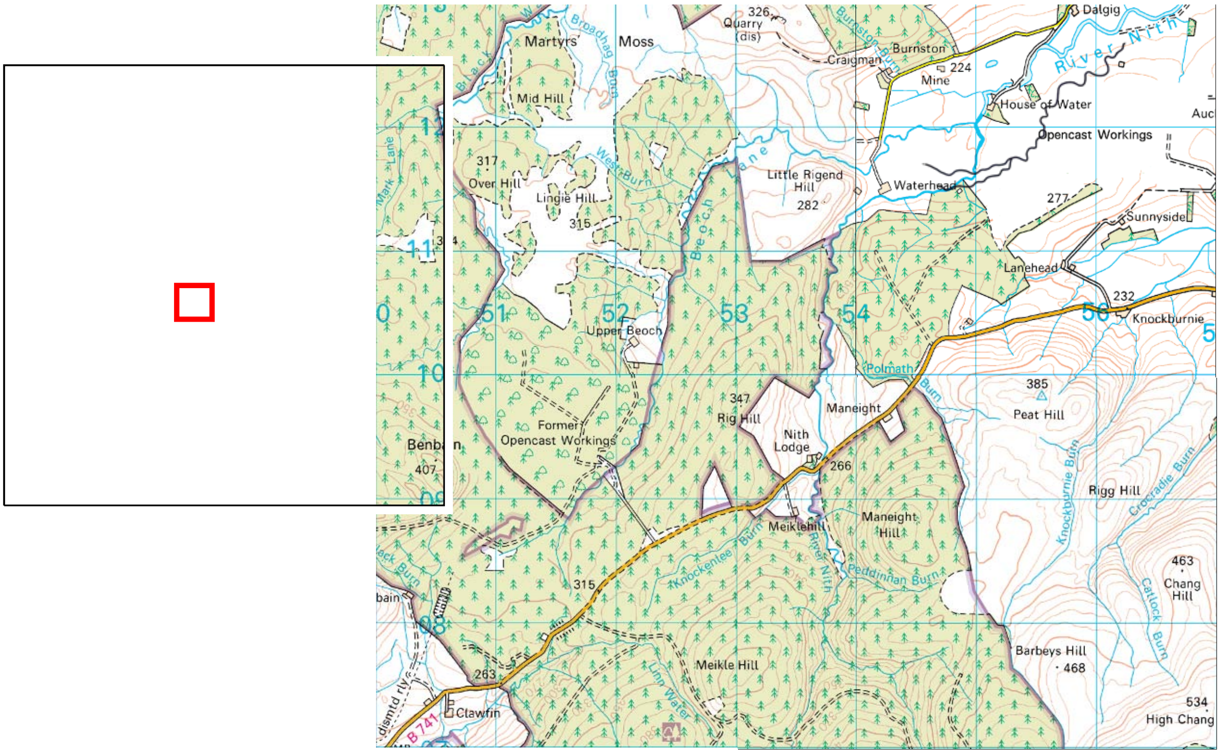


Figure 1.1: Ordnance Survey map of the River Nith catchment upstream of the House of Water site (visible in the top right of the map) and its location in Ayrshire, Scotland (inset).

The River and its tributaries offer great value for nature conservation as well as offering ecosystem services to the leisure and tourism industries. It is an important salmon river. Ten-year rod averages of over 1000 salmon and 2300 sea trout (Brydon, 2007) are an attraction for many anglers. Leslie (2002) estimates that the Nith contributes £2.2 million to the local economy. Under much of upland Ayrshire lie significant coal deposits also of considerable economic value to the local economy. The Scottish Coal House of Water opencast mine site (BNG NS547142), shown as ‘Opencast Workings’ in figure 1.1 extends across the route of the River Nith and the Beoch Lane Burn, as well as several other minor tributaries. It is around operations on this site that the River has been diverted on two occasions.

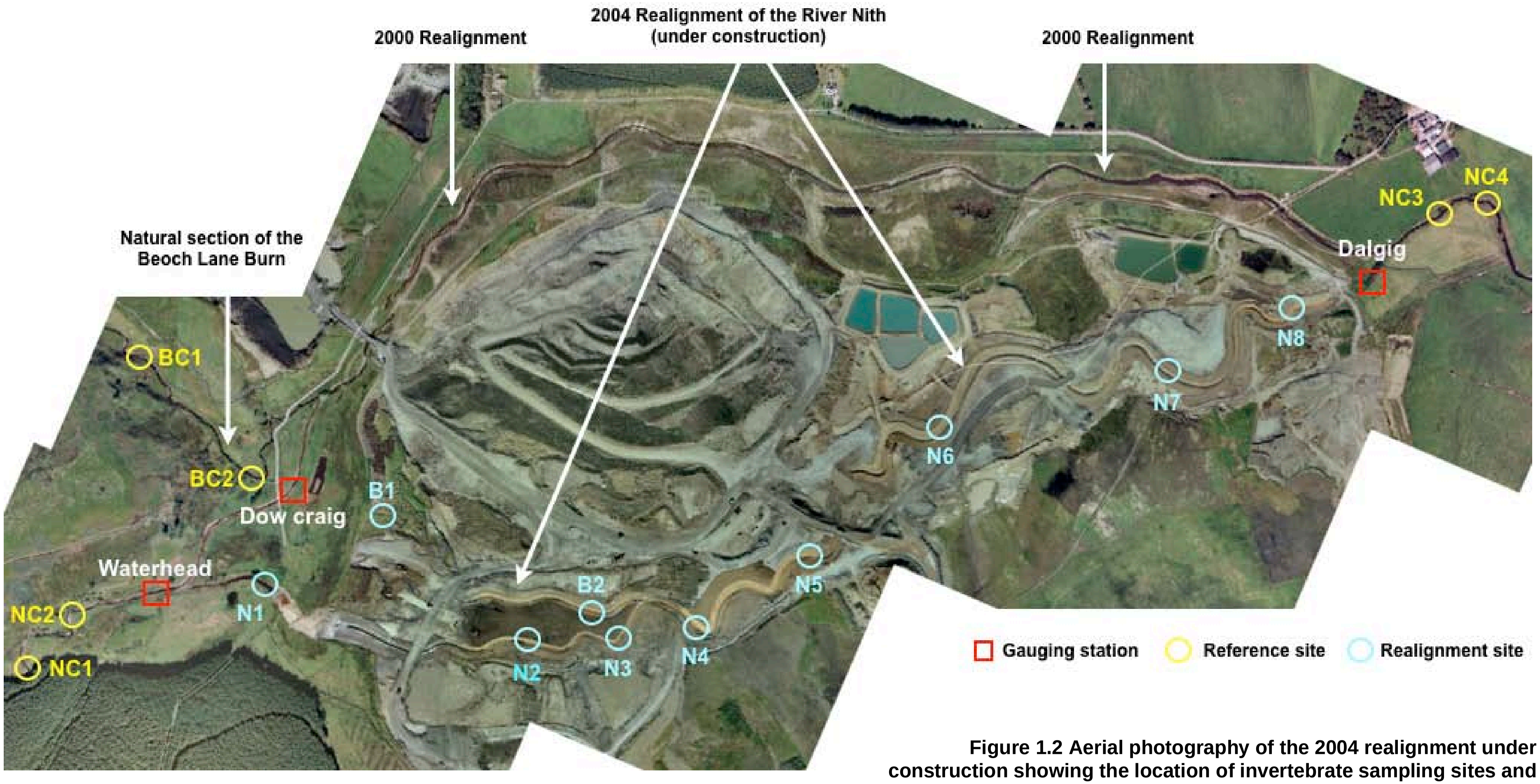


Figure 1.2 Aerial photography of the 2004 realignment under construction showing the location of invertebrate sampling sites and the three SEPA gauging stations relevant to the site

1.4 Hydrology

Three gauging stations are currently operational around the House of Water site (See figure 1.2). The Dalgig station records flow at the lower end of the diversion on the main stem of the River Nith. At this site an annual mean flow of 0.935m^{3}s^{-1} was recorded for 2006. This is less than the combined inflow recorded at the top of the site (0.668m^{3}s^{-1} at Waterhead + 0.669m^{3}s^{-1} at Dow Craig on the Nith and Beoch respectively). Instantaneous monthly maximum flows during 2006 recorded at Waterhead reached 54.82m^{3}s^{-1} (November) and 45.10m^{3}s^{-1} (December). These are considerably higher than readings at the downstream Dalgig gauging station for the same period (12.88m^{3}s^{-1} and 12.99m^{3}s^{-1} respectively). The upstream stations are newly installed and problems with calibration may explain the discrepancies. For this reason data has only been used from the established Dalgig monitoring station.

Despite the inconsistencies the 'flashy' hydrology of the catchment is clear, with water levels that rise sharply and recede almost as rapidly (figure 1.3). Maximum instantaneous flows are considerably higher than the mean daily flows, which rarely top 2m^{3}s^{-1} and are for much of the time less than 1m^{3}s^{-1} even during the winter months. The hydrology for the period of the study is illustrated in figure 1.4 which shows the water height recorded at the Dalgig station. It is worth noting that the largest flood during the three years of monitoring was within two weeks of connection and that the first winter was particularly flood-rich. Following this was a very dry first summer during which there were no notable flood events.

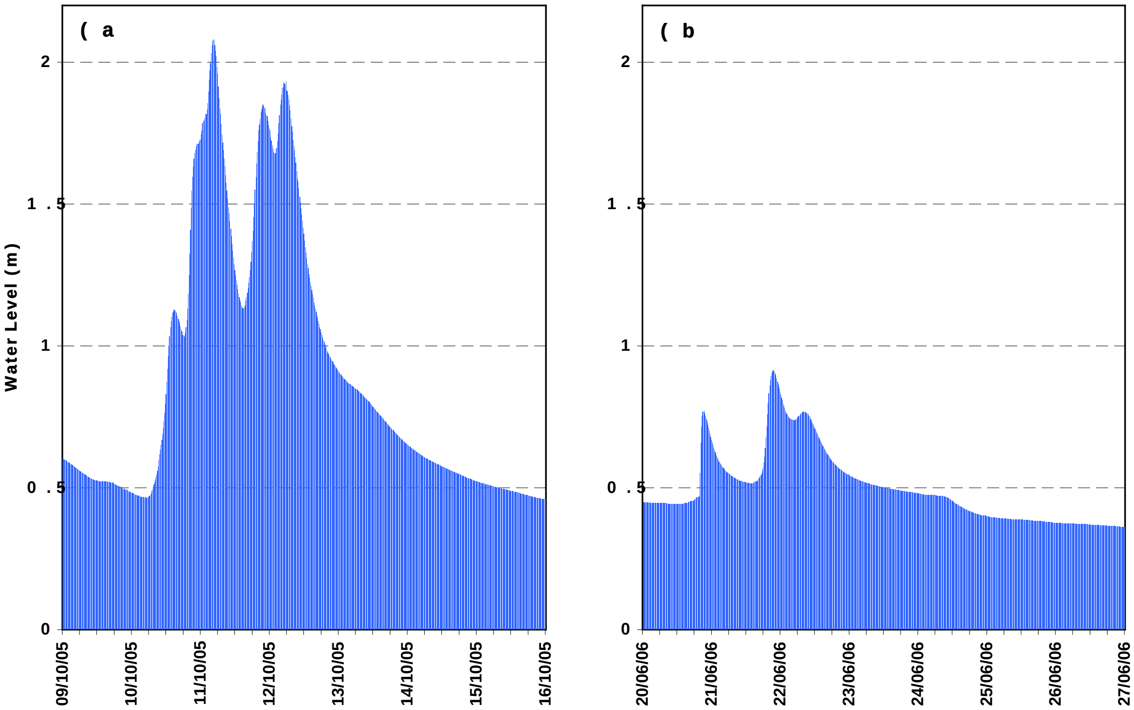


Figure 1.3: Hydrographs showing typical winter (a) and summer (b) high flows through the Nith diversion (Water levels recorded at the Dalgig gauging station)

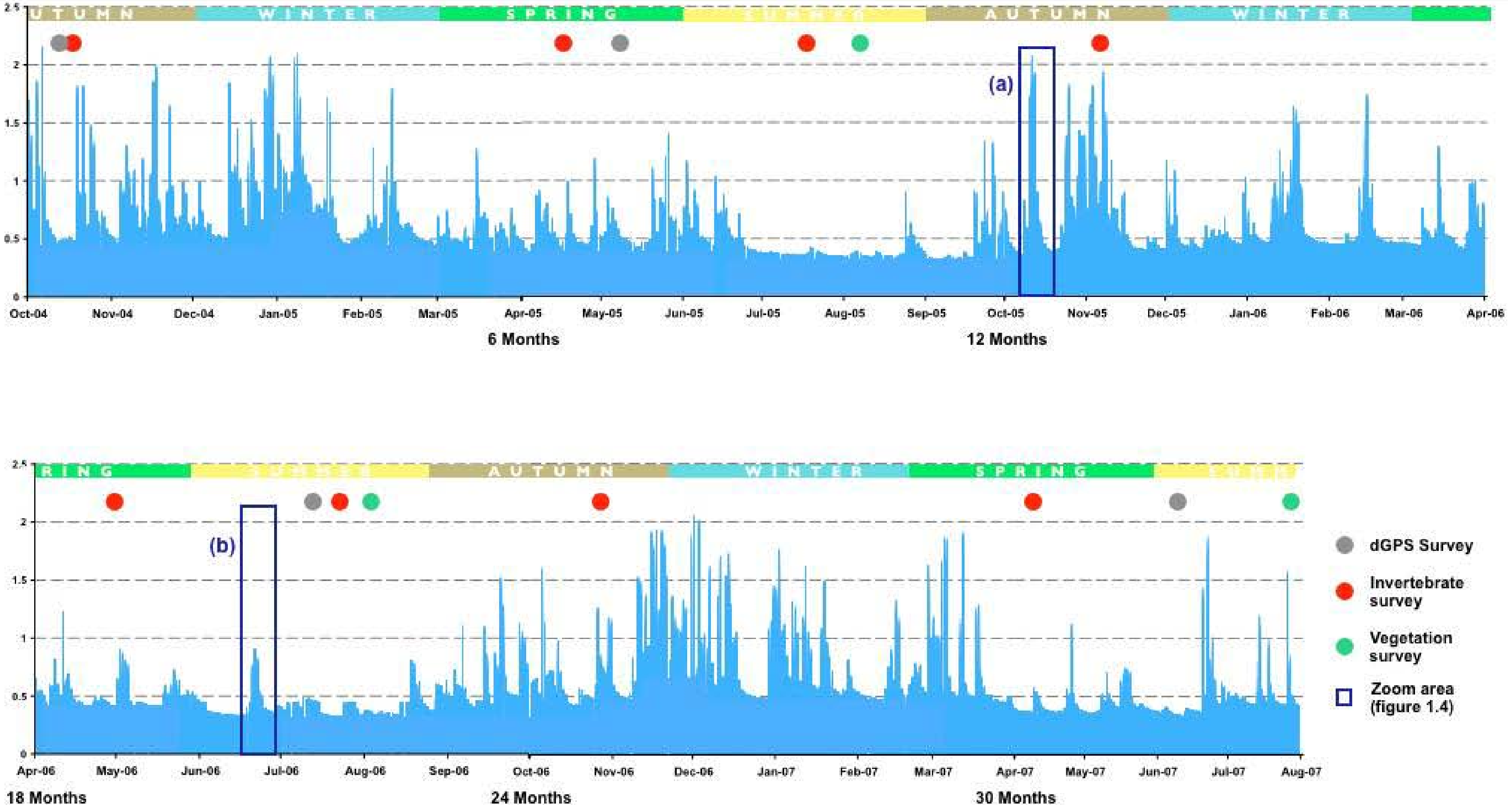


Figure 1.4: Hydrograph for the River Nith covering the study period starting at the beginning of October 2004 two weeks prior to connection into the catchment. Data shown is the water height recorded at the Dalgig SEPA gauging station Located at the lower end of the 2004 realignment

1.5 Condition

Mining operations have a long history of disrupting or altering catchment processes (Rowan, 2002; Wohl, 2006). Surface mining in particular often requires an element of landscape restoration and several examples of stream reinstatement exist (Mutz, 1998; Schulz & Wiegleb, 2000). In the case of the House of Water site the area to be restored had already been considerably altered. The morphology and riparian corridor of the original channel could be described as agriculturally degraded, flowing through semi-enriched pasture and providing a source of drinking water for cattle and sheep (plate 1.1). Macroinvertebrate communities in the pre-realigned river were monitored by SEPA in the springs of 1993 ’94 ’98 and ’99. It scored respectably on the BMWP index with average scores per taxa of 6.8, 6.5 6.7 and 7 respectively gaining it an A1 SEPA river quality score (Griffin, 2005).

Plate 1.1: Photographs of the original river and floodplain area prior to its diversion in 2000

1.6 Moving the river

The river was first rerouted in 2000 to allow excavation of coal lying under the riverbed. The diverted section of river starts at 220m OD and marks a significant change in character as the valley opens out. The gradient remains relatively steep at 1:134 for the top 1100 metres of the Nith and 1:154 for the 550m of channel conveying the Beoch Lane Burn. This then reduces to 1:271 marking a second change in character from a pool-riffle river type to a plane bed. From this point down the flow velocity is reduced and deeper glide type habitat dominates. Examples of the character of different sections can be seen in the photographs in section 7.1. The average slope over the length of the realignment recorded as the regression slope of height elevations measured along the channel centreline in autumn 2004 was 0.00559. A little less than the slope of 0.00568 proposed in the original designs (Halcrow, 2004) and the slope of the 2000 diversion (0.00562).

To enable the 2004 realignment an area previously excavated to a depth of 80m was backfilled in 2004 with compacted mine spoil. Along the realigned course a 50m wide 2m deep bed of impermeable clay provides a relatively impermeable barrier below the channel to prevent flow being lost to the highly permeable backfill material. The bed-layers of the sinuous planform were then cut into this layer. A near-trapezoidal shaped channel was constructed with bank gradients of between 1:2 and 1:3. Banks were protected with coir fibre geotextile along the length of the realigned sections to reduce excessive erosion and aid vegetation colonisation. A layer of sorted well-rounded gravels defined by the substrate composition of the natural riverbed was placed to a depth of 750mm with the lower 500mm being mechanically compacted.

The current realigned section of the Nith is approximately 2600m in length. At the top end there is a further 250m of relic channel from the original route of the river prior to the 2000 and another 100m at the downstream end. In addition to this, 700m of channel was constructed to connect the Beoch into the Nith at a point 500m down from the top end. Constructed channel bed widths ranged between 4m and 5m and had bankfull channel depths of between 1m and 1.5m. Cross-sectional area varies between 6.5m^2 and 13.13m^2. One of the design criteria for the 2004 realignment was that the channel convey a predicted mean annual flood of 22.4 m^{3}s^{-1} for the Nith and 9.4m^{3}s^{-1} for the Beoch Lane Burn (Halcrow, 2004). However, this was not achieved and the capacity of the constructed channel is estimated to be between 17.65m^{3}s^{-1} and 25.21m^{3}s^{-1} depending on bed roughness.

1.7 Existing Data

The 2000 diversion was monitored as part of a PhD (Griffin, 2005) from 2000 to 2003. The benthic invertebrate community was sampled using a kick-net based survey in the autumn of 2000, spring summer and autumn of 2001, and the spring and summer of 2002. Development of gravel bar vegetation was also recorded to either side of a 10m transect on gravel bars that formed along the new course of the river.

1.8 Science

Whilst channelization and straightening of comparable lengths to the Nith realignments are not uncommon in our long history of manipulating rivers (Chapter 2), the combination of the following factors make the realignments on the Nith unique in the UK.

i) The large scale of the project
ii) The requirement for novel construction techniques including new materials
iii) The attention given in the design to ensuring the continuation of geomorphological processes
iv) The river type and geography
v) The 'ground-zero' state over the length of the constructed channel

The design not only maintains continuity of sediment transport but also provides sediment sources and storage within the diversion channel itself. This takes the form of fluvial features including gravel bars, eroding cliffs, riffles and pools. These features are key in the development and maintenance of species richness and diversity in both benthic and riparian communities. They increase habitat heterogeneity and are created and maintained by fluvial processes (Wintle & Kirkpatrick, 2007; Hupp & Rinaldi, 2007), (see Chapter 3).

The act of diverting the same section of the Nith twice, provides additional scientific value. A major limitation to the contribution that large ecosystem projects can have to the field of restoration ecology is the lack of replication. Replication is fundamental in traditional approaches to scientific research and while statistical tools have been developed to avoid some of the issues related to limited replication, the two realignments will give the opportunity to investigate some of the colonisation patterns more robustly. Added to this, data collected from the natural channel in the summer of 2000 prior to the first realignment provides baseline data. Despite the interpretation of post-project results being greatly assisted by analysis of baseline data, sadly its collection is often neglected by restoration practitioners.

This PhD research arises out of a requirement for geomorphological and ecological monitoring of the River Nith and Beoch Lane Burn realignments. The research tracks the ecosystem development within the engineered channel. This not only provided the opportunity to observe changes but also to test some of the assumptions made in river restoration as well as some more scientific ecological models.

Specific terms and abbreviations will be defined at various stages. The following are widely used throughout the document. River Nith will be referred to as the 'Nith' and the Beoch Lane Burn as the 'Beoch'. 'Macroinvertebrates' will always refer to species that live for at least part of their life cycle on the benthic substrate of the river or river bottom structures such as macrophytes, boulders and woody debris.

CHAPTER 2

MOVING RIVERS

This chapter introduces the realignment of rivers. Starting with a brief history of diversions and historical presidents it then focuses on the UK experience of moving rivers and the motivations behind them, including specific Scottish examples. It talks about more modern approaches to rerouting and realignment of rivers and the desire to build functioning ecosystems. River restoration is introduced together with the science of restoration ecology and the contribution each is having to the design of modern river diversions. 'Best practice' in this area is discussed before moving on to the value of studying such projects including the important contributions that can be made to the field of restoration ecology.

Aims

Provide some historical background to the diversion of rivers
Summarise existing diversion projects in the UK
Highlight the change in approach to the rerouting of rivers
Discuss the importance and relevance of restoration ecology
Summarise best practice design and monitoring of ecosystem architecture projects

2.1 When was the first river ‘moved’

2.1.1 Pre-twentieth century

Man’s increasing ability to control rivers has allowed a change from the location of rivers determining much of urban and agricultural development to the modern day where it is ‘development’ that determines where and how exactly a river flows. These modern approaches to moving rivers have been influenced by, and have evolved from, a long history of river engineering: The way in which we move rivers now is very much a response to past interventions, both positive and negative, dating back several mellenia. ‘River engineering’ in the form of irrigation ditches was being carried out over 5,000 years ago in Egypt (Smith, 1971) and evidence still exists of a mud dam built near Memphis in Ancient Egypt. The Sadd el Katara Dam (Dam of the Pagans) was built around 4,900 years ago to supply irrigation water (McNeil, 2000) and is the oldest existing example of river engineering (Pender, 1998). Although simple in its design, it shows an early desire and ability to control and reroute the flow of water. Nine hundred years later a Pharaoh went one better, widening and deepening the natural channel to Lake Qarun for navigation, creating possibly the first ever canal (Overman, 1968).

The realisation that water could be used for power dates back at least 2000 years but mills could generally be located by the river requiring only modest modifications of the flow (McNeil, 2000). It was a need for drinking water in growing cities that provided the real motivation to divert water great distances. The earliest large-scale diversion was for the purpose of supply Rome with drinking water[1]. The first aqueduct was a modest 16km but as demand increased there was a need divert water from much greater distances (Overman, 1968). Whilst they are not river diversions as such, the diversion of 600,000 m^3d^{-1} and possibly considerably more (Hansen, 1983) would have had a serious ecological impact on affected streams and rivers. It was the Romans that brought river engineering of a significant scale to the UK. The Foss Dyke connected the River Trent to the River Witham and together with the Car Dyke from Lincoln to Peterborough represented nearly 100 miles of man-made channel (Hopkins & Brassley, 1982; Petts et al., 1989) and, although no match for the engineering of the Rome aqueducts, their construction nevertheless required an understanding of hydrological processes. It is likely that the ecosystem that must have developed was directly influenced by the engineering approach adopted. The Egyptians, Romans, Greeks, Ottomans all relied on the diversion of river water for services but a river had yet to be *moved*. Furthermore, despite attracting the attention of some of history’s greatest minds[2] no true appreciation of the ecosystem, and little appreciation even of the hydrological cycle, existed (Pender, 1998)

---

[1] It has been hypothesised that the levels of lead that the drinking water picked up from the lining of the aqueducts poisoned much of the aristocracy contributing to the downfall of the Roman Empire (Hansen, 1983) and the end of river engineering in Europe for hundreds of years.

[2] Among the many great minds that have contributed significantly to the science of hydrology, the more household names include Ecclesiastes, Michael Faraday, Rene Descartes, Aristotle, Edmund Halley, Clemens Herschel and Thomas Telford (Hershy, 1998)

Many of the large rivers in Europe were channelised during the 1900s (Buijse *et al*, 2002; Petts, 2002), an era when Tulla's statement "no stream or river needs more than one bed' was the mantra of many hydraulic engineers (Petts, 1989). Rivers were viewed as a resource for power, transport, irrigation and consumption and their floodplains, following construction of suitable flood defence structures, as 'prime land for agricultural, industrial or residential development' (Pender, 1998). As a result many rivers were channelised and embanked (Baattrup-Pedersen et al., 2005) resulting in considerable reduction of habitat diversity and area within the river channel (Tockner *et al*., 1998).

2.1.2 Twentieth century river engineering

The confidence gained through historical successes in controlling hydrology and development of the applied sciences of civil engineering, hydraulics and fluid mechanics from 1850 onwards (McNeil, 2000) paved the way for the extensive river engineering of the 20th century and the general belief that alterations to rivers had little consequence. The rate of river channelisation in the UK exploded during the middle of the twentieth century. In a survey of the period 1930 to 1980 Brooks *et al.* (1983) estimate that 8,500kms of river channel were heavily channelised equating to a density of 0.06kmkm^{-2}. The diversions of the early and mid twentieth century were constructed almost universally at considerable cost to the riparian and fluvial habitat they replaced. In England and Wales only 23% of Rivers can be classed near natural on the basis on their geomorphology (Sear et al., 1998). A similar figure has been reported for lowland rivers in Scotland (Werrity & Hoey, 2004), although the remote geography of much of Scotland may push the percentage pristine river channel closer to 50% for upland rivers (Werrity & Hoey, 2004). Whilst worldwide the supply of water for irrigation, drinking and power has remained a major driver, irrigated agriculture becoming the largest consumer (McNeil, 2000) pushing the diversion of water to extremes[3]; in the UK, prevention of flooding of both urban and agricultural land became perhaps the most important factor.

Larger diversions of UK rivers in the C20, often for power generation or drinking water supply, took the form of canals, as the most efficient, economic method of moving water from where it naturally flowed to where it was needed. Although they support a wide range of aquatic communities, which in some respects reflect slow flowing lowland rivers, canals lack the geomorphological variability that hydrological and hydraulic disturbance introduces to many streams and rivers. More common in Scotland is a pattern of relatively small localised works that together have a significant influence over morphology (Werrity & Hoey, 2004).

---

[3] Perhaps the most impressive sounding river diversion is the Great Man-made River in Libya. Constructed in 1986 with the help of an American Billionaire and capable of delivering an amount of water equivalent to 5% of the Niles flow. The river is in fact a major pipeline network running under the Sahara Desert from southern Libya to the costal regions around Tripoli and Benghazi (McNeil, 2000)

Agricultural intensification

Bringing land into intensive agricultural cultivation has required extensive channelisation of streams and rivers often involving straightening and embankment construction (Friberg et al., 1998; Buijse et al., 2002; Werrity & Hoey, 2004) with the River Nith being no exception (SEPA, 2007). It is probable that the majority of man-made channels in the UK and across the world are associated with agriculture. Artificial channels serve to both direct water to dry areas and drain areas of fertile wetland[4]. Post WWII, a drive for greater food security led to agricultural expansion and intensification, resulting in as much as 95% of the UK's wetlands being drained to exploit the fertile flat soils. This irrigation and draining of land together with the development of synthetic fertilisers allowed the intensification seen in the 20th century but also had a profound affect aquatic ecosystems (Harding *et al.*, 1998).

Urban development

The chaotic non-uniform nature of natural rivers did not fit with the ideals of 20th century cities. Rivers through urban areas have long been modified but the rediscovery of concrete at the beginning of the 20th century brought opportunities to confine and redirect the flow of rivers as never before. This was motivated by the desire to eliminate flooding as part of the new utilitarian ideals[5] and reached a peak in the 1960s. The post war era brought a public and political belief that planning would provide the solution to urban problems (Meller, 1997). Out, with the "inhumane conditions of the historic city" LeCorbusier (Norberg Schulz, 2000), went the chaotic un-predictable nature of rivers. The modernist utilitarian planning of the 60s demanded that river engineering focus on drainage. This was reflected in the design of channels at the time, for which hydraulic efficiency was a primary concern. Many miles of channel were replaced with networks of trapezoidal concrete channels in an attempt to reduce the risk of local flooding. Ironically[6], in the futuristic 'green-city' vision where health was idealised by the open spaces as much as the buildings, there was no room for ecological function. Floods were squeezed into channels just as communities were squeezed into tower blocks. In the most extreme examples, but widespread none the less, the riparian ecotones were transformed from complex gradients between aquatic and terrestrial habitats to 'edges' between a concrete river bed and a uniform carpet of green grass. In the countryside, agricultural grasses replaced riparian habitat; in the towns the grasses were ornamental. As the new channels had no capacity for the conveyance and storage of sediments the elimination of erosion also became an important objective. An example is the Whitecart catchment on the south side of Glasgow where the entire river network extending to streams a few

---

[4] The world's most extensive irrigation scheme is perhaps the Indus basin in the Punjab region of India and Pakistan where thousands of kilometres of channel divert water from the Indus to irrigate much of Bangladesh.

[5] Ebenezer Howard, inventor of the garden city is quoted as saying that, amongst many social ideals, "cities should be designed to express…the greater control over nature" and in his designs rivers were used to supply the water for water gardens and elaborate fountains (Beevers, 1988).

[6] One of the ecosystem services returned to society through the restoration of ecological function to urban rivers is said to be the promotion of human health and a sense of community togetherness (Findlay & Taylor, 2006).

feet across has been concreted in. Where rivers were not entirely replaced they were frequently embanked. As with channelisation, cost-benefit analyses of bank protection have, in the past, failed to recognise the ecosystem services and other benefits that the riparian zone provides (Piegay et al., 2005). Erosion control has long been seen more as an important aspect of stream management than a service. Bank stabilization on the River Nith, Paris, Canada, undertaken between 1971 and 1974 were credited with preventing serious flood damage in 1974 and 1975 (Gardener et al., 1978). Surveys of public opinion at the time showed a general popularity for such physical measures.

2.1.3 A new direction

Over the last few decades a number of factors have fundamentally changed the river engineering approaches used to tackle problems associated with the relationship that humans have with rivers.

i) Wider understanding of the ecosystem concept and its importance in sustaining river health.
ii) Change in approach to planning – less trust of large schemes
iii) The rise of an *environmental movement*.
iv) Growing appreciation of the importance of fluvial geomorphology.
v) The logic of a hydraulically efficient channel to expediate flows from one area only to cause problems elsewhere was questioned, as was the economic investment required to maintain heavily engineered structures[7].

By the 1980s ecosystem degradation in natural rivers became increasingly apparent, urban areas in particular having been hard hit by the loss of ecosystem services such as aesthetics and water conditioning. Channel engineering up to this point had been designed purely for the purposes of controlling channel erosion and flooding (Petts, 1989) and although degraded fish stocks may have motivated some physical rehabilitation for as much as 100 years (Harper *et al.*, 1998) restoration efforts were limited to instream habitat improvement devices for freshwater fisheries management (Swales & O’Hara, 1980). The failure of policy makers to understand ecological issues or communicate with experts in the field was highlighted by Biswas (1986) and is clearly seen in legislation passed during much the twentieth century. Since the first national legislation directed at the conservation of rivers – The Rivers Act of 1876 – nearly all legislation directed at rivers has been focused on the prevention of river pollution (Hershy, 1998) ignoring morphology and ecosystem concepts. This meant that the design of man-made channels could continue unchallenged. By the end of the 1980s the problem had been acknowledged by the scientific community and some effort made to address it (e.g. Petts 1989; and others). Environmental requirements started to play an increasingly important role in project planning and operation (Petts, 1989) and following the Brundtland Commission report of 1987 environmental objectives began to be integrated into social and economic

---

[7] Maintenance of river engineering works on the Mississippi are estimated to be approximately $180million per annum (Pender, 1998)

policies (Kitchen, 1997) and legislation[8] (Chave, 2001; McDonald *et al.*, 2004). One of the principles of sustainable development agreed at the Rio conference in 1992 was 'think globally, act locally' and adoption of this in Local Agenda 21 heralded the possibility for environmental projects to be developed and implemented at a local level. At this time, with the rise of sustainability as a concept, many began to focus on the degree to which ecosystems had been damaged. Naiman et al., (1993) quantified the disappearance of the riparian corridor area of North America and Europe at a staggering 80%. Nilsson (1992) goes further, suggesting that in Europe, all riverbanks have been subjected to some level and form of human perturbation. Work presented at 'River Restoration '96' showed a clear appreciation for the site and complexity of task ahead for restoration, but also a willingness to embrace new tools and develop a sound scientific basis (Boon, 1996). Of greatest concern is the structural degradation of stream morphology (Spänhoff & Arle, 2007). According to McDonald *et al.*, (2004) the growing awareness of river ecosystem decline was accompanied with an acknowledgment that rivers have multiple users, the needs of whom must be addressed by river management options. Restoration of rivers began to be talked about as a possibility. This can be contrasted with the important issues of a decade earlier when the idea was rarely mentioned. For example, in interviews conducted to gauge public perceptions of flood hazards, adjustments and mitigation measures diverting flow was mentioned but not in the context of restoring habitat (Gardener *et al.*, 1978).

Some meaningful progress towards the restoration of natural ecosystems was being investigated outside the UK. The Netherlands was already experimenting with the considerable power of natural recovery in river ecosystems (Swales, 1989). This approach; making use of 'undirected dynamics of abiotic processes', has been proposed as part of the solution for the restoration of opencast mining areas in Germany (Mutz, 1998). Whilst these solutions to degraded habitat raised the profile of restoration projects, there was still a desire, in the UK at least, for diversions and realignments e.g. to allow road widening projects (Gilvear and Bradley, 1997). Techniques developed for instream habitat improvement for fisheries management were applied to add a 'natural element' to many diversions to assist or promote natural processes, although there is little evidence that these were any more than aesthetic (e.g. Pretty et al., 2003)[9]. More recently research has proposed improvements to design such as the positioning of deflectors (Biron *et al.*, 2004). Harrison *et al.* (2004) found small-scale within channel structures brought little benefit to fish numbers in 13 such projects on lowland UK rivers. Riffle placement, which has been a popular tool for the rehabilitation of canalised rivers (Harper *et al.*, 1998; Clarke et al., 2003) have been shown to rapidly silt up or wash away when they are installed without due consideration for the hydraulic and geomorphological conditions (Harper *et al.*, 1998).

---

[8] European Union Habitats Directive 92/43/EEC (1992), UK Conservation regulations (1994)

[9] Although Ernst Haekle may have appreciated the aesthetic improvements I suspect he would also have predicted the ephemeral nature of the efforts. Haeckel would have been a fan of modern diversions for their social value. He hoped that "via awareness of, and interaction with, the natural world, people [would] become aware of the interconnection of all life on Earth and [would] lead to a betterment of these unintended consequences of modern society" (Gross, 2007). The unintended consequences being "problems arising out of the morphing of nature that lead to ugliness, degradation and destruction".

Consideration of restoration scale is also important. Lepori *et al.* (2005) found boulder placement and the use of deflectors restored habitat at the wrong scale for the target species. But perhaps of greater concern, with the exception of these few examples, the effectiveness of many stream restoration techniques remains at best uncertain and in many cases unknown (Alexander & Allan, 2006), rarely being rigorously assessed (Woolsey et al., 2007). This adds considerable complexity to the consideration of management options for stream conservation (Linke *et al.*, 2007)

Despite dressing them up, rivers were still essentially being diverted into trapezoidal channels. It was several years before the first demonstrations of the real potential for ecologically sensitive river engineering appeared. Once a realisation of the importance and value of natural, functioning rivers developed rivers began to be moved rather than simply having their flow diverted into man-made channels.

Diversions for restoration

Currently river diversion projects can be divided into two groups. Those where the prime motivation is the creation of a natural river of improved condition and those where the current alignment impedes land development. Historically the principal objectives of the latter, conveyance of flow and the permanence of the channel, have been achieved easily through hard engineering of the channel form and course. While economic and social factors/circumstances still necessitate the diversion of streams and rivers[10], a new set of rules apply. Flood management strategies that retain ecological value are being adopted in favour of more traditional *'heighten and harden'* approaches to flood control, such as concrete channelisation (Adams *et al.*, 2004; Tomkins & Kondolf, 2007).

In the UK the year 1995 produced examples of realignments of significant length motivated both by development and by restoration. In the first example motorway construction was going to require the diversion of 12 reaches on the Evan Water (Lanarksire) into straight trapezoidal channels. The eventual adoption, in 1993, of a more environmentally acceptable design marked a change in policy by the Scottish office and in 1995 river reaches totalling approximately 2700m in length were realigned into sinuous gravel bed channels. Post project monitoring showed rapid development of fluvial features such as gravel bars and pool-riffle forms (Gilvear & Bradley, 1997). The second example marks the first significant use of river realignment as a serious tool for restoration of UK rivers. The EU Life funded projects on the River Skerne and Cole realigned considerable lengths of channel. Re-meandering work was carried out on 2km of the River Cole in 1995. The project involved both new channel creation and some reshaping of existing channel sections, extending the length to 2.16km

---

[10] Stream diversions have been proposed for a range of reasons. Many of the historical reasons such as improving navigation, alleviating flooding and topping up canals (Gardiner, 1978) are less relevant in the modern UK. However many rivers are still being diverted to allow development of urban areas and infrastructure deemed more important than leaving the river alone such as airport expansion and motorways (ref). However, often where the river is already degraded this development can be an opportunity to source funds for channel restoration.

and raising bed levels by up to a metre. Designs were based on sound geomorphological and ecological principles and extensive post project monitoring was conducted (Biggs *et al.*, 1998). These projects together with other rehabilitation projects have been effective in reshaping the approaches of engineers, managers and politicians. However, the majority of restoration projects largely continued to progress on an ad-hoc, site and situation specific basis, possibly because of the inevitable bureaucratic complexities of large-scale projects (Holmes & Nielsen, 1998) and the sporadic, evanescent nature of funding (RRC Conference, 1997). There has been little development of general principles that would allow the transfer of methodologies from one situation to another (Hobbs & Norton, 1996). Only a fraction of the restoration projects taking place annually benefited from the combined efforts of practitioners and scientists (Michener, 1997) partly because of the considerable effort required to develop interdisciplinary partnerships (Holmes & Nielson, 1998)

By the turn of the century the drive to restore both terrestrial and freshwater environments was rapidly growing (Lake 2001). Reasons for restoration are profuse and various, they include efforts to counter the detrimental effects of flow regulation, over grazing, mining and urbanisation (Follstad shah *et al.*, 2007), and where degraded rivers cross international borders there may even be a role for restoration as a solution in political disputes (Asaf *et al.*, 2007). In a survey of restoration projects in the south west of the US the commonest motivations were riparian management, water quality management, instream habitat improvement and flow modification (Follstad shah et al., 2007). Some advocated a move from small, isolated, opportunistic habitat enhancement to large scale restoration of channel morphology (Sear *et al.*, 1998) while others saw a benefit of multiple small-scale projects co-ordinated across a catchment (Harper et al., 1999). Either way channel realignments were, and still are very much the exception (Tompkins & Kondolf, 2007) in part because the case for rivers is still being argued in many areas (Findlay & Taylor, 2006).

There are few examples of a truly holistic approach to restoration where it can be said that the entire river and floodplain have been viewed as a single connected system. Restoration of the Skjern (Denmark) recreated the pre-channelised physical habitat including connections between the river and floodplain (Pedersen *et al.*, 2007). The project was vast, requiring the excavation of 40km of river channel and the removal of 2.7 million cubic metres of soil (Pedersen *et al.*, 2007). Such a holistic valley based approach paid off. Reports from initial monitoring reveal changes in vegetation to more diverse natural communities and a decrease in domestic, cultivated grasses and traditional weeds (Pedersen *et al.*, 2007). An internationally important community of resident and migratory birds have developed and a number of rare and endangered species are now found in the area (Pedersen *et al.*, 2007).

2.1.4 The Scottish perspective

Table 2.1: Table summarising recent diversions in Scotland, designed to varying extents on geomorphological principles of sediment transport and storage.

Name (Location)	*Date*	*Length*	*Motivation*	*Appraisal*
River Nith (Ayrshire)	2000	2.8km	Diversion around opencast mining	Detailed monitoring and PhD
Evan Water (Lanarkshire)	1995	All < 1km	Make space for road widening	Geomorphology study and MSc
Niddry Burn (Edinburgh	Proposed	2km	Housing development/restoration	None
Abby burn (Lanarkshire)	2004	300m	Reinstatement following quarrying	Some invertebrate data collected
Gogar Burn (Edinburgh)	2003	200m	Make space for office development	None
Gogar Burn (Edinburgh)	Proposed	3.2km	Airport expansion	None planned
Eye water	1997	<1km	Highway widening	
River Nith (Ayrshire)	2004	2.9km	Re-diversion around opencast mining	Detailed monitoring and PhD
River Nith (Ayrshire)	Proposed	2x800m	Diversion around opencast mining	Proposed
Ponesk Burn (Ayrshire)	Proposed	1.2km	Diversion around coal mining	
Small Burn (Ayrshire)	2009	<1 km	Diversion around coal mining	None
Lane Burn (Ayrshire)	Proposed	1km	Diversion around opencast mining	Proposed
Docharty Burn (Ross-shire)	2003	2x200m	Make space for road widening	None

As elsewhere, many Scottish rivers have been heavily modified over the past 200 years (Werritty & Hoey, 2004). Modifications include:

i) Impoundment and flow regulation for power generation
ii) Diversion of urban rivers into straightened concrete channels for flood alleviation
iii) Straightening and embankment to expand agricultural cultivation

Previously diversion options to allow development included concrete lined channels and many examples exist throughout Scottish cities. Many miles of channel are concreted in across the Whitehart catchment in southeast Glasgow. Sections of the Gogar burn have been canalised to allow runway construction at Edinburgh Airport. A more sustainable approach is now required in response to new social and legal frameworks of acceptable approaches to river engineering. Over the last 10 years there have been a number of river diversions in Scotland (Table 2.1) that have been designed with this ‘ecologically sensitive’ approach. Consideration of geomorphological processes has been key to achieving this. Despite this, the opportunity to draw on the Scottish experience in a scientifically rigorous manor (e.g. Gilvear & Bradley, 1997) has only rarely been realised. The majority of UK river diversions where restoration has been a goal have been ‘demonstration’ projects on low gradient lowland rivers and much is still unknown about the processes of colonisation and development (Malmqvist, 2002). Uniquely many of the Scottish diversions are on high-energy upland streams with very different ecosystems. It has been hypothesised that frequent disturbance from spate flows may provide for more rapid recovery because of the evolutionary pressures on biota to recolonise disturbed areas (Griffin, 2005).

2.2 River restoration ecology

The differentiation is made in this text between *ecological restoration* as the practice of restoring ecological systems, and *restoration ecology* as the science that explains the associated patterns and processes. That the study of restored habitats has become formally recognised as a scientific discipline in its own right (Andel & Aronson, 2006) is testament to the importance of ecological restoration to modern society.

2.2.1 Definitions

There is a general consensus around what is meant by ecological restoration and the importance of what it aims to achieve. Definitions follow the same themes and share terminology but vary in the extent to which they are applied. In the strictest sense, Ecological restoration is ‘an attempt to return a system to some historical state’ (Palmer *et al*., 2006) creating an exact replica of the habitat present prior to being degraded or destroyed. This is often difficult or impossible to achieve and questions have been raised as to whether it is a necessary requirement (Pfadenhauer, 2001) or even appropriate. Restoration *sensu stricto* is rarely feasible and use of this definition may be counter productive by setting unrealistic expectations.

The view of restoration as the process of ‘inducing and assisting abiotic and biotic components of an environment to recover to a pre-existing state’ (Lake, 2001), “something approaching an original state” (Harper *et al*., 1998) or a “quasi-natural” state (Mutz, 1998) reflects the projects of the 90s; generally isolated enhancements to degraded stream ecosystems (Sear, 1998). However, even this relaxed definition seems rather narrow in the light more recent relatively ambitious large scale ‘restoration projects’ that have rebuilt entire rivers from scratch, such as that of the lower reaches of the River Skjern (Pedersen *et al*. 2007). It is understandable that with the changes in scientific thinking and nature conservation policy over the last few decades, terminology has been allowed to evolve. The Society for Ecological Restoration re-evaluated its definition of restoration at least five times in the 90s (Palmer *et al*., 1997) to keep pace with advances in principles and practice. With the ever-increasing ambitions of practitioners to return extensively damaged ecosystems to a sustainable condition and to maximise ecosystem services within extensively altered watersheds, historic-states are unlikely to be the most appropriate option across much of the industrialised world, especially with the potential for climate change to alter environmental conditions (Choi, 2007; Lake *et al*., 2007; Hobbs & Cramer, 2008). At the same time any definition must cover both large-scale projects as well as more traditional small-scale projects and simple management options that remove perturbations, which continue to play an important role in restoration of rivers (Palmer *et al*., 2006). Perhaps a more useful definition focuses on the process e.g. Halle (2007); the directed and accelerated succession [of degraded systems] by active management. The shear diversity of approaches and goals of projects at a variety of scales will continue to require a broad definition of the term restoration. Any definition must also be understandable to the large range of people likely to be involved in a restoration project (figure 2.1).

It may be that ecologists will have to pin down the definitions of other more scientific terminology to improve communication for research and debate.

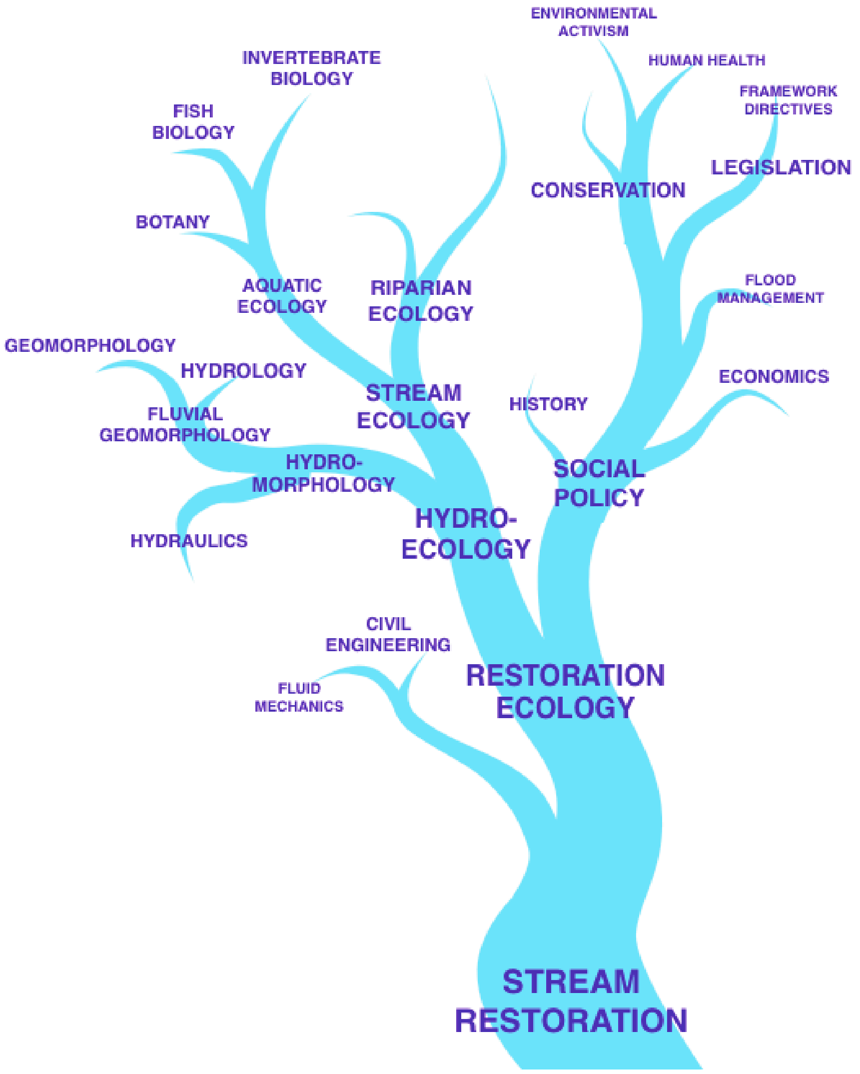


Figure 2.1: indicating the diversity of aspects to stream restoration when channel construction is used and highlighting the importance of a multidisciplinary approach

Stream restoration is multidisciplinary (Boon, 1998; Binder, 2006). This is particularly true when projects involve the construction of new channel. The diagram in figure 2.1 shows how a relatively specialised science such as restoration ecology and its application to ecological restoration in fact draws from a very broad scientific knowledge base. Calls for technical working groups behind projects to reflect this breadth are becoming increasingly common (Pedrioli, 2006; Lake, 2007) and are slowly being adopted into practice.

2.2.2 Relevance

Restoration, specifically that of streams and rivers, is discussed in depth for a number of reasons. Firstly, although the motivations behind river restoration projects and river realignment projects may differ, they generally have very similar aims. In most cases restoration projects have ecological (Lake 2001) and occasionally social (aesthetic and recreational) goals (Holl & Howarth, 2000; Hopfensperger et al., 2007). These include increasing biodiversity, enhancing water retention capacity and reducing degrading forces such as soil erosion (Pfadenhauer, 2001). The goals both provide the motivation for the project and are the objectives. Whilst the multiplicitus motivations for river realignment projects provide different reasons for moving rivers, the ecological and to some extent social goals very similar to those for restoration. The ultimate aim is implicit within all river projects: To create a self-supporting ecosystem, resilient to perturbation without further assistance (Ruiz-Jaen & Aide, 2005). Further to this, the recreation of landscapes following large-scale operations such as mining is a costly enterprise (Holmes & Nielsen, 1998). The incorporation of long-term goals for comprehensive ecological enhancement has been suggested as a means of adding value (Mutz, 1998) although there is considerable uncertainty in pricing the socio-economic benefits (Pederson et al., 2007).

Secondly, rerouting rivers into man-made channels is increasingly practised as a method for river habitat restoration. Although still a small proportion of the projects to date, for example only 38 in a database 4,000 Californian projects (NRRSS) (Tompkins and Kondolf, 2007), many of the larger restoration projects implemented recently have involved channel construction (e.g. Gurnell *et al*., 1998; Pedersen, 2007). The restoration of rivers has the potential to contribute significantly to the conservation of habitats and specific species (Tockner et al., 1998). This potential extends to many realignment projects regardless of the initial motivation.

Next, restoration ecology extends beyond restoration projects *per se*. The strands of ecology central to restoration ecology - succession, population dynamics (Palmer et al., 1997) - are equally relevant to realignment projects. As a result, understanding rates and patterns of colonisation observed in the realignment of the Nith (the succession processes involved in ecosystem development) by informing the science of restoration will help guide and inform many types of river project.

Finally, far more discussion exists around the evaluation of restoration projects than projects driven by the need to divert a stream course[11] This has meant many of the principles upon which sound, ecologically sensitive, river engineering is based originate in the field of restoration ecology. Likewise, this project has the potential to contribute significantly to the field of restoration ecology.

---

[11] The complex relationship between science and restoration has made for some interesting and occasionally heated debate (e.g. Cabin, 2007a; Giardina *et al.*, 2007; Cabin, 2007b). Nevertheless, that science has plenty to offer to ecological restoration is generally accepted

2.3 The Art of moving rivers

2.3.1 The state of the art

Concern that restoration has been viewed as an ‘Art’ rather than a ‘Science’ by many resource managers, funding agencies and policy makers as well as many scientists (Michener 1997), is misplaced. In many cases, where the natural river form is unknown, an element of creativity will inevitably have a role to play in the design process and the unpredictable nature of rivers means there is always a requirement for an adaptive element to any project. Indeed, Ernst Haeckel regarded the copying of nature as the highest form of artistic and cultural expression (Gross, 2007). Furthermore, hard engineering to control and direct the flow of rivers is also considered by some to be an *Art* (Pender, 1998). The ‘Art view’ is not, as Michener asserts, the reason why science has failed to keep pace with the ever increasing ambitions of practitioners. Artistic expression has pushed the boundaries of many sciences as is evident in many *architectural* buildings and *artificial* landscapes. The real concern lies in the gap (Palmer et al., 1997) gulf (McDonald *et al.*, 2004) discrepancy (Pfadenhauer, 2001) chasm (Michener, 1997) between theory and practice. The need and social appetite for restoration projects has driven development of realignment as a tool a requirement for projects rather than scientific know how and science has lagged behind for a number of reasons. Firstly, despite the growing number of restoration projects and the possibility that physical rehabilitation to benefit fish has been ongoing for many decades (Harper *et al.*, 1998), the relatively recent appearance of “River restoration” in the scientific literature suggests a delayed interest from the scientific community (figure 2.2). Although the exponential increase since 1990 does match reports a reported increase in the number of river restoration projects instigated in the US southwest during the same period (Follstad shah *et al.*, 2007).

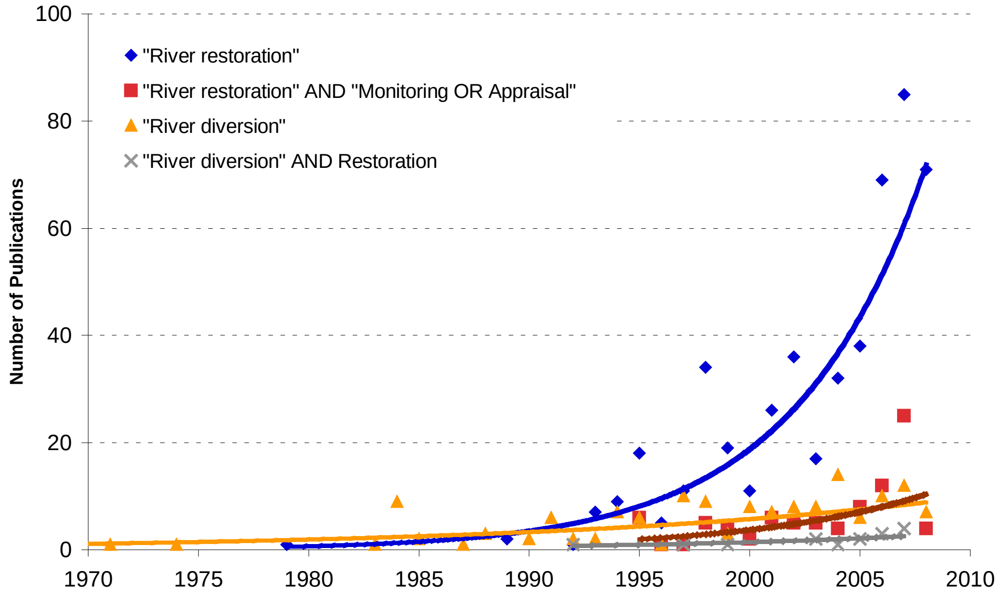


Figure 2.1. Graph showing the increase in number of publications containing key words relating to river restoration and diversions. Results obtained using the advanced search feature of the Web of Science database.

Secondly, the opportunities presented potential for scientific research, when ecological restoration ventures beyond the realms of current understanding restoration ecology, are frequently missed because of poor communication or an unwillingness to dedicate resources (Downs & Kondolf, 2002). This has perhaps been reinforced by widely held opinions that while up-to-date scientific theory is important for guiding projects, money is better spent on ecological restoration projects (RRC Conference, 2007).

Factors that complicate the design process (adapted from Johnson & Brown, 2001) include:

i) Objectives are often vague – making it difficult to develop specific solutions
ii) Interference in the design such as bridges and utilities
iii) Degrading pressures may be complex
iv) Requirement for interdisciplinary working – communication issues
v) Geomorphic and ecological responses can be difficult to predict

Where science is unable to inform project design, there are number of potential pitfalls.

i) In appropriate and costly techniques may be employed
ii) The ecosystem may develop towards an unexpected endpoint
iii) Excessive changes in planform or gradient may put local infrastructures at risk

2.3.2 Current best practice in river channel construction

The conveyance of flow and the permanence of the course are no longer the sole objectives of manmade channels. The potential of river diversions to seriously disrupt natural ecosystems, sometimes for only limited economic benefit (Karwacki, 2003) has led to an overall change in approach to channel design and construction. Best practice design needs to be economically and environmentally sustainable. Theoretical discussion of restoration divides it into two approaches – top-down strategy based and bottom-up tactic driven (Landers, 1997), although in best-practise restoration it is not an either/or situation. Top-down elements such as a clear overall guiding image (Palmer *et al.*, 2005) and a comprehensive design based on sound scientific principals are combined with site specific adaptive management and maximising the use of existing site landscape features. The same approach is also suited to the design of realignments.

Recent design has moved towards restoration of valued physical processes rather than the creation of features. This eco-hydromorphic approach, consistent with the Water Framework directive, is important if a sustainable system without on going management issues and costs is to be achieved (Rheinhardt *et al.*, 1999; Clarke *et al.*, 2003). Harper *et al.* (1998) emphasise the importance of basing river restoration projects on first principles. A study of artificial riffle placement as a rehabilitation technique on Harper's Brook, a tributary of the River Nene, UK, revealed only riffles placed according to first principles produced the desired geomorphological and ecological changes. Man-made channels will only be sustainable if undertaken within a “process driven” and “strategic framework” (Clarke *et al.*, 2003).

Re-meandering

Reconstruction of meanders can be a very effective strategy for conservation of lotic macroinvertebrates of lowland rivers (Nakano & Nakamura, 2006). It offers a range of immediate environmental benefits including increased aesthetic value, greater streambed area (over the same direct distance) and increased flood storage capacity (Friberg *et al.*, 1998)

Longitudinal connection

Maintaining connectivity through the length of a man-made channel is an element of best practice. In some projects this has included removal of barriers to hydrological and ecological connectivity, such as dams (Hart *et al.*, 2002). The appreciation for up- and down-stream linkages comes in part from the River Continuum Concept proposed by Vannote *et al.* (1980) as a model for energy transfer and population dynamics along natural river systems. However there has been little research into the role that longitudinal pathways pay in the recovery habitats following restoration or creation. One concern is that removing barriers may lead to the spread of non-native species (Rahel, 2007). There have been some reports where restoration of habitat processes has favoured the regeneration of regionally native species (RRC conference, 2007) but better understanding of the processes involved could be used to ensure that this is the case.

Lateral connection

Floodplain inundation is beginning to become accepted as important in stream restoration. The use of natural processes driven by the hydrological connection between river and floodplain to restore the structure and function on wetland and floodplain habitat has been used in numerous restoration projects. Surveys suggest that projects in Northern California designed to inundate the floodplain by flows with return periods greater than 1-5 years (Tompkins & Kondolf, 2007) have typically been for the purposes of restoration, however the connection is likely to be important to the natural functioning of any man-made channel. It has been emphasised by the Flood Pulse Concept (Tockner et al., 2000). Tompkins & Kondolf (2007) found that out of five relatively established channel restoration projects (surveyed 5-20 years after construction) two of the projects needed follow-up repairs. They conject that the wider stream corridors of the remaining three projects enabled them to be more self-sustaining.

Uncertainty

Uncertainty is an important aspect of any channel modification which should be incorporated in to modification designs. The many sources of uncertainty from natural randomness to uncertainty at the modelling stage can result in deviations from the predicted habitat that is created or develops (Johnson & Brown, 2001; Sear & Darby, 2008). This will obviously have the potential to affect ecosystem development and there is a scarcity of research in this area. Best practice design incorporates the potential for uncertainty into the system without necessarily constraining channel processes. Whilst adaptive management techniques allow informed decisions that embrace the uncertainty, they need to be supported by scientific evidence of the likely ecological effects whenever possible. Trial and error should be avoided in large scale ecosystem projects (Hobbs & Harris, 2001; Suding et al., 2004)

Stability

Although methodologies developed for designing ‘stable’ channels (e.g. Hey, 2006) may be advocated for channel construction projects in urban areas, scientific best practice would be to provide the sufficient space to allow a natural dynamic system to evolve.

Use of woody debris

The benefit of woody debris to river habitat is supported extensive scientific research and is widely accepted to be a powerful restoration tool. However, to be sustainable, it requires the planting of riparian woodland as has been proposed by a number of authors (e.g. Collins & Montgomery, 2002).

The rate of development

Best practice aims to minimise the damage and maximise the potential for recovery or development within man-made channels. Rate and extent of development is obviously very important in this respect, yet little scientific evidence exists. This is one of the key areas to be addressed by this research

2.4 The opportunity for scientific investigation

2.4.1 <u>Learning from river diversions</u>

Learning lessons from ecosystem engineering projects to help in the design of future rivers is a widely called for outcome of appraisal work (e.g. Tomkins & Kondolf, 2007; Follstad shah *et al.*, 2007) and despite spending vast amounts on river restoration, very little has been learnt in terms of ecological outcomes (Lake, 2007) with few projects being subject to any formal evaluation (Michener, 1997; Kondolf, 1998). This has contributed to an inadequate scientific basis in many restoration projects (Harper *et al.*, 1998). Evaluation has typically failed to report either the degree to which function is restored, or how closely projects approximate a natural ecosystem (Rheinhardt *et al.*, 1999). In a recent survey of stream restoration project managers in the US Northwest although 23% intend to monitor projects for more than 10 years, 34% of projects had insufficient monitoring for even basic evaluation project success (Rumps *et al.*, 2007). While this is an improvement compared with a decade ago, there are still many projects without any formal monitoring based on specific ecological indicators (Alexander & Allan, 2007). Furthermore post project appraisals typically lack sufficient replication to allow results to meaningfully contribute to learning and adaptive management for future projects (Tomkins & Kondolf, 2007). The majority of interviewees in a survey by Follstad shah *et al.* (2007) based project evaluation on observation of biota or public reaction rather than analysis of field data, and in a survey of stream restoration projects in Victoria, Australia, only 10–14% were monitored in any meaningful way (Lake, 2007).

The value of scientific learning from the outcomes of restoration projects is clear. By advancing understanding of ecosystem structure and functioning, required if restoration science is to catch up with enlightened application (Michener, 1997; Lake 2001), it will be possible to better inform future projects, increasing success and reducing costs. This includes providing feedback on how the ecosystem responds to various restoration techniques, identifying the best techniques in given circumstances and developing novel restoration techniques (Francis & Hoggart, 2008). The study of realignment projects is particularly valuable because of the ‘ground zero’ starting point, which reduces the number of confounding variables. Landscape scale projects provide the opportunity to contribute to the River Sciences in a way that is not possible in laboratory based simulations. The scale is beyond that of any flume and results are directly applicable to similar realignment projects.

This potential has been demonstrated by the handful of projects that have been investigated scientifically. Insights have been given into the behaviour of step-pool channels (Chin *et al.*, 2009), (Friberg *et al.*, 1998, Biggs *et al.*, 1998) and Plane-riffle channels (Gilvear & Bradley, 1997). Many of the advances are in applied area of the science. Study of the Nith is a case in point where lessons learnt from the 2000 realignment were used to guide the design of the 2004 realignment and the results of this study have been used in the design of proposed future realignments.

2.4.2 Measuring success

Monitoring is low down on the list of priorities for many restoration projects (Lake et al., 2007). This seems absurd given the amount of money that is spent on it annually[12]. It is nevertheless an essential component to any project that reroutes river courses such as restoration or channel realignment (Johnson & Brown, 2001; Edgar *et al.*, 2001). “Monitoring has to be undertaken so that we can learn from what we do” (Lake et al., 2007). A consensus on what constitutes success in restoration and other ecosystem architecture type projects is only now beginning to be reached (Giller, 2005; Palmer *et al.*, 2005). Ecology is central to the new ‘success standards’, and pre- and post-project monitoring have a major role to play (Palmer *et al.*, 2005). It is only possible gain meaningful insights into restoration ecology if the tools exist to properly investigate and monitor development and if projects are carefully designed and implemented (Wohl, 2005; Roni et al., 2008).

A list of desirable ecosystem attributes presented by the Society for Ecological Restoration International includes suggestions that are equally applicable to realignment projects:

i) Characteristic species assemblages similar to reference sites.
ii) Indigenous species especially of high conservation value
iii) Supports all the functional groups necessary for continued ecosystem development.
iv) Sustainable and diverse physical environment and habitat
v) Evidence of normal ecosystem function/ absence of signs of dysfunction.
vi) Resilience to normal, periodic stress events.
vii) Self-sustaining.

The majority of these can be said to be characteristics of a functioning stream ecosystem and monitoring the development of them may tell us lots about the patterns and processes important in restoration ecology. Such a holistic approach requires the application of a range of sciences in a careful and considered way. For instance compatibility issues may arise. Many indices developed for investigating geomorphological form operate at a different spatial scale to methods of collecting biological data. New indices designed to address this (such as those proposed by Bartley & Rutherfurd, 2005) have yet to be fully tested.

Macroinvertebrates are frequently used as indicators of ecosystem development and functioning in realignment type projects (e.g. Biggs *et al.*, 1998; Nakano & Nakamura, 2007), and are widely recommended as indicators of biodiversity (Bilton *et al.*, 2006; Dziock et al., 2006). They have numerous advantages over fish and plants. Although the use of target fish communities has also been recommended (Bain & Meixler, 2008), their mobility may make them inappropriate for studying changes at the reach scale.

---

[12] The cost of freshwater restoration extends to $billions in the US alone (Rumps et al., 2007)

CHAPTER 3

THE STREAM AS AN ECOSYSTEM

The background provided in this chapter provides information on current state of river science in relevant areas. Rates, patterns and processes of geomorphological and ecological development need to be based within the context of the river as an ecosystem. Many different elements have the potential to change and evolve, and will influence the direction of development and endpoint achieved. This chapter sets out current theories and understanding of rivers as conveyor belts of sediment transport and substrates of ecological habitat. It starts with the relevance of catchment hydrology to channel construction, the expected geomorphological features, and then builds the ecological elements up to the ecosystem level. To investigate the development of the Nith realignment from an engineered channel to a functioning stream ecosystem some consideration of natural geomorphological and ecological processes is required.

Aims

Provide an overview of hydraulic and geomorphic processes likely to shape ecosystem development
Review ecosystem concepts that may help explain observed patterns of development
Relate the realignment projects objectives to desirable properties of river ecosystems
Discuss the concept of diversity in relation to ecosystems and the development of habitat

Semantics

It is necessary to first define what is meant by ‘engineered channel’ and ‘functioning stream ecosystem’. The simplest definition for a river, ‘a path of moving water’ tells us little about its nature or form. A hydrologist might leave the definition simple (e.g. Gordon *et al.*, 2004) reflecting an interest focused on flow. Likewise the definition given by a fluvial geomorphologist reflects an interest in geomorphological aspects; “Rivers are essentially agents of erosion and transportation” (Knighton, 1998). An ecologist is likely to focus on biotic aspects such as benthic invertebrates and riparian vegetation. (e.g. Moss, 1998; Giller & Malmqvist, 1998; Everard & Powell, 2002) Rather than producing a definition reflecting the overall objective of the project to build a ‘river’ rather than just a channel or conduit, the phrase functioning stream ecosystem is adopted. This is used to describe a health condition with the various biotic and abiotic components interacting in a manner expected of a natural channel. By being more specific it avoids any miss-understanding. ‘Engineered channel’ has already been used in reference to river channels constructed to engineering specifications as existed at the House of Water mine site at the time of connection. As discussed (Chapter 2), the approach to engineering river channels has changed dramatically resulting in many different types, however, and from here on ‘engineered channel’ will be used to refer to fluvial forms of channel – channels designed and constructed to allow fluvial processes.

3.1 Introduction

The driving force behind rivers is the flow of water from the source to estuary. The study of this, hydrology, forms the basis of much river research today. Magnitude, frequency and duration of formative flows shape the physical habitat of streams and rivers (Brierley & Fryiars, 2005; Biggs *et al.*, 2005 and many others). A stream’s hydrology, the responding structure of the physical habitat and the hydraulic habitat that it, in turn, creates will shape the riverine and riparian flora and fauna communities present (Giller & Malmquist, 1998). Some of these species interact with the hydraulics and hydrology, stabilise sediments, process nutrients and in doing so begin to alter their environment, creating new habitat and opportunities for new species to invade. In this way rivers are mosaics of continually changing patches at different stages of development. Distance between populations, local climate, migratory fauna all influence and additionally help shape the ecosystem to different degrees at different times. The result is a highly complex ecosystem with many interacting components. This is concisely summarised by Harper & Everard (1998) who describe rivers as ‘…dynamic ecotones, controlled by processes that operate over a range of timescales and geographic extent, and that compromise a matrix of interdependent, transient habitats’. Essentially the aim of the realignment project at House of Water was to create an engineered channel that would rapidly evolve into this natural dynamic form but within limits due to mining proximity. To gauge the success and inform similar projects we need to understand how the system develops, rates and patterns of development, and barriers to development. This will be more definitive when based on sound scientific knowledge.

3.2 Hydrology

Hydrology is a well-studied area of the river sciences, not least because of the importance of understanding the nature of high flows and flooding to society. Patterns of variability in flow can have a major influence on geomorphological and ecological aspects that provide the structure and function of lotic ecosystems (Moss, 1998; Tockner, 1998; Biggs *et al.*, 2005). Natural disturbance in a stream system is introduced by variability of flow, which acts at a range of scales from destructive spate flows and droughts to disturbance at the micro-scale caused by the lapping of river margins. In upland gravel bed channels, hydrology, specifically spate flows, is the dominant force shaping the physical habitat (Moss, 1998) and is likely to play and important role in the morphological development of the engineered channel. Many argue that it is equally dominant over the development of invertebrate communities (e.g. Brown, 2007) and may override the effects of competition and predation. The most evident influence of hydrology is the presence of fluvial features such as gravel bars but it is also a determining factor of channel form and planform (Brooks, 1995; Gilvear, 2004). The influences within the channel are well documented but also extend well into the riparian zone (Hupp & Osterkamp, 1996; Francis, 2006; Wintle & Kirkpatrick, 2007) and across the floodplain such as through processes of scouring and deposition (Gurnell *et al.*, 2006). The relative importance of low frequency large-scale spate flows versus smaller persistent flows in shaping physical habitat has been a classic question in fluvial geomorphology (Costa & Connor, 1995).

Aside from the indirect effects that hydrology has on biota through shaping the physical habitat, large scale events such as infrequent spate flows may also determine “high level characteristics of ecosystem structure” (Biggs *et al.*, 2005). A major mechanism which may be important in this respect is the effect on colonisation patterns.

- Generation of propagules
- Redistribution of seeds and propagules between channel and floodplain
- Dispersal and redistribution of benthic invertebrates (passive)

In this respect hydrology may have a significant influence over the colonisation patterns in engineered channels. Understanding these and other influences on biota is important for river management (Biggs *et al.*, 2005). For example high flow events can severely disturb benthic communities, but equally may be important in maintaining biodiversity (Townsend *et al.*, 1997). Hydrology should therefore be taken into account in conservation strategies that are dependent on the effects of engineering projects (Vaughan et al., 2009), land use (Davies et al., 2008; Krause et al., 2008), and other river modifications. The effect of flood flows in shaping biological communities is still only partly understood (Renofalt *et al.*, 2007), with the exception of some studies that report behavioural responses to increasing flow velocities (e.g. Holomuzki & Biggs, 2000). The effect of flooding, important for the sustainability of natural systems, may also have the potential to reduce ecosystem integrity in modified and fragmented riparian zones (Hawkins *et al.*, 1997). Understanding the process of development will be important in understanding how degraded systems may respond to flooding and to a fragmented approach to restoration.

Biggs *et al*., (2005) hypothesise ‘that a hierarchy of flow variability is probably the “underlying reason for many temporal and spatial patterns of biological characteristics at different scales in lotic ecosystems” The large scale low frequency variability in flow (spate flows) have been demonstrated to shape the physical habitat.

3.3 Geomorphology

The physical structure of stream and river environments has been widely studied for many years. A range of methods and indices have been developed to describe the river channel form (Fozzard, 1997; Western et al., 1997; Barbour et al., 1999) and specifically, the variability (Bartley & Rutherfurd, 2005). Further studies have attempted to elucidate geomorphological processes behind channel form and relate both form and process to the creation and persistence of ecologically functional habitat (Gurnell *et al.*, 1998; Gurnell *et al.*, 2006). This research has extended to the four corners of the globe (Gilvear *et al.*, 2000; Boruah et al., 2008; Thorndycraft et al., 2008) but has generally remained focused on natural or regulated rivers. Less research has focused on whether geomorphological processes can be relied upon to create complex riverine and riparian habitat following construction of a basic engineered channel form. This is critical to the success of many restoration projects and is expected to be central to development on the realigned River Nith. The alternative approach focused on restoring a static physical element of the stream structure fails to recognise that the very nature of stream channels is to move (Palmer *et al.*, 1997) continually clearing and re-creating habitats. Physical stasis in no way ensures ecological sustainability and in the long term is likely to negatively affect morphology. Hence the importance of gathering data about the potential restorative effect of geomorphological processes. It has been proposed that an approach to channel design inline with modern geomorphic principles is more likely to achieve desired levels of dynamism and sustainability (Kondolf, 1998; Gilvear, 1999) but as yet there have been few attempts to establish whether of not empirical data supports this view. Where published data relating to river ecosystem ‘architecture’ type projects does exist, results show relatively rapid development of specific aspects of physical habitat in rivers. Changes in cross-sectional form can occur within a few years (Griffin, 2005) and the development of in-channel sediment stores such as mid and point bars and riffles, within six months (Sear *et al*., 1998). However, predictions of geomorphic development are difficult to develop and often unreliable (Hughes et al., 2005; Vaughan et al., 2009). This is in part due to the complex inter-relationships of channel factors such as form and substrate, and external factors such as discharge and a general paucity of empirical data.

It is possible that designs relying solely on hydrological and geomorphological processes fail to recognise the importance of the physical environment. Geomorphology acts on more than just the sediment that it transports. Differences in bank substrate and external factors such as woody debris serve to add variability to the fluvial processes driving a greater level of physical heterogeneity. In this way sediment systems can be said to have memory such that present geomorphological patterns in natural rivers are strongly influenced by past form and processes (Newson, 2002). An appreciation of geomorphological history is important for interpreting present processes, and predicting responses of river channels to human interventions (Sear *et al*., 1998). In this regard little is known about the long term effects of engineered channel form, where there is no geomorphological history to speak of, on the direction of ecosystem development.

The structure of the stream has a major influence on the range of habitats available. Some elements of a rivers structure are widely recognised; the streambed and banks, islands and backwaters, pools and riffles. It has long been known that invertebrate species respond to factors relating to the range of habitat elements (Badcock, 1949; Maitland & Penny, 1967; Egglishaw, 1969). There is often a deterministic relationship between stream biota and the physical features of river systems (Giller & Malmquist, 1998). Some habitat elements are less obvious because they are less visible; the hyporeic zone, or less well defined; the riparian zone. Communities respond to extent and diversity of each habitat, which in turn is very much defined by the hydrological regime. The circle is completed by the strong feedback mechanisms that allow the flora and fauna present in a habitat to extensively change ecosystem structure and processes.

3.4 Elements of river ecosystem habitat

3.4.1 Morphology

Planform

Any development of the engineered planform will be principally through channel migration and, as such, dependent on rate and patterns of bank erosion and point bar deposition. Much of this is expected to occur at meander beds where the determining factors will be bank erodibility and near-bank-stream-flow erosivity (Wallick *et al.*, 2006). These depend on dependent on causal links between independent and dependent variables (Sear *et al.*, 1998, Piegay *et al.*, 2005; Wallick *et al.*, 2006) Dependent variables: Sediment load, discharge regime, boundary materials - grain size and degree of cementation, flow velocity, frictional resistance. Independent variables: Slope, depth, width, bedform, planform geometry. Studies have shown a very close relationship between channel width and meander form (Leopold, 2004). Greater curvature results in greater secondary flow strength producing an increase in near-bank-flow erosivity, exerting large influences on channel movement, possibly irrespective of bank material (Wallick *et al.*, 2006). However, there is considerable inaccuracy and uncertainty in quantitative models of planform adjustment through space and time for alluvial rivers (Sear *et al.*, 1998; Piegay *et al.*, 2005). This may be because limitations associated with factoring in the diversity of hydrology and sediments (Piegay *et al.*, 2005) and in some cases the influence of channel stabilisation upstream, which can affect the downstream patterns of erosion by limiting sediment replenishment (Piegay *et al.*, 2005). It remains to be seen whether or not these models can be applied to engineered channels. The need for predictive models is hampered by the very diversity in structure and stability that is responsible for such a highly variable and biodiverse habitat, which in turn drives the desire for predictive models. The uniform nature of bank material on the diversion could therefore have consequences for the habitat diversity that develops. However, it also provides an opportunity to investigate channel migration patterns without having to factor in variability in bank material.

The plan form exists within a zone of potential geomorphologic activity in which the river moves over long timescales, eroding and depositing material. This zone has been referred to by many terms in the literature. Erodible corridor concept (Piegay *et al*. (2005), streamway, stream corridor, inner river zone, riparian corridor. Use of the concept has been advocated in the literature for years but is only now beginning to be adopted by practitioners (Piegay *et al*., 2005). In the early days of stream engineering there was some appreciation of the importance of giving rivers the space to meander and shift within this zone (Springer 1903 IN: Piegay *et al.,* 2005), which can be seen, for example, under bridge structures. More recent development has had a tendancy to encroach on this area. As a direct result erosion within this area has often been referred to as a problem (Gardener, 1978). There is little legislation in place in the UK that recognises the importance of this area although some progress has been made elsewhere: Since 2001 mine sites in France have not been allowed within the ‘space of mobility’ of rivers following a décret from the French Environmental Ministry (Piegay *et al*., 2005)

Cross-sectional form

The controlling variables of channel form can be divided into the driving variables, boundary characteristics, and existing channel form (Newson, 2002). Driving variables include the discharge hydrograph and sediment supply and transport capacity. Boundary characteristics include valley slope and topography, bed and bank materials, and riparian vegetation and woody debris and rocks (Olson-Rutz & Marlow, 1992). Channel form includes cross-sectional geometry, long profile and planform. All have a strong influence on channel shape which, in turn, determines the local hydraulics. The variability in width, depth and water velocity that this introduces has important implications for the nature and extent of habitats available to biotic communities. Despite the importance of variability in stream channel form being widely known (Ward & Tockner 2001), and furthermore, widely used to evaluate responses to management (Bartley & Rutherfurd, 2005) little is known about its development in man-made channels. This is possibly because channels were for many years designed to a fixed width for maximum flow conveyance. Nevertheless is an important consideration for modern realignment projects. The little research done in the area suggests factors are scale dependent and include vegetation and bank material (Anderson *et al.*, 2004). In fact feedback from vegetation growth may occur early on in development (Gurnell *et al.*, 2006). The use of natural bank materials in modern diversions allows erosion processes to develop the channel form. Because the literature has historically focused on the many perceived negative impacts of bank erosion (e.g. Gardener, 1978) such as damage to property and infrastructure or the loss of land and the potential impacts that the sediment can have on channel morphology and capacity, bank erosion is often considered a hazard (Piegay *et al.*, 2005).

Although precise predictions of erosion patterns have been difficult, the development of asymmetry on meander bends is expected. Re-meandering is a widely used restoration technique, partly for this reason, and has many benefits for stream health (Friberg *et al.*, 1998). However information of the exact benefits to ecological diversity is limited. One recent study (Nakano & Nakamura, 2008) shows a high level of invertebrate richness at the reach scale compared to channelised ‘controls’. They establish a relationship between both invertebrate density and richness and the shear velocity at each sampling location, with the highest richness at low velocities. No similar studies have been done on upland realignment projects.

Cross-sectional area, width, depth and width-depth ratio provide some information about geomorphological characteristics. Channel shape can reveal further information but is often reported as observations rather than repeatable measurements. Indices of channel shape can provide information about change undetected by the more traditional width depth ratio (Olson-Rutz & Marlow, 1992). Changes in channel area associated with movements of the riverbed and riverbank materials may represent natural flux or act cumulatively leading to substantial long-term changes (Olson-Rutz & Marlow, 1992).

The hydraulic diversity that results from development of a complex channel form including eddies and slack water areas are the most effective features in trapping seeds under natural flow conditions (Merritt & Wohl, 2002). Vegetation may have a key role in this respect; trapped seeds along the edge of the riparian zone positively correlates with plant species richness (Andersson *et al*., 2000b). Eddies and slack zones may also play an important role in invertebrate drift.

3.4.2 Within-channel habitat

The nature and diversity of the stream bed

The most indisputable element of the stream ecosystem is perhaps the bed since it is permanently or semi-permanently aquatic. Bed sediments can be characterised by differences in size, variability, sorting and packing and are influenced by water depth, gradient, sediment supply and the frequency and magnitude of flood flows (Jowett, 2003). It has been argued that the morphology of the streambed accurately reflects the range of flows that move through the channel (Bartley & Rutherfurd, 2005). Stream morphology would then provide a surrogate measure for flow conditions. Differences in bed sediment are likely to have a major influence the range of habitats available to colonisation by benthic communities (Wene & Wickliff, 1940; Williams & Smith, 1996). ASCE (1992) highlights bed sediment characteristics as ‘the primary influence of community composition and density’ (although this conflicts with the intermediate disturbance hypothesis). As such, development of a natural physical form of streambed is likely to be vital for ecosystem development within engineered channels. The size of interstitial spaces, for example, has been reported to affect colonisation (Townsend et al., 1997; Schmude et al., 1998). The large interstitial spaces provide little in the way of shelter for smaller invertebrates that will colonise gravels to a much higher degree. Large pebbles (>40mm) provide a more stable substrate to attract clinging sedentary invertebrates (Malmqvist & Otto, 1983; Khalat & Tachet, 1980). In this respect, the initial un-cohesive, uncompacted nature of the bed sediments in engineered channels will be more easily mobilised than the armoured layer of natural gravel beds. However the resulting supply of sediment can be an important driver of morphological development in downstream reaches (Sear *et al*., 1998). Whilst much can be learned studying the development of form under ‘laboratory conditions’ using flume experiments (e.g. Weichert *et al.,* 2008) there is also value to testing the results and associated channel design techniques under natural conditions.

Considering the clear link between bed substrate and invertebrate community it is critical that the physical habitat of the stream bed develops to a desirable end-point matching natural ‘reference’ reaches. However a functional stream habitat requires more than a *natural substrate structure*, rather success will depend on the development of a diversity of bed conditions. A study by Sarriquet *et al*., (2007) illustrates this point. In a project that aimed to restore bed sediments, interventions produced little increase in species richness. This, they explain was because the intervention simply changed the habitat from one condition to another, significantly altering the composition of invertebrate assemblages but not increasing richness. Natural rivers, even across single geomorphic units are a mosaic of variable bed substrate conditions.

The development of geomorphological features

The topography of the river bed is characteristically diverse in gravel bed rivers (Knighton, 1998; Gilvear, 1999) resulting in a range of different habitat features or ‘Morphological Habitat Units’ and associated variability in flow patterns (Bartley & Rutherfurd, 2005). Classic examples are pools and riffles, which have been used to represent hydraulic character and substrate composition (Parsons *et al.*, 2003). Development of inchannel sediment storage in the form of gravel bars can dominate the early morphological development of new channels (Sear *et al*., 1998). Rather than being a gradual process however, formation is rapid and related to bedload transport capacity and stream power during early flood events (Gilvear & Bradley, 1997). Morphological habitat units have been shown to support distinct invertebrate assemblages (Parsons & Norris, 1996), including on the upper catchment of the Nith (Griffin, 2005). Resh *et al*. (1988) relate increased abundances of stream insects on the upper sections of riffle features to increased stability but provide no explicit evidence. Conversely, high levels of disturbance in pool areas of gravel bed rivers (Andrews, 1984) have also been used to explain their relatively low invertebrate diversity. Even riffles with a very similar general morphology can have very different composition, structure and hydraulic conditions, with differences existing both between riffles and across individual riffles (Pedersen & Friberg, 2007).

As discussed diversion of river flows into man-made channels is not new. However, many of these channels have been designed with no capacity for sediment transport, severely limiting the development of geomorphological formations such as point bars and gravel islands. Sedimentary structures are characteristic of gravel-bed rivers and, because they vary greatly in size, form and type, contribute considerably to habitat complexity (Ward, 1998; Gilvear & Bradley 2006).

Organic matter

The importance of organic matter and nutrient cycling in stream ecosystems is not always fully acknowledged (e.g. Parsons *et al.*, 2003). The amount of organic matter trapped by different morphological units, for example, is likely to be an important aspect affecting colonisation and development towards a functioning stream ecosystem. It has been shown correlate well with the benthic diversities of stream invertebrates (Egglishaw, 1964). Furthermore, Bird and Hynes (1981) have shown that given a lack of organic matter, new colonisers will rapidly move on to a new area. CPOM is an important habitat for Microbial activity promoting nutrient retention (Aldridge *et al*., 2009) as well as being used directly by macroinvertebrates as a food resource and building material (Merritt & Cummings, 1996). Many stream ecosystems depend on organic matter inputs (Vannote *et al*., 1980, Mulholland *et al*., 2001) as a result retention of organic matter is likely to play a major factor controlling channel development and functioning. This could happen in a number of ways. However, studies of alluvial gravel riffle-pool streams by Brussock & Brown (1991) found physical parameters appeared to be of greater importance than trophic-related process in determining longitudinal patterns in assemblages of benthic invertebrates.

Large woody debris is an important component of many stream ecosystems with the potential to significantly enhance development through providing habitat niches (MacInnis et al., 2008) and altering channel dynamics (Muotka et al., 2002; MacInnis et al., 2008) and even establishing a tree population (Opperman & Merentender, 2007). Although fine woody material may not influence stream habitat heterogeneity in the same way that course debris has been demonstrated to (Gippel *et al*., 1996; Bennet *et al*., 2008) it may still represent important invertebrate habitat (Milner & Gloyne-phillips, 2005)

3.4.3 Near-channel habitat

Riparian zone

The defined extent of the riparian zone is a little vague with some authors restricting it to the bankfull discharge (Hupp & Osterkamp, 1996) while others extend it out across the floodplain (Stamford *et al*., 1996). Riparian zones can be physically, geomorphically, biologically and ecologically diverse, often considered more so than other ecosystems (e.g. Tockner *et al*., 1998). Naiman et al. (1993), for example, state "Natural riparian corridors are the most diverse, dynamic and complex biophysical habitats on the terrestrial portion of the Earth". There is certainly plenty of corroborative evidence[1] although this is best described and documented for vascular plants and tends to be limited elsewhere (Nilsson, 1992). The high levels of biodiversity and productivity observed in the terrestrial-aquatic transition of riparian zones may be related to high levels of available resources, disturbance regime and the corridor structure of riparian zones (Tabacchi *et al*., 2005). Although dependent a little on where the outer edge of the riparian zone is deemed to be, the main hydrogeomorphic processes behind the disturbance regime that shapes the riparian habitat are thought to be flooding, erosion and the accumulation and reworking of sediment (Salo, 1990; Stieger *et al*., 2005). The value of these processes in channel construction projects has been demonstrated on the River Cole though overbank deposition of gravels (Sear et al., 1998). Certainly most research into the influence of hydrology on the riparian zone has been focused on the effects of high flows and flood events (Hupp & Osterkamp, 1996; Hakins et al., 1997; and others). Flooding can cause considerable devastation to extensive areas of riparian zones. Hawkins et al., (1997) found up to 40% of pre-flood vegetation cover could be stripped away during exceptionally high flows. Obviously catastrophic levels of flooding can seriously reduce species diversity they may serve a similar ecological function to forest fires in adapted ecosystems, initiating a sequence of successional processes (Hawkins *et al.,* 1997) assuming

---

[1] >260 species of vascular plants (representing 13% of the vascular plants found in Sweden) were recorded along a single Swedish river, Vindel River (Nilsson, 1992). This river also holds the record 131 species of plant in 200m of river bank (Nilsson and Lunberg, 1985 IN: Nilsson, 1992). Around 900 species were found in a survey of the Ardour River, France (Tabacchi *et al*., 1990). A survey of approximately 200m of riparian zone along the Ore River, Sweden, found 264 species of invertebrates and a study looking at patterns in diversity across a the riparian transition identified 426 morphospecies of invertebrates (Dangerfield et al., 2003)

they are sufficiently infrequent. An intermediate level of disturbance (in both frequency and scale) has been hypothesised to maintain high levels of biodiversity (Townsend & Scarsbrook, 1997)

Studies generally conclude that flooding has a positive effect on structure and function, and supports biodiversity. For example, when the influence of floods is removed by flow regulation the species richness in the riparian zone has proved to be lower than comparable rivers with a natural flood regime (Jansson *et al*., 2000). Local extinctions during long lasting floods and recolonisation in inter-flood periods seem to be important processes for maintaining biodiversity, the recolonisation process specifically resulting in rapid increases in species richness within a 10-year period (Renofalt *et al*, 2007). There are a number of additional ways that floods contribute to shaping riparian vegetation composition and structure. These include creating habitat heterogeneity by opening up space, redistribution of nutrients, formation of new habitat such as gravel bars, sorting of bed gravels and the maintenance of habitat gradients.

As such a diverse and dynamic element of the stream ecosystem, it has been suggested that the riparian vegetation provides a key indicator of channel condition (Montgomery and MacDonald, 2002) and that the riparian system should serve as a framework for understanding the wider fluvial ecosystem (Naiman *et al*. 1993). As well as indicating condition of the Nith realignment, bank vegetation is likely to play a key role in the development of ecological functioning within and along the river channel with different elements providing different ecosystem functions (table 3.1). The complexity of the riparian zone and rate of development of particular elements such as the slow growth of woody vegetation is likely to have implications for ecosystem functioning. Litter inputs from woody vegetation are an important source of carbon in the channel (Shields et al., 2008) and may influence community structure both within the immediate reaches and in downstream reaches. Whether herbs and grasses make a similar contribution to the stream ecosystem is less certain although a limited amount of evidence suggests that the carbon inputs might be significant (Menninger & Palmer, 2007). The structure of the riparian zone may also be an important element of ecosystem development. Trailing riparian habitat, for example, has been shown play a role in assisting colonisation within developing streams by some invertebrate taxa, for example colonisation of unconsolidated sediments by EPT taxa in new branches of braided rivers (Milner & Gloyne-Phillips, 2005). Vegetation also has a role in controlling erosion, stabilises soil and reduces current velocity during floods, as shown in a number of studies (e.g. Nilsson, 1992; Anderson *et al*., 2004; Francis 2006)

Restoration of the floodplain area is increasingly recognised as a desirable and effective objective in any ecosystem architecture type project (Asselman, 1999; Palmer *et al.,* 2005). Connectivity between the river and floodplain is a major element of theories on river ecosystem function and loss of connectivity in this respect can reduce the diversity and productivity of aquatic habitats (Rahel, 2007) affecting stream structure and function (Aspetsberger, 2002). However, constraints to floodplain restoration or creation often exist. These include limited scientific understanding, the complexity of floodplain governance and the generally high economic value of the land (Adams & Perrow, 1999). Much of the floodplain area around the Nith realignment at house of water remains heavily managed and natural patterns of ecosystem development are unlikely to be observable. For this reason, and because of limited resources, research in this area was not undertaken as part of this PhD. As such the floodplain is not considered further.

Table 3.1: Summary of the ecosystem functions of the different elements of the riparian zone.

Element		Action	Ecosystem function
Trees	Canopy	Provides shade controlling instream light and temperature levels	Controls primary production within the channel
		Source of large and fine plant detritus	Provision of food and building materials
		Habitat corridor in form of line of trees across landscape	Communication corridor connecting populations and genetic material
	Large woody debris	Routing water and sediment	Shapes habitat creating complexity and refugia
		Substrate for biological activity	Provides food and space
	Roots	Deflection of flow	Increases bank stability
		Absorption of water and nutrients	Reduces enrichment of river water
Vegetation	Leaves and stems	Source of detritus	Provision of food and building materials.
		‘Matting’ of leaves and roots	Provide a erosion resistant surface

3.4.4 Catchment

It is important to remember that patterns of benthic macroinvertebrate distribution also occur independently of those determined by geomorphological boundries (Parsons *et al*., 2003). An example is catchment land use which can be a major factor influencing stream communities (Brisbois et al., 2008) and the effect can continue for many years after the use has changed (Harding et al., 1998). Biogeographical factors are also important such as the climate of the catchment, dependent on its wider geographical position.

3.4.5 Refugia

Sedell et al. (1990) defines refugia as habitats or environmental factors that confer spatial and temporal resistance and resilience on to biotic communities impacted by disturbances. This include both flood and drought flows but could equally be extended to refuge from predation. Parker *et al*. (2007) provide initial evidence that small sedentary herbivores in freshwater systems can gain enemy free space by feeding on plants that are chemically defended from larger consumers. Furthermore refuge from flow for invertebrates may also provide a space where organic matter can accumulate during and after floods. Nikora *et al*. (1998) suggested that bryophytes may create hydraulically quiescent regions around them, and that these regions could in part explain the high invertebrate densities and the algal and detrital biomass within these plants. This may also explain the high abundances of macroinvertebrates that have been found associated with dead wood in streams (Milner & Gloyne-Phillips, 2005). The nature of refugia is highly variable. It can be hydro-geomorphological (e.g. Lancaster, 2000), biological (e.g. Lancaster and Hildrew, 1993) or chemical (e.g. Parker, 2007) and as a result tends to vary in spatial and temporal extent. Examples vary from live vegetation to woody debris, single boulders to large debris dams, and can be in backwater, pool and riffle areas. The existence of many of these elements is dependent on there being sufficient physical diversity. Refugia may therefore be limited in the relative uniformity of engineered channels, at least initially. The availability of refugia is likely to be an important factor in maintaining a diverse and healthy ecosystem on the River Nith.

3.5 Ecosystem processes

These ecosystem ‘processes’ include models of disturbance, retention, nutrient cycling, and succession. Some of these have already been touched upon earlier in this chapter. An appreciation of these additional factors is a key ingredient in the design of ecosystem architecture projects (Kondolf, 1998) such as the realignment of the River Nith.

3.5.1 Disturbance

The physical effects of hydrological disturbances are evident though the formation and reshaping of fresh fluvial features, and in the riparian zone where frequent disturbance by flood and debris flows create a complex shifting mosaic of landforms. (Niaman *et al*., 1993). High flows have in the past been considered disturbances in terms of their localised damaging effects (Scott, 1950; Eljabi & Rousselle, 1987; Steinman, 1992), and as the term has taken on a more scientific meaning, floods have often been identified as the main disturbance within river ecosystems. However with the increasing appreciation for the role of disturbance in shaping habitat[2] the term has become more useful as a subtle concept to describe a specific ecosystem process. Disturbance in streams occurs at a range of spatial and temporal scales and drives the dynamic element of the system central to the maintenance of high biodiversity (Muotka & Virtanen, 1995; Ward, 1998).

Focus on the role of disturbance in shaping invertebrate assemblages within streams is common in the literature (Mackay & Currie, 2001; Death, 2002; Brown, 2007). Although developed as a model to explain patterns in vegetation development (Grime, 1979) the Intermediate Disturbance Hypothesis has been applied to benthic communities (Townsend & Scarsbrook, 1997; Death, 2002) and used to explain the exceptional diversity found in some riparian zones. Its application predicts that habitats disturbed intermittently produce communities composed of both pioneer and climax taxa and are expected to be the most diverse (Death & Winterbourn, 1995). Evidence for the influence of high flows on physical habitat is extensive. In upland gravel bed rivers where sediments tend to be less cohesive and stream power can reach very high levels, the entire riverbed can be mobilised and reshaped (Gilvear & Bradley, 1997). Jowett (1997) found that the types of bed movement associated with frequent floods reduced abundance and affected species composition. At the extreme, low frequency high flow events can reset entire benthic invertebrate communities (Scarsbrook & Townsend, 1993). Turbulence introduced by bed topography also seems to play a role in reducing anoxia in riverine and riparian sediments, reducing plant mortality during flooding (Renofalt *et al.,* 2007)

---

[2] Disturbance has been recognised as a processes central to the functioning of stream ecosystems. As such it has been categorised by form. However, one should highlight the importance of distinguishing between the disturbance e.g. high flows or drought flows, and the effect of the disturbance e.g. structural changes. Other studies use the term disturbance to refer to the response eg population changes (e.g. Muotka & Vituranen, 1995)

When disturbance levels are suitably low in magnitude or infrequent there is the potential for macrophyte growth. A range of ecological functions provided to small freshwater invertebrates by plants has been hypothesised and tested. The provision of habitable living spaces (Lodge 1985), entrainment of particulate matter (Brusven *et al*. 1990), surfaces for epiphytic algal growth (Suren 1991), shelter from turbulent flow (Linhart *et al*. 2002) and chemically defended refugia from predation/consumption (Parker et al. 2007)

Low flow

Hydrology continues to have an influence at during times of low flow. Drag related disturbance declines in relative importance with decreasing time scales replaced by the increasingly dominant mass-transfer processes which dominate the high frequency low magnitude variability that characterises stable flow (Biggs et al., 2005). Mass transfer is a broad term encompassing processes such as food uptake by invertebrates through grazing/predation, predation by fish, uptake of inorganic nutrients by autotrophes. At small frequent scales of variation mass transfer processes are likely to control growth and sustainability of individuals with moderate to high frequency flow events influencing processes that act on the organisation of populations within the community (Biggs et al., 2005)

- Nutrient transport
- Packing sediment
- Deposition of fines
- Oxygenation of gravels

Moderate to high frequency, low magnitude events also influence invertebrate communities (Biggs et al., 2005). The development of community structure and the existence of guilds may be an indicator that ecosystem functioning relies on the existence of small scale high frequency variability introduced by this type of disturbace. Small scale disturbance can have a negative effect on invertebrate communities. By turning over rocks Robinson and Minshall (1986) changed the flow conditions for the resident invertebrates. Different feeding strategies and levels of mobility can affect and invertebrates ability to adapt to this type of disturbance (Mackay 1992). The result of the Robinson and Minshall study was a significant reduction in invertebrate density and species richness with increasing disturbance frequency.

Spate flow and drought flow

Both floods and droughts might be considered to cause high magnitude disturbance. The drag disturbance that occurs during floods and associated dislodgement of biomass by high water velocities and associated abraision by mobilised bed sediments will have a strong influence over invertebrate and plant communities. Likewise the deprivation of water during period of drought. The conditions in both situations are outside the tolerances of individuals of most species.

Shallow stable areas of river bed have been found to provide important habitat to macroinvertebrate communities where they are generally poorly adapted to high magnitude disturbance (e.g. Nakano & Nakamura, 2007). However, in Scottish upland rivers it is possible that communities will be better adapted, given the frequence of high magnitude disturbances and its influence over ecosystem processes.

Erosion

In many circumstances bank erosion ought to be viewed in a positive light and may need to be preserved (Piegay et al., 2005). Related processes are essential for balanced sediment transport result in gravel bar formation and maintenance promoting healthy aquatic and riparian ecosystems (Wallick et al., 2006). The use of channel stabilisation efforts that reduce disturbance processes may reduce a rivers ability to develop new geomorphic surfaces (Wallick et al., 2006). However, a reduction in geodiversity can result from excess fine sediment entering the system (Bartley & Rutherfurd, 2005)

3.5.2 Retention

Although recognised as an important ecosystem process within riverine systems there has been limited published research in this area. Moutka & Laasonen (2002) found the retention capacity of newly restored channels, bare of bryophytes, to have reduced retention capacity. Experimental releases of wooden dowel (Bocchiola et al., 2006) and plastic leaves (Speaker et al., 1988) have been used in an attempt to assess the retention capacity of river reaches but not of new sections of channel.

3.5.3 Nutrient cycling and trophic linkages

Macroinvertebrates have a key role in nutrient cycling and the transfer of trophic energy through the ecosystem. They have been shown to play a significant role in the breakdown of leaves at certain times of year. Exclusion of invertebrates from leaf packs by Nelson and Anderson (2007) reduced the loss of organic matter by 25% and nitrogen loss be 65%. Forms of disturbance such as dessication, sedimentation and freezing are cited as reasons for the limited macroinvertebrate processing at other times of year. Drifting invertebrates are a major source of food for a number of freshwater fishes (Merritt & Cummins, 1997). Macroinvertebrate density and diversity were found to be among six variables which best explain patterns in salmonid biomass (Annoni et al., 1997). Trophic interactions such as herbivory and predation are often altered in degraded systems creating patterns resilient and resistant to restoration efforts (Suding et al., 2004)

Riparian corridors can also play a potentially important role in the removal of nutrients from runoff from terrestrial areas of the catchment (Lowrance et al., 1984). The nutrients taken up by the riparian zone may eventually reach the stream and coarse particulate organic matter. This has been identified as an importance allochthonous carbon source in many streams (Pozo, 1997), often identified as an important role of riparian woodlands. However, inputs from herbs and grasses, although poorly understood, may represent a significant terrestrial-aquatic linkage and an important allochthonous energy source for non-forested streams (Menninger & Palmer, 2007)

3.5.4 Feedback

It is possible that feedback will be a significant mechanism in the development of the engineered channel. For example as aquatic plants colonise they will modify habitat conditions that feedback into hydrological, geomorphological and ecological processes (Naiman et al., 1999). Some aquatic bryophytes may actually increase substrate stability by lowering the drag of the rocks on which they grow. Results from flume experiments by Suren et al. (2000) suggest that they may stream-line rocks, making them less prone to movement and increasing their suitability for additional colonisaters. As well as changing physical conditions such as changing water velocities, blocking sunlight, increasing oxygen levels reducing the availability of primary substrate (Bare rock) they can change the ecology providing attachment surfaces, ovipositioning sites, trapping detritus, providing food and refugia (Biggs 1996). This may result in increase suitability to colonising invertebrates.

Feedback mechanisms also have the potential to reinforce ecosystem development to an undesirable endpoint. More attention is now being paid to the ecological constraints that create internally reinforcing feedback by managers of restoration projects (Suding et al., 2004). It is important to recognise that the dynamics of the engineered state may be very different from those of the pristine or target condition. Barriers to the development of restored or created ecosystem environment into a functional ecosystem can develop as a result of feedback mechanisms between the habitat and internal or external factors such as invasion of inappropriate species, remoteness of native colonists, landscape fragmentation. Failure to accept the possibility that feedback mechanisms may internally reinforce a degraded system/state risks inappropriate management strategies resulting in unexpected and undesired new state, or the failure to perturb the system from the degraded state (Suding et al., 2004). Development may require innovative management to over come the constraints that starting condition of the system invariably imposes (Suding et al., 2004)

3.5.5 Succession and End points

It is generally accepted that many ecosystems are subject to temporal changes in community composition. The term succession, used to describe this change can be slip into primary and secondary succession (Grime, 2002). Primary succession occurs during colonisation and development of a new ‘skeletal’ habitat. Secondary succession occurs in circumstances where habitat is disturbed and recolonised. Succession is thought to be a major driver of the structure and function of riparian vegetation (Milner & Gloyne-Phillips, 2005). Tied into this view of habitat evolution is the concept of the ‘end-point’, which represents the habitat conditions towards which successional changes are headed. However, for political and economic reasons, development in ecosystem architecture type projects rarely parallels natural trajectories of succession or degradation (Kondolf *et al*., 2006). There are many examples of restoration efforts relying on successional pathways that have had unexpected endpoints (Suding et al., 2004)

A more successional based approach assumes that the re-establishment of the historical physical environment will allow natural successional processes to reinstate the origional ecosystem condition and biota (Sudin et al., 2004). Although in many of the examples given by Suding et al. (2004) achieving the desired endpoint of a historical physical environment simply by removing degrading pressures is obviously unrealistic. Removing degrading pressures such as grazing intensity may be a viable rehabilitation option in some circumstances. The expectation that this should always be sufficient to allow successional recovery is naïve. The persistence and resilience of degraded ecosystems is increasingly documented by research (Suding et al, 2004) and can represent ‘alternative states’. This is indicated of evidence that degraded communities being highly resilient to restoration efforts (Suding et al., 2004). A link between theoretical models of alternative ecosystem states and restoration ecology is beginning to emerge (Suding et al., 2004).

3.6 Ecosystem properties

Some important concepts contribute to our wider understanding of the 'Ecosystem' in addition to the hydrological, geomorphological and biological components. These might be considered properties of the habitat and include, variability, diversity, heterogeneity, connectivity, sustainability, resilience and integrity.

3.6.1 Variability and heterogeneity

Stream habitats can be considered as a variety of interconnected patches of variable size each with different physical characteristics. This mosaic form is promoted by the dynamic conditions in the riverine and riparian ecosystems and can be particularly complex in upland river ecosystems (Arscott et al., 2000). The study of variability can focus on almost any aspect or element of the ecosystem but is often overlooked. For example, most biotic research fails to explicitly recognise the effects that flow variation at different temporal scales can have on ecosystem components and processes (Biggs et al., 2005) despite the links becoming increasingly clear. Biggs et al. (2005) demonstrate a range of species interactions and behaviours which are a direct response to flow variability from a scale of years, through weeks and days to minutes, seconds and less. Other studies although not explicitly looking at the hierarchical scales, also reveal direct biotic responses (Egglishaw, 1969; Rader & Belish, 1999) that act over and above the influences flow can have through shaping the physical habitat. Flow conditions characteristic of continually changing, natural flow regimes create a zone subject to wetting and drying. This is known as the varial zone and is considered unsuitable habitat for benthic invertebrates by some authors. (eg Jowett, 2003)

More specifically heterogeneity, a particular form of variability, and a concept becoming increasingly prevelant in ecology. The distribution of stream biota is known to be highly variable, with patches of high and low taxa density and richness. This patchy distribution is likely, in part, to result from the heterogeneous nature of the habitat. Certainly invertebrate densities have been related to habitat factors including flow conditions, distribution of organic debris (Egglishaw, 1969) and the physical nature of the substrate. Large scale ecosystem elements play a key role in introducing heterogeneity to the disturbance caused by a flood. i.e. refuge patches. The size of these elements determine the scale at which the largest habitat patches are created.

Careful consideration of the variability inherent to dynamic river systems is required prior to any scientific investigation. Many research projects aim to reduce the variability in the samples by sampling a range of habitats to encapsulate the entire ecosystem community within the reach or habitat unit such as riffle/pool. However, because this patchiness may be an indicator of ecosystem functioning, it may be more insightful to sample the variability, giving a clearer picture of the true scale of biodiversity and physical habitat diversity. The study of species richness demonstrates this point. Overall species richness is often used as an indicator of diversity, but variability in species richness across a habitat or reach may be more appropriate indicator of diversity, functioning or stream health.

3.6.2 Connectivity

Riverine-riparian connections are now widely recognised as key to the sustainability of natural levels of functioning in many stream-systems. The importance of re-establishing connectivity between the riparian and riverine ecosystems has recently become widely advocated. (e.g. Gurnell et al., 1995; Harper et al., 1999). Loss of connectivity can reduce the diversity and productivity of aquatic habitats (Rahel, 2007). The splash made by the river continuum concept focused many environmental projects on the importance of longitudinal connectivity. Lateral connectivity lagged behind a little but was soon addressed by the flood pulse concept. Ecosystem functions that operate in a highly connected system include provision of food, supply of woody debris (Gurnell et al., 1995) and the transfer and storage of sediment, which can be significant in the early development of a new channel (Sear et al., 1998).

River continuum concept

One of the most important recent developments in stream ecology has been the River Continuum Concept presented by Vannote et al. (1980). The concept generated much productive debate (Giller & Malmqvist, 1998). Criticisms included the failure to tackle anthropogenic modifications. There is still a large body of evidence to support the theory although much of it comes from the area where the idea was developed. It did serve to emphasise the importance of longitudinal connectivity as well as provoking real discussions on the true nature of streams.

The river pulse concept

This model focused more on lateral connectivity and the importance of processes that cross the riparian-riverine ecotone allowing transfer of sediment and nutrients. The maintenance of lateral connections through frequent inundation of the floodplain is important in balancing out sediment budgets (Pringle, 2003).

3.6.3 Diversity

Biological diversity or ‘biodiversity’ is a key concept in ecology. Technical definitions cover a wide variety of aspects in addition to species richness but the frequent use of diversity as a synonym in this regard (e.g. Waide et al., 1999) has rather simplified its definition. Although it is intuitively understood, defining and calculating diversity has proved challenging ever since the term was coined in descriptions the natural world. There are many facets to diversity that can be applied to many scales. The focus of this study requires the interpretation of diversity at the stream ecosystem level. This will include aspects of community diversity and physical habitat diversity.

Increasing biodiversity of aquatic invertebrates strengthens the ecological integrity of streams (Spänhoff & Arle, 2007) and must therefore be a desirable aim of realignment projects in themselves. A surprising demonstration of this, conflicting with classical niche theory, was shown in a study by Franzén (2001) that showed species richness in seed mixtures enhanced recruitment.

Renofalt et al., (2007) found that riparian diversity along turbulent reaches was less affected by flood disturbance than diversity in more tranquil reaches suggesting that hydraulic conditions during low flow periods can influence the susceptibility of the riparian zone to damage under high flows.

The riparian community has long been recognised for its high species richness and seed banks within a riparian zone may be a hotspot for the reinstatment of biodiversity (Tabacchi et al., 2005). Riparian corridors are interfaces between terrestrial and aquatic ecosystems. As such they tend to encompass sharp environmental gradients (Niaman et al., 1993). They are zones of active ecological processes resulting in a diverse mosaic of landforms, communities and environments. Unusually so (Niaman et al., 1993). An analysis of species distributions within this zone would substantially improve understanding of these active ecological processes (Nilsson, 1992)

3.6.4 Productivity

There is much debate around the relationship between species richness and productivity (Rosenzweig, 1995; Waide et al., 1999) with a potential influence on many ecological theories. Those listed by Waide et al. (1999) include theories on interspecific competition, habitat heterogeneity, evolutionary maturity, habitat homogenisation, predator-prey, species energy. That productivity can effect species diversity is widely accepted for example manipulations of productivity using fertilisers has been shown to decrease plant diversity (Waide et al., 1999) but more recent research has also suggested the reverse can be true with species richness exerting an influence over productivity (Naeem et al.,1999; Hooper, 1998) although the idea is not new (MacArthur, 1955). Whether effects on productivity result from increased species richness per se, or if the mechanism relates to different functional groups or particular species is the source of current debate (Waide et al., 1999)

3.6.5 Sustainability and Resiliance

Ecoloigcal resilience can be described as the speed at which a system returns to its former state after it has been perturbed and displaced from that state. In the context of restoration resilience can refer to both a systems return to a restorative ‘goal’ state following a degradative perturbation or a systems return to a degraded state following a management perturbation (Suding et al., 2004) Ecological resistance is the amount of change or disruption (or management perturbation) that can be absorbed before processes change that control the structure and behaviour of a system (Suding et al., 2004). Healthy ecosystems generally exhibit an ability to withstand (resistance) or recover rapidly (resilience) from disturbance events. It is obvious that diversity may take a temporary knock but the nature of the ecosystem as a heterogeneous entity and the presence of colonisation processes ensure diversity can be maintained in the longer term.

The concept of ecosystem adaptability/flexibility will also be important here. With mounting evidence of climate change influencing habitat conditions a healthy bio-diverse ecosystem (species and genetic) will be in a stronger position to adapt to the changing conditions and continue to provide valuable ecosystem services

3.6.6 Integrity

“ The capacity of supporting and maintaining a balanced integrated adaptive community of organisms, having a species composition , diversity and functional organisation comparable to that of a natural system” (Angermeier & Karr, 1994). In societal terms integrity might be judged on the ecosystem services provided. Ecosystem services provided by the riparian zone include flood mitigation, aquifer recharge and maintenance of water quality (Stieger et al., 2005). Riparian zones may be particularly susceptible to colonisation by invasive and exotic species (Tabacchi et al., 2005).

CHAPTER 4

THE NATURE OF COLONISATION

This chapter focuses on the ecology of dispersal and colonisation by invertebrates and plants including possible barriers. It specifically looks at the traits of specific species with a view to interpreting the patterns that are revealed by the research. It then looks at community structure and its development.

Aims

Discuss the potential modes of colonisation for the realignment project
Review the roles that colonisation plays in the different approaches to river restoration
Discuss the importance of Species traits

4.1 The development of new ecosystems

The aim of any habitat creation is to achieve a self sustaining system with a full complement of species, features and processes as would be found in a natural unimpacted system. The best way to achieve this is at best uncertain and for many habitats, unknown; and is likely to vary considerably between projects.

The initial floristics model proposes that all desired species must be introduced (Palmer et al., 1997). Given theories on succession and well-established strategies of dispersal in many organisms, adopting this approach in project design seems pessimistic. However, a city restoration project the extent of surrounding urban area may be an insurmountable barrier to all but the most hardy colonists.

The presence of specific species may be important providers of ecosystem services and be included in project objectives. Without a balance of colonists it is possible the successional pathway in an undesired direction to an end point that differs from the project aims. Without the moderating effect of existing diversity, certain species may establish and achieve a level of productivity and growth that dominates to the exclusion of new colonists. There is some evidence that rates of colonisation are higher in diverse communities.

However problems can arise if design is based solely on an initial floristics approach. Examples exist of the reintroduction of the desired species without due consideration for habitat structure or processes. Several risks are associated with this approach. Each species introduced will need an optimum set of environmental conditions to which it is better suited than any existing competitors if it is not to be out competed. Without a diversity in habitat structure the few species best suited to the environmental conditions may out compete and dominate others.

Perhaps the most common approach to ecosystem creation focuses on the construction of habitat structure. This approach relies on the ‘Field of Dreams’ hypothesis (Palmer et al., 1997), which asserts that given the appropriate environmental conditions colonists will arrive and the ecosystem will develop. As the phrase ‘field of dreams’ implies, this adopting this approach for a project design might be considered optimistic. There are situations to which this approach is well suited. In areas with closely located species pools that would ensure an abundant supply of colonists, assuming the pool contains the desirable species. In areas where disturbance processes are relatively weak; unable to shape the habitat and unlikely to undo the effort of construction. This approach has been applied in many lowland river restoration projects.

The final design approach focuses on the reinstatement of processes and is based on theories of disturbance ecology. The importance of physical processes in the creation of ecosystems has often been overlooked. Where processes are weak or slow this may not be a problem and often where processes are highly dependent on morphology the recreation of processes may be implicit. Biological processes may be more problematic. They rely on complex interactions between all three elements to the habitat and could be seen as the holy grail of the ecosystem. The time taken for succession and the growth of many species to maturity (such as trees) before natural processes are successfully reinstated may be an argument for the initial floristics approach.

The importance of processes in sustaining the physical features of an ecosystem should not be underestimated and will be important for the long term sustainability of the system and minimising maintenance and associated costs. The existence of processes operating at different scales may also be important in creating heterogeneity ultimately insuring the creation of a biologically diverse system (Chapter 3)

Key to channel construction is the premise that an ecosystem will develop around the man-made habitat. Greater understanding of rates, patterns and processes of ecosystem development in newly formed habitat is needed to guide the design and maximise the success of river realignment projects. It is also important, if realignment is to be implemented as a restoration option, that strong, supporting, scientific evidence exists.

There is much discussion over the relative strengths as weaknesses of different approaches. Despite the debate, the field of dreams hypothesis (Palmer et al., 1997, Bond & Lake, 2003) it is widely adopted as an acceptable approach recently backed up by a number of high profile river restoration projects (Pedersen et al., 2007) The extent of physical habitat development is likely to depend on the success to which natural geomorphological processes have been reinstated. Whilst much progress has been made in recent years regarding interactions between fluvial processes and river channel form (e.g. Weichert et al., 2008), little work has been published on the relevance of theories to large scale channel construction projects. The development of the physical habitat will have important implications for the subsequent development of ecosystem function and the provision of ecosystem services.

4.2 Colonisation

4.3 Ecosystem development in new channels.

Many factors affect the development of the river ecosystem within man-made channels, in both the short and long term. The design approach adopted for the River Nith focused mainly on reinstatement of processes. Although some seeding was undertaken at the site, it was not a priority for the river corridor area. Consequently, ecosystem development will be dependent on natural processes for the formation of habitat features together and the arrival and establishment of biota. This chapter focuses on the colonisation aspect (Factors relating to the development of habitat features are discussed in chapter 3).

The process of colonisation has two stages, each with distinct ecological circumstances. Immigration; the act of individual colonists departing from the source pool and following a dispersal routes to the new habitat, will depend on the condition of the source environment and the availability of pathways. Establishment; the survival and propagation of colonists in the new habitat, will depend on the suitability of the new habitat. Rates of colonisation will be specific to species or groups of species depending on life history, densities and trait characteristics, and are expected to have a major influence over community structure. The physical habitat and ecosystem that develops around it will determine which taxa are able to colonise and settle permanently and these in turn may affect the availability of niche space to further waves of colonisation. The relative importance of different factors will vary both spatially and temporally and the return frequency of disturbance flows is likely to have a major influence.

<u>In the short term</u>, colonisation will be the significant agent in ecosystem development. This Contributing factors will include the rarity of long distance movements by individuals of most taxa (Bond & Lake, 2003) and the existence of hydrological connectivity (Lake 2001). When and by what means taxa colonise, affected by activity levels and life cycles of individuals, will be reflected in the relative rates of increase in species density (Mackay,1992). Many barriers to development exist at this stage with the potential to limit development (Bond & Lake, 2003).

<u>Long-term development</u> may be more dependent on habitat development (Suren et al., 2005), hydrological factors such as the frequency and size of disturbances (Brown, 2007) and ecological factors such as the presence of resources and predators (Jansson et al., 2007). In restoration ecology much disagreement over the relative importance of each exists. The importance of resource acquisition and community interactions emphasised by niche theory have been questioned by theories concerning disturbance ecology (Scarsbrook and Townsend, 1993). A lack of empirical evidence on both sides has hampered the debate making the true nature behind benthic community structure unclear (Gortz, 1998) although it is likely that a combination of factors is important (Lake et al., 2007). Physical and ecological barriers, existing at a range of scales, have the potential to further limit ecosystem development (Bond & Lake, 2003). The connection of an engineered channel into the Nith

catchment provides the opportunity to investigate ecological hypotheses relating to ecosystem development and potentially contribute to adaptive management strategies.

The following areas of research will therefore be important to this study
- Colonisation
- Habitat development
- Community-species-habitat interactions

4.4 Colonisation and establishment

Colonisation, as a major processes driving the ecological development of the engineered channel, requires as a first immigration of colonists.

The first stage in the biotic stages of ecosystem development will be the immigration of colonists. However, the relationship between dispersal/colonisation dynamics and habitat arrangement, especially concerning new habitat space, is poorly understood in most ecosystems (Palmer et al., 1997). Yet they are key to basic assumptions behind ecosystem architecture projects. Although many studies have investigated the factors that affect colonisation pathways such as propensity to drift by invertebrates (Elliot, 2002, Lauridsen & Friberg, 2005) and plant propagules (Riis & Jensen, 2006), aerial movement of invertebrate adults and wind blow seeds, and swimming by invertebrates (Humphries, 2002); there has in the past been limited published research into how these processes contribute to the colonisation of bare streambed and bank substrates in channels (Mackay, 1992). More recently, the adoption of channel construction as a restoration technique has seen a flurry of publications (Friberg et al., 1998; Kondolf, 2006; Pedersen et al., 2007) but studies have generally been limited in temporal and spatial extent.

Factors that will affect success include: taxa present in source habitat their traits and life cycles; habitat and community conditions in the source pools, specifically their effect on departure rates; the available pathways of colonisation and how conducive they are to dispersal; and factors affecting settlement and establishment of colonists such as retention and habitat condition.

4.4.1 **The source population**

Almost all organisms have evolved a well developed dispersal strategy. There are a number of hypothesised reasons why such adaptations might confer advantage. Arguments commonly focus on evading predation, genetic dispersal, maintenance of genetic diversity, poor resource value and resource crowding (Bullock, 2002). These strategies also allow flora and fauna to colonise and re-colonise habitat and where they affect the rates of entry to the drift then they will have an influence over rates and patterns. As a result the species present in source pools and their associated traits and life histories will affect the colonisation pathways available and the speed at which they move along them, ultimately affecting how rapidly a new ecosystem is able to develop. Friberg et al. (1998) noted a difference in the colonisation rates between species with less mobile taking three years or longer to move into the restored reaches. Species with poor dispersal rates included the leach *Glossiphonia complanata* (L.) and the Beetle *Limnious volkmari* Panzer. The location of the species pool is equally important. If it has become degraded, has been lost altogether or is simply to far away to allow colonists to reach the restored area there is the potential for limited project success.

4.4.2 Pathways

Colonists will enter the diversion along a number of routes or pathways. These are likely to be principally aquatic or aerial. Whilst invertebrates and plants (the focus of this study) are very different they move along very similar colonisation pathways since both are highly susceptible to the unidirectional forces of flow and wind. A range of studies have identified hydrochory as an important dispersal mechanism for many aquatic and riparian plant species (Merrit & Wohl, 2006; Riis & Jensen, 2006) whilst the equivalent for invertebrates - drift – is argued to be the most important factor in colonisation of fresh benthic habitat (Bird & Hynes, 1981). Colonisation is likely to be predominantly in the downstream direction (Elliott, 2003) although upstream movement has been observed in some species of insect by swimming and crawling (Williams & Hynes, 1978)[1]. Aerial dispersal is the second major pathway of colonisation in both plants and insects (Madsen et al., 1977; Malmqvist, 2002). Both are highly dependent on lifecycle patterns.

Along the river

Downstream

Invertebrates

Many studies report drift as the most significant path of colonisation by invertebrates. The propensity for individual taxa to colonise via drift depends on a number of factors. Behaviour patterns and life cycle stage can have a strong influence (Townsend & Hildrew, 1976; Bird and Hynes, 1981). Much of the research in this area focuses on invertebrate densities with mixed results producing disagreement over the role of density dependent processes (Elliott, 1971), which continues today (ref). In a study of the mayfly *Baetis rhodani* (L.), Humphries (2002) makes a case for density dependent dispersal. Other studies show the proportion of larvae entering the drift has been shown to be independent of the benthos density and amount of detritus for two different species of mayfly; *Baetis acaudatus* Dodds and *Ephemerella inermis* Eaton (Ciborowski) (Wise, 1980)

---

[1] Drift was found to be the most important pathway for colonisation on the River Nith's namesake in Ontario, US, contributing 41.4% of the colonising invertebrates in a 28 day period (Williams and Hynes, 1976). Significant numbers were also found to use other pathways including upstream movement (18.2%), vertical movement from the hyporeos and substrate (19.1%) and aerial colonisation[1] (28.1%).

Under certain conditions it may be disadvantageous to drift such as during the day where predators are active or during dangerous spate flows. Adaptations that enable individuals to avoid entering the drift, control drifting rates and return to the substrate will also affect rates of colonisation. These can be behavioural (e.g. moving to refugia, cementing) or morphological (eg streamlined body shape, anchoring). *Baetis spp.* exhibit a range of behaviours[2] that could prolong or reduce the time spent in the drift (Campbell, 1985; Bird and Hynes 1981). Other studies have observed similar behavioural traits in Numouridae (Lancaster et al. 1996). Poff and Ward (1991) showed it was possible to alter the drifting behaviour of a range of species by varying velocity, the drift densities of most taxa increasing with reduced flow and a few decreasing. Elliot (2002) found a reduction in the amount of time spent in the drift at a site that had macrophyte growth. Rates and patterns of entering drift in this way are species specific. The presence of plants reduced the time *Baetis rhodani* (L.) and *Gammarus pulex* spent drifting by 50% whereas Simmulium and *Ephemerella ignita* spent 80% less time drifting (Elliot, 2002). Similar effects have been demonstrated using artificial plants (Corkum and Clifford, 1980).

The timing, duration, magnitude and rate of change of flow form the hydrologic regime of a stream. They are key factors in the distance travelled by invertebrates (Malmqvist, 2002; James et al., 2009) and may explain why estimates of the distances drifted by invertebrates vary so considerably. In early studies (Waters 1965; Mclay, 1970) some invertebrates were observed to drift over 100m. Later experiments showed drift to be fairly unimportant with invertebrates drifting little over 2m (Townsend and Hildrew, 1976). Interpretation of results may also help explain the discrepancies. Despite placing more emphasis on average times spent in the drift (which are considerably lower), Elliot (2002) reports drifting times of well over 2 minutes for a small proportion of individuals from a wide range of species giving them the potential to drift 100s of meters. So while the conclusions highlight that the majority of invertebrates returning to the substratum after travelling only short distances, given that the dispersal of a species requires only a small proportion of individuals to remain drifting for lengthy periods, the more intrepid few may have been more significant. Individuals of Baetis and Gammarus taxa remains drifting for 43 seconds. Despite being just 1% of released individuals this still represents a significant potential to colonise new habitat. 1% of both *Ephemerella ignita* (Poda) and *Hydropsyche siltalai* Döhler drifted for over 100 seconds and *Ecdyonurus venosus* (Fabr.) for 76 seconds.

Density dependent drifting has ben observed in Plectonemia conspersa which is competitive for net spinning space (Matczak and Macaky, 1990). A number of New Zealand species demonstrated refugia seeking behaviour including rapid migration into the hyporheos by the common mud snail *Potamopyrgus* and absailing to more sheltered patches by the common caddis fly larvae *Pycnocentrodes* using silk-like drag lines (Holomuzki & Biggs, 2000). Behaviour was in response to increasing skin friction as flow velocity increased with increasing water levels.

---

[2] Behaviours of both *Baetis muticus* (L.) and *Baetis rhodani* (L.) included swimming in various directions, and parachuting by spreading there legs out and lifting their abdomens (Cambell, 1985) Or somersaulting there way to the substrate (Bird and Hynes, 1981)

Plants

Like invertebrates the majority of seeds of riparian species are transported on the water surface or in suspension (Merrit & Wohl, 2006). However, many of the early plant colonists to the riparian zone may not be riparian species. Many of the species well adapted to colonising newly opened habitat space have aerially dispersed seeds. The obvious divide is between those adapted dispersal by hydrochory and those with structures that assist aerial dispersal (discussed later). Species well adapted to dispersal in the water include species of willow, found to be the most abundant taxa in hydrochic samples from mid May to mid July (Merrit and Wohl, 2006)

seed buoyancy and season are likely to be overriding factors. Propagules depart from or are washed from the habitat and deposited downstream. Morphological adaptations of the same kind are seen in the shape and structure of seeds.

Hydrologic regime is equally important for the distance travelled and final location of seeds deposited to the riparian zone and river bank margins (Merrit & Wohl, 2002). Floods can contribute to long distance dispersal bringing seeds from tens of kilometres away (Tabacchi et al., 2005) They have been found to contribute high numbers of species in low densities, particularly outside the growing season (Tabacchi et al., 2005). However the relationship between seed inputs and extant vegetation may be relatively weak (Thompson & Grime, 1979). Tabacchi et al. (2005) found that while flood events are likely to increase the diversity of seed inputs to the edges of the riparian corridor they have little effect on the riparian vegetation that develops on the floodplain side of the ‘riparian transverse gradient’. By 1992 the ecological implications of dispersal along water courses had never been evaluated for seeds and propagules (Nilsson, 1992)

Lancaster et al. (1996) conject that the return rates/ drift distances “of the entire assemblage are an average of the component species’ return rates weighted by their relative abundance in the community and constrained by channel hydraulics”[3]. Although there is currently little supporting evidence, it is easy to conject that all the factors which play a role in drift densities are going to affect rates and patterns of development. Taxa commonly listed as early colonists of bare substrate include Baetidae, Gammarus, Simulidae, Chironomidae (Mackay, 1992). These same taxa have been observed to reach high densities in the drift. A more detailed investigation of which species arrive when will help develop these theories.

The substrate also provides a pathway for seed movement. Transport along the bed of the stream can be as high as 25% of incoming seeds (Merrit & Wohl, 2006). Whilst driven by flow the properties of the riverbed are likely to affect the pathway. Disturbance of sediments by floods can also redistribute

---

[3] Larkin and McKone (xx) put this another way. “…a hodge-podge phenomena” proclaiming “there is little point in trying to be more profound”. Whilst others in the field are clearly agitated by this “defeatist attitude” (Lancaster et al., 1996) it does serve to highlight the difficulty in elucidating processes in such a variable and complex environment.

stored seeds and propagules again providing a role for substrate character in influencing colonisation rates.

An important aspect of drift which will affect the rates and patterns of development within engineered channels is the seasonal variability known to be significant in many species (Robinson et al., 2004). The drift pathway of recolonisation has been reported as particularly important for relatively sedentary invertebrates such as filter feeding caddisflies, black flies and chironomids (Mackay, 1992)

Catastrophic drift generally recognised for the potential to clear areas of benthic fauna (Gibbins et al., 2007) can also relocate surviving individuals to new areas of substrate. Mobilisation of bed material, large woody debris, organic matter and disturbance of refugia during large spate events. Morphological and behavioural adaptations are likely to be less effective in such circumstances. However, Baetis has been shown to drift in large numbers during low flows and possibly seek shelter during high flows explaining their low numbers recorded during a spate flow (Bird and Hynes, 1981; Weninger 1968; Cokum et al., 1977; Walters 1969)

The question remains; does Hydrochory matter to plant populations, plant communities and riparian ecosystems, and if so in what ways? (Merrit & Wohl, 2006). Merrit & Wohl (2006) found that most Hydrochoric plants occur within about 2m of the stream channel and dramatically reduce with increasing distance.

Upstream

Disagreements regarding the upstream dispersal abilities of invertebrates have been around for as long as the studies themselves and have yet to be resolved. Neave (1930) observed the large numbers of the mayfly *Blastrurus cupidus* Say (now *Leptophlebia cupidus*) moving over 1km upstream in as little as few months to colonise newly wetted channels. This, he explains, is “not in any way a tactic response, but is the result of an internal instinct working in combination with the physical structure of the animal”. Other early studies find little evidence but are reluctant to rule out upstream dispersal (e.g. Bishop and Bishop, 1968)

Bird and Hynes (1981) assess the significance of upstream and other movements of freshwater insects. While they found drift to be the predominant mode of dispersal they also provide evidence for significant upstream and lateral movements. As a proportion of the movements recorded, those in an upstream direction accounted for between 2.1% and 15.2%. They found no difference between upstream and across stream movements and conclude that upstream movement is purely a random event. Random or otherwise, this still suggests it may be an important pathway for colonisation of new habitat Higher levels (25.2%) have been reported elsewhere (Elliot, 1971). This would of course be dependent on the drift levels at the time, which are known to vary diurnally and with season.

Mayflies ability to swim by dorsal ventral undulations makes them strong swimmers (ref). Specific family groups known to be vigorous swimmers include Baetidae, Siphlonuridae, and some species within Oligoneuridae and Gammarus (ref ref). As a result these groups might be expected to colonise by swimming from downstream species pools, ahead of groups that require aerial pathways at the adult lifestage. However crawling, an option open to nearly all taxa, may be more significant. Colonisation of bare substrate areas by crawling invertebrates has been shown to be a small but significant pathway. Individuals can spread as a result of behavioural migration or random movement. Random movement by crawling or swimming will result in a net movement from areas of high density to areas of low density. Although movement is likely only over small distances it can occur during all season and in the upstream as well as downstream direction (Williams & Hynes, 1978). Conclusions of past studies are uncertain. Doeg et al. (1989) highlight the potential of experimental design to influence conclusions. They report that some substrate holding trays hinder the accessibility to crawling invertebrates. In a study if the Acheron River, Australia, they found drift contributed less than 50% of immigrants over a 10 day period.

Upstream movement has been linked to the life stages of the invertebrate larvae. High values of upstream movement seem to correspond to early instars (Bishop and Hynes 1969; Hultin et al., 1969) and to large individuals about to emerge (Hultin et al., 1969; Ulfstrand, 1968; Elliott, 1971) The swimming behaviour of some Leptophlebidae is linked to periods of their life cycle (Söderstrom, 1987)

Aerial pathways

Invertebrates

Dispersal by via aerial pathways will make a significant contribution to colonisation patterns. There has been considerable interest in the upstream dispersal of insect as a result of the ‘Drift Paradox’ (Humphries & Ruxton, 2001; Pachepsky et al., 2005). Regardless of whether or not aerial dispersal counters any effect that drifting might have on invertebrate populations[4], research in this area has demonstrated the potential of aerial dispersal to act as a significant colonisation pathway and show the movement of adults insects over considerable distances. Aerial routes are particularly significant, for all propagules, as they provide for downstream reaches as significant species pools, otherwise limited by the unidirectional nature of flow. There will be strong temporal patterns in the importance of this pathway.

Patterns of aerial dispersal by adult insects will be strongly influenced by habitat development. Most are weak flyers and as such will be affected local air movements (Bond & Lake, 2003). For these taxa development of the riparian corridor will be essential to encourage movement of adults into the realigned reaches, an important aspect of population dynamics and full ecological functioning.

---

[4] Results from studies by Masden and Butz (1976) support the view that there is a pronounced tendency for egg baring females of *Brachyptera risi* to fly upstream

Vegetation

In studies of dispersal Poaceae, Betulaceae, Asteraceae and Salicaceae were all well represented in the aerial seed rain (Merrit & Wohl, 2006). Aerially dispersed seeds were equally likely to occur at the stream edge as they were far from the channel (Merrit & Wohl 2006). Species of willow were the most abundant species to be found in hyrochoric samples and peaked between mid May and mid July (Merrit & Wohl, 2006)

4.4.3 Habitat condition on arrival

Colonisation depends on invertebrate mobility along the pathways discussed above. Additionally substrate texture and associated food supplies, competition and predation may affect rates of colonisation (Mackay, 1992)

- Ovipositing adult female Hydropsychids favour smooth clean surfaces, tending to lay their eggs on the smooth underside of rocks that are free from silt (Mackay, 1992)
- Filter feeders such as Simulidae are able to take early advantage of bare scoured substrates which have been flushed clean of detritus and epilithon because they use the stream flow as their source of food. (Mackay 1992)
- Some species of baetidae are able to feed on the thinnest of epilithic films before the development of an algal layer (Boulton et al. 1988) These can develop very rapidly. Levels have been observed to recover within 13 days following disturbance and reach maximum primary productivity within 21 days (Osbourn, 1983)
- Hydropsychid caddis fly larvae require surface irregularities for the attachment of their retreats and capture nets (Mackay, 1992)
- Dominant predators tend to be later immigrants (Refxx) which could provide an initial period of reduced predation pressure.
- Investigations into the importance of detritus have drawn various conclusions. Peckasky only during summer in a stream bed with minimal leaf litter input, did detritus increase colonisation. Other studies have found detritus levels to be of overriding importance (Culp et al., 1983).
- Peckasky (1985) found that predaceous perlids and perlodids consistently reduce the density and therefore rate of prey-community establishment including Baetis and Ephemerella species as well as Simulium, Chironomids and others

4.4.4 Limitations to ecosystem development

Lake and Bond (2003) present five broad issues covering the different barriers to restoration projects. These have been modified below to cover the potential issues relating specifically to the development of engineered channels.

1. Barriers to the dispersal of biota
2. Existence of the full range of habitats used by different life stages
3. Competition from introduced species/early vigorous colonisers
4. Long-term or large scale driving processes
5. The large scale of projects

4.4.4.1 Barriers to dispersal

Results of Stein et al. (2008) stress the serious consequences of dispersal limitation for local species diversity and for ecosystem processes. They showed it was possible to enhance both diversity and productivity by the introduction of dispersal-limited and seed-limited species. The limitation on local species richness imposed by the availability of seeds has been demonstrated in both lab and field based studies (Stein et al., 2008). As dispersal will largely be along the stream channel longitudinal connectivity will be critically important (Wiens, 2002). Barriers to dispersal may include both hard (dams, wiers) and soft such as isolation and sheer distance or poor stream habitat in intervening reaches. High flows will play a key role in overcoming these obstacles. Propagule dispersal in plants can vary considerably between years (Andersson and Nilsson, 2002)

4.4.5 Big picture

It is important that linkages with the catchment are recognised, Catchment landuse is usually a dominant driver in stream condition (Lake 2001) and can continue to dominate ecosystem patterns many years later (Harding et al., 1998). Evaluation of habitat allows us to estimate the potential impacts of different management alternatives, plan conservation and compensation strategies to reduce habitat losses. Many communities exist in perpetual states of non-equilibrium or dynamic equilibria where natural disturbance prevents most populations from reaching maximum densities (Palmer et al., 1997). However it does not necessarily follow that species interactions are not important in shaping community structure. Variabillity, both physical and biological, is part of nature (Palmer et al., 1997)

Ecosystems have long been exploited as a resource (McNeil, 2000) It is only the relatively recent realisation that these resources are finite that we have come to view then more as a service provided by the ecosystem. Potential services provided by functioning stream ecosystems include

i) Flood retention
ii) Support of economically valuable fish species
iii) Amenity value
iv) Scientific knowledge
v) Support of species with tourism value
vi) Support of rare or endanger species with intrinsic value

Whilst there is still value in many heavily modified environments they are by definition different from the genuine article

CHAPTER 5

MODELS AND HYPOTHESES

This chapter starts by setting out the importance of hypothesis testing in progressing science then looks at the difficulties of testing them in a natural setting. It presents the models and hypotheses to be investigated which relate to the development of physical habitat, riverbed invertebrate communities and riverbank vegetation. The rational behind models and hypotheses is also given.

Aims

To establish the validity of testing hypotheses on the Nith realignment
To set out clear goals and objectives of the research
To develop a model for the development of a functioning stream ecosystem
To develop a set of hypotheses and explain the rational behind them

5.1 Hypotheses and Models

5.1.1 The importance of hypothesis testing

While there has been increasing interest in restoration, it is clear that there has not been an accompanying increase in knowledge and the development of principles of restoration ecology, much to the frustration of many experts in the field (Hobbs & Norton, 1996; Palmer et al., 1997; Ehrenfeld, 2000, Lake, 2001). According to Lake (2001), understanding the causal links and critical steps of ecosystem development requires well-designed hypothesis testing experiments at small scales rather than the generation of models and simply carrying out monitoring. The view is echoed elsewhere in the literature; Palmer et al., (1997), argue that reporting the results from soundly designed monitoring projects is the only way restoration ecology is going to make significant progress as a serious science. Hypothesis testing will play an important role in closing the knowledge gap but is rare in restoration projects (Lake 2001). This includes realignment projects, a commonly used tool in the restoration of streams. Michener (1997) highlights the opportunity that hypothesis testing in restoration projects provides for advancing more general ecological theory. However, the presumptions made in following a single hypothesis may lead to an overly simple conclusion (Chamberlin, 1897). Adequate explanations for a complicated phenomenon may be necessarily complex. The alternative involves modelling a range of responses to a range of factors. Models can offer greater explanatory power and provide general rules, compared to the specific nature of hypothesis, but they are more difficult to falsify as they make less specific predictions, and as such, may struggle to provide much mechanistic understanding (Michener, 1997).

The nature of ecosystem architecture type projects, such as restoration or realignment of rivers, make them ideal for testing important models and hypotheses (Palmer et al., 1997) for a number of reasons.

i) Many ecosystem processes occur over large spatial and temporal scales making their study in a laboratory impractical, whereas field observations, when combined with information from the literature, offers a powerful comparative approach (Southwood, 1988).
ii) A single large-scale project allows a graded sequence of hypotheses from the short term to the long term may be considered together to cover the different rates of response of different elements at different scales within the ecosystem (Lake, 2001).
iii) Long-term studies offer a relatively unique opportunity to study questions related to slow processes, rare events, episodic phenomena, processes with high variability, subtle processes and complex phenomena (Michener, 1997).

5.1.2 Difficulties of hypothesis testing

Poor design, planning and implementation of research of projects is considered a major obstacle to the contribution of projects [like the Nith diversion] to ecology (Lake 2001). A strong starting point is to work with a hypothesis or model and view the realignment as an experiment on a large scale. All the same rules about replication and control apply. The lack of input into design and implementation of the projects can present an enormous challenge in this respect (Michener, 1997). However careful design can ensure conclusions are robust and not vulnerable to serious criticism. This will also provide a clearer picture of project success. The nature of large scale 'ecosystem architecture projects': tight time scales, lack of communication between disciplines, limited funds and reluctance to evaluate results, all present challenges to the experimental approach. Such challenges or constraints and are far from ideal, often including lack of experimental control and insufficient replication. That said, such problems are not unique to ecology (Michener, 1997)

5.1.3 The value of hypothesis testing

The reasons for persevering with research into large-scale ecosystem projects are manyfold. Realignment is one of the many tools used in the restoration of streams. It is frequently the only option in the restoration of canalised rivers and has been practiced at very large scales. The potential impact can be huge. Testing hypotheses that explore ecological models, for example the ecology of macroinvertebrate and plant dispersal along stream corridors, will improve understanding of the ability of stream ecosystems to regenerate within an engineered channel and the value of designs based around sound geomorphological principles. This type of community ecological theory has an extremely relevant role to play in restoration ecology (Palmer et al., 1997). Knowledge is relevant to restoration practitioners and ecologist alike. Learning from past river architecture projects through the study of ecosystem development will be key to the improved success of future projects (McDonald, 2004).

5.2 Habitat development

Regardless of how many individual organisms arrive in the realignment, they cannot be considered 'colonists' unless they are able to establish a persistent community. For some species, well adapted to colonising new areas, the early habitat condition may represent suitable territory. However, the habitat, as constructed, is unlikely to be very diverse or to offer many niches for colonisation. Desirable levels of colonisation and increases in ecosystem diversity are unlikely to occur without some degree of habitat development. A conceptual model is presented in figure 5.1 that links the project design criteria to fluvial and riparian habitats that are expected to develop. The effectiveness of implementing these design criteria will influence the rate and extent to which these habitats develop and are sustained. The criteria that should be incorporated into the design of an upland gravel bed river are shown at the centre of the diagram in blue text. Major habitat units are shown in black text and are illustrated in the photographs. Linking the design criteria to the habitats are the ecosystem processes and interactions, which are expected to develop (green text); the success to which these processes are triggered will depend on the implementation of the design and will ultimately determine whether or not the diversion is colonised to its full potential.

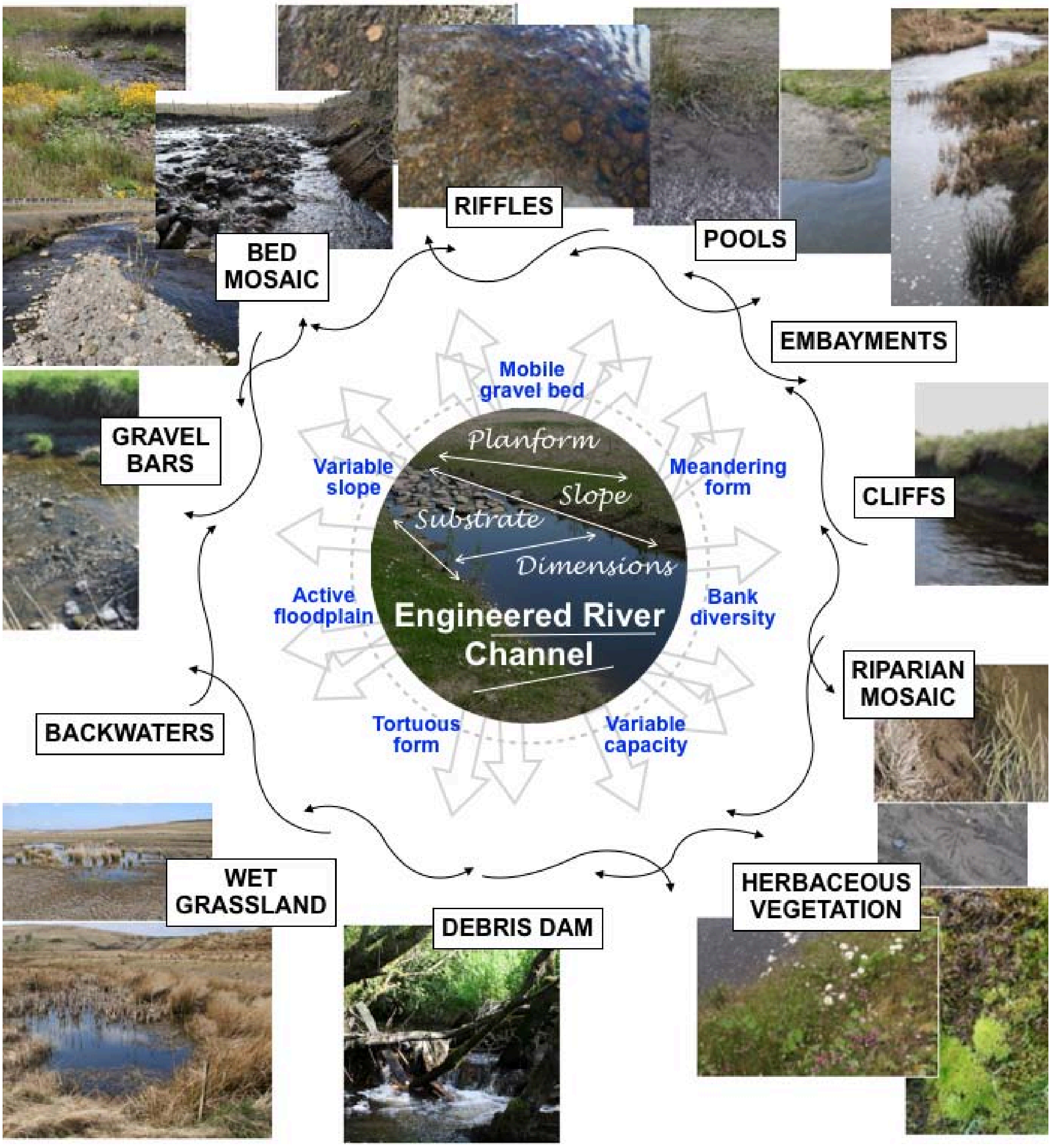


Figure 5.1. Conceptual model linking the design considerations for the engineered channel (blue) through the triggered processes (red) to the habitats that are expected to develop within the system (black). Habitats are illustrated in the photographs

5.3 Dispersal pathways

Barriers to dispersal are frequently cited as a major obstacle to the success of ecosystem architecture projects (Chapter 4). However they are not on/off factors; barriers tend not to be either present or absent, rather they can be considered as pressures that reduce the availability of a pathway to an individual. Many barriers may be countered by species-specific adaptations that have evolved to increase an individual's dispersal ability. Alternatively, where a barrier results from poor habitat quality, elements of project design may be aimed at countering its influence.

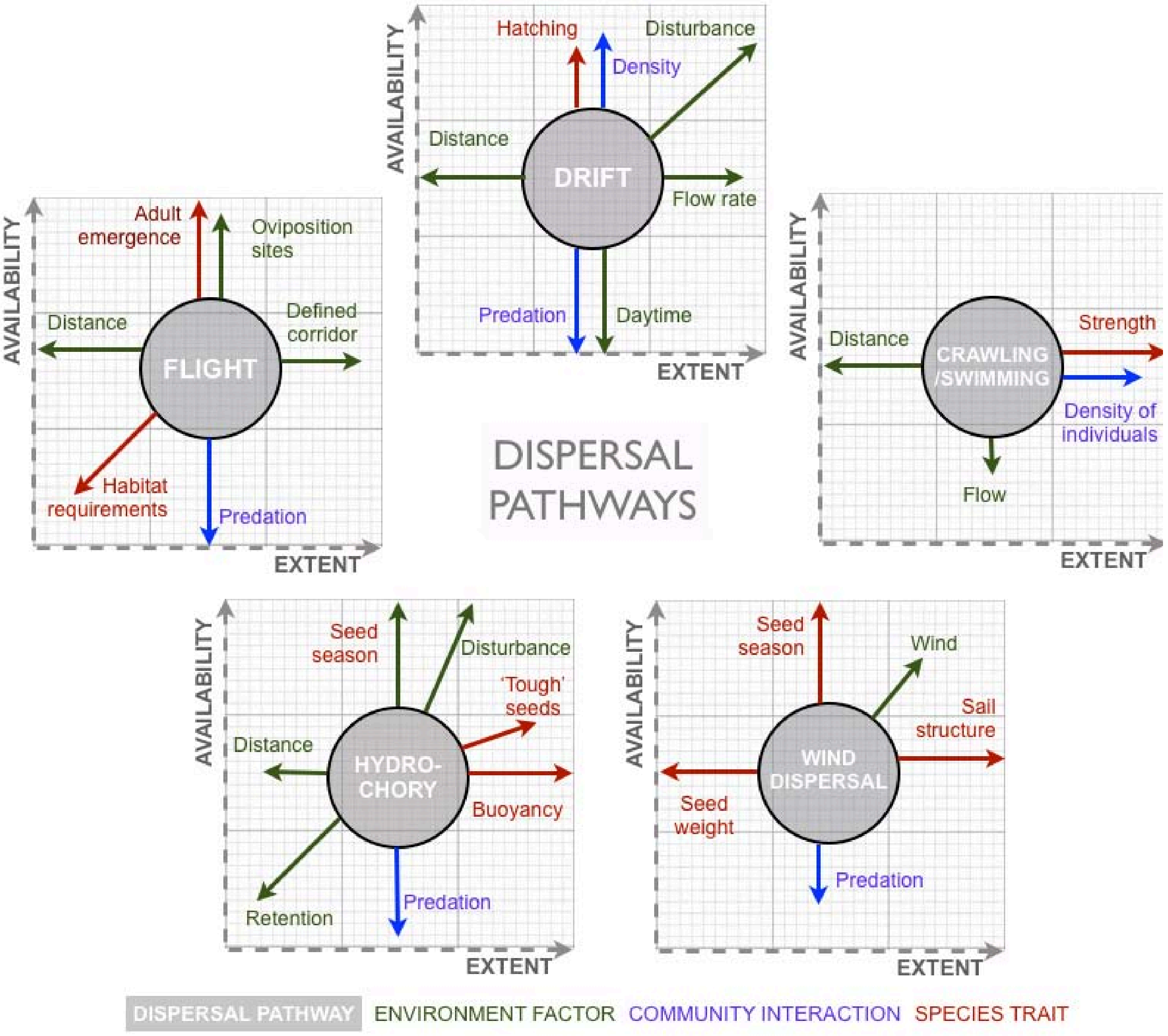


Figure 5.2. Conceptual model showing the potential influence of environmental and life-history factors on the number of individuals that can make use of the principle dispersal pathways and the effectiveness of a pathway in opening up new areas for colonisation.

Figure 5.2 shows a conceptual model of dispersal highlighting the important influences on the number of individuals to which pathways are available and how effective, or useful, the pathway is likely to be. The model shows the pathways, expected to be important in the development of a diverted stream or river, as grey circles. Their position on two axes, effectiveness and quantity, indicates their availability to colonists based on how many individuals are likely to use the pathway and how far it will get them; they might be taken to indicate the relative importance of each. However, positioning will be unique to each species and depend on the influence of each of the factors shown. These may be environmental barriers inhibiting movement, species traits permitting movement or community interactions that affect the likelihood of an individual using or surviving along a dispersal pathway.

Habitat development (figure 5.1) and dispersal (figure 5.2) are two meta-processes assumed to occur in many ecosystem architecture projects. They are essential to the development and sustainment of a functioning ecosystem and together will control the rate at which it develops. Despite their importance there is little empirical evidence in the literature that explicitly demonstrates to what extent they can be relied upon when creating an ecosystem.

5.4 Geomorphology

Changes in the condition and complexity of the physical habitat will have implications for ecosystem development. This is an area of uncertainty in restoration projects which are frequently undertaken without sound understanding of the likely rates and patterns of development, or an appreciation of the processes that will be important to the project’s success (Kondolf, 2000). It is likely to vary between river systems depending on hydraulic and geophysical character. The model in figure 5.1 relates some of the geomorphological features associated with upland gravel bed rivers to factors that can be considered in the design of a river channel. Two general hypotheses that relate to the model presented in Figure 5.1 are considered here.

1. From a homogeneous starting point, the 3km of engineered channel will develop a diverse and heterogeneous form comparable to a natural sinuous pool riffle river.

With respect to this, we would expect to observe gravel bar development, variable channel form, pool/riffle features and development of substrate diversity. Development is expected with the first major mobilisation of the gravel bed. In such a flashy catchment this is expected to occur during the early life of the channel. Physical habitat diversity will be important both along the length and across the channel. Rivers are naturally both highly patchy and highly dynamic environments. This should therefore be detectable longitudinally along the path of the thalweg. A diversity of channel forms will be important in the development of a biodiverse riparian zone and ultimately the health of the river. Near uniform banks were constructed under the assumption that a more natural form would develop.

A second hypothesis following on from the first, relates to the specific elements of the design included to propagate or enhance the process of development towards a natural form.

2. The influence of design specifications, included to promote development towards a natural form and propagate ecosystem development, will be evident in the pattern of geomorphological development

Potential design specifications based on combinations of channel shape, planform, slope and substrate are shown in figure 5.1 (blue text). The design for the realignment of the Nith included three of these; a meandering form, mobile gravel bed and a variable slope (approximating to the gradient of the natural channel). Sediment mobility and the sinuous planform are important features that, together with natural hydrological variability, will help to establish the desired physically and biologically diverse stream habitat. Additionally, construction included the creation of pool and riffle features, rock armouring of high energy meander bends, geotextile protection of the river banks and a program of seeding and planting. Many design recommendations implemented in ecosystem architecture projects currently lack the scientific foundation which monitoring of their influence could provide. Results will have implications for conflicting theories on the importance of habitat creation and species introductions.

Answering these two linked hypotheses will require the investigation of changes in channel form, including profiles of channel cross-sections and thalweg and changes in the sediment structure of the installed gravel bed. It would be fair to ask if this design approach can be relied upon to create the desired morphology and level of diversity. By testing this assumption it is hoped to better inform and guide future designs.

5.5 Invertebrates

Surveys of benthic invertebrate communities will be used to investigate a number of hypotheses relating to rates, patterns and mechanisms of dispersal and establishment. It is often assumed that these processes are strong enough for the colonisation of man-made habitats despite little documented evidence. The model presented in figure 5.2 summarises the major factors likely to influence colonisation of the channel and species traits with the potential to counter the barriers. We investigated two interrelated hypotheses relating to colonisation.

3. There are predictable environmental barriers to colonisation, which affect the dispersal and establishment of macroinvertebrate colonists. These must be considered when designing ecosystem architecture projects.

4. There will be specific morphological and behavioural traits that have evolved in some species to overcome environmental barriers to colonisation affecting dispersal and establishment. Species possessing these traits will be superior colonists.

Three principle dispersal pathways will be important to the development of the realignment: drifting, crawling and flying. The effectiveness of each pathway will depend on environmental factors or barriers such as distance and season. Distance is expected to be a major barrier in relation to all pathways with more remote sites being colonised at a much slower rate than sites close to the species pool. Those species with adaptations suited to specific pathways are likely to appear on the diversion earlier than those that are more poorly adapted. The results from drift experiments suggest that downstream drift of invertebrates will be the most important pathway of colonisation (Chapter 4), at least initially. This should be evident as a higher rate of colonisation at upstream sites and will result in an inverse relationship between species diversity and distance from the upstream source. This relationship has been observed in a number of man-made channels (Gore, 1982; Griffin 2004) but not extensively investigated. If the rate of increase in richness at any given site along the diversion is solely related to distance from the upstream species pool this would confirm the overriding importance of this colonisation pathway. However it is unlikely that the process of ecosystem development will be so simple. The development of physical habitat and the contribution of other dispersal pathways are likely to interrupt any relationship with downstream distance. Considered in this respect, the distance of downstream sites to upstream species pools is just one of many barriers that may inhibit development. At the furthest downstream locations where the influence of drift dispersal is likely to be weakest other dispersal pathways (e.g. upstream crawling and swimming) may contribute greater numbers of individuals. Likewise during the summer months when many of the invertebrates emerge as adults, aerial dispersal is likely to be a significant factor.

Pathways are likely to be used differently by invertebrates depending on their species traits and life cycle stage and, as such, will be important to different degrees at different times. Species reported as long drifters, having high drift abundance and being rapid dispersers are expected to appear in the diversion more rapidly than species reported to have poor dispersal abilities. If some species are reaching the bottom end of the realignment more rapidly than others, then there exists a species-specific barrier to dispersal or establishment. This could be propensity to enter the drift, habitat conditions affecting settlement, or limitations imposed by the lifecycle stage. Establishment will then depend on how well suited an organism is to the habitat that has developed and on the habitat quality (figure 5.1) with generalists initially having an advantage over species with a more specialist life history. The importance of habitat development may be indicated by differential rates of colonisation of patches that differ in habitat type, complexity or quality.

The potential barriers will affect different sites and different taxa to different degrees and at different times and are discussed in more detail in Chapter 4. Eventually ecosystem recovery will be marked by a transition from a general pattern of increasing local and regional diversity to a dynamic equilibria where the interplay between disturbance, recolonisation and community interactions produce a fluctuating diversity at a local scale, but high diversity and heterogeneity at the reach scale. By studying the invertebrate community composition and structure in different locations along the realignment and at different time points in the development of the ecosystem it will be possible to investigate the hypotheses and the implications for invertebrate ecology and future channel design.

5.6 Vegetation

In a functional riparian zone, plant species richness should vary considerably in space and time along stream margins (Naiman et al., 1993). Just as with the colonisation of the streambed by invertebrates, development of this diversity will depend on both dispersal pathways and habitat development and on the strategies and life histories that have evolved in the colonising flora The creation of what is in effect a new riparian zone allows both of these groups of processes to be investigated.

Studying the arrival rate and sequence of colonising flora with respect to specific dispersal pathways will allow the investigation of the following hypothesis and the importance of dispersal in the development of ecosystem architecture projects

5. Environmental barriers to the dispersal of plant propagules will be evident in the rate and pattern of plant colonisation, with those species better adapted to using specific dispersal pathways colonising more rapidly and extensively.

The influence of hydrochory is likely to reduce with increasing distance from the river, whereas wind dispersal provides access to all areas of the riparian zone. Furthermore hydrochory provides a uni-directional colonisation pathway compared to the omnidirectional dispersal provided by the wind. Each pathway requires specific adaptations to ensure propagule viability on arrival. The influence of hydrochory will decrease downstream (i.e. at the upstream end the majority of colonists will be water dispersed while species at the downstream end must be dispersed by wind or animals) initially, but this pattern will fade with the establishment of new proapgule sources distributed along the diversion.

6. Environmental barriers exist to the establishment of plant propagules will be evident in the rate and pattern of plant colonisation following patterns of habitat development and suitability as the ecosystem develops

Riparian environments are recognised as being ecotones, often with a strong physical gradient between aquatic and terrestrial habitats. As such, different plant communities are likely to develop at the top and bottom of the riverbank depending on the suitability and level of disturbance present. As colonisation sources develop within the realigned reaches, habitat factors will become increasingly important for explaining differences between sites. It is expected that the development of habitat diversity will be driven by the reworking of the riparian zone by the river. This influence is likely to reduce with increasing distance from the river. While the levels of disturbance may limit the diversity of individual quadrats, beta diversity (dissimilarity between quadrats) is likely to be higher closer to the river. The sinuous planform will play an important role in the development of physical habitat diversity.

Chapter 6

DATA COLLECTION AND ANALYSIS

This section first describes the rationale behind the research approach. It details the parameters that were measured as indicators of ecosystem development. Methods of data collection and processing are also described. Finally the indexes used to demonstrate and summarise patterns in the data and statistics used to analyses the results are discussed and explained.

6.1 RATIONAL

6.1.1 Experimental appraoch

There are many ways of collecting data to test theories about the natural world. These can be broadly classed as either experimental or observational approaches (Scheiner, 2001)

One of the most powerful is the manipulative experiment in which all variables are controlled before changing one to investigate the measured response of another. Removing the range of variability that exists in nature from ecological experiments has advantages. The ability to control all other factors provides for robust conclusions about the effect of test variable on the response variable. This is both a strength and weakness of the approach. Control of variables often requires the use of laboratories or glasshouses. Such un-natural conditions introduce the possibility that the results are not a fair reflection of the behaviour of a natural system and as a result, may not advance substantially our understanding of natural communities (Waide et al., 1999). This is, in part, because they are limited in space and time (Diamond, 1986). Large scales tend to require more observational approaches.

Observational experiments are another way of investigating phenomena. They tend to be weaker as a tool for testing specific hypotheses because of the lack of manipulative control. Conclusions may also have an element of deductive reasoning and as such are susceptible to the many pitfalls associated with a deductive approach (Muray, 2001). There are also advantages. First and foremost the results will accurately reflect real processes under ‘natural conditions’ although how transferable the results are between river catchments is a question yet to be fully addressed. Incorporating the variability in size and function of species and individuals in natural communities into the analysis takes the concept one step further and may assist in their interpretation (Waide et al., 1999). Observation experiments are particularly important in stream ecosystem ecology because of the scale over which processes operate and because ethical considerations often preclude manipulative experiments[1] although this maybe less of an issue in degraded habitats. Some authors have cited the failure to recognise the scientific value of basic monitoring restoration projects as a serious hindrance to the development of restoration ecology (Michener, 1997)

A third approach falling between manipulative and observational forms of experimental research is the ‘Natural Experiment’ (Diamond, 1986; Scheiner, 2001). In natural experiments one predicts the outcome of a naturally occurring perturbation. This category could be expanded to include large scale

---

[1] In the past there have been countless studies that have disrupted and damaged natural systems to investigate the effects of a particular chemical of the response of a particular population or community. This type of experiment now rarely occurs. There are numerous ethical implications to damaging a system that we ultimately seek to protect as well as conservation issues concerning damaging pristine habitat that may be in decline. Although whether or not this applies so strongly to degraded habitats that ecological restoration seeks to improve has been questioned, ethical considerations are, and should be, an important element of scientific discovery.

habitat manipulations as carried out in river restoration projects. As natural variability is uncontrolled in this type of approach, its usefulness in scientific investigation is reliant on the response of specific habitat characteristics being detectable above the levels of natural variability. To achieve this reference data must be collected to gauge the degree and pattern of habitat change that results from the uncontrolled variables. If environmental projects of this type are rigorously studied with robust study designs they can be of considerable scientific value to a number of areas of river restoration ecology and can be used to test specific hypotheses (Leps & Smilauer, 2003). Landscape scale manipulative experiments may be key in furthering our understanding of the importance of catchment scale processes (Merrit & Wohl, 2006) and will be required to test which aspects of alternative ecosystem state theory are important in indicating or determining states and thresholds that affect resilience to restoration efforts (Suding et al., 2004). In addition to a clear contribution to restoration ecology, results are directly transferable to the field of habitat restoration and the development of guidance for the design of future projects (Michener, 1997; McDonald, 2007).

The rerouting of the Nith should be considered as a large-scale outdoor experiment. A careful study design guided by this approach has great potential to contribute to stream ecology through the testing of specific scientific hypotheses and the development of theoretical models. The research also offers the opportunity to study and evaluate some of the methods used for measuring success of realignment projects. This should be an important aim included in the planning and implementation of stream restoration (Palmer et al., 2005)

In adopting this approach there will be a requirement to give specific consideration to a number of limitations:

1. The lack of a true control set up against which responses to the impact can be compared
2. The inability to exclude the potential confounding effects from the data.

Care can be taken to design a sampling protocol that minimises the influence that these limitations have over the results and conclusions (Leps & Smilauer, 2003)

6.1.2 **Study Site**

The research was conducted on a realigned reach of the River Nith. Relevant hydrological and geomorphological facts about the site are presented in Chapter 1 along with a geographical and historical background and an aerial photo of the realignment under construction (figure 1.2).

6.1.3 Study Design

Theory dictates optimal design. Long term monitoring of salient patterns and processes is important. Adequately replicated control and experimental units are essential (Roni et al., 2005). The appropriate spatial and temporal scales and robust statistical analysis of the results must also be considered and that all this can rarely be achieved is widely acknowledged (Michener, 1997; Roni, 2005). Constraints generally involve timing - often a project is underway before the potential for research is even considered (Palmer et al., 2005); and resources – detailed study design is both time consuming and labour intensive, especially for large scale projects. In the design of this study, consideration had to be given to the value of the data collected and the resources required for collection and processing. More often than not study design represents a trade off. The ecologist must decide between frequent sampling (at cost to extent or intensiveness), sampling many parameters (at cost to low spatial and temporal extent) or sampling many locations (sacrificing an element of intensiveness and frequency).

In this study, priority was given to the investigation of the spatial and temporal extent ecosystem development. Sampling design therefore had to match the spatial and temporal scale of the realignment project. Given the size of the river realignment project a more extensive and less intensive study design was chosen which meant many sites were studied at relatively frequent intervals but at low intensity. For physical habitat development this meant trying to capture changes at the reach scale. GPS surveys of cross-sectional form and thalweg movement are appropriate at this scale (Bartley & Rutherfurd, 2005). It is evident that spatial scale is equally important for biological development, when the range of hierarchical levels that exists within ecosystem ecology is considered (Rosenberg & Resh, 1993). Research needed to be focused at elements of the ecosystem that would also respond at the reach scale. Here freshwater macroinvertebrate populations and species assemblages and community measures were appropriate as they encompass spatial patterns over the 10s to 1000s of metres and temporal changes from months to years (Cooper & Barmuta, 1993). Macroinvertebrate responses are widely studied at this level for summarising the magnitude and consequences of specific ecological changes or events (Rosenberg & Resh, 1993). Being sedentary, patterns in vegetation richness and abundance are expected to develop at a similar scale.

The propensity for studies to focus on fish, despite the value of macroinvertebrates to both the evaluation process and the ecological integrity of streams has been highlighted by a number of authors (CAins & Pratt, 1993; Jowett, 2003) Aspects intrinsic to the biology of benthic macroinvertebrate communities; their ubiquity, high richness, their sedentary nature and long life cycle – make them ideally suited to investigating spatial and temporal changes of disturbances to stream habitats (Rosenberg & Resh, 1996)

Before After Control Impact Studies

A carefully implemented BACI design is the most powerful of all study designs (Roni et al. 2005) Because of the observational nature of much of this research and the pit falls associated with it (section 6.1.1) a BACI design was considered appropriate for testing the hypotheses detailed in the previous chapter, and for fulfilling evaluation obligations. Specifically a Multiple BACI (Before-After-Control-Intervention) design was chosen. The components of this are as follows:

Multiple – More than one site was surveyed both on the diversion and reference reaches. Replication of control and impact sites is essential for statistical inference (Conquest, 2000).
Before/After – Sites were monitored both before and after the diversion of the river.
Control/Intervention – Monitoring sites were located on both control reaches (natural ‘reference’ reaches up- and down- stream) and intervention reaches (i.e. the diversion).

It should be noted that the multiple BACI design differs from a replicated BACI design, which would require replication of the diversion on a number of rivers. A replicated approach would be required if we were interested in the *effect* of the diversion. Although the multiple BACI would not be ‘statistically correct’ in this respect (Hurlbert, 1984), it does allow meaningful investigation of the patterns and processes of development when considered as many individual colonisation events.

6.1.4 Experimental Control and Reference Sites

In a scientific approach to investigating ecosystem architecture type projects, replication and reference sites are “minimum requirements” for the following reasons.

i) To control for variability in the measured response.
ii) They are fundamental in statistical inference and the interpretation of responses.
iii) Provide information for ecological inference on habitat condition and the potential species pool.

This means that sites selected must be suitable to serve as control for variability, provide reference for condition and provide ecological information.

Control

A true control reach would have all the features that would have existed had the river not been diverted. In the absence of any natural channel identical to the pre impact condition, reference reaches provide the closest possible alternative. Results from surveys of the realigned reaches were compared to hypothetical reference communities based on data collected at up- and down- stream reference sites. This enabled the ecosystem development following realignment to be distinguished from variability attributable to natural processes, random variation and sampling-error which could have masked significant patterns. Survey sites in reference reaches were also ensured that any environmental trend through time was not falsely interpreted as a colonisation pattern. Monitoring then allowed variation introduced by external factors such as climate and hydrology common to both control and treatment reaches to be factored in to any subsequent analysis. In this respect it was important that the reference sites chosen closely tracked the realigned reaches (e.g. responded in the same way to climatic variation)

Reference

There is much debate around what constitutes a reference site (Palmer et al., 1997). Here these are not pristine sites against which restoration aims of the diversion should aspire to (Chapter 2), rather they are sections of the river unimpacted by the realignment. Maddock (1999) takes the analogy of human health, often alluded to in discussions on habitat quality. The average persons health may be perceived as poor if compared to that of an Olympic athlete. In this way using control sites on a pristine Scottish upland river would be inappropriate.

Information

Locating control sites within the potential species pool area means that conclusions about colonisation can be interpreted in the light of the ecology of species at the control sites. The orientation of control sites is also important in this respect since different pathways will be available to species pools located in different directions.

Location of sites

Riverbed sites

Invertebrate communities were surveyed at multiple sites located on both reference (upstream) and response (realigned channel) reaches. Four reference sites were positioned upstream of the realignment, two above the Nith section and two above the Beoch. A further two reference sites were also positioned downstream of the realigned river. Reference reaches downstream provide a suitable comparison for the lower reaches. They also better represent the pre-realignment condition as active agricultural land. A total of 10 response sites were surveyed along the length of the realignment.

Cross-section sites

25 channel cross-sections were surveyed on the realigned sections of the Nith and a further 7 were surveyed on the Beoch. These were located every 100m along the length of the realigned channel. Both cross-sectional form and vegetation development were surveyed at these locations. Additionally in the summer of 2007 habitat features were also surveyed within the vegetation quadrats: Data for moisture, slope, silt depth and coir fibre condition were collected.

The layout and location of monitoring sites is detailed in figure 6.1.

6.1.5 Surveys of the physical habitat, river planform and slope

As discussed the physical environment in streams is a complex combination of geomorphological and hydrological factors (Chapter 3). When investigating rivers the considerable variability in hydrology can make comparisons difficult. Hydraulic data must be collected and compared under many different conditions before a fair assessment can be made. The use of spatial measures of variability over measures such as hydraulics that require repetition through time means comparisons can be made between streams or stream reaches, despite differences in water levels (Bartley & Rutherfurd, 2005). For the same reason it allows changes in reach morphology to be tracked through time without undue influence from the different flow conditions associated with seasonal variability.

Bartley & Rutherfurd (2005) propose a variety of measures that quantify the physical variation in a stream reach independently of flow (see section 6.3) encompassing different scales of geomorphic variability. To collect data for these indicies the following aspects of the realignment were surveyed.

Differential GPS long profile (Thalweg)

Longitudinal profile as represented by the thalweg – The deepest path of water and path of fastest flow within a reach. The changes in gradient along this path represent reach scale geomorphic diversity. These topographic undulations produce hydraulic variability such as pool and riffle flows (Bartley & Rutherfurd, 2005)

The use of Global Positioning System satellite signals and a fixed-point base station allows, differential GPS to pin point locations to an accuracy of 3cm. This allowed us to survey both gradual and rapid changes in bed level along the length of the diversion and pick out movement of the thalweg and development of riffle and pool habitats. Heights were recorded approx. every 10 m along the thalweg and closer together at regions of changing gradient. Two Leica SR3000 Differential GPS units were used; one as a fixed-point base station, the other as the roving device. As well as providing information on the bed habitat, the thalweg survey also allowed us to document changes in the form and position of the meanders.

Cross-section profiles

The stream morphology at the scale of the cross-section is considered to be sensitive to large-scale disturbances (Bartley & Rutherfurd, 2005). This provides a record of lateral variability complementing longitudinal data from the thalweg survey. The cross-sectional scale can represent a rapid spatial gradient in habitat conditions from very slow, zero or even reverse flow of marginal habitat to the fast flow of the thalweg.

32 cross section sites were located every 100m along the diverted channel to track any changes in channel shape. Whilst this is not intensive enough to record every area of erosion and deposition, it does give a detailed enough picture to show channel shape and capacity and follow general patterns at the reach scale and identify areas of the diversion that are responding differently. The locations of cross-sections are marked with x-sec and are numbered from bottom to top. They were surveyed using the Leica SR3000 dGPS with heights recorded at every change in slope across the width of the river channel between fixed points at the top of the river banks. This produced three dimensional data for approximately 15 points from which cross-section profiles were created.

Pebble counts

Bed substrate – Three well established measures (Bartley & Rutherfurd, 2005) exist that are useful for describing and quantifying bed sediments, namely Skewness, sorting and Kurtosis (Briggs, 1977). Further measures include substrate heterogeneity

Pebble counts were conducted at each of the riverbed sites in 2005 and 2007 using a Wolman pebble plate to size approximately 100 pebbles. Stones were selected from under the toe at each step along a zig-zag path within the region of each of the riverbed sampling sites (figure 6.1). At each site both riffle and glide habitats were surveyed.

6.1.6 Riverbed Sampling

Benthic macroinvertebrate surveys

This study followed standardised procedures for sampling benthic macroinvertebrates (Wright et al., 1984) to an extent. However whilst the main objectives of Wright et al., (1984) was to obtain the most comprehensive species list possible, the testing of our hypothesis required a quantitative element to the data. This necessitated a more structured approach.

Macroinvertebrates communities were sampled using a 500um mesh kicknet with a 30cmx30cm opening. Sediment and the invertebrates are disturbed by agitating a 0.5m2 area of the sediment with the net positioned downstream and its base tightly against the riverbed. A discrete 20 second kick is repeated a total of nine times to include the range of habitat present, including pool, riffle and glide habitat in equal measure. This forms a combined 3 minute kick sample, which is then transferred to a plastic bag, preserved with methylated spirits and sealed. This method was chosen to allow comparison of invertebrate data from the 2004 realignment with data collected from a previous rerouting of the same reach in 2000. During the autumn surveys pool riffle and glide habitats were sampled separately as three one-minute samples (each collected as three discrete 20 second kicks) to provide information about development in specific habitats before being combined to form the standard three-minute samples for comparison with other seasons.

6.1.7 Riverbank Sampling

Vegetation

The colonisation of riverbanks and subsequent development of a vegetation community was tracked at a 36 river bank sites positioned at 100m intervals along the length of the realigned reaches. Plant community composition, richness and cover over a 1m^2 area were surveyed using a quadrat placed at the top, middle and toe of the bank. Vascular plants were identified to species in most cases. Some mosses were left at genus or family level. An estimate of cover was made by eye for each species present. By positioning the riverbank sites at fixed 100m intervals we were able to investigate the effects of channel form, meander position and disturbance on colonisation and development without the factors having influenced site choice, avoiding sampling bias

The riverbank survey sites were all positioned on the north bank and, to varying degrees, generally faced south. As such cover and diversity values may be marginally higher than might have been shown in a survey of both banks. This aspect was not considered to be of interest to the study, given that the effect would be the same on any channel design. Excluding it from the study enabled the inclusion of more sites focusing with greater intensity on the effects of distance, in addition to eliminating an additional source of variability.

Physio-chemical habitat

In August 2007, a record was made of the physical habitat within each of the vegetation quadrats. The quadrat was divided into 16 25x25cm squares. Three measurements of moisture were made in each square using a Theta probe pushed in to the surface substrate. This gave us 16 averages for each quadrat. The depth of silt accumulation, the condition of the coir fibre and the angle of slope of the ground were also recorded for each of the 16 squares.

6.1.8 The Nith Experiments

Mossy boulders

The mossy boulder experiment was set up to investigate the importance of habitat at early stages of ecosystem development and the potential of moss to enhance the recovery of the stream bed ecosystem. A number of studies have pointed to the value of moss to stream health (section 3.4.5 and 3.5.2) but its role in recovery processes is still unknown. To test hypotheses related to this question the following experiment was set up.

Bryophyte patches were created in the upper reaches of the realigned section of the Nith. Mossy boulders were collected from a section of relic river bed cut off by the realignment project. Boulders were vigorously scrubbed to remove all resident invertebrates before being transported across to the test reach in the engineered channel. Boulders with only loosely attached moss or of a size smaller than 128mm (approx.) were discarded to improve the longevity of the patches.

The experimental layout consisted of alternating patches of mossy and non-mossy boulders. Three patches of approximately $0.25m^2$ were created 1m apart across the width of the channel. Eight such rows were created separated by 3m forming an alternating pattern of 24 patches within an area of riverbed 3.5m by 25m.

Leaf-release

The retention capacity of the channel is important to the health of upland streams. Coarse Particulate Organic Matter (CPOM) is a major source of carbon to upland stream systems (Shields et al., 2008). Hydraulic diversity and zones of slack water may be a factor in the retention of CPOM and drifting invertebrates (Section 3.4.1). However, quantifying levels of retention or hydraulic diversity can be difficult. An experiment using the travel time of leaves through a reach is proposed as a method of directly surveying both retention and hydraulic diversity.

Medium sized (8-12cm) fresh alder (*Alnus glutinosa*) leaves were picked and divided into bags of 200. The bright green fresh leaves had the advantage of being easily identifiable over the existing leaflitter in the stream and were visible to a depth of several feet. The matched closely the behaviour of natural debris (more so than plastic leaves) travelling at the surface, in the water column and along the bed of the channel and provided no pollution risk to the stream.

A batch of 200 leaves were sprinkled evenly across the channel at an upstream point. The time taken for each leaf to travel through a 100m reach was then recorded. After the last floating leaf passed through, the location and distance of any trapped leaves were recorded.

Leaf packs

Fallen willow leaves collected from the vicinity of trees in the upstream control sites were washed clean of any mud and fragments, and spun dried in a salad spinner for 1 minute. Pyramid shaped mesh bags were then filled with 30grams of leaves. A total of 16 coarse mesh (5mm) and 16 fine mesh (0.5mm) bags were filled. Four bags of each mesh size were then placed at each location. Bags of alternate mesh size were tied to a line stretched between to steaks so that the lay perpendicular to the flow in a marginal area of deep slow flowing water. Locations of the bags were NC1, N2, N7 and NC3 (See figure 6.1)

6.1.9 Sampling dates

Riverbed sites were first sampled for macroinvertebrates in the autumn of 2004 two weeks after the connection of the realigned channel into the river course. Sampling was then repeated in the spring, summer and autumn seasons of 2005 and 2006 with a final survey completed in Spring 2007. Pebble counts took place in autumn of 2004 and the summers of 2005, 2006 and 2007. Bank vegetation surveys were carried out in July of each year from 2005 to 2008. Gravel bar vegetation was surveyed in August 2007. Moisture levels and silt accumulation along the banks were recorded in late August 2007. GPS data was collected for 2004 in October and for 2005 in April and May. In 2006 and 2007 GPS data was collected in July.

The diversion was also visited on a number of other occasions to undertake additional scientific studies including experimental work on the role of mossy boulders as invertebrate habitat and leaf release experiments as indicators of organic matter retention capacity.

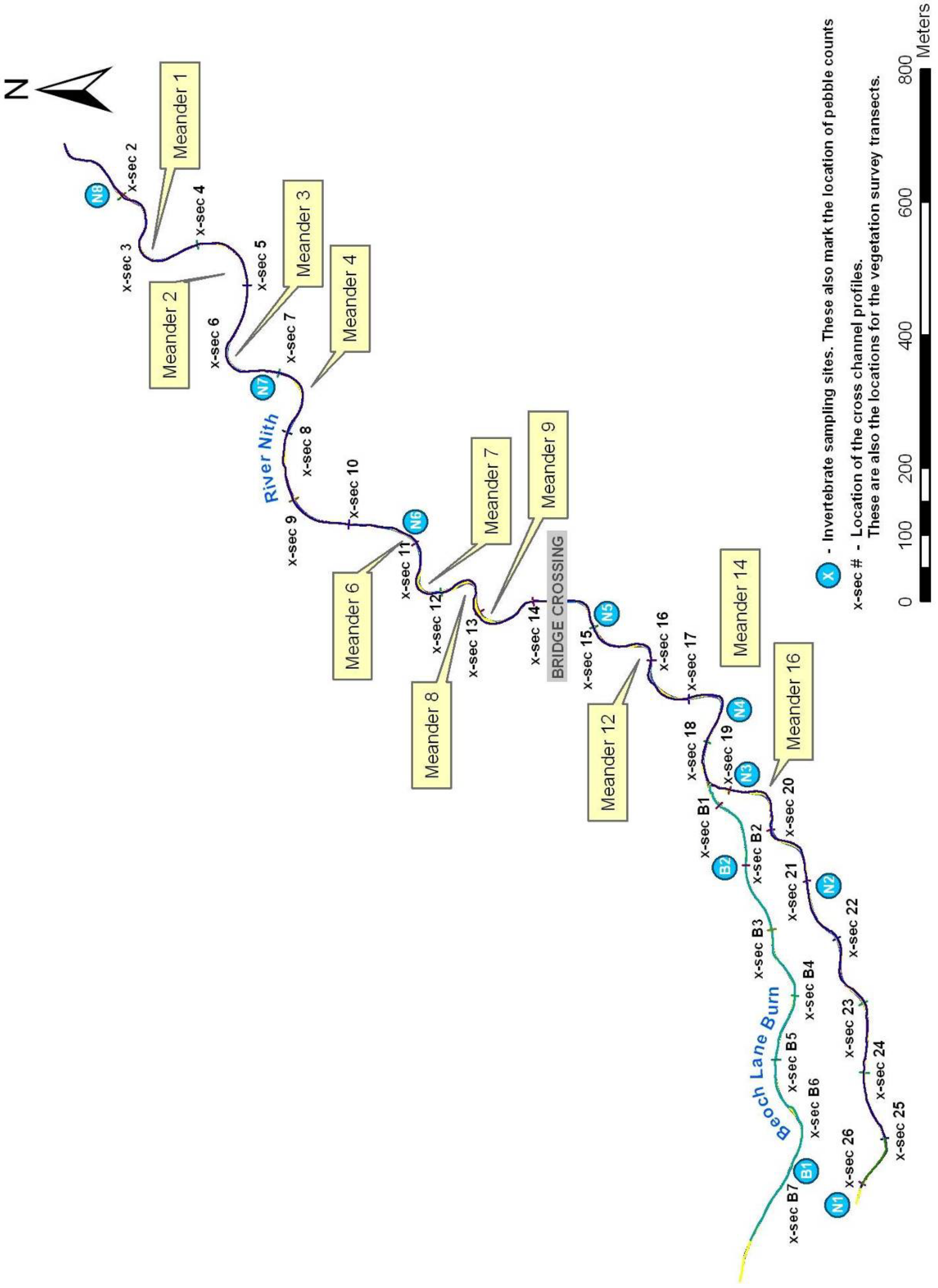


Figure 6.1. Map of the River Nith 2004 Diversion showing the approximate location of invertebrate monitoring sites, positioning of channel cross sections, meander numbers and locations of instream experiments. Six-figure OSGB grid references for all sites and cross-sections are given in Appendix I

6.2 Laboratory analysis

6.2.1 Sorting of macroinvertebrates

The macroinvertebrate samples were returned to the University and stored at 5ºC until ready to be sorted. They were then graded into size classes through a set of sieves for easier sorting. A small amount of the sample would then be transferred to a white tray with a little water. All the invertebrates were removed to Petri dishes and identified under a microscope. The samples were sorted exhaustively and in their entirety so as to minimise any margin of error to the already variable sample diversities. Sub-sampling was ruled out because it causes some biotic indices to behave erratically. Cao and Hawkins found that changes in population evenness following disturbance can compensate for true taxa loss in 200-300 count samples of macroinvertebrates. This would adversely affect the sensitivity to change of a number of indices.

6.2.2 Identification

Identification of the invertebrates was based mainly on the Freshwater Biological Association and additional Environment Agency keys and was performed to species level. We considered the investment of time in such a high level of identification appropriate because it provides so much more information. There is considerable diversity within the family groups on the Nith and Beoch, especially for the EPT taxa, which could potentially reveal patterns not visible at the family level. As discussed in Chapter 5, individual species can have characteristic behavioural traits and life histories.

6.3 Data analysis

The types of challenges faced by field research are not unique to ecology (Michener, 1997). As a result a range of statistical tests have been developed to deal avoid the problems associated with in sufficient replication, a lack of true experimental control and other constraints.

Despite much debate over the best specific metrics to use to evaluate river projects (Palmer et al., 1997), diversity is widely believed to be useful ecosystem property which at the very least provides an appropriate surrogate for ecosystem health. Two challenges exist for the study of diversity; reliable measurement and effective summation (Rosenzweig, 1999). To reliably measure the complexity of the physical environment a huge volume of measurements must be collected. The shear volume of data means that before any patterns or trends can be detected appropriate methods of summarising the data must be chosen. Likewise counts of species can amount to an overwhelming amount of data. Because many facets to diversity exist the use of a number of different indices may be appropriate. Furthermore, the assumption that biological indicators give unbiased estimates of true biological condition remains largely untested (Cao & Hawkins, 2005), making this all the more important. Variability in certain parameters of the physical habitat, such as moisture or silt depth, can be taken as a form of diversity. Where this is the case the standard deviation of the variable can be used as a diversity index. When the variable is not normally distributed, non-parametric equivalents such as the upper quartile or inter quartile range may be appropriate alternatives.

6.3.1 <u>Software</u>

The software packages Excel 2003, MiniTAB 15 and Graphpad PRISM were used to analyse the invertebrate and vegetation data. Further ordination and resampling analysis of data was done using CANOCO 4.5 for Windows. PAST was used for the calculation of Sørenson’s, Bray-Curtis, rarefaction and diversity indices including Shannon’s H and *distinctness*. Analysis of geomorphological data (including differential GPS data) made use of Excel 2003, ArcGIS 9.1 and ArcView 3.2.

6.3.2 <u>Indices of physical habitat</u>

Many studies have focused on single variables to quantify physical habitat diversity in streams and rivers. However, more ‘well rounded’ approach to describe physical structure or quantify levels of geomorphic diversity can be achieved by investigation diversity at a range of scales using a number of different indices (Bartley & Rutherfurd, 2005). Although the relation of these indices to levels of biological diversity has yet to be assessed some methods have been shown to be sensitive the degradation of physical habitat diversity (Bartley & Rutherfurd, 2005). Approaches include quantification of habitat area, complexity and variability. Ideal properties of a suitable index include the summation of one or more of these aspects and a sensitivity to environmental changes. Avoiding uncommon or abstruse indices will facilitate the comparison of results with past studies.

6.3.3 Bed substrate data

Pebble count data was analysed by plotting cumulative frequency curves with a logarithmic x axis. This allowed spatial and temporal comparison of the spread of gravel sizes present and the coarseness of the material at each of the invertebrate sampling sites.

6.3.4 Thalweg data

It was possible to visualise directly the changes in bed level along the thalweg by converting xy co-ordinates to path distance along the thalweg and plotting against the elevations. Fixed point photography was used to complement the resulting graphs. Data for each individual 200m reach was also analysed using the following indices of physical habitat.

6.3.5 Channel cross-section data

Profiles of the cross-sections were reconstructed by converting xy co-ordinates into distances across the channel which were then plotted against the elevations allowing visual inspection of the cros-sectional change. Data for each cross-section was also analysed using the following indices of physical habitat.

Bankfull width to bankfull wetted perimeter ratio

This is a simple indicator of the complexity of channel form. Bankfull wetted perimeter is in effect the distance of the path as measured down the banks and across the bed. Greater complexity leads to a higher cross channel diversity score.

Hydraulic radius

Hydraulic radius (R) = Cross-sectional area (A) / Wetted perimeter

Absolute percentage change in area (Olson-Rutz & Marlow, 1992).

$|\Delta A\%|$ = erosion + deposition and quantifies all movement of stream bank or streambed materials.
It avoids the risk of erosion in one area of the bed being balanced out by deposistion in another giving a false impression of stream bed and stream bank stability (Olson-Rutz & Marlow, 1992).

Vector dispersion (Bartley & Rutherfurd, 2005)

Vector dispersion was calculated as the deviations of angles around each data point. This takes the width of the cross-section into account

Channel Asymmetry

The development of an asymmetrical shape to channel cross-sections was calculated using the ratio between the left and right halves of the channel. Ratios of wetted perimeter, volume and hydraulic radius were calculated.

6.3.6 Hydraulic diversity and retention

Data from experiments using leaf release to investigate hydraulic diversity were analysed using cumulative frequency curves. Four measures were taken from the data.

L_1 Time for the first leaf to travel 100m (assumed to represent the most direct route)
L_{50} Time for 50% of floating leaves to travel 100m (The median value is assumed to give an indication of the length of the average path of flow)
L_{90} - L_{50} The time period range for the slower leaves to travel the 100m (Taken as an indication of hydraulic diversity)
p(R) The probability of a leaf being retained within the 100m reach. This was the proportion of recovered leaves that were retained by the stream.

6.3.7 Ecological Indices

A considerable range of parameters have been presented as suitable indicators of ecological character, such as diversity, for evaluating projects (Ruiz-Jaen & Aide; 2005). Commonly used indices used include taxa richness, abundance, community composition and turnover, Shannon-diversity and dissimilarity. They are widely used within ecological research, giving insights into community structure, ecological health and integrity, ecosystem processes and successional development. It has been suggested (e.g. Rosenberg & Resh, 1993) that the multiplicity of indices available for the analysis of ecological data may indicate that workers are not satisfied with the results that they provide. Each has its own strengths and weaknesses and careful consideration of their limitations is required before any individual index is chosen (Norris & Georges, 1993).

Ecological indices can be categorised into two distinct groups - diversity indices that attempt to summarise composition and structure; and biotic indices that summarise functional or biological characteristics (see figure 6.2). Many indices can be applied to at any taxonomic level. Identification to species level can provide greater insight than more generic family level analysis. However, this does require accurate identification of the individuals throughout all data sets analysed. Individuals were grouped taxonomically to both family level and species level allowing a comparison of the usefulness of each level to be made. Individuals were also grouped into functional groups according to feeding guilds.

There is considerable discussion over the interpretability of observed differences in biotic index scores when comparing two or more sites (Stewart-oaten et al., 1986) for example sites taken before and after an impact. Because replicate samples are rarely taken, some authors have called for more reliance to be placed on knowledge of ecosystem processes (Conquest, 2000; McDonald, 2007). Since the study of diversity is appropriate to answering many questions in ecology, many indices have been developed that attempt to summarise information to a level that can be used to compare impact and reference habitats (Rosenberg & Resh, 1993). Whether or not the same level is appropriate to investigation ecosystem development or recovery has received less attention.

Diversity indices	Taxa Richness (Alpha diversity)	Count data for specific groups	Species, genera or families in a community Specific gourps of taxa such as EPT richness (Ephemeroptera, Plecoptera, Trichoptera)
	Enumeration (Gamma diversity)	The abundance of specific groups	Total abundance - Counts of all organisims Dominant abundance - counts of the most common taxa
	Diversity' (Delta diversity)	Combinations of richness and abundance into a summar statistic	Shannon's diversity Simpson's eveness
	Similarity (Beta diversity)	Relative numbers of share species or groups	Qualtiative - Presence/Absence of shared taxa eg Sorensen's Quanitative - Relative abundances of shared taxa eg Bray-curtis, Morisita
Ecological indices	Tolerance/Preference	Taxa grouped and scored according to survival under certain conditions	BMWP score Ellenberg scores (vegetation)
	Functional	Taxa grouped acording to life history traits	Functional feed groups of macroinvertebrates Oviposition behaviour of adult macroinvertebrates Established strategy in plants

Figure 6.2. Types of Ecological indices with the tyoe of data on which each is based. Examples for each type are also shown

6.3.8 Invertebrate data

A number of different indices were applied to the species count data. Values were then plotted against downstream distance for each sampling season. Richness. Fisher's alpha, Shannon's H and Simpson's 1-D were applied to both species and family level count data.

Taxonomic diversity

Richness

Richness is simply calculated as the total of the taxa either at the family or species level.

Fisher's alpha $S = -\alpha \ln(1-x)$

In a community described by the log series model the number of species in a sample can be described by the above formula where α is a constant dependent on the sample diversity and x is a variable that depends on the sample size. The variable x satisfies $S/N = [(x-1)/x]\ln(1-x)$ where N is the number of individuals in the sample. As such, Fishers α is an artefact of the log series distribution. Small incomplete samples of other distributions approximate well to the log series distribution (Rosenzweig, 1999). It is particularly useful as it allows investigation of the diversity present at each site without influence of sample size. For this reason Fishers α is considered relatively robust.

Shannon's Index of Diversity $H = -\sum P_i \ln P_i$

Where P_i is the proportion of the ith species in the sample and is equal to the number of individuals in a species divided by the total number of individuals in the sample. The Shannon index of diversion is very popular despite being biased and very insensitive to rare species

Beta diversity

β-Diversity is the measure of dissimilarity between habitats or samples in relation to the variety of species recorded. The considerable range of beta diversity indices available may have resulted from the variability in results that they produce, a lack of understanding of their basic properties or dissatisfaction with their performance (Koleff et al., 2003). In a study investigating the properties of 24 published indices based on presence-absence data, Koleff et al. (2003) were able to relate the performance of the indices to underlying way in which they compare the data. However additional indices that include comparisons of taxa abundance between samples are also popular (Cao & Hawkins, 2005). Much of this research is concerned with the species populations at each of the sampling locations and comparing sampling locations based on the number and abundance of species present. The indices listed below each achieve this in slightly different ways and as a result compare different aspects of the sample similarity. By using a range of indices should provide greater detail for interpreting the patterns of colonisation.

Beta diversity measures were used to compare the composition of samples from the realignment with the sampled communities at upstream and downstream reference sites. In addition samples from the realignment were compared with estimates of the regional species pool. This was assumed to match fairly closely the entire species list for all samples collected from the reference reaches over the 3 year sampling period.

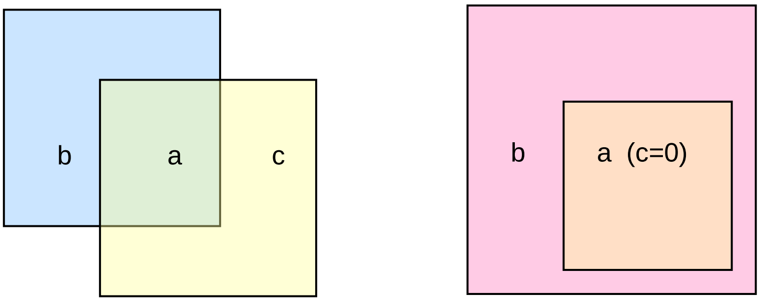


Figure xx showing the possible relationship in terms of shared species between to separate species pools (i), and between local and regional species pools (ii). a = shared species, b = species unique to the first species pool, c = species unique to the second species pool

Ruggiero et al. (1998) **a / (a+b)**

This is perhaps best understood as the percentage of taxa found in the first sample that are shared with the second sample (described by Koleff et al. (2003) as a measure of continuity and loss).

Jaccard (1912) **a / (a+b+c)**

Jaccard's index could be described as the proportion of the total taxa recorded that are shared between the two samples. High values indicate greater similarity between the samples; lower beta diversity.

Whittaker (1960) **(a+b+c) / 0.5(2a+b+c)**

This index behaves in a similar way (but reversed) to Jaccard's, scaling negatively with increases in the proportion of shared taxa. As such it can be considered a direct measure of beta diversity. It is included as the most widely used measure of beta diversity to date (Koleff et al., 2003)

Routledge (1977) $\mathbf{(a+b+c)^2 / ((a+b+c)^2 - 2bc)}$

Values of Routledge's index reflect the degree of overlap of species distributions, focusing more on compositional differences than differences in species richness (Koleff., 2003)

Distinctness

Taxonomic distinctness and diversity

A disadvantage of species-abundance distributions is that they contain no information about taxonomic composition (Cao &Hawkins, 2005). An alternative approach developed by Clark & Warwick (1998) is based on the how taxonomically distinct the assemblage is; where a community with many families each with a few number of species would be considered more distinct than an alternative community with the same number of species shared between few families. A diversity measure based on this same approach was also developed.

Functional distinctness and diversity

This approach looking at taxonomic groupings of families has been extended here to look at ecological groupings, specifically functional feeding groups.

6.3.9 Vegetation data

A large number of quadrats were surveyed, positioned within a variety of environmental conditions. Gradient analysis was considered an appropriate approach to summarise and investigate patterns within the data. Vegetation data was visualised using indirect gradient analysis and analysed using direct gradient analysis. Remoteness, chainage and bank position were included as explanatory variables to investigate their role in vegetation development. Physiochemical conditions within each quadrat, recorded in 2007, were included in analysis for that years vegetation data.

Quadrats were also scored based on the plant attributes according to Grime *et al*., (2007) and Ellenberg, revised for UK populations (CEH, 1999). The average of the scores achieved by each plant in the quadrat is taken as the quadrat score. The technique of weighting scores according to plant cover was not used as, in many cases, it made little difference to the observed patterns and in some instances was found to give undue weight to the most abundant group of species. Quadrat scores were calculated for the following plant attributes.

Dispersule weight

Calculated is the average class (1-6) achieved by plants in the quadrat. Classes as reported by Grime *et al.*, (2007) and based on the dried weight of the seed, achene or other indehiscent germinule; 1, <0.21mg; 2, 0.21-0.50mg; 3, 0.51-1.00mg; 4, 1.01-2.00mg; 5, 2.001-10.00mg; 6, >10.00mg

Seed persistence

Calculated is the average class (1-4) achieved by plants in the quadrat. Classes as reported by Grime *et al.,* (2007) and described as follows; 1, transient seed bank present during the summer and germinating synchronously during the autumn; 2, transient seed bank present during the winter and germinating synchronously in late winter or spring; 3. A small amount of the seed persists in the soil often for greater than 5 years but after intial high numbers following seed shed, concentrations are low; 4 there is a large bank of persistent seeds in the soil throughout the year.

Seed fragment ratio

This is calculated as the ratio of plants found in the quadrat with a dispersule and germinule in the from of a fruit against the those whose dispersal and germinule is in the form a seed. The majority of plants fell into one of these two categories.

Ellenberg F indictor value

Calculated as the quadrat average for the Ellenberg indicator score indicating plant preference for soil moisture levels (1-12)

Ellenberg N indicator value

Calculated as the quadrat average for the Ellenberg indicator score indicating plant prefrence for Nitrogen rich soil (1-12)

Species status.

Calculated as the proportion of species found in the quadrat that are known to be nationally in decline across the UK as indicated by Grime et al., (2007)

6.3.10 Data quality

In a study by Clark et al. (2002) results suggested that 12% of the overall variation was attributable to inter operator effects and emphasise the importance of proper training of field technicians. To minimise this element of variability it was considered important to use the same person to take all kicknet samples.

Standard deviation and the coefficient of variation (CV=SD/$\bar{x}$x) are commonly used to measure the precision of estimating indvidual assemblage attributes and biotic indices (Cao et al., 2003) In fact since accuracy is often impossible to measure precision is generally the only measure of data quality used[2] despite often being inappropriate for assessing assemblage data. Cao et al. (2003) describe three reasons related to the fact that a range of indices are used to characterise different aspects of community composition and hence vary in different ways and to different degrees. The precision of any single index is far from representative of the entire assemblage.

A number of further complications also exist:

i) High quality sites have been shown to have higher sampling variability in the number of taxa (Clark et al., 2002; Cao et al., 2003)

ii) Rare species are unlikely to be detected without very extensive sampling. Common species can be so widespread that they obscure diversity differences from place to place (Rosenzweig, 1999).

iii) Whereas taxonomic groupings requires a detailed understanding species ecology to interpret patterns, classification of groups according to ecological traits such as functional role within the ecosystem can provide insights in to the mechanisms that underlie patterns (Lancaster, 2000)

---

[2] Accuracy is a measure of how close the measured value is to the true value (i.e. how representative a sample composition is of the community in the channel). Precision is a measure of how close repeated measurements are to each other (i.e. the similarity of one sample to another).

CHAPTER 7

RESULTS I: GEOMORPHOLOGY

This chapter presents the results from surveys of physical habitat development including data from fixed point photography, differential GPS and pebble counts. Geomorphic adjustments are described using a number of different indices and appropriate results are selected for comparison with indices of biological development. The rate and pattern of geomorphic adjustment to the engineered physical habitat will be important to the development of a natural form and are expected to underpin patterns of ecological development. A key aspect of this study was to link geomorphic processes to to the development of physical habitat form and heterogeneity.

Aims

Investigate development of the riverbed substrate including relationships with planform and erosion
Track changes in channel morphology in relation to bedslope, hydrology and planform of reaches
Investigate the development of physical habitat along the path of the thalweg using diversity indices
Investigate the development of physical habitat across the channel using diversity indices
Describe the retention capacity of the realignment using leaf release data

7.1 Bed sediments

7.1.1 Substrate development at invertebrate monitoring sites

Figure 7.1 compares the structure of the sediments making up the surface layer of riffles with the surface layer of pool/glide habitat in 2005 and 2007 using cumulative histogram plots of the pebble count data. The initial gravel substrate placed in the bed during the construction phase (2004) is shown to mark the stating point of the realignment.

Site N1 is located in a section of relict channel complete with the original bed sediments. This site was not surveyed in 2004 and the 2004 gravel mix (dotted line) is simply shown for comparison purposes. The coarseness of sediment structure is comparable with that recorded at the construction stage of the realignement but the sediment is less well sorted. Glide habitat in 2007 shows an increase in the predominance of smaller sediment sizes.

Site N2 is within the engineered reach. In 2005 the surface substrate in both glide and riffle areas is noticeably finer than the construction mix and than the natural and relic reaches upstream (Figure 7.1 N2). Some areas of channel adjustment are evident upstream of N2 during this period (Plates 7.10j-l and 7.11a-f). In 2007 the bed can be seen to have coarsened again and is comparable in structure to the relic bed at site N1 (Figure 7.1 N3). Also in 2007 the riffle habitat is coarser than the pool/glide habitat.

The 2005 survey shows little change in bed substrate from the construction mix at site N3. Pebbles placed across the channel at two cross-sections in the vicinity of this site (cs18&19) remained unmoved throughout 2005, eventually being displaced or buried in 2006. There was also little channel adjustment in this reach during the first winter (Plates 7.9g). In 2007 the riffle remains relatively course and a slight fining of the substrate is visible in the glide. Localised erosion occurred upstream of the N3 glide site between 2005 and 2007 (Plates 7.9h-i).

The pebble survey of glide habitat at site N4 in 2005 indicated the deposition of finer sediments, although, the substrate is shown to coarsen again between 2005 and 2007 (Figure 7.1 N4) This site is downstream of some significant gravel movement associate with the confluence of the Nith and Beoch Lane (Plate 7.9a-c) and changes in bed level (Figure 7.12)

To summarise, substrate in the upper reaches (N1 to N4) are broadly comparable with a very similar degree of sorting. Gravels are more poorly sorted over the scale of the survey – approx. 10m of channel. The appearance of a greater proportion of finer material has contributed in this respect. The d90 is courser in the relic channel than the constructed (approx 128mm compared to between 60-90mm) reflecting the absence of course material in the construction mix and the bank substrate.

Sediment surveys at sites N5 and N6 show substantial fining of the surface substrate between 2004 and 2005 in both riffle and glide habitat. Both N5 and N6 are located in a region of extensive channel adjustment (Plates 7.7g-I and 7.5g-l respectively) with a high input of sediment from bank erosion. Site N5 shows some resilience to this with the substrate having coarsened again by time of the 2007 survey. Although the substrate at N6 is coarser in 2007 than in 2005 it remains notably finer than the constructed mix as well as the substrate in natural reference reaches (Figure 7.1 N6). Pebbles placed in this reach were rapidly displaced or buried after during flood season 2005/06 with only one 90mm pebble recovered from the channel edge of cross-section 10 in 2006

Site N8 shows deposition of finer sediment in the glide habitat between 2004 and 2005. Although the proportion of coarser fractions increases again by 2007, finer material is still a significant proportion of the surface area. Riffle habitat at the downstream end of the realignment remains coarse and relatively well sorted. Pebbles placed across the channel in the downstream end of the diversion (cs4) remained un-mobilised until the winter of 2006/07.

To summarise, the middle geomorphically active reaches show marked fining of the sediments but exhibit resilience with sediments re-coarsening after the initial adjustment of the channel. Less activity is seen at the bottom end of the diversion with only some deposition of finer sediment in slow moving glide habitat at the downstream end.

Fining of the sediment at site B1 on the Beoch Lane Burn is consistent with the patterns observed on the Nith and is in a region with significant local erosion of the river banks. Although data is not available for 2005 it is seems likely from the similarity between the substrate in 2007 and 2004 that the lower reach (B2) has seen very little change. This in reinforced by the fact that pebbles placed immediately upstream, across cross-section b3, remained in place for the length of the study period.

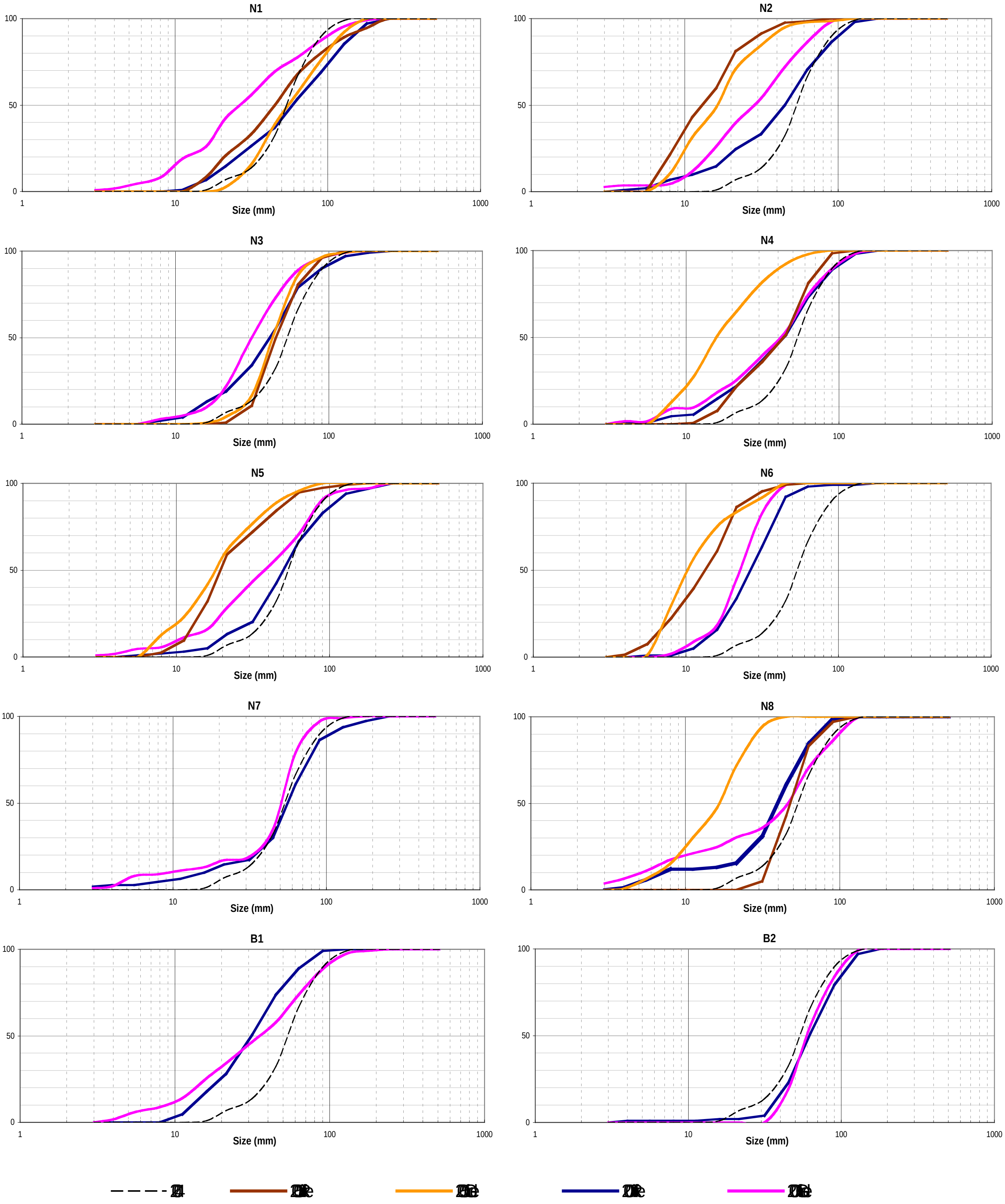


Figure 7.1. Cumulative pebble size count data for sites N1 to N8 on the Nith realignment showing the change in substrate structure from the initial gravel mix (2004) to a more diverse substrate

7.1.2 Substrate change around meanders

The role of the sinuous plan form in the development of the streambed is shown in figures xx and xx. The graphs show a clear grading of the substrate from the inside to the outside of the meander on all the meanders surveyed. Statistical analysis of the d_{50} using ANOVA confirms this. ($P<0.001$) The inside of the meanders consistently found to have a finer substrate that the outside edge. The bed substrate round the outside of meander 4 matches the gravel mix used for the construction of the bed and finer substrate along the inside edge could simply result from deposition in this area. Meanders 1 3 8 9 12 14 show the outside edge getting coarser than the construction mix indicating active resorting of gravels related to the hydraulics of the meander form. The same pattern of resorting across the channel width is not observed on straight sections of channel and, although the substrate is finer than the original gravel mix, there is generally very little change in substrate structure associated with lateral position across the channel (figure 7.2)

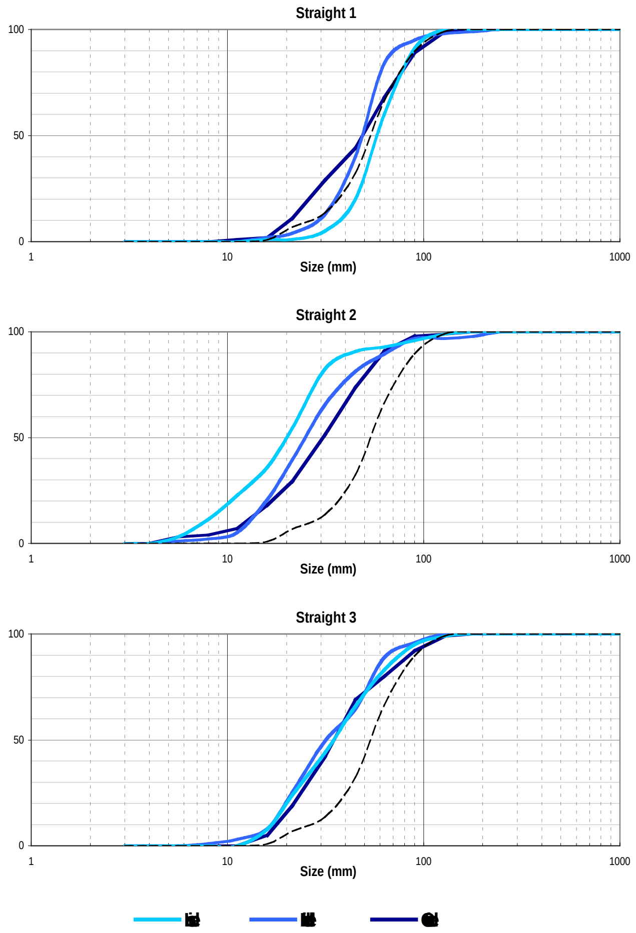


Figure 7.2. Cumulative pebble size count data for straight reaches surveyed on the Nith realignment in 2007 comparing the sediment structure along the centre and edge of the channel

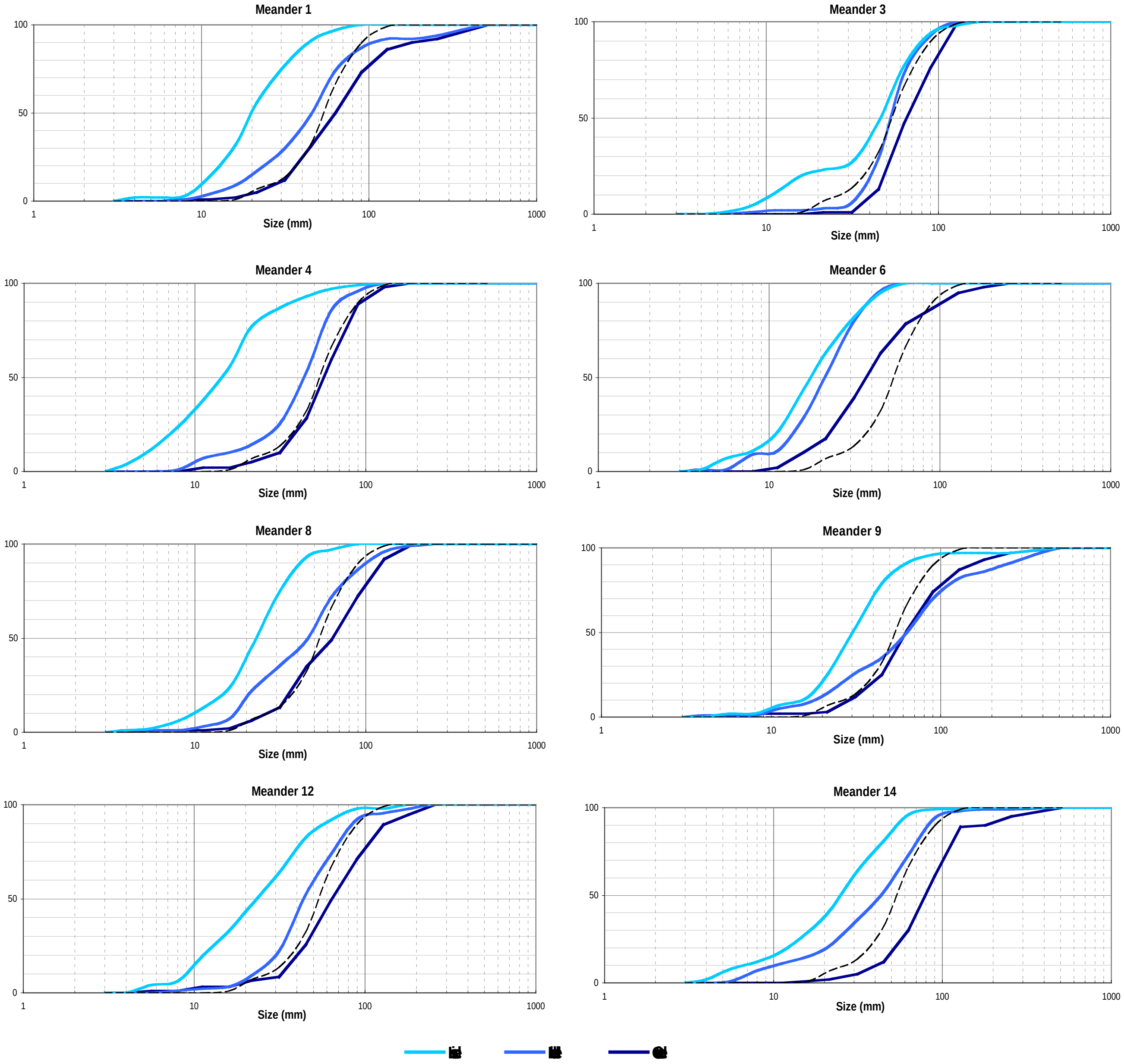


Figure 7.3. Cumulative pebble size count data for meanders surveyed on the Nith realignment in 2007 comparing the sediment structure along the inside middle and out side edge of the channel

7.2 Development of channel form

7.2.1 Development of the longitudinal profile.

It is clear from the elevation profiles for the channel thalweg that considerable geomorphic activity has occurred varying in both spatial and temporal aspect. The greatest activity is observed in the middle reaches of the Nith realignment and is associated with changes in sediment transport. Here slope and discharge combine to produce the greatest stream power eroding down the stream bed through the reorganisation and export of gravels. Cross-sections at the top end of this section show bed levels to have dropped across the full width of the channel with much of the change occurring between 2004 and 2005. Significant lowering of the thalweg profile occurred during this time from the confluence with the Beoch Lane (Chainage 720m, figure xx) for approximately 500m to the bridge crossing. Further lowering of the bed is generally associated with pool development and the out side of meander bends and has been gradual (e.g. chainage 870m, chainage 1025m, chainage 1215m). Below this stretch, a second region of extensive geomorphic activity is characterised by the deposition of gravels generally in areas of very low bed gradient (chainage 1200m-1300m and 1400m-1500m) promoted in places by tight meander bends (from chainage1200m to 1400m). This is clearly visible in photos (plates 7.5 and 7.6) and cross section profiles 10, 11 and 12 (figure 7.25). In the lower reaches of the realignment (Chainage 1600m-2400m) little change in the thalweg profile was recorded (figures 7.4-7.6) and the channel appears to be relatively uniform (Plates 7.1-7.4)

Chainage 2400m to 2200m

The thalweg profile shows little change in the relatively flat bed topography and the two engineered pools at 100m and 170m located on meander bends are maintained. A step in bed level at ch40m on a straight channel section has filled in to form a more even profile. Bed and bank erosion at the tail end of meander 2 from ch5m to ch20m is evident in the thalweg profile and in plate 7.1l. In 2005 the reach scores unexpectedly high on diversity indices considering it's relative uniformity (Vector dispersion 1.9, Wiggliness 0.8). Ranked 3/15 on a scale of least to most ative.

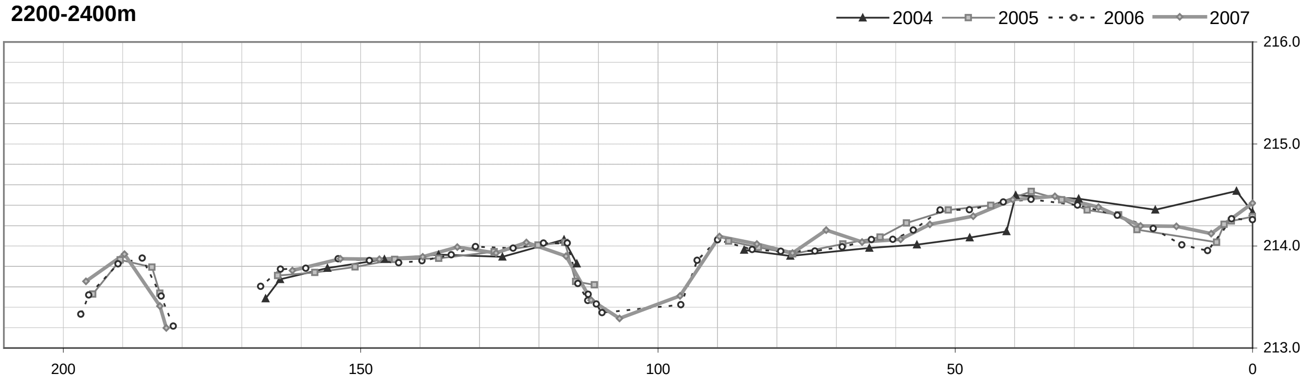


Figure 7.4 Thalweg profile for chainage 2200m to 2400

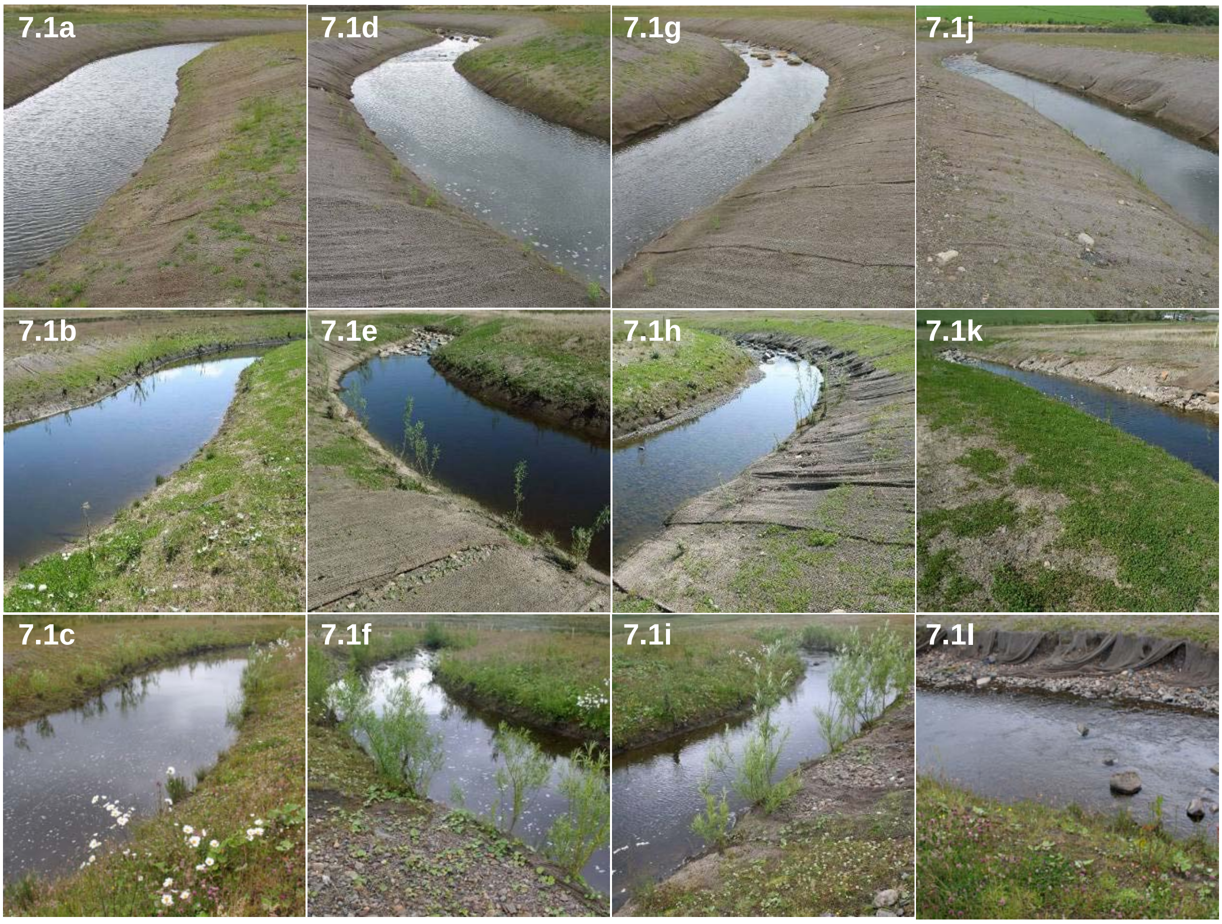


Plate 7.1. Fixed point photography from the left bank showing river habitat development between cross sections 2 and 4 (Chainage 2400m to 2200m) for 2005, 2006 & 2007. (a)-(c) upstream from cs2 (d)-(f) downstream from cs3 (e)-(i) upstream from cs3 (j)-(l) downstream from cs4.

Chainage 2200m to 2000m

The thalweg profile (figure 7.5) shows no significant changes in bed elevation. The positioning of the lower pool (185m) on a very gentle meander has been sufficient to keep it free from infilling. Falls in scores for diversity indices based on the thalweg profile data between 2005 and 2007 (Vector dispersion 0.11 to 0.3, Wiggliness 0.39 to 0.21) suggest a change in physical habitat not evident from close inspection of the data (figure xx) or from fixed point photography. Plates 7.2d -7.2l indicate little change in the bed and no significant bank erosion. Ranked 2nd least active.

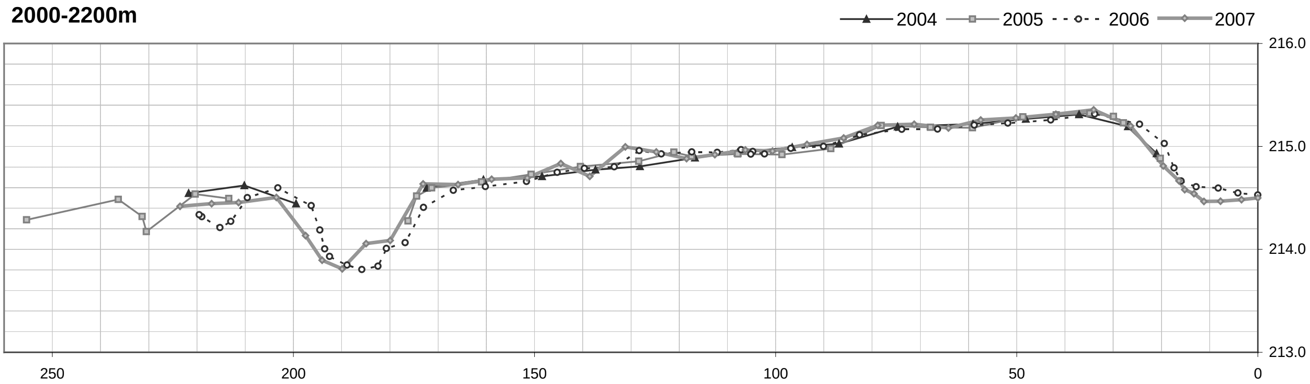


Figure 7.5 Thalweg profile for chainage 2000m to 2200m

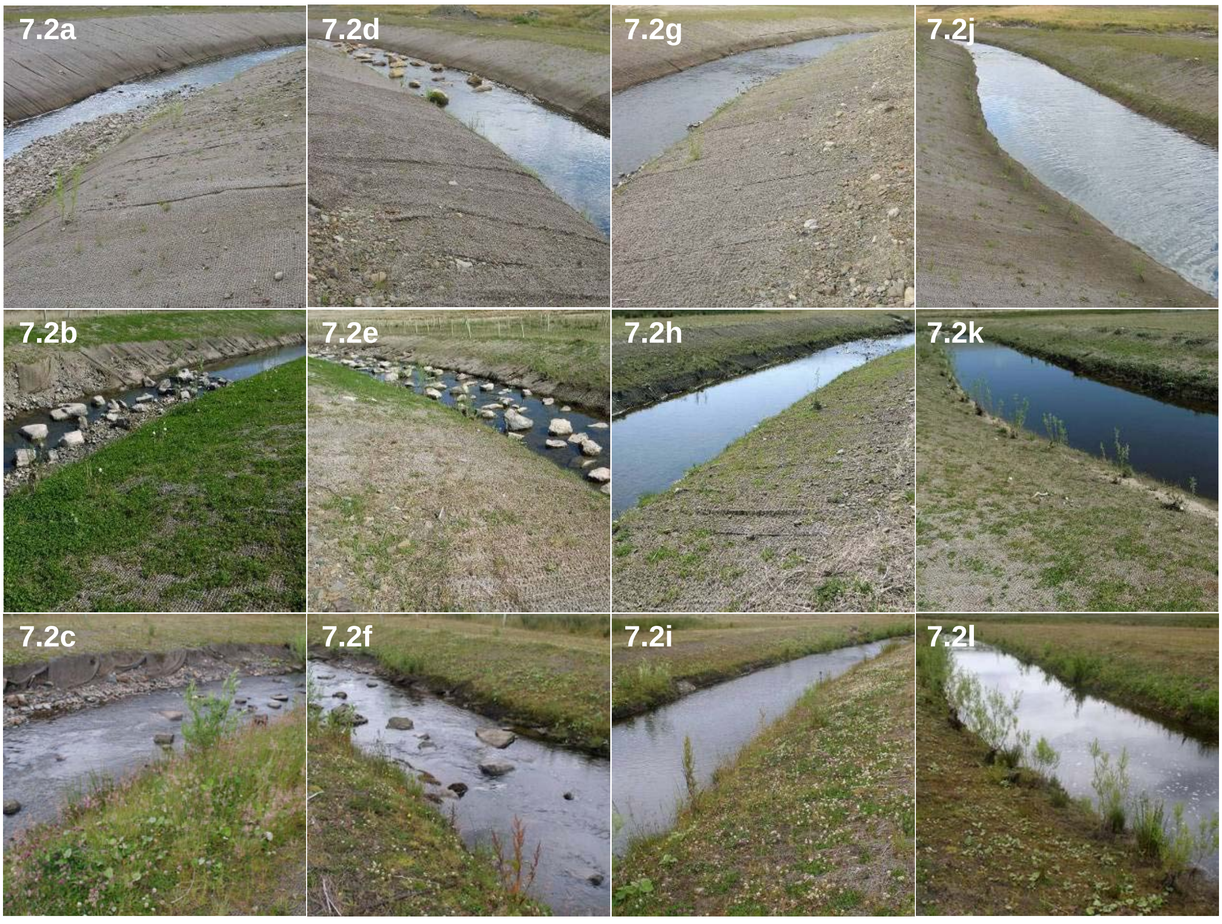


Plate 7.2. Fixed point photography from the left bank showing river habitat development between cross sections 4 and 6 (Chainage 2200m to 2000m) for 2005, 2006 & 2007. (a)-(c) upstream from cs4 (d)-(f) downstream from cs5 (e)-(i) upstream from cs5 (j)-(l) downstream from cs6.

Chainage 2000m to 1800m

Little geomorphic change is evident between chainage 1800m to 2000m other the deposition of a small side bar at approximately ch30m (Plates 7.3k&l) and the suggestion of some deposition in the pool at ch70m. The slight drop in bed level detected in 2006 (ch150m to ch175m) may be a result of scour around boulders and is not maintained (figure 7.6). The photography shows bank morphology through the reach remains uniform despite three years of flood flows.

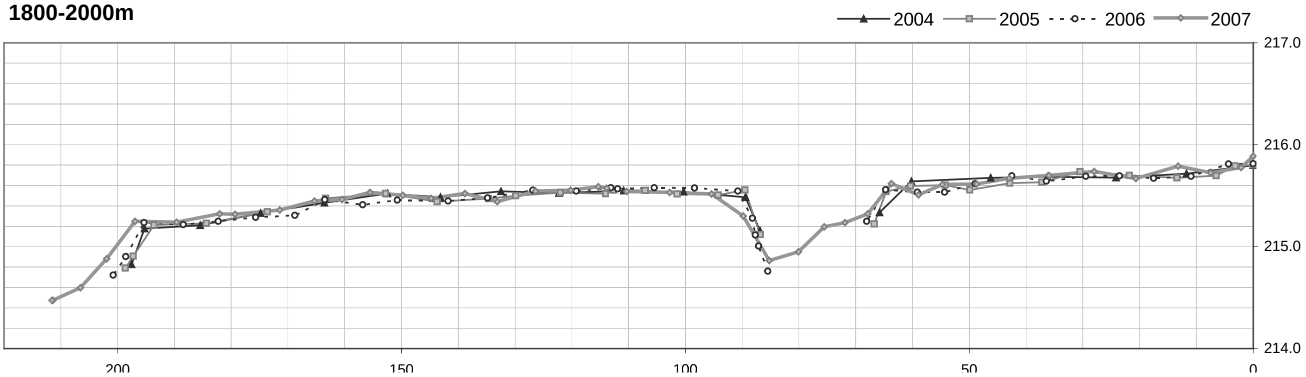


Figure 7.6. Thalweg profile showing changes in elevation between 2004 and 2007

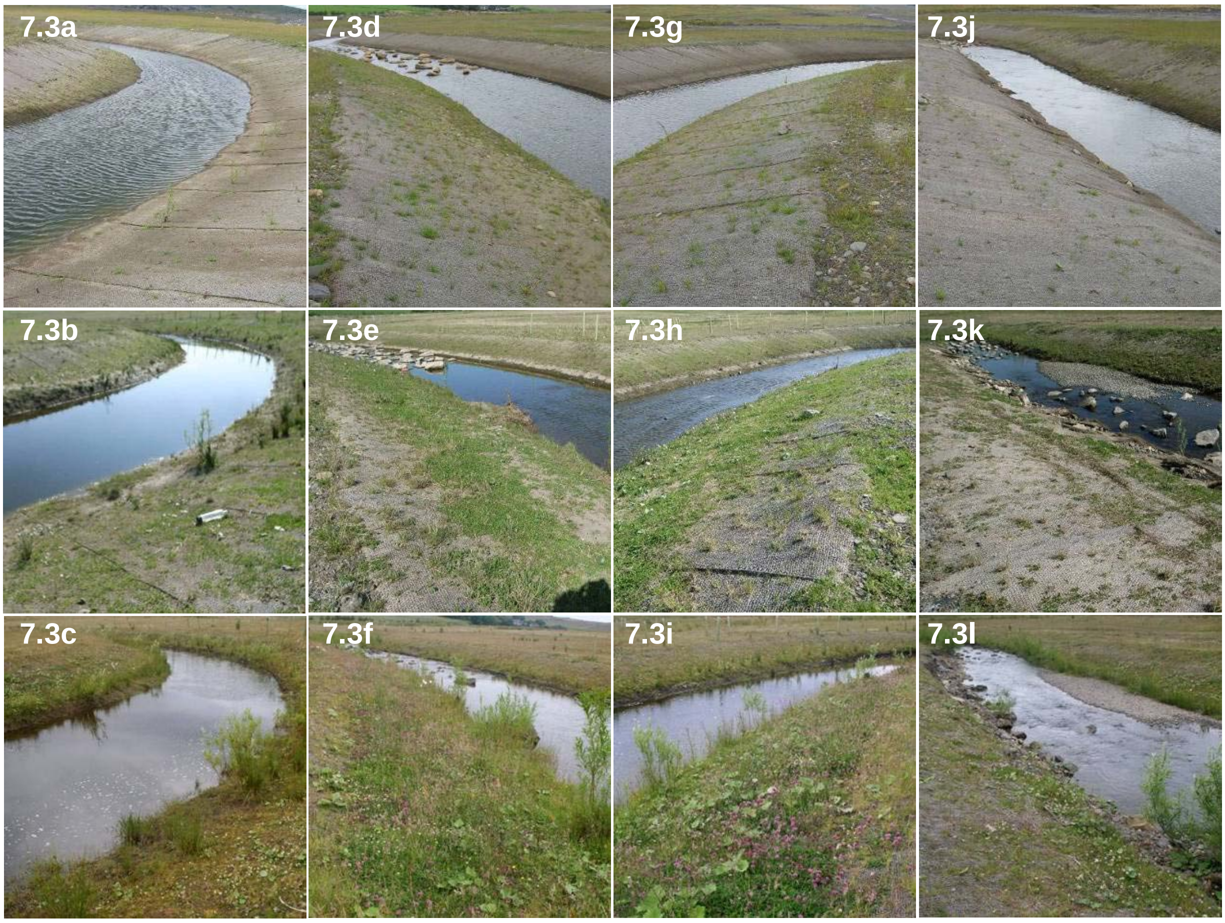


Plate 7.3. Fixed point photography from the left bank showing river habitat development between cross sections 6 and 8 (Chainage 2000m to 1800m) for 2005, 2006 & 2007. (a)-(c) upstream from cs6 (d)-(f) downstream from cs7 (e)-(i) upstream from cs7 (j)-(l) downstream from cs8.

Chainage 1800m to 1600m

Figure xx shows changes in profile elevation, most notably in the upper section (ch0m to ch100m) and deposition in a deep section from ch140m to ch170m. However the bed remains relatively flat. This is reflected in the physical diversity index scores, which are consistently low for this reach (VD<0.03, 'w' approx. 0.2). Inflow from settlement lagoons at ch100m has little evident effect although a covering of very fine silt was observed on the bed for several hundred metres downstream of this point. An extensive gravel bar formed during the first few floods has persisted unchanged (Plates 7.4g-i).

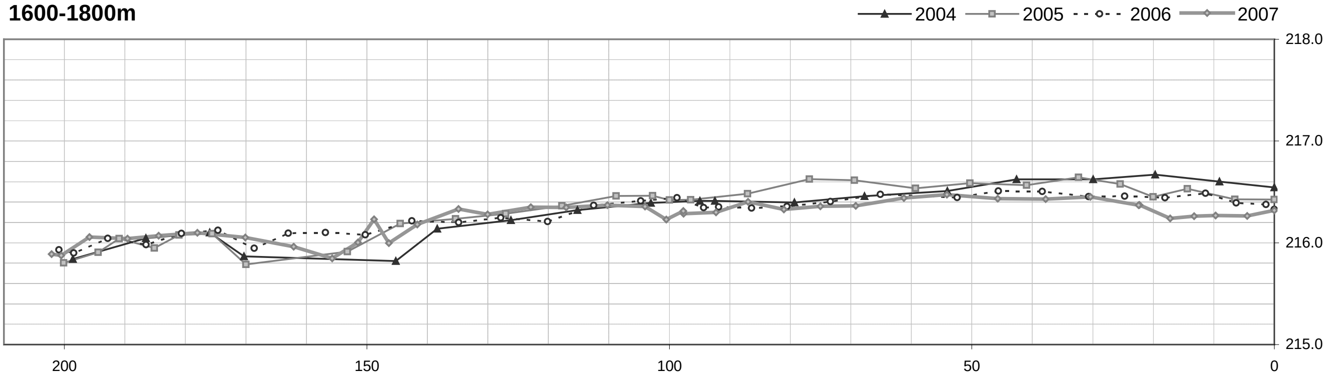


Figure 7.7. Thalweg profile showing elevations from chainage 1600m to 1800m

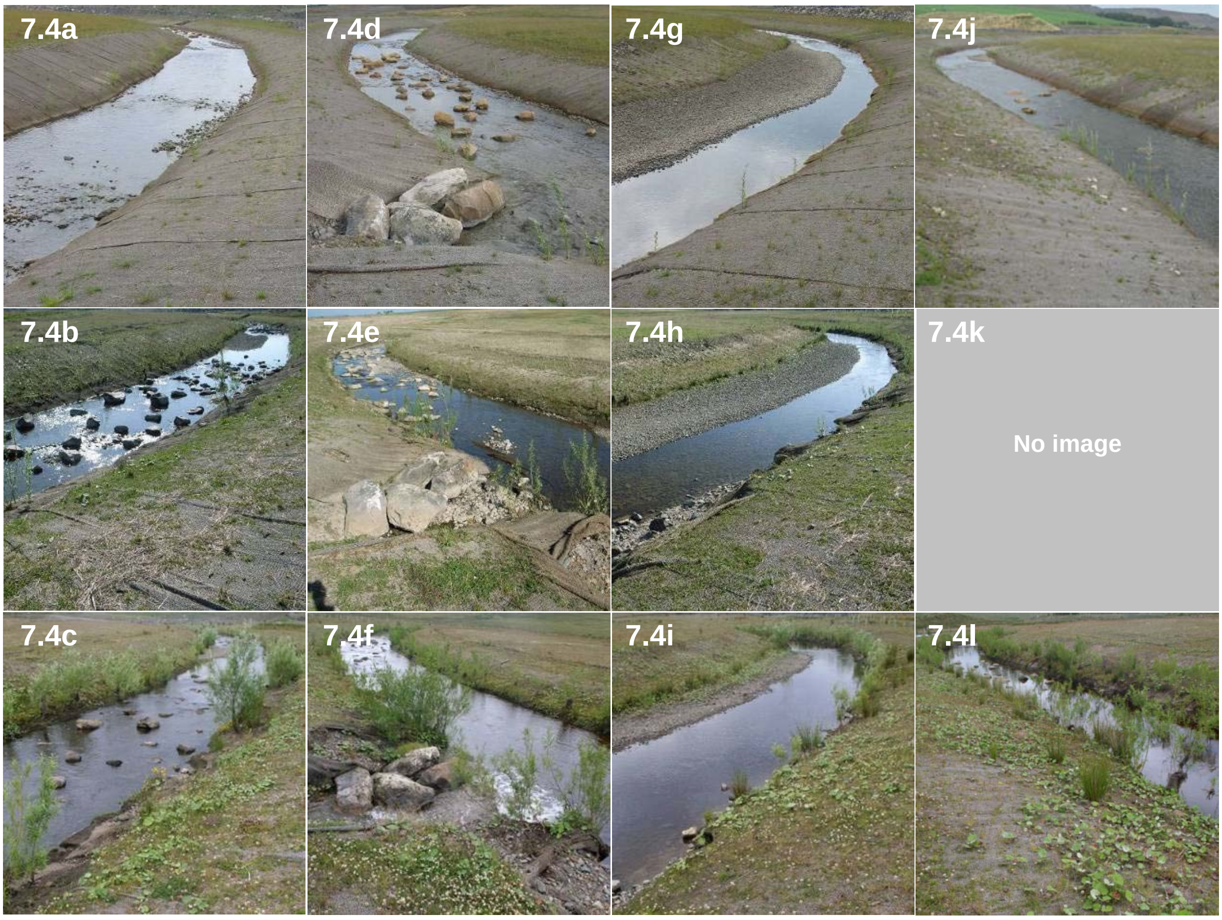


Plate 7.4. Fixed point photography from the left bank showing river habitat development between cross sections 8 and 10 (Chainage 1800m to 1600m) for 2005, 2006 & 2007. (a)-(c) upstream from cs8, (d)-(f) downstream from cs9, (e)-(i) upstream from cs9, (j)-(l) downstream from cs10.

Chainage 1600m to 1400m

Both fixed point photography and the thalweg profile indicate significant geomorphological activity within this reach. This reach is through the area of ground settlement, approx. 0.2m between ch0m and ch100m by 2007 (figure 7.8). Plates 7.5(d)-7.5(i) show significant gravel movement and deposition in this area. A fall in thalweg profile of as much as 0.6m suggests erosion patterns are associated with this deposition. This reach is immediately downstream of a region of extensive erosion (Plate 7.6).

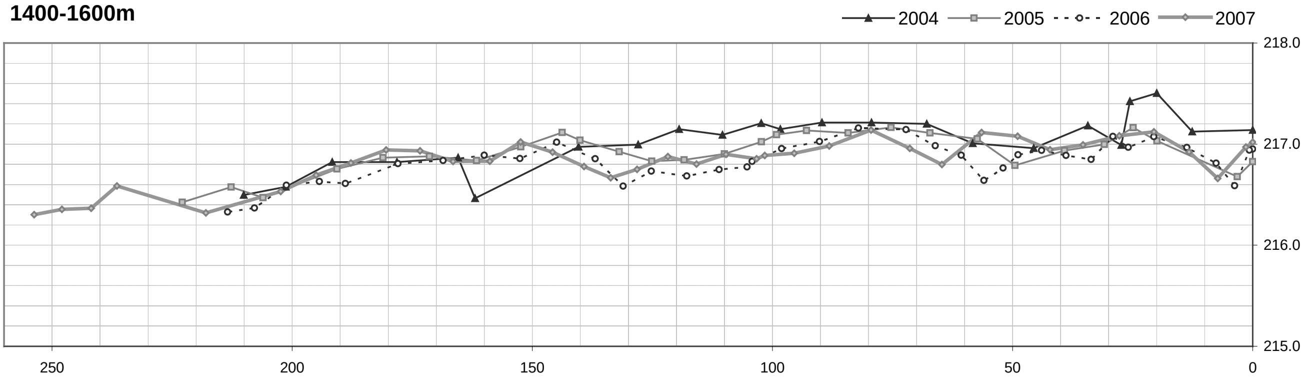


Figure 7.8. Thalweg profile showing elevations from chainage 1400m to 1600m

Plate 7.5. Fixed point photography from the left bank showing river habitat development between cross sections 10 and 12 (Chainage 1600m to 1400m) for 2005, 2006 & 2007. (a)-(c) upstream from cs10, (d)-(f) downstream from cs11, (g)-(i) upstream from cs11, (j)-(l) downstream from cs12.

Chainage 1400m to 1200m

It is evident from figure xx and plate 7.6 that there has been significant geomorphic activity through this reach including extensive erosion of outerbanks, pointbar deposition, and scouring of the thalweg. The resulting physically diverse topography by 2007 is not reflected by the score achieved for vector dispersion (0.07) although it scores a little better for wiggliness (0.4). Constructed pools were rapidly infilled (ch100m and ch140m) and replaced through a more gradual development (ch20m)

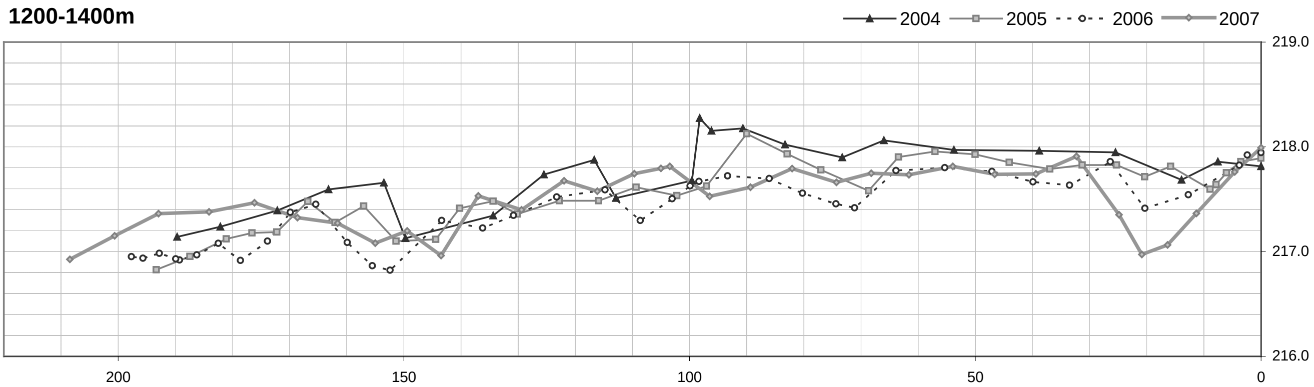


Figure 7.9. Thalweg profile showing elevations from chainage 1200m to 1400m

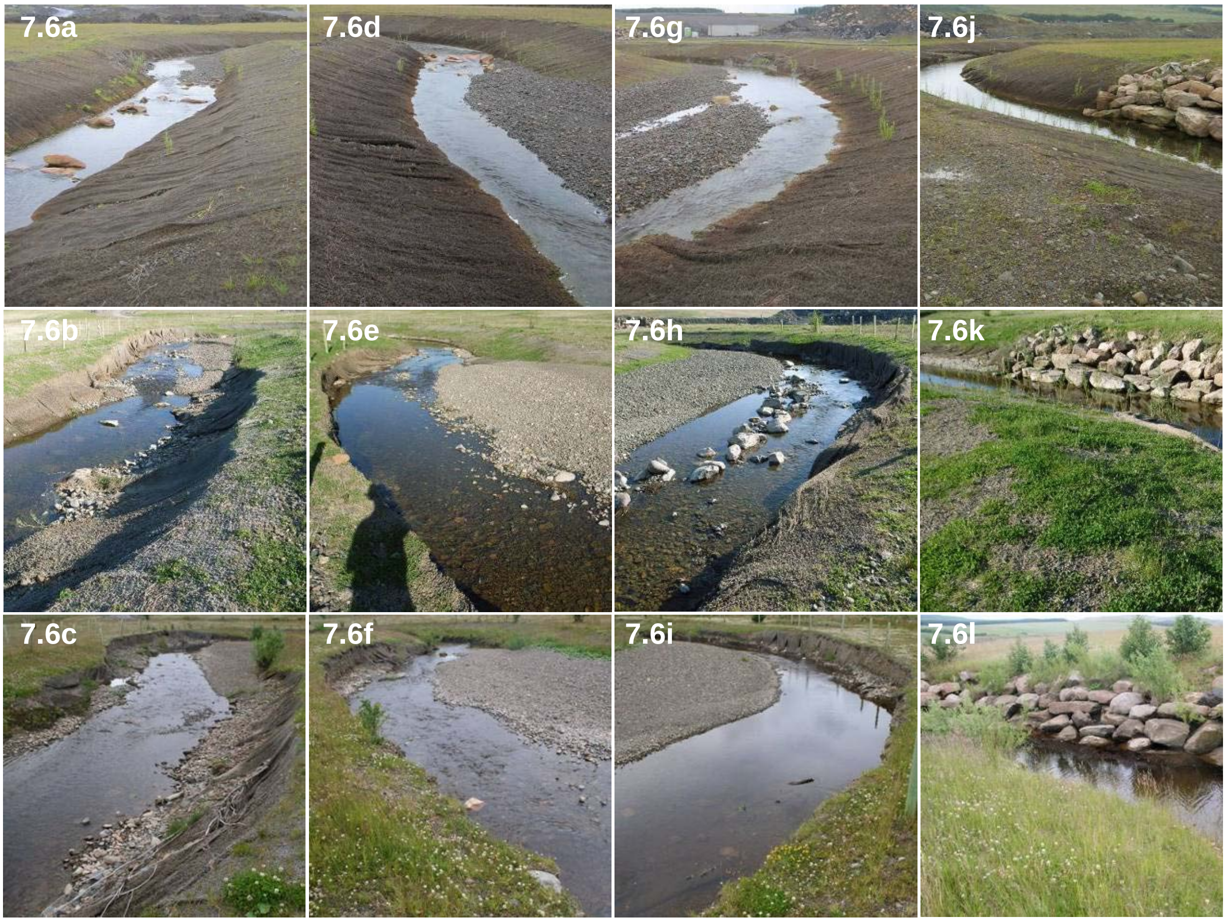


Plate 7.6. Fixed point photography from the left bank showing river habitat development between cross sections 12 and 14 (Chainage 1400m to 1200m) for 2005, 2006 & 2007. (a)-(c) upstream from cs12, (d)-(f) downstream from cs13, (g)-(i) upstream from cs13, (j)-(l) downstream from cs14.

Chainage 1200m to 1000m

This has been a geomorphologically active reach with some gravel bar deposition and a significant amount of gravel exported out of the upper section (figure xx) resulting in a widening and deepening of the channel (e.g. plates 7.7g-i). Scouring of deep pools associated with bank armouring (ch25m and ch150m) may explain why this is the highest scoring reach for both Vector Dispersion (0.13) and Wiggliness (0.59) in 2007.

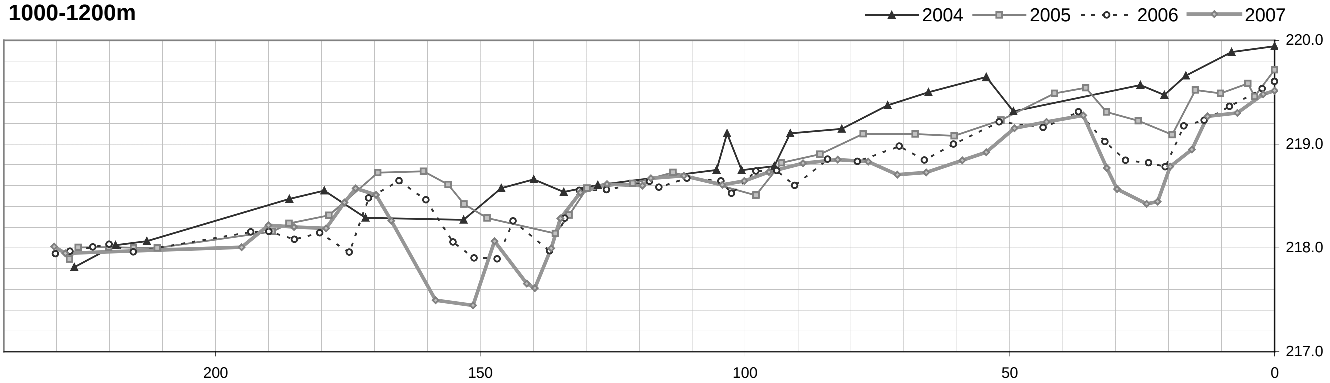


Figure 7.10. Thalweg profile showing elevations from chainage 1000m to 1200m

Plate 7.7. Fixed point photography from the left bank showing river habitat development between cross sections 14 and 16 (Chainage 1200m to 1000m) for 2005, 2006 & 2007. (a)-(c) upstream from cs14, (d)-(f) downstream from cs15, (g)-(i) upstream from cs15, (j)-(l) downstream from cs16.

Chainage 1000m to 800m

Lowering of the bed level has occured along the length of this reach along the line of the thalweg (figure 7.11) and across the width of the channel as indicated by cross section profiles (figure xx). The greatest changes, up to 0.5m, occurred in the first year and are not attributable to ground settlement which can only account for a 15mm change in this region. Pool development is evident at chainages 70m and 200m and infilling occurred at 160m.

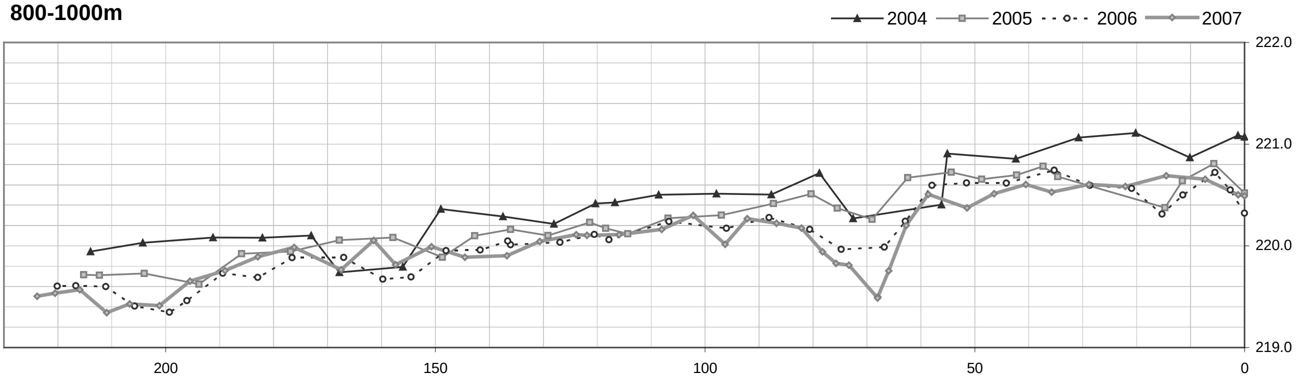


Figure 7.11. Thalweg profile showing elevations from chainage 1000m to 800m

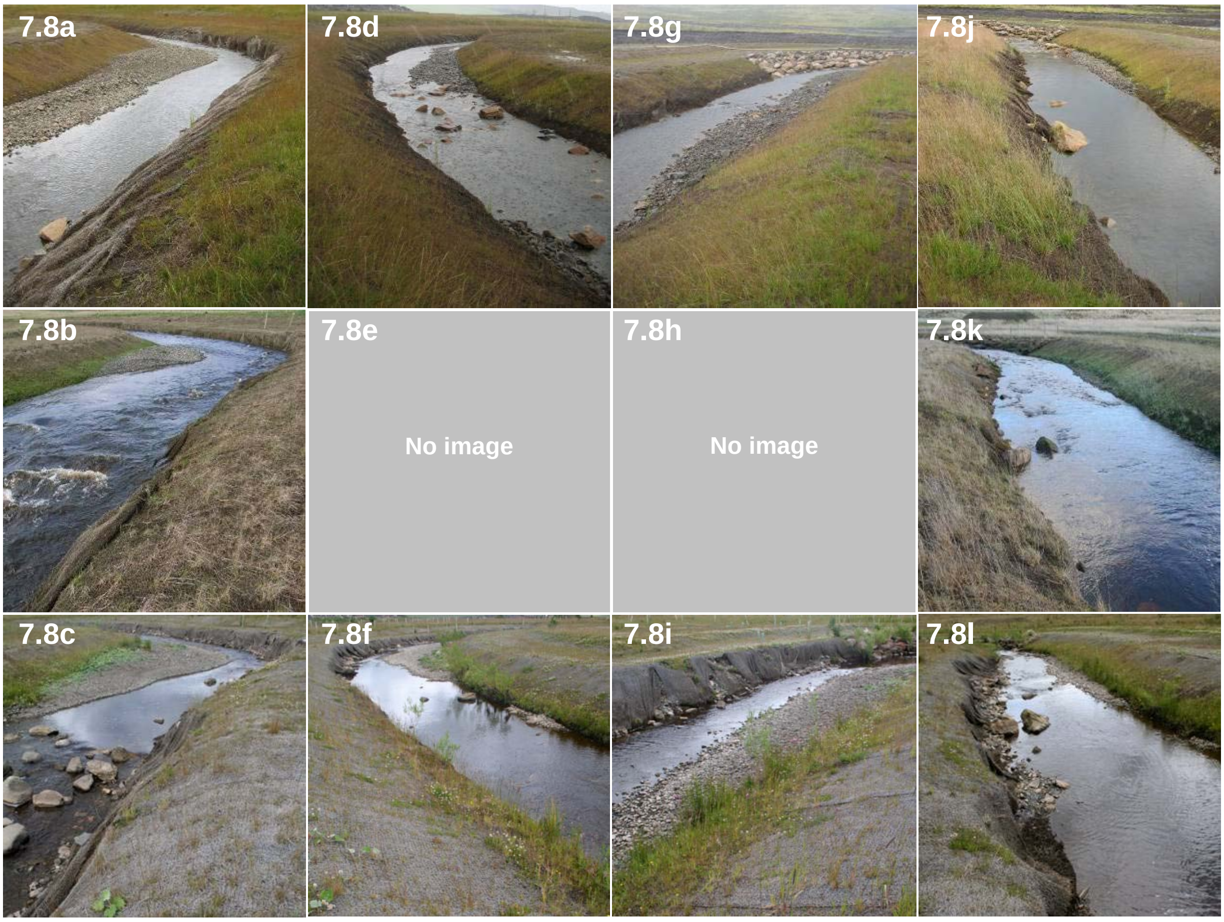


Plate 7.8. Fixed point photography from the left bank showing river habitat development between cross sections 16 and 18 (Chainage 1000m to 800m) for 2005, 2006 & 2007. (a)-(c) upstream from cs16, (d)-(f) downstream from cs17, (g)-(i) upstream from cs17, (j)-(l) downstream from cs18.

Chainage 800m to 600m

The confluence with the Beoch Lane occurs at 130m along this reach and the change in geomorphic activity resulting from the combined discharge of the two rivers is evident in the profile (figure 7.12) with gravel bars clearly visible in the plate 7.9 a-c. The greatest change in bed level and character occurs between 2004 and 2005. Any outer bank erosion features that might have developed is limited by extensive rock armouring. Above the confluence the erosion and bar formation, associated with a meander bend, is the only morphological change. This reach is ranked 12/15 for activity.

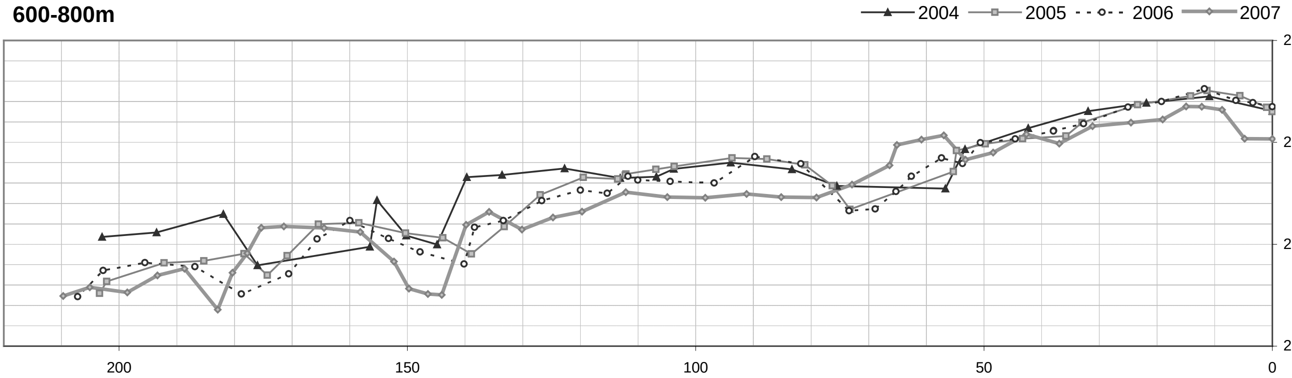


Figure 7.12. Thalweg profile showing elevations from chainage 600m to 800m

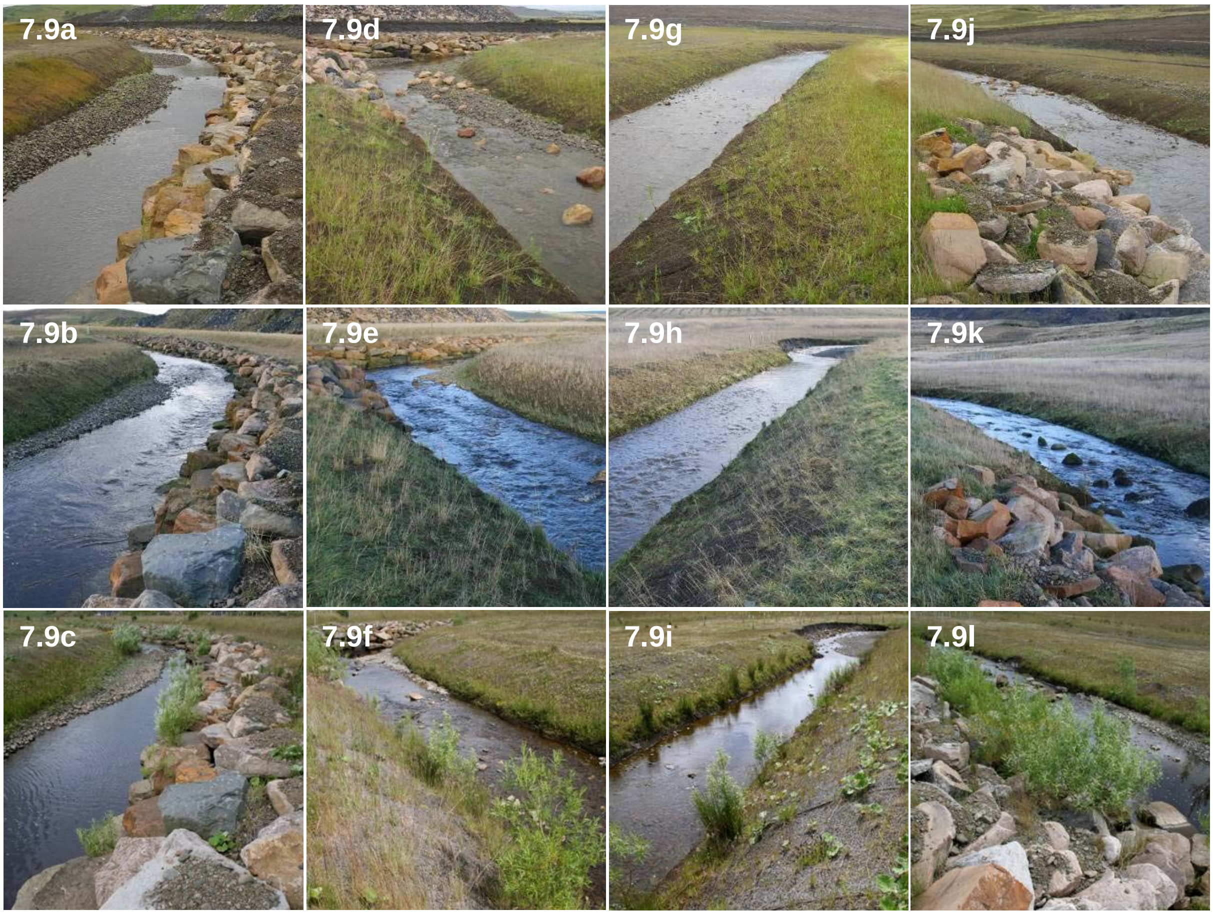


Plate 7.9. Fixed point photography from the left bank showing river habitat development between cross sections 18 and 20 (Chainage 600m to 800m) for 2005, 2006 & 2007. (a)-(c) upstream from cs18, (d)-(f) downstream from cs19, (g)-(i) upstream from cs19, (j)-(l) downstream from cs20.

Chainage 600m to 400m

Despite little change in bed level, an increase in complexity along the path of the thalweg is evident from profile data (figure 7.13). Changes are particularly clear in the lower 50m of channel and are associated with the development of gravel bars (plates 7.10 a-c) on the inside of relatively tights meander bends. Elsewhere some reorganization of bed sediments across the channel has produced a slight asymmetry likely to improve flow conditions during drought periods. This reach is ranked 10/15 (15 being most active).

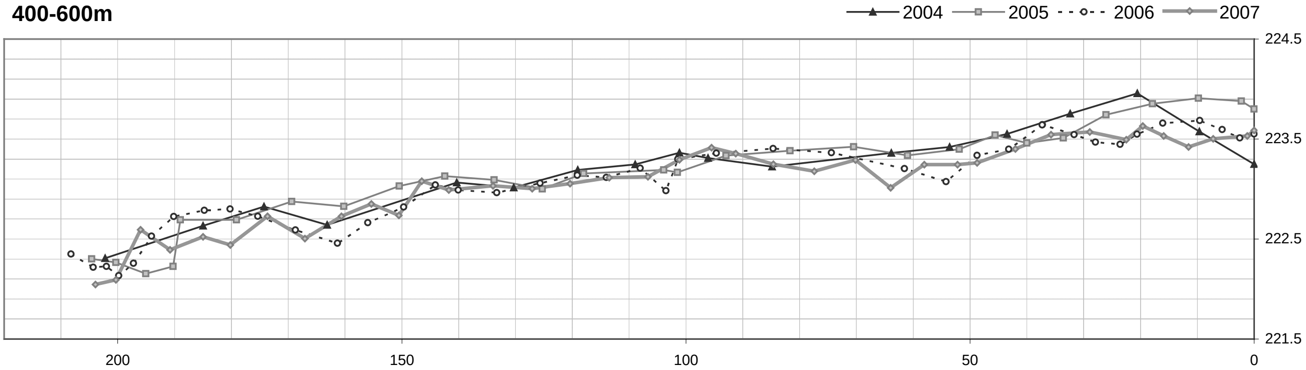


Figure 7.13. Thalweg profile showing elevations from 400m to 600m

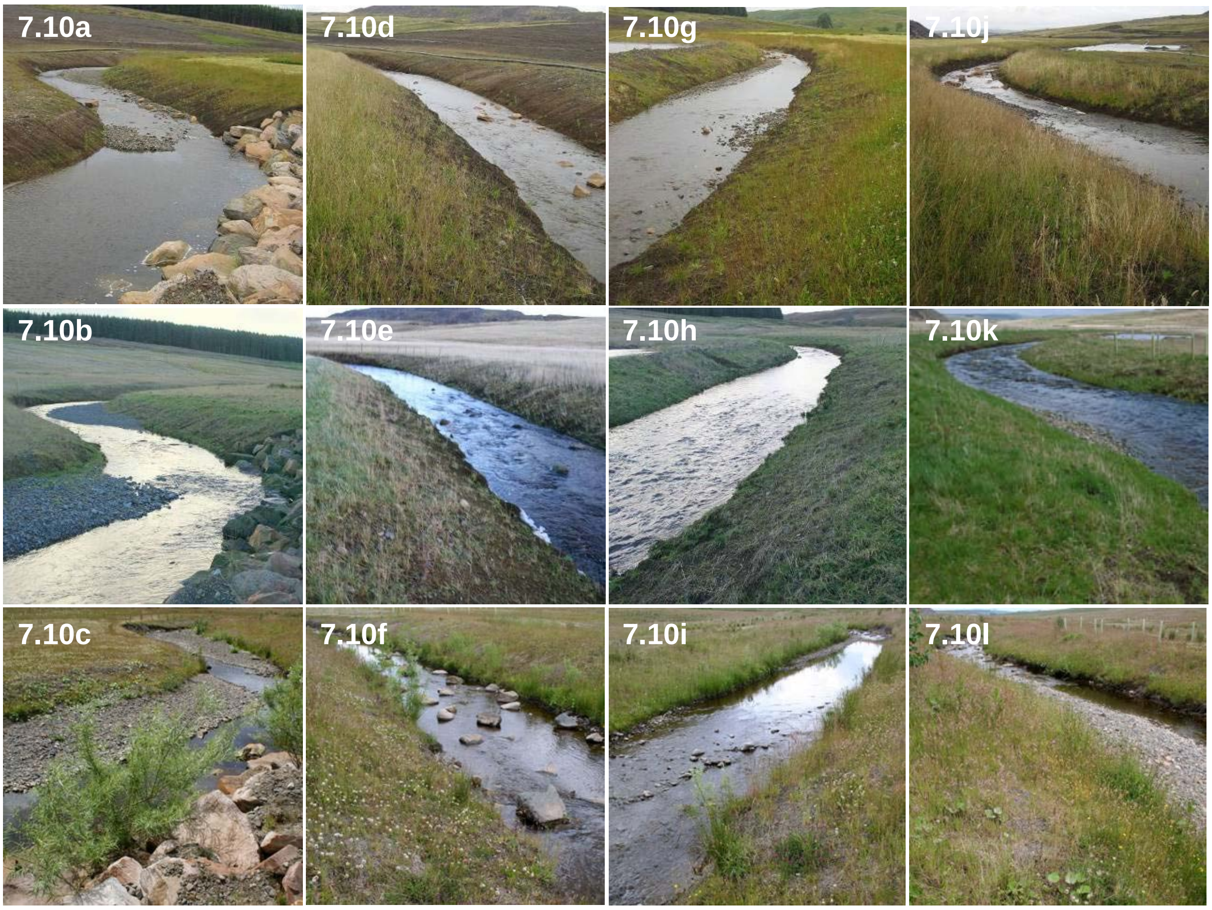


Plate 7.10. Fixed point photography from the left bank showing river habitat development between cross sections 20 and 22 (Chainage 1200m to 1000m) for 2005, 2006 & 2007. (a)-(c) upstream from cs20, (d)-(f) downstream from cs21, (g)-(i) upstream from cs21, (j)-(l) downstream from cs22.

Chainage 400m to 200m

Despite little change in bed level, an increase in complexity along the path of the thalweg is evident from profile data (figure 7.14). The profile clearly indicates area of sediment deposition at the lower end of this reach likely transported from the middle of the reach where there has been a lowering of the bed level (figure xx). These changes were temporary indicating an initial adjustment phase. This reach is ranked 8/15

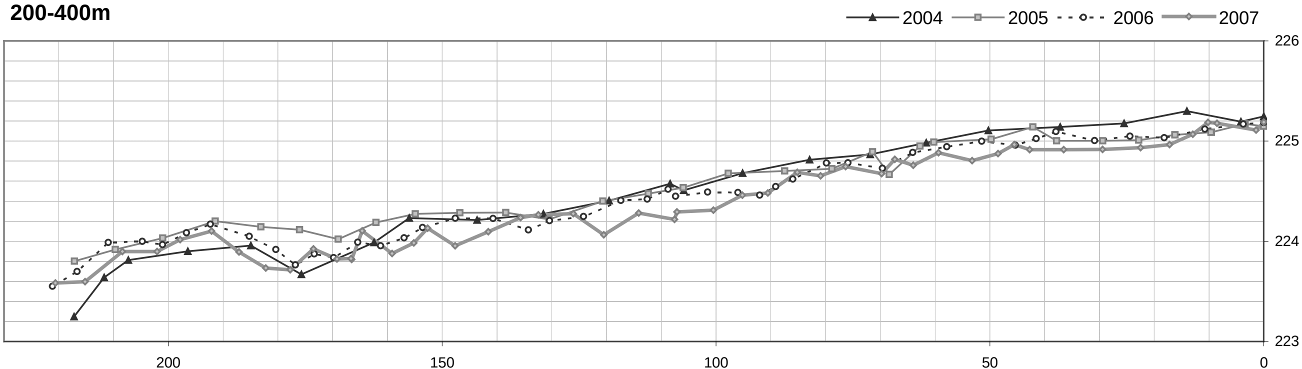


Figure 7.14. Thalweg profile showing elevations from chainage 200m to 600m

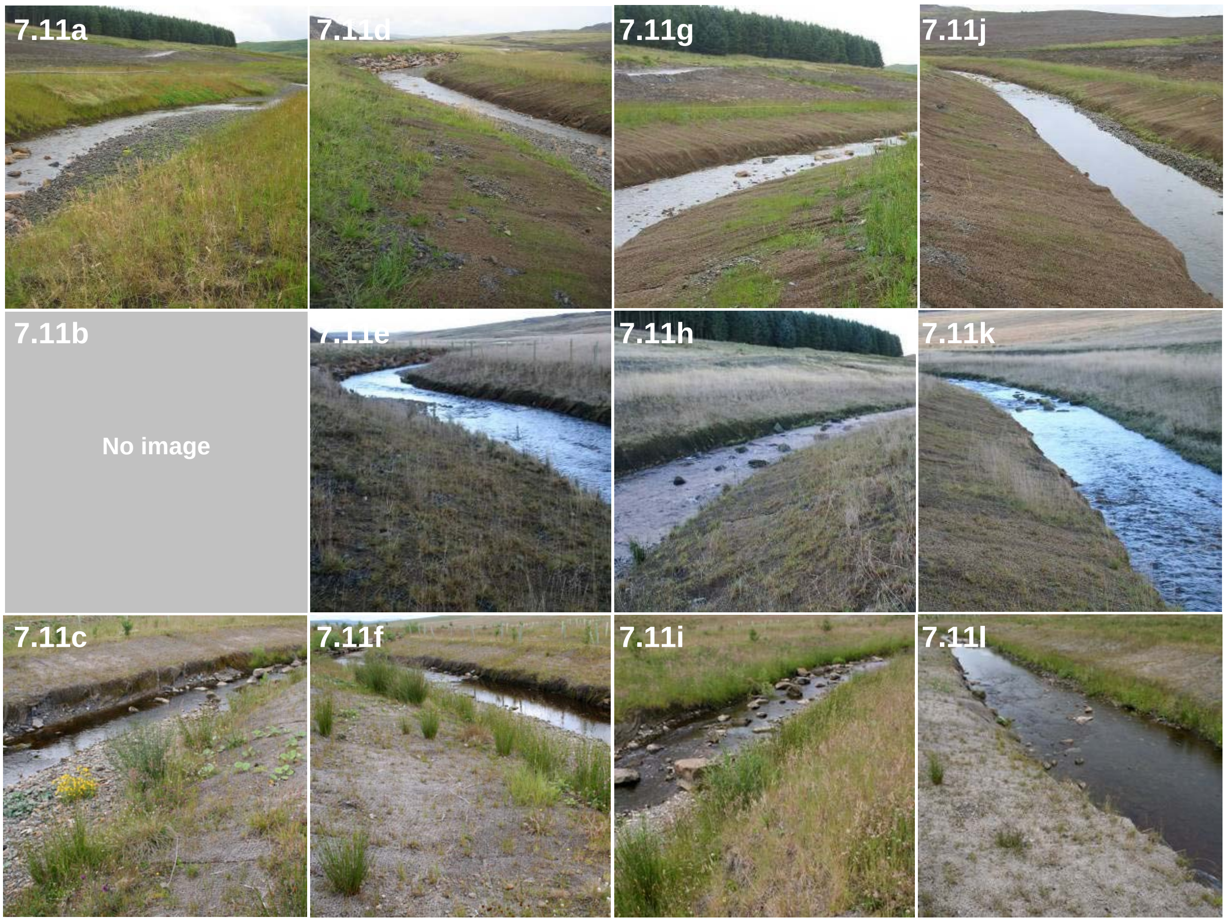


Plate 7.11. Fixed point photography from the left bank showing river habitat development between cross sections 22 and 24 (Chainage 1200m to 1000m) for 2005, 2006 & 2007. (a)-(c) upstream from cs22, (d)-(f) downstream from cs23, (g)-(i) upstream from cs23, (j)-(l) downstream from cs24.

Chainage 0m to 200m

The upper 60m, a section of relic channel with its original naturally armoured gravel bed, changes little over the study period (figure 7.15). Below this, the consecutive surveys show the bed to become more topographically complex. There is little evidence of gravel bar deposition. Armouring at cross section 25 was extended in 2006 to prevent bank erosion (Plate 7.12i). This reach is ranked 9/15.

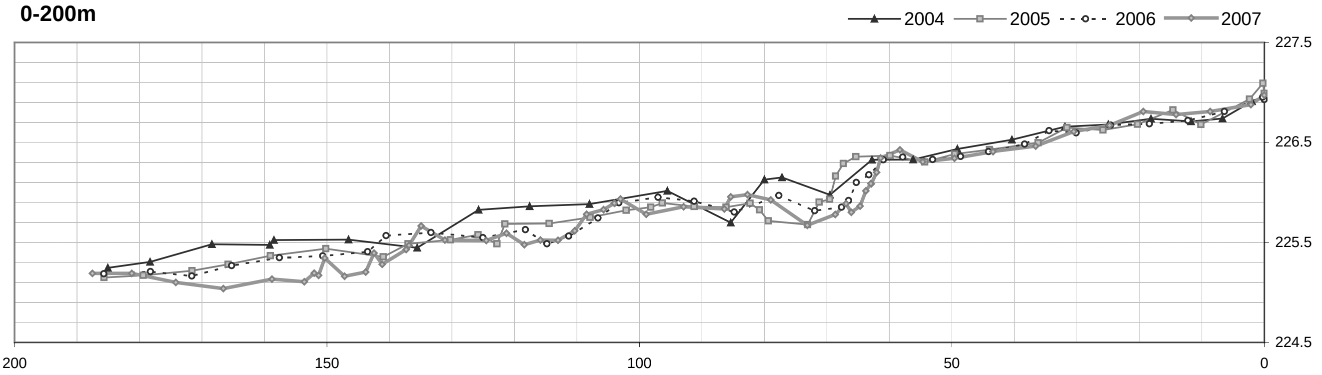


Figure 7.15. Thalweg profile showing elevations from chainage 0m to 200m

Plate 7.12. Fixed point photography from the left bank showing river habitat development between cross sections 24 and 26 (Chainage 1200m to 1000m) for 2005, 2006 & 2007. (a)-(c) upstream from cs24, (d)-(f) downstream from cs25, (g)-(i) upstream from cs25, (j)-(l) downstream from cs26.

Beoch Lane chainage 600m to 400m

Very little geomorphic activity was observed along this reach during the study period and this is reflected in the thalweg profile. No sedimentation of constructed pools has occurred and no bank erosion is visible in the photos taken in 2007. On a scale from least to most geomorphologically active this reach is ranked 3/15.

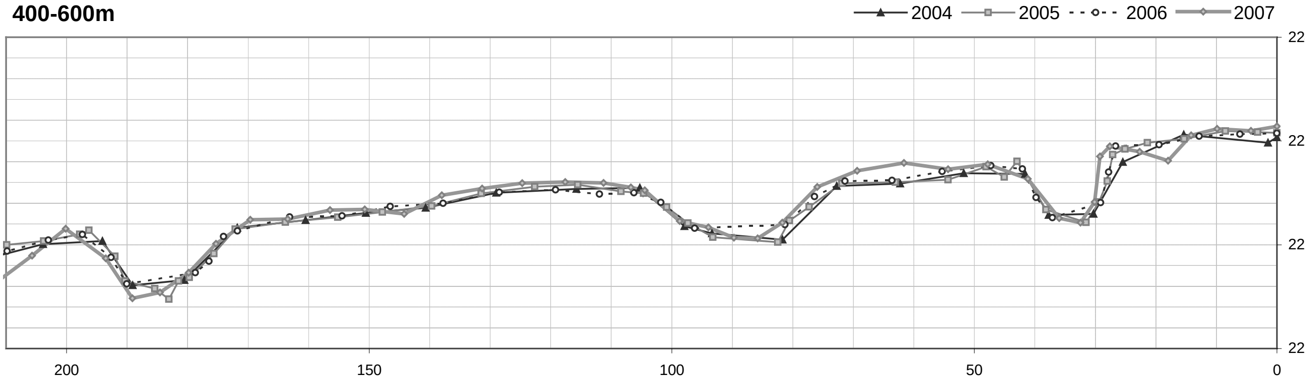


Figure 7.16. Thalweg profile showing elevations from chainage 0m to 200m

Plate 7.13. Fixed point photography from the left bank showing river habitat development between cross sections b1 and b3 (Chainage 600m to 400m) for 2005, 2006 & 2007. (a)-(c) upstream from csb1, (d)-(f) downstream from csb2, (g)-(i) upstream from csb2, (j)-(l) downstream from csb3.

Beoch Lane chainage 400m-200m

Little geomorphic activity is evident in this reach. There is some aggradation of the bed observable between 10m and 40m and infilling of the pool at the lower end (figure 7.17). This suggests sediment is being transported into the reach, possibly from the erosion feature in the upstream section. However, there are no fluvial features of the scale expected or this size of channel. This reach is ranked 4/15.

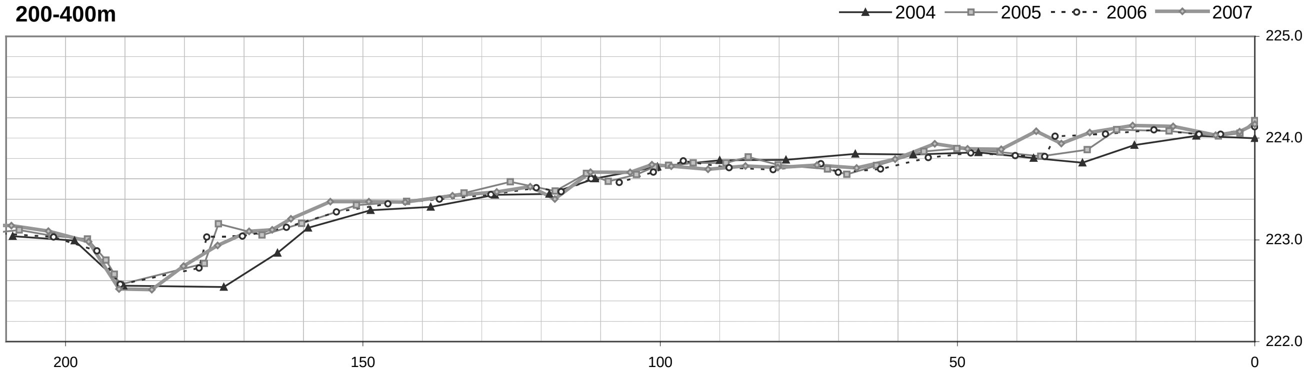


Figure 7.17. Thalweg profile showing elevations from chainage 400m to 200m

Plate 7.14. Fixed point photography from the left bank showing river habitat development between cross sections b3 and b5 (Chainage 400m to 200m) for 2005, 2006 & 2007. (a)-(c) upstream from csb3, (d)-(f) downstream from csb4, (g)-(i) upstream from csb4, (j)-(l) downstream from csb5.

Beoch Lane chainage 200m to 0m

The agredation of the bed in the upper section of this reach between 2004 and 2005 coincides with erosion of the unprotected bank. There has been extensive geomorphic activity below the point where the channel crosses an old bentonite wall. Notably widening of the channel and the formation of a gravel bar.

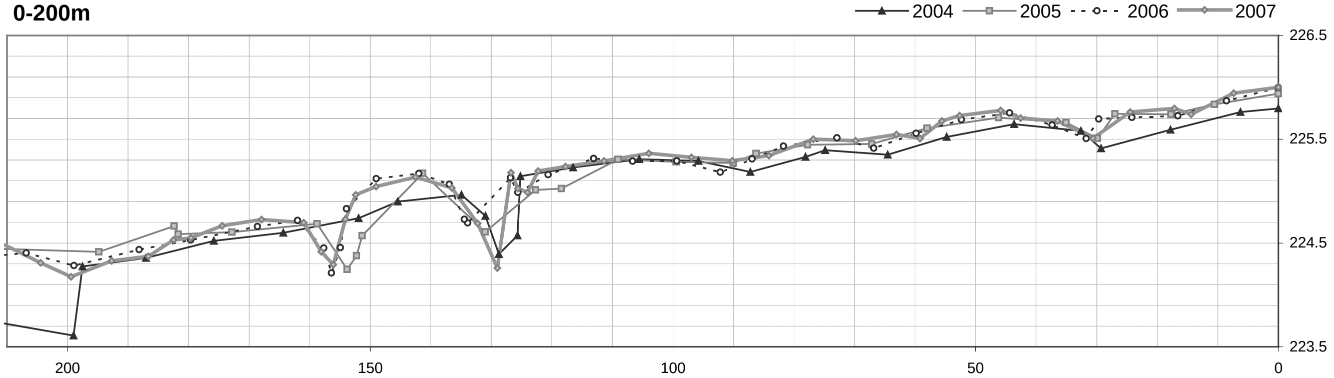


Figure 7.18. Thalweg profile showing elevations along the Beoch Lane Burn from chainage 200m to 0m

7.15a No image

7.15d No image

7.15g No image

7.15j No image

7.15b No image

7.15e No image

7.15h No image

7.15k No image

7.15l No image

Plate 7.15. Fixed point photography from the left bank showing river habitat development between cross sections b5 and b7 (Chainage 200m to 0m) for 2005, 2006 & 2007. (a)-(c) upstream from csb5, (d)-(f) downstream from csb6, (g)-(i) upstream from csb6, (j)-(l) downstream from csb7.

7.2.2 Application of diversity indices to the thalweg profile data

Although the height profiles and photographic evidence show clearly the patterns of geomorphic development, quantification of physical habitat diversity was desirable. A number of diversity indices were applied to the data presented in the form of reach thalweg profiles (Section 7.2). These attempt to summarise the physical habitat diversity into a figure that would facilitate the comparison of reaches and relate differences in development to landscape and design factors. This would allow the application of formal statistics to rates of development for investigation the driving factors behind development.

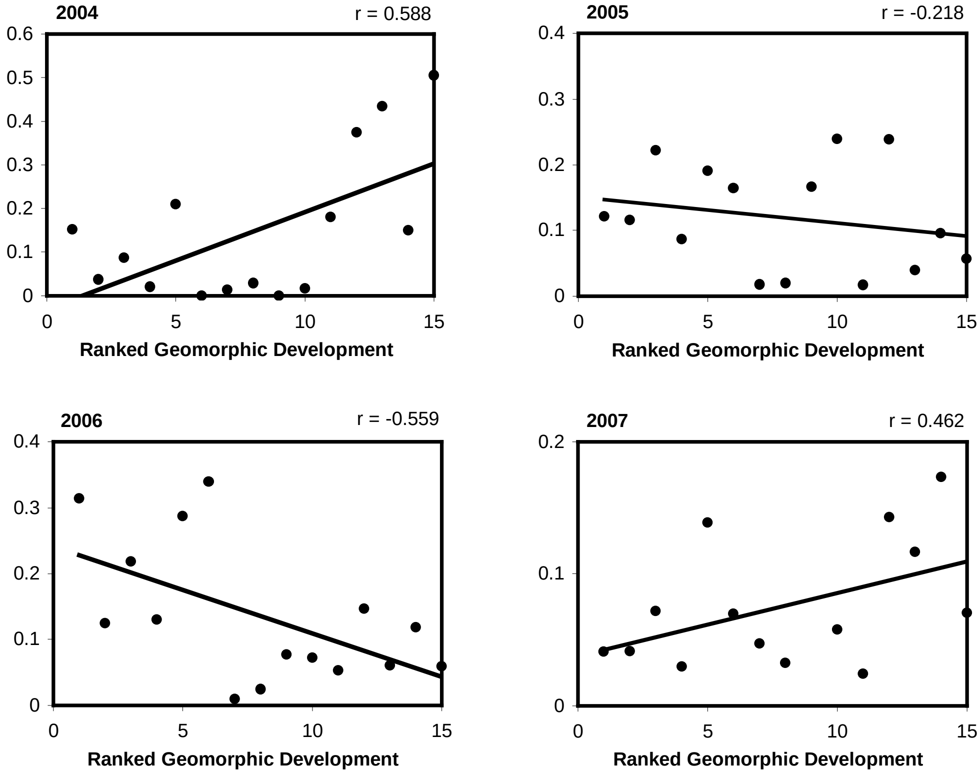


Figure 7.19. Graphs correlate the different scores achieved by reaches within the realignment based on visual ranking and calculation of vector dispersion. Pearson's correlation coefficient (r) is shown.

Figure 7.19 shows no consistent meaningful correlation between the development of reaches assessed by expert judgement and by the proposed index of physical habitat diversity; vector dispersion. Although scores achieved by vector dispersion seem to reflect the rank scores in 2007, albeit weakly (pearson's r = 0.462) the same index negatively correlates with the rank scores in the previous year (r= -0.559). A similar pattern is observable analysing the same data using the Wiggle index (Figure 7.20). Calculation of the standard error of the residual elevations shows a very weak positive correlation in all years (figure 7.21). However this index is strongly affected by reach gradient and it is likely that the relationship of steeper reaches having more energy is showing through this index.

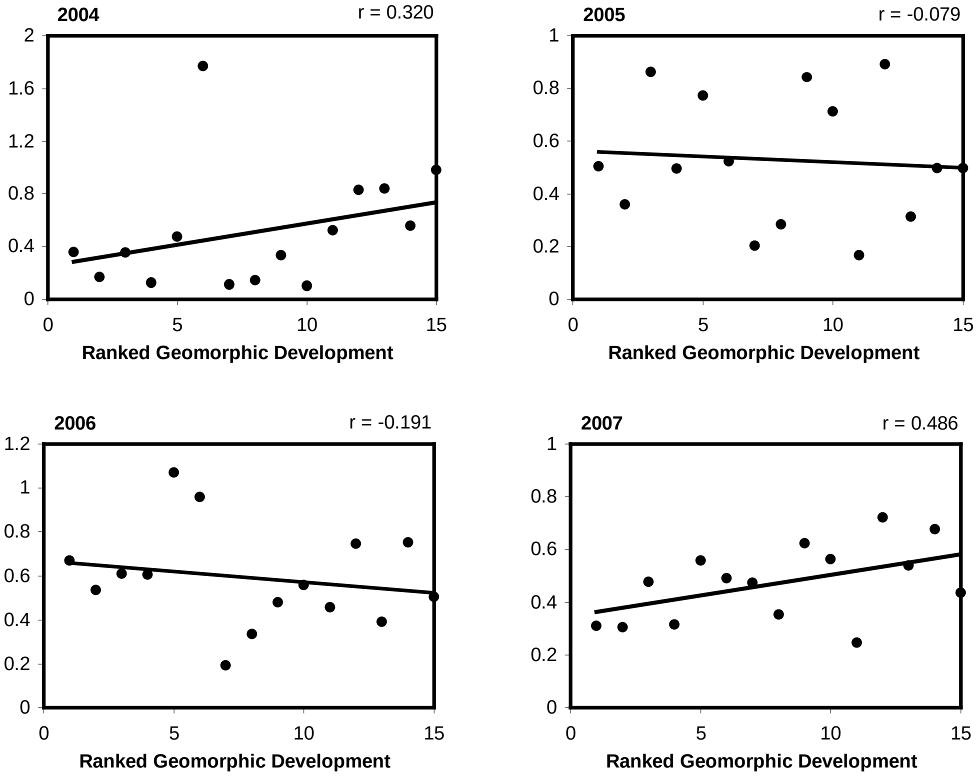


Figure 7.20. Graphs correlate the different scores achieved by reaches within the realignment based on visual ranking and calculation of vertical wiggle. Pearson's correlation coefficient (r) is shown.

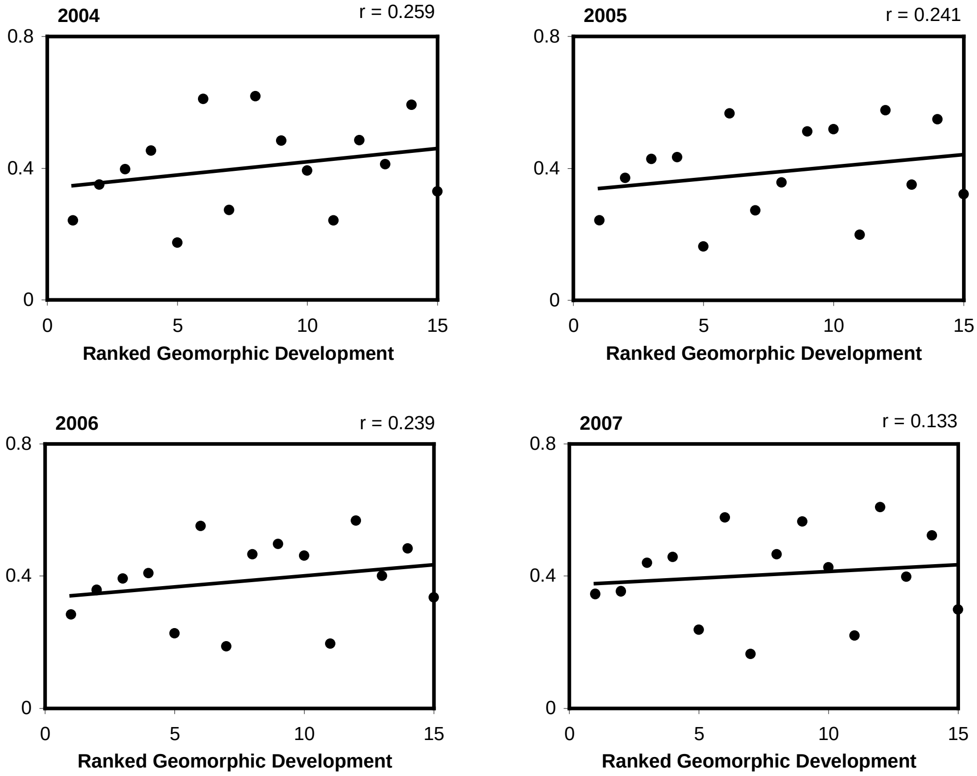


Figure 7.21: Graphs correlate the different scores achieved by reaches within the realignment based on visual ranking and calculation of the standard error of the residual elevations. Pearson's correlation coefficient (r) is shown.

In addition to the indices of physical habitat diversity recommended in the literature, values of within channel sinuosity were also calculated as indicators of physical habitat diversity that are more directly related to hydraulic diversity. Sinuosity was calculated for both the increased flow path as a result of vertical variability (relative to channel centreline) (figure 7.22) and the increased flow path length as a result of lateral variability (relative to the channel centreline) (figure 7.23)

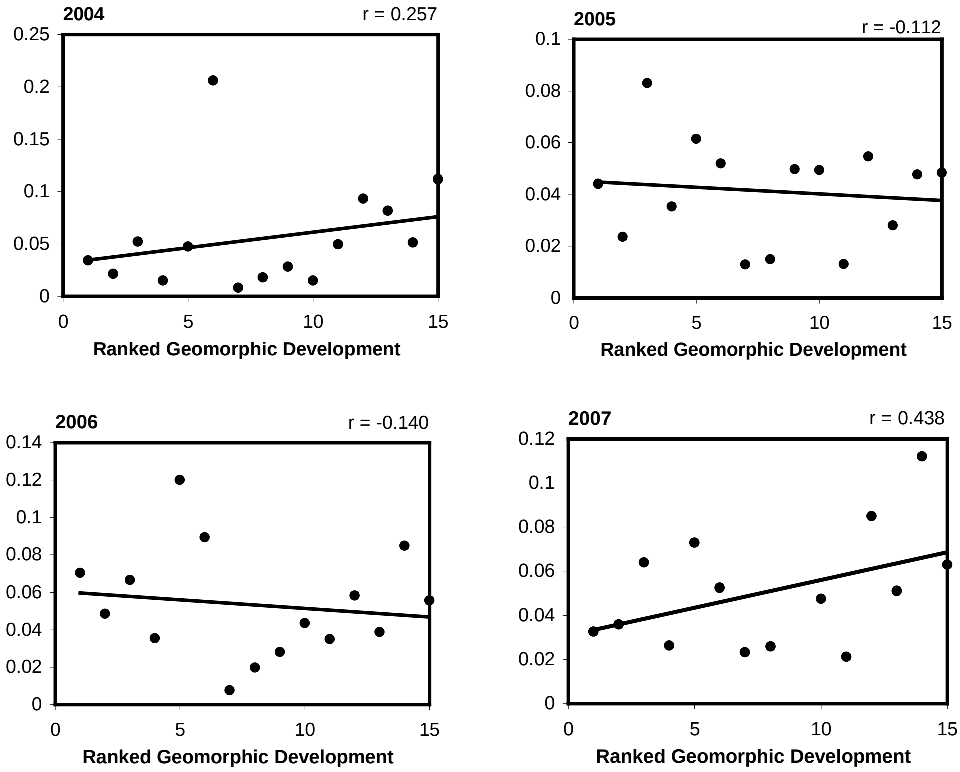


Figure 7.22. Graphs correlate the different scores achieved by reaches within the realignment based on visual ranking and calculation of lateral sinuosity. Pearson’s correlation coefficient (r) is shown.

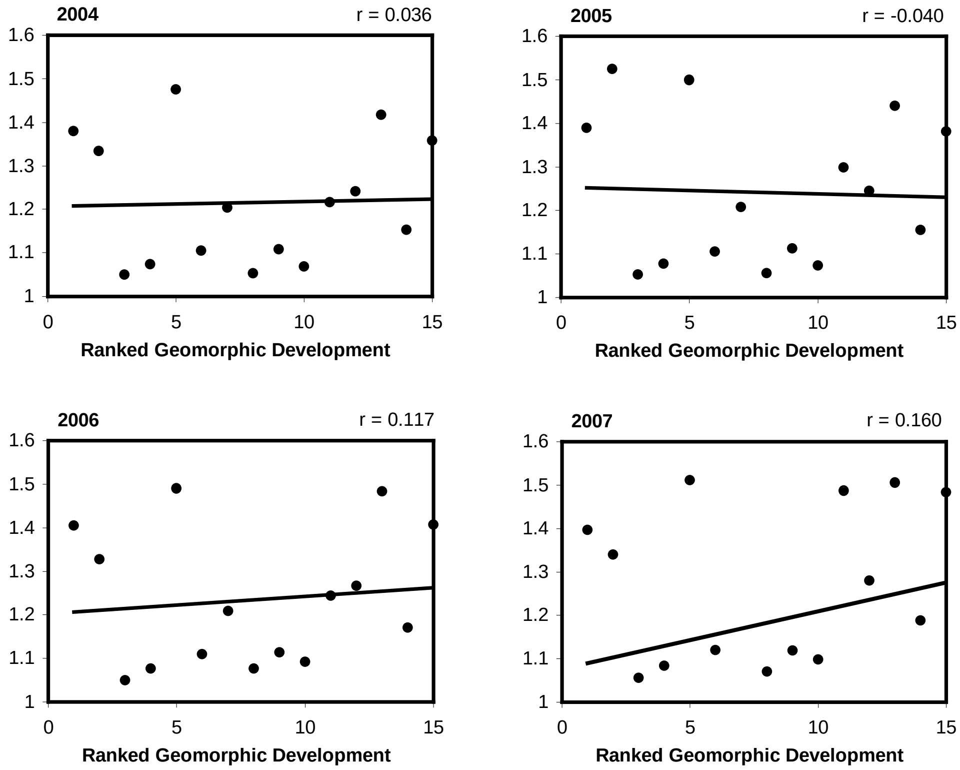


Figure 7.23. Graphs correlate the different scores achieved by reaches within the realignment based on visual ranking and calculation of lateral sinuosity.

The bed gradients of each successive 200m reach are shown in figure 7.24. Gradients are calculated as the regression slope of survey points along the thalweg taken over reaches of channel 200m in length. It is clear there has been very little change in overall gradient of the different sections of the realignment. The exception being the middle reaches where gradient fallen. This is within a region of observed settlement (Appendix) predicted to occur where the river passes across the area deepest backfill.

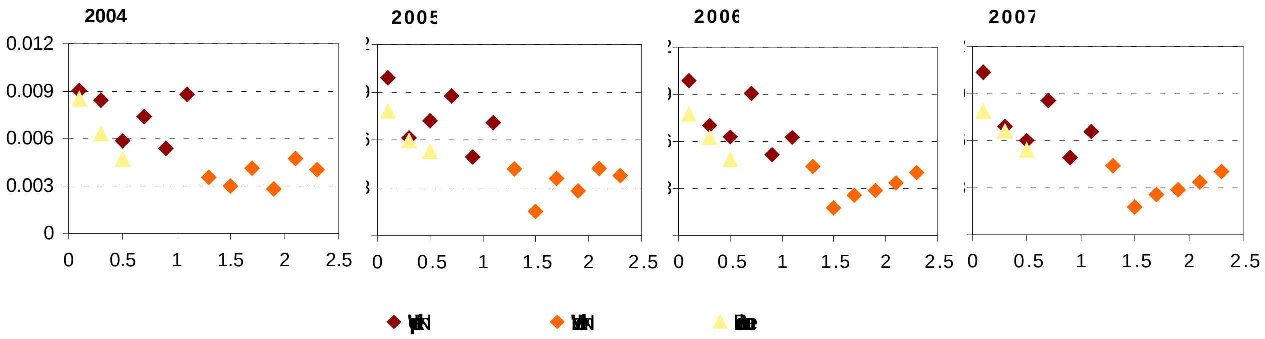


Figure 7.24. Graphs showing the regression slope of successive 200m reaches along the length of the realignment. Recorded in successive years. Distances downstream from the top of the realignment are plotted on the x-axis against the regression slope on the y-axis

The channel changes in character along the length of the realignment with the upstream sites being steeper than those in the middle and lower reaches. The different gradients will have different hydraulic character and hence are expected differentiate geomorphologically.

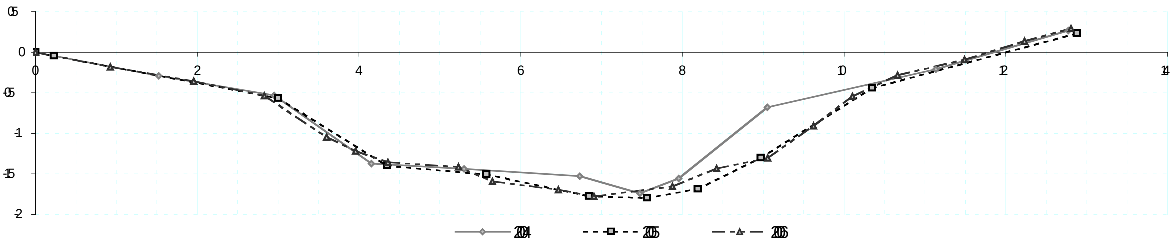
The gradient in the constructed channel reflected the natural slope of the valley with a steeper upper section flattening out at a point approximately halfway along the realignment. The constructed bed gradient however was not consistent from reach to reach although this is likely to partly reflect the positioning of pools and riffles.

It is clear from the photos in Sction 7.2 and profile graphs (figure 7.25) that there was very little change in cross-sectional form at the majority of cross-sections on the beoch diversion. This is confirmed by the patterns of movement of tracer pebbles placed on the river bed in April 2005 (table xx). All pebbles 26.5- 128mm remained in a straight line along the cross-section B2 (figure 7.26) throughout the study and were still present in 2008.

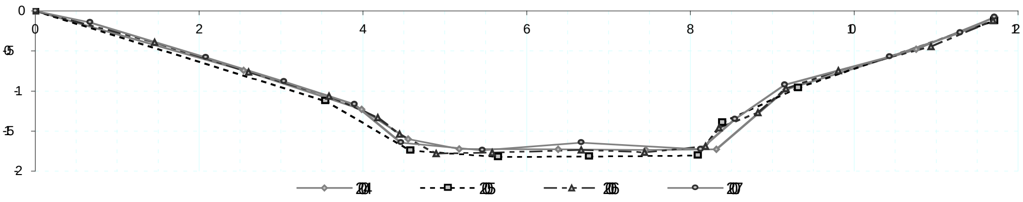
7.3 Cross-section profiles

7.3.1 Development of cross channel profiles.

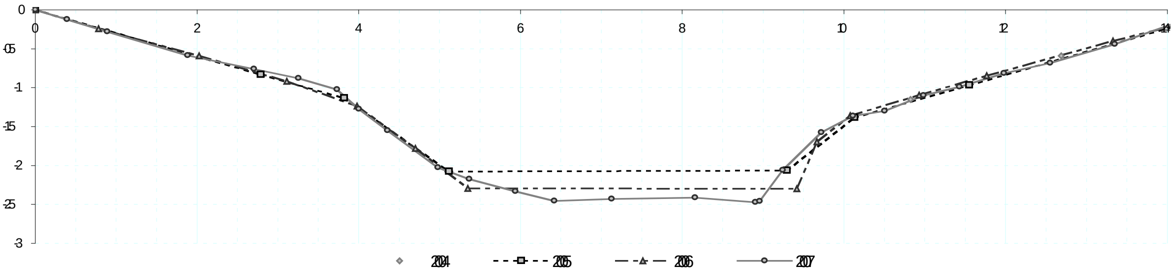
Cross-channel profiles indicate relativity limited development of channel form along much of the realignment. The deposition of point bars is evident where cross-sections co-inside with the apex of meanders bends and in fixed point photography. Where this has occurred the development of cross channel diversity has been captured (Figure 7.25; cross-section 13,) although significant levels of diversity are limited to areas of erosion and deposition. Variability in cross-sectional form has developed to the greatest degree, in the middle, geomorphically active reaches (Cross-sections 10-18, figure 7.25). In the lower reaches of the diversion there has been little or no change in cross-section profiles (Cross-sections 4-9, figure 2.5) and banks have a uniform appearance (Plates 7.1-7.4). Cross-sectional profiles also show an absence of morphological development at the bank top positions at locations.



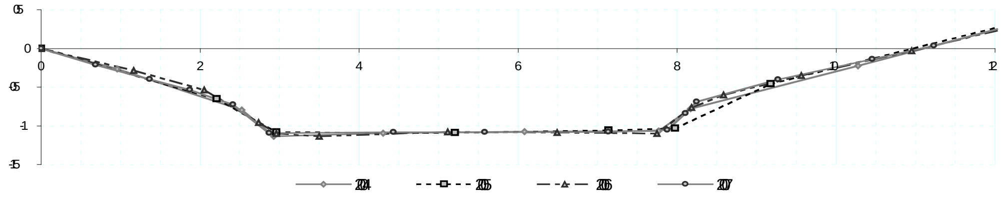
Cross-section 4



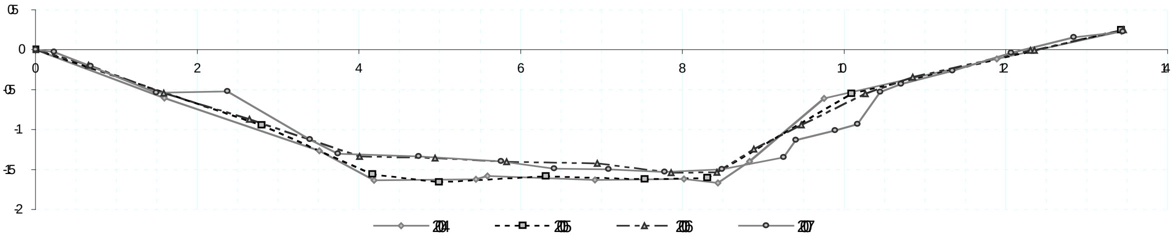
Cross-section 5



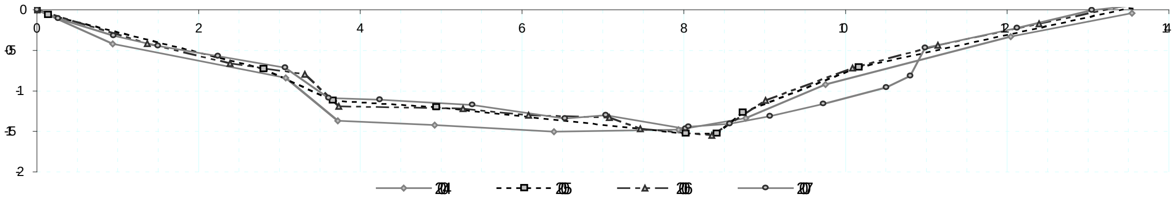
Cross-section 6



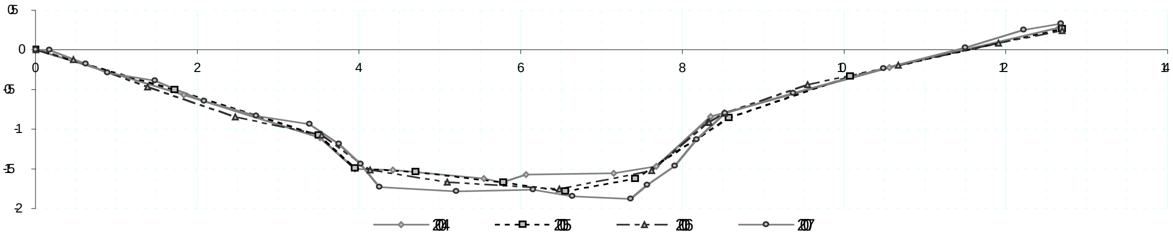
Cross section 7



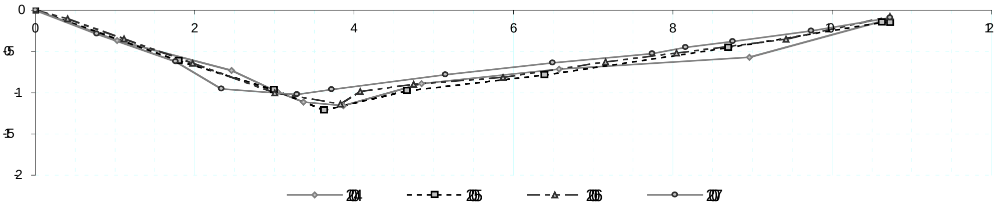
Cross-section 8



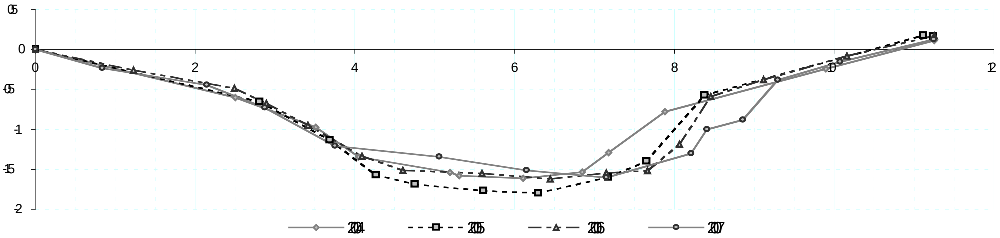
Cross-section 9



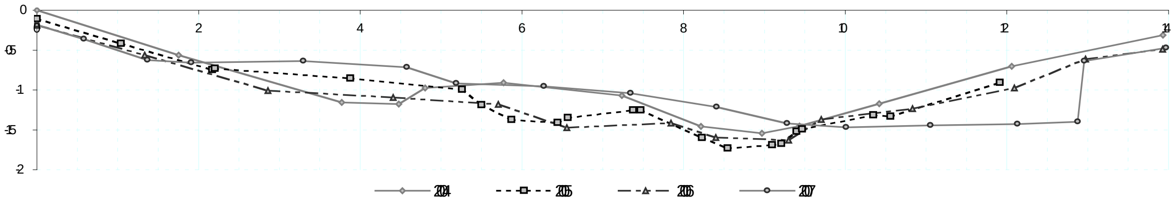
Cross section 10



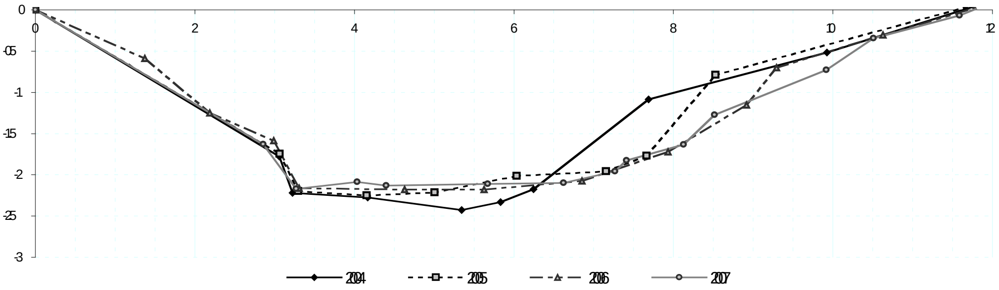
Cross-section 11



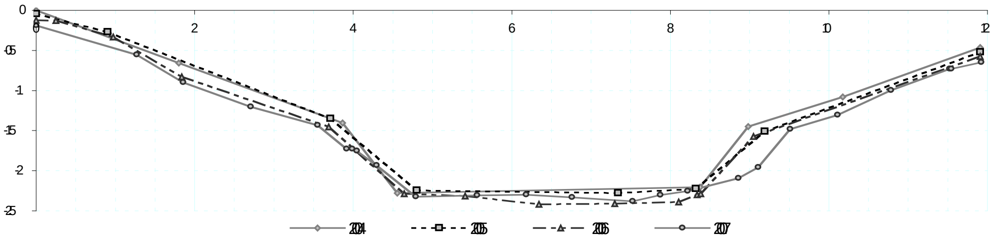
Cross-section 12



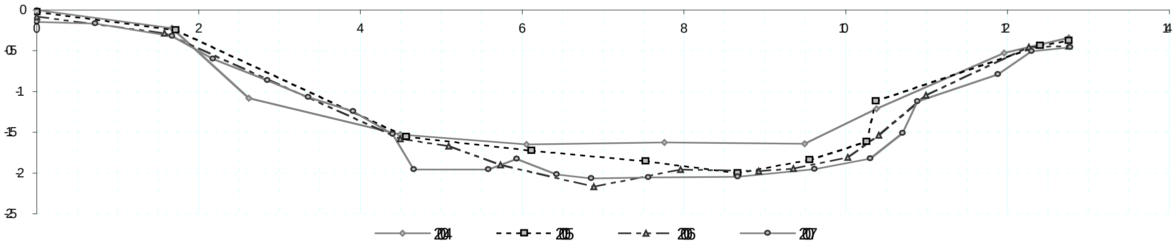
Cross-section 13



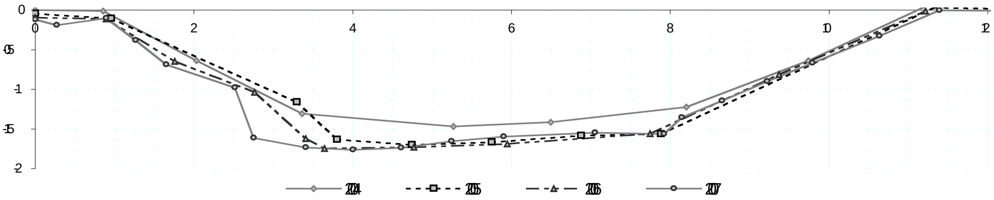
Cross-section 14



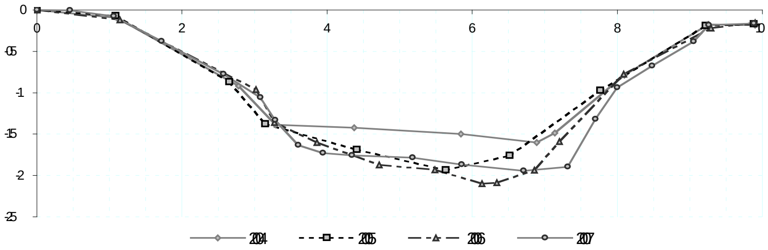
Cross-section 15



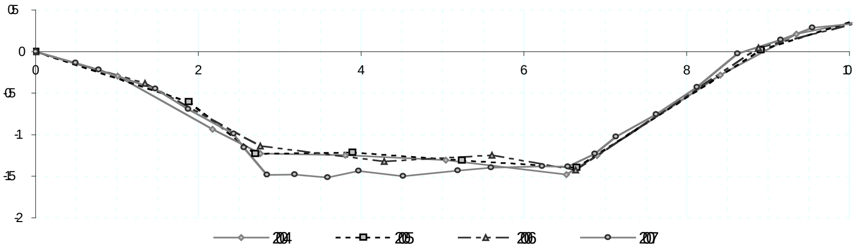
Cross-section 16



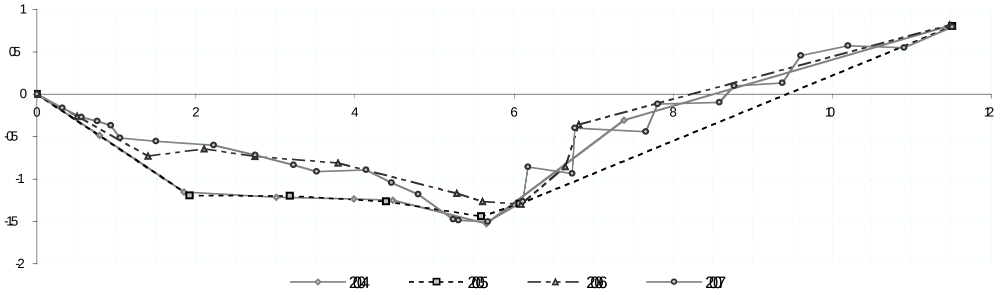
Cross-section 17



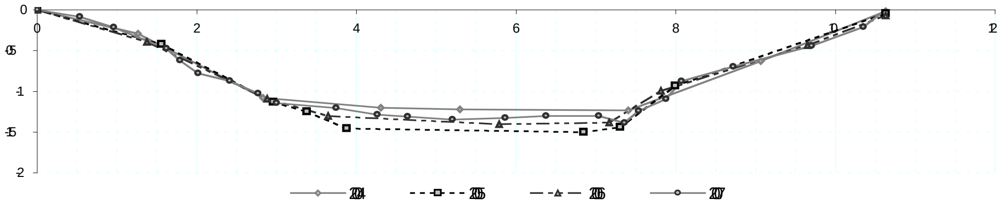
Cross-section 18



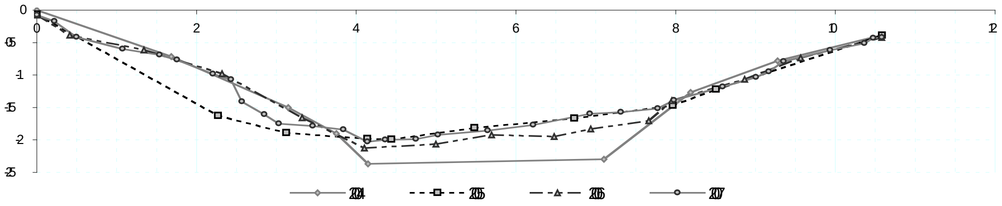
Cross-section 19



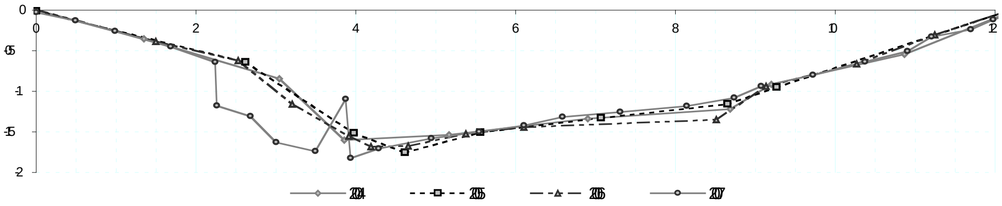
Cross-section 20



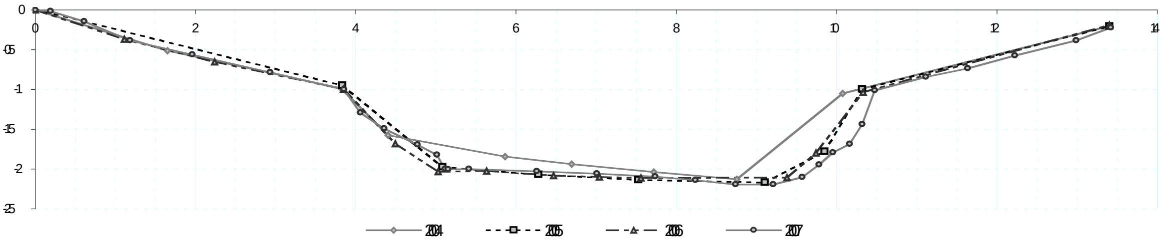
Cross-section 21



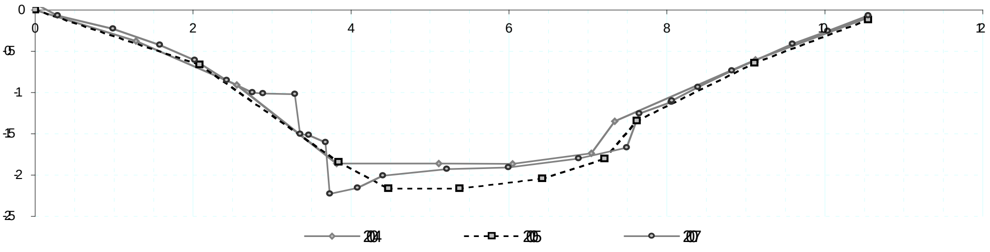
Cross-section 22



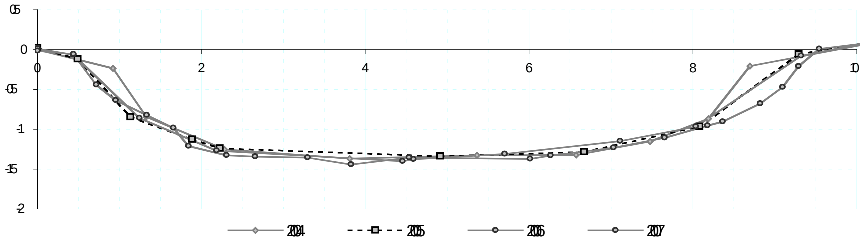
Cross-section 23



Cross-section 24



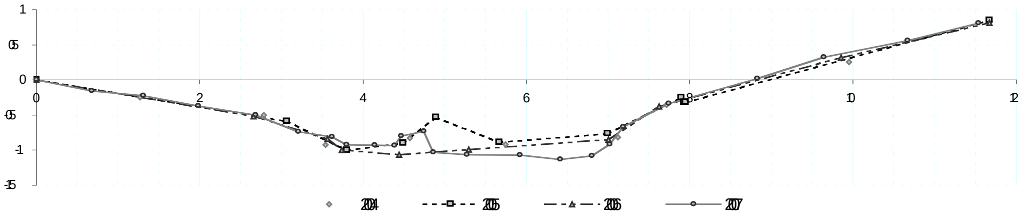
Cross-section 25



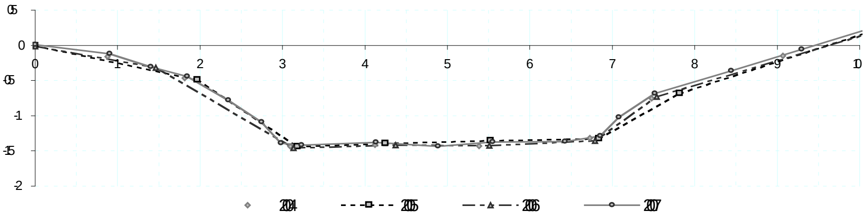
Cross-section 26

Figure 7.25. Profile graphs showing cross-channel elevations for transects positioned every 100m from the bottom to the top of the river realignment. Cross-sections are numbered from the downstream end.

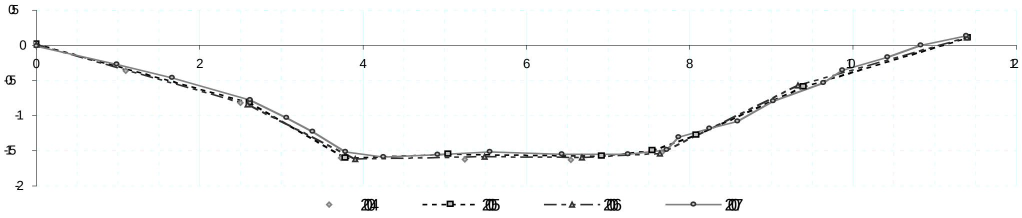
Cross-section profile of the beoch lane show there has been no significant geomorphological adjustment of the stream bed.



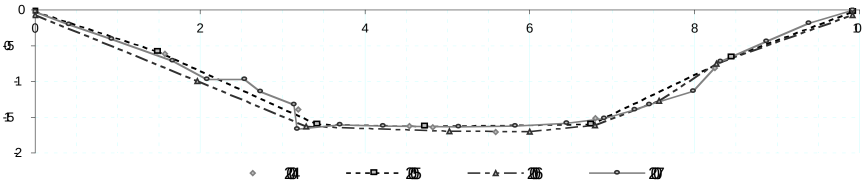
Cross-section b1



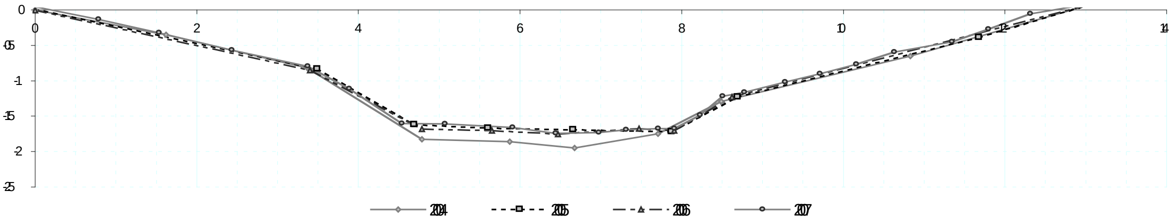
Cross-section b2



Cross-section b3



Cross-section b4



Cross-section b5

Figure 7.26 Profile graphs showing cross-channel elevations for transects positioned every 100m along the realigned reach of the Beoch Lane Burn realignment. Cross-sections are numbered from the downstream end.

7.3.2 Application of diversity indices to cross-section data

Wetted perimeter to bankfull width ratio and vector dispersion both measure the topographic complexity of the channel cross-sections. Marked increases were detected by vector dispersion for all river section in 2007. These were most notable for the steeper reachs on the upper Nith and Beoch Lane (figure 7.27) this pattern does not fit with visual interpretations of the cross-section profiles that show no changes on the Beoch lane profiles (figure 7.26)

At 14 of the 25 cross-sections the morphological development is asymmetrical. Bank erosion features are frequently accompanied by deposition on the opposite channel margin (Figure 7.25; cross-sections 8, 9, 11, 12, 13, 23). The pattern is less clear but still present at cross-sections where asymmetrical development is accompanied by vertical adjustment of bed height. (Figure 7.25; cross-sections 17, 18). Figure 7.28 shows moderately rapid development of an asymmetrical form at cross-sections in the middle reaches. The greatest development in on cross-sections in close proximity to a meander apex. In other sections, where asymmetry develops it is a very gradual change but consistent across the years. Where cross-sections start asymmetrical, development may increase the symmetry of the channel.

There is no appreciable change in the variability of cross-sectional area of channel depth at either the realignment scale or section scale (figure 7.29). Nor did any significant differences between river sections develop despite having very different discharges and slopes (figure 7.24).

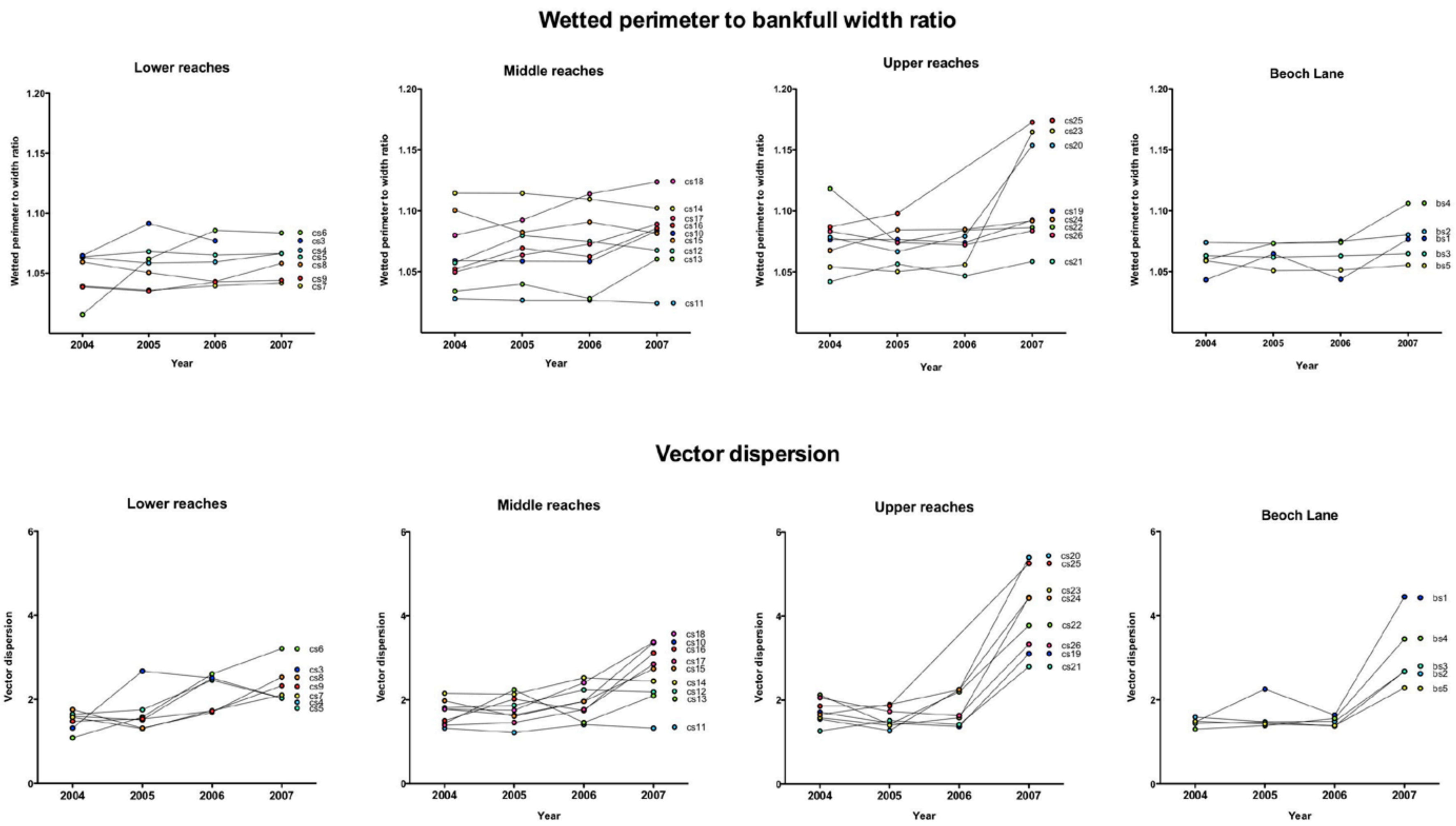


Figure 7.27. Graphs showing the changes in cross-sectional character over the four year study period. The realignment is divided into four sections presented separately, each containing reaches of similar hydrology and gradient. Values of wetted perimeter to bankfull width ratio and cross-channel vector dispersion are presented.

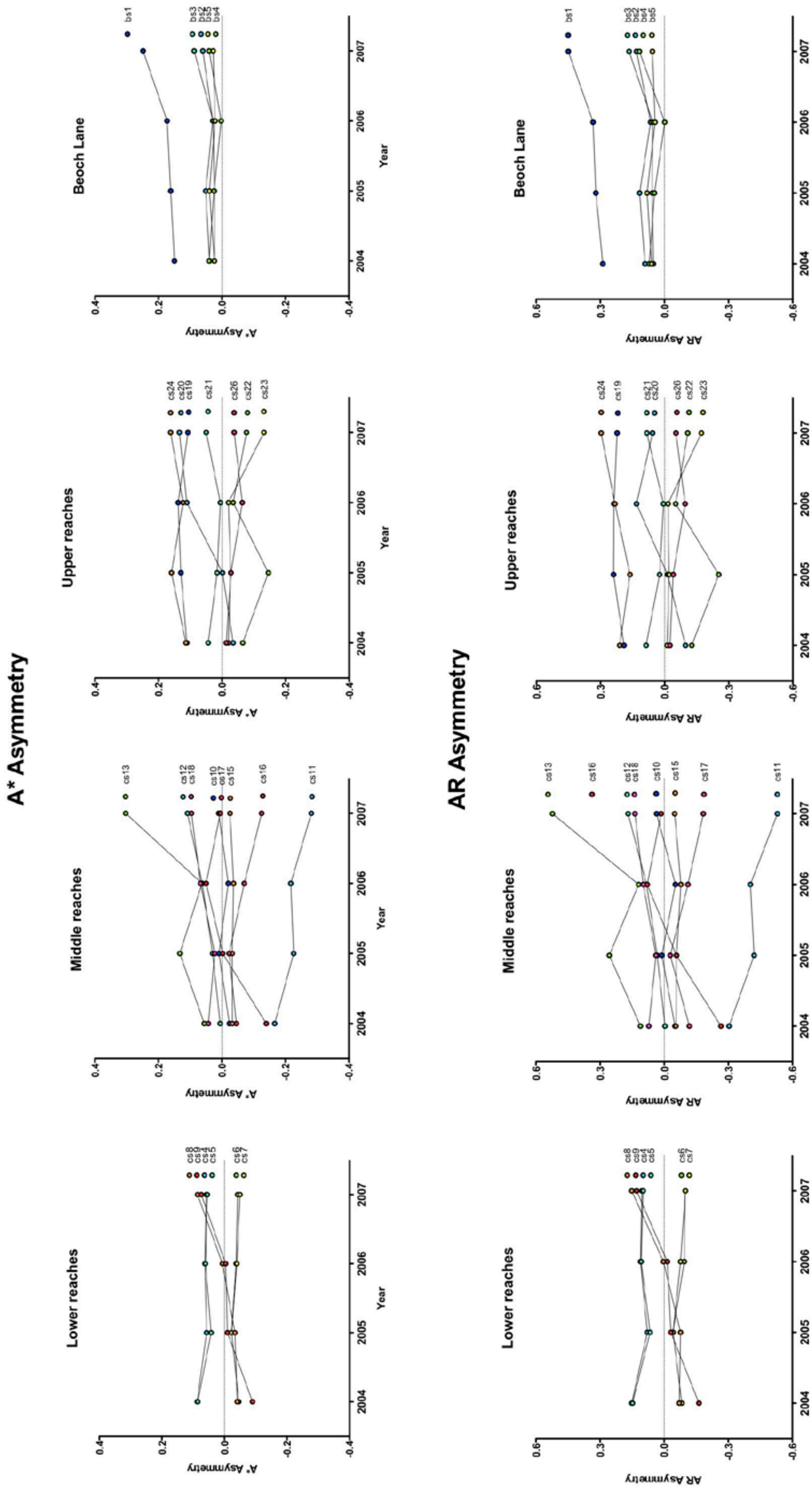


Figure 7.28. Graphs showing the changes in cross-sectional character over the four year study period. The realignment is divided into four sections presented separately, each containing reaches of similar hydrology and gradient. Values of asymmetry in area and hydraulic radius are presented are presented.

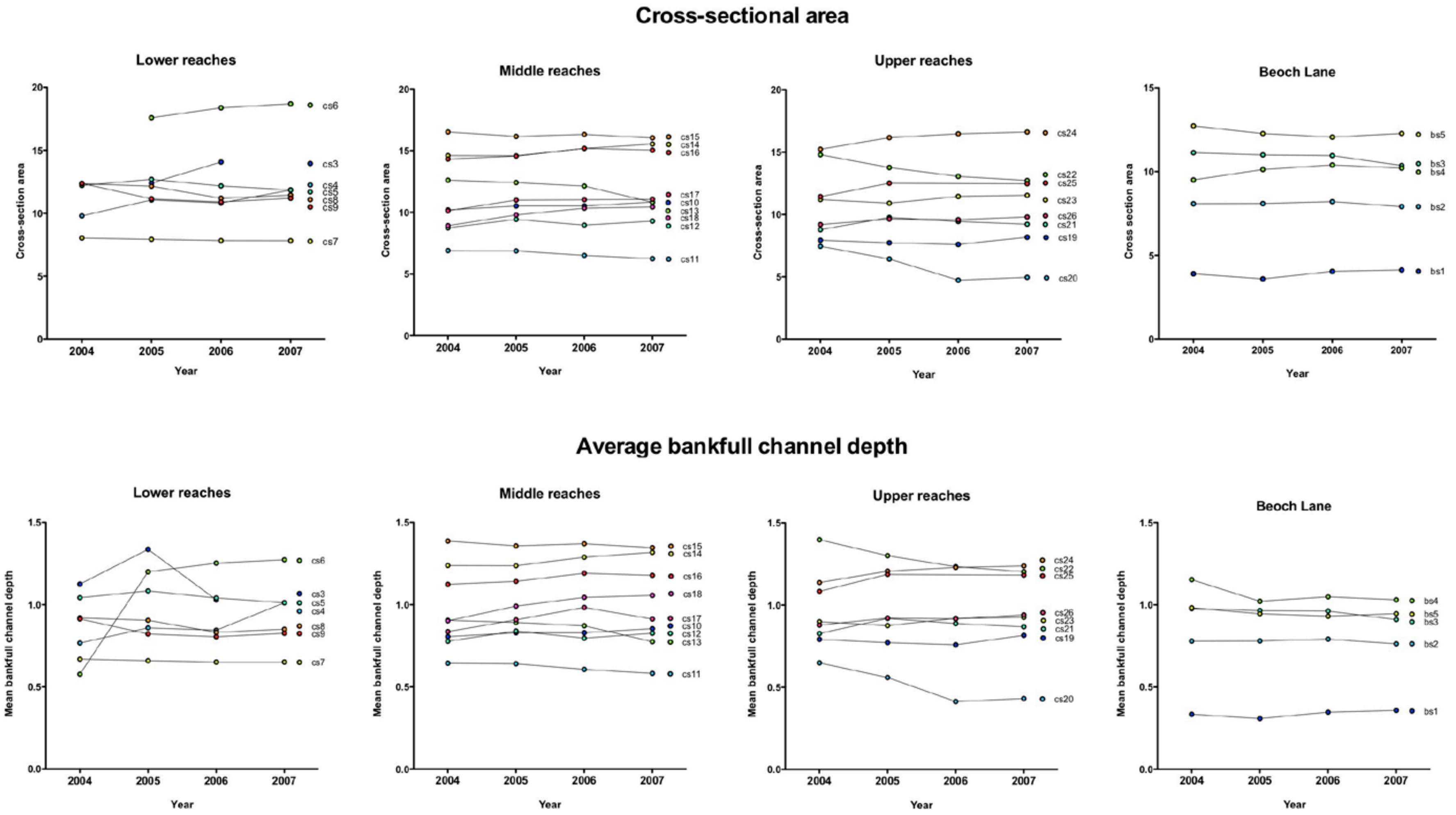


Figure 7.29. Graphs showing the changes in cross-sectional character over the four year study period. The realignment is divided into four sections presented separately, each containing reaches of similar hydrology and gradient. Values of cross-sectional area and average bankfull channel depth are presented.

7.4 Retention capacity

7.4.1 Results of leaf release experiments

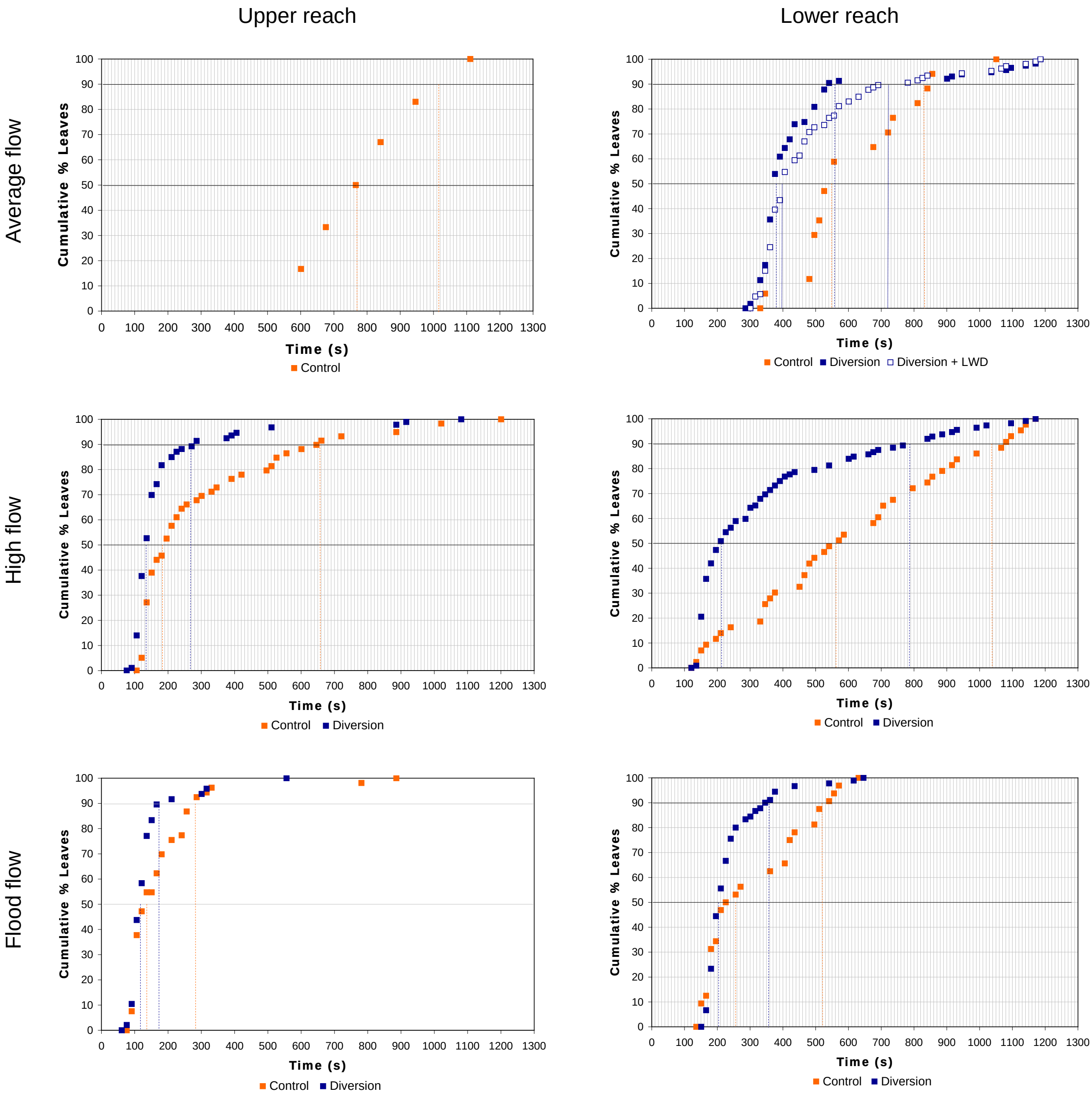


Figure 7.30: Results of leaf release experiments showing the time taken for leafs to travel through a 200m reach of the diversion. Graphs show the cumulative percentage of leaves that travelled through the reach (y-axis) over time (x-axis). Leaves retained within the reach are excluded.

The results from leaf release experiments are shown in figure 7.30. In all cases the first leaf took the same time to travel through the realignment reach as its respective reference reach indicating the path of fastest flow. This was a fairly simple sinuous path under high flow conditions. The average flow path is indicated by the 50th percentile leaf. Under flood conditions both the upper and lower reaches of the realignment are comparable with there respective reference reaches. However under all other conditions the average flow path is longer in the reference reaches than the realignment. The difference between the realignment and respective reference reaches generally increased up to the 80th-90th percentile leaf. Under low flow conditions the all leaves were retained at all locations with the exception of the reach on the lower realignment where two leaves travelled the full 200m (taking 1530s and 1710s).

CHAPTER 8

RESULTS II: INVERTEBRATES

This chapter presets results from surveys of the invertebrate assemblages colonising benthic habitat within the engineered realignment. Data is interpreted through the application of indices that reveal different aspects of community diversity. Surveys of benthic invertebrates are widely recommended for the evaluation of restoration projects. However, perhaps as a result of the rare implementation of monitoring, there is little consensus on the most appropriate indices for extracting meaningful ecological patterns. Results are therefore presented here using a range of approaches. Rates and patterns of development reflect the dispersal pathways used by colonists and the influence of habitat factors.

Aims

Present relationships that provide insight into dispersal processes
Investigate the biological traits behind colonist communities
Investigate the development of community structure

8.1 Diversity relationships with downstream distance

At early stages of development there are clear relationships between measures of ecosystem diversity and distance downstream from the top of the realigned section (Figure 8.1). However it is clear that the different indices differ in sensitivity to the pattern.

The simplest index – species richness – is the clearest, remaining significant in the very early stages of development when the number of 'pioneering' individuals is very low. In October 2004 sample abundance is less than 50 at all realignment sites with the exception of N1 and the lowest two sites, N7 and N8, remained un-colonised by any individuals. A strong correlation is shown between distance and richness from October 2004 ($r = -1.00$, $p=0.008$) through April ($r = -0.847$, $p=0.005$) to July 2005 ($r = -0.886$, $p=0.004$). The strength of the correlation suggests that no dispersal pathways are feeding the diversion with colonists from alternative sources or directions. It is clear that the effectiveness of the dispersal pathway is reduced with increased distance from the upstream end.

The slope of the distance-richness correlation reduces through time, as distant sites achieve higher levels of colonisation. This relationship is plotted in figure 8.2. The slope of this graph, representing the rate at which the distance diversity relationship degrades is clearly the same in both the 2000 and 2004 realignments. This would imply the processes governing recovery are time dependent. The generalised relationship is shown in figure 8.3.

Most sites see a jump in *richness* between July and November 2005. With the exception of sites N5 and N6 (figure 8.4), *richness* effectively doubles from 20 to 40 species. It is not until the following October that N5 and N6 are again equivalent in richness to the other rest of the sites. This pattern is also evident, but not as pronounced, in other measures of diversity presented. Interpretation of this pattern needs to be based on more than just *richness*. Graphs of *abundance* highlight further differences in diversity in November 2005. At this time sites N7 and N8 located at the downstream end have over twice the sample abundance of any other site indicating a higher density of invertebrates on the riverbed (figure. 8.5). Rarefaction of the November data suggests that this difference may be sufficient to explain the high richness at these sites relative to N5 and N6. Analyses based on rarefaction suggest that the underlying pattern of alpha diversity persists until at least April 2006. Patterns in *Fisher's alpha* suggest this relationship persists through to October 2006. It's possible that the non significance in April 2006 maybe a type II error.

Patterns in scores from *Shannon's* and *Simpson's* indices show a relationship of decreasing diversity with distance in April 2006). In later surveys the patterns are complicated by the fact that both indices attempt to summarise richness and abundance, which appear to develop through different biological and ecological processes.

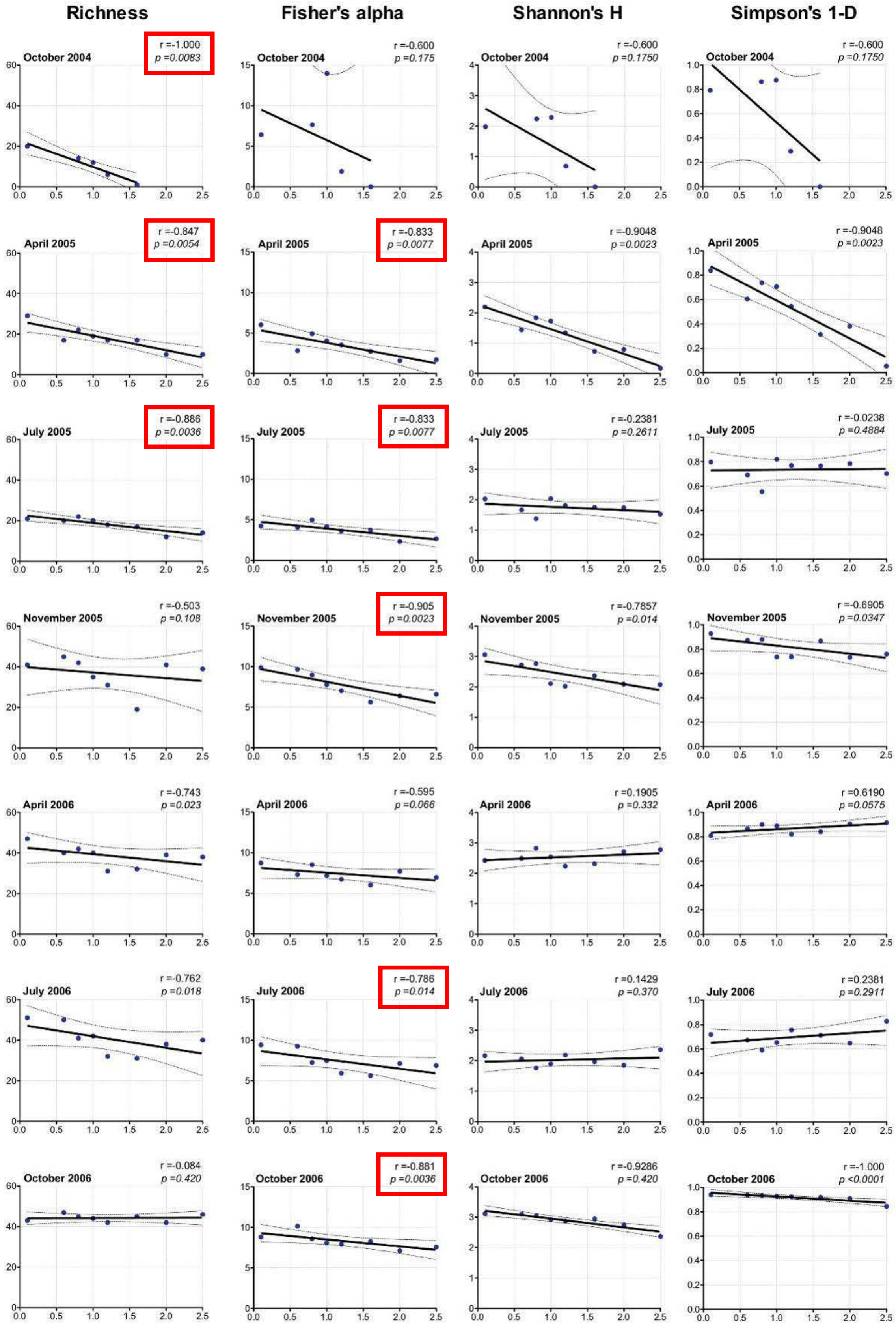


Figure 8.1: Graphs showing the patterns of colonisation along the length of the diversion as indicated by a range of diversity indices applied to species abundance data; Richness, Fisher's alpha, Shannon's H and Simpson's 1-D. Distances downstream from the top of the Nith realignment (x-axis) are plotted against index scores (y-axis) for each invertebrate sample collected.

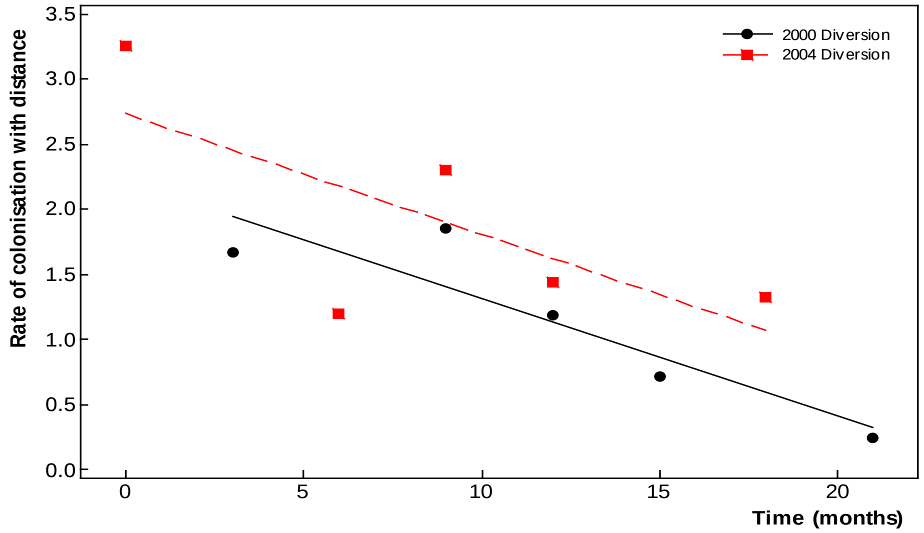


Figure 8.2. Graphing showing how the relationship between invertebrate diversity (richness) and distance from the upstream source changes through time. Data is presented for both the 2000 and 2004 realignments.

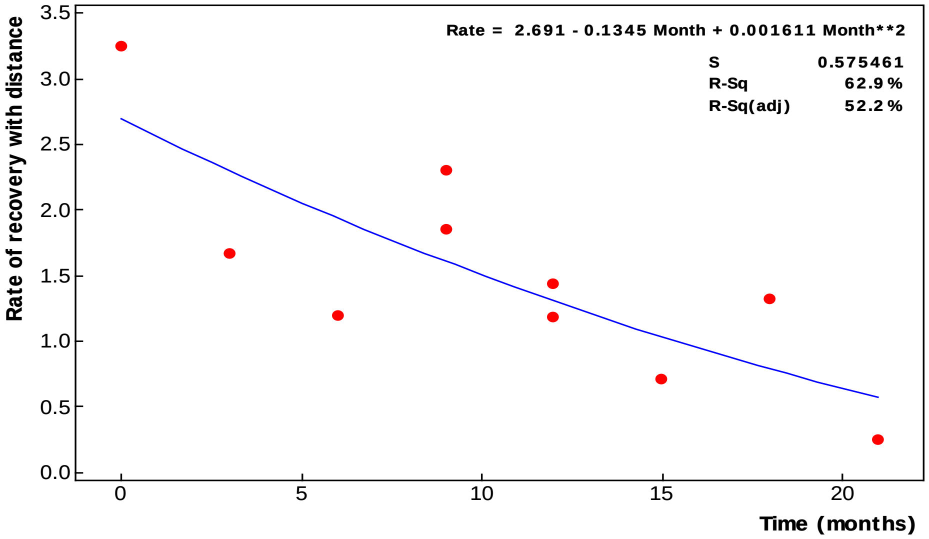


Figure 8.3. Graphing showing how the generalised relationship between invertebrate diversity (richness) and distance from the upstream source changes through time combining data from both the 2000 and 2004 realignments.

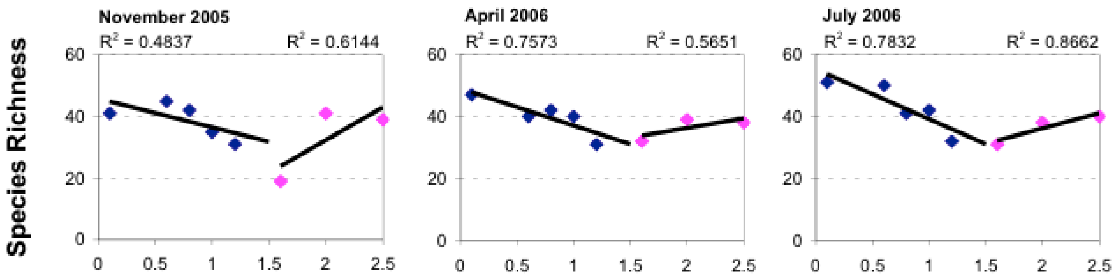


Figure 8.4 Species and Family richness along the length of the realignment showing the possible influence of colonisation from both upstream and downstream sources.

8.2 Abundance

Patterns of abundance reflect the patterns of alpha diversity already discussed (figure 8.5). Decreasing diversity (using square-root abundance scores) with increased distance from the upstream end is visible from October 2004 to July 2005. The use of square root abundance reduces problematic influences of life history patterns. For example individual species can reach very high densities for short periods immediately after hatching or cluster in high numbers within small patches of substrate. Using square root abundance values can give a better indication of the true abundance of taxa over the impression given by the total sample abundance, which gives a very limited spatial and temporal view.

Rarefaction was used to investigate whether the distance-abundance relationship was sufficient to explain patterns of decreasing richness with distance at early stages of development discussed in 8.1.1. Patterns shown in figure 8.6 demonstrate this not to be the case. Patterns of underlying alpha diversity do exist; downstream sites were more impoverished than upstream sites in all 2005 surveys, in terms of colonising species.

During the autumn surveys abundance is greater at the downstream end of the realignment. The pattern is consistent across years and samples but not across the community. While some taxa are present in considerably greater abundance than they are at upstream and reference sites, others remain below average levels (Appendix)

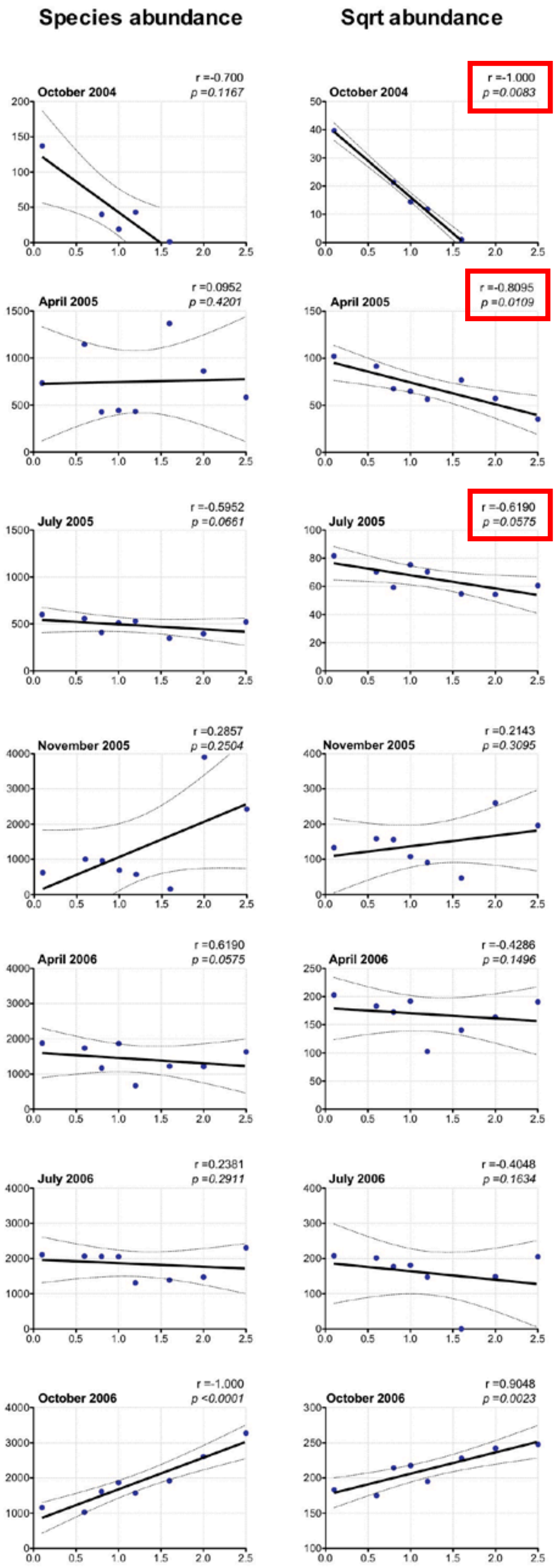


Figure 7.5 Showing patterns of colonisation as detected by abundance and square root abundance of sample counts.

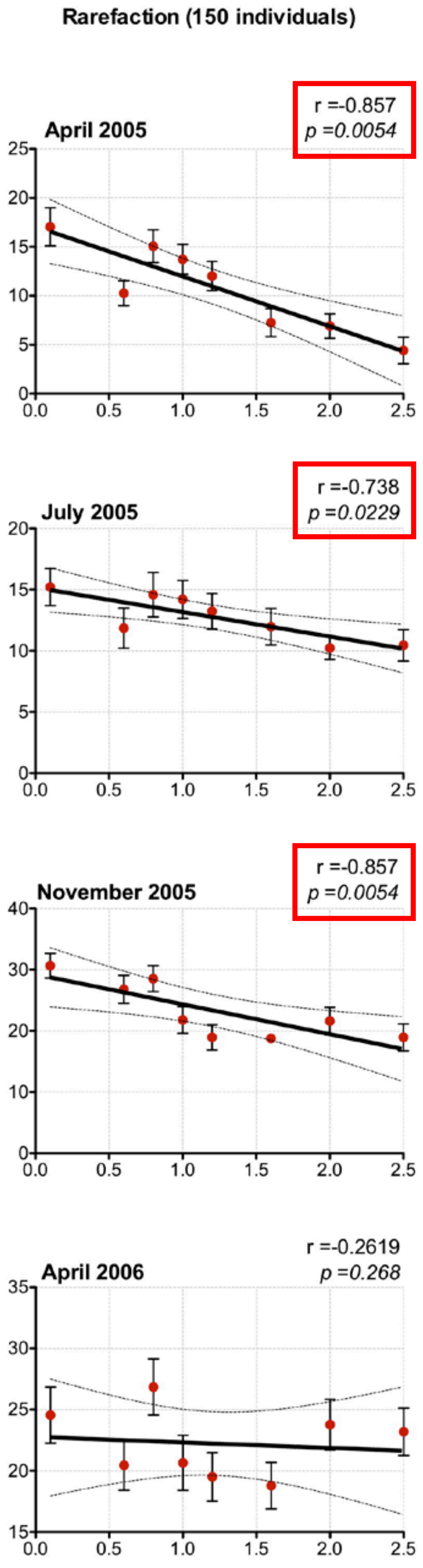


Figure 7.6 Showing patterns alpha diversity as detected by Rarefaction of species abundance data.

8.4 Taxonomic distinctness

Taxonomic diversity shows a clear decrease in diversity with downstream distance for April 05 only (r= -0.905, p= 0.002). No relationship is detected in taxonomic distinctness. Exactly the same patterns are detected when the same indices are applied to species abundance data grouped according to functional feeding type rather than genus and family groupings. A significant distance diversity relationship is only detected in April 2005 (r= -0.905, p= 0.002). The two sets of results seem to be closely related as a result of the functional feeding group research used has been focused at the family level. It is notable that there no obvious pattern in the index scores for all other surveys.

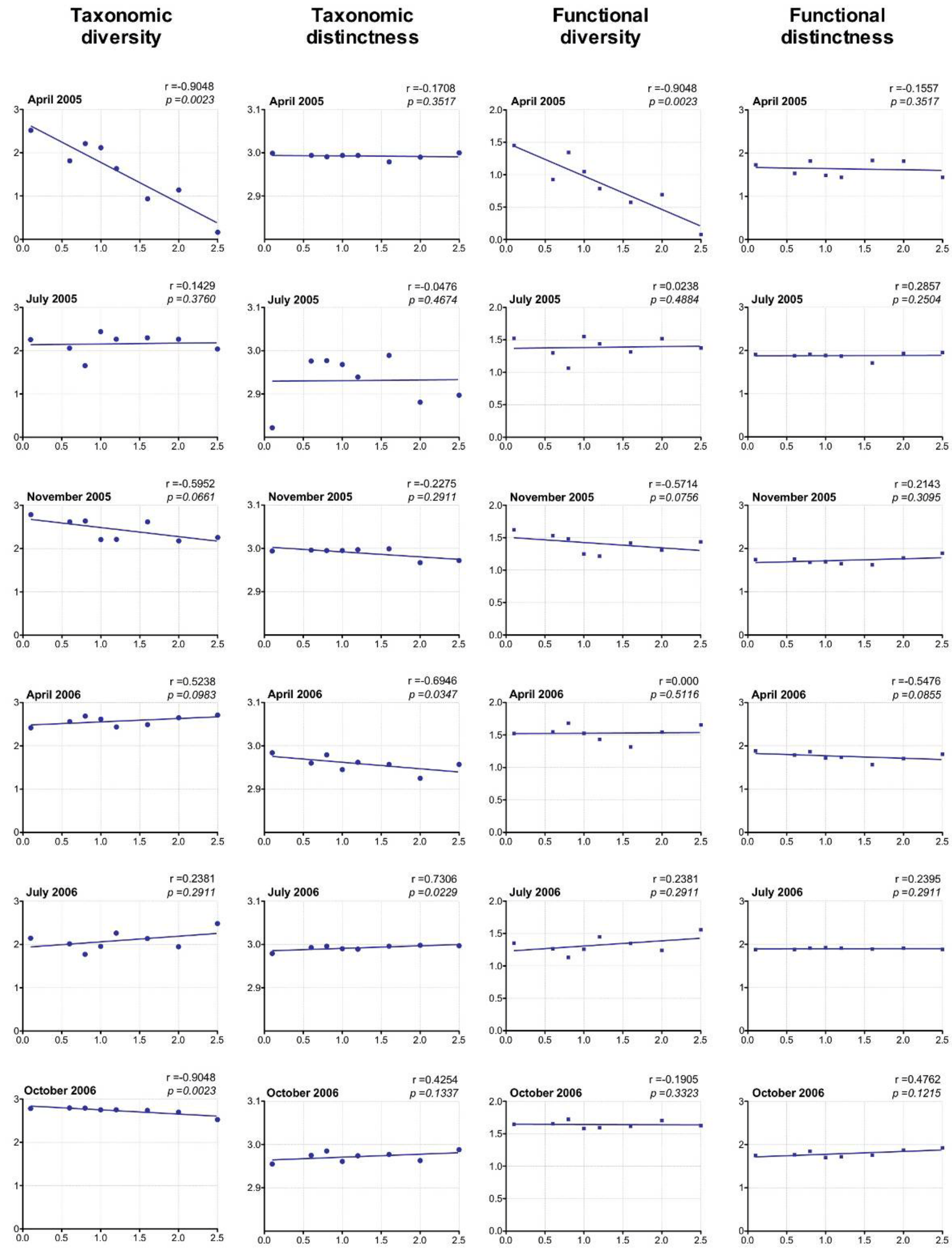


Figure 8.7: Graphs showing the diversity and distinctness of samples based on splitting data into taxonomic groups and into functional feeding groups.

8.5 Beta diversity

It is possible for many aspects of diversity to develop within the diversion and for communities to remain relatively uniform. This can be investigated using measures of beta diversity. Two aspects are of particular interest. First the levels of similarity to reference reaches as an indication of river health and the closeness to a desired ecological end-point and second, levels of dissimilarity within the realigned reaches as an indication of habitat variability – an important element of diversity important to the sustainability of river health.

Despite measuring beta diversity in different ways, three out of the four methods used to compare the realignment communities to the local invertebrate diversity (Ruggiero, Jaccard, Whittaker) show very similar patterns of development (figure 8.8) As a measure of dissimilarity, Whittaker appears to show a reverse pattern but all show similarity to decrease with distance downstream (an increase in beta diversity/dissimilarity) and increase with time (decrease in beta diversity/dissimilarity). These patterns are likely to be linked to patterns of underlying diversity that factor in the calculations of beta diversity.

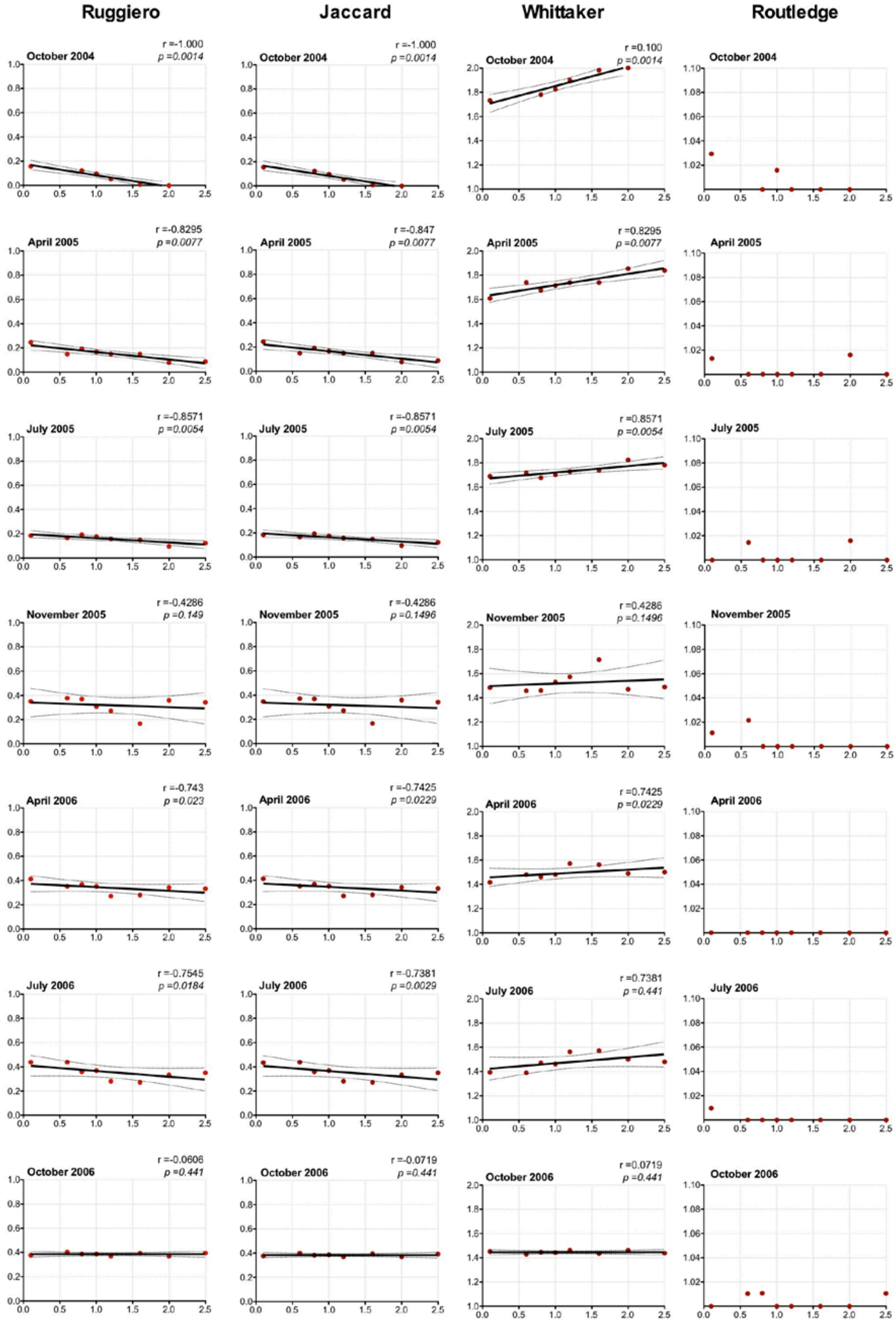


Figure 8.8: Graphs showing the comparison of development of local species diversity with the regional diversity as recorded at reference sites over a 4 year period. Comparisons have been made using a four beta diversity indices; Ruggiero, Jaccard, Whittaker and Routledge. Distances downstream from the top of the Nith realignment (x-axis) are plotted against index scores (y-axis) for each invertebrate sample.

The use of beta diversity to investigate changes in community structure through time presents a much clearer pattern about development. Figure 8.9 is an ordination of PCA analysis of the species abundance data collected from the realignment. Seasonal differences between samples are clearly evident with samples collected in July surveys standing out from those collected in the spring and autumn surveys. Also clear is the greater 'within survey' beta diversity at earlier stages of development, which then decreases each year. That is to say samples collected from the realignment are more different from each other at early stages than they are as the ecosystem develops. A decrease in between year beta diversity also through time is also visible with less change in community structure from Year 2 to Year 3 than was detected between Years 1 and 2. This is clearer when the seasonal variability is removed from the ordination (figure 8.10). Patterns can be contrasted with the Beta diversity recorded between control sites (figure 8.11), which show less change over time representing sampling variability and species turnover.

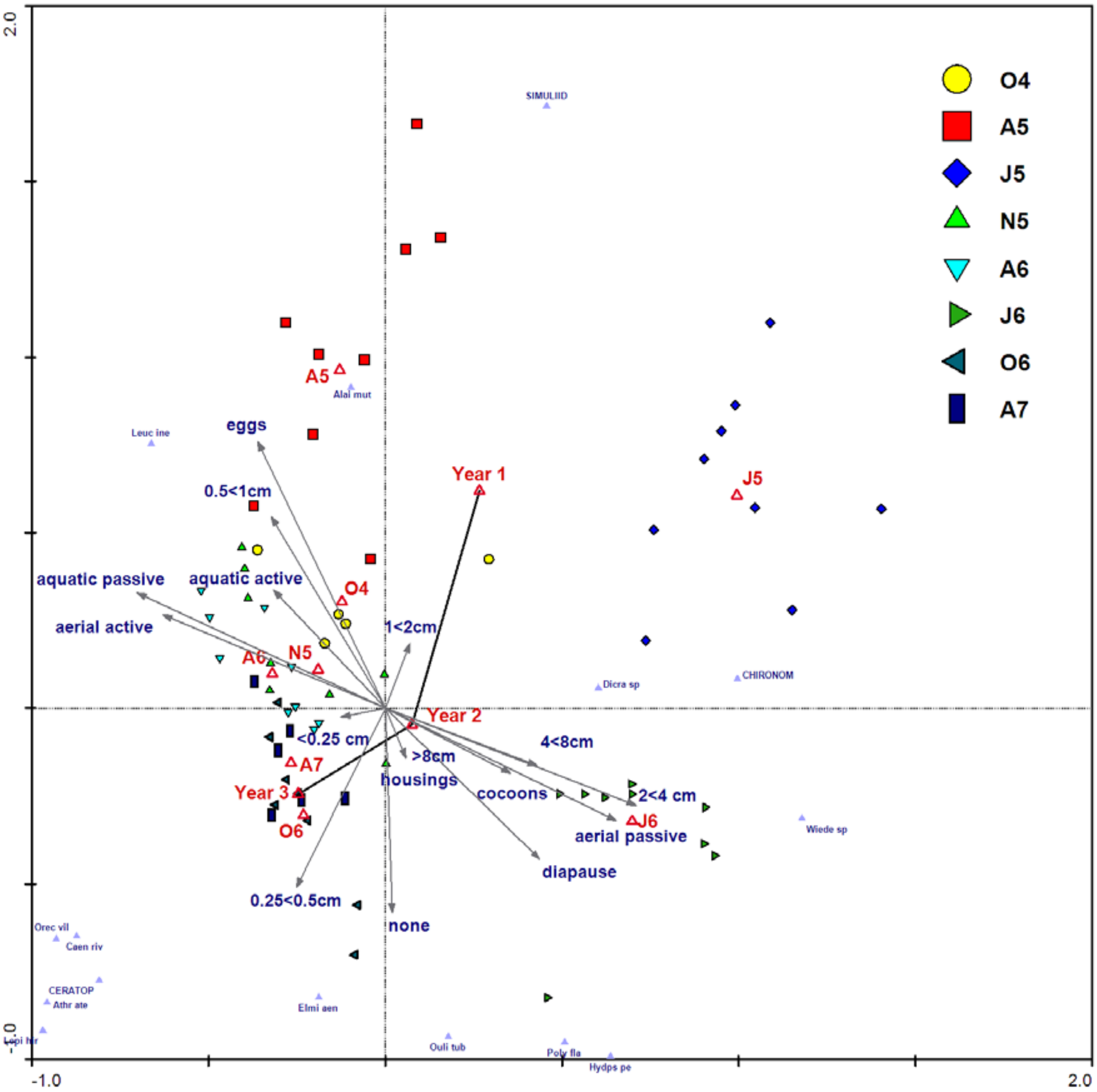


Figure 8.9. Unconstrained ordination of correspondence analysis showing the relationship of samples based on shared species abundance. Year on year change is indicated by arrows linking the centre of samples taken in each year. Community trait expression is superimposed over the ordination.

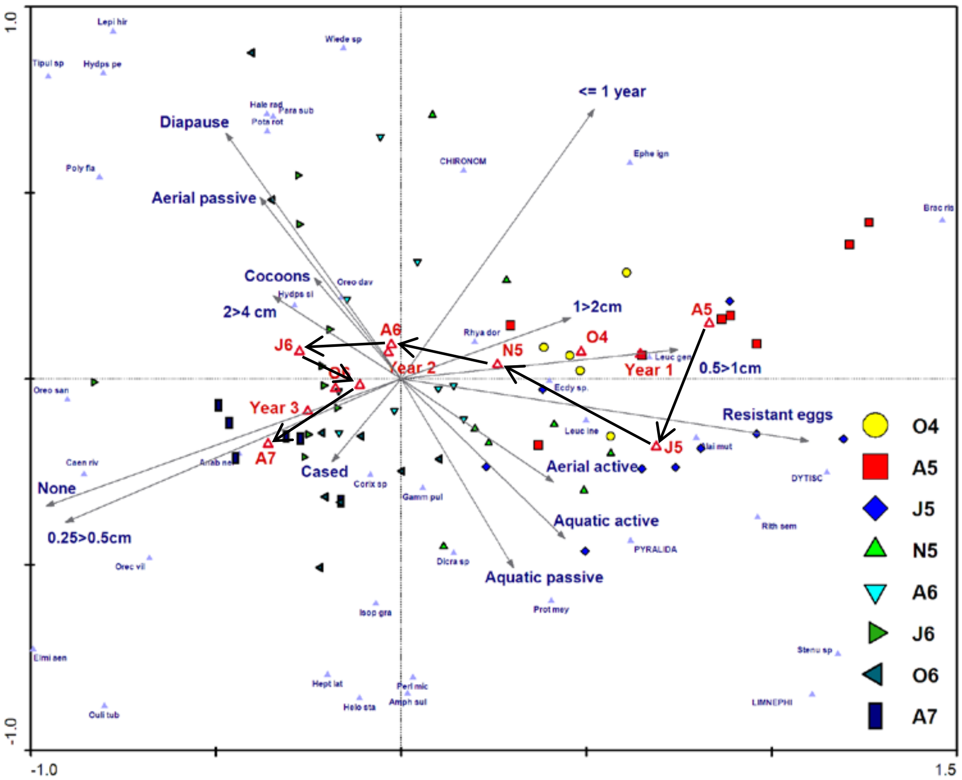


Figure 8.10 Unconstrained ordination of correspondence analysis with seasonal variance set as a covariable, showing the relationship between realignment samples based on shared species abundance. Between survey change is indicated by arrows linking the centre of samples taken in each season. Community trait expression is superimposed over the ordination.

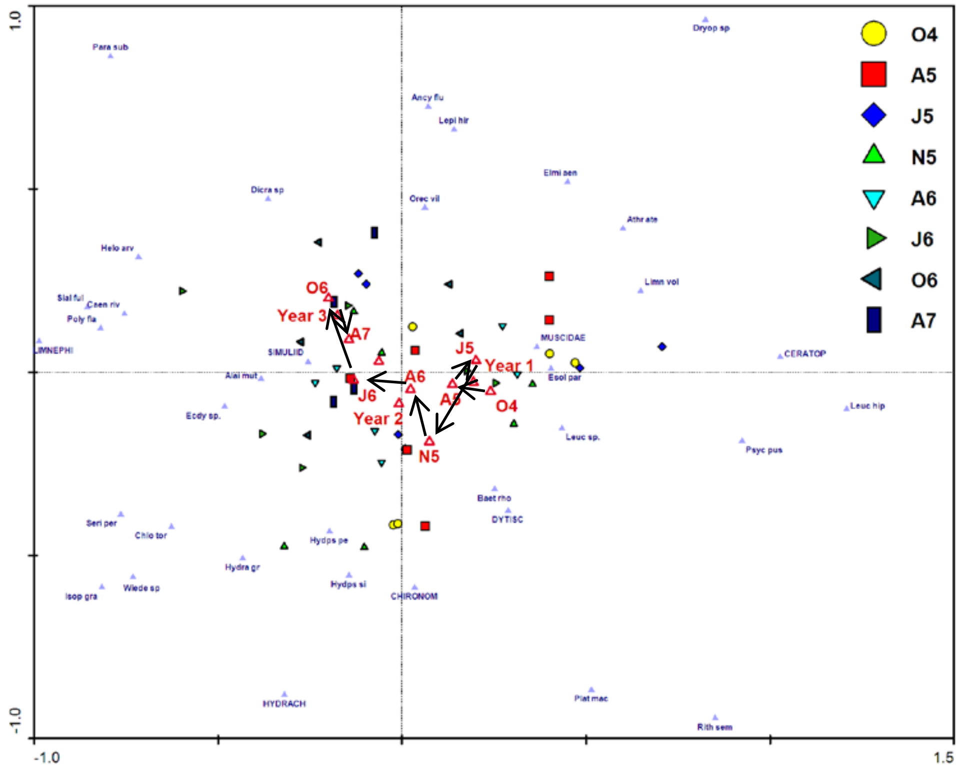


Figure 8.11 Unconstrained ordination of correspondence analysis with seasonal variance set as a covariable, showing the relationship between reference samples based on shared species abundance. Between survey change is indicated by arrows linking the centre of samples taken in each season.

8.6 Community expression of species traits 2006

Analysis using diversity indices simplify the large data-set into more easily interpretable values that can indicate patterns and trends but in doing so lose much of the ecological information linked to specific groups of taxa and guilds. The use of biotic indices goes some way to addressing this problem.

The ordination diagrams below present principal components analysis of community trait expression. Sites were scored according to the traits of the species present in the survey samples. Figure xx shows the relatedness of sites based on the community expression of traits relating to the presence of resistant stages in the organisms' life cycles. Both the likelihood of a site being a reference site and having a community with a week expression of any form of resistance are correlated with the first axis whereas many of the sites surveyed along the realignment show strong community expression for a resistant form at the egg stage of the life cycle.

Figure xx show the relatedness of sites based on the community expression of traits relating to dispersal modes. The reference sites have a stronger expression of active aerial dispersers suggesting species that actively make use of aerial pathways have a lower prevalence in the realignment than they do in the reference reaches.

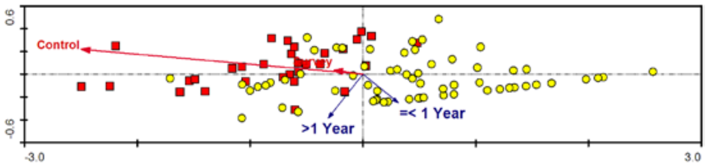


Figure 8.12. Ordination of PCA showing the relationships between sampling sites (scaling focused on inter-sample distance) based on the community trait expression relating to the length of the life cycle. Data is from 2 years of invertebrate surveys of the 2000 realignment. Data was untransformed and standardised. Eigen values for successive PCA axes are shown. The first two axes account for 100% of the variation.

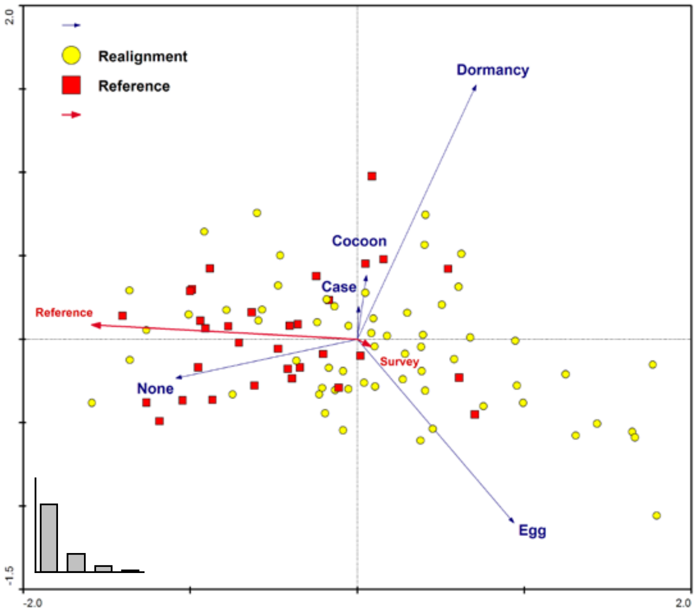


Figure 8.13. Ordination of PCA showing the relationships between sampling sites (scaling focused on inter-sample distance) based on the community trait expression relating to resistant life cycle stages. Data is from 2 years of invertebrate surveys of the 2000 realignment. Data was untransformed and standardised. Eigen values for successive PCA axes are shown. The first two axes account for 91% of the variation.

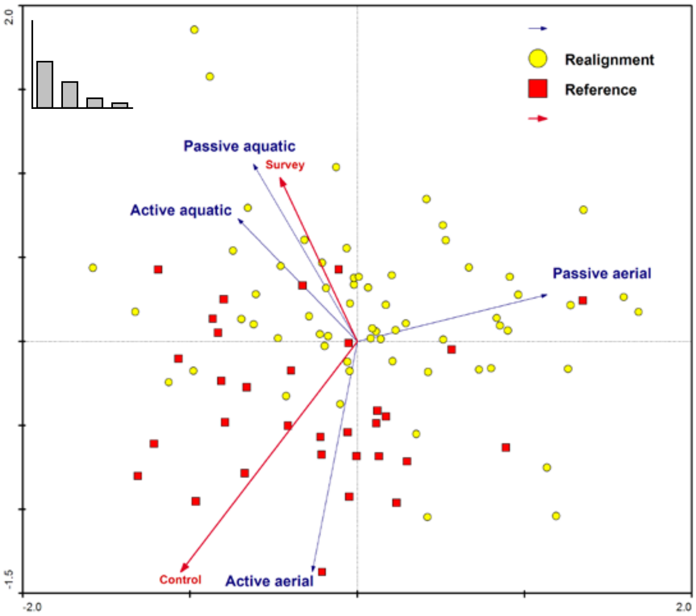


Figure 8.14. Ordination of PCA showing the relationships between sampling sites (scaling focused on inter-sample distance) based on the community trait expression relating to general modes of dispersal. Data is from 2 years of invertebrate surveys of the 2000 realignment. Data was untransformed and standardised. Eigen values for successive PCA axes are shown. The first two axes account for 83% of the variation.

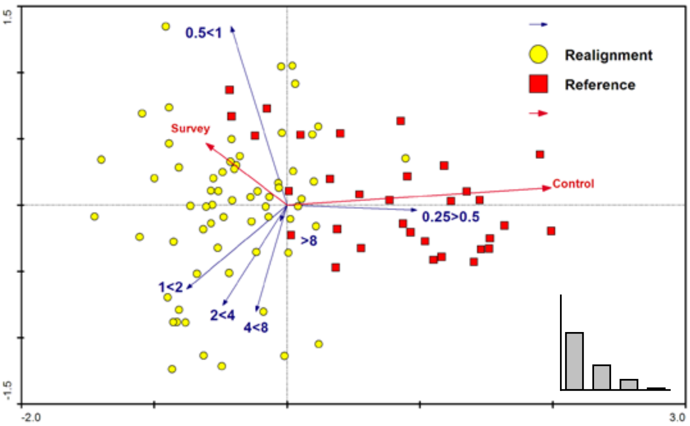


Figure 8.15. Ordination of PCA showing the relationships between sampling sites (scaling focused on inter-sample distance) based on the community trait expression relating to maximum size of individuals. Data is from 2 years of invertebrate surveys of the 2000 realignment. Data was untransformed and standardised. Eigen values for successive PCA axes are shown. The first two axes account for 87% of the variation.

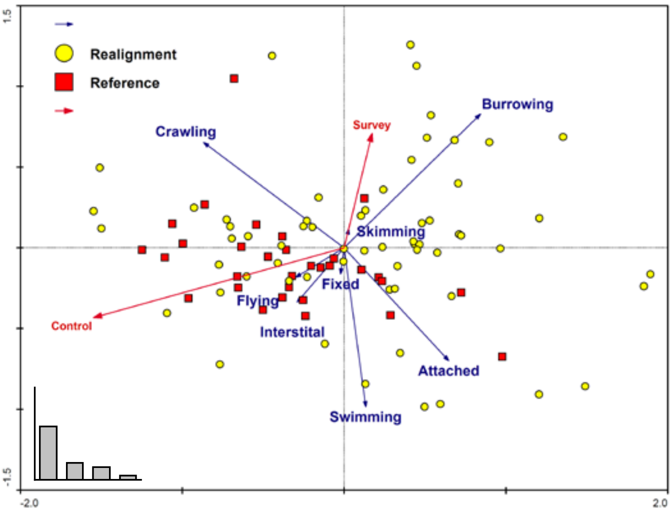


Figure 8.16. Ordination of PCA showing the relationships between sampling sites (scaling focused on inter-sample distance) based on the community trait expression relating to general lifestyle. Data is from 2 years of invertebrate surveys of the 2000 realignment. Data was untransformed and standardised. Eigen values for successive PCA axes are shown. The first two axes account for 76% of the variation.

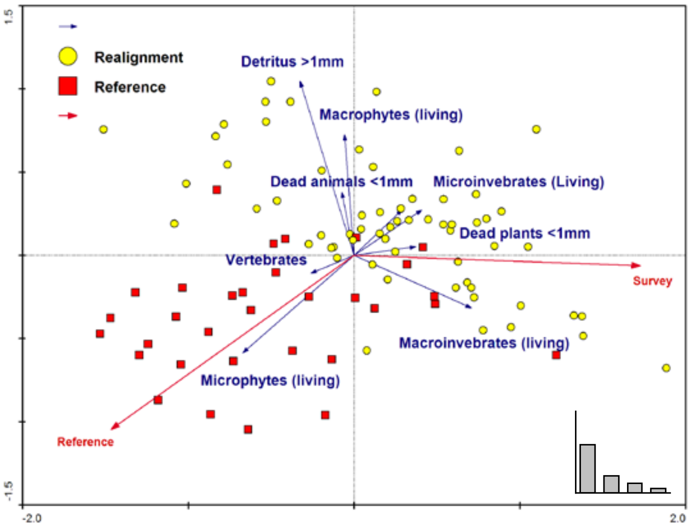


Figure 8.17. Ordination of PCA showing the relationships between sampling sites (scaling focused on inter-sample distance) based on the community trait expression relating to the preferred food type. Data is from 2 years of invertebrate surveys of the 2000 realignment. Data was untransformed and standardised. Eigen values for successive PCA axes are shown. The first two axes account for 80% of the variation.

8.7 Community expression of species traits 2005

The following figures show the differences in community trait expression between the colonists of the 2005 realignment and the reference reaches. Of the six traits proposed to confer advantage to colonists, four were differentially expressed in samples from reference and realignment communities.

Figure 8.17 indicates a stronger presence of species with a short lifecycle amongst colonising communities relative to the reference reaches at early stages of development. The difference was found to be significant throughout the period of study with the exception of November 2005 (Mann-Whitney U test, P<0.05). Figure 8.18 indicates a stronger presence of species with eggs that are robust to desiccation or impact amongst colonising communities relative to the reference communities. The difference can be seen to gradually reduce over time but remains significant until April 2006 (Mann-Whitney U test, P<0.05). Figure 8.20 indicates a weaker presence of species with a smaller maximal size (<0.5cm) amongst colonising communities relative to the reference communities at early stages of development. The difference is observed to reduce with time but remains significant until July 2006 (Mann-Whitney U test, P<0.05). Figure 8.22 indicates a weaker presence of species with a feeding preference for microphytes amongst colonising communities relative to the reference communities, at early stages of development. The difference is significant for April and July of 2005 (Mann-Whitney U test, P<0.05). However, variance at this time is not comparable.

Table 8.1. Summary of statistically significant differences between community expression of traits of possible advantage or disadvantage to colonists

Trait **(Modality)**	**April 2005**	**July 2005**	**November 2005**	**April 2006**	**July 2006**	**October 2006**
Life cycle (Less than 1 year)	**W=10 (p=0.004)**	**W=10 (p=0.004)**	**NS**	**W=14 (p=0.025)**	**W=11 (p=0.007)**	**W=14 (p=0.025)**
Resistant life cycle stage (Egg)	**W=10 (p=0.004)**	**W=14 (p=0.025)**	**W=12 (p=0.011)**	**W=11 (p=0.007)**	**NS**	**NS**
Mode of dispersal (Active aerial)	**NS**	**W=41 (p=0.006)**	**NS**	**NS**	**W=40 (p=0.011)**	**NS**
Maximal size (Less than 0.5cm)	**W=42 (p=0.004)**	**W=42 (p=0.004)**	**W=40 (p=0.011)**	**W=42 (p=0.004)**	**W=42 (p=0.004)**	**NS**
General life habit (Burrowing)	**NS**	**NS**	**NS**	**NS**	**W=68 (p=0.004)**	**NS**
Preferred food type (Microphytes)	**W=42 (p=0.004)**	**W=42 (p=0.004)**	**NS**	**NS**	**NS**	**NS**

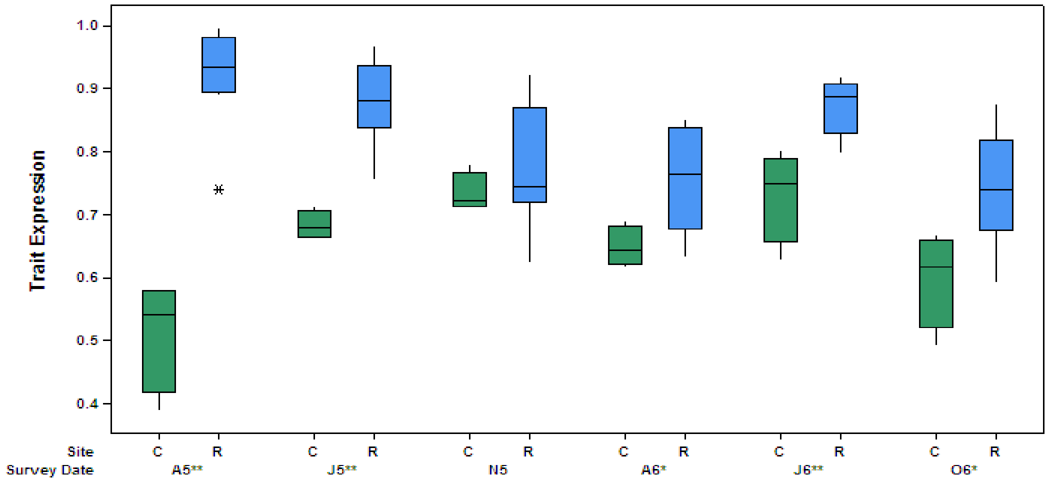


Figure 8.17. Box and whisker plots showing the differences, between the reference and realignment samples, in community expression of the *less than one year* modality in the *length of lifecycle* trait for each sampling season. Significant differences at each season are indicated by ** (P<0.01) * (P<0.05) Control Realignment

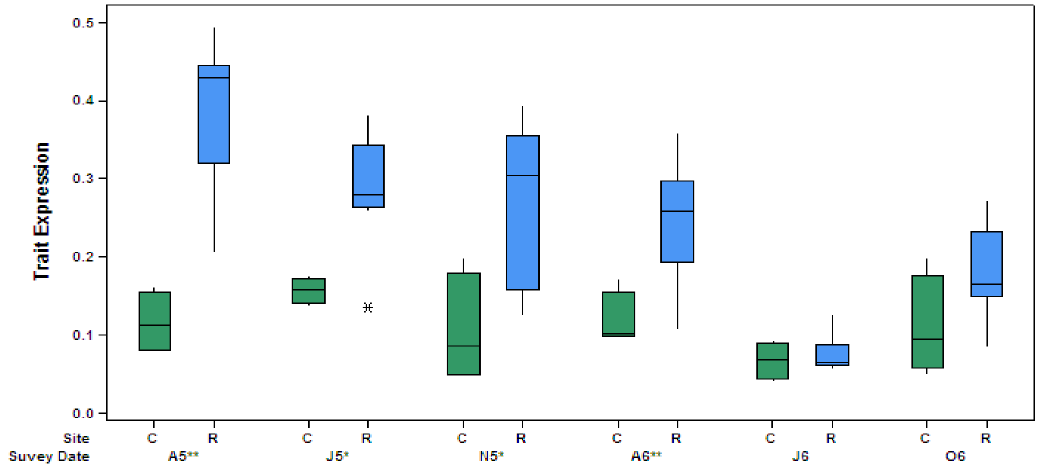


Figure 8.18. Box and whisker plots showing the differences, between the reference and realignment samples, in community expression of the *resistant egg* modality in the *resistant lifecycle stage* trait for each sampling season. Significant differences at each season are indicated by ** (P<0.01) * (P<0.05) Control Realignment

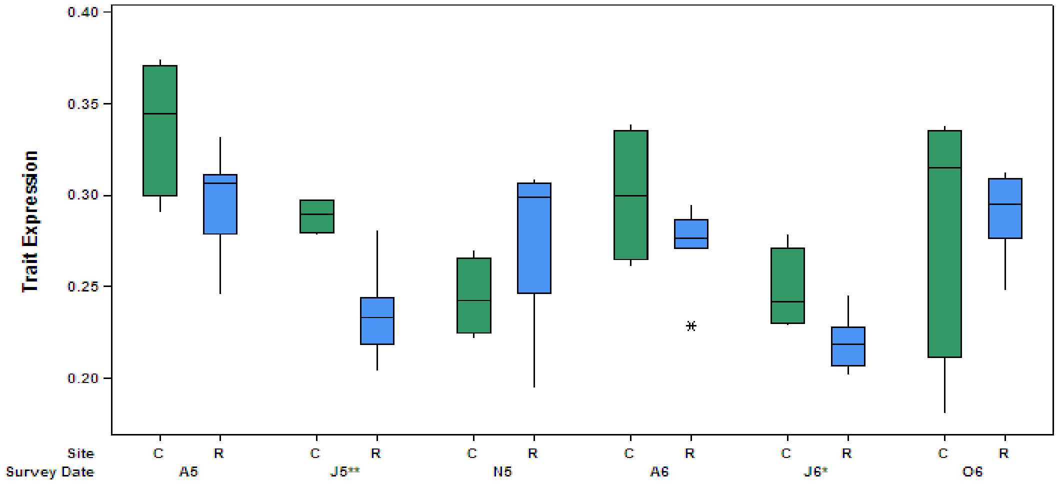


Figure 8.19. Box and whisker plots showing the differences, between the reference and realignment samples, in community expression of the *active aerial* modality in the *mode of dispersal* trait for each sampling season. Significant differences at each season are indicated by ** (P<0.01) * (P<0.05) Control Realignment

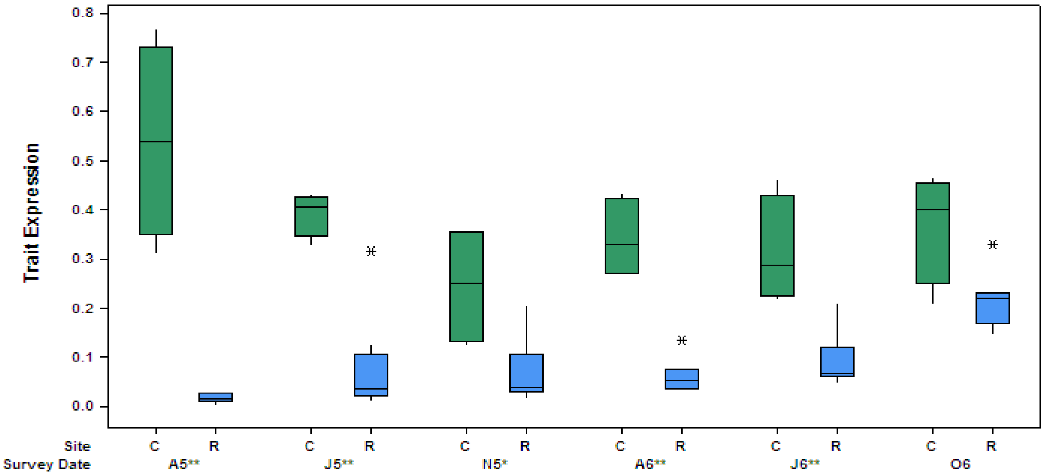


Figure 8.20. Box and whisker plots showing the differences, between the reference and realignment samples, in community expression of the *<0.5cm* modality in the *maximal size* trait for each sampling season. Significant differences at each season are indicated by ** (P<0.01) * (P<0.05) Control Realignment

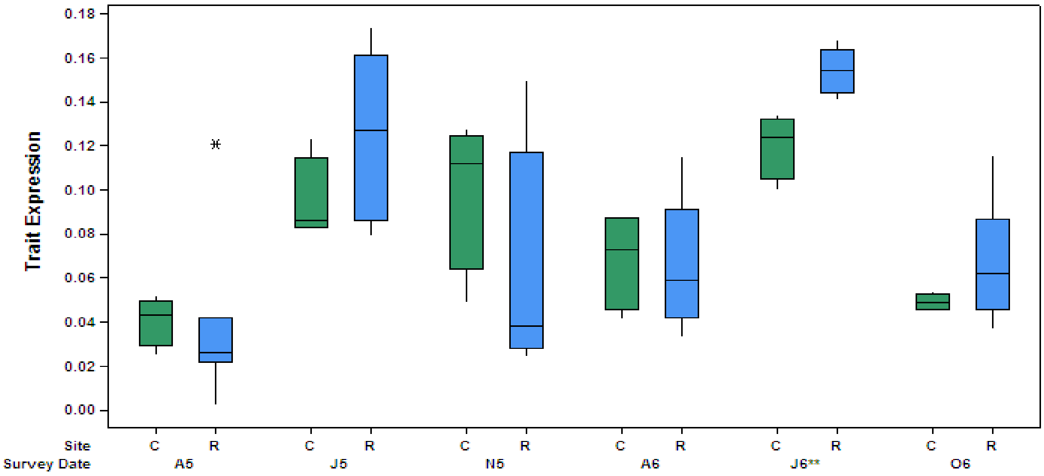


Figure 8.21. Box and whisker plots showing the differences, between the reference and realignment samples, in community expression of the *burrowing* modality in the *general habit* trait for each sampling season. Significant differences at each season are indicated by ** (P<0.01) * (P<0.05) Control Realignment

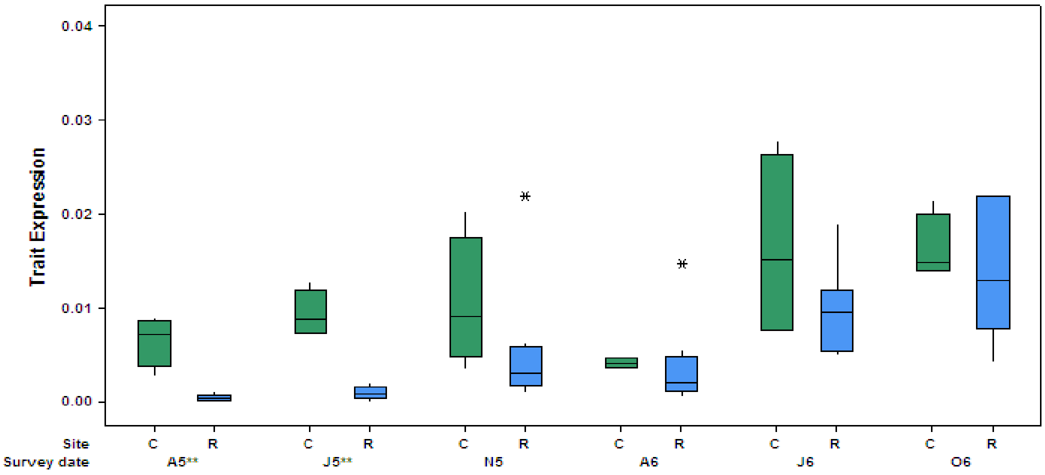


Figure 8.22. Box and whisker plots showing the differences, between the reference and realignment samples, in community expression of the *microphyte* modality in the *food type* trait for each sampling season. Significant differences at each season are indicated by ** (P<0.01) * (P<0.05) Control Realignment

8.8 Community structure

Species abundance models have been developed in response to the observation that species are rarely equally common in natural ecosystems. Species sample abundances have been used here to investigate the development of a structure to the colonising community. Samples from the reference sites fit the log normal species abundance model as shown in figures 8.23 and 8.24. A realignment sites the community at early stages of development are better described by a geometric series model. However at site N1 there is a clear development of community structure to the log normal model between July and November 2005 (figure 8.25). A similar change is shown at the downstream end of the diversion (site N7, figure 8.27). In the middle reaches this change is more gradual, not maturing to the log normal structure until October 2006.

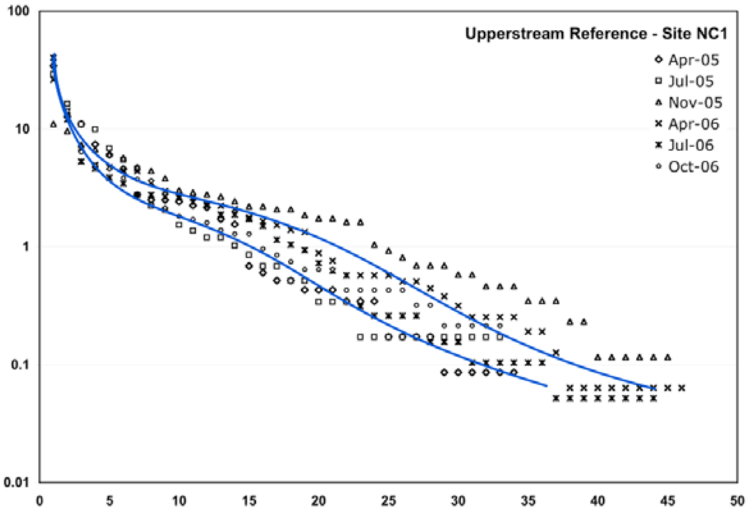


Figure 8.23. Rank abundance plot of species rank (x-axis) plotted against percentage species abundance (y-axis). Graphs show community structure through time at site NC1 in the upstream reference reaches

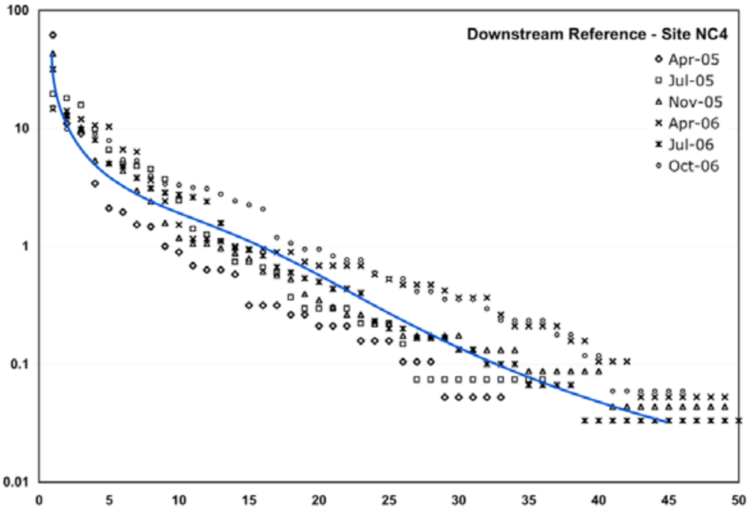


Figure 8.24. Rank abundance plot of species rank (x-axis) plotted against percentage species abundance (y-axis. Graphs show community structure through time at site NC4 in downstream reference reaches

Figure 8.25. Rank abundance plot of species rank (x-axis) plotted against percentage species abundance (y axis). Graphs show community structure through time at site N1 in the upper reaches of the realignment

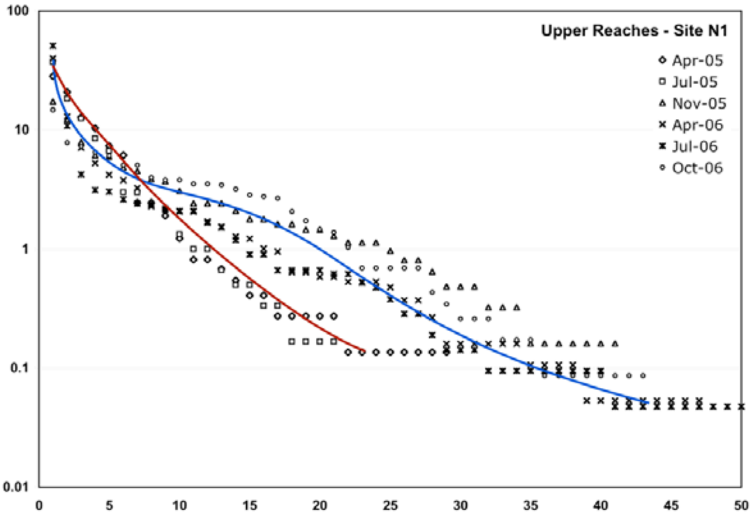


Figure 8.26. Rank abundance plot of species rank (x-axis) plotted against percentage species abundance (y-axis). Graphs show community structure through time at site N5 in the middle reaches of the realignment

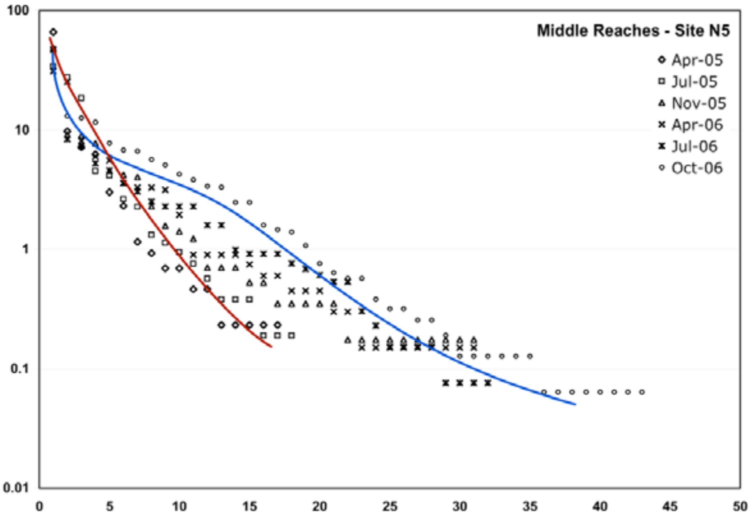
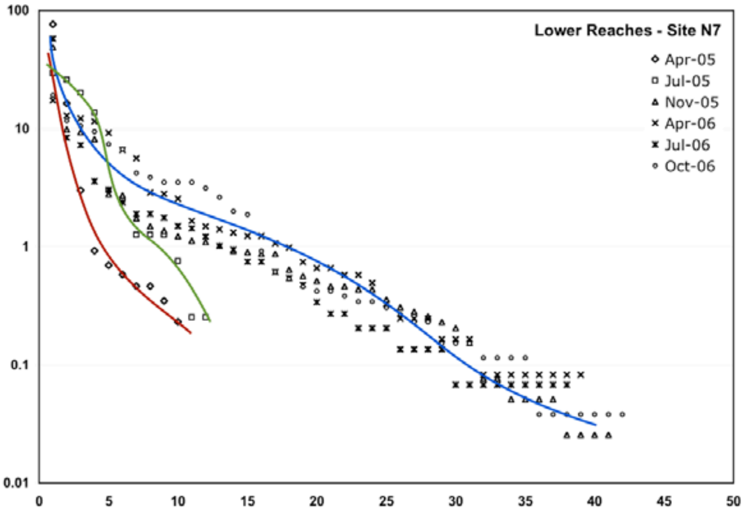


Figure 8.27. Rank abundance plot of species rank (x-axis) plotted against percentage species abundance (y-axis). Graphs show community structure through time at site N7 in the lower reaches of the realignment



8.9 Habitat development

The influence of a lack moss within the channel was investigated using an experimental set-up of alternating patches of mossy and non-mossy boulders. The affect on the diversity of colonising biota is shown in figures 8.28 and 8.29. There was a noticible difference in patch species richness between the treatments 6 weeks after installation although not found to be significant (Mann-Whitney U test, P>0.05). This difference had grown and was significant after a further 12 weeks (18 weeks; Mann-Whitney U test, P=0.02). An even clearer difference between treatments was observable when total patch abundance was considered, statistically significant at both 6 weeks (Mann-Whitney U test, P=0.03) and 18 weeks (Mann-Whitney U test, P<0.01) after installation. Severe disturbance of the mossy patches after 18 weeks meant no further surveys could be meaningfully carried out.

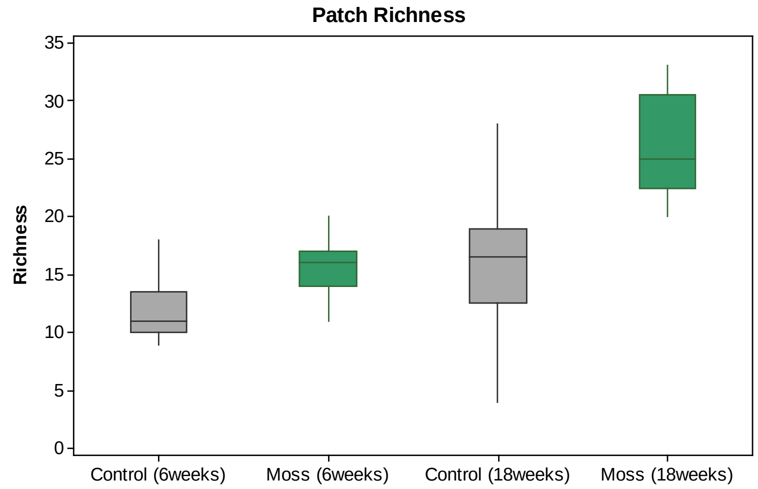


Figure 8.28. Box plots comparing the species richness of mossy and non-mossy patches 6 weeks and 18 weeks following installation in the upper reaches of the Nith realignment.

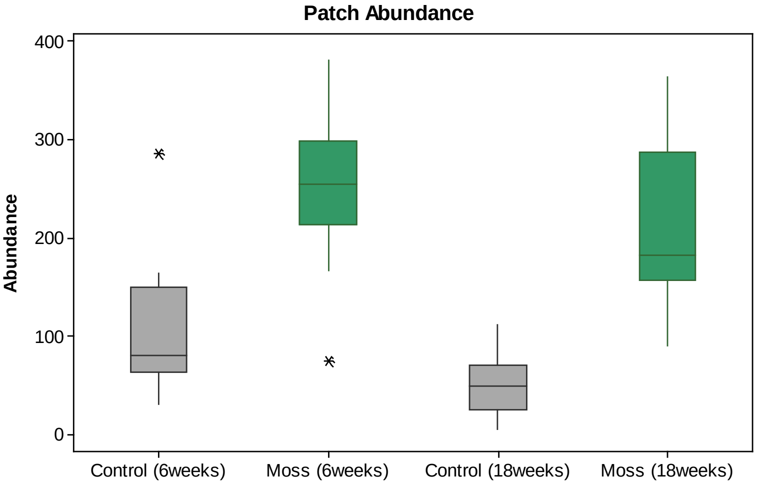


Figure 8.29. Box plots comparing the total species abundance of mossy and non-mossy patches 6 weeks and 18 weeks following installation in the upper reaches of the Nith realignment.

The influence of the channel planform investigated using Surber sample surveys of the inside and outside edge of two meanders. The affect of the different environmental conditions on colonising biota is shown in figure 8.30. Comparisons of the inside and outside edges of the channel revealed significant differences in community richness and abundance around meander 12 (Richness; Mann-Whitney U test P<0.01. Abundance; Mann-Whitney U test, P<0.01). However, no difference was detected in the invertebrate community richness or abundance between the inside and outside edges of meander 9.

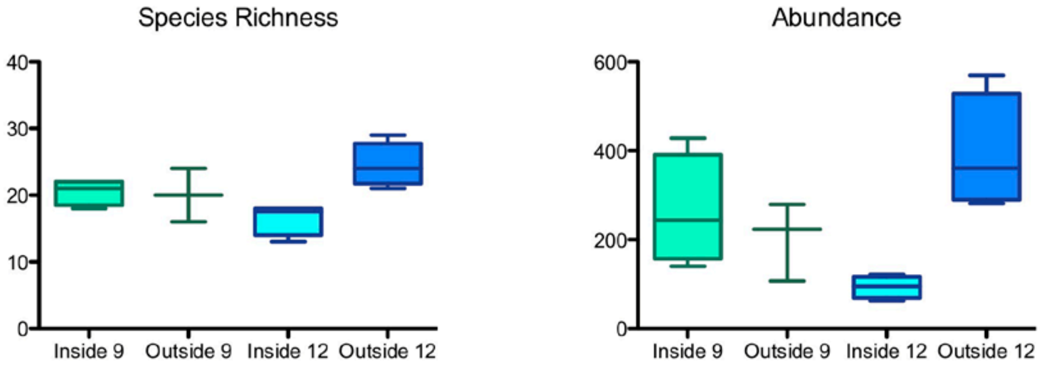


Figure 8.30. Box plots comparing the benthic invertebrate community richness and abundance around the inside and outside of Meanders 9 and 12 in the middle reaches of the Nith realignment

CHAPTER 9

RESULTS III: VEGETATION

9.1 Vegetation development

The riparian zone is clearly a vital element of a healthy stream ecosystem. Development of structure and diversity will therefore represent an important element of ecosystem recovery. Initial investigations of the data used canonical correspondence analysis to illustrate patterns within the data and provided an intial impression of whether or not hypotheses were likely to be supported by the data collected.

Figures 9.1 to 9.4 show detrended correspondence analysis (unimodal) for each survey year. Indirect gradient analysis was used for the ordination diagrams and the ordination was detrended by segments to remove the arc pattern common in the ordination of ecological data. The method results in the best distribution of sites through the ordination space based on relative species composition, allowing the identification of patterns in sites that relate to similarities in richness and percentage cover of species. Species are clustered around sites where they are typically found. To prevent relatively dominant species, principally Trifolium spp. and in early years – Lolium perenne, from masking the colonisation patterns of less successful species, percentage cover values were square root transformed. Further to this, rare species were not down weighted in the analysis because they made up the majority of the species richness and were considered important in the project success. Furthermore, although absence of a rare species may be linked to stochastic factors, their presence can still be taken as a positive indication of colonisation potential or habitat condition. Environmental variables are included in the diagrams as red arrows indicating the direction and strength of a gradient in value across the diagram but were not used to produce the ordination. Likewise, the changes in Ellenberg scores across the ordination indicate the differences in habitat preferences of the communities within each quadrat.

Table 9.1. Description of the explanatory variables used in the ordination analysis

Explanatory variable	*Description*
Angle	*The angle between downstream slope and the quadrat from the centre of the meander radius*
Aspect	*Whether the quarat lies on the inside, straight or out side of a meander*
Chainage	*Distance downstream from the top of the diversion*
Reach	*The section of the diversion in which the quadrat is located.*
Remoteness	*Distance from any existing natural river channel*

Figure 9.1 shows the relationship between quadrats as detected by the 2005 vegetation survey. Vegetation richness and cover was relatively low during this period. A total of 28 species were recorded in 75 quadrats on 25 transects along the length of the realignment and of these species 7 were found just once and only 13 occurred in more than 10% of the quadrats. Of the 28 only 4 were included in a prescribed seed mix for the restoration of vegetation across the site. These were *Ranunculus repens, Cardamine pratensis Rumex acetosa* and *Persicaria maculosa*. The most frequent species *Lolium perennne* also had the highest total cover, followed by Trifolium repens, next in both frequency and total cover.

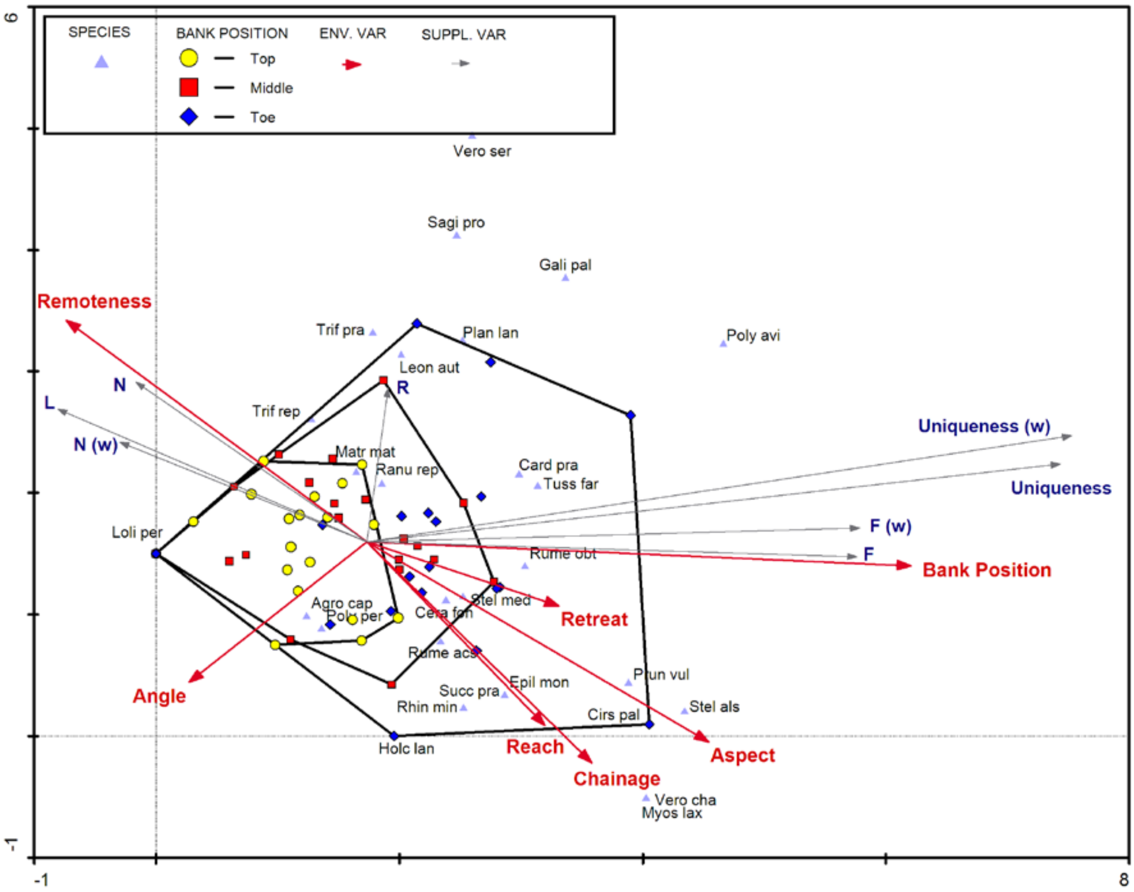


Figure 9.1: Canonical correspondence analysis (detrended) relating 2005 VEGETATION DATA to explanatory variables; Bank position (top middle or toe of the bank), Aspect (inside or outside of meander), Bank retreat (degree of bank erosion), Remoteness (distance from natural channel reaches), Chainage (distance from the top of the realignment) and River section shown as red arrows. Ellenburg scores for moisture (F), reaction (R), Light (L) and Nitrogen (N) preference are shown as blue arrows. Scores weighted by species cover are indicated by (w). Envelopes have been drawn around quadrats at the same bank position, which have been differentiated using different symbols (see key).

The large envelope encompassing the quadrats positioned at the bank toe is larger that the bank top suggesting a greater variety of habitat condition my be developing compared to a more uniform bank top, although at this stage the majority of sites are still relatively closely clustered. Strong gradients in geocraphic variables can be seen to relate to community composistion. The most evident is bank position further highlighted by low level of overlap between the area occupied by bank to an bank toe sites on the ordination diagram. Whether or not tansects were positioned on the inside of a meander, on a straight section of channel or on the outside of a meander (Apsect) also appears to be a factor in community composition and may relate to the deposition of seeds. A tendency for species colonising the toe of the bank to have a tolerance of moist conditions is indicated by the parallel gradients of Ellenburg’s F and Bank position. The strongest gradient through the ordination is a score of uniqueness indicating the relative rarity of the species found in the quadrats. The distance from potential sources of colonists represented in the ordination by Chainage, Reach and Remoteness also has an evident relationship with species composition and cover. Significance testing of the explanatory power of variables for composition and cover of quadrats was performed by the Monte-Carlo test and are shown in table 9.2

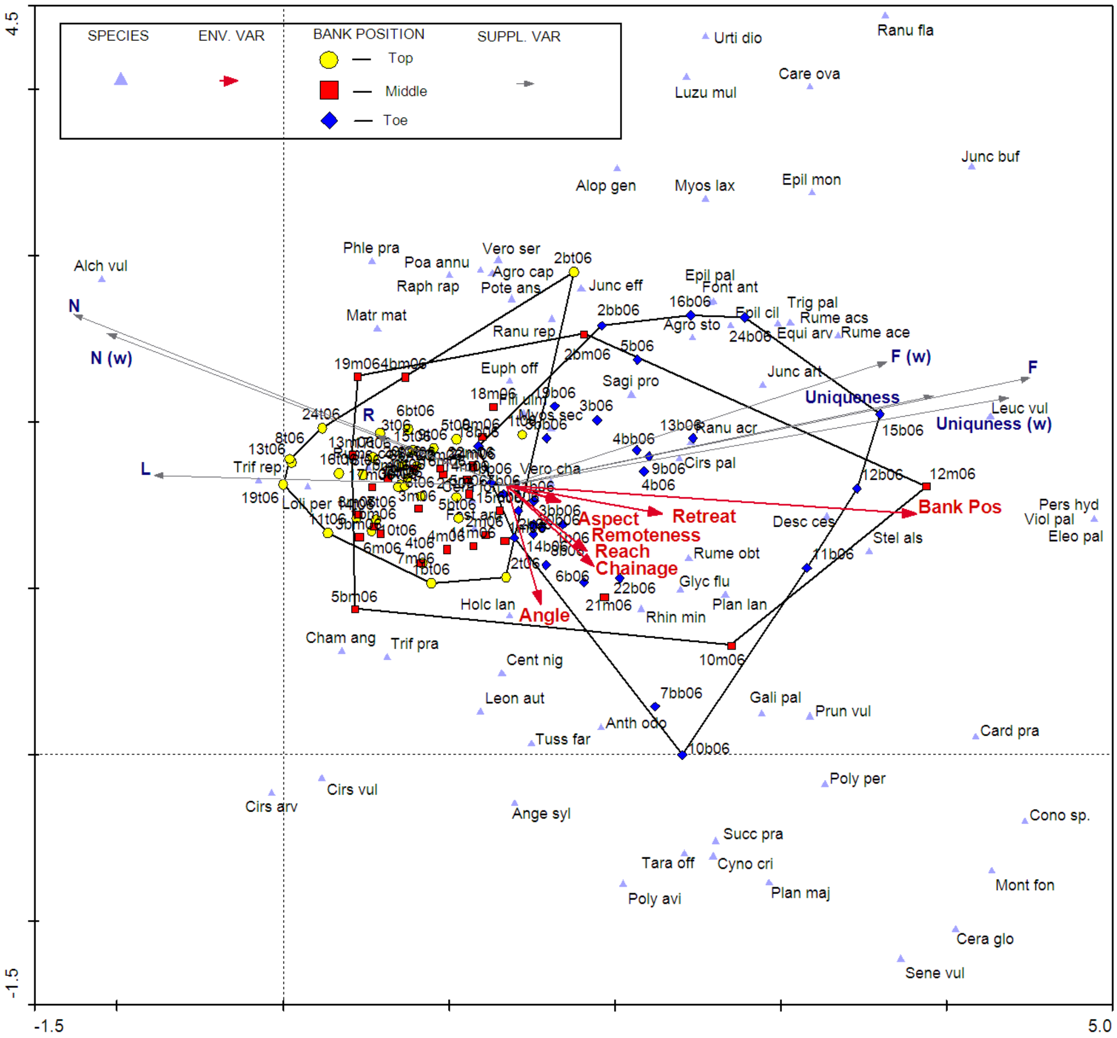


Figure 9.2: Canonical correspondence analysis (detrended) relating 2006 VEGETATION DATA to explanatory variables Bank position, Bank retreat, Remoteness, Chainage and River section shown as red arrows. Ellenburg scores for moisture (F), reaction (R), Light (L) abd Nitrogen (N) preference are shown as blue arrows. Scores weighted by species cover are indicated by (w). Envelopes have been drawn around quadrats at the same pabnk position whic are differentiated using different symbols (see key).

Considerably greater species richness was recorder along the realignment in 2006 compared to 2005. A total of 76 species were recorded of which 25 were found in greater than 10% of quadrats. The most frequently occuring five species, occurring in over 50% of quadrats were all competitive ruderal species. In order of dominance these were *T.repens, L.perene, H.lanatus, Agrostris stolonifera and R repens.* However the majority of species although infrequent were ‘mire’ species (figure 9.5). The ordination indicates that bank position is still the most important factor related to quadrat composition (P<0.01) and that factors relating to the lenght of colonisation pathways. The overlap in species composition between bank toe and other bank positions is further reduced with the community at the toe of the bank continues to be represented by species with a preference for moisture as indicated by the Ellenburg scores for the quadrats. Species are distributed relatively evenly with little clustering around particular groups of sites.

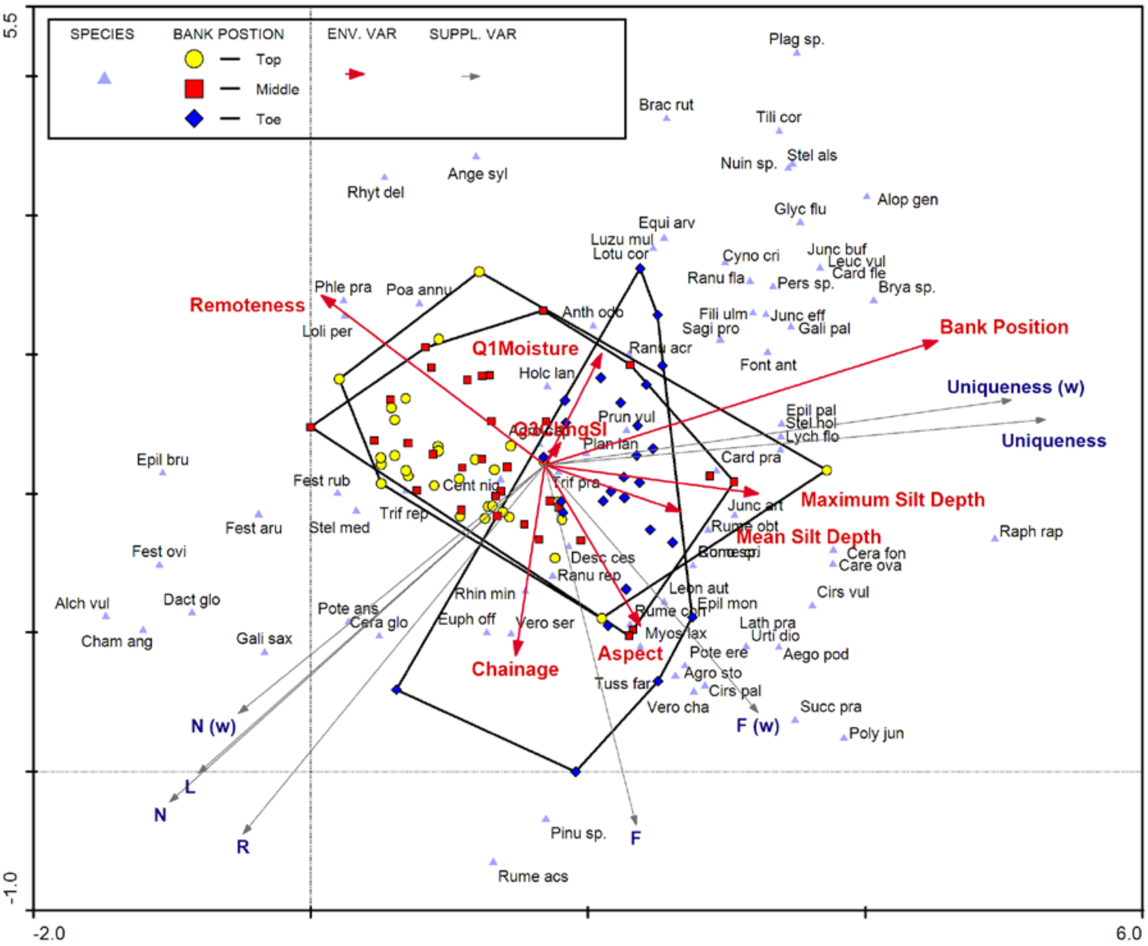


Figure 9.3: Canonical correspondence analysis (detrended) relating 2007 VEGETATION DATA to explanatory variables Bank position, Bank retreat, Remoteness, Chainage and River section shown as red arrows. Ellenburg scores for moisture (F), reaction (R), Light (L) abd Nitrogen (N) preference are shown as blue arrows. Scores weighted by species cover are indicated by (w). The bank positions of the quadrats are differentiated using different symbols.

In 2008 data (figure 9.4) the same factors exist as gradients through the ordination. Species show a greater degree of clustering together indicating that the are increasingly occurring together.

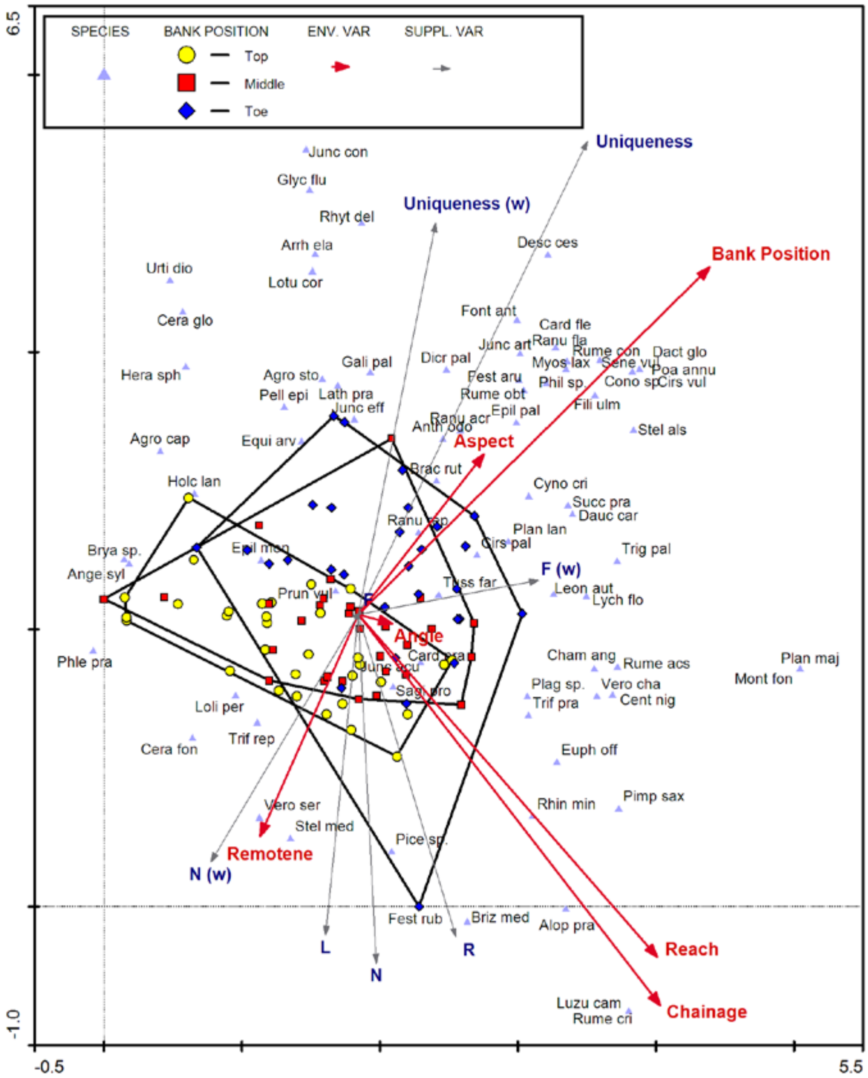


Figure 9.4: Canonical correspondence analysis (detrended) relating 2008 VEGETATION DATA to explanatory variables Bank position, Bank retreat, Remoteness, Chainage and River section shown as red arrows. Ellenburg scores for moisture (F), reaction (R), Light (L) and Nitrogen (N) preference are shown as blue arrows. Scores weighted by species cover are indicated by (w). The bank positions of the quadrats are differentiated using different symbols.

To test the significance of the environmental variables for the patterns of community composition along the banks direct gradient canonical correspondence analysis was used. Scaling focused on inter sampling distances using the recommended Hills scaling (Leps & Smilauer, 2003). Percentage cover was again square root transformed and rare species were not downweighted. Forward selection was used to rank the environmental variables according to their importance in determining the patterns in the species data. Selection was done automatically and variables selected sequentially on the basis of maximum extra fit. The significance of each variable was then estimated using a Monte-carlo permutation test using 499 permutations. Results for each survey year are presented in table 9.2.

		2005	*2006*	*2007*	*2008*
Bank Position	*L1*	***0.24***	***0.21***	***0.21***	***0.17***
	P	0.002	0.002	0.002	0.002
	F	*(3.72)*	*(3.57)*	*(3.68)*	*(3.28)*
Chainage	*L1*		***0.1***	***0.15***	***0.17***
	P	*ns*	0.002	0.002	0.002
	F		*(1.70)*	*(2.84)*	*(3.35)*
Moisture (Lower quartile)	*L1*			***0.12***	
	P	-	-	0.002	-
	F			*(2.22)*	
Mean Silt Depth	*L1*			***0.11***	
	P	-	-	0.02	-
	F			*(2.04)*	
Maximum Silt Depth	*L1*			***0.1***	
	P	-	-	0.026	-
	F			*(1.88)*	
Remoteness	*L1*	***0.14***	***0.12***	***0.09***	***0.1***
	P	0.002	0.002	0.002	0.002
	F	*(2.25)*	*(1.99)*	*(1.67)*	*(2.05)*
Change in Slope (Upper quartile)	*L1*			***0.08***	
	P	-	-	0.018	-
	F			*(1.57)*	
Meander position	*L1*			***0.07***	***0.1***
	P	*ns*	*ns*	0.038	0.002
	F			*(1.35)*	*(1.91)*
River section	*L1*	***0.12***	***0.11***		***0.07***
	P	0.002	0.004	ns	0.004
	F	*(2.00)*	*(1.80)*		*(1.54)*
Angle to Valley Slope	*L1*		***0.09***		
	P	*ns*	0.006	*ns*	*ns*
	F		*(1.66)*		

Table 9.2: Shows output from direct gradient analysis of environmental variables with the vegetation data. Lambda 1 (L1) indicates the proportion of variation explained together with its significance derived from Monte-carlo permutations (P) and F statistic (F). Analysis was done using CANOCO 4.5 for Windows.

Bank position can be used to explain less of the variability in quadrat composition with each successive year. With distance from the top of the realignment (Chainage) the reverse is true with increasing amount of variability explained through time. It seems unlikely that a trend of length of colonisation pathway should appear late on in the channels development. It may be that the relationship with distance existed but that colonists had poor establishment success because of the slower development of physical habitat factors such as the deposition of silt. Alternatively chainage could be acting as a surrogate for an aspect of habitat development such as disturbance or physical diversity which appear to be greater in the upper reaches.

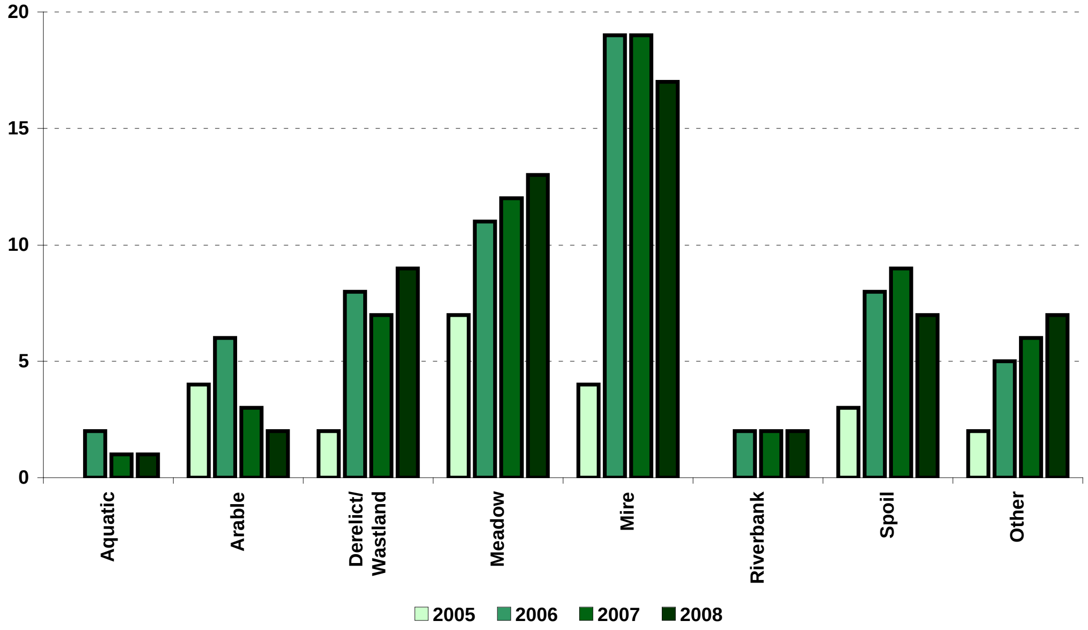


Figure 9.5. Graph showing the number of species recorded along the realignment according to the habitat types in which they are most frequently found (Grime et al., 2008)

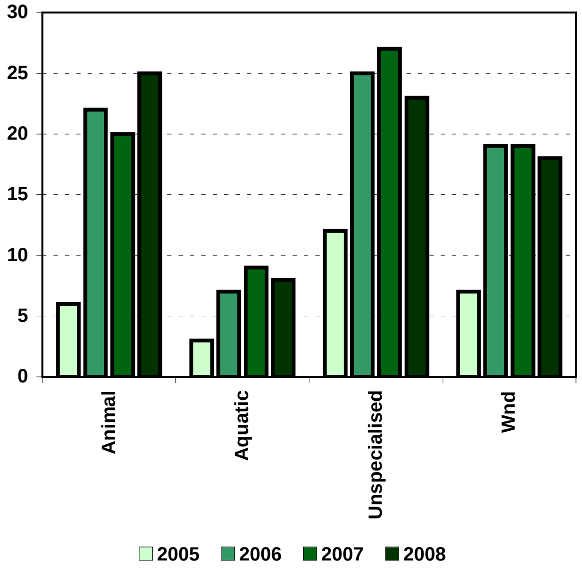


Figure 9.6. Graph showing the number of species recorded along the realignment grouped according to their dispersal strategy as indicated by the morphology of the seeds (Grime et al., 2008)

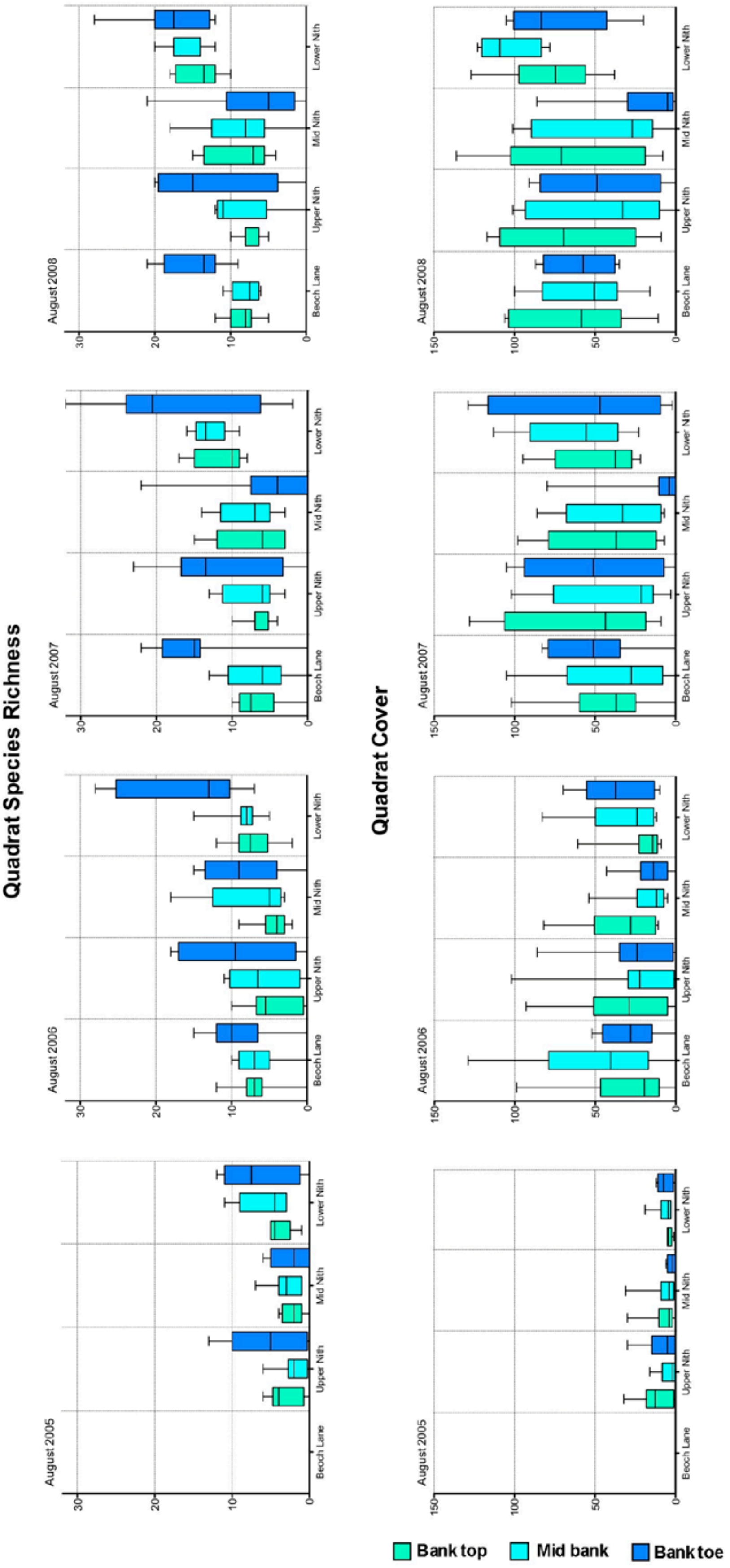


Figure 9.7. Graphs showing the changes bank vegetation diversity from 2005 to 2010 with each of the distinct river sections.

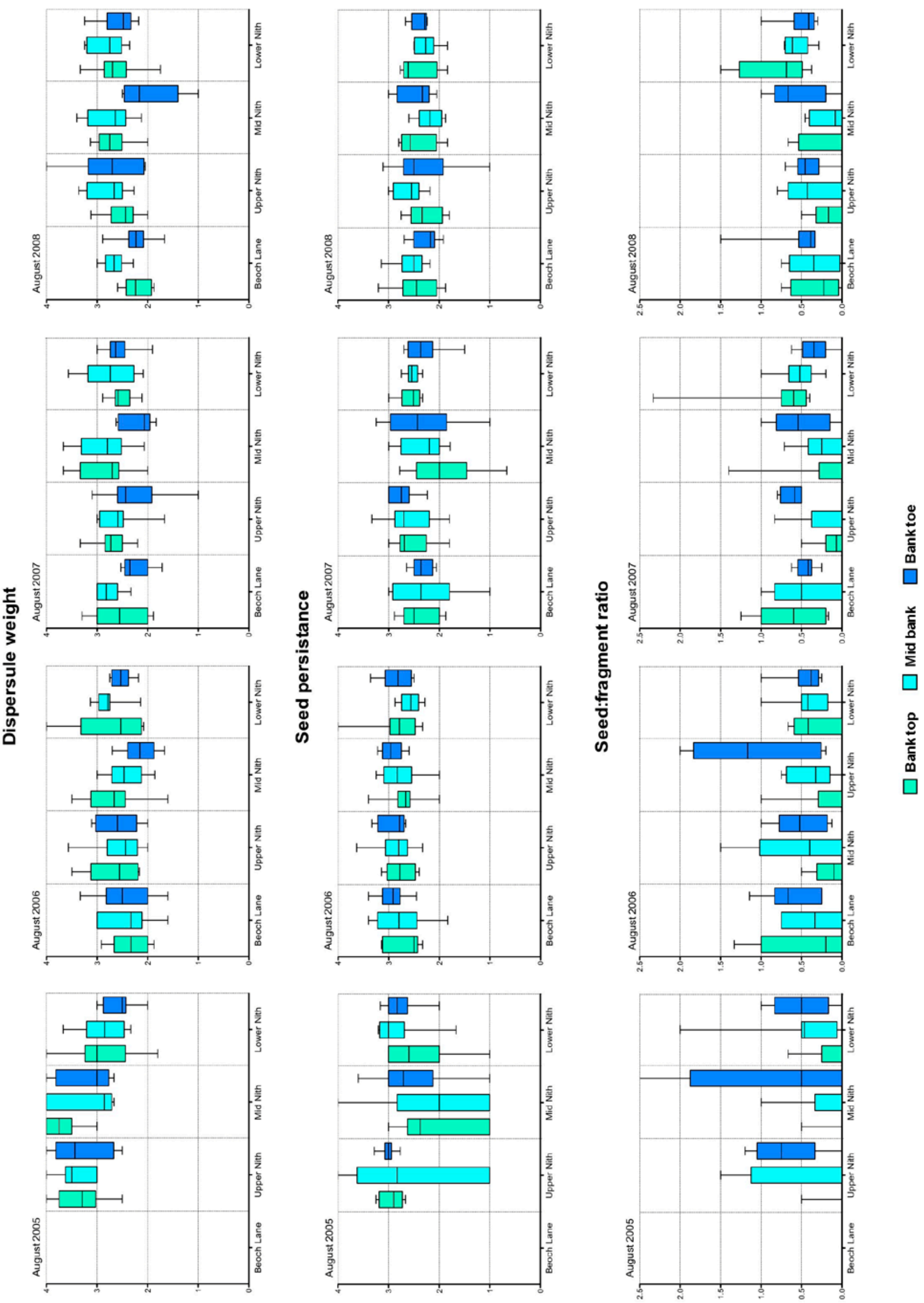


Figure 9.8. Graphs showing the changes bank vegetation dispersal strategy from 2005 to 2010 with each of the distinct river sections.

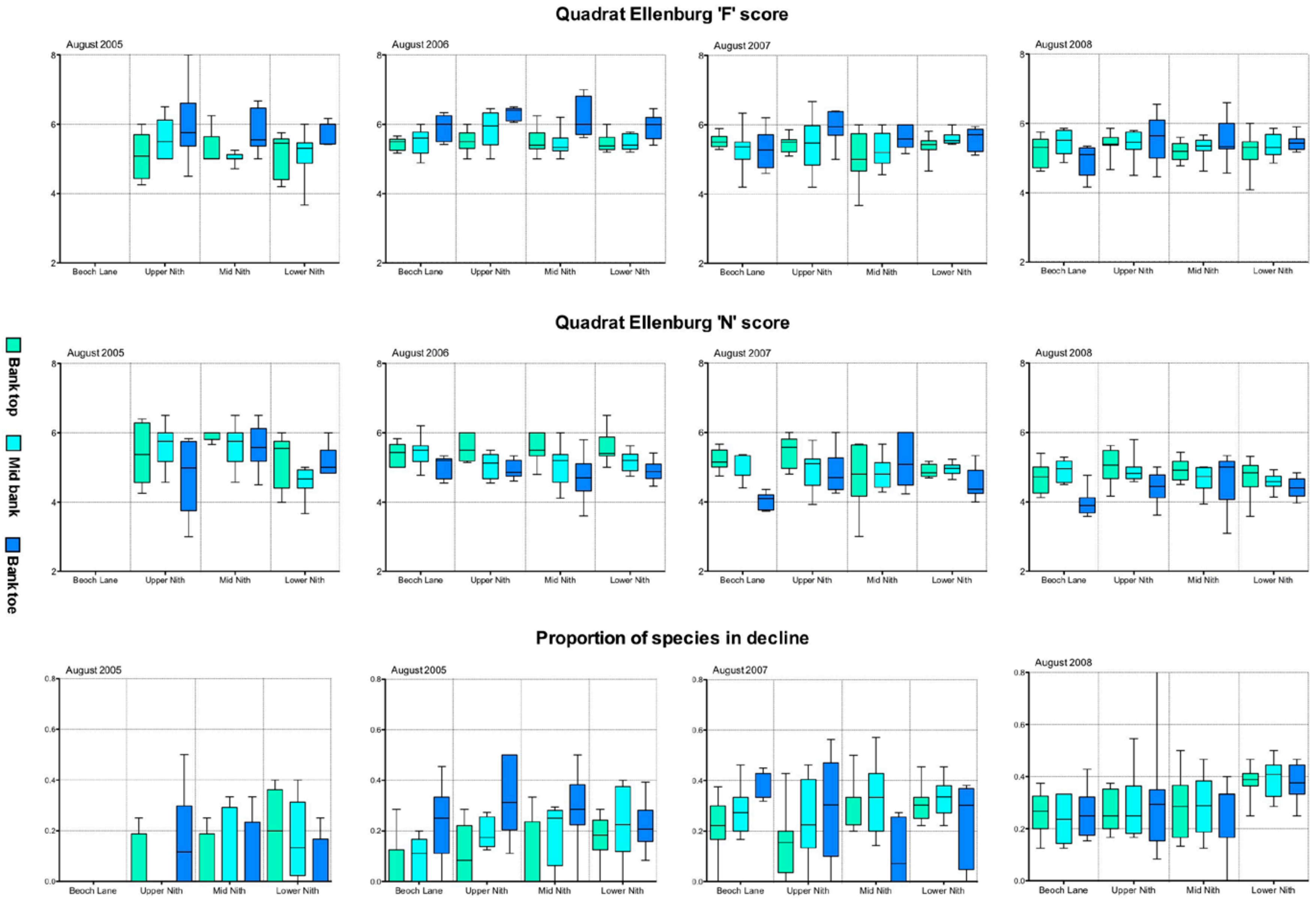


Figure 9.9. Graphs showing the changes bank vegetation character from 2005 to 2010 with each of the distinct river sections.

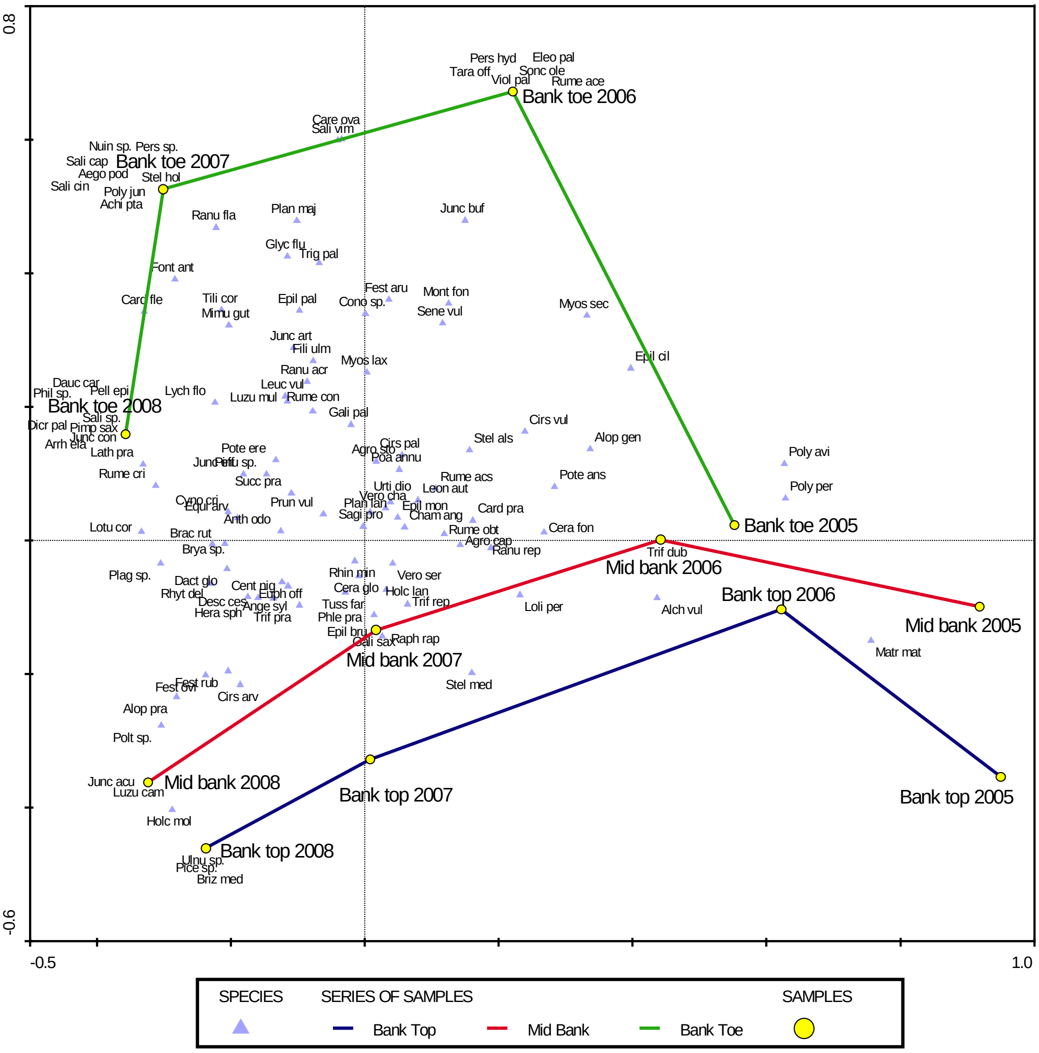


Figure 9.10: Ordination using canonical correspondence analysis showing changes in quadrat vegetation composition and abundance between surveys. Data for quadrats at each bank position in each year are combined to show the different paths of development for each bank position through time

Further analysis of the colonising communities was performed using Indval. Table 9.3 shows the first run of the procedure identifying typical species for each bank position in each year. Table 9.4 shows the second identifying any speces typical of the different 'sections' of river.

_**2005**_	_**2006**_	_**2007**_	_**2008**_
**Bank Top**			
Lolium perenne ** _**(55.7)**_ Trifolium repens _(26.8)_	**Lolium perenne** ** _**(41.1)**_ Trifolium repens _(36.7)_	Festuca rubra _(27.2)_ Holcus lanatus _(28.7)_ **Trifolium repens** ** _**(36.0)**_	Festuca ovina** _(23.6)_ Festuca rubra _(27.0)_ Holcus lanatus _(36.1)_ Lolium perenne _(32.1)_ **Phleum pratense** ** _**(31.2)**_ **Trifolium repens** ** _**(43.0)**_
**Middle Bank**			
-	Holcus lanatus _(25.7)_ Trifolium pratense* _(21.9)_	Lolium perenne _(34.5)_	-
**Bank Toe**			
Cardamine pratense ** _**(27.0)**_ Cirsium palustre** _(16.7)_ **Epilobium montanum** * _**(27.0)**_ Prunella vulgaris* _(16.7)_	**Agrostis stolonifera** ** _**(49.8)**_ Cardamine pratense** _(16.5)_ Deschampsia cespitosa** _(17.2)_ Epilobium palustre** _(15.2)_ Galium palustre** _(15.0)_ Glyceria fluitans** _(18.2)_ **Juncus articulatus** ** _**(34.7)**_ **_Plantago lanceolata** (25.8)_** **Prunella vulgaris** ** _**(26.9)**_ Rumex acetosa** _(19.4)_ **Sagina procumbens** ** _**(27.6)**_	Anthoxanthum odoratum** _(23.4)_ Cynosurus cristatus** _(16.1)_ **Fontinalis antipyretica** ** _**(36.4)**_ **Galium palustre** ** _**(32.4)**_ **Juncus articulatus** ** _**(36.4)**_ Juncus effusus** _(21.8)_ **Leontodon autumnalis** ** _**(25.7)**_ Nuinum sp**_(15.2)_ **Prunella vulgaris** * _**(26.5)**_ **Ranunculus flammula** ** _**(27.3)**_	**Anthoxanthum odoratum** ** _**(27.0)**_ **Dicranella palustris** ** _**(27.3)**_ Epilobium palustre** _(12.1)_ Filipendula ulmaria** _(15.2)_ **Fontinalis antipyretica** ** _**(30.3)**_ Galium palustre** _(18.7)_ **Juncus articulatus** ** _**(26.1)**_ Juncus effusus** _(19.8)_ **Leontodon autumnalis** * _**(25.1)**_ **_Plantago lanceolata** (29.3)_** **_Prunella vulgaris** (30.3)_** Ranunculus flammula** _(15.2)_

*** Species significantly associated with the bank position. ** Highly significant association**
Table 9.3: Shows species indicative of the different BANK POSTIONS in each survey year. Species with indicative of the habitat have Indicator Values greater than 25 and strong indication is shown by species with a value greater than 50 (Indicator values given in brackets)

_**2005**_	_**2006**_	_**2007**_	_**2008**_
**Beoch Lane**			
No data	**_Lolium perenne (25.7)_** _Matricaria discoides** (12.5)_	**_Fontinalis antipyretica* (26.2)_** _Leucanthemum vulgare** (12.5)_ **_Lolium perenne* (32.9)_** _Luzula multiflora** (12.5)_ _Raphanus raphanistrum** (12.5)_	_Cirsium arvense** (24.7)_ _Dicranella palustris** (16.3)_ _Fontinalis antipyretica* (18.3)_ **_Holcus lanatus** (43.2)_**
**Upper Nith**			
Ranunculus repens** (31.3)	_Alchillia vulgaris** (9.5)_ _Filipendula ulmaria** (15.2)_ _Glyceria fluitans* (13.7)_ _Succisa pratensis** (9.5)_	**_Holcus lanatus** (39.3)_**	_Equisetum arvense** (14.3)_ _Ranunculus acris** (11.1)_
**Middle Nith**			
-	_Montia fontana** (13.3)_	-	**_Phleum pratense** (25.1)_**
**Lower Nith**			
Cerastium fontanum** (24.9) **_Epilobium montanum** (27.9)_** _Prunella vulgaris** (17.0)_	_Centaurea nigra** (17.1)_ _Cirsium palustre** (16.7)_ _Prunella vulgaris** (19.2)_ _Trifolium pratense** (24.9)_ **_Tussilago farfara** (34.4)_**	**_Agrostis capillaris** (31.4)_** **_Euphrasia officinalis** (35.3)_** **_Plantago lanceolata** (29.1)_** **_Prunella vulgaris** (29.7)_** **_Rhinanthus minor** (35.3)_** _Rumex acetosa** (24.8)_ **_Rumex conglomeratus** (25.0)_**	_Alopecurus pratenis** (16.7)_ _Cardamine pratense** (13.5)_ _Centaurea nigra** (19.7)_ **_Euphrasia officinalis** (58.0)_** **_Festuca rubra** (60.1)_** **_Leontodon autumnalis** (34.1)_** **_Lolium perenne** (32.2)_** **_Plantago lanceolata* (28.0)_** **_Rhinanthus minor** (37.6)_** **_Sagina procumbens** (45.1)_** _Stellaria media* (17.3)_ **_Trifolium pratense** (60.2)_** **_Trifolium repens* (36.2)_** **_Tussilago farfara** (36.1)_** _Veronica chamaedfys** (20.8)_

*** Species significantly associated with the bank position. ** Highly significant association**
Table 9.4: Shows species indicative of the different SECTIONS OF THE REALIGNMENT in each survey year. Species with indicative of the habitat have Indicator Values greater than 25 and strong indication is shown by species with a value greater than 50 (Indicator values given in brackets)

Bank top *Lolium perenne* (Indval = 55.7; P**Max) is the only species that can be said to be indicative of the quadrats at the bank top position although *Trifolium repens* (Indval=26.8; P NS Max) is also most commonly found at the top of the bank.

No species are significantly indicative of the middle bank position.

Two species can be said to be indicative of the bank toe - *Cardamine pratense* (Indval=27.0; P**Max) and Epilobium montanum (Indval=27.0; P**). Two further species are also significant but only weekly indicative, *Cirsium palustre* (Indval=16.7; P**Max) and

CHAPTER 10

DEVELOPMENT OF A FUNCTIONAL ECOSYSTEM

In this chapter geomorphological and ecological results of observational and manipulative experiments are discussed in the context of published literature. Interpretation of the results and likely implications are presented, with particular respect to various ecological models on which the different approaches to river restoration are based. The extent to which empirical data supports the alternative viewpoints is important to both the design principals and justification of river realignments, regardless of the project motivation. Recommendations for project design and management, and for the development of monitoring protocols presented in chapter 11 are based on conclusions from this discussion.

Aims

To assess the extent to which the findings of the research presented in the previous chapters support ecological models on which river restoration is based.

To refine our understanding of ecosystem development within engineered channels

10.1 Introduction

The natural development of physical habitat is central to a process based approach to river restoration. Over the period of the study, physical habitat development was observable within all reaches on the realignment, albeit to varying degrees. This was evident in the formation of fluvial features, cross-sectional change and changes in the nature, and complexity, of the bed substrate. Modern river restoration also relies on the capacity of flora and fauna to disperse and establish in areas of new habitat. Patterns of colonisation reflect underlying biological and ecological processes driving development.

10.2 Development of channel habitat

10.2.1 Introduction

It is important that in-channel habitat develops at a range of scales from meso-habitat patches to reach scale diversity (Frissel et al., 1986). Given the size of the realignment, the main focus of the study was at the reach scale. Different reaches adjusted their morphology to different degrees, which provided information on (i) the potential for natural processes to develop the physical habitat, (ii) relative importance of individual processes, and (iii) the influence of channel design specifications in promoting development. The first hypotheses to be considered in the light of the results, proposed in chapter 5, regard the development of physical habitat:

From a homogeneous starting point, the 3km of engineered channel will develop a diverse and heterogeneous form comparable to a natural sinuous pool riffle river.

The influence of design specifications, included to promote development towards a natural form and propagate ecosystem development, will be evident in the pattern of geomorphological development.

10.2.2 **Model development**

Figure 10.1 (reproduced from chapter 5) shows the framework (planform, slope, dimension and substrate) within which the process driven development was expected to occur. This process based *eco-hydromorphic* approach recognises that spatial and temporal heterogeneity are fundamental characteristics of fluvial systems and advocates creating a framework within which natural processes such as sediment transport can occur (Clarke et al., 2003). It was hypothesised that design choices (blue) would support development from a relatively homogeneous starting point to a diverse 'natural' river form without having to artificially create habitat (black).

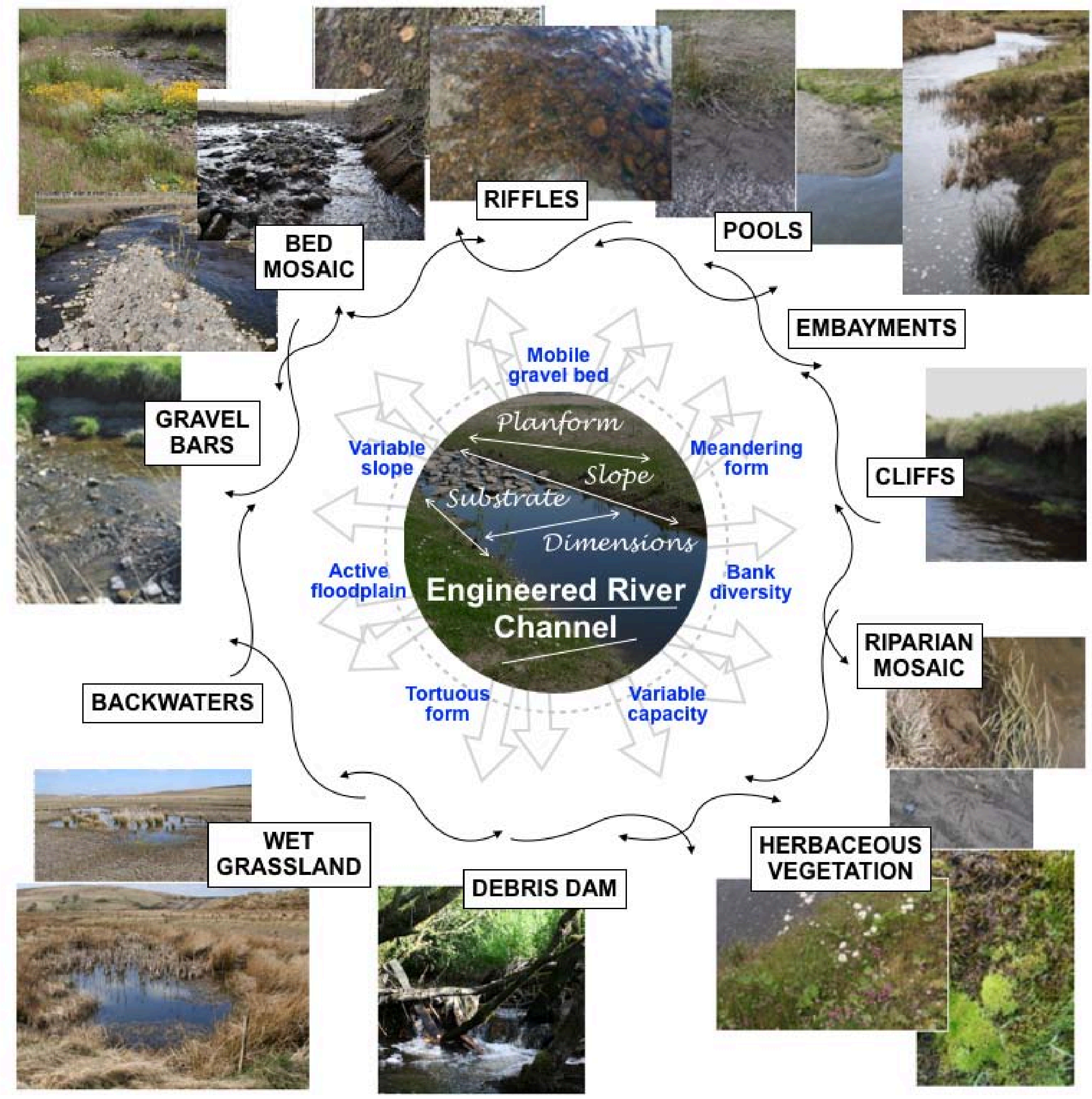


Figure 10.1. Conceptual model linking the design considerations for the engineered channel (blue) to the habitats that are expected to develop (black). Habitats are illustrated in the photographs.

Figure 10.2 shows where the results from this study have demonstrated the links between the framework and habitat development (black arrows) and possible links that are notable in their absence within the realignment design (red arrows). The way in which the results support an eco-hydromorphic approach to river restoration are discussed in the following sections.

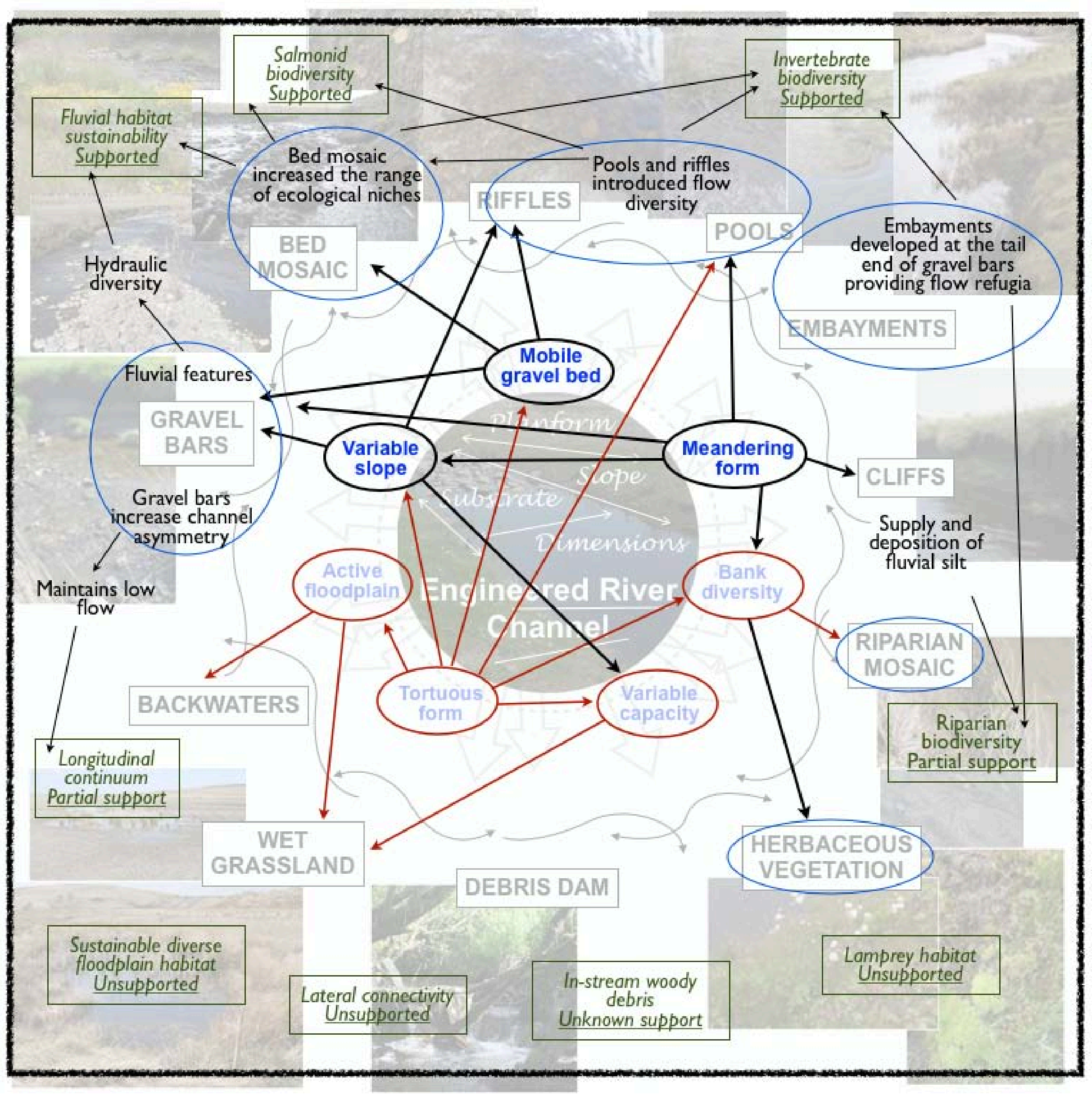


Figure 10.2 development of the model presented in figure 10.1 showing links between the design features specific to the House of Water realignment and the resulting development of physical habitat.

Key to the creation of a functional stream ecosystem is a dynamic sediment system continuously providing and changing areas of bare submerged oxygenated gravels, deep pool areas, exposed gravel bars and eroding banks.

10.2.3 Riverbed – development of a habitat mosaic

A habitat creation focused approach would advocate the artificial creation of a diverse riverbed habitat. However, even in ambitious restoration projects, this if often unfeasible because of the technical complexity, practicalities and inhibitive cost. Minor modifications of the streambed substrate have been used in the past, specifically the addition of cobbles and gravels to create spawning habitat (Kondolf, 2000). Results of this study demonstrate explicit links between the mobile gravel bed, the meandering planform and the development of a bed substrate mosaic. Furthermore substrate diversity is maintained under the dynamic conditions inherent to high-energy streams.

Underlying processes

Mobilisation and resorting of existing bed gravels and the introduction of new sediment from erosion features were both found to be contributing factors to the development of a diverse benthic habitat. Processes can be separated in time. The main development process at early stages of the riverbed evolution must have been transport and resorting of the installed gravels, since no significant sources of sediment had developed within the channel during the first year. Geotextile remained in place along the length of the channel successfully preventing areas of bank erosion from developing. With no new sediment sources, changes in the sediment profile must be attributable to processes of re-sorting. Differentiation of riffle and glide features occurred at a number of sites on the Nith confirming this pattern. Without the installation of gravels that can be mobilised by floods with an annual return period, development would have been delayed.

Resorting of gravels

The apparent reduction in sediment size in a number of profiles may relate to a smaller flood in March 2005. This flood appears to have mobilised the smaller fraction of gravels and selectively deposited them in lower energy sections burying the courser un-mobilised fractions. However, the early changes in substrate size appear to have been temporary. The deposition of finer gravels was not sustained with most sites 'recoarsening' over the following 2 years. It is likely that the selective redistribution of smaller sediment fractions would not have occurred following the development of an armoured layer and provides an early demonstration of the importance of a loosely installed gavels and variable flood sizes play in the development of physical habitat heterogeneity.

Input of new sediment

Bed sediments in 2007 were found to be more poorly sorted than either the placed substrate, or that which developed in the early stages. This is principally a result of the introduction of finer sediment fractions (4-24mm) as indicated by the stretching of the cumulative particle size plots in the direction of smaller gravels at most sample sites. It is likely that areas of channel bank erosion have contributed significantly in this regard. It may be significant that this did not occur at reaches where bank erosion was notable in its absence. Sediment packing by finer grained material, as a feature of natural channels was an important element of habitat development. Fine grains are an important aspect of habitat variability (Knighton, 1998) may introduce an element of stability to areas of channel bed and are ecologically important to many species (Merritt & Cummins, 1997). It was expected that the introduction of fine grains would be a mass transfer process (Briggs et al., 2005) associated with frequent low magnitude disturbance. In this study the introduction of fines seems to have been associated with the more turbulent flow at reaches in the upper section of the Nith realignment. In contrast the stronger flow through bed gravels observed on the lower reaches kept them relatively free of fine material despite significant deposits on the banks and marginal areas of channel bed.

Diversification

A natural bed form has been observed to rapidly develop despite the absence of large keystones, which are thought to play an important role in the development of sedimentary structures (Lamarre & Roy, 2008). However, the long term stability of these structures is unknown and anecdotal evidence indicate the middle reaches have remained geomorphologically dynamic, which may go some way to explain the slower rate of biological development in this area. Boulder placement as fish habitat may serve to provide keystones in areas where they have been placed. The even spacing off boulders to enhance fish habitat, differs from the ideal placement; two boulders having close surface contact with each other, for maximum effect as key stones (Lamarre & Roy, 2008).

Armouring

The resistance to major riverbed profile change beyond 2005 suggests that armouring of the bed developed within the first year along much of the realignment.

Rate of development

Steady development

The steady development of the benthic habitat as compared to the development of identifiable fluvial features as commented by Gurnell et al. (2006) is also observed here. The role of flooding for introducing inter-reach variability in substrate structure was evident after 6 months. Further development including armouring, differentiation of riffle and glide habitat and grading of gravel around meander bends required the reworking actions of successive flood events over the following year. This differs from findings of Lamarre & Roy (2008) who observed bed sediments to reach an *Equilibrium Bed Surface* within two bed mobilisations. Two possible explanations are offered here and should be considered at the design stage of future projects. Firstly the lack of keystones in the installed substrate mix may have prolonged bed instability.

An initial phase of geomorphic adjustment has been identified in previous studies. Development of the armoured layer appears to have significantly reduced sediment mobility during the smaller floods. Assessing the health of channel architecture projects during this stage and basing assumptions about future sediment dynamics on early data is clearly a dangerous strategy cautioned against by a number of authors (Tilman, 1989)

Zero development

These processes are clearly important for ecosystem development, as demonstrated by the lack of geomorphic activity and physical habitat diversity on the Beoch Lane Burn where bed mobilisation was largely absent. Tracer pebbles placed across the channel in 2005 remained in situ for the duration of the study indicating that the low energy conditions were insufficient to mobilise gravels through much of the reach. Although data was not collected for 2005, the similarity of the 2007 sediment profile with the installed gravel profile supports this, as well as highlighting the lack of new sediment inputs. The bed morphology, as a result, remains as constructed along the majority of the length, with only very limited development. The overly stable channel will require a larger flood with a longer return period to generate a natural morphology and until then will remain simplistic in form. Extensive bryophyte growth in later years can also be linked to the static morphology.

Investigation and discussion of riverbed substrates refer only to the surface levels, the section of flood-mobilised gravels, which reach down to the depth of light discontinuity (Bretschko, 1998), and imply development of this section only. The development of deeper gravels forming levels of the Hyporeos remains unknown and is likely to be a considerably slower process. Despite the potential importance of this zone to ecosystem health (Bretschko, 1998), restoration of the hyporeos has yet to be considered in a river architecture project.

10.2.4 Fluvial features – development of pools, riffles and gravel bars

Underlying processes

Variable sediment conveyance

Development was driven by sediment transport and deposition processes, initially acting on the unconsolidated riverbed, and later on the input of riverbank sediments. Fluvial features were most pronounced below sections where channel slope coincided with high discharge. On the Nith this was below the confluence with the Beoch Lane where the combined discharges were sufficient to drive geomorphic processes during floods with an annual return period (plates 7.7-7.9). High sediment transport, relative to the rest of the realignment, exported material from this reach and depositing it in the form of fluvial features where stream power was reduced. On the Nith, this was where gradient reduced and on the inside of meander bends (plates 7.4-7.6) demonstrating the importance of variable bed gradients and sinuous planform for interrupting sediment conveyance. This could be considered an extension of the zones of sediment erosion and storage described by Lane & Richards, (1997). Projects designed with uniform sediment and flow conveyance to ensure channel stability (ref xx) may develop much slower as a result. Gilvear & Bradley (1997) relate rapid physical habitat changes to flood flows shortly after construction of realignments on the Evan Water.

The contrasting channel development of the Nith and Beoch demonstrate the difference between *stable* and *static* bed morphologies, both having had the same gravel mix installed despite the notable difference in flood hydrology. Initial mobility of bed sediments along the River Nith allowed the development of bed sedimentary structures, consistent with a natural river-form in an upland landscape, that contribute significantly to bed stability (Lamarre & Roy, 2003, Weichert et al., 2009). Channels with an ‘artificial armoured layer’ using overly course sediments for instant bed stability would fail to benefit from this initial phase of adjustment. This inadvertently occurred on the Beoch Lane, which has so far remained uniform along much of its length.

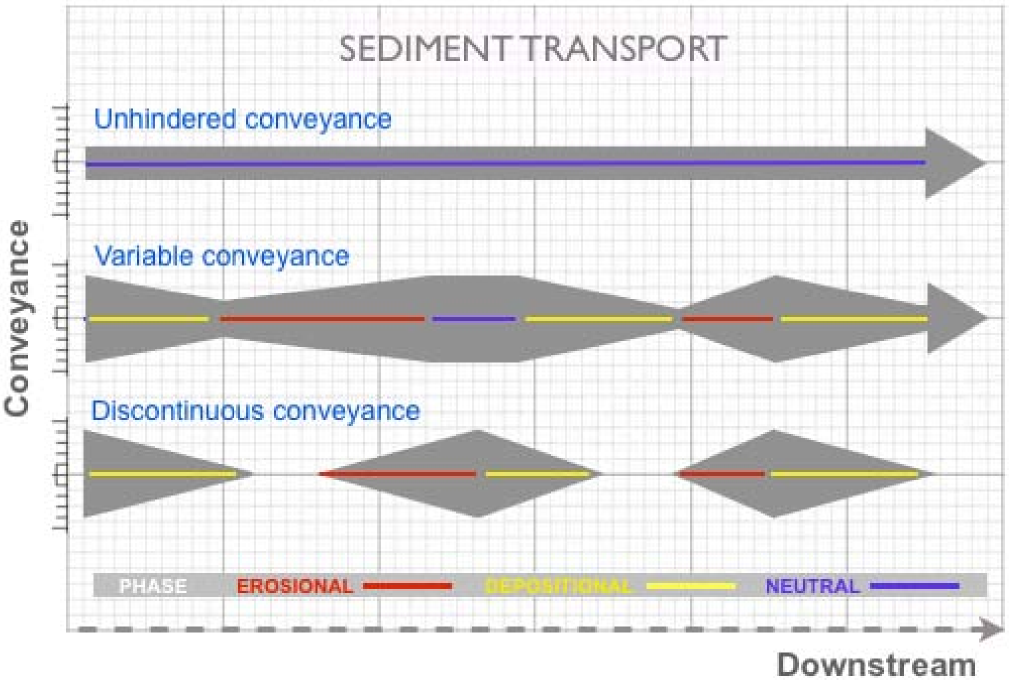


Figure 10.3. Diagram illustrating that changes in sediment conveyance produce alternating phases of erosion and deposition along the channel

Bank erosion

Sediment sources also evolved during the study period; dominance switching from reorganisation of the bed sediments to meander apexes as processes of bed armouring and bank erosion developed. The changes in the gravel bed predominately occurred in the first year of development. Although this included a particularly flood rich winter an equally flood rich period recorded during the third year did not produce the same marked change in the profile. In contrast change to the toe of the river bank occurred across all years and in many cases (e.g cross-sections 12 and 13, figure 7.25) the changes were greatest in later years. Changes to the upper riparian zone have yet to occur along most of the realignment. As inherently dynamic systems, upland rivers are expected to be in a state of constant sediment flux. This is an important characteristic of the ecosystem. The intermediate disturbance hypothesis (Grime, 1979), for example, proposes the associated disturbance plays an important role in the maintenance of biodiversity. Whilst a mobile gravel bed may act as a sediment source fuelling the development of fluvial features at early stages, without the development alternative sediment sources, the early habitat development may not be maintained. Erosion features on the riverbanks are important in this respect and the implementation of design strategies, aimed at minimising erosion, will have a negative impact on long-term sustainability that is not evident in early development.

Bed erosion

Sinuosity alone may not be sufficient for natural habitat forms to develop. Natural rivers rarely have 'symmetrical sinuosity' i.e. where planform resembles a sine-wave parallel to channel slope (except in areas of very low sediment adhesion). However stream architecture projects often 'restore' such form. Examples of where this has occurred on the Nith realignment have produced meander apexes positioned at the point of highest bed-slope. These develop as riffle features rather than pools, or, where the meander is confined, fast flowing deep water confirming observations that they are unlikely to result in habitats that would naturally exist at the site (Kondolf, 2006). Adoption of a more tortuous planform design equivalent to a later stage of channel evolution would create meanders perpendicular to valley slope producing scour conditions under high flows but more genuine 'still' pool conditions at times of normal and low flow.

Diversification

The development of a variable bed topography during the early stages represents an increase in habitat diversity, an important aspect of channel naturalisation creating distinct habitat-velocity relationships key to functional development e.g. salmon egg laying habitat (Kemp et al., 1999). Visual examination of the physical form provides qualitative evidence of the strong influence that stream power and channel size have over habitat development. The most prominent development was associated with sediment transport on meander bends producing pool habitat and reaches with high discharge and slope resulting in a dynamic geomorphic character. This is clearly visible in the thalweg profiles (section 7.2).

The habitat complexity introduced by heterogeneity influences many aspects of aquatic ecosystems (see Shumway et al., 2007) Development of instream heterogeneity will be important for flow refugia that allows flora and fauna to survive disturbance events (Clarke et al., 2003). Where a variable topography develops is across the channel it allows for the existence of a range of surface flow depths at low flow conditions. And maintains longitudinal connectivity. The cross-sectional profile can represent a rapid gradient in habitat conditions from very slow, zero or even reverse flow of marginal habitat to the fast flow of the thalweg within a small spatial scale.

Rate of development

Early development of distinct fluvial features including point bars, pools and riffles demonstrated the potential capacity of engineered channels for very rapid development of diverse stream habitat. Photographic evidence shows this to occur within 6 months, although the flood hydrograph suggests that it was likely to have been much quicker. This is supported by data collected just two weeks after connection of the new channel providing some evidence of gravel bar formation in the most active middle reaches (See cross-sections at 1500m and 1300m chainage. Figure 7.25). Rapid development of fluvial features have been observed elsewhere in the literature (e.g. Gilvear & Bradley, 1997; Gurnell et al., 2006; Lamarre & Roy, 2008) typically reporting development within 6 to 12 months. This research confirms the same rate of development is possible for much larger scale projects.

An adjustment period is a feature of engineered channels. It is not clear whether these early adjustments are simply a naturalisation of the constructed form or result from the unconsolidated nature of the installed gravels prior to the development of an armoured layer. Regardless, it is clear from the results that the two processes are linked with naturalisation permitted when the gravels match the flow conditions.

Point bars continued to grow and evolve throughout the study, as they do in naturally migrating rivers, evident in both fixed point photography (e.g. plate 7.6) and the cross-sectional profiles (figure 7.25). In many cases just small amounts of erosion or deposition led to the development of features, distinct in the fixed point photography. Changes at a scale not practicable with current contouring methods (Sawyer et al, 2009).

10.2.5 Riparian zone and cross-sectional form

Rate of development

Limited morphological change of the riverbanks demonstrate that, when constructed in a trapezoidal form, process driven development may be unacceptably slow for the purposes of river restoration.

Underlying processes

Erosion and deposition

The process of point bar deposition and associated erosion of the outer bank created an asymmetrical channel form, akin to a natural river, as well as reducing bank height and slope. This altering of the channel form represents process driven evolution but was limited in extent developing a very small proportion of the channel length. Furthermore, additional development from gravel bar habitat to riparian habitat relies on the slow processes of weathering and sedimentation to stabilise the substrate.

Weathering

The effects of exposure to wind and rain were not investigated but observations suggest little impact on riverbank morphology, other than two locations where runoff from the floodplain produced down cutting creating a channel that bisects the riverbank.

Self-organised criticality

The lack of physical habitat development on the upper bank over the three year period, as indicated at every cross-section, suggests that this area may not be strongly influenced by hydromorphological processes in the same way that in-channel habitat is. Instead, riparian development may develop and evolve through processes relating to channel failure and course change. These processes operate on an entirely different time scale and as such are more likely to be considered as large disturbance events. A true eco-hydromorphic base project would aim to restore these processes although the timescale and resource investment required would likely make this unacceptable. Instead a habitat-based approach may achieve higher habitat quality and ecosystem service provision, if not ecosystem integrity. Allowing and promoting criticality would in the long term, not only provide insight into processes important for the sustainability of a natural riparian zone, but may also increase the sustainability and health of the wider catchment system (Fonstad & Marcus, 2003)

10.2.6 Planform

Meander development combined all four forms of migration but was dominated by downstream translation. The maximum rate of outer bank erosion consistently occurred on the downstream side of the apex across a range of meander amplitudes. Likewise, although point bars stretched around the inside of the meanders they were widest at this same downstream angle. This produces meander migration best represented by the translation type although was really a combination of all four types. Bank material will have played a role in the repeated pattern of meander erosion. Material used to reconstruct the flood plain was highly uniform and free of large objects such as boulders and tree roots that would otherwise have introduced another dimension to physical habitat development and more complex, irregular meanders.

Rip-rap around the outside of meander bends on the upper reaches of the Nith and Beoch Lane clearly prevented channel migration. It is possible that this will have slowed ecosystem development in a number of ways. Meander erosion would have provided a sediment source, already established as an important aspect of the developing riverbed and riparian ecosystems. Limiting meander migration also resulted in smaller steeper point bars, an important river habitat unit with the potential to support unique plant and invertebrate communities. These stunted point bars also appear to be more readily disturbed and have lower vegetation richness. Bars are less complex with well sorted gravels and no associated silt deposition or slack water.

10.2.7 Summary – Process driven development of physical habitat

The process-based approach assumes that a relatively simple morphology can be constructed as a framework to support ecosystem development. This research has demonstrated not only that this is a practical proposition, but also that the rate of development can be very rapid if the framework is correctly implemented. Consideration of design elements in terms of the enhancement or hindrance of processes rather than the creation of habitats is an important part of the approach. As such, the creation of morphology is as important as it would be in a habitat based approach but should be aimed at maximising processes such as sediment resorting, bank erosion and variable sediment conveyance.

This can be exemplified by considering two techniques for targeting restoration at salmonid numbers:

1. The installation of spawning gravel habitat ideal for salmonids will not be restructured into active fluvial features such as riffles if flow conditions are not sufficient to mobilise the gravels. The channel remains uniform and the bed static, leading to extensive macrophyte growth and zero spawning habitats. Installing a more diverse profile promotes the process of restructuring producing and maintaining specific areas of spawning gravels within a habitat mosaic.

2. The installation of boulders, equally spaced across the streambed serve as enhancements to fish habitat but could also be used as a surrogate for natural keystones in the process of bed development if placement and size were varied. The improved habitat suitability for invertebrates would improve valuable food resources.

Furthermore, as with any other flora or fauna, sustainable numbers are likely to be at a level in proportion to the wider ecosystem.

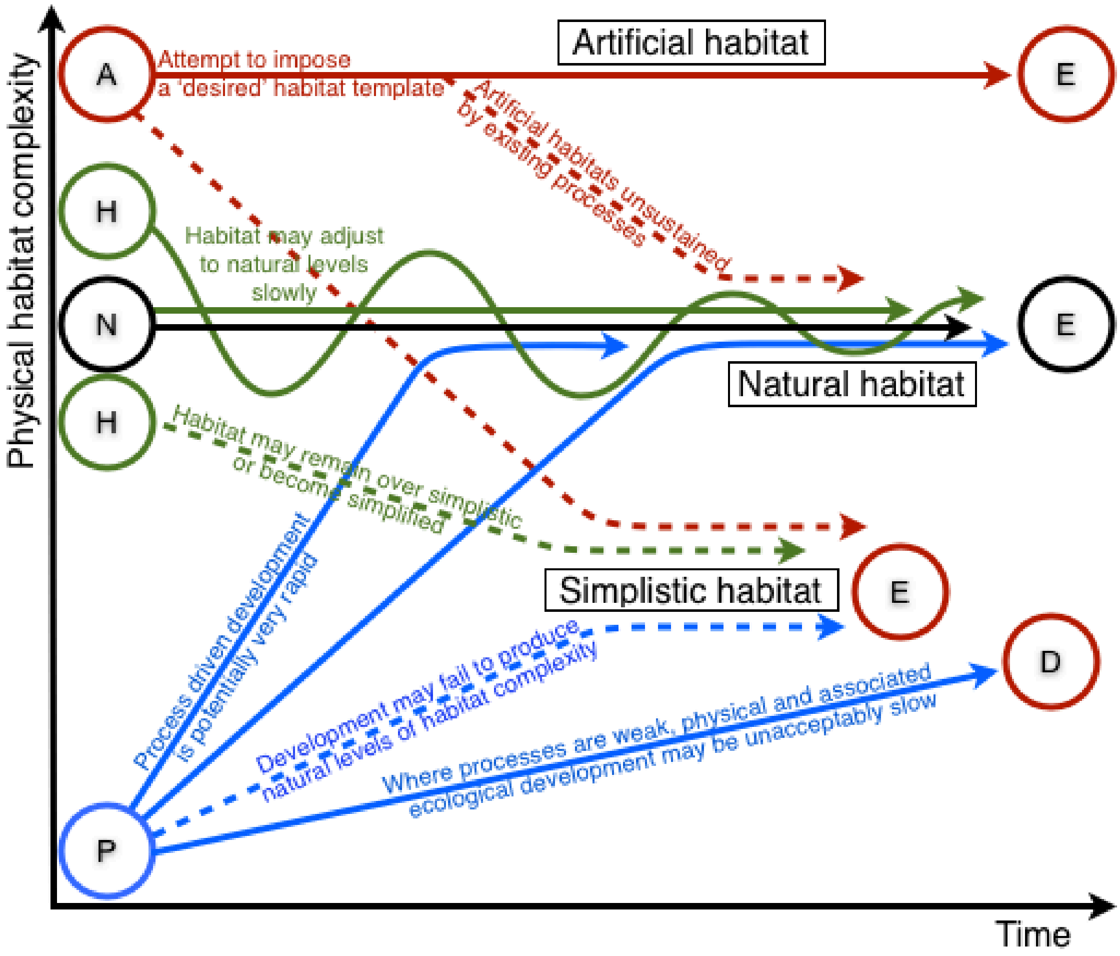


Figure 10.4. Conceptual diagram showing the development pathways of different approaches to habitat creation

The two approaches to the creation of physical habitat in river realignments and restoration projects are shown in figure 10.4. From time zero the ecosystems begin to develop from different levels of physical habitat complexity.

Eco-hydromorphic – process based approach

The process based eco-hydromorphic approach starts with low complexity (P). It has been demonstrated on upper riverbanks of the realignment that where processes are weak, relying on processes to drive change may result in a very slow or delayed development (D) of ecosystem structure and functioning. It is also possible for development to reach a simplified or alternative habitat condition as observed on the bed of the Beoch Lane (Es). It is also possible, given the right framework, for natural processes to rapidly drive ecosystem development to the desired sustainable endpoint (E) and, where design consideration is given to maximising these processes, this can be very rapid. This includes the careful design of planform, bed slope and the installed substrate. It remains to be seen whether this is faster than the adjustment phase of the habitat approach.

The field of dreams – habitat based approach

In the habitat based field of dreams approach an attempt is made to construct the habitat in the natural form (N) and at best may be somewhere near to this state (H). The deliberate creation of pool and riffle structures at a low level within the realignment were intended to enhance the rate of ecosystem development by providing elements of physical habitat diversity from day one. It is possible that the construction of habitat may result in a simplistic but stable endpoint equilibrium (Es). The pools at the lower end of the realignment may turn out to approximate this development as did the overly wide 2000 diversion, where, as a result of the habitat structure the stream power is reduced and is unable to rework the sediments in to a more complex pool form. Equally misplaced habitats could by the same means prevent development of a more natural form. Alternatively the constructed habitat may simply adjust to the natural form or complexity, initially fluctuating around the equilibrium level because of feedback mechanisms. The length of this adjustment phase is likely to depend on the efficiency of the same processes driving the eco-hydromorphic approach. Furthermore if natural processes are not restored there is no guarantee that adjustments will take the habitat to the desired end point (E). In the Nith example, habitat structures degraded rapidly through a combination of bed elevation reduction and infilling of pools in the more active reaches and were replaced by process driven development such as the scour of pools on the outside of meander apexes.

Artificial habitat

Under some circumstances there may be a desire to restore to an artificial habitat (A). However it is unlikely that this will be sustained, at a satisfactory level of ecosystem diversity, by the natural processes that maintain the natural equilibrium. And without ongoing management or alteration of the natural processes, development will more likely proceed towards the natural endpoint (E) or the simplistic endpoint (Es) rather than the artificial one (Ea). This was demonstrated by the positioning of evenly spaced boulders along much of the Nith realignment, which were steadily worked into the streambed by erosive flows undercutting them.

The field of dreams hypothesis, that advocates creation of physical habitat has been widely implemented but should be done so with caution. The results of this research show that desirable physical habitat features, constructed without due consideration of their scale and positioning, are unlikely to be maintained by geomorphological processes. Past examples include 'poor performance of riffles' (Harper et al., 1998) due to sedimentation of riffle areas or erosion and dispersal of riffle cobbles. Positioning of flow deflectors to alter conditions and keep riffles clear is a solution, in certain circumstances, for maintaining *desirable* fluvial features but should not be considered natural or sustainable. Such approaches may be appropriate in projects that aim to restore flow diversity when restoration of geomorphological diversity is not possible. However, in high-energy upland rivers, the results demonstrate a considerable capacity for the rapid formation of pools and riffles, and that engineered features rapidly degrade during this processes. There are also low energy sections of channel in upland rivers, for example the lower reaches of the realignment, which would require much higher discharges to produce 'formative flows'. In these areas many of the constructed pools and riffle have remained, although in the case of some of the slow flowing pools, deposition of fine sediments is beginning to reduce pool length and depth. Observations suggest that this may be much slower process. It should be noted that reaches downstream of the realignment suggest that the character of the lower reaches might be better described as a plain bed than pool/riffle channel in which case pool riffle sequences will be inappropriate in the long term and unlikely to be sustained.

10.3 Invertebrate colonisation

10.3.1 Introduction

Freshwater macroinvertebrates are important to the health of river ecosystems. They are involved in nutrient processing as well as being a food resource for many fish, birds and small mammal species. The development of a diverse and functional community therefore represents an important stage of development of the overall ecosystem. However, little is known about the processes involved, nor the influences of ecological and biological factors. For example, suggestions that significant barriers to colonisation have the potential to prevent successful development of river restoration projects are rarely able to draw on data from formal scientific research. By studying different facets of diversity it was possible to provide insights into the ecology of colonisation. Gradients in alpha diversity, or level of underlying community richness, gave insights into the direction from which colonists were entering the realignment and rates of change could be related to the effectiveness of the pathways. Beta diversity, how similar benthic communities were, was applied to both taxonomic and trait scores allowing investigation of the biological and ecological limitations to colonisation. This allowed the following two hypotheses to be addressed:

3. There are predictable environmental barriers to colonisation, which affect the dispersal and establishment of macroinvertebrate colonists. These must be considered when designing ecosystem architecture projects.

4. There will be specific morphological and behavioural traits that have evolved in some species to overcome environmental barriers to colonisation affecting dispersal and establishment. Species possessing these traits will be superior colonists.

10.3.2 Patterns in alpha diversity and abundance

Pathway availability

The availability of pathways was an influential ecological process behind the development of the House of Water realignment. This has important implications for river restoration projects.

The dominance of drift during early development

Drift was immediately available to the upstream species community as a route for colonists to enter the diversion. Conversely, no pathway was available to downstream sources. Despite kick-net sampling being considered only semi-quantitative, a dominant pattern of decreasing community density (as indicated by Sqrt Abundance; figure 7.5) with increasing distance from the upstream species pool was observed within two weeks of connection and persisted for the first nine months of ecological development. Square root sample abundance is used as the preferred index for community density because it accounts for the risk of sampling large numbers of some species simply because of the patchy nature of the distribution. The reduction in numbers of individuals with increased distance along any colonisation pathway is expected because as individuals ‘settle out’ on to the habitat the number of ‘dispersers’ is reduced. This is consistently observed under experimental conditions (Elliot, 2002). A reduction in the density of colonists (sample abundance) with increasing distance along the drift pathway was also observed at early stages of development on the 2000 diversion (Griffin, 2006). Although the gradient of the distance-density relationship reduces through time as would be expected with the pathway constantly in use, the correlation remains strong accounting for the significant proportion of variability. Sampling variability is likely to be sufficient to account for the remainder, suggesting no other colonisation pathways are available during this period.

The distance-density relationship was reflected in patterns of richness. Although often interpreted as a simple measure of alpha-diversity, richness scores alone were not considered reliable enough as they are not independent of sample abundance in situations where communities of different densities are compared. Lower sample richness would always be expected in fixed sample sizes from lower density communities, but this does not necessarily indicate lower community richness. Instead *Fishers-alpha* and *Rarefaction* were used as indicators of alpha diversity. Both confirm the relationship of decreasing community alpha diversity with distance after accounting for the differences in density. A lower level of Alpha-diversity in the realignment compared to control sites indicated that drift, as a colonisation pathway is selectively available, acting as a barrier to some specific species. Furthermore the observation of reducing alpha diversity with increasing distance indicated that specific species are more effective than others at dispersing via drift. In this respect, downstream distance can be considered as a species-specific barrier to the dispersal of invertebrates. Biological and ecological traits conferring a colonisation advantage are investigated later with the use of Beta-diversity.

The contribution of additional colonisation pathways and habitat factors

Patterns of underlying Alpha-diversity show that the high sample richness at the downstream end of the realignment in later stages of development, relative to other sites (figure 7.4) can be explained by the considerably higher levels of abundance in the downstream samples. This also highlights the indication given by sample abundance, that the distance-density relationship breaks down at this point in time and an explanation for the relative high densities is required. It is likely that additional pathways are contributing to development at later stages of development. Abundance and richness of specific groups of species associated through shared biological and ecological traits proved to be of limited use for investigating colonisation because they failed to summarise the data sufficiently to highlight patterns (Appendix) and were complicated by significant seasonal changes. However, for the investigation of the communities at specific sites and times they are helpful. They show that in November 2005 sites through out the realignment share very similar proportions of group richness but that at sites N7 and N8, where high density is indicated, the community is dominated by high abundance of biological trait group e and ecological trait group F. This is also observed at the downstream control sites. Other sites within the realignment biological group f and ecological group A are dominant.

Table 10.1. Summary of biological and ecological traits shared by the dominant populations in the downstream reaches and controls compared to the middle and upstream reaches

		Downstream reaches e(e1)/F(F3)	Upstream/middle reaches f/A
Biological Traits	**Size**	Small/Medium	Medium
	Dispersal	Mixed	Mixed
	Life cycle	Short-lived uni/pluri-voltine	Monovoltine
	Eggs	Cemented	Cemented or attachment structure Quiescence
	Active stage	Crawlers	Crawlers
	Feeding habit	Varied	Shredders
	Typical families	Crustaeca, Diptera, Tricoptera	Tricoptera, Ephemeroptera, Plecoptera
Ecological traits	**Habitat type**	Mixed but not the active main channel	River channel
	Substrate	Macrophytes	Course mineral substrates or vegetation
	Trophic status	Oligotrophic	Mesotrophic
	Typical families	Gastropoda, Hirudinea, Trichoptera	Tricoptera, Plecoptera

10.3.3 Beta diversity

Methods of beta diversity measurement, when applied directly to species data, gave little insight in to patterns of colonisation. Most indices struggle to compare communities with very different levels of abundance, as was the case at early stages of channel development. The lower scores achieved at downstream locations of the diversion are unsurprising given the lower number of individuals and do not necessarily mean that downstream communities are less similar to the species pool community. All realignment taxa at this stage are shared with the reference sites, since they represent the only source of colonists.

Colonisation traits

The second hypothesis relating to community development proposes that biological traits should be evident within the colonising community that allow individuals to overcome the barriers to ecosystem development. Patterns of alpha diversity suggest that at early stages barriers exist to both the dispersal and establishment of species. Traits were narrowed down to those expected to be relevant to dispersal and colonisation processes. The chosen traits expected to related to dispersal were Maximal size, Life cycle duration, Potential number of lifecycles per year, dispersal mode, and resistance form. Traits thought to confer an establishment advantage were food, feeding habit, substrate preference and transversal distribution. Using community data from the 2000 realignment as an independent data set it was possible to establish particular biological characteristics shared by colonists and then test the patterns using data from the 2004 realignment.

Traits conferring a dispersal advantage

Some traits were clearly better represented in the diversion than reference sites or baseline samples. One such trait was a resistant form of egg. Although the best represented ‘trait modality' in both realignment and reference samples was to have no resistant form, having eggs with some resistance to destruction or desiccation is selectively favoured in the processes leading to colonisation. There was also evidence of a selection pressure favouring taxa with a fast life cycle of less than or equal to a year. This points to the possibility that the dispersal of eggs may represent a significant colonisation strategy. This is an idea not discussed in the literature. It is reasonable to suggest that at times of high flow and bed mobilisation that mortality of invertebrate eggs, larvae or nymphs may be high (ref xx) conferring an advantage in terms of community persistence on species with tough eggs, that, once settled in the realignment, hatch and develop rapidly as a short life cycle requires. Because the gradient of species richness is not mirrored in the community expression of resistant egg forms it is likely that the eggs enhance survival and establishment rather than dispersal. This is supported by the patterns of trait expression, which show a strong presence of resistant egg characteristics in areas of geomorphic dynamism and a week presence, at a similar level to the reference sites in the lower reaches of the Beoch Lane where there has been no geomorphic activity.

Taxa maximal size data was also looked at because larger size might confer a survival advantage of colonising individuals. The size class of 0.25>0.5mm although smaller than the net size was fairly well represented at control sites. However, scores were consistently lower in diversion samples for the 2000 data. This could be related to survival during dispersal, dispersal distance or habitat condition. Small taxa require finer sediments for case construction in the case of Hydroptilla, and ingest smaller sized food items. The validity of the pattern was supported by its recurrence in the 2004 diversion. It

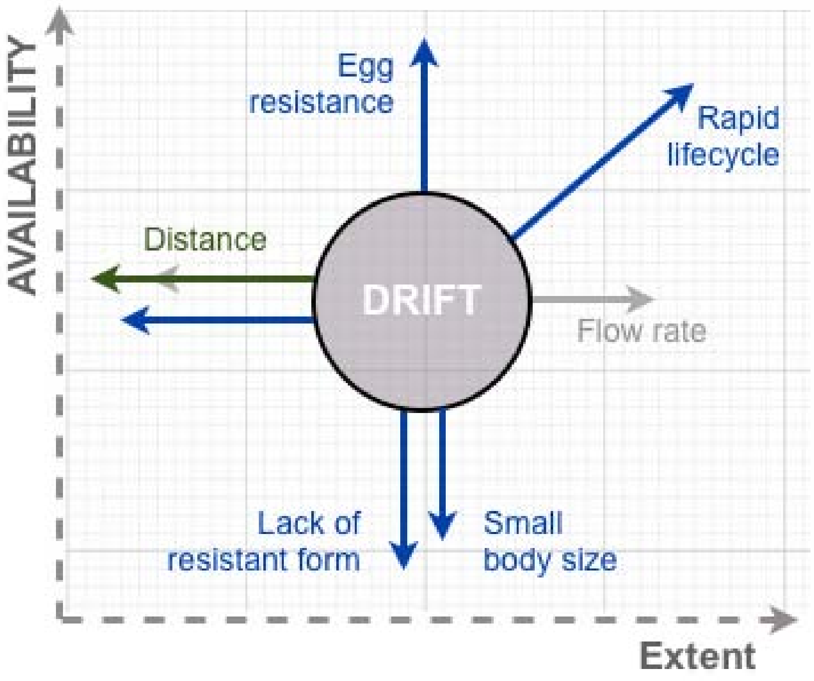


Figure 10.5. Diagram illustrating the established factors relating to the speed of the drift pathway and its availability to potential macroinvertebrate colonists.

Circumstantial evidence is presented for the knock on effects that a lack of physical habitat diversity in riparian and flood plain areas can have on invertebrate communities. This was not investigated directly with regard to riparian and floodplain fauna. However it is expected to affect lateral connections that are thought to be important to river health. A trait association with bank and connected side arms is under-represented in the 2004 diversion until the end of its second year of development. The pattern was not obvious in the 2000 diversion, perhaps because of the construction of back waters and the presence of an established floodplain area.

Traits conferring an establishment advantage

Habitat development is clearly important in river architecture projects. Biota will not successfully establish in areas that fail to provide their necessary ecological requirements. It was therefore expected that community characteristics would reflect habitat conditions. A distinct habitat gradient was expected to develop along the diversion as a result of the project scale and the changing valley characteristics.

The upstream end of the project characterised by steeper gradients, greater hydraulic diversity, pools and riffles, boulders and patches of bed rock. The lower sections by lower bed gradients, relatively simple hydraulics marked by a predominance of glide habitat and areas of gravel and sand in addition to the cobble and pebble sections of channel. Only in one regard does community trait expression reflect the different character of the upper and lower control sites. This is the association of the community with the river banks and channel side arms which is consistently more strongly expressed in the invertebrate samples collected from the downstream reference sites than it is in the upstream sites

10.3.4 Community structure

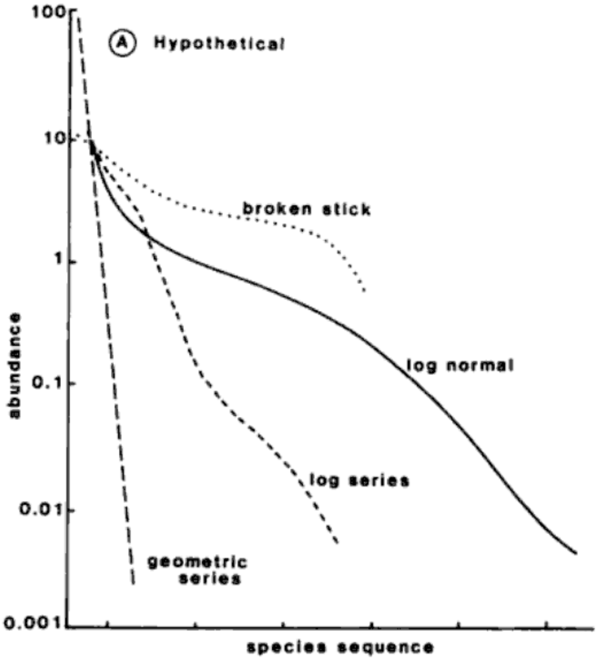


Figure 10.6. Rank species abundance plot showing the different hypothetical models for community structure. (From Magurran, 1988)

10.3.5 Influence of habitat

Experiments using mossy boulders showed that potential colonists are present within the realignment given habitat suitable for establishment. Specific taxa that are otherwise very rarely found in the realignment, a level of scarcity that suggests individuals found represent a transient population.

Influence of disturbance
Not about mixture of climax and pioneer species - in a lotic system they are one and the same. Pioneer species must exist in the climax community else they wouldn't be available to 'pioneer' colonisation of new and freshly disturbed habitat.
The confined valley upstream would impose a different geomorphological form. Other studies have shown a dissimilarity in species assemblages corresponding to this zonation (Parsons et al., 2003)

Rates of development are meaningless without a desired endpoint against which development can be compared. Here a comparison is made to reference sites for reasons outlined in the methods.
Intermediate disturbance and may be important for maintaining heterogeneity
Sites were located both along the diversion and on reference reaches to show whether or not the ecological status of the 2004 Nith diversion compares to that at relatively un-impacted natural conditions.

It is clear from the results that for some species drift is not the principal mode of dispersal. This seems to include many of the beetle taxa etc etc To encourage this species into the diversion it is clear that other dispersal pathways should be supported by the management strategy

A well developed riparian corridor will be invaluable in encouraging and attracting flying immigrants.

It is difficult to make predictions since there have been very few studies specifically investigating the influence that valley morphology can have on aquatic insect communities. (Parsons et al., 2003)

Rate of development

Species richness is reported as being generally rapid in newly created sinuous channels, reaching near pre-realignment levels within a year. (Biggs et al. 1998). The flow dependent changes in drift density and drift rate observed for many taxa following experimental discharge manipulations (Poff and Ward, 1991) emphasises the importance of hydrological regime in influencing the potential colonisation of a man made channel. Data from Bird and Hynes (1981) supports drift as the most significant pathway for colonisation. The model states that drift is the dominant mechanism behind the community development on the streambed.

Observations that distance travelled by drifting invertebrates is a function of water velocity (Elliot, 2001) would imply that rate of colonisation of the benthic habitat will be driven by hydraulic conditions. If this were true the two realignment projects would be expected to recover at similar rates being that hydraulic conditions are likely to be so similar. However, the extent to which the community develops will be governed by the integrity of the biotic and abiotic habitat.

If drift were the dominant over patterns of colonisation one would expect the community that developed to be a function of the dynamic dispersal (stochastic) processes. The alternative is that community development is more deterministic. That is to say shaped by the habitat, and species interactions including competition for resources and predation.

The power of indices

Whilst species richness is often an informative measure, more complex diversity indices have the potential to offer a considerable degree of ecological insight (Magurran, 1988). Many of the indices comparing two communities that take into account the abundances of individual tax are strongly influenced by species richness and sample size (Magurran, 1988). Morisita has been recommended b a number of authors because it is relatively uninfluenced by these factors (ref ref; Hammer, 2002)

The also results show that very low densities of individuals, in this case at early stages of development reflected in the low sample abundances present in the October 2004 survey, are problematic for many indices of diversity. The diversity of low abundance communities are particularly susceptible to random sampling variability producing some samples with very low numbers of a few species. Diversity measures with a strong 'evenness' aspect interpret the small but comparable population abundances as even and score these systems highly, incorrectly implying good ecosystem health. This has serious implications for the monitoring of projects targeting impoverished systems.

Perhaps one of the most interesting modern indexes of diversity is Clark and Warwick’s index of Taxonomic distinctness. This summarises the taxonomic relatedness of species at each sample site. Like morisita's index, Taxonomic distinctness is unaffected by sample size and sampling effort.

10.4 Development of riparian vegetation

The development of diverse environmental conditions in response to hydro-morphic interactions and the consequences biodiversity has been amply demonstrated with respect to the riparian zone. Although the patterns are complicated by greater physical diversity within the vegetation quadrats at the bank toe than between bank positions it is still evident that more beta diversity exists between quadrats positioned at the toe of the bank that between those positioned at the top, even after levels of alpha diversity have been accounted for.

The hydromorphological processes shaping the streambed (and to some extent lower bank) are rapid, with both low and high flow events contributing to the diversity. The 'disturbance' processes, as referred to by (Brown, 2007), play a key role in the development of the ecosystem. Hydromorphic development of the upper bank and floodplain, on the other hand, occurs only during high flows. Other formative processes (wind erosion, animal interactions) known to occur in this area (Johansson et al., 1996; Bond et al., 2000) are much weaker and slower than the potential influence of hydrology. Constructing a 'pre-existing' morphology or topology in these areas is more important in order to achieve ecological development within an acceptable timeframe. A morphologically simple floodplain will reduce the effectiveness of processes relating to lateral connectivity. Variability in strength of these connections will influence variability in in-channel characteristics longitudinally. In this respect variable inundation will also be important to introduce areas of the floodplain to different disturbance regimes. In a natural channel this would result from variable bank height and channel volume.

Although not formally investigated anecdotal evidence suggests that placement of geotextile served to stabilise the riverbanks in the absence of vegetation cover. Channel widening as a result of erosion was observed at locations along the realignment where geotextile was not used. The slow rate at which vegetation cover was shown to develop means that without geotextile erosion and excessive sediment input into the channel would be an issue. However, geotextile represents a significant project cost. It is also possible that the geotextile may have limited vegetation development by shading and drying the substrate where it is not carefully installed reflecting similar findings in previous studies where geotextile was only loosely anchored (Gilvear & Bradley, 1997). An alternative solution that should be considered at the planning stage would be management to encourage significant vegetation cover prior to the introduction of flow.

Establishment of vegetation

The value of seeding the riparian zone appears to have been limited as indicated by the low levels of vegetation diversity (principally richness and cover but also beta diversity) during the first year. This is likely to be due to floods scouring the seeds from the river banks, limited propagule trapping potential of the relatively uniform banks and conditions unsuitable to establishment of seedlings. In subsequent years both the physical habitat diversity and biodiversity were seen to increase. Analysis showed the tow to be closely related. An important component of the biodiversity is quadrat beta diversity. Greater beta diversity was strongly related to physical habitat variability in 2007 when physiochemical parameters within the quadrats were recorded.

It is clear from the patterns of plant colonisation that habitat plays a more important role than distance to source populations. This is likely to be because of the very efficient dispersal strategies adopted by plants. There is still a possibility that barriers exist to the recovery of specific species that relate to the dispersal from species pools but enough species reach the diversion to demonstrate a clear pattern of habitat suitability

It is clear from the ordinations of the vegetation data for each year that relatively distinct groups of species colonised the top, mid and bank toe areas. The pattern develops through time and seems to be strongest in 2007. It seems likely that the pattern would have been even clearer had smaller quadrats been used: At the time of surveying a distinct banding is visible within many of the quadrats located at the toe of the bank. As a result of the chosen quadrat size being too large the bottom quadrats included many mid-bank species. It is possible that the pattern is reduced in 2008 because of the developing dissimilarity between quadrats at the same bank positions (Figure 7.9). As the channel developed it was expected that the factors controlling the colonisation and establishment success of species would change. The results would seem to suggest that the influence of bank position may be strongest in the first few years of development. Bank position was the most significant variable tested.

Not so clear in the diagram is the influence of distance from the source populations. It has been suggested that this may be a significant barrier to the colonisation success of species (Bond & Lake, 2003; Rahel, 2007). Testing the influence of distance was done using direct gradient analysis. The results revealed that distance is an important factor. More specifically the 'remoteness' (physical distance to either the up- or downstream species pool) rather than the 'chainage; (downstream distance from the top of the diversion) of sites was the most significant environmental variable after bank position for the first few years. Chainage, unsignificant in the first year accounts for an increasing proportion of the variability through time.

Dispersal strategy

The importance of dispersal strategies behind the influence of remoteness and chainage were further investigated. Seed weights were taken from Grime et al., (2007) were correlated against two diversity indices. Number of sites present as a surrogate for colonisation success and cover as a surrogate for establishment success.

A possible explanation for Lolium prenne and Trifolium repens being the most successful colonisers during the first year of development is their economic importance and wide spread use as forage crops. The vast tonnage of both sown annually in the UK (Grime et al 2008) may explain their dominance within the diversion especially considering the lack of evidence for a persistent seed bank for L.perenne. Together with Holcus lanatus they stay in the top five most successful colonisers through all surveys.

The barrier to dispersal imposed by distance is clear in the results with fewer individual organisms successfully colonising the more remote areas. As well as the fall off in the number of colonists with increasing dispersal distance it is possible that habitat condition is poorer in more remote regions making establishment less likely.

Habitat condition is likely to become increasingly important as more dispersal pathways become available from natural reaches and with time from sources that develop within the realignment.

The objectives of this section were to: establish a link between the instatement of geomorphological processes and the development of physical habitat diversity along the river banks; provide supporting evidence for the link between physical habitat diversity and the development biodiversity - specifically relating to ecosystem integrity.

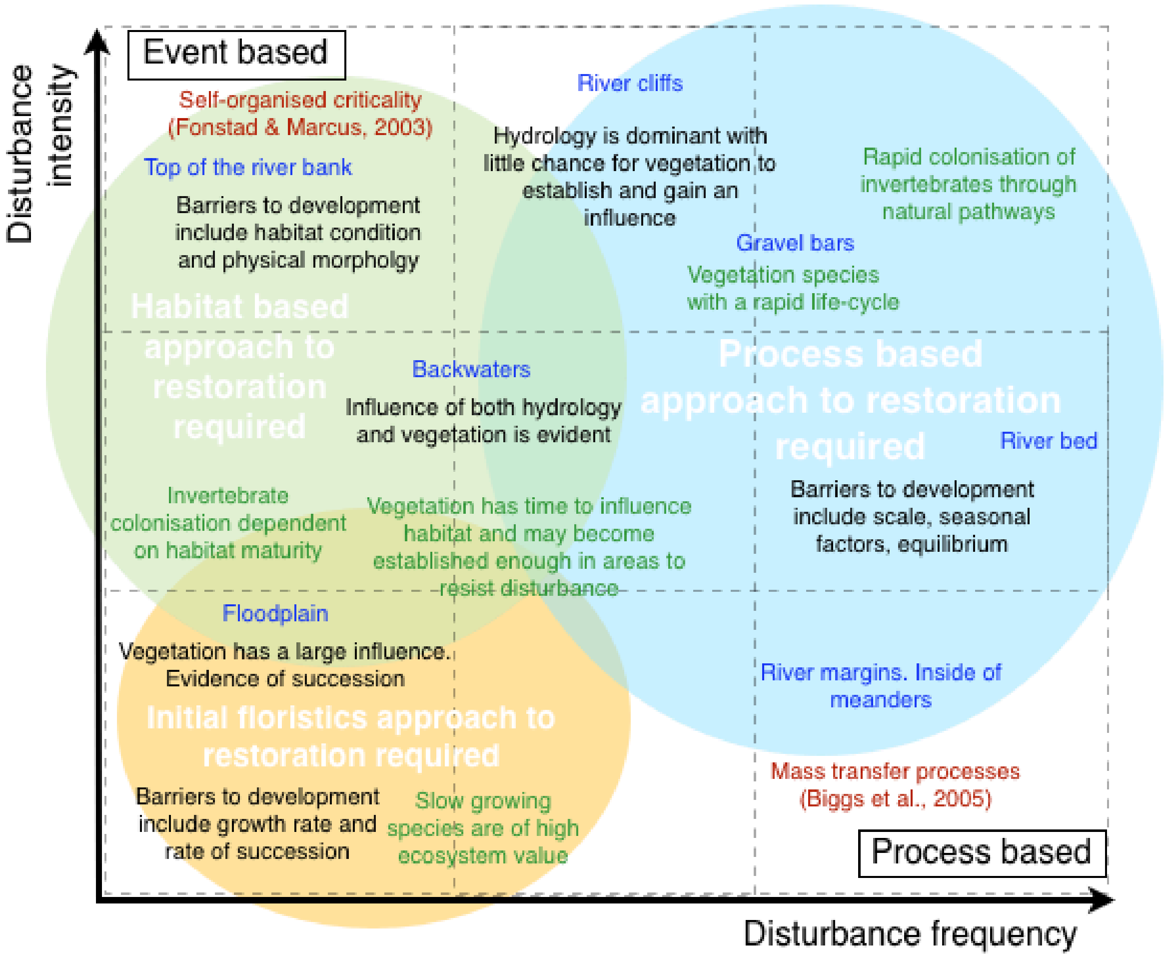


Figure 10.7. Conceptual model indicating the importance of disturbance frequency and disturbance intensity to different stream habitats and the implications for restoration strategies adopted.

CHAPTER 11

CONCLUSION

8.1 Introduction

Most rivers have evolved though years of interaction with their landscapes into highly complex systems. However in recent history these systems have frequently been degraded by being regulated and simplified for agricultural, industrial and commercial benefit. Current recognition of the cultural and ecological value of naturally functioning rivers has driven the restoration of degraded systems and is demanding a more ecologically sensitive approach to new resource development involving rivers. In a growing number of projects this has necessitated the construction of channels with full ecological function with respect to both geomorphological and biological processes, processes that are critical to the health and long-term sustainability of streams and rivers. The questions addressed by this research relate to: (i) The possibility and practicality of achieving full ecological function in a man-made channel. (ii) How quickly processes operate and whether they have the potential to drive the ecosystem development. (iii) The influence of barriers to development and biological adaptations to surmount them. (iv) The role river design can play in improving the overall chances of long-term project success.

Six principal hypotheses were considered. These are presented below together with the key findings related to each one.

1. From a homogeneous starting point, the 3km of engineered channel will develop a diverse and heterogeneous form comparable to a natural sinuous pool riffle river.

ACCEPTED: Despite the length of the realignment, point bars, glides and riffles formed within the first few months. The channel deepened at the outside of meander apexes gradual over successive floods. Sediment sorting occurred and new sources of sediment input developed. The degree of diversity also changed along the length of the diversion with areas of different disturbance frequency and intensity. Development of the riverbanks was less pronounced and many stretches remained morphologically simple and obviously engineered.

2. The influence of design specifications, included to promote development towards a natural form and propagate ecosystem development, will be evident in the pattern of geomorphological development.

ACCEPTED: Mobile gravel bed allowed significant morphological development. Early development of point bars occurred prior to significant scour on the outside of meander bends highlighting the importance of sediment mobility. This was confirmed on the Beoch Lane where gravel size was not suited to the energy conditions. Meanders directly contributed to sediment sorting creating ecologically distinct conditions. Changes in bed gradient created produced sediment transport promoting patterns of erosion and deposition

3. There are predictable environmental barriers to colonisation, which affect the dispersal and establishment of macroinvertebrate colonists. These must be considered when designing ecosystem architecture projects.

ACCEPTED: Availability of dispersal pathways was a barrier to development with drift acting as the only way for individuals to migrate into the realignment for the first year of development. Related to this drifting distance proved to be a species-specific barrier with more distant sites being more impoverished. Increases in abundance and changes in the community composition at downstream sites indicated the influence of habitat on development at the 12 month mark. Mossy boulder placement showed that habitat condition is a barrier to the establishment of individuals with specific habitat requirements, and also provided evidence for the potential to enhance the rate of development.

4. There will be specific morphological and behavioural traits that have evolved in some species to overcome environmental barriers to colonisation affecting dispersal and establishment. Species possessing these traits will be superior colonists.

ACCEPTED: Colonist communities expressed distinct biological traits that were less prevalent in reference communities. Some traits appeared to relate to the effective use of dispersal pathways. A ruderal strategy of investment in large numbers of rapidly developing progeny was not an effective strategy at early stages of development because recruitment was not initially from within the realignment. Species that took an early hold in the diversion were those able to rapidly disperse and survive in the diversion. Colonists shared had short life cycles and eggs resistant to damage, which would enable early and effective dispersal of eggs into the realignment. Trait expression also indicated a relative absence of small bodied species, which are typically ruderal, compared to the reference reaches.

5. Environmental barriers to the dispersal of plant propagules will be evident in the rate and pattern of plant colonisation, with those species better adapted to using specific dispersal pathways colonising more rapidly and extensively.

UNCLEAR: It was difficult to tease apart dispersal and establishment patterns. Poor habitat quality is likely to have limited the number of species that were able to establish. However a relationship between remoteness (from up or downstream sites) rather than a linear relationship with distance from upstream suggest that wind or zoo dispersal of seeds is a more significant dispersal pathway than hydrochory. The majority of species found along the banks were generalist dispersers in the first year but wind and zoo dispersed species made up a significant proportion of species in following years.

6. Environmental barriers exist to the establishment of plant propagules will be evident in the rate and pattern of plant colonisation following patterns of habitat development and suitability as the ecosystem develops

ACCEPTED: The rapid differentiation of the environment at the top and bottom of the riverbank clearly demonstrated the effect that habitat condition can have on the establishment of seedlings. Communities were taxonomically distinct by 2005 with almost zero overlap of sites. Habitat preference scores for the communities, weighted for species cover indicated that species traits matched the habitat type. Silt deposition significantly added to habitat diversity and was directly related to increased biological diversity.

8.2 Implications of findings

The long history of river engineering has a strong influence over the design of river realignments as the Nith demonstrates. The role of design remains with engineers, as has been the tradition for many of years. The consequence of this status quo, a focus on achieving hydraulic efficiency, is responsible for the poor ecological performance of many modern ecosystem architecture projects. This research has shown that even in high energy channels, if consideration is given to the geomorphic processes, the considerable cost of hard forms of channel engineering and revetment are at best unnecessary, and can jeopardise the long term sustainability of the ecosystem. Whilst river restoration has demonstrated the benefits of embracing interdisciplinary discussions, the design of river channels where restoration is not the primary goal remains largely entrenched in tradition. The improvement of future river designs is dependent on a move from a focus on hydrology an unhindered hydraulics to an approach based on strong input from geomorphologists and ecologists in addition to hydrologists.

In this regard, constraints on flow conveyance are as important to consider as continuity of flow conveyance. A connection is made here between the lack of narrow and wide channel sections that reduce local longitudinal conveyance under specific conditions and the limited lateral connectivity to the floodplain. Lateral connections that positively influence community structure, and are so important to long-term ecosystem health. Likewise the absence of natural weir and dam structures, e.g. large boulders and large woody debris that stall flow, form pools and encourage overspill on to the floodplain, similarly places limitations on the potential of laterally acting processes to enhance eoclogical diversity.

The restoration of meanders in central to many restoration projects as the represent a more natural channel form. We investigated the effect this had in producing a diverse cross-sectional form, on flow variability and on structuring the riverbed habitat. We were then able to link this to the development of distinct macroinvertebrate fauna.

Several areas of the research support the argument that management techniques and design considerations can serve to enhance the development or 'recovery' of engineered channels, but that natural processes are key to long-term sustainability and in many cases can be relied upon to drive habitat development.

Whilst hydrological processes would be sufficient to develop or restore river habitat, given a long enough time frame and a pool of potential colonists, it is evident from this research that it is possible to improve the potential for ecological development. The role of river channel design and management is to speed up this development so that a functional stream ecosystem can be achieved within an acceptable time. The designs adopted has often been based on theoretical principals and despite the existence of restoration ecology as a science for several decades, little data has been collected in support of the approaches.

Research into the 2000 and 2004 realignments of the River Nith have given a clear picture of the development that can be expected in a large-scale diversion. The project is unique within the UK with respect to size and location and is hydromorphologically distinct from lowland river architecture projects where much of previous research has been focussed. Results have highlighted the central role that hydromorphological processes play in the development of ecological units, or geomorphological units around which ecological diversity is promoted. Conclusions apply to high-energy environments where the engineered or modified channel sits within a wider biodiverse setting. However many of the findings are relevant to other geographical and geophysical riverscapes. It must be recognised that large-scale river architecture projects develop within a different time frame to small-scale projects. Many demonstration projects where scientific data is more often collected are large-scale projects. As such data for the time that more typical smaller restored river ecosystems take to develop may be overly pessimistic. However, this errs on the side of caution, sitting well with the precautionary principle advocated by the EU Water Framework Directive (Chave, 2001)

The findings should provide reassurance to practitioners, that heavy engineering is unnecessary in river architecture projects. Strategies appropriate to ensuring the integrity of drainage infrastructure often recommended by engineering firms represents a poor use of budget that would be better directed towards natural erosion barriers and habitat enhancement techniques.

8.3 Management recommendations for the River Nith at House of Water

The future of the house of water site will now depend on the habitat management that is put into practice. The results from this research suggest that the following management options would maximise the development of ecological diversity

i) Focus should be on the development of the stream corridor. The value of this to the wider environment would be manifold

ii) The stream would benefit from the input of large woody debris. To be a sustainable feature of the channel this would require the creation of a diverse tree line.

8.4 Recommendation for developing future monitoring strategies

It is argued here that this research is relevant to the design and management of future projects, and it is clear that further studies are required. Whilst it is widely acknowledged that investigating the outcome of projects is critical to the future of river restoration ecology, whether or not the current techniques and tools are adequate for guiding monitoring is rarely questioned. Recommendations for monitoring and analysis of data are also included and are considered equally important.

Future restoration projects will of course not benefit from past experience unless results are documented and readily available. Calls for even the most informal of evaluations have been widespread in the literature for a number of years (Michener, 1997; Lake 2000)

New indices need to be generated for assessment of river heterogeneity at a number of scales. The index will have to be based on appropriate types of data. It needs to be easily collectable at relatively low cost. At the same time it must lack ambiguity, have high sensitivity to the ecosystem responses and ideally offer the ability to incorporate hypothesis testing.

The use of macroinvertebrate species traits and community structure provide a sound base for the assessment of ecological condition relvealing information about ecosystem development that is missed by traditional diversity indices. A 1-10 traits scale index similar to the BMWP index for pollution, could be used for the assessment of ecosystem development after major engineering disturbances.

The development of physical diversity visible in the profile graphs was not summarised effectively by the various indices recommended in the literature. The GPS data was collected at a scale that summarised the variability present at the time of sampling. As variability increased with time more data points were collected in successive surveys. However, indices proved not to be robust to this sampling strategy with the increased number of sampling points having a strong influence over the index scores. Reducing the data of later years to match the sampling resolution of the early surveys, effectively increasing the scale at which diversity was assessed, showed that diversity did not increase at this scale (10-15m resolution) but missed the development of diversity at a smaller scale that corresponds more closely to the development of macrohabitat.

The creation of artificial pool and riffle features also helped to hide changes since the indices were more far more sensitive to pool features than the more complex topography of intermediate sections of stream bed. Development of a more variable bed topography observable in the thalweg profiles were not detected. The creation and in-filling of pools were approximately equal in number. The thalweg profile also failed to capture the diversity introduced by the formation of gravel bars as, by definition, the thalweg circumvented these features. To track this level of change a more productive approach would have involved using differential GPS to document individual fluvial features in three dimensions. This would have been a time intensive strategy further complicated by uncertainty as to exactly when and where new features were going to develop. Instead, the qualitative equivalent, fixed point photography was used.

8.5 Recommendations for stream architecture projects

Natural river processes must be restored to achieve rapid development of stream morphology and so that management and repair is not continually needed to maintain the river condition. The correct gravel mix is crucial in this respect.

Geomorphically inappropriate placement of pools at construction to provide fish habitat does not harm the river or impede ecological development under conditions of high sediment transport but represent an unneccessary project expense. Pool creation at meander bends and development of slack water at the tail end of large point bars represent sufficient provision of fish habitat.

The direct links between the health of the riparian zone and hydraulic interactions has important implications for project design. Reducing bank slope at appropriate locations will increase the area of interaction. Additionally varying the channel form has the potential to introduce flow diversity and associated variability in sediment conveyance.

As a final point, it should be noted that realignment projects should aim to achieve an ecological condition greater than that observed at ‘reference reaches’. Virtually all European river corridors have been substantially regulated and are likely to be less heterogeneous and more stable than they would be in the natural state (Ward et al., 2001). It is only by taking the opportunity to allow and study the response of river channels to unhindered fluvial disturbance that an idea of the true ‘reference condition’ will be attainable. Allowing disturbances to shape rivers at the landscape level has demonstrated the high levels of temporal and spatial heterogeneity and the strong longitudinal and lateral connections achieved by large unregulated rivers across Europe (e.g. Talgiamento). A similar approach taken on smaller UK rivers may yield insights in to the management of rivers for conservation and nature value.

8.6 Recommendations for future research

i) An area of weak scientific understanding only addressed in part by this research concerns the importance of habitat diversity at early stages of development. One possibility would be to compare colonising communities to dispersal communities by directly measuring individuals at active stages of dispersal. This could include analysis of seston and aerial adult populations. Efforts to address this question as part of this research failed because of the difficulty in anchoring drift nets to the gravel bed of the river.

ii) Replicating the research under a range of different channel types and realignments with different design specifications.

iii) Greater experimental work on new realignments. Research could target techniques with the potential enhance development such as the incorporation of organic matter into the minerogenic component of the bed material in promoting invertebrate colonisation and ecosystem development of the incorporation of large woody debris into channel design to diversify patterns of sediment transport and habitat types.

iv) Long term monitoring of development on realignments can contribute significantly to restoration ecology and should be perused whenever possible.

8 References

Adams, W.M. & Perrow, M.R. (1999) Scientific and institutional constraints on the restoration of European floodplains. *Floodplains : Interdisciplinary Approaches***,** 89-97.

Adams, W.M., Perrow, M.R. & Carpenter, A. (2004) Conservatives and champions: river managers and the river restoration discourse in the United Kingdom. *Environment and Planning A,* **36,** 1929-1942.

Alexander, G.G. & Allan, J.D. (2006) Stream restoration in the Upper Midwest, USA. *Restoration Ecology,* **14,** 595-604.

Alexander, G.G. & Allan, J.D. (2007) Ecological success in stream restoration: Case studies from the midwestern United States. *Environmental Management,* **40,** 245-255.

Allan, J. & Johnson, L. (1997) Catchment-scale analysis of aquatic ecosystems. *Freshwater Biology,* **37,** 107-111.

Andel, J. & Aronson, J. (2006) *Restoration Ecology,* Blackwell Science, Oxford.

Andersen, H.E. & Svendsen, L.M. (1997) Suspended sediment and total phosphorus transport in a major Danish river: methods and estimation of the effects of a coming major restoration. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **7,** 265-276.

Anderson, R.J., Bledsoe, B.R. & Hession, W.C. (2004) Width of streams and rivers in response to vegetation, bank material, and other factors. *Journal of the American Water Resources Association,* **40,** 1159-1172.

Andersson, E. & Nilsson, C. (2002) Temporal variation in the drift of plant litter and propagules in a small boreal river. *Freshwater Biology,* **47,** 1674-1684.

Andersson, E., Nilsson, C. & Johansson, M.E. (2000a) Effects of river fragmentation on plant dispersal and riparian flora. *Regulated Rivers-Research & Management,* **16,** 83-89.

Andersson, E., Nilsson, C. & Johansson, M.E. (2000b) Plant dispersal in boreal rivers and its relation to the diversity of riparian flora. *Journal of Biogeography,* **27,** 1095-1106.

Annoni, P., Saccardo, I., Gentili, G. & Guzzi, L. (1997) A multivariate model to relate hydrological, chemical and biological parameters to salmonid biomass in Italian Alpine rivers. *Fisheries Management and Ecology,* **4,** 439-452.

Armitage, P.D., Bowes, M.J. & Vincent, H.M. (2007) Long-term changes in macroinvertebrate communities of a heavy metal polluted stream: The river nent (Cumbria, UK) after 28 years. *River Research and Applications,* **23,** 997-1015.

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,** 143-170.

Arscott, D.B., Tockner, K. & Ward, J.V. (2000) Aquatic habitat diversity along the corridor of an Alpine floodplain river (Fiume Tagliamento, Italy). *Archiv fur Hydrobiologie,* **149,** 679-704.

Asaf, L., Negaoker, N., Tal, A., Laronne, J. & Al Khateeb, N. (2007) Transboundary stream restoration in Israel and the Palestinian authority. *Integrated Water Resources Management and Security in the Middle East***,** 285-295.

Aspetsberger, F., Huber, F., Kargl, S., Scharinger, B., Peduzzi, P. & Hein, T. (2002) Particulate organic matter dynamics in a river floodplain system: impact of hydrological connectivity. *Archiv fur Hydrobiologie,* **156,** 23-42.

Asselman, N.E.M. (1999) The use of floodplain sedimentation measurements to evaluate the effects of river restoration works. *Floodplains : Interdisciplinary Approaches***,** 111-122.

Baattrup-Pedersen, A., Friberg, N., Larsen, S.E. & Riis, T. (2005) The influence of channelisation on riparian plant assemblages. *Freshwater Biology,* **50,** 1248-1261.

Badcock, R.M. (1949) STUDIES IN STREAM LIFE IN TRIBUTARIES OF THE WELSH DEE. *Journal of Animal Ecology,* **18,** 193-208.

Bain, M.B. & Meixler, M.S. (2008) A target fish community to guide river restoration. *River Research and Applications,* **24,** 453-458.

Bakker, J.P. & Berendse, F. (1999) Constraints in the restoration of ecological diversity in grassland and heathland communities. *Trends in Ecology & Evolution,* **14,** 63-68.

Baldigo, B.P. & Warren, D.R. (2008) Detecting the response of fish assemblages to stream restoration: Effects of different sampling designs. *North American Journal of Fisheries Management,* **28,** 919-934.

Barbour, M.T., Poff, N.L., Norris, R.H. & Allan, J.D. (2008) Perspective: communicating our science to influence public policy. *Journal of the North American Benthological Society,* **27,** 562-569.

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

Beechie, T., Pess, G. & Roni, P. (2008) Setting river restoration priorities: a review of approaches and a general protocol for identifying and prioritizing actions. *North American Journal of Fisheries Management,* **28,** 891-905.

Beechie, T.J., Pess, G.R., Pollock, M.N., Ruckelshaus, M.H. & Roni, P. (2009) Restoring Rivers in the Twenty-First Century: Science Challenges in a Management Context. *Future of Fisheries Science in North America,* **31,** 697-717.

Bennett, S.J. (2004) Effects of emergent riparian vegetation on spatially averaged and turbulent flow within an experimental channel. *Riparian Vegetation and Fluvial Geomorphology,* **8,** 29-41.

Bennett, S.J., Pirim, T. & Barkdoll, B.D. (2002) Using simulated emergent vegetation to alter stream flow direction within a straight experimental channel. *Geomorphology,* **44,** 115-126.

Bennett, S.J., Wu, W.M., Alonso, C.V. & Wang, S.S.Y. (2008) Modeling fluvial response to in-stream woody vegetation: implications for stream corridor restoration. *Earth Surface Processes and Landforms,* **33,** 890-909.

Bergerot, B., Lasne, E., Vigneron, T. & Laffaille, P. (2008) Prioritization of fish assemblages with a view to conservation and restoration on a large scale European basin, the Loire (France). *Biodiversity and Conservation,* **17,** 2247-2262.

Bernhardt, E.S. & Palmer, M.A. (2007) Restoring streams in an urbanizing world. *Freshwater Biology,* **52,** 738-751.

Bernhardt, E.S., Palmer, M.A., 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) Ecology - Synthesizing US river restoration efforts. *Science,* **308,** 636-637.

Bernhardt, E.S., Sudduth, E.B., Palmer, M.A., Allan, J.D., Meyer, J.L., Alexander, G., Follastad-Shah, J., Hassett, B., Jenkinson, R., Lave, R., Rumps, J. & Pagano, L. (2007a) Restoring Rivers One Reach at a Time: Results from a Survey of U.S. River Restoration Practitioners. *Restoration Ecology,* **15,** 482-493.

Bernhardt, E.S., Sudduth, E.B., Palmer, M.A., Allan, J.D., Meyer, J.L., Alexander, G., Follastad-Shah, J., Hassett, B., Jenkinson, R., Lave, R., Rumps, J. & Pagano, L. (2007b) Restoring rivers one reach at a time: Results from a survey of US river restoration practitioners. *Restoration Ecology,* **15,** 482-493.

Bertoldi, W., Zanoni, L. & Tubino, M. (2009) Planform dynamics of braided streams. *Earth Surface Processes and Landforms,* **34,** 547-557.

Beschta, R.L. & Platts, W.S. (1986) MORPHOLOGICAL FEATURES OF SMALL STREAMS: SIGNIFICANCE AND FUNCTION1. *Journal of the American Water Resources Association,* **22,** 369-379.

Bhuiyan, A.B.M.F. & Hey, R.D. (2001) Instream J-vane for bank protection and river restoration. *Hydraulics of Rivers Water Works and Machinery, Vol Ii, Theme D, Proceedings - 21St Century: the New Era for Hydraulic Research and Its Applications***,** 161-166.

Biggs, B.J.F., Nikora, V.I. & Snelder, T.H. (2005) Linking scales of flow variability to lotic ecosystem structure and function. *River Research and Applications,* **21,** 283-298.

Biggs, J., Corfield, A., Gron, P., Hansen, H.O., Walker, D., Whitfield, M. & Williams, P. (1998) Restoration of the rivers Brede, Cole and Skerne: a joint Danish and British EU-LIFE demonstration project, V - Short-term impacts on the conservation value of aquatic macroinvertebrate and macrophyte assemblages. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **8,** 241-255.

Bilton, D.T., Mcabendroth, L.O.U.I., Bedford, A.L.A.N. & Ramsay, P.M. (2006) How wide to cast the net? Cross-taxon congruence of species richness, community similarity and indicator taxa in ponds. *Freshwater Biology,* **51,** 578-590.

Binder, W. (2006) Case study: River restoration projects along the Isar (Germany). *River Flow 2006, Vols 1 and 2***,** 51-60.

Bird, G.A. & Hynes, H.B.N. (1981) MOVEMENT OF IMMATURE AQUATIC INSECTS IN A LOTIC HABITAT. *Hydrobiologia,* **77,** 103-112.

Biron, P.M., Robson, C., Lapointe, M.F. & Gaskin, S.J. (2004) Deflector designs for fish habitat restoration. *Environmental Management,* **33,** 25-35.

Bishop, J.E. & Hynes, H.B.N. (1969) UPSTREAM MOVEMENTS OF BENTHIC INVERTEBRATES IN SPEED RIVER ONTARIO. *Journal of the Fisheries Research Board of Canada,* **26,** 279-&.

Biswas, A.K. (1986) Water Development Policies. *Resources Policy,* **12,** 290-292.

Blakely, T.J., Harding, J.S., Mcintosh, A.R. & Winterbourn, M.J. (2006) Barriers to the recovery of aquatic insect communities in urban streams. *Freshwater Biology,* **51,** 1634-1645.

Bocchiola, D., Rulli, M.C. & Rosso, R. (2006) Flume experiments on wood entrainment in rivers. *Advances in Water Resources,* **29,** 1182-1195.

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.

Boedeltje, G., Bakker, J.P., Bekker, R.M., Van Groenendael, J.M. & Soesbergen, M. (2003) Plant dispersal in a lowland stream in relation to occurrence and three specific life-history traits of the species in the species pool. *Journal of Ecology,* **91,** 855-866.

Boedeltje, G., Bakker, J.P., Ten Brinke, A., Van Groenendael, J.M. & Soesbergen, M. (2004) Dispersal phenology of hydrochorous plants in relation to discharge, seed release time and buoyancy of seeds: the flood pulse concept supported. *Journal of Ecology,* **92,** 786-796.

Bond, N.R. & Lake, P.S. (2003) Local habitat restoration in streams: Constraints on the effectiveness of restoration for stream biota. *Ecological Management & Restoration,* **4,** 193-198.

Bond, N.R., Lake, P.S. & Arthington, A.H. (2008) The impacts of drought on freshwater ecosystems: an Australian perspective. *Hydrobiologia,* **600,** 3-16.

Boon, P.J. (1998) River restoration in five dimensions. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **8,** 257-264.

Boulton, A.J. (2007) Hyporheic rehabilitation in rivers: restoring vertical connectivity. *Freshwater Biology,* **52,** 632-650.

Bracher, R. (1929) The ecology of the Avon banks at Bristol. *Journal of Ecology,* **17,** 35-81.

Bradshaw, A.D. (1984) Ecological Principles and Land Reclamation Practice. *Landscape Planning,* **11,** 35-48.

Brinkhurst, R.O. (1993) Future directions in freshwater biomonitoring using benthic macroinvertebrates. *Freshwater biomonitoring and benthic macroinvertebrates***,** 442-460.

Brisbois, M.C., Jamieson, R., Gordon, R., Stratton, G. & Madani, A. (2008) Stream ecosystem health in rural mixed land-use watersheds. *Journal of Environmental Engineering and Science,* **7,** 439-452.

Brookes, A. (1995a) Challenges and Objectives for Geomorphology in Uk River Management. *Earth Surface Processes and Landforms,* **20,** 593-610.

Brookes, A. (1995b) THE IMPORTANCE OF HIGH FLOWS FOR RIVERINE ENVIRONMENTS. *Ecological Basis for River Management***,** 33-49.

Brooks, S.S. & Lake, P.S. (2007) River restoration in Victoria, Australia: Change is in the wind, and none too soon. *Restoration Ecology,* **15,** 584-591.

Brown, A.G. (2002) Learning from the past: palaeohydrology and palaeoecology. *Freshwater Biology,* **47,** 817-829.

Brown, A.V. & Brussock, P.P. (1991) COMPARISONS OF BENTHIC INVERTEBRATES BETWEEN RIFFLES AND POOLS. *Hydrobiologia,* **220,** 99-108.

Brown, B.L. (2007) Habitat heterogeneity and disturbance influence patterns of community temporal variability in a small temperate stream. *Hydrobiologia,* **586,** 93-106.

Brown, C., Pemberton, C., Birkhead, A., Bok, A., Boucher, C., Dollar, E., Harding, W., Kamish, W., King, J., Paxton, B. & Ractliffe, S. (2006) In support of water-resource planning - highlighting key management issues using DRIFT: A case study. *Water Sa,* **32,** 181-191.

Brown, R.A. & Pasternack, G.B. (2008) Engineered channel controls limiting spawning habitat rehabilitation success on regulated gravel-bed rivers. *Geomorphology,* **97,** 631-654.

Brydon, D. (2007) *Linking meander form to variability in flow*, University of Stirling, Stirling.

Buckley, M. & Haddad, B.M. (2006) Socially strategic ecological restoration: A game-theoretic analysis shortened: Socially strategic restoration. *Environmental Management,* **38,** 48-61.

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,** 889-907.

Burke, S.M. & Mitchell, N. (2007) People as Ecological Participants in Ecological Restoration. *Restoration Ecology,* **15,** 348-350.

Buzas, M.A. & Culver, S.J. (1999) Understanding regional species diversity through the log series distribution of occurrences. BIODIVERSITY RESEARCH. *Diversity and Distributions,* **5,** 187-195.

Cabin, R.J. (2007a) Science and Restoration Under a Big, Demon Haunted Tent: Reply to Giardina et al. (2007). *Restoration Ecology,* **15,** 377-381.

Cabin, R.J. (2007b) Science-Driven Restoration: A Square Grid on a Round Earth? *Restoration Ecology,* **15,** 1-7.

Cairns, J., Jr. & Pratt, J.R. (1993) A history of biological monitoring using benthic macroinvertebrates. *Freshwater biomonitoring and benthic macroinvertebrates***,** 10-27.

Campbell, R.N.B. (1985) COMPARISON OF THE DRIFT OF LIVE AND DEAD BAETIS NYMPHS IN A WEAKENING WATER CURRENT. *Hydrobiologia,* **126,** 229-236.

Cao, Y. & Hawkins, C.P. (2005) Simulating biological impairment to evaluate the accuracy of ecological indicators. *Journal of Applied Ecology,* **42,** 954-965.

Cao, Y., Hawkins, C.P. & Storey, A.W. (2005) A method for measuring the comparability of different sampling methods used in biological surveys: implications for data integration and synthesis. *Freshwater Biology,* **50,** 1105-1115.

Cao, Y., Hawkins, C.P. & Vinson, M.R. (2003) Measuring and controlling data quality in biological assemblage surveys with special reference to stream benthic macroinvertebrates. *Freshwater Biology,* **48,** 1898-1911.

Cariboni, J., Gatelli, D., Liska, R. & Saltelli, A. (2007) The role of sensitivity analysis in ecological modelling. *Ecological Modelling,* **203,** 167-182.

Casas, J.J., Zamora-Munoz, C., Archila, F. & Ba-Tercedor, J. (2000) The effect of a headwater dam on the use of leaf bags by invertebrate communities. *Regulated Rivers-Research & Management,* **16,** 577-591.

Cassey, P., Blackburn, T.M., Lockwood, J.L. & Sax, D.F. (2006) A stochastic model for integrating changes in species richness and community similarity across spatial scales. *Oikos,* **115,** 207-218.

Chin, A., Anderson, S., Collison, A., Ellis-Sugai, B.J., Haltiner, J.P., Hogervorst, J.B., Kondolf, G.M., O'hirok, L.S., Purcell, A.H., Riley, A.L. & Wohl, E. (2009) Linking Theory and Practice for Restoration of Step-Pool Streams. *Environmental Management,* **43,** 645-661.

Chin, A., Daniels, M.D., Urban, M.A., Piegay, H., Gregory, K.J., Bigler, W., Butt, A.Z., Grable, J.L., Gregory, S.V., Lafrenz, M., Laurencio, L.R. & Wohl, E. (2008) Perceptions of wood in rivers and challenges for stream restoration in the United States. *Environmental Management,* **41,** 893-903.

Choi, Y.D. (2007) Restoration Ecology to the Future: A Call for New Paradigm. *Restoration Ecology,* **15,** 351-353.

Clark, B.T. (2004) Agenda setting and issue dynamics: Dam breaching on the lower Snake River. *Society & Natural Resources,* **17,** 599-609.

Clark, B.T. (2009) River Restoration in the American West: Assessing Variation in the Outcomes of Policy Change. *Society & Natural Resources,* **22,** 401-416.

Clarke, K.R. & Warwick, R.M. (1998) A taxonomic distinctness index and its statistical properties. *Journal of Applied Ecology,* **35,** 523-531.

Clarke, R.T., Furse, M.T., Gunn, R.J.M., Winder, J.M. & Wright, J.F. (2002) Sampling variation in macroinvertebrate data and implications for river quality indices. *Freshwater Biology,* **47,** 1735-1751.

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,** 439-450.

Clifford, N.J. (2007) River restoration: Paradigms, paradoxes and the urban dimension. *Sustainable and Safe Water Supplies,* **7,** 57-68.

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.

Collins, B.D. & Montgomery, D.R. (2002) Forest development, wood jams, and restoration of floodplain rivers in the Puget Lowland, Washington. *Restoration Ecology,* **10,** 237-247.

Conquest, L.L. (2000) Analysis and interpretation of ecological field data using BACI designs: Discussion. *Journal of Agricultural Biological and Environmental Statistics,* **5,** 293-296.

Cooper, S.D. & Barmuta, L.A. (1993) Field experiments in biomonitoring. *Freshwater biomonitoring and benthic macroinvertebrates***,** 399-441.

Corkum, L.D. & Clifford, H.F. (1980) THE IMPORTANCE OF SPECIES ASSOCIATIONS AND SUBSTRATE TYPES TO BEHAVIORAL DRIFT. *Flannagan, J. F. and K. E. Marshall (Ed.). Advances in Ephemeroptera Biology; 3rd International Conference, Winnipeg, Manit., Canada, July 4-10, 1979. Xiii+552p. Plenum Press: New York, N.Y., USA; London, England. Illus. Maps***,** P331-342.

Corkum, L.D., Pointing, P.J. & Ciborowski, J.J.H. (1977) INFLUENCE OF CURRENT VELOCITY AND SUBSTRATE ON DISTRIBUTION AND DRIFT OF 2 SPECIES OF MAYFLIES (EPHEMEROPTERA). *Canadian Journal of Zoology-Revue Canadienne De Zoologie,* **55,** 1970-1977.

Dangerfield, J.M., Pik, A.J., Britton, D.A.V.I., Holmes, A.N.D.R., Gillings, M.I.C.H., Oliver, I.A.N., Briscoe, D.A.V.I. & Beattie, A.J. (2003) Patterns of invertebrate biodiversity across a natural edge. *Austral Ecology,* **28,** 227-236.

Davies, B.R., Biggs, J., Williams, P.J., Lee, J.T. & Thompson, S. (2008) A comparison of the catchment sizes of rivers, streams, ponds, ditches and lakes: implications for protecting aquatic biodiversity in an agricultural landscape. *Hydrobiologia,* **597,** 7-17.

Death, R.G. (2002) Predicting invertebrate diversity from disturbance regimes in forest streams. *Oikos,* **97,** 18-30.

Death, R.G. (2006) Colonisation in New Zealand streams: predictable patterns or chance events? *New Zealand Natural Sciences,* **31,** 93-112.

Death, R.G. & Winterbourn, M.J. (1995) DIVERSITY PATTERNS IN STREAM BENTHIC INVERTEBRATE COMMUNITIES - THE INFLUENCE OF HABITAT STABILITY. *Ecology,* **76,** 1446-1460.

Doberstein, C.P., Karr, J.R. & Conquest, L.L. (2000) The effect of fixed-count subsampling on macroinvertebrate biomonitoring in small streams. *Freshwater Biology,* **44,** 355-371.

Doeg, T.J., Marchant, R., Douglas, M. & Lake, P.S. (1989) EXPERIMENTAL COLONIZATION OF SAND, GRAVEL AND STONES BY MACROINVERTEBRATES IN THE ACHERON RIVER, SOUTHEASTERN AUSTRALIA. *Freshwater Biology,* **22,** 57-64.

Downes, B.J., Bellgrove, A. & Street, J.L. (2005) Drifting or walking? Colonisation routes used by different instars and species of lotic, macroinvertebrate filter feeders. *Marine and Freshwater Research,* **56,** 815-824.

Downs, P.W. & Kondolf, G.M. (2002) Post-project appraisals in adaptive management of river channel restoration. *Environmental Management,* **29,** 477-496.

Downs, P.W. & Thorne, C.R. (1998) Design principles and suitability testing for rehabilitation in a flood defence channel: The River Idle, Nottinghamshire, UK. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **8,** 17-38.

Doyle, M.W., Shields, D., Boyd, K.F., Skidmore, P.B. & Dominick, D. (2007) Channel-forming discharge selection in river restoration design. *Journal of Hydraulic Engineering-Asce,* **133,** 831-837.

Dziock, F., Henle, K., Foeckler, F., Follner, K. & Scholz, M. (2006) Biological indicator systems in floodplains - a review. *International Review of Hydrobiology,* **91,** 271-291.

Eden, S., Tunstall, S.M. & Tapsell, S.M. (1999) Environmental restoration: environmental management or environmental threat. *Area,* **31,** 151-159.

Edgar, B., Schofield, N. & Campbell, A. (2001) Informing river management policies and programs with science. *Water Science and Technology,* **43,** 185-195.

Egglisha.Hj (1969) DISTRIBUTION OF BENTHIC INVERTEBRATES ON SUBSTRATA IN FAST-FLOWING STREAMS. *Journal of Animal Ecology,* **38,** 19-&.

Egglishaw, H.J. (1964) THE DISTRIBUTIONAL RELATIONSHIP BETWEEN THE BOTTOM FAUNA AND PLANT DETRITUS IN STREAMS. *Journal of Animal Ecology,* **33,** 463-476.

Eljabi, N. & Rousselle, J. (1987) A FLOOD DAMAGE MODEL FOR FLOODPLAIN STUDIES. *Water Resources Bulletin,* **23,** 179-187.

Elliott, J.M. (2002) Time spent in the drift by downstream-dispersing invertebrates in a Lake District stream. *Freshwater Biology,* **47,** 97-106.

Elliott, J.M. (2003) A comparative study of the dispersal of 10 species of stream invertebrates. *Freshwater Biology,* **48,** 1652-1668.

England, J., Skinner, K.S. & Carter, M.G. (2008) Monitoring, river restoration and the Water Framework Directive. *Water and Environment Journal,* **22,** 227-234.

Engstrom, J., Nilsson, C. & Jansson, R. (2009) Effects of stream restoration on dispersal of plant propagules. *Journal of Applied Ecology,* **46,** 397-405.

Everard, M. & Powell, A. (2002) Rivers as living systems. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **12,** 329-337.

Findlay, S.J. & Taylor, M.P. (2006) Why rehabilitate urban river systems? *Area,* **38,** 312-325.

Follstad Shah, J.J., Dahm, C.N., Gloss, S.P. & Bernhardt, E.S. (2007) River and Riparian Restoration in the Southwest: Results of the National River Restoration Science Synthesis Project. *Restoration Ecology,* **15,** 550-562.

Fonstad, M. & Marcus, W.A. (2003) Self-Organized Criticality in Riverbank Systems. *Annals of the Association of American Geographers,* **93,** 281-296.

Fozzard, I.R., Davidson, M. & Moffett, G. (1997) River habitat survey in Scotland. *Freshwater Quality: Defining the Indefinable?***,** 235-240.

Francis, R.A. (2006a) Allogenic and autogenic influences upon riparian vegetation dynamics. *Area,* **38,** 453-464.

Francis, R.A. (2006b) Allogenic and autogenic influences upon riparian vegetation dynamics. *Area,* **38,** 453-464.

Francis, R.A. (2007) Size and position matter: riparian plant establishment from fluvially deposited trees. *Earth Surface Processes and Landforms,* **32,** 1239-1243.

Francis, R.A. & Hoggart, S.P.G. (2008) Waste not, want not: The need to utilize existing artificial structures for habitat improvement along urban rivers. *Restoration Ecology,* **16,** 373-381.

Francis, R.A., Hoggart, S.P.G., Gurnell, A.M. & Coode, C. (2008) Meeting the challenges of urban river habitat restoration: developing a methodology for the River Thames through central London. *Area,* **40,** 435-445.

Franzen, D. (2001) The role of species richness for recruitment in a seminatural grassland. *Oikos,* **95,** 409-415.

Friberg, N., Kronvang, B., Hansen, H.O. & Svendsen, L.M. (1998) Long-term, habitat-specific response of a macroinvertebrate community to river restoration. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **8,** 87-99.

Gauch, H.G., Jr. (1973) A Quantitative Evaluation of the Bray-Curtis Ordination. *Ecology,* **54,** 829-836.

Giardina, C.P., Litton, C.M., Thaxton, J.M., Cordell, S., Hadway, L.J. & Sandquist, D.R. (2007) Science Driven Restoration: A Candle in a Demon Haunted World-Response to Cabin (2007). *Restoration Ecology,* **15,** 171-176.

Gibbins, C., Vericat, D. & Batalla, R.J. (2007) When is stream invertebrate drift catastrophic? The role of hydraulics and sediment transport in initiating drift during flood events. *Freshwater Biology,* **52,** 2369-2384.

Gibbons, P. & Freudenberger, D. (2006) An overview of methods used to assess vegetation condition at the scale of the site. *Ecological Management & Restoration,* **7,** S10-S17.

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

Gillilan, S., Boyd, K., Hoitsma, T. & Kauffman, M. (2005) Challenges in developing and implementing ecological standards for geomorphic river restoration projects: a practitioner's response to Palmer et al. (2005). *Journal of Applied Ecology,* **42,** 223-227.

Gilvear, D. & Bradley, S. (1997) Geomorphological adjustment of a newly engineered upland sinuous gravel-bed river diversion: Evan Water, Scotland. *Regulated Rivers-Research & Management,* **13,** 377-389.

Gilvear, D. & Willby, N. (2006) Channel dynamics and geomorphic variability as controls on gravel bar vegetation; River Tummel, Scotland. *River Research and Applications,* **22,** 457-474.

Gilvear, D.J. (1999) Fluvial geomorphology and river engineering: future roles utilizing a fluvial hydrosystems framework. *Geomorphology,* **31,** 229-245.

Gilvear, D.J., Heal, K.V. & Stephen, A. (2002) Hydrology and the ecological quality of Scottish river ecosystems. *Science of the Total Environment,* **294,** 131-159.

Gippel, C.J., Finlayson, B.L. & Oneill, I.C. (1996) Distribution and hydraulic significance of large woody debris in a lowland Australian river. *Hydrobiologia,* **318,** 179-194.

Gore, J.A. (1982) BENTHIC INVERTEBRATE COLONIZATION - SOURCE DISTANCE EFFECTS ON COMMUNITY COMPOSITION. *Hydrobiologia,* **94,** 183-193.

Gregory, K.J. & Chin, A. (2002) Urban stream channel hazards. *Area,* **34,** 312-321.

Griffin, I. (2005) *Biological colonisation and geomorphological development of a Scottish river diversion*, University of Stirling, Stirling.

Gross, M. (2007) Restoration and the Origins of Ecology. *Restoration Ecology,* **15,** 375-376.

Gurnell, A.M., Bickerton, M., Angold, P., Bell, D., Morrissey, I., Petts, G.E. & Sadler, J. (1998) Morphological and ecological change on a meander bend: the role of hydrological processes and the application of GIS. *Hydrological Processes [Hydrol.Process.].vol.12,* **122,** 981-993.

Gurnell, A.M., Morrissey, I.P., Boitsidis, A.J., Bark, T., Clifford, N.J., Petts, G.E. & Thompson, K. (2006) Initial adjustments within a new river channel: Interactions between fluvial processes, colonizing vegetation, and bank profile development. *Environmental Management,* **38,** 580-596.

Halle, S. (2007a) Present State and Future Perspectives of Restoration Ecology-Introduction. *Restoration Ecology,* **15,** 304-306.

Halle, S. (2007b) Science, Art, or Application-the "Karma" of Restoration Ecology. *Restoration Ecology,* **15,** 358-361.

Hannah, D.M., Sadler, J.P. & Wood, P.J. (2007) Hydroecology and ecohydrology: a potential route forward? *Hydrological Processes,* **21,** 3385-3390.

Hansen, H.O. (2001) European Centre for River Restoration and a database for river restoration. *International Association of Theoretical and Applied Limnology, Vol 27, Pt 3, Proceedings,* **27,** 1528-1531.

Harding, J.S., Benfield, E.F., Bolstad, P.V., Helfman, G.S. & Jones, E.B.D. (1998) Stream biodiversity: The ghost of land use past. *Proceedings of the National Academy of Sciences of the United States of America,* **95,** 14843-14847.

Harnischmacher, S. (2007) Thresholds in small rivers? Hypotheses developed from fluvial morphological research in western Germany. *Geomorphology,* **92,** 119-133.

Harper, D., Ebrahimnezhad, M. & Cot, F.C.I. (1998) Artificial riffles in river rehabilitation: setting the goals and measuring the successes. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **8,** 5-16.

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

Harper, D.M. (1995) River bottom habitats: Biological reality and practical value in river management. *Ecological Importance of River Bottom,* **91,** 35-45.

Harper, D.M., Ebrahimnezhad, M., Taylor, E., Dickinson, S., Decamp, O., Verniers, G. & Balbi, T. (1999) A catchment-scale approach to the physical restoration of lowland UK rivers. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **9,** 141-157.

Harrison, S.S.C., Pretty, J.L., Shepherd, D., Hildrew, A.G., Smith, C. & Hey, R.D. (2004) The effect of instream rehabilitation structures on macroinvertebrates in lowland rivers. *Journal of Applied Ecology,* **41,** 1140-1154.

Hart, D.D., Johnson, T.E., Bushaw-Newton, K.L., Horwitz, R.J., Bednarek, A.T., Charles, D.F., Kreeger, D.A. & Velinsky, D.J. (2002) Dam removal: Challenges and opportunities for ecological research and river restoration. *Bioscience,* **52,** 669-681.

Hassett, B.A., Palmer, M.A. & Bernhardt, E.S. (2007) Evaluating stream restoration in the Chesapeake Bay watershed through practitioner interviews. *Restoration Ecology,* **15,** 563-572.

Hawkins, C.P., Bartz, K.L. & Neale, C.M.U. (1997) Vulnerability of Riparian Vegetation to Catastrophic Flooding: Implications for Riparian Restoration. *Restoration Ecology,* **5,** 75-84.

Haynes, H., Haynes, R. & Pender, G. (2008) Integrating socio-economic analysis into decision-support methodology for flood risk management at the development scale (Scotland). *Water and Environment Journal,* **22,** 117-124.

Heiler, G., Hein, T., Schiemer, F. & Bornette, G. (1995) Hydrological connectivity and flood pulses as the central aspects for the integrity of a river-floodplain system. *Regulated Rivers-Research & Management,* **11,** 351-361.

Hein, T., Heiler, G., Pennetzdorfer, D., Riedler, P., Schagerl, M. & Schiemer, F. (1999) The Danube restoration project: Functional aspects and planktonic productivity in the floodplain system. *Regulated Rivers-Research & Management,* **15,** 259-270.

Hey, R.D. (2006) Fluvial geomorphological methodology for natural stable channel design. *Journal of the American Water Resources Association,* **42,** 357-374.

Hillman, M. & Brierley, G. (2005) A critical review of catchment-scale stream rehabilitation programmes. *Progress in Physical Geography,* **29,** 50-76.

Hobbs, R.J. (2007a) Restoration Ecology: Are We Making an Impact? *Restoration Ecology,* **15,** 597-600.

Hobbs, R.J. (2007b) Setting Effective and Realistic Restoration Goals: Key Directions for Research. *Restoration Ecology,* **15,** 354-357.

Hobbs, R.J. & Cramer, V.A. (2008) Restoration Ecology: Interventionist Approaches for Restoring and Maintaining Ecosystem Function in the Face of Rapid Environmental Change. *Annual Review of Environment and Resources,* **33,** 39-61.

Hobbs, R.J. & Harris, J.A. (2001) Restoration ecology: Repairing the Earth's ecosystems in the new millennium. *Restoration Ecology,* **9,** 239-246.

Hobbs, R.J. & Norton, D.A. (1996) Towards a conceptual framework for restoration ecology. *Restoration Ecology,* **4,** 93-110.

Holl, K., Bonilla, M., Carle, R., Carvill, S., Concilio, A., Gabbe, A., Hulvey, K., Kolden, E., Krawack, M.L., Schweizer, D. & Vasey, M. (2007) Foundations of Restoration Ecology - Edited by Donald A. Falk, Margaret A. Palmer, and Joy B. Zedler. *Restoration Ecology,* **15,** 592-593.

Holl, K.D. & Howarth, R.B. (2000) Paying for restoration. *Restoration Ecology,* **8,** 260-267.

Holmes, N.T.H. & Nielsen, M.B. (1998) Restoration of the rivers Brede, Cole and Skerne: a joint Danish and British EU-LIFE demonstration project, I - Setting up and delivery of the project. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **8,** 185-196.

Holmes, N.T.H., Snape, A. & Spray, C.J. (1999) A demonstration river restoration project incorporating surface water sewer improvements. *Sustaining Urban Water Resources in the 21St Century, Proceedings***,** 280-298.

Hooke, J. (2003) Coarse sediment connectivity in river channel systems: a conceptual framework and methodology. *Geomorphology [Geomorphology].vol.56,* **566,** 79-94.

Hooke, J.M. (2007) Spatial variability, mechanisms and propagation of change in an active meandering river. *Geomorphology [Geomorphology].vol.84,* **844,** 277-296.

Hopfensperger, K.N., Engelhardt, K.A.M. & Seagle, S.W. (2007) Ecological feasibility studies in restoration decision making. *Environmental Management,* **39,** 843-852.

Hughes, F.M.R., Adams, W.M., Muller, E., Nilsson, C., Richards, K.S., Barsoum, N., Decamps, H., Foussadier, R., Girel, J., Guilloy, H., Hayes, A., Johansson, M., Lambs, L., Pautou, G., Peiry, J.L., Perrow, M., Vautier, F. & Winfield, M. (2001) The importance of different scale processes for the restoration of floodplain woodlands. *Regulated Rivers-Research & Management,* **17,** 325-345.

Hughes, F.M.R., Colston, A. & Mountford, J.O. (2005) Restoring riparian ecosystems: The challenge of accommodating variability and designing restoration trajectories. *Ecology and Society,* **10**.

Hughes, F.M.R. & Rood, S.B. (2003) Allocation of river flows for restoration of floodplain forest ecosystems: A review of approaches and their applicability in Europe. *Environmental Management,* **32,** 12-33.

Hultin, L., Svensson, B. & Ulfstran.S (1969) UPSTREAM MOVEMENTS OF INSECTS IN A SOUTH SWEDISH SMALL STREAM. *Oikos,* **20,** 553-&.

Humphries, S. (2002) Dispersal in drift-prone macroinvertebrates: a case for density-independence. *Freshwater Biology,* **47,** 921-929.

Humphries, S. & Ruxton, G.D. (2001) Re-examining the drift paradox. *Trends in Ecology & Evolution,* **16,** 486.

Humphries, S. & Ruxton, G.D. (2002) Is there really a drift paradox? *Journal of Animal Ecology,* **71,** 151-154.

Hupp, C.R. & Osterkamp, W.R. (1996) Riparian vegetation and fluvial geomorphic processes. *Geomorphology,* **14,** 277-295.

Hupp, C.R. & Rinaldi, M. (2007) Riparian vegetation patterns in relation to fluvial landforms and channel evolution along selected rivers of Tuscany (Central Italy). *Annals of the Association of American Geographers,* **97,** 12-30.

Hurlbert, S.H. (1984) PSEUDOREPLICATION AND THE DESIGN OF ECOLOGICAL FIELD EXPERIMENTS. *Ecological Monographs,* **54,** 187-211.

Jaehnig, S.C. & Lorenz, A.W. (2008) Substrate-specific macroinvertebrate diversity patterns following stream restoration. *Aquatic Sciences,* **70,** 292-303.

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

Jahnig, S.C., Brunzel, S., Gacek, S., Lorenz, A.W. & Hering, D. (2009) Effects of re-braiding measures on hydromorphology, floodplain vegetation, ground beetles and benthic invertebrates in mountain rivers. *Journal of Applied Ecology,* **46,** 406-416.

Jahnig, S.C., Lorenz, A. & Hering, D. (2008) Hydromorphological parameters indicating differences between single- and multiple-channel mountain rivers in Germany, in relation to their modification and recovery. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **18,** 1200-1216.

James, A.B.W., Dewson, Z.S. & Death, R.G. (2009) The influence of flow reduction on macroinvertebrate drift density and distance in three New Zealand streams. *Journal of the North American Benthological Society,* **28,** 220-232.

Jansson, R., Backx, H., Boulton, A.J., Dixon, M., Dudgeon, D., Hughes, F.M.R., Nakamura, K., Stanley, E.H. & Tockner, K. (2005) Stating mechanisms and refining criteria for ecologically successful river restoration: a comment on Palmer et al. (2005). *Journal of Applied Ecology,* **42,** 218-222.

Jansson, R., Nilsson, C., Dynesius, M. & Andersson, E. (2000) Effects of river regulation on river-margin vegetation: A comparison of eight boreal rivers. *Ecological Applications,* **10,** 203-224.

Jansson, R., Nilsson, C. & Malmqvist, B. (2007) Restoring freshwater ecosystems in riverine landscapes: the roles of connectivity and recovery processes. *Freshwater Biology,* **52,** 589-596.

Jansson, R.O.L.A., Zinko, U.R.S.U., Merritt, D.M. & Nilsson, C.H.R.I. (2005) Hydrochory increases riparian plant species richness: a comparison between a free-flowing and a regulated river. *Journal of Ecology,* **93,** 1094-1103.

Johansson, M.E. & Nilsson, C. (2002) Responses of riparian plants to flooding in free-flowing and regulated boreal rivers: an experimental study. *Journal of Applied Ecology,* **39,** 971-986.

Johnson, P.A. & Brown, E.R. (2001) INCORPORATING UNCERTAINTY IN THE DESIGN OF STREAM CHMNEL MODIFICATIONS1. *Journal of the American Water Resources Association,* **37,** 1225-1236.

Johnson, R.K., Furse, M.T., Hering, D.A.N.I. & Sandin, L.E.O.N. (2007) Ecological relationships between stream communities and spatial scale: implications for designing catchment-level monitoring programmes. *Freshwater Biology,* **52,** 939-958.

Johnson, R.K., Wiederholm, T. & Rosenberg, D.M. (1993) Freshwater biomonitoring using individual organisms, populations, and species assemblages of benthic macroinvertebrates. *Freshwater biomonitoring and benthic macroinvertebrates***,** 40-158.

Jowett, I.G. (1997) Instream flow methods: A comparison of approaches. *Regulated Rivers-Research & Management,* **13,** 115-127.

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

Jungwirth, M., Muhar, S. & Schmutz, S. (2002) Re-establishing and assessing ecological integrity in riverine landscapes. *Freshwater Biology,* **47,** 867-887.

Karwacki, P. (2003) The Kipawa River versus the Tabaret River diversion projects. *Applied Energy,* **75,** 221-233.

Kasahara, T. & Hill, A.R. (2006) Hyporheic exchange flows induced by constructed riffles and steps in lowland streams in southern Ontario, Canada. *Hydrological Processes,* **20,** 4287-4305.

Katz, S.L., Barnas, K., Hicks, R., Cowen, J. & Jenkinson, R. (2007) Freshwater habitat restoration actions in the Pacific Northwest: A decade's investment in habitat improvement. *Restoration Ecology,* **15,** 494-505.

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

Kemp, P.S., Gilvear, D.J. & Armstrong, J.D. (2003) Do juvenile Atlantic salmon parr track local changes in water velocity? *River Research and Applications [River Res.Appl.].vol.19,* **199,** 569-575.

Khalaf, G. & Tachet, H. (1980) COLONIZATION OF ARTIFICIAL SUBSTRATA BY MACRO-INVERTEBRATES IN A STREAM AND VARIATIONS ACCORDING TO STONE SIZE. *Freshwater Biology,* **10,** 475-482.

Kitchen, T. (1997) Cities of tomorrow - Hall,P. *Regional Studies,* **31,** 823-823.

Koebel, J.W. (1995) An Historical-Perspective on the Kissimmee River Restoration Project. *Restoration Ecology,* **3,** 149-159.

Koed, A., Baktoft, H. & Bak, B.D. (2006) Causes of mortality of Atlantic salmon (Salmo salar) and brown trout (Salmo trutta) smolts in a restored river and its estuary. *River Research and Applications,* **22,** 69-78.

Koleff, P., Gaston, K.J. & Lennon, J.J. (2003) Measuring beta diversity for presence-absence data. *Journal of Animal Ecology,* **72,** 367-382.

Kondolf, G.M. (1995) 5 Elements for Effective Evaluation of Stream Restoration. *Restoration Ecology,* **3,** 133-136.

Kondolf, G.M. (1998) Lessons learned from river restoration projects in California. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **8,** 39-52.

Kondolf, G.M. (2000) Some suggested guidelines for geomorphic aspects of anadromous salmonid habitat restoration proposals. *Restoration Ecology,* **8,** 48-56.

Kondolf, G.M. (2006) River restoration and meanders. *Ecology and Society,* **11**.

Kondolf, G.M., Anderson, S., Lave, R., Pagano, L., Merenlender, A. & Bernhardt, E.S. (2007) Two Decades of River Restoration in California: What Can We Learn? *Restoration Ecology,* **15,** 516-523.

Kondolf, G.M., Angermeier, P.L., Cummins, K., Dunne, T., Healey, M., Kimmerer, W., Moyle, P.B., Murphy, D., Patten, D., Railsback, S., Reed, D.J., Spies, R. & Twiss, R. (2008) Projecting Cumulative Benefits of Multiple River Restoration Projects: An Example from the Sacramento-San Joaquin River System in California. *Environmental Management,* **42,** 933-945.

Kondolf, G.M., Boulton, A.J., O'daniel, S., Poole, G.C., Rachel, F.J., Stanley, E.H., Wohl, E., Bang, A., Carlstrom, J., Cristoni, C., Huber, H., Koljonen, S., Louhi, P. & Nakamura, K. (2006) Process-based ecological river restoration: Visualizing three-dimensional connectivity and dynamic vectors to recover lost linkages. *Ecology and Society,* **11**.

Kondolf, G.M., Piegay, H. & Landon, N. (2007) Changes in the riparian zone of the lower Eygues River, France, since 1830. *Landscape Ecology,* **22,** 367-384.

Kopp, M., Jeschke, J.M. & Gabriel, W. (2001) Exact compensation of stream drift as an evolutionarily stable strategy. *Oikos,* **92,** 522-530.

Krause, S., Jacobs, J., Voss, A., Bronstert, A. & Zehe, E. (2008) Assessing the impact of changes in landuse and management practices on the diffuse pollution and retention of nitrate in a riparian floodplain. *Science of the Total Environment,* **389,** 149-164.

Kronvang, B., Svendsen, L.M., Brookes, A., Fisher, K., Moller, B., Ottosen, O., Newson, M. & Sear, D. (1998) Restoration of the rivers Brede, Cole and Skerne: a joint Danish and British EU-LIFE demonstration project, III - Channel morphology, hydrodynamics and transport of sediment and nutrients. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **8,** 209-222.

Lacey, R.W.J. & Millar, R.G. (2004) Reach scale hydraulic assessment of instream salmonid habitat restoration. *Journal of the American Water Resources Association,* **40,** 1631-1644.

Lake, P.S. (2001) On the maturing of restoration: Linking ecological research and restoration. *Ecological Management & Restoration,* **2,** 110-115.

Lake, P.S. (2005) Perturbation, restoration and seeking ecological sustainability in Australian flowing waters. *Hydrobiologia,* **552,** 109-120.

Lake, P.S., Bond, N. & Reich, P. (2007) Linking ecological theory with stream restoration. *Freshwater Biology,* **52,** 597-615.

Lamarre, H. & Roy, A.G. (2008) A field experiment on the development of sedimentary structures in a gravel-bed river. *Earth Surface Processes and Landforms,* **33,** 1064-1081.

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

Lancaster, J. & Hildrew, A.G. (1993a) CHARACTERIZING IN-STREAM FLOW REFUGIA. *Canadian Journal of Fisheries and Aquatic Sciences,* **50,** 1663-1675.

Lancaster, J. & Hildrew, A.G. (1993b) FLOW REFUGIA AND THE MICRODISTRIBUTION OF LOTIC MACROINVERTEBRATES. *Journal of the North American Benthological Society,* **12,** 385-393.

Lancaster, J., Hildrew, A.G. & Gjerlov, C. (1996) Invertebrate drift and longitudinal transport processes in streams. *Canadian Journal of Fisheries and Aquatic Sciences,* **53,** 572-582.

Landers, D.H. (1997) Riparian Restoration: Current Status and the Reach to the Future. *Restoration Ecology,* **5,** 113-121.

Landers, D.H., Haggerty, P.K., Cline, S., Carson, W. & Faure, F. (2000) The role of regionalization in large river restoration. *International Association of Theoretical and Applied Limnology, Vol 27, Pt 1, Proceedings,* **27,** 344-351.

Lane, S.N. & Richards, K.S. (1997) Linking river channel form and process: Time, space and causality revisited. *Earth Surface Processes and Landforms,* **22,** 249-260.

Larkin, P.A. & Mckone, D.W. (1985) AN EVALUATION BY FIELD EXPERIMENTS OF THE MCLAY MODEL OF STREAM DRIFT. *Canadian Journal of Fisheries and Aquatic Sciences,* **42,** 909-918.

Lauridsen, R.B. & Friberg, N. (2005) Stream macroinvertebrate drift response to pulsed exposure of the synthetic pyrethroid lambda-cyhalothrin. *Environmental Toxicology,* **20,** 513-521.

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

Lepori, F., Palm, D. & Malmqvist, B. (2005) Effects of stream restoration on ecosystem functioning: detritus retentiveness and decomposition. *Journal of Applied Ecology,* **42,** 228-238.

Leps, J. & Smilauer, P. (2003) Multivariate analysis of ecological data using CANOCO. *Multivariate analysis of ecological data using CANOCO***,** i-xi, 1-269.

Leslie (2009) *Economic value of salmon fishing to Scotland*, University of Stirling, Stirling.

Liermann, M. & Roni, P. (2008) More sites or more years? Optimal study design for monitoring fish response to watershed restoration. *North American Journal of Fisheries Management,* **28,** 935-943.

Linhart, J., Vlckova, S. & Uvira, V. (2002) Bryophytes as a special mesohabitat for meiofauna in a rip-rapped channel. *River Research and Applications,* **18,** 321-330.

Linke, S., Pressey, R.L., Bailey, R.C. & Norris, R.H. (2007) Management options for river conservation planning: condition and conservation re-visited. *Freshwater Biology,* **52,** 918-938.

Looy, K.V., Honnay, O., Pedroli, B. & Muller, S. (2006) Order and disorder in the river continuum: the contribution of continuity and connectivity to floodplain meadow biodiversity. *Journal of Biogeography,* **33,** 1615-1627.

Macinnis, C., Floyd, T.A. & Taylor, B.R. (2008) Large woody debris structures and their influence on Atlantic salmon spawning in a stream in Nova Scotia, Canada. *North American Journal of Fisheries Management,* **28,** 781-791.

Mackay, R.J. (1992) COLONIZATION BY LOTIC MACROINVERTEBRATES - A REVIEW OF PROCESSES AND PATTERNS. *Canadian Journal of Fisheries and Aquatic Sciences,* **49,** 617-628.

Mackey, R.L. & Currie, D.J. (2001) The diversity-disturbance relationship: Is it generally strong and peaked? *Ecology,* **82,** 3479-3492.

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

Mader, H. & Maier, C. (2008) A method for prioritizing the reestablishment of river continuity in Austrian rivers. *Hydrobiologia,* **609,** 277-288.

Madsen, B.L., Bengtsson, J. & Butz, I. (1977) UPSTREAM MOVEMENT BY SOME EPHEMEROPTERA SPECIES. *Archiv Fur Hydrobiologie,* **81,** 119-127.

Madsen, B.L., Boon, P., Lake, P.S., Bunn, S.E., Dahm, C.N., Langford, T.E. & Zalewski, M. (2006) Ecological principles and stream restoration. *International Association of Theoretical and Applied Limnology, Vol 29, Pt 4, Proceedings,* **29,** 2045-2050.

Maitland, P.S. & Penney, M.M. (1967) ECOLOGY OF SIMULIIDAE IN A SCOTTISH RIVER. *Journal of Animal Ecology,* **36,** 179-&.

Malcolm, I.A., Soulsby, C., Youngson, A.F. & Petry, J. (2003) Heterogeneity in ground water-surface water interactions in the hyporheic zone of a salmonid spawning stream. *Hydrological Processes,* **17,** 601-617.

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

Malmqvist, B. & Otto, C. (1987) THE INFLUENCE OF SUBSTRATE STABILITY ON THE COMPOSITION OF STREAM BENTHOS - AN EXPERIMENTAL-STUDY. *Oikos,* **48,** 33-38.

Maloney, K.O., Feminella, J.W., Mitchell, R.M., Miller, S.A., Mulholland, P.J. & Houser, J.N. (2008) Landuse legacies and small streams: identifying relationships between historical land use and contemporary stream conditions. *Journal of the North American Benthological Society,* **27,** 280-294.

Mander, U., Kuusemets, V. & Hayakawa, Y. (2005) Purification processes, ecological functions, planning and design of riparian buffer zones in agricultural watersheds. *Ecological Engineering,* **24,** 421-432.

Manners, R.B., Doyle, M.W. & Small, M.J. (2007) Structure and hydraulics of natural woody debris jams. *Water Resources Research,* **43**.

Marshall, K.E. (1993) The literature of biomonitoring. *Freshwater biomonitoring and benthic macroinvertebrates***,** 28-39.

Masters, Z., Peteresen, I., Hildrew, A.G. & Ormerod, S.J. (2007) Insect dispersal does not limit the biological recovery of streams from acidification. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **17,** 375-383.

Matczak, T.Z. & Mackay, R.J. (1990) TERRITORIALITY IN FILTER-FEEDING CADDISFLY LARVAE - LABORATORY EXPERIMENTS. *Journal of the North American Benthological Society,* **9,** 26-34.

Mcbride, M. & Booth, D.B. (2005) URBAN IMPACTS ON PHYSICAL STREAM CONDITION: EFFECTS OF SPATIAL SCALE, CONNECTIVITY, AND LONGITUDINAL TRENDS1. *Journal of the American Water Resources Association,* **41,** 565-580.

Mccoy, E.D. & Mushinsky, H.R. (2002) Measuring the success of wildlife community restoration. *Ecological Applications,* **12,** 1861-1871.

Mcdonald, A., Lane, S.N., Haycock, N.E. & Chalk, E.A. (2004) Rivers of dreams: on the gulf between theoretical and practical aspects of an upland river restoration. *Transactions of the Institute of British Geographers,* **29,** 257-281.

Mcdonald, T. (2007) Visions of stream restoration; the need for long-term, big-picture projects: Interview with Sam Lake. *Ecological Management & Restoration,* **8,** 5-10.

Mcewen, L.J., Brazier, V. & Gordon, J.E. (1997) Evaluating the geomorphology of fresh waters: An assessment of approaches. *Freshwater Quality: Defining the Indefinable?***,** 258-281.

Mclay, C. (1970) A THEORY CONCERNING DISTANCE TRAVELLED BY ANIMALS ENTERING DRIFT OF A STREAM. *Journal of the Fisheries Research Board of Canada,* **27,** 359-&.

Mcneil, J.R. (2000) *Something New Under the Sun: An Environmental History of the Twentieth Century World,* W W Norton, New York.

Mehan, G.T., Naiman, R.J. & Latterell, J.J. (2006) River restoration. *Issues in Science and Technology,* **22,** 17-19.

Merritt, D.M. & Wohl, E.E. (2002) Processes governing hydrochory along rivers: Hydraulics, hydrology, and dispersal phenology. *Ecological Applications,* **12,** 1071-1087.

Michener, W.K. (1997) Quantitatively evaluating restoration experiments: Research design, statistical analysis, and data management considerations. *Restoration Ecology,* **5,** 324-337.

Miller, J.R. & Hobbs, R.J. (2007) Habitat Restoration-Do We Know What Were Doing? *Restoration Ecology,* **15,** 382-390.

Millington, C.E. & Sear, D.A. (2007) Impacts of river restoration on small-wood dynamics in a low-gradient headwater stream. *Earth Surface Processes and Landforms,* **32,** 1204-1218.

Milner, A.M. & Gloyne-Phillips, I.T. (2005) The role of riparian vegetation and woody debris in the development of macroinvertebrate assemblages in streams. *River Research and Applications,* **21,** 403-420.

Monteith, H. & Pender, G. (2005) Flume investigations into the influence of shear stress history on a graded sediment bed. *Water Resources Research,* **41.**

Montgomery, D.R. & Macdonald, L.H. (2002) DIAGNOSTIC APPROACH TO STREAM CHANNEL ASSESSMENT AND MONITORING1. *Journal of the American Water Resources Association,* **38,** 1-16.

Mouton, A.M., Schneider, M., Depestele, J., Goethals, P.L.M. & De Pauw, N. (2007) Fish habitat modelling as a tool for river management. *Ecological Engineering,* **29,** 305-315.

Muhar, S., Schmutz, S. & Jungwirth, M. (1995) River Restoration Concepts - Goals and Perspectives. *Hydrobiologia,* **303,** 183-194.

Muotka, T. & Laasonen, P. (2002) Ecosystem recovery in restored headwater streams: the role of enhanced leaf retention. *Journal of Applied Ecology,* **39,** 145-156.

Muotka, T., Paavola, R., Haapala, A., Novikmec, M. & Laasonen, P. (2002) Long-term recovery of stream habitat structure and benthic invertebrate communities from in-stream restoration. *Biological Conservation,* **105,** 243-253.

Muotka, T. & Syrjanen, J. (2007) Changes in habitat structure, benthic invertebrate diversity, trout populations and ecosystem processes in restored forest streams: a boreal perspective. *Freshwater Biology,* **52,** 724-737.

Muotka, T. & Virtanen, R. (1995) THE STREAM AS A HABITAT TEMPLET FOR BRYOPHYTES - SPECIES DISTRIBUTIONS ALONG GRADIENTS IN DISTURBANCE AND SUBSTRATUM HETEROGENEITY. *Freshwater Biology,* **33,** 141-160.

Murray, B.G. (2001) Are ecological and evolutionary theories scientific? *Biological Reviews,* **76,** 255-289.

Mutz, M. (1998) Stream system restoration in a strip-mining region, eastern Germany: dimension, problems, and first steps. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **8,** 159-166.

Nagle, G. (2007) Evatuating 'natural channel design' stream projects. *Hydrological Processes,* **21,** 2539-2545.

Naiman, R.J., Decamps, H. & Pollock, M. (1993) The Role of Riparian Corridors in Maintaining Regional Biodiversity. *Ecological Applications,* **3,** 209-212.

Naiman, R.J., Elliott, S.R., Helfield, J.M. & O'keefe, T.C. (1999) Biophysical interactions and the structure and dynamics of riverine ecosystems: the importance of biotic feedbacks. *Hydrobiologia,* **410,** 79-86.

Nakamura, K., Tockner, K. & Amano, K. (2006) River and wetland restoration: Lessons from Japan. *Bioscience,* **56,** 419-429.

Nakano, D., Nagayama, S., Kawaguchi, Y. & Nakamura, F. (2008) River restoration for macroinvertebrate communities in lowland rivers: insights from restorations of the Shibetsu River, north Japan. *Landscape and Ecological Engineering,* **4,** 63-68.

Nakano, D. & Nakamura, F. (2006) Responses of macroinvertebrate communities to river restoration in a channelized segment of the Shibetsu River, Northern Japan. *River Research and Applications,* **22,** 681-689.

Nakano, D. & Nakamura, F. (2008) The significance of meandering channel morphology on the diversity and abundance of macroinvertebrates in a lowland river in Japan. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **18,** 780-798.

Neilsen, M. (2002) Lowland stream restoration in Denmark: Background and examples. *Journal of the Chartered Institution of Water and Environmental Management,* **16,** 189-193.

Newson, M.D. (2002) Geomorphological concepts and tools for sustainable river ecosystem management. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **12,** 365-379.

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.

Nielsen, M.B. (1996) River restoration: Report of a major EU life demonstration project. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **6,** 187-190.

Nikora, V.I., Suren, A.M., Brown, S.L.R. & Biggs, B.J.F. (1998) The effects of the moss Fissidens rigidulus (Fissidentaceae : Musci) on near-bed flow structure in an experimental cobble bed flume. *Limnology and Oceanography,* **43,** 1321-1331.

Nilsson, C. (1992) Conservation management of riparian communities. *Conservation Ecology Series; Ecological principles of nature conservation: Applications in temperate and boreal environments***,** 352-372.

Nilsson, C., Jansson, R., Malmqvist, B. & Naiman, R.J. (2007) Restoring riverine landscapes: The challenge of identifying priorities, reference states, and techniques. *Ecology and Society,* **12**.

Nolan, P.A. & Guthrie, N. (1998) River rehabilitation in an urban environment: examples from the Mersey Basin, North West England. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **8,** 685-700.

Norberg, S. (2000) *Principles of Modern Architecture,* Andreas Papadakis, London.

Norris, R.H. & Georges, A. (1993) Analysis and interpretation of benthic macroinvertebrate surveys. *Freshwater biomonitoring and benthic macroinvertebrates***,** 234-286.

Nuttle, T. (2007) Evaluation of Restoration Practice Based on Environmental Filters. *Restoration Ecology,* **15,** 330-333.

O'donnell, T.K. & Galat, D.L. (2008) Evaluating success criteria and project monitoring in river enhancement within an adaptive management framework. *Environmental Management,* **41,** 90-105.

Opperman, J.J. & Merenlender, A.M. (2007) Living trees provide stable large wood in streams. *Earth Surface Processes and Landforms,* **32,** 1229-1238.

Ormerod, S.J. (2004) A golden age of river restoration science? *Aquatic Conservation-Marine and Freshwater Ecosystems,* **14,** 543-549.

Osborne, L.L. & Kovacic, D.A. (1993) Riparian vegetated buffer strips in water-quality restoration and stream management. *Freshwater Biology,* **29,** 243-258.

Osmundson, D.B., Ryel, R.J., Lamarra, V.L. & Pitlick, J. (2002) Flow-sediment-biota relations: Implications for river regulation effects on native fish abundance. *Ecological Applications,* **12,** 1719-1739.

Pagliara, S., Das, R. & Palermo, M. (2008) Energy dissipation on submerged block ramps. *Journal of Irrigation and Drainage Engineering-Asce,* **134,** 527-532.

Palmer, M., Allan, J.D., Meyer, J. & Bernhardt, E.S. (2007a) River restoration in the twenty-first century: Data and experiential future efforts. *Restoration Ecology,* **15,** 472-481.

Palmer, M., Allan, J.D., Meyer, J. & Bernhardt, E.S. (2007b) River Restoration in the Twenty-First Century: Data and Experiential Knowledge to Inform Future Efforts. *Restoration Ecology,* **15,** 472-481.

Palmer, M.A. & Allan, J.D. (2006) Restoring Rivers. *Issues in Science and Technology [Issues Sci.Technol.].vol.22,* **222,** 40-48.

Palmer, M.A., Ambrose, R.F. & Poff, N.L. (1997) Ecological theory and community restoration ecology. *Restoration Ecology,* **5,** 291-300.

Palmer, M.A. & Bernhardt, E.S. (2006) Hydroecology and river restoration: Ripe for research and synthesis. *Water Resources Research,* **42**.

Palmer, M.A., Bernhardt, E.S., Allan, J.D., Lake, P.S., Alexander, G., Brooks, S., Carr, J., Clayton, S., Dahm, C.N., Shah, J.F., 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,** 208-217.

Parasiewicz, P., Eberstaller, J., Weiss, S. & Schmutz, S. (1998) Conceptual guidelines for nature-like bypass channels. *Fish Migration and Fish Bypasses***,** 348-362.

Parker, J.D., Burkepile, D.E., Collins, D.O., Kubanek, J. & Hay, M.E. (2007) Stream mosses as chemically-defended refugia for freshwater macroinvertebrates. *Oikos,* **116,** 302-312.

Parsons, M. & Norris, R.H. (1996) The effect of habitat-specific sampling on biological assessment of water quality using a predictive model. *Freshwater Biology,* **36,** 419-434.

Parsons, M., Thoms, M.C. & Norris, R.H. (2003) Scales of macroinvertebrate distribution in relation to the hierarchical organization of river systems. *Journal of the North American Benthological Society,* **22,** 105-122.

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,** 125-139.

Patten, D.T., Harpman, D.A., Voita, M.I. & Randle, T.J. (2001) A managed flood on the Colorado River: Background, objectives, design, and implementation. *Ecological Applications,* **11,** 635-643.

Peckarsky, B.L. (1980a) INFLUENCE OF DETRITUS UPON COLONIZATION OF STREAM INVERTEBRATES. *Canadian Journal of Fisheries and Aquatic Sciences,* **37,** 957-963.

Peckarsky, B.L. (1980b) PREDATOR-PREY INTERACTIONS BETWEEN STONEFLIES AND MAYFLIES - BEHAVIORAL OBSERVATIONS. *Ecology,* **61,** 932-943.

Peckarsky, B.L. (1991) HABITAT SELECTION BY STREAM-DWELLING PREDATORY STONEFLIES. *Canadian Journal of Fisheries and Aquatic Sciences,* **48,** 1069-1076.

Pedersen, M.L. & Friberg, N. (2007) Two lowland stream riffles - linkages between physical habitats and macroinvertebrates across multiple spatial scales. *Aquatic Ecology,* **41,** 475-490.

Pedersen, M.L. & Friberg, N. (2009) Influence of disturbance on habitats and biological communities in lowland streams. *Fundamental and Applied Limnology,* **174,** 27-41.

Pedersen, M.L., Friberg, N., Skriver, J., Baattrup-Pedersen, A. & Larsen, S.E. (2007) Restoration of Skjern River and its valley - Short-term effects on river habitats, macrophytes and macroinvertebrates. *Ecological Engineering,* **30,** 145-156.

Pedersen, T.C.M., Baattrup-Pedersen, A.N.N.E. & Madsen, T.V. (2006) Effects of stream restoration and management on plant communities in lowland streams. *Freshwater Biology,* **51,** 161-179.

Pedroli, B., De Blust, G., Van Looy, K. & Van Rooij, S. (2002) Setting targets in strategies for river restoration. *Landscape Ecology,* **17,** 5-18.

Pedroli, B., Pinto-Correia, T. & Cornish, P. (2006) Landscape - what's in it? Trends in European landscape science and priority themes for concerted research. *Landscape Ecology,* **21,** 421-430.

Pedroli, G.B.M. & Postma, R. (1998) Nature rehabilitation in European river ecosystems: Three cases. *New Concepts for Sustainable Management of River Basins***,** 67-84.

Pereira, A., Duarte, P. & Norro, A. (2006) Different modelling tools of aquatic ecosystems: A proposal for a unified approach. *Ecological Informatics,* **1,** 407-421.

Petersen, I., Masters, Z., Hildrew, A.G. & Ormerod, S.J. (2004) Dispersal of adult aquatic insects in catchments of differing land use. *Journal of Applied Ecology,* **41,** 934-950.

Petts, G. (1989) Historical Analysis of Fluvial Hydrosystems. In: *Historical Change of Large Alluvial Rivers: Western Europe.* (Ed^Eds, pp. 1-18. John Wiley & Sons, New York.

Petts, G. (2001) Sustaining our rivers in crisis: setting the international agenda for action. *Water Science and Technology,* **43,** 3-16.

Petts, G.E. (2004) Perspectives for Ecological Management of Regulated Rivers. *IN: Alternatives in Regulated River Management.CRC Press***,** 24.

Petts, G.E., Nestler, J. & Kennedy, R. (2006) Advancing science for water resources management. *Hydrobiologia,* **565,** 277-288.

Petts, J. (2007) Learning about learning: lessons from public engagement and deliberation on urban river restoration. *Geographical Journal,* **173,** 300-311.

Pfadenhauer, J. (2001) Some remarks on the socio-cultural background of restoration ecology. *Restoration Ecology,* **9,** 220-229.

Phelps, D., Ward, M. & Bailey, L. (2004) Similkameen River restoration: A work in progress. *Protection and Restoration of Urban and Rural Streams***,** 281-286.

Piegay, H., Darby, S.E., Mosselman, E. & Surian, N. (2005) A review of techniques available for delimiting the erodible river corridor: A sustainable approach to managing bank erosion. *River Research and Applications,* **21,** 773-789.

Piegay, H. & Hicks, D.M. (2005) Sediment management in river systems: A need to assess changing processes in the long term and at a large scale. *River Research and Applications,* **21,** 689-691.

Piegay, H., Mutz, M., Gregory, K.J., Rinaldi, M., Bondarev, V., Chin, A., Wyzga, B., Dahlstrom, N., Zawiejska, J., Elosegi, A., Gregory, S.V. & Joshi, V. (2005) Public perception as a barrier to introducing wood in rivers for restoration purposes. *Environmental Management,* **36,** 665-674.

Poff, N.L. & Ward, J.V. (1991) DRIFT RESPONSES OF BENTHIC INVERTEBRATES TO EXPERIMENTAL STREAMFLOW VARIATION IN A HYDROLOGICALLY STABLE STREAM. *Canadian Journal of Fisheries and Aquatic Sciences,* **48,** 1926-1936.

Pohlon, E., Augspurger, C., Risse-Buhl, U., Arle, J., Willkomm, M., Halle, S. & Kusel, K. (2007) Querying the obvious: Lessons from a degraded stream. *Restoration Ecology,* **15,** 312-316.

Postel, S. (2002) Rivers of Life: the challenge of restoring health to freshwater ecosystems. *Water Science and Technology,* **45,** 3-8.

Pozo, J., Gonzalez, E., Diez, J.R., Molinero, J. & Elosegui, A. (1997) Inputs of particulate organic matter to streams with different riparian vegetation. *Journal of the North American Benthological Society,* **16,** 602-611.

Pretty, J.L., Harrison, S.S.C., Shepherd, D.J., Smith, C., Hildrew, A.G. & Hey, R.D. (2003) River rehabilitation and fish populations: assessing the benefit of instream structures. *Journal of Applied Ecology,* **40,** 251-265.

Purcell, A.H., Friedrich, C. & Resh, V.H. (2002) An Assessment of a Small Urban Stream Restoration Project in Northern California. *Restoration Ecology,* **10,** 685-694.

Pyrce, R.S. & Ashmore, P.E. (2005) Bedload path length and point bar development in gravel-bed river models. *Sedimentology,* **52,** 839-857.

Rader, R.B. & Belish, T.A. (1999) Influence of mild to severe flow alterations on invertebrates in three mountain streams. *Regulated Rivers-Research & Management,* **15,** 353-363.

Rahel, F.J. (2007) Biogeographic barriers, connectivity and homogenization of freshwater faunas: it's a small world after all. *Freshwater Biology,* **52,** 696-710.

Rayburg, S.C. & Neave, M. (2008) Assessing Morphologic Complexity and Diversity in River Systems Using Three-Dimensional Asymmetry Indices for Bed Elements, Bedforms and Bar Units. *River Research and Applications,* **24,** 1343-1361.

Reeve, T., Lichatowich, J., Towey, W. & Duncan, A. (2006) Building science and accountability into community-based restoration: Can a new funding approach facilitate effective and accountable restoration? *Fisheries,* **31,** 17-24.

Renofalt, B.M., Merritt, D.M. & Nilsson, C. (2007) Connecting variation in vegetation and stream flow: the role of geomorphic context in vegetation response to large floods along boreal rivers. *Journal of Applied Ecology,* **44,** 147-157.

Renofalt, B.M., Nilsson, C. & Jansson, R. (2005) Spatial and temporal patterns of species richness in a riparian landscape. *Journal of Biogeography,* **32,** 2025-2037.

Resh, V.H. & Mcelravy, E.P. (1993) Contemporary quantitative approaches to biomonitoring using benthic macroinvertebrates. *Freshwater biomonitoring and benthic macroinvertebrates***,** 159-194.

Reuss, M. (2005) Ecology, planning, and river management in the united states: Some historical reflections. *Ecology and Society,* **10**.

Rheinhardt, R.D., Rheinhardt, M.C., Brinson, M.M. & Faser, K.E. (1999) Application of reference data for assessing and restoring headwater ecosystems. *Restoration Ecology,* **7,** 241-251.

Richardson, D.M., Holmes, P.M., Esler, K.J., Galatowitsch, S.M., Stromberg, J.C., Kirkman, S.P., Pysek, P. & Hobbs, R.J. (2007) Riparian vegetation: degradation, alien plant invasions, and restoration prospects. *Diversity and Distributions,* **13,** 126-139.

Richter, B.D., Baumgartner, J.V., Braun, D.P. & Powell, J. (1998) A spatial assessment of hydrologic alteration within a river network. *Regulated Rivers-Research & Management,* **14,** 329-340.

Riis, T. & Sand-Jensen, K. (2006) Dispersal of plant fragments in small streams. *Freshwater Biology,* **51,** 274-286.

Riley, A. (2005) What role should landscape architects play in river restoration and flood control projects? *Landscape Architecture,* **95,** 50-50.

Rinaldi, M. & Johnson, P.A. (1997) STREAM MEANDER RESTORATION1. *Journal of the American Water Resources Association,* **33,** 855-866.

Rios, S.L. & Bailey, R.C. (2006) Relationship between riparian vegetation and stream benthic communities at three spatial scales. *Hydrobiologia,* **553,** 153-160.

Robinson, C.T. & Minshall, G.W. (1986) EFFECTS OF DISTURBANCE FREQUENCY ON STREAM BENTHIC COMMUNITY STRUCTURE IN RELATION TO CANOPY COVER AND SEASON. *Journal of the North American Benthological Society,* **5,** 237-248.

Robinson, C.T., Tockner, K. & Burgherr, P. (2004) Drift benthos relationships in the seasonal colonization dynamics of alpine streams. *Archiv Fur Hydrobiologie,* **160,** 447-470.

Rohde, S., Hostmann, M., Peter, A. & Ewald, K.C. (2006) Room for rivers: An integrative search strategy for floodplain restoration. *Landscape and Urban Planning,* **78,** 50-70.

Rohde, S., Kienast, F. & Burgi, M. (2004) Assessing the restoration success of river widenings: A landscape approach. *Environmental Management,* **34,** 574-589.

Rohde, S., Schutz, M., Kienast, F. & Englmaier, P. (2005) River widening: An approach to restoring riparian habitats and plant species. *River Research and Applications,* **21,** 1075-1094.

Roni, P., Hanson, K. & Beechie, T. (2008) Global review of the physical and biological effectiveness of stream habitat rehabilitation techniques. *North American Journal of Fisheries Management,* **28,** 856-890.

Rosenberg, D.M. & Resh, V.H. (1993) Freshwater biomonitoring and benthic macroinvertebrates. *Freshwater biomonitoring and benthic macroinvertebrates***,** ix+488p.

Rosenfeld, J.S., Raeburn, E., Carrier, P.C. & Johnson, R. (2008) Effects of side channel structure on productivity of floodplain habitats for juvenile coho salmon. *North American Journal of Fisheries Management,* **28,** 1108-1119.

Roth, M., Weber, M. & Bezzola, G.R. (2001) Physical modeling of sediment deposits in a river delta case study: The Alpenrhein delta in Lake Constance. *Hydraulics for Maritime Engineering, Theme E, Proceedings - 21St Century: the New Era for Hydraulic Research and Its Applications***,** 187-194.

Ruiz-Jaen, M.C. & Aide, T.M. (2005) Restoration success: How is it being measured? *Restoration Ecology,* **13,** 569-577.

Rumps, J.M., Katz, S.L., Barnas, K., Morehead, M.D., Jenkinson, R., Clayton, S.R. & Goodwin, P. (2007) Stream restoration in the Pacific Northwest: Analysis of interviews with project managers. *Restoration Ecology,* **15,** 506-515.

Salo, J. (1990) EXTERNAL PROCESSES INFLUENCING ORIGIN AND MAINTENANCE OF INLAND WATER-LAND ECOTONES. *Ecology and Management of Aquatic-Terrestrial Ecotones,* **4,** 37-64.

Sarriquet, P.E., Bordenave, P. & Marmonier, P. (2007) Effects of bottom sediment restoration on interstitial habitat characteristics and benthic macroinvertebrate assemblages in a headwater stream. *River Research and Applications,* **23,** 815-828.

Sawyer, A.M., Pasternack, G.B., Merz, J.E., Escobar, M. & Senter, A.E. (2009) Construction Constraints for Geomorphic-Unit Rehabilitation on Regulated Gravel-Bed Rivers. *River Research and Applications,* **25,** 416-437.

Scarsbrook, M.R. & Townsend, C.R. (1993) STREAM COMMUNITY STRUCTURE IN RELATION TO SPATIAL AND TEMPORAL VARIATION - A HABITAT TEMPLET STUDY OF 2 CONTRASTING NEW-ZEALAND STREAMS. *Freshwater Biology,* **29,** 395-410.

Schadler, B. (2005) Synergy between Flood Management and River Ecosystem Protection-15 Years Experiences in Switzerland. *Proceedings of the 2Nd International Yellow River Forum on Keeping Healthy Life of the River, Vol Iv***,** 337-344.

Schiemer, F., Hein, T. & Reckendorfer, W. (2007) Ecohydrology, key-concept for large river restoration. *Ecohydrology & Hydrobiology,* **7,** 101-111.

Schlapfer, F. & Witzig, P.J. (2006) Public support for river restoration funding in relation to local river ecomorphology, population density, and mean income. *Water Resources Research,* **42**.

Schmidt, J.C., Webb, R.H., Valdez, R.A., Marzolf, G.R. & Stevens, L.E. (1998) Science and values in river restoration in the Grand Canyon - There is no restoration or rehabilitation strategy that will improve the status of every riverine resource. *Bioscience,* **48,** 735-747.

Schmitt, L., Maire, G., Nobelis, P. & Humbert, J. (2007) Quantitative morphodynamic typology of rivers: a methodological study based on the French Upper Rhine basin. *Earth Surface Processes and Landforms,* **32,** 1726-1746.

Schmitz, O.J., Grabowski, J.H., Peckarsky, B.L., Preisser, E.L., Trussell, G.C. & Vonesh, J.R. (2008) From individuals to ecosystem function: Toward an integration of evolutionary and ecosystem ecology. *Ecology,* **89,** 2436-2445.

Schmude, K.L., Jennings, M.J., Otis, K.J. & Piette, R.R. (1998) Effects of habitat complexity on macroinvertebrate colonization of artificial substrates in north temperate lakes. *Journal of the North American Benthological Society,* **17,** 73-80.

Schmutz, S. & Jungwirth, M. (1999) Fish as indicators of large river connectivity: the Danube and its tributaries. *Archiv fur Hydrobiologie***,** 329-348.

Schneider, S., Nestmann, F. & Lehmann, B. (2006) Interaction of vegetation, current and sedimentation. *River Flow 2006, Vols 1 and 2***,** 647-653.

Schulz, F. & Wiegleb, G. (2000) Development options of natural habitats in a post-mining landscape. *Land Degradation & Development,* **11,** 99-110.

Schwartz, J.S. & Herricks, E.E. (2007) Evaluation of pool-riffle naturalization structures on habitat complexity and the fish community in an urban Illinois stream. *River Research and Applications,* **23,** 451-466.

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.

Scott, A. (1950) Fighting the deluge. The border floods of August. 1948. *Scottish Agric,* **29,** 127-132.

Sear, D.A., Briggs, A. & Brookes, A. (1998) A preliminary analysis of the morphological adjustment within and downstream of a lowland river subject to river restoration. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **8,** 167-183.

Sear, D.A., Darby, S.E., Thorne, C.R. & Brookes, A.B. (1994) Geomorphological Approach to Stream Stabilization and Restoration - Case-Study of the Mimmshall Brook, Hertfordshire, Uk. *Regulated Rivers-Research & Management,* **9,** 205-223.

Sedell, J.R., Reeves, G.H., Hauer, F.R., Stanford, J.A. & Hawkins, C.P. (1990) ROLE OF REFUGIA IN RECOVERY FROM DISTURBANCES - MODERN FRAGMENTED AND DISCONNECTED RIVER SYSTEMS. *Environmental Management,* **14,** 711-724.

Sellin, R.H.J. & Van Beesten, D.P. (2004) Conveyance of a managed vegetated two-stage channel. *Proceedings of the Institution of Civil Engineers-Water Management,* **157,** 23-35.

Sepa (2008) The River Nith Catchment Plan (incorporating the Lochar Water). Scottish Environmental Protection Agency, Dumfries.

Shields, F.D. (1998) Woody vegetation in river restoration: Problems and opportunities. *Water Resources Engineering 98, Vols 1 and 2***,** 660-665.

Shields, F.D., Knight, S.S. & Cooper, C.M. (2007) Can warmwater streams be rehabilitated using watershed-scale standard erosion control measures alone? *Environmental Management,* **40,** 62-79.

Shields, F.D., Knight, S.S. & Stofleth, J.M. (2008) Stream bed organic carbon and biotic integrity. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **18,** 761-779.

Shields, F.D., Pezeshki, S.R., Wilson, G.V., Wu, W.M. & Dabney, S.M. (2008) Rehabilitation of an Incised Stream Using Plant Materials: the Dominance of Geomorphic Processes. *Ecology and Society,* **13**.

Simons, J.H.E.J., Bakker, C., Schropp, M.H.I., Jans, L.H., Kok, F.R. & Grift, R.E. (2001) Man-made secondary channels along the River Rhine (the Netherlands); results of post-project monitoring. *Regulated Rivers-Research & Management,* **17,** 473-491.

Simons, R.K. & Simons, D.B. (1998) River restoration and rehabilitation. *Tailings and Mine Waste '98***,** 701-710.

Skinner, K.S. & Bruce-Burgess, L. (2005) Strategic and project-level river restoration protocols-key components for meeting the requirements of the water framework directive (WFD). *Water and Environment Journal,* **19,** 135-142.

Slate, L.O., Shields, F.D., Schwartz, J.S., Carpenter, D.D. & Freeman, G.E. (2007) Engineering design standards and liability for stream channel restoration. *Journal of Hydraulic Engineering-Asce,* **133,** 1099-1102.

Soderstrom, O. (1987) UPSTREAM MOVEMENTS OF INVERTEBRATES IN RUNNING WATERS - A REVIEW. *Archiv Fur Hydrobiologie,* **111,** 197-208.

Sondergaard, M. & Jeppesen, E. (2007) Anthropogenic impacts on lake and stream ecosystems, and approaches to restoration. *Journal of Applied Ecology,* **44,** 1089-1094.

Spanhoff, B. & Arle, J. (2007a) Setting attainable goals of stream habitat restoration from a macroinvertebrate view. *Restoration Ecology,* **15,** 317-320.

Spanhoff, B. & Arle, J. (2007b) Setting Attainable Goals of Stream Habitat Restoration from a Macroinvertebrate View. *Restoration Ecology,* **15,** 317-320.

Speaker, R.W., Luchessa, K.J., Franklin, J.F. & Gregory, S.V. (1988) THE USE OF PLASTIC STRIPS TO MEASURE LEAF RETENTION BY RIPARIAN VEGETATION IN A COASTAL OREGON STREAM. *American Midland Naturalist,* **120,** 22-31.

Steiger, J., Tabacchi, E., Dufour, S., Corenblit, D. & Peiry, J.L. (2005) Hydrogeomorphic processes affecting riparian habitat within alluvial channel-floodplain river systems: A review for the temperate zone. *River Research and Applications,* **21,** 719-737.

Stein, C., Auge, H., Fischer, M., Weisser, W.W. & Prati, D. (2008) Dispersal and seed limitation affect diversity and productivity of montane grasslands. *Oikos,* **117,** 1469-1478.

Stewartoaten, A., Murdoch, W.W. & Parker, K.R. (1986) ENVIRONMENTAL-IMPACT ASSESSMENT - PSEUDOREPLICATION IN TIME. *Ecology,* **67,** 929-940.

Storey, A.W. & Lynas, J. (2007) Application of the functional habitat concept to the regulated Lower Ord River, Western Australia, part I, macroinvertebrate assemblages. *Hydrobiologia,* **592,** 499-512.

Strayer, D.L. (2006) Challenges for freshwater invertebrate conservation. *Journal of the North American Benthological Society,* **25,** 271-287.

Sudduth, E.B., Meyer, J.L. & Bernhardt, E.S. (2007) Stream restoration practices in the southeastern United States. *Restoration Ecology,* **15,** 573-583.

Suding, K.N., Gross, K.L. & Houseman, G.R. (2004) Alternative states and positive feedbacks in restoration ecology. *Trends in Ecology & Evolution,* **19,** 46-53.

Suren, A.M. (1991) BRYOPHYTES AS INVERTEBRATE HABITAT IN 2 NEW-ZEALAND ALPINE STREAMS. *Freshwater Biology,* **26,** 399-418.

Suren, A.M., Riis, T., Biggs, B.J.F., Mcmurtrie, S. & Barker, R. (2005) Assessing the effectiveness of enhancement activities in urban streams: I. Habitat responses. *River Research and Applications,* **21,** 381-401.

Suren, A.M., Smart, G.M., Smith, R.A. & Brown, S.L.R. (2000) Drag coefficients of stream bryophytes: experimental determinations and ecological significance. *Freshwater Biology,* **45,** 309-317.

Swales, S. & Ohara, K. (1980) Instream Habitat Improvement Devices and Their Use in Fresh-Water Fisheries Management. *Journal of Environmental Management,* **10,** 167-179.

Tabacchi, E., Correll, D.L., Hauer, R., Pinay, G., Planty-Tabacchi, A.M. & Wissmar, R.C. (1998) Development, maintenance and role of riparian vegetation in the river landscape. *Freshwater Biology,* **40,** 497-516.

Tabacchi, E., Planty-Tabacchi, A.M., Roques, L. & Nadal, E. (2005) Seed inputs in riparian zones: Implications for plant invasion. *River Research and Applications,* **21,** 299-313.

Taniguchi, H. & Tokeshi, M. (2004) Effects of habitat complexity on benthic assemblages in a variable environment. *Freshwater Biology,* **49,** 1164-1178.

Temperton, V.M. (2007) The Recent Double Paradigm Shift in Restoration Ecology. *Restoration Ecology,* **15,** 344-347.

Thomson, J.R., Taylor, M.P., Fryirs, K.A. & Brierley, G.J. (2001) A geomorphological framework for river characterization and habitat assessment. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **11,** 373-389.

Thorndycraft, V.R., Benito, G. & Gregory, K.J. (2008) Fluvial geomorphology: A perspective on current status and methods. *Geomorphology,* **98,** 2-12.

Tilman, D. (1989) ECOLOGICAL EXPERIMENTATION STRENGTHS AND CONCEPTUAL PROBLEMS. *Likens, G. E. (Ed.). Long-Term Studies in Ecology: Approaches and Alternatives; 2nd Cary Conference, Millbrook, New York, USA, May 13, 1987. Xvi+214p. Springer-Verlag New York, Inc.: New York, New York, USA; Berlin, West Germany. Illus. Maps***,** 136-157.

Tockner, K. & Schiemer, F. (1997) Ecological aspects of the restoration strategy for a river-floodplain system on the Danube River in Austria. *Global Ecology and Biogeography Letters,* **6,** 321-329.

Tockner, K., Schiemer, F., Baumgartner, C., Kum, G., Weigand, E., Zweimuller, I. & Ward, J.V. (1999) The Danube restoration project: Species diversity patterns across connectivity gradients in the floodplain system. *Regulated Rivers-Research & Management,* **15,** 245-258.

Tockner, K., Schiemer, F. & Ward, J.V. (1998) Conservation by restoration: The management concept for a river-floodplain system on the Danube River in Austria. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **8,** 71-86.

Tokita, K. (2006) Statistical mechanics of relative species abundance. *Ecological Informatics,* **1,** 315-324.

Tompkins, M.R. & Kondolf, G.M. (2007) Systematic postproject appraisals to maximise lessons learned from river restoration projects: Case study of compound channel restoration projects in Northern California. *Restoration Ecology,* **15,** 524-537.

Townsend, C.R., Doledec, S. & Scarsbrook, M.R. (1997) Species traits in relation to temporal and spatial heterogeneity in streams: A test of habitat templet theory. *Freshwater Biology,* **37,** 367-387.

Townsend, C.R. & Hildrew, A.G. (1976) FIELD EXPERIMENTS ON DRIFTING, COLONIZATION AND CONTINUOUS REDISTRIBUTION OF STREAM BENTHOS. *Journal of Animal Ecology,* **45,** 759-772.

Townsend, C.R., Scarsbrook, M.R. & Doledec, S. (1997) The intermediate disturbance hypothesis, refugia, and biodiversity in streams. *Limnology and Oceanography,* **42,** 938-949.

Tullos, D.D., Penrose, D.L., Jennings, G.D. & Cope, W.G. (2009) Analysis of functional traits in reconfigured channels: implications for the bioassessment and disturbance of river restoration. *Journal of the North American Benthological Society,* **28,** 80-92.

Urban, M.C., Skelly, D.K., Burchsted, D., Price, W. & Lowry, S. (2006) Stream communities across a rural-urban landscape gradient. *Diversity and Distributions,* **12,** 337-350.

Van Looy, K., Jacquemyn, H., Breyne, P. & Honnay, O. (2009) Effects of flood events on the genetic structure of riparian populations of the grassland plant Origanum vulgare. *Biological Conservation,* **142,** 870-878.

Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R. & Cushing, C.E. (1980) 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,** 113-125.

Vinson, M.R. (2001) Long-term dynamics of an invertebrate assemblage downstream from a large dam. *Ecological Applications,* **11,** 711-730.

Vivash, R., Ottosen, O., Janes, M. & Sorensen, H.V. (1998) Restoration of the rivers Brede, Cole and Skerne: a joint Danish and British EU-LIFE demonstration project, II - The river restoration works and other related practical aspects. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **8,** 197-208.

Walker, J., Diamond, M. & Naura, M. (2002) The development of Physical Quality Objectives for rivers in England and Wales. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **12,** 381-390.

Wallick, J.R., Lancaster, S.T. & Bolte, J.P. (2006) Determination of bank erodibility for natural and anthropogenic bank materials using a model of lateral migration and observed erosion along the Willamette River, Oregon, USA. *River Research and Applications,* **22,** 631-649.

Walsh, C.J., Waller, K.A., Gehling, J.A.N.E. & Nally, R.M. (2007) Riverine invertebrate assemblages are degraded more by catchment urbanisation than by riparian deforestation. *Freshwater Biology,* **52,** 574-587.

Ward, J.V. (1998) Riverine landscapes: Biodiversity patterns, disturbance regimes, and aquatic conservation. *Biological Conservation,* **83,** 269-278.

Ward, J.V., Tockner, K., Arscott, D.B. & Claret, C. (2002) 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,** 709-+.

Warwick, R.M. & Clarke, K.R. (1998) Taxonomic distinctness and environmental assessment. *Journal of Applied Ecology,* **35,** 532-543.

Waters, T.F. (1965) INTERPRETATION OF INVERTEBRATE DRIFT IN STREAMS. *Ecology,* **46,** 327-334.

Weber, C., Woolsey, S. & Peter, A. (2006) A strategy to assess river restoration success. *Archives des Sciences,* **59,** 251-256.

Weichert, R.B., Bezzola, G.R. & Minor, H.E. (2009) Bed erosion in steep open channels. *Journal of Hydraulic Research,* **47,** 360-371.

Weiher, E. (2007) On the Status of Restoration Science: Obstacles and Opportunities. *Restoration Ecology,* **15,** 340-343.

Wene, G. & Wickliff, E.L. (1940) Modification of a stream bottom and its effect on the insect fauna. *Canadian Ent,* **72,** 131-135.

Western, A.W., Finlayson, B.L., Mcmahon, T.A. & Oneill, I.C. (1997) A method for characterising longitudinal irregularity in river channels. *Geomorphology,* **21,** 39-51.

Wharton, G. (1995) The Channel-Geometry Method - Guidelines and Applications. *Earth Surface Processes and Landforms,* **20,** 649-660.

Wheaton, J.M., Darby, S.E., Sear, D.A. & Milne, J.A. (2006) Does scientific conjecture accurately describe restoration practice? - Insight from an international river restoration survey. *Area,* **38,** 128-142.

Wiens, J.A. (2002) Riverine landscapes: taking landscape ecology into the water. *Freshwater Biology,* **47,** 501-515.

Wilkinson, D.M. (1999) The disturbing history of intermediate disturbance. *Oikos,* **84,** 145-147.

Williams, D.D. & Smith, M.R. (1996) Colonization dynamics of river benthos in response to local changes in bed characteristics. *Freshwater Biology,* **36,** 237-248.

Williams, J.B. & Poff, N.L. (2006) Informatics software for the ecologist's toolbox: A basic example. *Ecological Informatics,* **1,** 325-329.

Williams, L., Reich, P., Capon, S.J. & Raulings, E. (2008) Soil seed banks of degraded riparian zones in southeastern Australia and their potential contribution to the restoration of understorey vegetation. *River Research and Applications,* **24,** 1002-1017.

Wintle, B.C. & Kirkpatrick, J.B. (2007) The response of riparian vegetation to flood-maintained habitat heterogeneity. *Austral Ecology,* **32,** 592-599.

Wise, E.J. (1980) SEASONAL DISTRIBUTION AND LIFE HISTORIES OF EPHEMEROPTERA IN A NORTHUMBRIAN RIVER. *Freshwater Biology,* **10,** 101-111.

Wohl, E. (2005) Compromised rivers: Understanding historical human impacts on rivers in the context of restoration. *Ecology and Society,* **10**.

Wohl, E. (2006) Human impacts to mountain streams. *Geomorphology,* **79,** 217-248.

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**.

Wohl, E. & Merritts, D.J. (2007) What Is a Natural River? *Geography Compass,* **1,** 871-900.

Wood, P.J. & Armitage, P.D. (1997) Biological effects of fine sediment in the lotic environment. *Environmental Management,* **21,** 203-217.

Woolsey, S., Capelli, F., Gonser, T., Hoehn, E., Hostmann, M., Junker, B., Paetzold, A., Roulier, C., Schweizer, S., Tiegs, S.D., Tockner, K., Weber, C. & Peter, A. (2007) A strategy to assess river restoration success. *Freshwater Biology,* **52,** 752-769.

Zalewski, M., Bis, B., Lapinska, M., Frankiewicz, P. & Puchalski, W. (1998) The importance of the riparian ecotone and river hydraulics for sustainable basin-scale restoration scenarios. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **8,** 287-307.

CHAPTER 13

APPENDIX

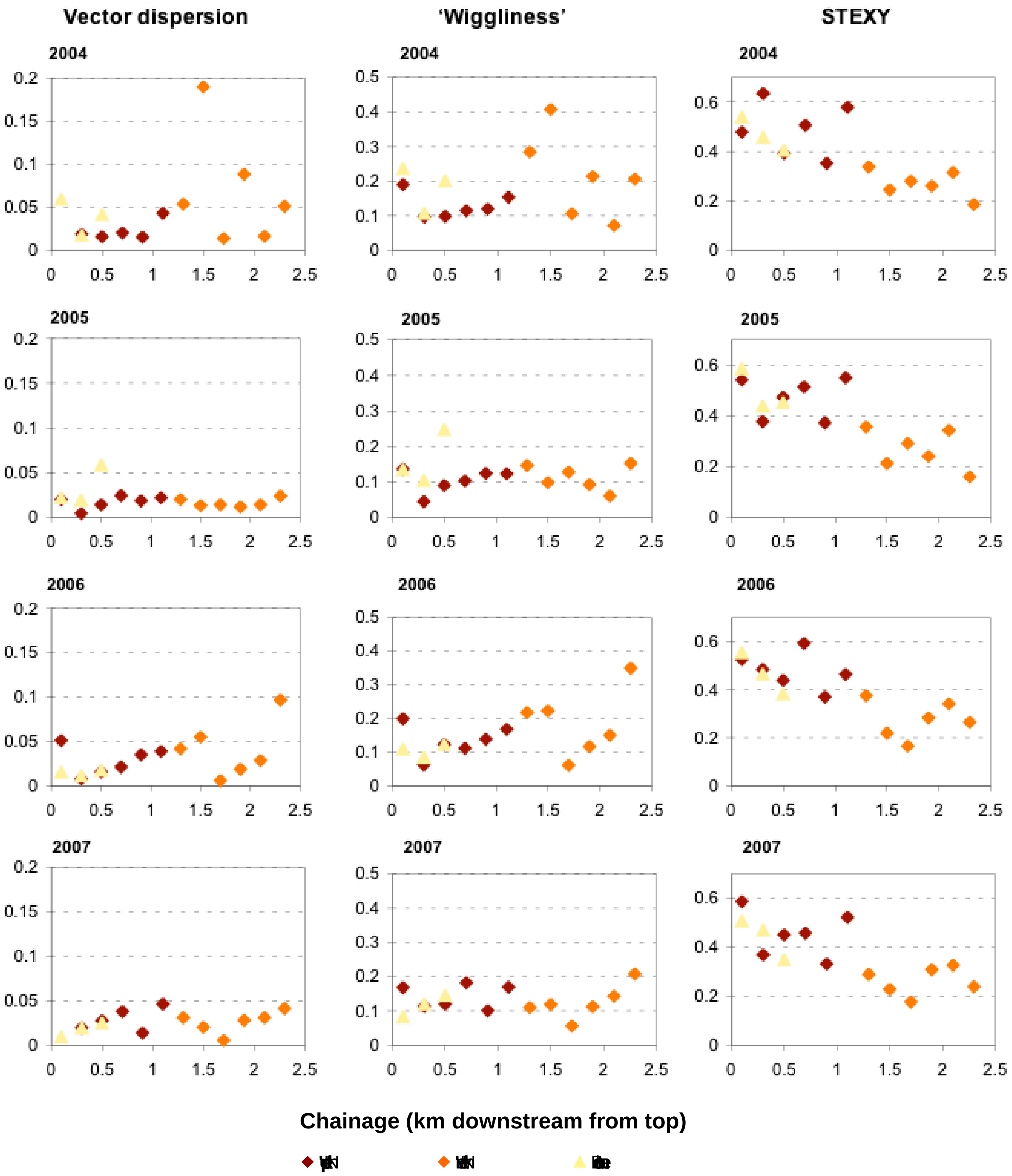


Figure A.1: Shows the development of physical habitat along the length of the realignment as detected by three recommended indices of diversity; Vector dispersion, 'wiggliness' and STEXY. Sampling intensity is standardised to 18 points per reach. The chainage (km) is plotted on the x-axis against the index score on the y-axis

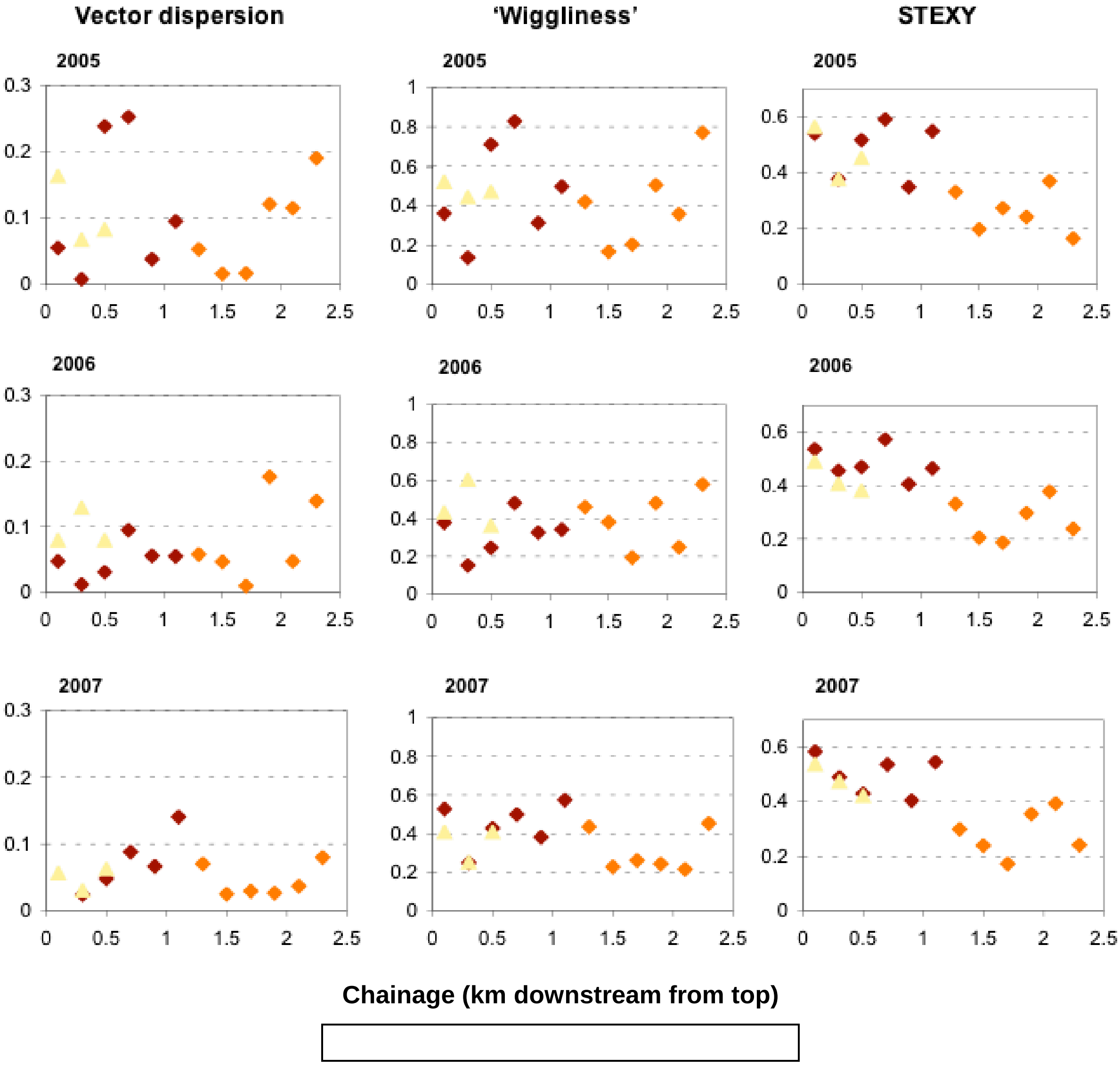


Figure A.2: Shows the repeated analysis at a sampling intensity standardised to 33 points per reach. The development of physical habitat along the length of the realignment is preented as detected by three recommended indices of diversity; Vector dispersion, 'wiggliness' and STEXY. The chainage (km) is plotted on the x-axis against the index score on the y-axis

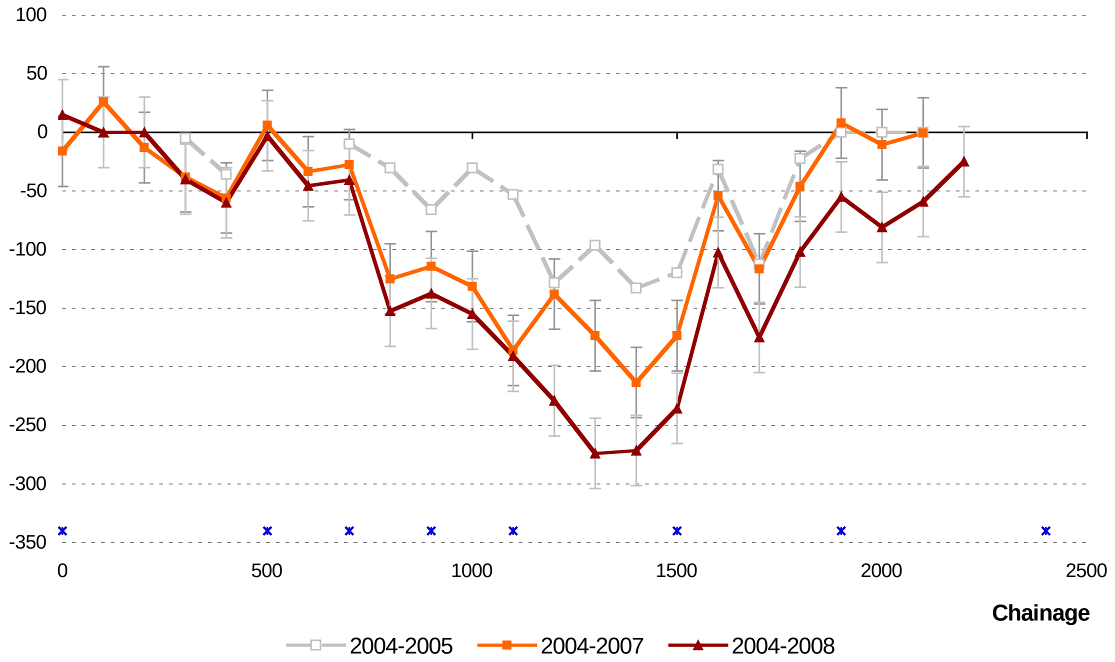


Figure A.3: Graphs shows the degree of bed settlement along the length of the realignment at 100m channel intervals from the top of the Nith tributary. Ground settlement shown in mm as an average of movement recorded at left and right bankfull bank-tops. The greatest level of settlement is recorded for chainage1200m to 1400m. The degree of settlement in this region is around 250mm is in line with the predicted best case scenario - α=0.5% (Halcrow, 2004) Error bars show the accuracy of the dGPS equipment (±30mm) Blue asterisks show approximate locations of invertebrate sampling locations

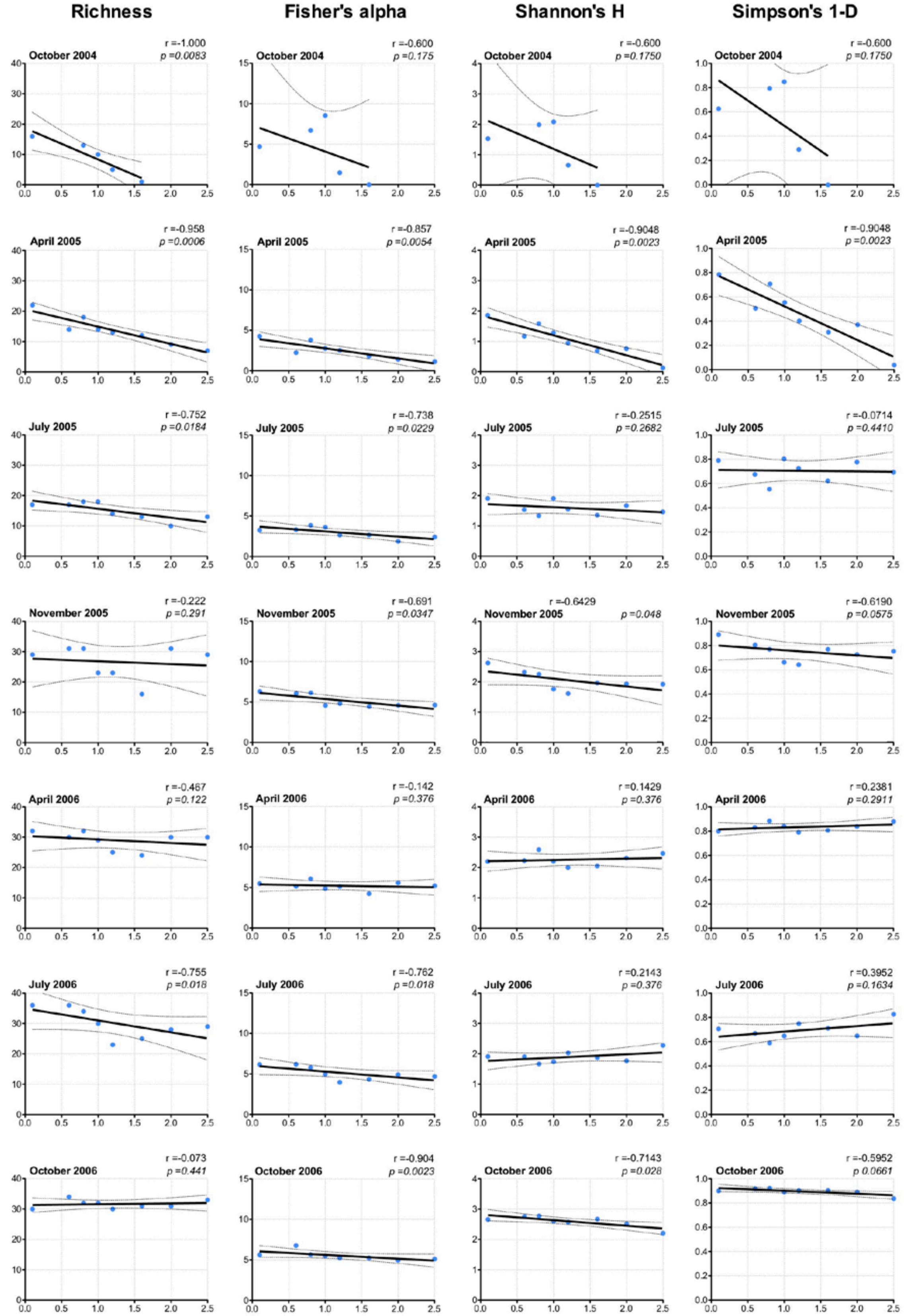


Figure A.4: Graphs showing the patterns of colonisation along the length of the diversion as indicated by a range of diversity indices applied to FAMILY abundance data; richness, Fisher's alpha, Shannon's H and Simpson's 1-D. Distances downstream from the top of the Nith Realignment (x-axis) are plotted against index scores (y-axis) for each invertebrate sample collected

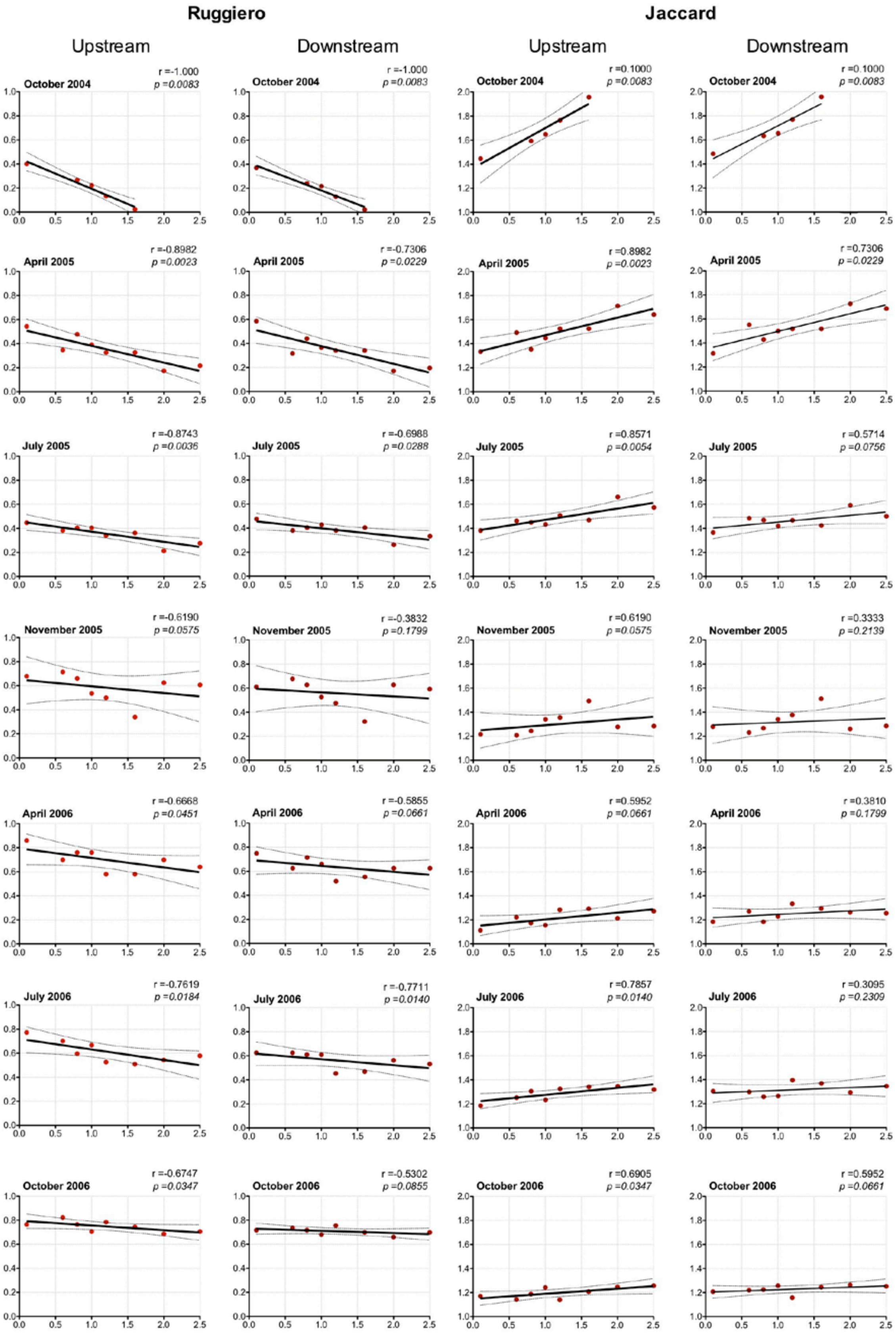


Figure A.5 Beta diversity measures (y axis) based on presence-absence SPECIES data indicating similarity (Ruggiero) and dissimilarity (Jaccard) to the upstream and downstream invertebrate populations of site along the length of the realignment (x axis)

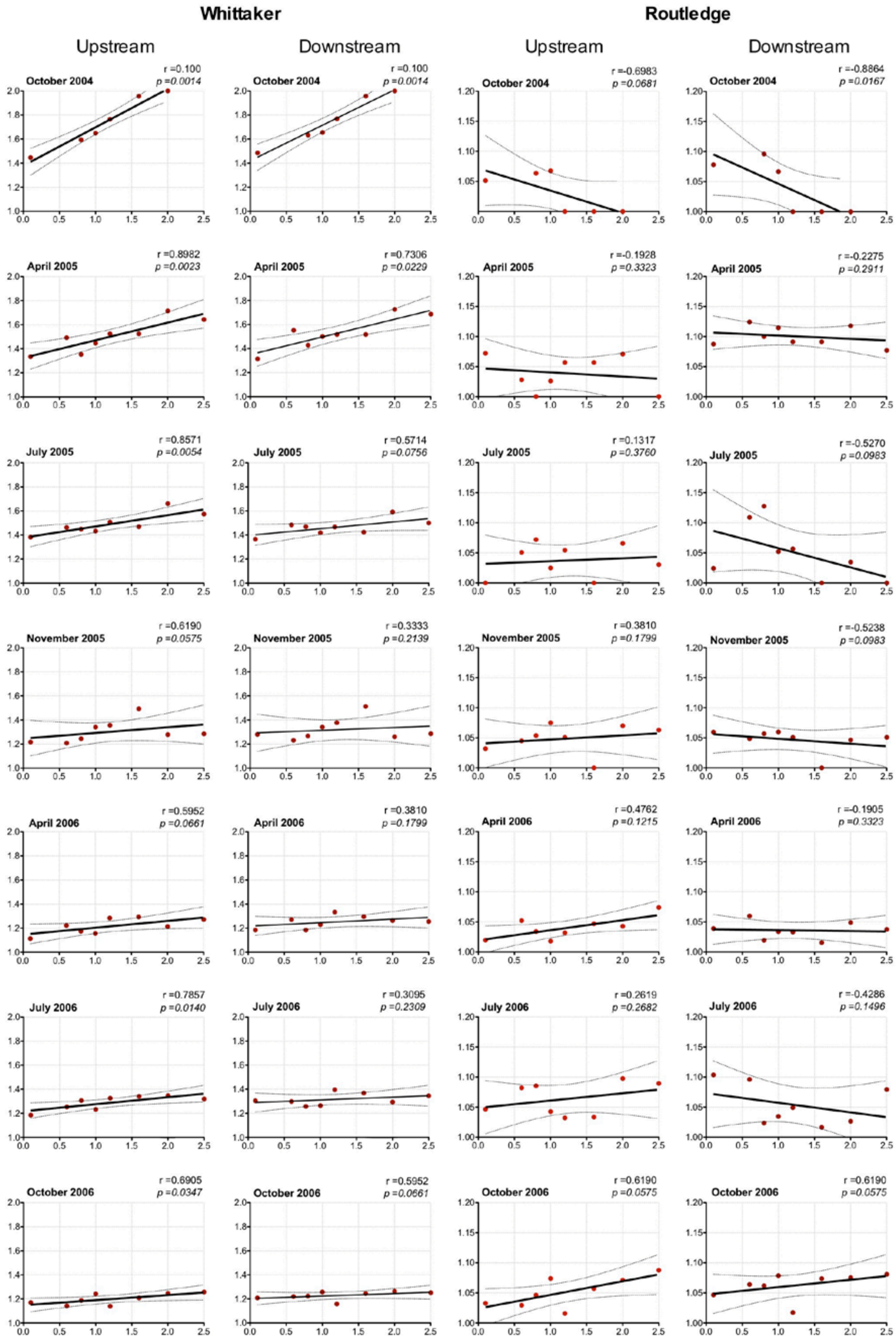


Figure A.6 Beta diversity measures (y axis) based on presence-absence SPECIES data indicating dissimilarity (Whittaker) and similarity (Routledge) to the upstream and downstream invertebrate populations of site along the length of the realignment (x axis)

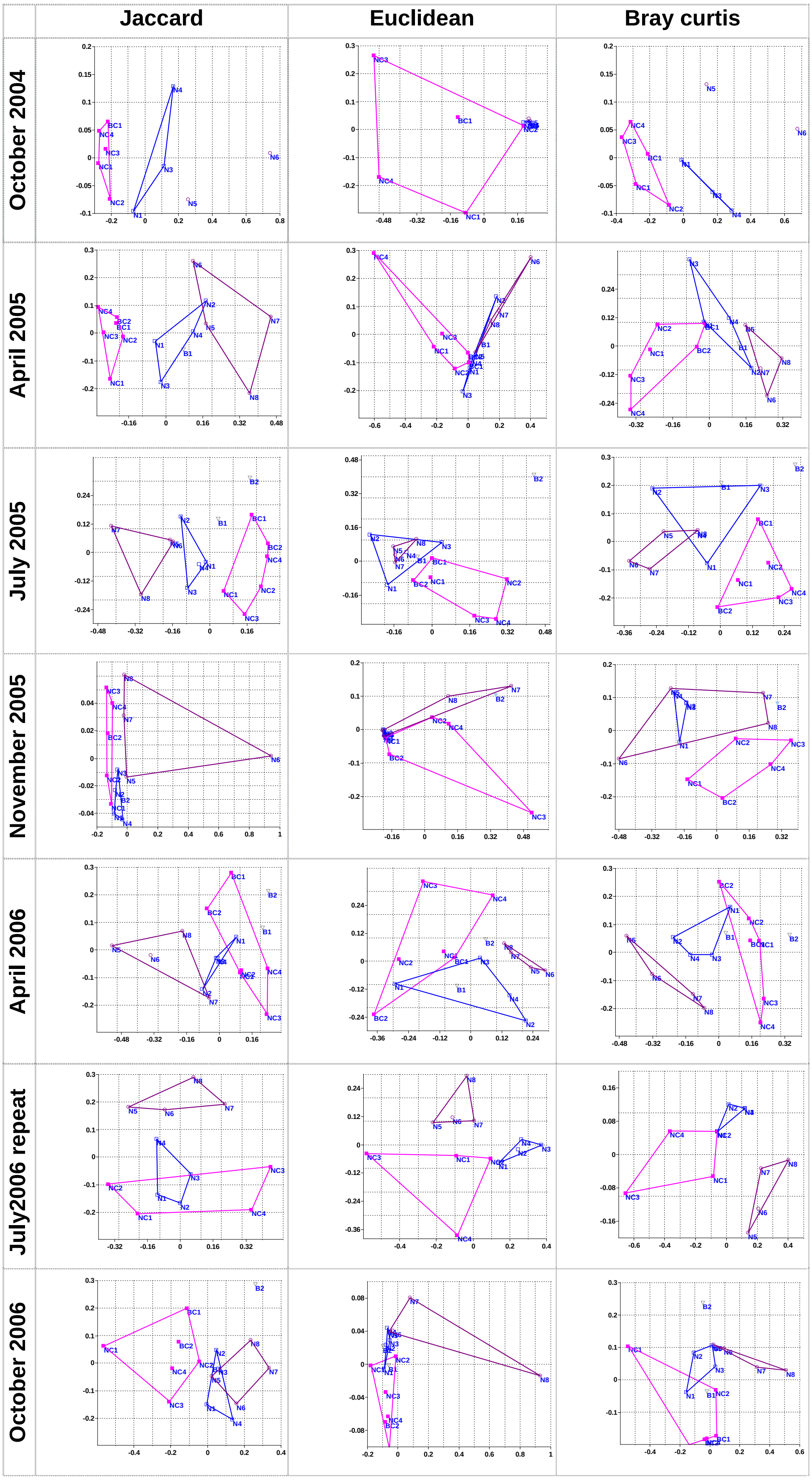


Figure A.7. NMDS ordination plots showing the dissimilarities between sites based on three distance measures; Jaccard (presence-absence), Ecuclidean distance and Bray-curtis dissimilarity. Patterns are shown for each invertebrate survey and sites are grouped into 'upper', 'lower' and 'reference' classes.

Figure A.8. Bar charts showing the richness of different biological trait groups for different sampling sites along the realignment on different survey dates

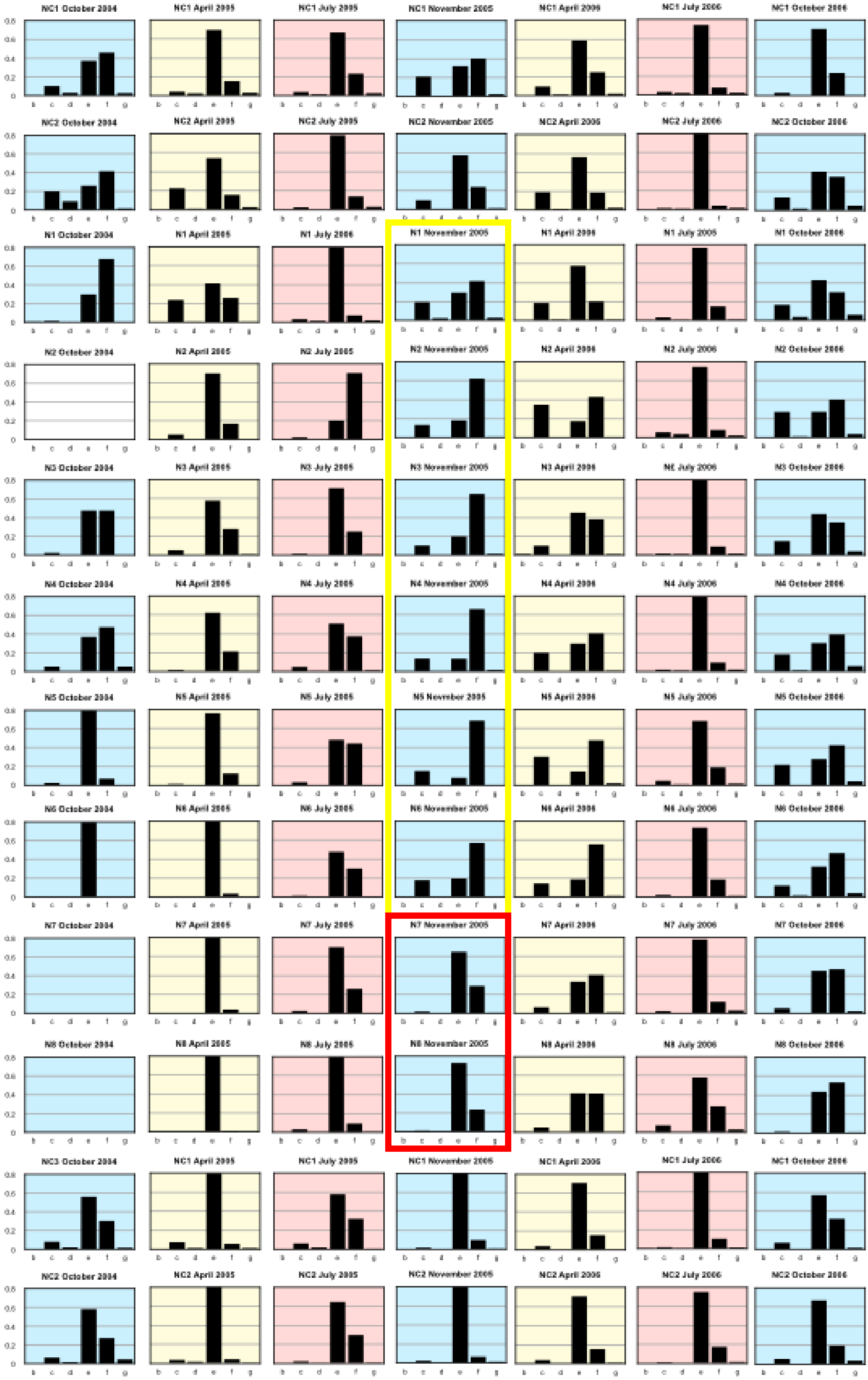


Figure A.9. Bar charts showing the abundance of different biological trait groups for different sampling sites along the realignment on different survey dates

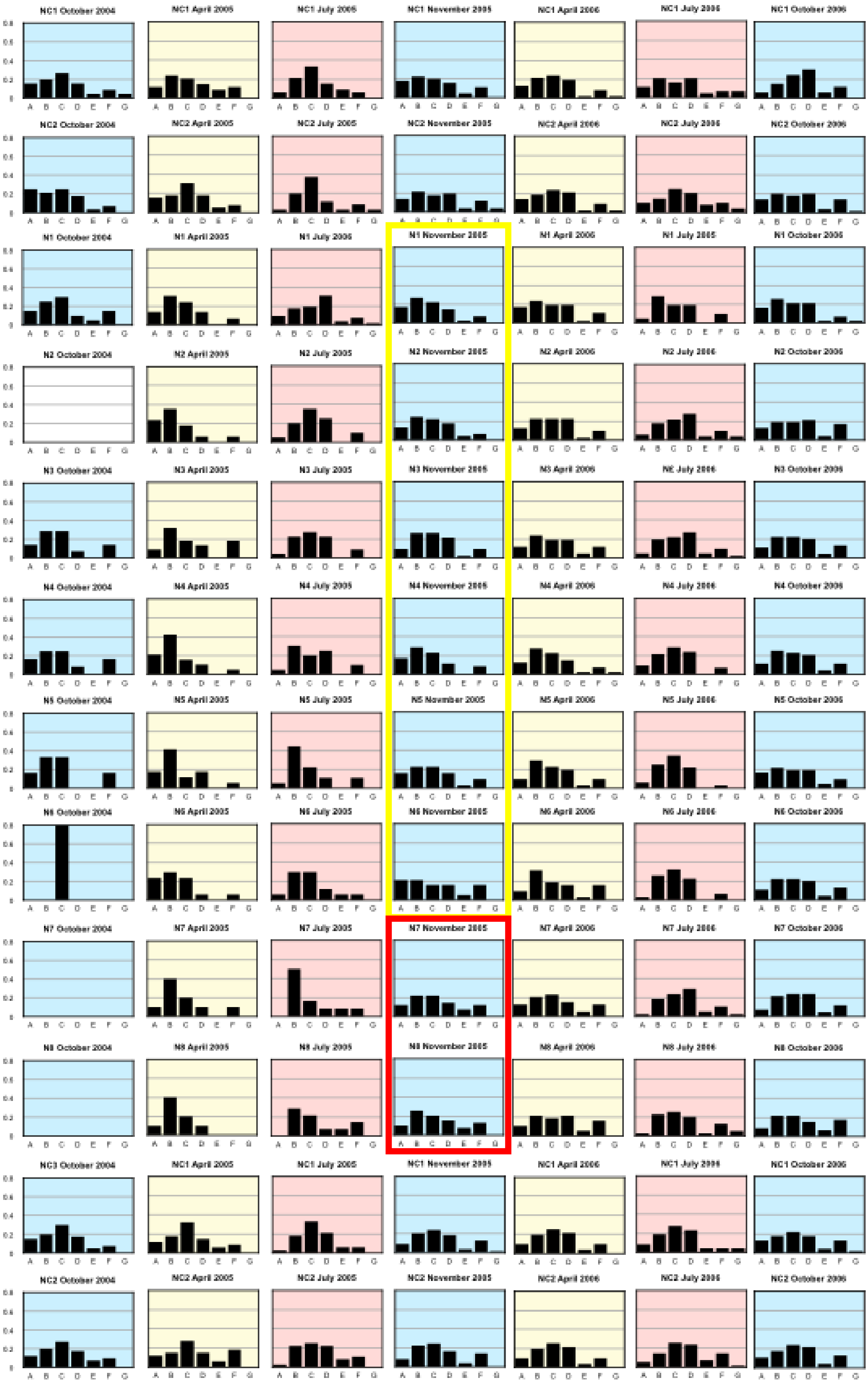


Figure A.10. Bar charts showing the richness of different ecological trait groups for different sampling sites along the realignment on different survey dates

Figure A.11. Bar charts showing the abundance of different ecological trait groups for different sampling sites along the realignment on different survey dates