CATCHMENT SCALE INFLUENCES ON BROWN TROUT FRY POPULATIONS IN THE UPPER URE CATCHMENT, NORTH YORKSHIRE.

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Catchment-scale influences on brown trout fry populations in the upper Ure catchment.

A multi-scale approach for restoration site selection is presented and applied to an upland catchment, the River Ure, North Yorkshire. Traditional survey methods, advances in remote sensing, Geographical Information Systems (GIS) and risk-based fine sediment modelling using the SCIMAP module are combined to gather data at the catchment-scale through to the in-stream habitat-scale. The data gathered have been assessed against spatially distributed brown trout fry populations using Pearson's correlation and multiple stepwise regressions.

Fine sediment was shown to have a positive correlation with fry populations when upland drainage channels (grips) were added to the SCIMAP model. This suggests risk from peatland drainage is realised further down the catchment where eroded sediments are deposited. Farm-scale SCIMAP modelling was tested against farmers' knowledge with variable results. It appears there is a cultural response to risk developed over generations. Management of meadows and pasture land through sub-surface drainage and stock rotation resulted in the risk being negated or re-routed across the holding. At other locations apparently low-risk zones become risky through less sensitive farming methods.

This multi-scale approach reveals that the largest impacts on brown trout recruitment operate at the habitat-adjacent scale in tributaries with small upstream areas. The results show a hierarchy of impact, and risk-filters, arising from different intensity land management. This offers potential for targeted restoration site selection. In low-order streams it seems that restoration measures which exclude livestock, and provide bankside shading, can be effective. At such sites the catchment-scale shows a reduced signal on in-stream biota. Thus, brown trout stocks could be significantly enhanced by targeting restoration at riffle-habitat zones and adjacent land in order to disconnect the stream from farm-derived impacts and through adding structure to the stream channel.

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Declaration of copyright

I confirm that no part of the material presented in this thesis has previously been submitted by me or any other person for a degree in this or any other university. In all cases, where it is relevant, material from the work of others has been acknowledged.

Signed: D. Higgins

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Acronyms

ART	Association of Rivers Trusts
BOD	Biological Oxygen Demand
CEH	Centre for Ecology and Hydrology
CSA	Critical Source Area
CSS	Countryside Stewardship Scheme
DEM	Digital Elevation Model
DO	Dissolved Oxygen
DOC	Dissolved Organic Carbon
ELS	Entry Level Scheme
EA	Environment Agency
ES	Environmental Stewardship
ESA	Environmentally Sensitive Area
EU	European Union
GIS	Geographical Information Systems
HLS	Higher Level Scheme
LCM	Land Cover Map
LIFE	Lotic-invertebrate Index for Flow Evaluation
NE	Natural England
NGO	Non-Governmental Organisation
PC	Principal Components
POM	Particulate Organic Matter
RSPB	Royal Society for the Protection of Birds
SAC	Special Area of Conservation
SCIMAP	Sensitive Catchment Integrated Modelling and Analysis Platform
$SCIMAP_G$	SCIMAP weighted by land use and with grips added to the model
$SCIMAP_L$	SCIMAP weighted by land use
$SCIMAP_U$	SCIMAP unweighted by land use
SSSI	Site of Special Scientific Interest
UELS	Upland Entry Level Scheme
WFD	Water Framework Directive
WWTW	Waste Water Treatment Works
YDNP	Yorkshire Dales National Park
YDRT	Yorkshire Dales Rivers Trust

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Chapter 1: Aim and Objectives of the Thesis

The aim of this research is to combine advances in remote sensing. Geographical Information Systems (GIS), catchment-scale modelling and ecological survey techniques with current awareness of salmonid species, specifically brown trout fry populations (*Salmo trutta*), to develop an effective approach to the ecological restoration of habitat through the prioritisation of location and management options. The research will be developed and applied to the upper Ure catchment which has a resident brown trout stock cut off from upstream migration of anadromous[1] forms by Aysgarth Falls a series of three natural waterfalls. The aim of this research will be achieved through the following objectives:

Objective 1: To review and synthesise in-stream, riparian and catchment scale controls on salmonid habitat, focusing on brown trout fry populations, in order to formulate a set of hypothesis for further investigation.

Objective 2: To employ advances in remote sensing, GIS and modelling to explore land use risk at the catchment scale that links to the in-stream habitat scale, in particular the risk of fine sediment delivery from the wider catchment.

Objective 3: To identify qualitative methods in data-poor catchments for testing model predictions and to employ the experience of agricultural communities in testing these predictions.

Objective 4: To use the data acquired under 2 to investigate hypothesis formulated in 1 to test which impacts on brown trout fry populations are important and to discuss the results in the context of model testing and ecological restoration.

Conceptualising the linkages between catchments, land use and brown trout populations is important in order to understand both process and response and to predict future population dynamics under scenarios of land use change including intensification, and extensification, as well as wider ecosystem pressure such as may arise from climate

[1] Anadromous forms of brown trout (sea trout) follow a similar life cycle to atlantic salmon (*Salmo salar*) by migrating to coastal zones at ages above 2+ to 6+, in order to feed, returning to their natal river systems to spawn (Kallio-Nyberg *et al.*, 2002). Unlike atlantic salmon sea trout are able to perform this migratory feat numerous times.

change. Without this level of understanding on catchment and ecosystem response systems may respond to pressures in unpredictable ways. As threshold dominated systems upland rivers can rapidly flip between states resulting in new systems that are difficult to reverse. This can result in restoration effort becoming overly complicated often carried out on an ad hoc basis with success based on little more than chance and good fortune.

It is clear that brown trout utilise a wide habitat range and type with specific life-stage dependent requirements. Whilst there are specific habitat types required at different life stages, there is no definitive cut-off point and habitat requirements overlap. If brown trout are to be effectively conserved, then conservation and restoration must consider all the processes acting on the river habitat at all stages of the life cycle following the species throughout its migratory routes. However, there may be critical life stages that place strong controls on the whole population. If these are recognised, then conservation effort should concentrate on improving the success of specific life-stage populations to enable these to pass through potential bottlenecks. With brown trout, it appears that the life stages most vulnerable to impacts occur between the egg and fry stage due to a combination of low dispersal ability and high sensitivity to pollution. This research focuses on brown trout fry survival due to their response to impacts coupled with the ability to carry out relatively simple and rapid surveys at specific habitat types, to be discussed in later chapters.

Chapter 1 explores the context and conceptualisation of the project in terms of brown trout requirements, catchment-scale investigations, restoration ecology, ecohydrology, connectivity and the legal drivers for large-scale research. In so doing, the research will be justified in terms of research gaps as well as economic and environmental drivers.

1.1 Justification of research

This thesis will explore brown trout fry populations in terms of multiple pressures on their populations in order to decipher which pressures matter, where they act and at what scale they operate. In so doing there will be an attempt to test fine sediment modelling through farmers experience of the landscape and through ecological components of rivers, notably brown trout fry. Testing model predictions on organisms is not a straightforward exercise yet Lane (2008) argues that it is important to consider what organisms tell us through their behaviour, and survival, when placed under pressure. If it is possible to utilise species of interest in order to test model predictions, then confidence in their ability to make useful predictions at other locations can be developed.

The following paragraphs will explore scientific and policy drivers that justify the theme of this research and provide a brief overview of brown trout fry ecology in preparation for subsequent chapters. It will also introduce the case study catchment and explain why it was a focus for the research notwithstanding some difficulties that were immediately apparent. Chapter three will provide a full overview of the case study catchment. The final section will provide a summary of the subsequent chapters in terms of the objectives and aim of this thesis.

Reductionist science has raised awareness of the impacts on freshwater organisms that has been invaluable in understanding how individual pressures affect organisms. This has provided useful insight into the impacts of fine sediment, changes in hydrological regimes, acid flushes and nutrient export. However, reductionist science has failed to help river managers and restoration ecologists understand how these individual impacts interact, and which are the most important, in natural systems given issues of multiple scales and processes. Understanding the river in terms of the whole catchment requires different approaches.

Riverscapes[2]are connected to the land by hydrological pathways, stock access and land use and thus are easily affected by multiple pressures (Fausch, 2010) that are the result of complex interactions between socioeconomic and natural systems (Hart and Calhoun, 2010). As a consequence, identifying which stressors are posing the strongest limiting factor on organisms, species and communities is difficult (Heathwaite, 2010). Overcoming these complexities may well require the development of new methods and approaches to the sciences of freshwater ecology, ecological restoration, and hydrology that could result in the dissolution of disciplinary boundaries to allow new disciplines to

[2] Haslam (2008, p. 2) defines the riverscape as the, ‘sheet of water that covers the land; in whole or in part, permanently of intermittently...(it) is that part of the land that has (or had) a watercourse as its focus’. Here the riverscape takes a similar definition and includes vertical, horizontal, longitudinal and temporal hydrological connectivity at the stages that they have ecological significance to the river.

emerge (Lane *et al.*, 2006). Ecohydrology[3] is an emerging science that aims to combine hydrology and ecology in order to investigate process-driven impacts on freshwater ecosystems. The development of this transdisciplinary science provides a context for research into river catchments that covers all the embedded scales, multiple pressures and river ecosystems as the recipients of impacts (Petts and Morales, 2006).

During the 1970s and 1980s there developed a new approach to understanding rivers. Streams and river stretches were originally considered to be discrete, individual entities (Minshall *et al.*, 1985) whilst the new perspective viewed them as a continuum. The River Continuum Concept (RRC, Vannote *et al.*, 1980) frames the fluvial system as a, '*continuously integrating series of physical gradients and associated biotic adjustments...(streams are)...longitudinally linked systems in which ecosystem level processes in downstream areas are linked to those in upstream areas,*' (Minshall *et al.*, p. 1046, 1985). In conjunction, the concepts of the RRC and nutrient spiralling reframed freshwater ecology and hydrological approaches to rivers to one of a continuous and interacting series of biological and physical processes across a stream gradient (Minshall *et al.*, 1985). This conceptualisation of rivers as a connected continuum with interactions between reaches is a forerunner to the much more recent concept of ecohydrology (discussed later).

The reality of river function is likely to lie somewhere between continuum and discrete sections as individual entities. Rice *et al.* (2001) offered empirical evidence of the river "discontinuum" (Poole 2002) by revealing how tributary confluences reset invertebrate communities. Land use can be added to the complexities of river systems to add further intricacy to freshwater systems (Newsom, 2010). Severe impacts arising from point or diffuse pollution sources can create discrete sections of river ecologically distinct from the immediate upstream setting. Dams and larger impoundments act to create obvious situations of river discontinuum. Low flows arising from drainage in upland regions for example act to limit longitudinal and horizontal connectivity thus reducing habitat

[3] Ecohydrology is defined as a discipline that 'seeks to understand the interactions between the hydrological cycle and ecosystems. The influence of hydrology on ecosystem patterns, diversity, structure, and function coupled with ecological feedbacks on elements of the hydrological cycle and processes...(covering) both terrestrial and freshwater ecosystems and the management of our relationship with the environment,' (Porporato and Rodriguez –Iturbe, 2002).

availability. All such impacts can create localised situations that clearly breach the river continuum concept.

Burt and Pinay (2005) comment that the catchment is an appropriate scale for research due to its well-defined boundary. Hydrological connectivity is a major driver for numerous catchment processes at many spatio-temporal scales (Michaelides and Chappell, 2009). Not only does hydrological connectivity couple different compartments of the hydrological cycle, it also acts as a delivery pathway for diffuse pollution including agriculturally derived nutrients, sediments and pesticides. Many such process driven pollution transfers occur at the catchment scale and cascade through numerous scales until their impacts are felt in ecosystems downstream of the causal location (Burt and Pinay, 2005). Long-term data sets are highlighting that the catchment processes that drive these impacts are both non-linear and non-stationary (Tetzlaff *et al.*, 2008); thus, stream ecology could be better understood by the development of a science of multi-scale analysis (Palmer and Poff, 1997).

In recent years there has been a major shift in capacity for viewing locations remotely and manipulating data captured at distance which Lane *et al.* (2006) term 'surveillant science.' Aerial photos, for example, give a different view of a river (Haslam, 2008) that can reveal the riverscape as an integral component of the wider catchment. Such methods provide opportunities for viewing the river/landscape at numerous scales and so assist with identifying locations of risky land use **(**Pietroniro and Leconte, 2000). To complement such remote sensing capabilities, advances in GIS and modelling allow analysis of remotely sensed data in order to decipher land use patterns and pollution pathways at the catchment scale. Erosion management has long utilised advances in these methods (Paringit and Nadaoka, 2003) and new approaches to modelling landscape risk in terms of delivery pathways to river systems is providing qualitative and quantitative data on the relative risk of fine sediment delivery risk across whole catchments (Reaney *et al.*, 2010). If these methods can be shown to offer accurate assessments of multi-scale processes that impact riverscapes, then they can be provided for use by restoration ecologists to assist in their search for efficient targeting of resources.

Strong policy drivers exist for research into multi-scale impacts on water resources. In particular the EU Water Framework Directive (WFD, 2000) requires that all surface waters are brought up to "good ecological status" by 2015 across member states (Saz-Salazar *et al.*, 2009). However, the definition of 'good ecological status' offered by the WFD is weak and described (Moss, 2008) as being only slightly different from high ecological status which in turn is defined as having no, or minimal, human impact. It is hard to imagine such a river existing within the EU states, if anywhere. In addition to the loose definition of these categories, the WFD has been further criticised as being filled with political compromise (Moss, 2008) and spreading an 'ecological dream' in its aim for good ecological status (Bouleau, 2008). Yet it does offer an opportunity for improving river ecosystem health by providing a policy driver for ecological restoration coupled with dialogue between stakeholders which, under article 14 of the WFD, is a requirement of all member states. The WFD is also encouraging new tools and methods for measuring the ecological status of freshwater systems (Hatton-Ellis, 2008). This links well with advances in remote sensing, GIS and modelling and offers encouraging signs that these methods will be supported as tools for restoration ecologists and environment agencies if they are shown to offer good predictive ability.

Developments in policy and science, and in particular remote sensing, GIS and modelling, will provide a theme for this thesis. These will be developed to link spatially distributed brown trout fry populations to multiple pressures acting at numerous scales, from the catchment to the local habitat where brown trout fry exist. This will be carried out in order to explain variation in populations of the species at this life stage.

1.2 Summary of thesis structure

The structure of this thesis follows the objectives outlined above. The following paragraphs offer a brief overview of the thesis outline and direction. The research is contextualised in terms of present scientific awareness of riverscapes and research gaps in the field of ecohydrology. The concepts that drive the work are firmly embedded in hydrological connectivity and the role of process cascades through catchments that deliver matter to rivers and the impacts, positive and negative, on river ecosystems.

Chapter 1 has presented an overview of the research and introduced the research aim and objectives.

Chapter 2 will concentrate on objective 1 by providing a review of salmonid ecology focusing on brown trout fry requirements in terms of biotic and abiotic requirements to develop a set of hypothesis for later investigation. The literature review explores in-stream and adjacent habitat controls and catchment scale processes then moves on to multiple factors integrating the scales inherent within a catchment. The chapter will develop the context and conceptualise the research through an exploration of current awareness of brown trout fry ecology, the policy context and the numerous scales of a catchment. Here the discussion will concentrate on source areas of fine sediment and the recipient streams. The interactions between scales and land management in terms of diffuse pollution are also covered.

Chapter 3 introduces the case study to place the thesis into the context of the upper Ure catchment, Wensleydale, North Yorkshire. It will provide an overview of the physical conditions of the catchment, knowledge of brown trout populations, the institutional framework and a brief summary of the sub-catchments within the dale. This will be in the context of data capture and a demonstration of the factors that are important to brown trout fry. As part of this case study exploration hydrological data will be presented and conflicts between stakeholder groups will be explored. Finally, the chapter introduces the rivers trust movement concentrating on the Yorkshire Dales Rivers Trust which has been central in funding this research.

Chapter 4 introduces the methods in terms of the field data required to understand brown trout fry populations spatially distributed across an upland catchment. The methods will be developed in order to capture the pertinent factors that brown trout fry require along with factors that may limit populations including surrounding land use. The methodology continues by discussing the exploration of larger scale processes that are captured through GIS, remote sensing and modelling. This thesis is reliant on a large amount of spatial geo-referenced data, the manipulation of this data and application in terms of pressures on brown trout fry. Objective 2 will be the focus of the chapter which discusses the uses of remote sensing and GIS and their usefulness in catchment scale research. The chapter introduces the SCIMAP[4] module that aims to incorporate risky land use into the modelling process. There are two scales used during the SCIMAP application. The first explores fine sediment delivery at the farm-scale whilst the second looks at the full catchment. The chapter explores how land managers' knowledge can be incorporated into testing the model outputs at the farm scale. SCIMAP is then discussed in terms of catchment-scale application of the model.

Chapter 5 expands on chapter 4 by presenting the results of the research. This incorporates land managers' knowledge into the peer-review process as an attempt is made to test the SCIMAP model on 8 separate land holdings against land managers' knowledge. This takes the form of interviews and walk-over surveys with the farmers visiting locations that the model outputs suggest pose a risk of soil erosion and connection to the stream network. By incorporating land managers knowledge into model testing it meets objective 2 and 3. The chapter then discusses the catchment-scale SCIMAP results in terms of brown trout fry pressures looking at three different versions of the model which will be discussed later. This will provide a contrast to the more localised SCIMAP results; this will expand on objective 2. The field data which includes macroinvertebrate and, importantly, brown trout fry surveys is presented in this chapter. Land use factors that add detail to the SCIMAP modelling by exploring smaller scale processes that and may be impacting river ecology are also presented here.

Chapter 6 provides a discussion of the results in terms of catchment functioning and brown trout fry populations. The chapter discusses the results and explores how they map onto brown trout fry populations and at which scale the important factors operate. This is done in terms of management options for enhancing brown trout fry stocks coupled with an exploration of how the results of this research can be incorporated into catchment scale restoration to inform the process of achieving 'good ecological status' as defined in the WFD. This chapter concludes the thesis and summarises the findings by discussing the results in terms of the original aims and objectives set out in chapter one. It will consider the implications in terms of fisheries management to improve recruitment in upland streams and suggest the concepts and hypothesis that require

[4] SCIMAP (www.scimap.org.uk) stands for Sensitive Catchment Integrated Modelling and Analysis Platform. It is a hydrologically based model that can provide information on fine sediment mobilisation and delivery to streams at the catchment scale and at resolutions down to 5m

further research. The final analysis is a discussion on the implications of the findings. This chapter meets objective 4 and the overall aim of the thesis.

1.3 Research context

The research presented here is important for a number of reasons that range from advancing ecohydrological research in terms of how brown trout fry react to changes in water quality and hydrological connectivity to policy imperatives at a number of scales up to and including transnational obligations. Being able to map the spatial distribution of brown trout fry provides a rapid survey of the ecological condition of an upland river system across a range of scales and allows recruitment to be assessed. Utilising an early life stage of this species as a bioindicator offers a descriptor of habitat quality that will be discussed in the context of in-stream, riparian and catchment-scale controls on population. Such an approach offers the possibility of avoiding complications that arise from well known annual fluctuations in salmonid spawning by investigating relative[5] recruitment spatially distributed across a catchment over two breeding seasons. Rapid survey methods condense the time required to build a picture of brown trout populations that otherwise require long-term studies stretching to over ten years (Elliot, 1994). This approach allows a picture of relative populations to be identified allowing an appreciation of the locations where limiting factors may be most keenly felt. From this, habitat, land use and catchment factors can be tested against brown trout fry populations. By taking this approach, rapid knowledge can be gathered that allows restoration measures and locations to be prioritised in order of: 1) relevance, 2) appropriate scale and 3) effectiveness of restoration method. Moreover, by including spawning and juvenile brown trout habitat into research there is potential to improve wild brown trout populations through subsequent restorative measures (Summers *et al*, 2008).

Cresser *et al* (2000) argue that investigations into large-scale processes within ecology have great potential for regional-scale environmental management. Utilising different scales of SCIMAP fine-sediment modelling, in order to understand a catchment in terms

[5] Whilst year on year recruitment has much variability comparing river reaches that are subject to similar climatic controls over one season offers the opportunity to develop insights into the relative recruitment success across a catchment.

of process cascades and impacts on river networks, offers a two-way process whereby the model provides information on in-stream ecology which in turn provides a method of validating the model. Importantly, here, the model is additionally run at the farm-scale to see how it performs when exploring how hydrological connectivity combines with soil erosion risk to provide a fine-sediment delivery risk index. This gives a second and novel approach to validating a model in that the farm-scale outputs can be explored alongside farmer's knowledge of their holding to decipher accuracy at that scale. This enables the farming community to proffer information to the scientific community in a manner that has been uncommon as a validation method to date. Using this second validation approach offered the potential to 1) explore a number of farms in fine detail, 2) compare the model outputs at locations not ordinarily accessible, 3) examine the outputs alongside the farmers, and finally, 4) decipher the outputs in terms of farmer's knowledge and fine-scale nuances of the local hydrology. These two approaches to employing the SCIMAP model allows links between scales to be made in order to examine a catchment and provide interesting methods for model validation in a data poor catchment. SCIMAP thus enables a new and novel approach to exploring catchments.

Linking GIS, remote sensing and modelling technologies with more traditional data collection methods allows advances in computer processing power to be utilised in order to decipher a brown trout fishery in terms of recruitment. These methodologies provide opportunities to incorporate advances in remote survey capabilities into ecohydrological research to explore the appropriate scales placing controls on brown trout fry populations. This allows an exploration of catchment controls on localised biotic components of river ecosystems that accounts for 1) process cascades and 2) multiple factors that exist in all natural systems. Lijklema (1998, p.1) argues that, '*phenomena in the environment occur on a wide range of spatial and temporal scales. This puts demands on the ways we perform research and model systems*.' Thus, research should be pursued on all appropriate scales (Lijklema, 1998). The research presented here will offer a rapid approach to catchment exploration incorporating different scales in combination with multiple factors that can be picked up by other river trusts, conservation bodies and government agencies in order to identify the most appropriate restoration sites and methods.

By incorporating a mix of social and scientific methodologies into research, the links between human intervention and the natural world can be explored. From this, an understanding of the different forms of land management in terms of the social, practical and economic factors which govern them can be built. Moreover, land managers are best placed to describe the landscape they farm. Ormerod and Watkinson (2000) believe that stronger links between disciplines is required to improve large-scale research into ecological processes. This developing theme in ecohydrology is followed here through farmer interviews, employing different scales and forms of modelling techniques and remote sensing in combination with field work that explores land use along with the biotic and abiotic components of ecosystems. This form of research is better able to account for the large variation in ecological controlling factors, cascades, scales and land management styles.

River restoration is set to become a dominant feature of the conservation movement over the coming years due to policy drivers including the EU Water Framework Directive 2000 (WFD) and others such as the Habitats Directive and Biodiversity Action Plans (BAP). These policies require the ecological status of water courses to be brought up to at least "good ecological status" (GES) as well as identifying species and habitats that require specific action such as bullhead (Habitats Directive) and gravel bed rivers (BAP). Of all these drivers, the WFD is the most important in terms of reach, scale and the demands it places on EU member states. This policy provides an '*overarching piece of legislation that aims to harmonise existing European water policy and to improve water quality in all of Europe's aquatic environments'* (Kaika and Page, 2003, p.1). The WFD aims to supply a clear legal framework that manages catchment systems as a single entity in recognition of the connections between landscape processes and river networks (Holzwarth, 2002). This policy driver requires rapid exploration methods if the targets to bring river systems up to GES by 2015 (or 2027) are to be met. Thus, this work offers an approach that will allow knowledge of catchments to be gleaned rapidly allowing restoration plans to be drawn up and implemented that may push forward the move towards GES.

In order to bring river systems up to GES, it will be important to understand impacts, the scale they operate at and where there are underlying pressures that require restorative methods to be employed. By exploring a catchment in order to provide a rapid assessment of the controls, pressures and the multiple nature of the impacts and cascades on localised ecological components of river systems, this research offers potential to be employed in other upland catchments that are presently failing WFD criteria.

Chapter 2: Catchment processes and river ecology

The remainder of this chapter explores a catchment in terms of scale and process. It develops these as concepts that will be implicit throughout this body of work. The chapter will relate process and scale to salmonid species, in particular brown trout at the fry stage of the life cycle. This focus on the fry stage will be explained and justified in terms of how the organism responds to pressures. The literature review describes the many limiting pressures on the species and finally identifies hypotheses regarding the limiting factors on brown trout and the scale at which they operate. This is in line with Objective 1. These hypotheses will be tested in subsequent chapters, in particular chapters 5 and 6, in order to meet Objective 4.

2.1 Introduction.

In order to understand the ecology of river systems, it is important to develop knowledge about the processes that control habitat and ecosystem types (Tetzlaff *et al.*, 2007). Minshall *et al.* (1985) note that there has been a shift from descriptive autoecological[6] studies in the 1950s towards research that is synecological[7] and increasingly holistic. This trend has since continued and it is becoming increasingly recognised that stream and river research requires catchment scale perspectives (Wissmar and Beschta, 1998; Burt and Pinay 2005; Tetzlaff *et al.*, 2007). This moves the scale of enquiry beyond the channel reach towards the whole catchment. Such a shift in the scale of enquiry poses research difficulties that require novel approaches in order to make links between scales that are so intertwined they become difficult to disentangle. Moreover, natural processes combine with anthropogenic impacts including rural land use, urbanisation and habitat fragmentation which, when connected to river ecosystems, may impact upon habitat integrity.

Holmes and Hanbury (1995) argue that the potential of rivers to support wildlife has been severely depleted. Collares-Pereira and Cowx (2004) argues that most rivers have

[6] Autoecological studies explore the relationship between one species and its environment (Lawrence *et al.*,1988).
[7] Synecological studies explore the ecology of plant or animal communities (Lawrence *et al.*,1988).

been severely and negatively affected over the last one hundred years of human development. In order to assess this degradation, traditional reductionist approaches have focused upon tightly-defined scales, individual species and individual factors (Lane, 2008). However, a broadening of the scale of investigation is required if rivers are to be understood in the context of their natural settings and upstream processes. Understanding ecosystem function in such a context, whether aquatic or terrestrial, is an unsolved challenge (Reynolds, 1998). Fluxes of species, water, nutrients and weather systems all link communities and ecosystems with the wider landscape (Parker and Pickett, 1997) making identification of cause and effect complex. It is these wider connections that place direct and indirect controls on river ecosystems.

A catchment perspective becomes increasingly important in fragmented landscapes where ecosystems are likely to have been disrupted by human activity (agriculture, urban developments, quarrying, etc; Gosset *et al.*, 2006). This creates difficulties for dispersal both in and out of a given habitat (White and Walker, 1997). Thus, there is a need to restore connections between sites to ensure viability of distinct ecosystems through enhanced migration (Noss and Harris, 1986) and by ensuring systems connect with the process locations that govern ecosystem functions. It is such connectivity that ensures processes connect and allows organisms to respond to degradation in any one site through dispersal to less degraded sites.

2.2 The nature of salmonids

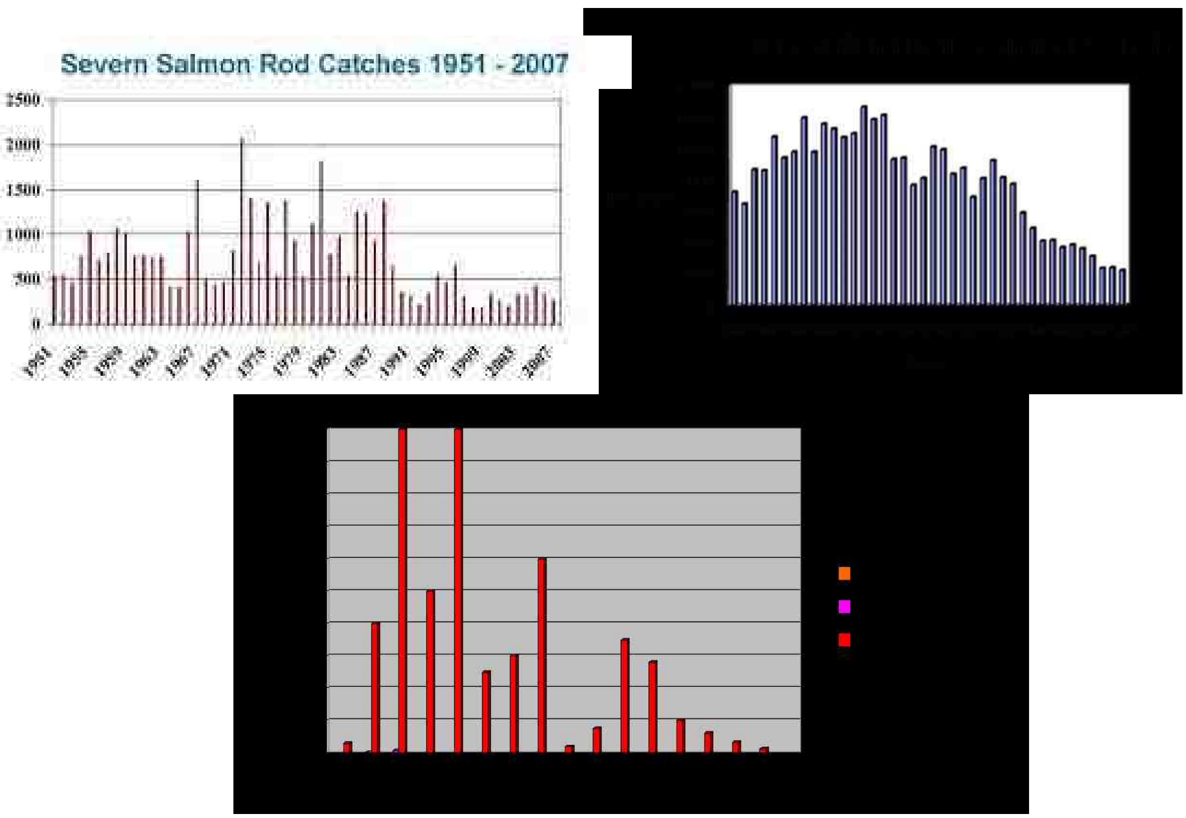
Catchment-scale studies are best able to account for the multi-spatial aspects of ecosystems and species throughout their life cycles. Hendry *et al.* (2003) state that there are three integral components of river systems to study when attempting to decipher salmonid habitat management: water quality; water quantity; and the physical structure of the riverine environment. Armstrong *et al.* (2003, P. 165) state that, '*there is a clear need for more advanced models of the relationships between habitat variables and fish production… (at the scale of)… catchment and sub-catchment.*'

On a global scale, salmonid species have been in decline for a number of years (Figure 2.1 a, b, c). There are numerous hypotheses put forward to explain this decline, none of which are mutually exclusive. This decline covers all salmonid species including the

British native species Atlantic salmon (*Salmo salar*), arctic charr (*Salvelinus alpinus*), grayling (*Thymallus thymallus*) and brown trout *(Salmo trutta*). Atlantic salmon and brown trout are species of high economic importance and degraded populations can impact local and regional economies. Both these species spawn in gravel-bed rivers with high gradients that provide the requirements for spawning success: these requirements include gravel-beds with well-oxygenated, oligotrophic waters (Mills, 1971; Frost and Brown, 1973; Mills 1991; Elliot, 1994; RSPB, 1994; Armstrong *et al.*, 1998; Klemetson *et al.*, 2003).

Figure 2.1: *Decline in the global nominal Atlantic salmon catch since 1960.*

*(**Source:** Salmonid 21C, www.salmonid21C.org).*



2.3 Brown trout (*Salmo trutta*)

As with most salmonid species brown trout have commercial value with a UK industry turnover of approximately £150 million per annum (British Trout Association, 2006) giving their conservation a high monetary value. In recognition of the importance of brown trout, and salmonids in general, there have been numerous international annual workshops dedicated to conservation and restoration of salmonid habitats (Duff, 2002).

Moreover, the species has been researched for many decades and there is a good amount of detail on their habitat requirements at a number of life stages and spatial scales. The following two sections will cover the biotic and abiotic controls on brown trout populations.

Armstrong *et al.* (2003) devised two diagrams that highlight the abiotic and biotic controls on salmon parr, these factors are very similar to controls on brown trout fry, except that fry have limited dispersal and so the response to pressure is reduced survival, (figures 2.2 and 2.3).

Figure 2.2: *Biotic controls on salmon parr. Source: Armstrong et al. (2003)*

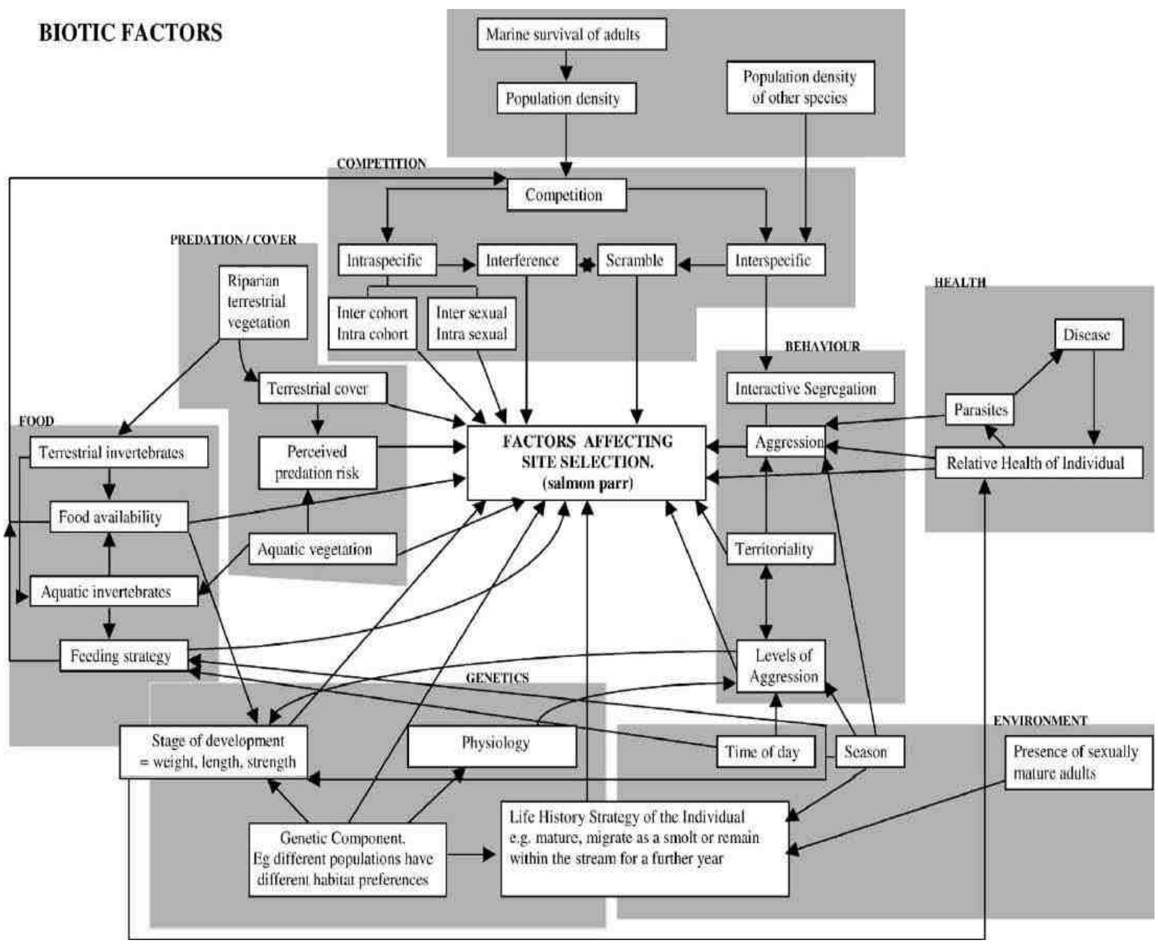
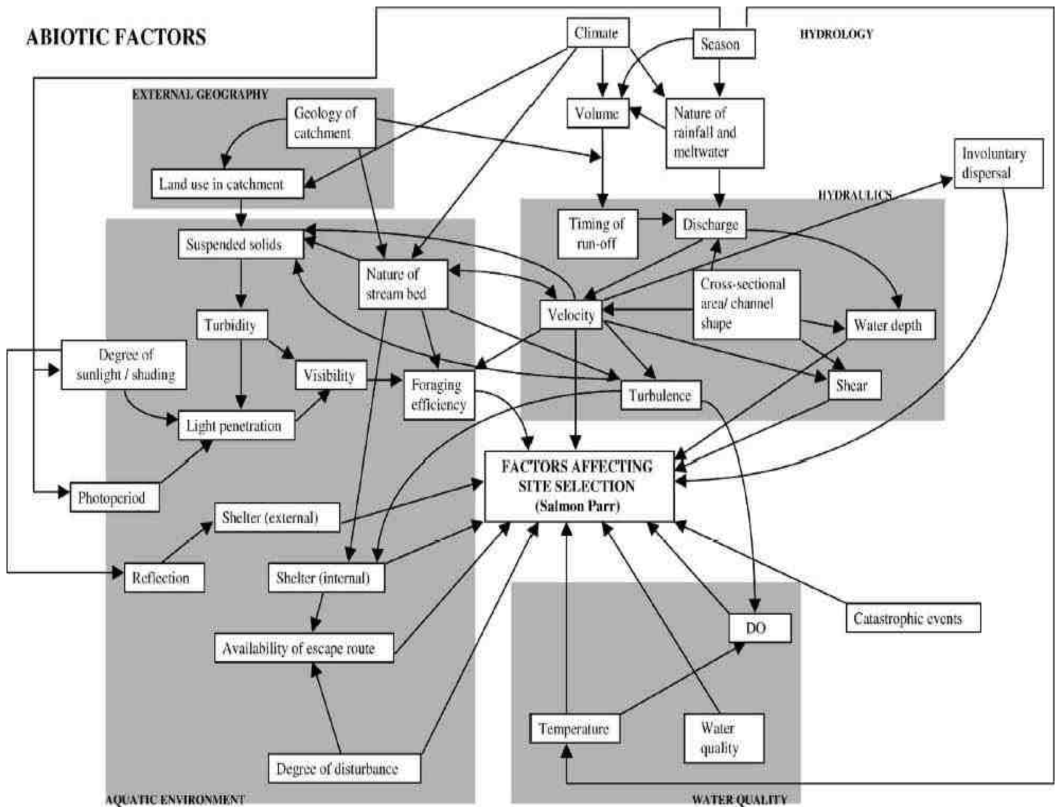


Figure 2.3: *Abiotic controls on salmon parr. Source: Armstrong et al. (2003)*



These figures (2.2 and 2.3) describe controls site selection for atlantic salmon parr but they also apply to brown trout fry with the caveat that there is limited dispersal ability and so the controls on site selection can be more profound. The biotic and abiotic controls are discussed below.

2.3.1 Biotic factors

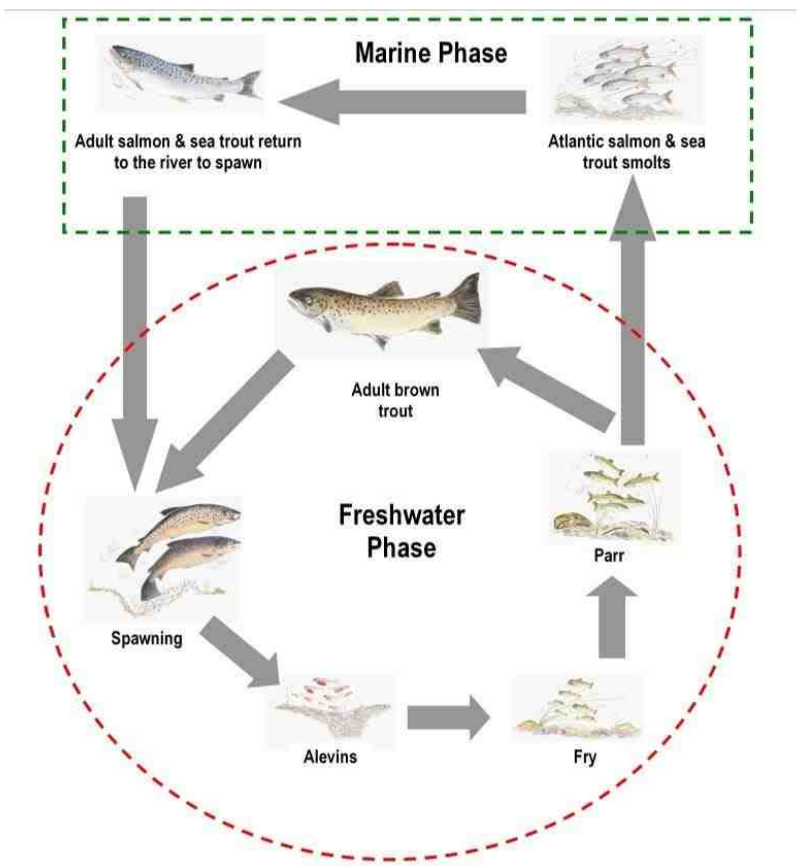
Brown trout (*Salmo trutta*) are a characteristic teleost[8] fish species of upland rivers and streams (RSPB *et al.*, 1994). The species has at least five possible life cycles including populations that remain in upland streams throughout their full life cycle to anadromous varieties that migrate to estuaries and coastal zones (slob trout and sea trout respectively) before returning to their natal streams to spawn (Figure 2.4; Table 2.1).

[8] A fish of the sub-class teleostei they are defined as having a consolidated internal skeleton, swim bladders and thin cycloid scales.

Table 2.1: *The different life strategies of brown trout.*

Life Strategy	Geographic Range
Resident	Short range, remains in natal streams
Migrate between natal stream and river stem	Short to medium range, generally travel short distances from natal streams to the main river, returning to spawn
Ferox[9] trout	Medium range, with migrations from natal streams to lakes
Slob and sea trout	Long range, migrate from natal streams as smolts to estuaries (slob trout) or to sea (sea trout)

Figure 2.4: *The salmonid life cycle (artwork courtesy of B. Berwick)*



[9] Ferox trout describes large predatory brown trout inhabiting deep lakes. It was thought to be a separate species (*Salmo ferox*) but is now known to be a form of brown trout showing the plasticity of the species.

Brown trout have a wide-ranging and varied life cycle with the ability to adapt to numerous in-stream habitats at different life stages (Klemetson *et al.*, 2003). The life cycle begins at the egg stage. Eggs are deposited in well-oxygenated gravel beds of small upland streams during the period October to December. The eggs require oxygen replenishment and adequate flows to remove wastes (Klemetson *et al.*, 2003). After overwintering as eggs, the fish emerge as alevins[10], typically in February. Whilst in the alevin stage, the fish feed on egg yolk that is attached to their bellies.

The alevins remain in the gravel bed until the egg yolk is close to depletion or fully consumed. This takes approximately six weeks (Frost and Brown, 1973). At this stage they emerge from the gravel bed as fry and set up territories close to the spawning area. There is evidence of staggered emergence with short upward migrations into the river column before returning to the gravel interstices (Williams *et al.*, 1981; Hemming *et al.*, 1982). This occurs only while the yolk provides nutrition. On full emergence as fry, it is essential that a territory is established (Ayllón *et al*, 2009; Elliot, 1994). Fry from larger eggs, and those that emerge early from gravel beds, are at a competitive advantage (Vøllestad and Lillehammer, 2000). Any fry that fail to establish territories are forced into downstream migration and those that fail to establish territories downstream generally die within a short period (Elliot, 1986; Elliot, 1994). Poorer habitats result in higher mortality rates (Heggenes *et al.*, 1999) and fry forced into downstream migration are always smaller than those that set up territories (Skoglund and Barlaup, 2006).

Brown trout begin exogenous feeding close to depletion of the egg yolk and always when they transfer from alevins to fry. The aim of 0+[11] (young of year) brown trout is to maximise energy intake, posing a trade-off between finding food and vulnerability to predation (Ayllón *et al.*, 2009). The optimal habitat at this stage provides boulders close to the spawning gravel bed, which act as refugia, with a depth range of 20 to 35 cm and a velocity range of 0.5 m s^{-1} to 0.8 m s^{-1} (Fausch and White, 1981; Hughes and Dill, 1990). The fry stage is the least plastic stage of the life cycle and fry are generally confined to narrow niches in riffle environments of upland streams (Ayllón *et al.*, 2009).

[10] This is the post egg stage when the yolk sac is still attached to the fish and it remains within the gravel interstices. On using up the yolk sac it emerges from the gravel bed to begin exogenous feeding. At this point the fish has begun the fry stage of the life cycle.
[11] Fish age is noted in terms of 0+, 1+, 2+ etc, at 0+ the fish is in its first year of life, at 1+ the second, 2+ the third etc.

It is this early life history that has a disproportionate effect on lifetime reproductive success (Vøllestad and Lillehammer, 2000).

During their first winter, trout migrate from the spawning gravel beds into pool habitats (Elliot, 1986). At approximate age 2+, brown trout start the next part of their life cycle where a more plastic approach to habitat selection is available to them. After approximately two years there are several possible life strategies most of which are habitat and water quality driven (Bridcut and Giller, 1993). Some fish will remain in their natal streams for the full duration of their life cycle; others will migrate from the natal streams and take up residency in the main river stem (see table 2.1).

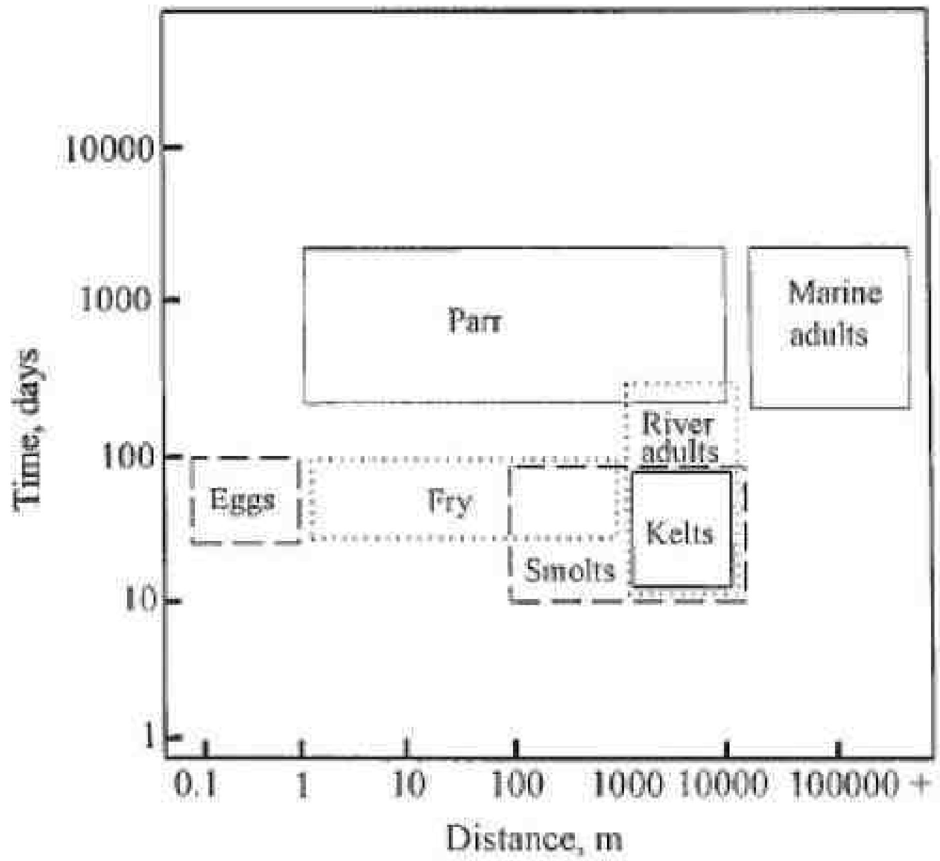
Now the life cycle becomes complicated with longer range migratory options available. If lakes are present in the catchment (and are accessible to the fish), then some will migrate to these and may grow rapidly turning to piscivory as the main food source. Brown trout that inhabit lakes grow to substantial sizes, reaching weights of >5kg. These are generally known as ferox trout. Others undergo smoltification[12] and become sleeker, silvery and develop the ability to cope with saline conditions. These migrate to estuaries (slob trout) or to coastal regions (sea trout; Bridcut and Giller, 1993). Smolt migration is more usual in shorter river systems but can be triggered in larger systems or even those with barriers to upstream migration preventing the fish from returning to their natal streams. Loss of fish in a system due to outward migration can be a limiting factor on populations where upstream migration is hindered due to barriers including weirs and waterfalls.

There is a definite sexual dimorphism with a higher proportion of females undergoing smoltification (Klemetsen *et al.*, 2003). Once at the coastal zone, smolts undergo rapid growth due to the greater abundance of prey and this possibly explains the greater proportion of females opting for this migratory life strategy. Egg production and egg size both increase with body length and mass and female fitness increases linearly with body mass (Wooton, 1988). The life cycle of atlantic salmon and brown trout is depicted in figure 2.4 whilst the dispersal ability of brown trout at different life stages can be seen

[12] Smoltification involves morphological, physiological and behavioural changes that allow the fish to migrate to coastal zones having developed the biological functions that allow the transition between fresh and salt water.

in figure 2.5. As can be seen, fry have the most limited migratory abilities and this stage is confined to small streams close to the spawning beds (Bridcut and Giller, 1993).

Figure 2.5: *Stages of the life-cycle of Atlantic salmon in relation to scales of space and time. These patterns hold for brown trout. (Armstrong et al., 1998)*



The final stage of the life cycle involves migration back to the natal stream for spawning. The length of the migration is dependent on the life strategy adopted. Brown trout have been shown to have a strong homing device that compels them to return to the stream in which they were spawned. Stuart (1957) carried out experiments that involved the removal of spawning brown trout from their selected tributaries to streams on the opposite side of the catchment. Out of 3000 fish only one failed to return to its original natal stream to spawn. Once at the gravel beds, the males compete to mate, the more dominant animals fertilise the most eggs. Females use their tails to thrash out scrapes in the gravel where they deposit their eggs. Dominant males then spread their milt over the eggs before the female fills the scrape in the same manner it was dug. It is rare for all the eggs to be deposited in one scrape. A series of these scrapes on the same gravel bed is termed a redd. Figure 2.4 summarises the life cycle stages of salmonid fish.

This is a simplified version of the life cycle. There are numerous differences within and between catchments that alter the timing of life stages and growth rates. Habitat quality can vary substantially between tributaries of the same river system. A simple snap-shot of a food web is unable to account for these temporal and spatial differences, seasonality or life stage.

Brown trout diet is broad and consists of macroinvertebrates including plecoptera, ephemoptera, trichoptera, diptera, gammarus, crustaceans, coleoptera, arachnids, molluscs and fish (Frost and Brown, 1967; Klemetsen *et al.*, 2003). Terrestrial invertebrates comprise 10 to 41% of diet, with a higher proportion taken during the months of June and July. Aquatic invertebrates comprise 57 to 90% of diet (Greenberg and Dahl, 1998). Larger fish, especially migratory sea trout and those inhabiting lakes and lochs, will turn to piscivory. Rosenzweig (1995) believes that behavioural differentiation between sea trout and resident brown trout may be competitive speciation, a form of sympatric speciation[13], arising from direct competition for resources. The ability to disperse, a behaviour now becoming more common within brown trout, is undoubtedly a favourable ecological coping mechanism. As with brown trout, their prey taxa are subject to similar habitat controls working at process scales above the river reach. It is becoming increasingly recognised that it is larger-scale processes that create the template in which the smaller scale functions (Armstrong *et al.*, 1998; Stauffer *et al.*, 2000).

Brown trout face inter-specific competition[14] with species such as atlantic salmon, grayling and bullhead (*Cottus gobio*). However, it is intra-specific competition[15] that is more important in limiting population size. Trout of all ages will aggressively protect territories against con-specifics in order to protect available resources (Klemetsen *et al.*, 2003). Figure 1.4 describes the biotic controls on feeding site selection for atlantic

[13] Sympatric speciation is the development of new taxa from ancestral taxon within the same geographical region (Allaby, 1994). Presently sea trout and resident brown trout interbreed, for this form of speciation to occur interbreeding between the two forms has to cease. Often the reason for cessation of interbreeding in this form of speciation is poorly understood.
[14] Inter-specific competition is competition between species operating at the same trophic level.
[15] Intra-specific competition is the same form of competition but between con-specifics.

salmon parr, these controls carry for brown trout fry. but there is limited dispersal ability and so the controls on site selection can be more profound.

2.3.2 Abiotic factors

Downstream migration in brown trout has been shown to be triggered by several factors including water temperature (Hegennes and Traaen, 1988), water flow (Ottoway and Clarke, 1981), rates of change of water flow (Crisp and Hurley, 1991), developmental stage (Ottoway and Clarke, 1981), river system (Klemetsen *et al.*, 2003), number of degree days (Klemetsen *et al.*, 2003), mean annual fish length (Klemetsen *et al.*, 2003), and population density (Crisp, 1991). Migration occurs at a variety of spatio-temporal scales and the greater the distance migrated, the higher the energy cost on the fish.

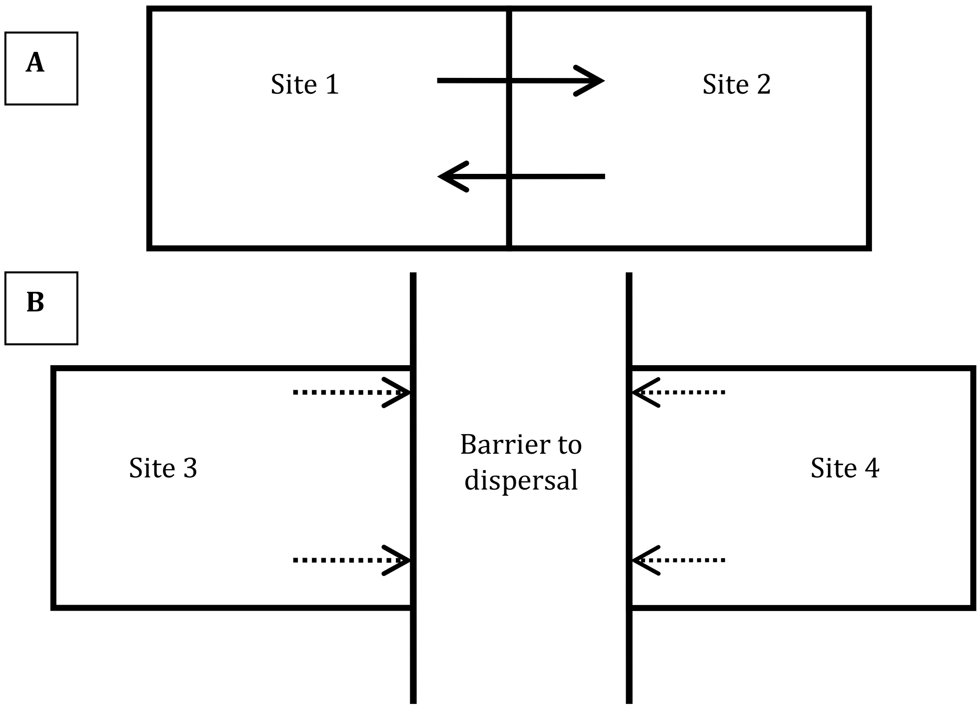
In upland river systems subject to barriers, either natural (waterfalls, sink holes) or anthropogenic (dams, weirs and culverts), brown trout exist as resident, non-migratory populations. There is anecdotal, and research, evidence to suggest that these populations have been severely suppressed due to land use impacts (Campbell, 1987; Theurer *et al.*, 1998; Luckenbach *et al.*, 2001; Gosset *et al.*, 2006). These impacts include changes in hydrological connectivity[16], diffuse pollution, reduced or increased water flows, fine-sediment delivery and accumulation, habitat fragmentation and temperature changes. Brown trout have specific habitat requirements at different life stages. The fry require gravel beds with low fine-sediment inputs (Theurer *et al.*, 1998). This provides habitat and refugia for the fish but also supports the macroinvertebrates on which they depend. At the parr stage, the presence of pools and boulders enhances survivorship by offering refugia whilst older fish (>2+) start migratory behavior often triggered by the condition of the habitat.

During migration barriers, may act to limit migratory behavior. For example, low flows can act as a barrier if sections of river become dry or when water is held back by weirs preventing fish from continuing downstream, or more commonly from returning to natal streams during the spawning season (figure 2.6). At a more local scale, fine-sediment

[16] Hydrological connectivity is taken to mean vertical, horizontal, longitudinal and temporal connectivity. In the context of this section it is longitudinal and temporal hydrological connectivity that are important.

delivery can act to hinder egg to fry survival (Theurer *et al.*, 1998) or reduce habitat availability for prey such as plecoptera or ephemeroptera.

Fig. 2.6: *Showing how a fragmented landscape can reduce a population's ability to disperse. With good connections dispersal is a two way process between habitats (A) whereas with fragmented landscapes (B) dispersal is hindered or not possible. In terms of rivers waterfalls, manmade structures including weirs and pollution hotspots can all fragment ecosystems and become barriers to dispersal.*



The physical structure of the habitat can enhance population size through the addition of refugia or greater habitat for a wide range of prey species (Armstrong *et al.*, 2003). Structure can come in the form of large woody debris (Lester and Wright, 2009), boulders within a gravel matrix, a hyporheic zone free from fine sediments or riffle – pool zones. Richter *et al.* (1997) identified water flows as the major limiting factor within a stream ecosystem. Too low or too high and the habitat is disrupted through physical displacement of structural features or reduction in the wetted channel. Low flows coupled with high summer temperatures can reduce oxygen levels within the water column. If this is combined with little or no bankside vegetation to offer shade, then the situation is further exacerbated. This highlights how biotic and abiotic aspects of an ecosystem interact to create conditions within the range species need or those that breach such thresholds.

2.3.3 Human impacts on salmonids

The more pernicious controls on population are not driven by competition but habitat quality, especially habitat patches that have been degraded by anthropogenic impacts including agricultural land use, waste water treatment works, industrial pollution including temperature changes and water stress (Klemetsen *et al.*, 2003; Ormerod, 2003; Gosset *et al.*, 2006). Negative anthropogenic impacts on water quality and ecosystem integrity can be witnessed through the response of species like brown trout that have specific habitat requirements. This is even more apparent during the early, poor-dispersing stages of the life cycle that demand good water quality in order to survive. Anthropogenic threats to water quality include;

- Eutrophication and high biological oxygen demand (BOD) via inputs of nutrients;
- Reduced pH via atmospheric acidic deposition, acid flushes and acid mine drainage;
- Thermal pollution;
- Mismanagement of riparian habitats including siltation of gravel beds;
- Pesticide and industrial toxins; (see Armstrong *et al.*, 1998; Armstrong *et al.*, 2003; Klemetsen *et al.*, 2003)

In recognition of the economic importance of brown trout, there exists a wealth of information on the limiting factors that humans impose upon the species. Luckenbach *et al.* (2001*)* found that the embryos and larvae have a high sensitivity to toxins. Negative correlations exist between temperature and stream width for older individuals, whilst the main limiting factor on fry is reduced dissolved oxygen (DO) content of the water (Ecklöv *et al.*, 1999). Both of these impacts can arise from altered hydrology due to, for example, upland drainage or soil compaction. Reduced DO can occur through nutrient inputs leading to eutrophication, again this can be exacerbated when flows are low. Dinham (1993) argues that severe degradation of water quality is due to high levels of fertilisers and pesticides in areas with highly intensive agriculture. These sources of pollution are pernicious and chronic resulting in eutrophication, high BOD, depleted invertebrate populations, bioaccumulation and magnification.

Ojanguren and Brana (2003) exposed brown trout embryos and fry to a range of temperatures ranging from 4º to 18º C and found that survival was maximal at between 8º and 10º C. If increased temperatures are coupled with effluents from nearby industries and sewage works, then the oxygen carrying capacity is further reduced under eutrophication and high BOD. Moreover, Ormerod and Durance (2009) found that a one degree increase in river temperature can reduce macro-invertebrate populations by 20%, significantly reducing a major food source for brown trout.

Changes in water flows from land drainage and fine-sediment accumulation in gravel spawning beds (Ojanguren and Brana, 2003) negatively impact on brown trout survival rates. Water quality and habitat needs vary throughout the life stages of the species. The fry stage (0+ fish) have the most specific water quality and habitat needs and are thus vulnerable to many of the pressures identified above (Ayllón *et al.*, 2009). This, coupled with the poor dispersal ability of the fry stage, make it a good indicator species for ecosystem health.

Brown trout are negatively affected by acidified waters. Signs of acidification include tail deformities (Campbell, 1987) and ultimately local extinctions (Maitland *et al.*, 2000). In granite rock areas, acid flushes occur after heavy snowmelt or rainfall. These flushes can be exacerbated by land drainage including the cutting of open drains (grips) in the upper reaches of UK catchments. Ormerod *et al.* (1989) carried out a study in 1987 at the headwaters of a stream in Wales. They gradually reduced pH from 7 down to between 4.2 and 4.5 and then increased the aluminium content of the water from 0.005g m^{-3} up to between 0.3 and 0.4 g m^{-3}. Brown trout responded with a 7 to 10% population decline in the acid zone and a 50 to 87% decline in the Aluminium zone. Jutila *et al.* (2001) found negative relationships between brown trout and the proportion of upstream peat soils and further correlations between brown trout numbers and upstream area, pool abundance and pH. All these effects reduce the ecosystem integrity and therefore carrying capacity of water courses. They directly and indirectly reduce brown trout numbers.

Brown trout move to spawning streams during the period October to November in response to either, or both, spate conditions (Munro and Balmain, 1957) and specific temperature ranges between 6 to 7º C (Stuart 1953). If flow rates are disrupted or temperatures remain significantly above or below this range (perhaps in response to climate change and altered flow rates from grips), then spawning is likely to become disrupted. If conditions derived from land management regimes degrade upland low-order streams to the extent that spawning becomes non-viable, then after several years there may be little or no spawning stock associated with specific tributaries. With a 1 in 3000 chance of a stray fish returning to the 'wrong' stream (Stuart, 1957), it is unlikely that natural restocking of tributaries will occur if populations drop below minimum viable population[17] levels. If altered hydrology, high sediment loads and changed water chemistry result in reduced availability of the spawning beds and streams, brown trout populations may be depleted for the foreseeable future. Grips, and other anthropogenic impacts, have to be studied within this multivariate reality of catchments that may contain synergistic relationships between variables.

2.3.4 Indicator species and brown trout

An indicator species provides a proxy for measuring ecosystem health through its response to pressures placed on their habitat. As a wide-ranging, migratory fish with several available life cycles involving adaptation to a number of habitats, brown trout are subject to numerous pressures at different stages of their life cycle (Ayllón *et al.*, 2009). The ability to migrate away from unsuitable locations is a feature of brown trout fish age >1+. At earlier stages of their life cycle, all salmonids are less able to disperse and so remain either within the gravel interstices (alevin stage) or close to natal spawning beds (fry stage; Armstrong *et al.*, 1998).

In terms of brown trout ecology, there are different issues of scale dependence at different life stages. For example, returning adult fish require high flows and strong longitudinal connectivity to reach the natal spawning streams whereas, at the fry stage, the organism is dependent on small riffle zones in upland stream systems (Armstrong *et al.*, 1998). The location of fry riffle habitat generally has a small upstream area with land use dominated by livestock and forestry as opposed to arable systems. The poor dispersal ability of fry, coupled with high mortality in sub-optimal conditions, allows detail on river condition to be assessed against fry survivorship. This has positive

[17] The minimum viable population is the smallest population size that can interbreed to maintain a population over time.

implications for river restoration as electro-fishing surveys can quickly reveal the relative condition of a river system across a whole catchment or sub-catchment. The scale required to enhance brown trout fry habitat is very different from the scale required in order to improve upstream migration of spawning fish. Moreover, the issues are very different. In order to enhance upstream migration of returning fish removal of weirs, or construction of fish passes, may be the overriding issue, whilst local and upstream land restoration is more likely the issue to ensure habitat quality for fry population viability.

Kondolf (2000) argues that it is important to identify life stages that place limiting controls on populations. If a life stage responds to a number of variables that are being impacted by anthropogenic changes, then it will provide a sound biological indicator for identifying limiting pressures. Because of the variety of scales and processes to which brown trout respond, they are an interesting species to study in order to elucidate catchment processes that impinge on river ecosystems. Moreover, in upland rivers the fry stage, with its high demands for habitat condition and water quality coupled with their poor dispersal capabilities, appears to be an ideal indicator when assessing land use and hydrological connectivity that combine to degrade water quality (Heggennes *et al.*, 1999). From this it is hoped that limiting natural and anthropogenic factors can be teased out to ascertain where human impacts are placing strong controls on populations.

Home ranges vary over a number of scales between life cycles (Figures 1.2 and 1.3). There is a high mortality rate during the early stages of the brown trout life cycle (Skoglund and Barlaup, 2006). This means that, if the habitat is not suitable, and there are no vacant territories available downstream, then the fish will not survive. It is this reduced survivorship, when the habitat conditions are impacted upon, that make them an important indicator species.

Therefore, to develop understanding on the limiting factors present in upland streams, it is necessary to explore pressures on a stage of the life cycle that: 1) occurs in upland streams; and 2) has low dispersal ability. With these specific requirements there are three possible stages to choose from being: 1) the egg stage; 2) the alevin stage; or 3) the fry stage. The fry stage is the simplest to sample using methods including torch-lighting and electrofishing which can provide rapid assessments of population.

2.4 Catchment function

Efficient targeting of ecological restoration requires good understanding of riverscapes in the context of how they sit within the wider catchment. Developing this knowledge necessitates the blurring of disciplinary boundaries (Lane *et al.*, 2006) in order to relate ecology to hydrological processes across spatio-temporal scales (Hannah *et al.*, 2004). Ecological restoration often fails by not accounting for these interacting components that include the varying scales of catchments, landscape processes, hydrology, land use and ecological communities (Kondolf, 2000).

Under sufficient pressure, the mechanisms maintaining ecosystems within the range required by the ecological communities present are likely to be breached leaving the system vulnerable to change (Reynolds, 2002). Poff (1992) argues that disturbance will always have ecological effects. It is the predictability and magnitude of disturbance that govern the level to which change occurs (Poff, 1992). Disturbance to habitats that is within the range from which a system can rebound includes factors such as low and high flows that are seasonal or in response to typical rainfall events. These are less likely to have long-term impacts than disturbance that is beyond the assimilatory ability of in-stream biota such as fine-sediment delivery, pesticide pollutants and eutrophication (Poff, 1992). White and Pickett (1985, p.3) define disturbance as, '*any relatively discrete event in time that disrupts ecosystem community, or population structure, and changes resources, substrate availability, or the physical environment.*' In terms of stream ecosystems, Resh *et al.* (1988, p434) add to this definition by focusing on events that are, '*outside a predictable range as organisms are adapted to predictable seasonal fluctuations of discharge, temperature, dissolved oxygen etc.*'

2.4.1 Catchment scale

The upland regions of the UK have extensive blanket peat coverage of approximately 8%, much of which is impacted by severe erosion with high levels of gullying (Evans *et al.*, 2006). In England and Wales, peat is defined as a deposit of at least 30cm depth and containing >50% organic matter (Johnson and Dunham, 1963). These upland blanket peat systems are a source of quickflow with brisk water movement resulting in rapidly rising and receding limbs of hydrographs as a response to rainfall (Holden, 2009). In dry periods, which can be as short as one week, peatland streams can contain very little or no flow (Holden and Burt, 2003a) highlighting the rapid response of catchment hydrology in upland catchments holding large proportions of peatland in their upper reaches. In extended dry periods, peat can become hydrophobic reducing its ability to retain its initial moisture content (Egglesman *et al.*, 1993); this can then enhance surface flow adding to the quickflow response of peatland hydrological processes. In most peat soils the water table is within 40 cm of the surface for approximately 80% of the year (Holden and Burt 2003b) and so the soil has very little surplus storage capacity which further explains the quickflow response during rainfall events (Holden 2009). Saturated peat is 90 to 98% water by mass with little difference above the saturated zone which is generally 90 to 95% water by mass (Holden, 2009).

Hydrological change plays a key role in peatland dynamics (Yu *et al.*, 2001). Hydrology in undisturbed blanket peats is dominated by overland flow or by throughflow within the upper few centimetres of the soil (Holden 2009). There are also flow paths at depth within the peat via macropores, commonly known as soil pipes, which can form networks extending to >100 metres in some locations (Holden, 2004). At the Maesnant catchment in Wales, it was shown that these macropores could contribute up to 50% of stream flow and responsible for enhanced sediment transfers from land to stream (Jones 1997; Jones 2004; Jones and Crane 1984). However, Holden and Burt (2002c) and Holden *et al.* (2009) have shown that contribution to streams via macropores is generally less than this in deeper peat, more in the order of 10 to 20%.

The position of the water table in peat soils places strong controls on the difference between accumulation and decomposition of organic matter and consequently the stability of peat soils of upland regions (Holden *et al.*, 2004). Yu (2006) comments that water table depth places a strong control on the residence time of organic matter in the acrotelm[18] and so determines the rate of peat transfer from acrotelm to catotelm. The sustainability of peat soils is therefore dependent on hydrological processes and thus changes to peatland hydrology, often primed by land use change and intensification, can

[18] The acrotelm is the upper portion of a peat soil where organic matter decomposes aerobically and much more rapidly than the lower catotelm which is generally waterlogged and subject to slower anaerobic decomposition. Here the peat soil accumulates in an intact functioning system.

degrade the ecosystem and create subsequent downstream impacts (Holden *et al.*, 2004; Wallage *et al.*, 2006).

Many peatland systems have been severely degraded due to a number of land management practices (Wallage *et al.*, 2006). Impacts that have been subject to intensive research include the ploughing of open drains into upland peat soils (grips) and extensive burning for grouse moor management. Drainage can be extensive on peatlands and was carried out intensively during the 1970s and 1980s in order to improve land productivity by enhancing the ground conditions for sheep and grouse (Worrall *et al.*, 2007; Waddell, 2006; Holden *et al.*, 2004) through the lowering of the water table to reduce surface water and thus alter vegetation cover (Holden *et al.*, 2004). However, Stewart and Lance (1983) could find little or no evidence to indicate that these aims were achieved through the cutting of grips.

Whilst grips have been cut into peatlands for centuries (Holden, 2009), the development of the Cuthbertson drainage plough, coupled with post-war agricultural policies that drove intensification of productivity, resulted in open drains being increasingly cut in peat soils (Robinson, 2006). On first cutting, the cross profile of a grip is trapezoidal; they are typically 50cm deep, 90 cm in width at the top and 40cm at the base (Worrall *et al.*, 2007). However, over time many become severely eroded whilst others fill through natural processes. At some locations grips have resulted in changes in hydrological flow paths across peatland systems (Holden *et al.*, 2006) reputedly both increasing and deceasing flood peaks (Holden *et al.*, 2004; Robinson, 2006; Waddell, 2006). Early research into the hydrological changes as a consequence of gripping carried out by Conway and Miller (1960) suggested that runoff generation in blanket peat systems became exceptionally rapid post gripping. This effect was also noted where extensive gullying or burning occurred. In contrast, Conway and Miller (1960) found that relatively intact peatlands displayed a smoother hydrograph with greater lag times and higher water retention within the peat soils (although still providing a flashy hydrological response compared to better drained lowland soils).

Effects on catchment hydrology from grips are complicated. For example, drainage reduces the water table of the peat soils adjacent to the drain which increases soil water holding capacity whilst the drain channel itself increases the transfer of water from land to stream (Holden, 2009). Moreover, drainage of peat soils increases soil piping, with older drainage networks being associated with increasing soil pipe density, and thus water flow through macropores is increased (Holden, 2005). This further increases transfers of water to stream networks. A number of factors have to be considered when assessing how drains affect hydrology including drain network design, slope and vegetation (Gilman, 2002). Peats shrink, crack and decompose when dried (Holden and Burt 2002b) and this has impacts on hydrology, water quality and ecology. Combined with a lowered water table, these changes increase the likelihood of peat soils becoming hydrophobic during periods of low rainfall thus further adding to quickflow response (Holden, 2009). Holden *et al.* (2008) showed that flow velocities across the surface of grip blocked (i.e. restored) peatlands was slower than in drained peatlands suggesting that the impacts of drainage can be reversed to some extent. Grips can alter the hydrochemistry of runoff water in particular through discolouration by dissolved organic carbon (DOC) which is an expensive cost for water companies.

Pearson (1972) found that numerous grips in Derbyshire had been prone to severe erosion resulting in the deepening and widening of downstream channels and the delivery of large amounts of peat matter and fine sediment to stream networks. This has been confirmed by many examples of research and anecdotal evidence. For example, Carling and Newborn (2007) found that sediment delivery to the stream network from drained peatlands were between 10 and 10000 times greater over a five-year period when compared to intact peatlands. Burt *et al.* (1983) noted a marked increase in suspended sediment following gripping and this was in agreement with Robinson (1980) who showed that sediment concentration in runoff increased by two orders of magnitude during drainage works and that the peat soils took several years to stabilise after the initial change. This suggests that there are spatial differences governing the order of magnitude increase of sediment yields; however, all these studies noted a significant increase showing that extensive drainage adds to the fine-sediment load of downstream channels. From a land manager’s perspective, it has been estimated that hundreds of thousands tonnes of peat soil have been lost from the Raby estate alone due to the presence of grips (Waddell, 2006) and that drains were responsible for losses of grouse broods that became trapped in highly eroded drainage channels. At Oughtershaw Moss, a catchment adjoining the case study location for this thesis, Holden *et al.* (2007)

showed that sediment yields increased substantially during and after drainage with 18.3% of sediment being derived from grips that drained just 7.3% of the area.

Worrall *et al.* (2007) found that in terms of DOC the method of blocking was not important to reduction and so the cheapest method appropriate to the local conditions, typically peat plugs, could be followed. Moreover, the volume of water at the drain channel outlet can be significantly reduced by blocking (Holden *et al.*, 2008a), mean water table recovery can be rapid (Holden 2009) and drain blocking was shown to reduce fine-sediment yields in upper Wharfedale by at least one order of magnitude (Holden *et al.*, 2007b). This appeared to be due to increased sphagnum moss cover around blocked drains which significantly slowed flow. This body of work suggests that locating grip networks and then making some form of assessment to ascertain those which would respond best to blocking could assist with targetting of time and money resources. However, it must be noted that in comparison to undisturbed peatlands, grip-blocked restored peat soils still show altered hydrology and recovery is slower than simple measures such as mean water table depth would suggest (Holden, 2009).

Catchment-scale studies reveal that hydrological response has been altered through land use change; this has changed runoff response to rainfall creating flashier responses (Bunn *et al.*, 2010). Such changes can result in greater erosion rates and enhanced delivery of fine sediment, nutrients and other pollutants from the wider catchment to stream networks. These changes in hydrological processes and sediment transfer rates are good examples of why changes at the river reach/habitat scale must be viewed in the context of the wider catchment (Kondolf, 1998). Many researchers support catchment-scale approaches and identify the catchment as the core unit for management of rivers (Chorley, 1969; Newson 1992; Burt and Pinay 2005). This approach has resulted from the recognition that many forms of degradation can occur across large areas of a catchment, often driven by land-use change (Bond and Lake, 2003).

2.4.2 Floodplain scale

The grasslands of the Pennines reflect a long history of exploitation (Atherden, 1992) and the intensity of this exploitation has increased in post-war years due to political and economic drivers (Marshall *et al.*, 2009). These changes have resulted in higher stocking rates. For example, sheep numbers in the UK increased from 19.7 million to 40.2 million between 1950 and 1990 (Fuller and Gough, 1999) and the growth of herd size on dairy farms more than tripled between 1960 and 1997 (Lowe *et al.*, 1997). These increases in stocking rates have been shown to decrease soil infiltration, porosity and hydraulic conductivity, as well as increasing soil bulk density (Nguyen *et al.*, 1998). Thus, runoff rates have increased in tandem with intensification (Elliot and Klemetsen, 2002). Moreover, soils have become more prone to erosion both by livestock poaching and heavy machinery compaction which further reduces infiltration (Marshall *et al.*, 2009). This results in the combination of high rates of surface flow and increased critical source areas[19] and thus fine-sediment delivery to streams. Sediment loss from agriculture is a cause for concern due to both on-farm practical and economic implications (Boardman *et al.*, 2003) as well as the impacts sedimentation has on stream habitats and ecology (Owens *et al.*, 2008; Theurer *et al.*, 1998).

Diffuse pollution from farming is a major issue in upland streams and is created when sediments and associated pollutants, including nutrients, heavy metals and pathogens (Edwards and Withers, 2008), are connected to watercourses by runoff generated through precipitation and snowmelt (Abaci and Papanicolauo, 2009). In upland catchments surface runoff is the primary pathway for agricultural diffuse pollutants to reach the stream network. Understanding the processes and delivery pathways that connect sediment sources to streams is necessary if mitigation to reduce transfers is to be implemented (Heathwaite *et al.*, 2005). Whilst surface runoff pathways can be highly visible (Deasey *et al.*, 2008), identifying these across a whole catchment is costly in terms of survey time and due to property rights which can restrict access to key areas (Dugdale, 2007). To compound this, landscape heterogeneity means that studying overland flow and erosion/deposition using traditional experimental approaches is not only costly, but also constrained by spatial and temporal variability (Tayfur and Singh, 2004). Abaci and Papanicolauo (2009) found that soil heterogeneity was not a significant factor in terms of the spatial heterogeneity of erosion whilst land management practices can enhance or diminish precipitation impacts on soils in

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[19] Critical Source Areas (Heathwaite *et al.*, 2005) are locations within a catchment that are sources of diffuse pollution such as fine sediment and nutrients. They must be connected to watercourses by surface flow for at least part of the year. They are generally small, sub-field locations and are notoriously difficult to locate at a meaningful scale.

agricultural catchments. This suggests that a more explicit focus on land use may better describe controls on soil erosion from farmland to streams.

In recent years evidence has highlighted how diffuse pollutants coupled with alterations of hydrological flow paths, derived from upland catchments, impact downstream river systems. For example, Marshall *et al.* (2009) noted that, whilst impacts such as flooding occur downstream in lowland regions, it is the flashy nature of upland catchments that are the source of much of the runoff generation. Hence, there is a need to place impacted in-stream habitats into the context of their upstream contributing area, so capturing the appropriate scales and land management. This is important since to decipher the impacts on river systems it is necessary to place the riverscape, its management and restoration firmly within the spatial scale that is most important (Lane *et al.*, 2008). The WFD aims to adopt this holistic catchment-scale approach and provides a driver for sustainable river management in order to bring water bodies across member states up to the standard of good ecological status (Brandt *et al.*, 2004).

In order to bring upland stream networks up to good ecological status, it is essential that pollution pathways and the ways in which land management practices modify these flow paths are understood. As discussed above, much of the spatial and temporal variation in diffuse pollution arise due to land management and there has long been concern that modern agricultural practices in the UK increase erosion rates and surface runoff (O'Connell *et al.*, 2007). How different types of land cover modify soil structure, surface flow and propensity for erosion must be understood in order for restorative measures to be taken. Marshall *et al.* (2009) found that shelter belts of trees as young as ten years old significantly reduce overland flow through 1) the presence of trees and 2) the absence of sheep. Mature forests are known to reduce peak flows due to a number of processes including evaporation of canopy interception, transpiration and an increase in soil water storage capacity beneath trees (Robinson and Dupeyrat, 2005). In comparison, pasture land reduces interception and, due to both livestock trampling and heavy farm machinery, soil compaction may occur which lowers soil water capacity. This inevitably increases runoff rates in comparison to woodland given the same topographical conditions (Marshall et al., 2009). These findings support the work carried out at the catchment scale at Pont Bren (Jackson *et al.*, 2008).

Gburek *et al.* (2000) argue that putting in place simple measures to add roughness to a landscape will reduce surface flow and therefore diffuse pollution. They also note that the delivery of diffuse pollution decreases with distance from a channel and high-magnitude, long return-period storm events are required in order to connect more distant locations. The measures they suggest to add roughness to a landscape include reducing grazing pressures, planting trees and hedgerows or fencing out riparian zones to create well vegetated buffer strips. These measures are supported by work carried out in the Pont Bren catchment (Jackson *et al.*, 2008) and work by Anderson and Flaig (1995) who identified other methods for reducing sediment in agricultural runoff which includes the installation of sediment traps, increased use of culverts and river bank stabilisation through the use of cover crops. These examples show that restoration can work; however, to provide this level of protection and enhancement, comprehensive planning for multiple uses is required. Moreover, linking these processes and impacts to the in-stream scale is an essential aspect when developing restoration plans.

2.4.3 In-stream scale

It is well understood that the delivery, entrainment and deposition of fine sediments are a significant impact on river systems worldwide (Larsen and Ormerod, 2009) and that organisms at all trophic levels are affected by fine sediment. For example, fine sediment can reduce light infiltration and therefore photosynthesis as well as reducing the efficiency of visual predators (Rowe and Dean, 1998; Parkhill and Gulliver, 2002), it alters substrate structure and habitat quality for benthic macroinvertebrates (Tunpenny and Williams, 1980), reduces feeding efficiency of filter feeders and grazers (Graham, 1990) and reduces oxygen supply to salmonid eggs via interstitial occlusion (Heywood and Walling, 2007; Grieg *et al.*, 2005). Yet, deciphering fine-sediment impacts on streams remains difficult due to: 1) other stressors that can either mask or exacerbate the effects; and 2) the scale differences in pollution sources across a catchment (e.g. sediment delivery from the immediate river bank or from drained peatlands many kilometres upstream).

The early life stages of brown trout have quite specific requirements. For example, at the reach scale, egg development requires gravel and pebbles (16 to 64mm) with a minimum dissolved oxygen concentration of approximately 5mg/l, although this can be as high as 7mg/l depending on the developmental stage of the egg (Louhi *et al.*, 2008). Any sustained reduction below these levels reduces survival. At the catchment scale, brown trout show a strong preference towards small streams during spawning periods and at the reach-scale gravel and pebbles must be located at pool – riffle transition zones (Crisp, 2000) whilst they actively avoid step pool and cascade zones (Moir *et al.*, 2004). These requirements carry over to the fry life stage. However, habitat heterogeneity can enhance survival of fry by providing refugia and increasing habitat availability for prey species including macroinvertebrates, whilst at the same time extending the factors that may limit populations. For example, both high macroinvertebrate abundance and richness is positively correlated with medium to large substrate heterogeneity which provide stability, interstitial space for refuge, oxygen exchange, attachment sites for filter feeders and diverse microbial, algal and detritus food supply (Minshall, 1984; Allan, 1995; Wood and Armitage, 1997; Moss *et al*, 1987) and these conditions become more important as brown trout begin exogenous feeding.

Whilst brown trout management must cross scales and habitats to account for the diverse needs through the full life cycle (Dugdale, *et al.*, 2005), by identifying bottlenecks in the life cycle, management can be targeted for specific habitat needs of the more limiting life cycle stages. As identified in section 1.4, the fry stage appears to pose the strongest control on population and has the narrowest niche and thus management can become more targeted than it would be if addressing the requirements of all life stages.

Nutrients including phosphate, nitrate and potassium, and micro-nutrients, pass between the biological and physical components of all ecosystems. In terrestrial systems this is known as a nutrient cycle. However in stream ecosystems there is downstream transport before a cycle is complete (Newbold *et al.*, 1981). Freshwater ecologists consider nutrient cycling within streams as a spiral to account for this movement (Webster 1975; Newbold *et al.*, 1981). Newbold *et al.* (1983) found that phosphorus moved downstream at an average velocity of 10.4 m/d completing a cycle every 18.4 days. The spiralling distance and duration is very dependent on current velocity, physical retention devices such as weirs and, to a lesser extent, the efficiency of the biological community (Minshall *et al.*, 1983). This downstream movement of nutrients and matter as it cycles between physical and biological components of the river ecosystem is an important process in freshwater systems. It means that each location has to be viewed in terms of what occurs upstream, either within the channel or on the floodplain, hillslope and wider catchment.

In terms of brown trout fry survival, fine-sediment delivery is a key concern in drainage basins affected by anthropogenic disturbance (Wood and Armitage, 1997). Impacts on stream ecosystems, including forestry and agriculture, have widely degraded river systems and habitats thus reducing natural reproduction of fish (Calow and Petts, 1994). For example, fine-sediment delivery to streams can reduce dissolved oxygen concentrations by deposition within the interstitial space, reducing flow and oxygen replenishment. Moreover, fine sediment has other impacts. For example, particle size <1mm can result in a film on the redd surface inhibiting fry emergence (Kondolf, 2000) whilst very fine sediment <0.125mm can block the micropore canals in the egg membrane thus reducing waste transfer (Lapointe *et al.*, 2004; Grieg *et al.*, 2005; Julien and Begereron, 2006). Moreover, these same impacts reduce brown trout prey availability thus providing a secondary limiting factor on populations. Numerous research projects have found that changes in macroinvertebrate assemblages are related to hydrological variability that are known to directly affect the physical habitat including structure of bed substrate such as the alteration of substrate composition through the inputs of silt and fine sediment (Chutter 1969; Mclelland and Brusven 1980; Lenat *et al.*, 1981; Bourassa and Morin, 1995; Zweig and Rabeni, 2001).

There is a plethora of research that has provided insight into how salmonid fish populations decline in response to impacts such as fine sediment (Theurer *et al.*, 1998), nutrient inputs (Pretty *et al*, 2003) and habitat loss (Waples and Hendry, 2008). Yet it is important to understand such effects in terms of a functioning (or malfunctioning) catchment that is subject to large-scale human influence. This is essential when aiming to identify the important impacts and contextualise these in terms of the landscape with all its processes and multiple impacts. Modelling, remote sensing and GIS methodologies can help in the search for possible impacts by providing opportunities to survey and map specific locations within the context of the surrounding land use and then model landscape processes that may deliver diffuse pollutants to watercourses.

Before reaching the stage where these novel methods are employed, an understanding of how the land processes impact on river biota is an important first step.

Stewart (1963) carried out studies in the Ribble and Hadden catchments and found that salmon catches fell from 1400 yr^{-1} to 380 yr^{-1} during the 8 years following peat drainage whilst in the nearby Lune catchment, that had little or no drainage, catches remained high and stable. Research carried out in Finland by Laine *et al.* (2001) showed that recapture rates of stocked yearling salmon were reduced in riffles receiving high inputs of particulate matter from drained peatlands in comparison to riffles receiving less loadings of particulate matter. Stewart (1963) discovered links between peatland drainage and slope instability with drains being capable of acting as failure points for mass movements. Moreover, production of particulate organic matter (POM) is significantly higher in drained peatlands than in undrained peatlands and mobilisation of POM has noticeable impacts on macroinvertebrate communities of peat streams (Ramchunder *et al.*, 2009).

Such studies have helped show the link between in-stream habitats and catchment scale processes. This shows how upstream management can place significant controls on river ecosystems through, for example, the delivery of sediments or changes in hydrological regimes. Such impacts transcend scale and move through catchments via pathways controlled by hydrological connectivity with negative impacts being noticeable at small-scale riffle habitats. Thus, understanding the cascades is essential for river managers. In many ways this is easier working from the point of impact by placing riverscapes and localised habitat patches into the context of the overlying scales operating upstream from the point of interest.

Riverbeds can be seen as a mosaic of spatially distinct surface to subsurface exchange patches where the timing and magnitude of exchange is temporally variable (Brunke and Gosner, 1997; Sophocleous, 2002). Exchange processes at the microhabitat scale are driven by subtle changes in topography, permeability and the roughness of the channel bed (Grieg *et al.*, 2007). Obstacles in the river such as large woody debris, logjams and boulders create pressure differentials that enhance surface to subsurface exchange within the hyporheic zone (Vaux, 1968; White, 1990). In systems with minimal human impacts, such microhabitat heterogeneity enhances the ecology of river networks by providing refugia that buffer against ecologically difficult circumstances such as rapid spate events or drought conditions (Poff, 1997). Even in heavily impacted systems, microhabitat heterogeneity provides refugia; however, in such circumstances the buffering capacity may well be reduced. For example, the infiltration of fines and biofilm growth reduces the porosity of gravel matrix surfaces which can then reduce salmonid egg survivorship, habitat availability, refugia and also increase macroinvertebrate drift response (Grieg *et al.*, 2007). Larsen and Ormerod (2009) showed that addition of fine sediment to riffle habitats increased macroinvertebrate drift density by 45% and propensity by 200% with these effects being greatest on the night following addition of sediment rather than biota displaying an immediate response. Whilst benthic macroinvertebrate composition remained the same, density did decline in treated reaches by 30 to 60% and the effects remained consistent between seasons and streams. Organic matter such as that delivered from drained peatlands can deposit within interstitial pores and encourage biofilm growth thus impacting systems as noted above (Grieg *et al.*, 2007). Biofilms form around sediment particles during the breakdown of organic matter and can result in cohesive matrices reducing gravel permeability and inter-gravel flow (Chen and Li, 1999).

Within stream ecosystems, organic matter is the principal nutrient source with the predominant forms being POM and DOM both of which can have autochthonous[20] and allocthonous[21] sources (Grieg *et al.*, 2007). Peat soils are a source of both forms with increasing amounts derived from eroding soils impacted by gully formation, drainage and burning (Holden, 2009). How these impacts interact with underlying scales and map out at the habitat scale is important if restoration effort is to be successful.

2.5 Conceptualisation of process cascades

An important aspect of river research is to conceptualise process cascades that traverse scale and link near and remote spatial locations to river reaches (Flodmark *et al.*, 2006; Lane, 2008). In order to test how these cascades impact in-stream ecological components it is essential to ascertain the important environmental factors for target

[20] Autochthonous describes material that deposits in-situ, for example organic matter in a peat soil.
[21] Allochthonous describes material that did not originate in its present position, for example POM within a stream may be derived from eroding peat soils.

species and develop the appropriately scaled process cascades. Without such conceptualisation, understanding the process controls on geomorphololology, ecology and, importantly, predicting the outcomes of interventions will escape scientific and conservation communities. The complexity of the interactions of catchment processes (that cascade through scales), coupled with land use poses a challenge that should not be ignored. Without such information, emergent behaviour at the river scale will be poorly understood (Tetzlaff *et al.*, 2008) and interventions to improve ecological conditions at the river reach scale are more likely to fail (Boon, 1998).

Gionnani *et al.* (2005) state that, '*catchment response is strongly influenced by the dynamics of water flow movement on the hillslope*.' Lane (2006) argues that upland catchments are threshold-dominated systems and rapid change brought on by multiple pressures may result in breaches and state changes in ecosystems. Assessing large scale and multiple factors affecting the survival of brown trout within upland river catchments is increasingly recognised as an imperative in order to identify limiting processes. Milner *et al.* (2003) argue that small-stream studies create difficulties when trying to scale up to larger ecosystems. For example, a single stream within a whole catchment may miss the pertinent information that a catchment study captures by providing information on the relative condition of a system and its tributaries. Setting the incorrect spatial scale in which to explore systems can result in dubious findings. For example, Larsen *et al.* (2009) found that sedimentation of gravel beds was directly linked to eroding banks within 500m upstream. When they increased the scale of inquiry, they discovered that the bank erosion was negatively correlated with riparian and catchment woodland extent. Small-scale processes such as bank erosion place limiting factors on brown trout and it is now becoming increasingly accepted that such processes must be viewed in the context of upstream land use such as extent of riparian cover and woodland (Jutila *et al.*, 2001; Lane, 2008; Larsen *et al.*, 2009).

River ecosystems are directly linked to larger scale patterns of precipitation, groundwater recharge and evapotranspiration (Ormerod and Durance, 2009). Each of these in turn relate to catchment features including soil type, geology, topography and land use. Emergent behaviour in river ecosystems is thus related to the interactions of these processes operating at scales that overlay the river reach. Research horizons must be broadened to account for these interacting layers. Acidic water can be directly toxic to fish but can also mobilise heavy metals from old mine workings, including aluminium, which are also toxic to fish. At the reach scale, water discolouration is correlated with acid flushes from upstream moorland which are strongly associated with fish kills (Jutila *et al.*, 2003). Discoloured water is associated with areas of hill peat on slopes <5° (Mitchell and McDonald, 1995). Some forms of land use, including upland drainage and peat burning, increase discolouration of water. Jutila *et al.* (2003) found that the upstream area of peatland was a good surrogate for pH. This confirms what Lane (2008) argues, that in order to conceptualise process cascades dropping down through scales and ultimately impacting river ecosystems, it is important to ascertain what matters to the organisms of concern. To do this, he argues that it is important to research where the organism exists and not locations that happen to appear suitable; moreover, choosing an organism, or life stage, with low dispersal properties and requirements that map on to interacting scales and processes is important when trying to ascertain the processes impinging on river quality. The importance of interacting scales and the controls they place on process cascades and river biota cannot be overestimated.

Investigating hydrological processes with little consideration of land-use pressures provides only a partial understanding of the processes and issues operating throughout a catchment that may disturb ecological conditions. Agricultural land use is just one aspect of the human domination of natural systems which add to the impacts of urbanisation, forestry, industry and transport networks. On a global scale, approximately 40% of land area is managed for agriculture. In the UK this rises to approximately 70% (Ormerod *et al.*, 2003). Upland catchments are sensitive to land use pressures with rapid responses in both the quantity and quality of water reaching the river network (Lane *et al.*, 2004). Forest clearance, high stocking rates, moorland management, including drainage and heather burning on grouse moors, all contribute to soil erosion and enhanced runoff rates (Watson, 1990). Ormerod and Durance (2009) argue that biodiversity conservation has shifted away from being solely a moral consideration to a survival imperative.

With regard to salmonids, Hendry *et al.* (2003) argue that the three important variables to study are, 'water quality, water quantity and the physical structure of the riverine environment.' Connections between cultural processes (economic, social, traditional) and their effects on physical processes need to be understood as do the meanings and reasons behind them. So land management across a catchment with its interplay of social, economic and natural processes highlights the requirement for multidisciplinary approaches to knowledge gathering (Kershner, 1997). Land use interacts with natural catchment processes and crosses spatio-temporal scales; moreover, different forms of land management impact water quality and habitat at a number of scales. For example, dredging of coarse sediments to reduce overbank flow may directly impact habitats, by reducing available gravel habitat, whilst grazing of gill sides can compact soils increasing overland flow which can be one of the root causes of the coarse sediment delivery (Lane *et al.*, 2008). If management of river systems is to become more sustainable, then it is these root causes of degradation that must be addressed (Boon, 1998; Lane *et al.,* 2008; Ormerod *et al.*, 2003). This may require a reframing of the issue from diffuse events to a series of distributed point sources (Lane *et al.*, 2008). Whilst all fields in a catchment pose risk to river ecosystems some are more risky than others. It is by placing some form of weighting that allows us to identify which fields have a high risk of delivering pollution that we can reverse the issue from a diffuse pollution problem to a series of small, spatially distributed, point sources (Lane *et al.*, 2008).

The following section (2.5.1) will explore a range of land use pressures that may operate within UK upland catchments and how, as with natural processes, these can cascade through catchments and scales until they impact the water quality of stream networks. Impacts of land management will be discussed in the context of suppressed brown trout populations which have been depleted due to human induced habitat fragmentation (Ayllón *et al.*, 2003; Gosset *et al.*, 2006). The discussion will follow the scales of a catchment from the upper reaches, down the hillslope to the floodplain eventually reaching low-order river networks in order to highlight the variety of land management types of upland river catchments and their effects on in-stream ecology. This brief overview of the issue looks at a catchment in a post-disturbance state after the initial woodland clearances of the previous few thousand years. It will try to identify and conceptualise issues in the context of post-war intensification of land use. It is also important to view the issues as multiple and linked. The impacts of land management on higher elevations are compounded by land use on lower elevations wherever there are routes for pollution sources to connect to rivers.

2.5.1 Interactions between scales and land management

Diffuse pollution of rivers is posing a major problem that hinders the achievement of good ecological status as defined by the WFD (Krause *et al.*, 2008). In order to prevent the delivery of pollutants such as fine sediment, substantial, and often controversial, changes in agriculture are being discussed (Krause *et al.*, 2008). These changes involve breaking the connections between CSAs and the river or changing the land use method that creates the initial problem of erosion. Such measures that can be carried out at catchment or field scale include gill planting, grip blocking, creating buffer strips along riparian zones which delimit terrestrial and aquatic systems (Hattermann, 2006; McGlynn and Seibert, 2003), moving gateways from the downslope section of fields to areas where water is less likely to accumulate or completely changing the farming method in some fields or farms. At the river-scale, methods for enhancement include installing sediment traps, the use of culverts and bank stabilisation through cover crops (Anderson and Flaig, 1995).

There exists a variety of methods to achieve such shifts in management including national schemes such as those run by Natural England: Environmental Stewardship schemes and the England Catchment Sensitive Farming Delivery Initiative, as well as local initiatives including those run by rivers trusts which are often grant led. However, for any of these schemes to be effective, and just as importantly efficient, there is a necessity to quantify or qualify the impact of land management on in-stream ecosystems at various scales (Ott and Uhlenbrook, 2004). The protection and restoration of downstream river systems requires comprehensive planning for multiple uses (Anderson and Flaig, 1995) but first the locations that require restoration must be identified as carrying out restoration across an entire catchment is not feasible (Dugdale *et al.*, 2005).

2.5.2 Multiple impacts on streams

Riverscape systems are beset with complex multiple impacts that are a combination of interactions between socio-economic and ecosystem factors. Whilst freshwater ecologists and hydrologists (and more recently ecohydrologists) have been able to reveal the poor condition of watercourses on a global scale, the problems appear to be increasing rather than decreasing (Hart and Calhoun, 2010). In agricultural catchments these impacts may be low level but acting at numerous locations over large spatial scales resulting in widespread suppression, and alterations to the structure and components, of in-stream biota.

It is pollution of this diffuse nature that is difficult to locate and remedy. This difficulty is compounded by the complex filtering that occurs within the landscape that places controls on the delivery of solutes and sediments (Dillon and Mollot, 1997). Such impacts, and how they combine with the numerous natural limiting factors, suppress brown trout populations and, if stocks are to be improved through restoration effort, it is a necessity to disentangle and place them within the appropriate spatial scale. The delivery of fine sediment is as (or possibly more) likely to be derived from upstream land use many kilometres from the impact as opposed to the nearby river bank.

Fausch *et al.* (2010) argue that, as streams are strongly connected to the wider landscape, they are quickly altered by multiple impacts which can affect uplands, floodplains, riparian zones and finally the streams themselves. Studies in New Zealand showed that, whilst the combined impacts of fine sediment and nutrients result in complex effects on stream macroinvertebrates, the overriding impact was negative (Magbanua *et al.*, 2010). Many of these land management-derived stressors are synchronous (e.g. sediment, nutrients and temperature) resulting in their combined effects being poorly understood (Battarbee *et al.*, 2005). As a result, it appears that the capacity to predict how human activity degrades riverscapes, and at what level this becomes unacceptable, is poor (Downes, 2010). Thus, freshwater and restoration ecologists are faced with complex demands on their effort due to: 1) the multiple impacts that stress riverscape systems; 2) the increasing degradation of stream ecology; and 3) the difficulty in predicting how impacts map out on to ecological components of freshwater systems.

At any given location, an organism has an engagement with numerous external factors of the environment. Some of these relationships are intimate and an organism encounters them as immediate and compelling controls on its survival. These include the quality of its habitat including prey and refuge availability; here these will be grouped as habitat controls. Other factors are less obvious and exist as more remote controls that link to the immediate habitat and impact the organism directly, for example as a food source entering its habitat from an adjacent habitat, or indirectly through the addition of nutrients to its immediate habitat that helps drive the food web. These will be discussed as habitat-adjacent controls. Whilst catchment-scale controls provide the underlying conditions for the continued existence of the habitat, they may also place great pressure on organisms. For example, catchment controls govern substrate and flow rates within the stream and order streams, thus providing habitats for sets of species and connect habitats for life cycles of complex species such as brown trout. Yet they may also pulse pressures through a habitat in the form of flood events or link with land use pressures to deliver devastating pollution events.

2.5.3 Habitat controls

Studying animals in situ requires information on a broad range of factors including the physical and biological components of the habitat. This takes in situ studies beyond the narrow selection of variables available in the laboratory (Pottinger, 2010). Local habitat is where species exist and if their populations remain steady or rise, it can be concluded that the habitat patch is of adequate quality to allow populations to endure through time. Many studies have identified habitat structure as key to an organism's population and community existence. Harper and Everard (1998, p. 395) comment that habitat is the '*result of predictable physical processes and ... sits between the forces which structure rivers and the biota which inhabit them*.'

Understanding river systems needs investigations into its structure (substrate, plant distribution, available refugia) as a method of understanding the overall system (Frissell *et al.*, 1986). Feedbacks within and between physical and biological components of a habitat invariably exist (Harper and Evarard, 1998) and so "cherry picking" single components for study will not elucidate much about the system itself. Brown trout respond to the multivariate nature of their habitat. The species requires distinct sediment structure (riffle habitat with gravel substrate) for spawning (Kennedy and Crozier, 1995) and this habitat encourages the macro-invertebrate community on which emerging fry will begin exogenous feeding[22]. Further structural diversity enhances brown trout populations. Waterside vegetation, undercuts, in-stream tree roots and pools all provide habitat for prey species, provide refugia for the fry at night or, in terms of pools, in low flow (Poff, 1997).

2.5.4 Adjacent habitat controls

Habitats rarely exist in isolation and require external inputs of energy and nutrients in order to function. This is especially so in upland stream systems which rely on allochthonous material to feed ecosystem dynamics. Allochthonous supply of invertebrates to the stream surface can provide up to 50% of brown trout diet and become more important during periods of low flow or during summer months when in-stream macroinvertebrate supply is low (Gustafsson *et al.*, 2010). The removal of bankside vegetation can reduce brown trout populations by removing the source of allochthonous prey (Gustafsson *et al.*, 2010; Edwards & Huryn 1995; Wipfli 1997; Bridcut 2000; Kawaguchi *et al.*, 2003; Zadorina 1988). Gustafsson *et al.* (2010) found that both 0+ and 1+ fish increased their consumption of terrestrial prey in their diet during periods, or at locations, where terrestrial prey sources were readily available; moreover, they found that brown trout of all age classes showed a preference for terrestrial prey. In addition to providing a source of prey, riparian vegetation, in particular trees provide a buffer against increasing summer water temperatures that helps maintain water temperature within the tolerance range of brown trout (Ormerod and Durance, 2009).

Buffer zones along the riparian zone help provide the conditions to promote vegetation that provides the services noted above. They also provide other, equally important, functions. The management of riparian zones provides a buffer against agriculturally derived diffuse pollutants such as nutrients or fine sediment (Clews and Ormerod, 2010). In addition, riparian buffer zones promote the stabilisation of stream banks and reduce stock access along river banks, reducing poaching and further deterioration of the river bank. In these ways buffer strips mitigate against floodplain and wider catchment effects on river systems (Malanson, 1993).

[22] Exogenous feeding occurs in brown trout after the alevin stage and refers to feeding on external food sources such as macroinvertebrates.

2.5.5 Moorland management: grips

Peat contains a high proportion of water which helps to create a process of carbon accumulation with peat providing a net carbon store under circumstances of low management intensity (Wallage *et al.*, 2006). Beyond the concept of carbon stores peat provides other ecosystem services including placing controls on hydrological runoff. Holden *et al.* (2004) comment that the interactions between land cover and management affect both the quality and quantity of run-off reaching streams and rivers. Peatlands create hydrological conditions that affect downslope water courses and these relationships are altered under disturbance.

On these higher elevations of a catchment, land management can affect water flows providing the conditions for downstream impacts. For example moorland drainage can directly affect both hydrological and hydrochemical aspects of rivers (Holden *et al.*, 2004). Changes in water quality from grips include increases in discoloration, H^+, Na, Mg, Ca, NH_4 and SO_4, (Richter *et al.*, 1997; Holden *et al.*, 2004; Adamson *et al.*, 2000; Wallage *et al.*, 2006). Such threats to both peatlands and rivers from altered hydrology may disturb flood and drought regimes with increased severity resulting in a degradation of in-stream habitat and therefore ecology (Richter *et al.*, 1997). Moreover grips alter hydrochemistry which includes an increased release of DOC and water discoloration (Wallage *et al.*, 2006); increase fine sediment loads delivered to rivers (Clausen 1980); increase piping and therefore runoff rates (Holden, 2006); and reduce peat water-holding capacity if it becomes hydrophobic (Holden *et al.*, 2004). Grips alter the nature of natural runoff channels and increase the density of channels overall (Robinson, 2006) thus shifting water across a catchment at enhanced rates.

It has been shown that up to 85% of summer base flow comes from the top 1 cm of a peat soil and 17% comes from the 1 to 5 cm layer (Holden, 2006). Moors with older grip networks have an increased density of soil macropores or pipes (Holden, 2006; Robinson, 2006). From the relationship between drainage and piping coupled with organic content of waters, the sediment load in the runoff can be ascertained (Holden 2006). Unsurprisingly the older the grips the more sediment and organic matter is found in the water. Comparison of data from the 1950s and 2002 to 2004 shows a 15% annual increase in runoff, lower peak flows and longer recession limbs. Grip cutting has a direct influence on river flows as well as soil and water chemistry. The general consensus is that grips will result in altered hydrology and hydro-chemistry (Holden *et al.*, 2004) with some biologists believing such hydrological disturbance regimes are the dominant factor in the depletion of in-stream ecology (Archer and Newson 2002).

It seems the cutting of grips was carried out without much research into either the environmental or economic costs and benefits. Holden and Burt (2002) comment that, '*many UK rivers drain areas of blanket peat, yet little is known about the exact hydrological processes responsible for runoff generation in these areas'*. It is unsurprising that we know little about the changed responses after gripping when we know little about the dynamics of intact systems. However, the research suggests changes in flow regimes, sediment delivery and downstream habitat quality arise after gripping.

Richter *et al.* (1997) argue that as most methods and models for deciding the correct in-stream flow regime have been either reductionist or overly simplistic; they have failed to fully assess the natural flow regime. They argue that a holistic approach is required in order to ascertain the flow regime within the natural variation and seasonality. Such an approach would place the effect of grips within the context of the whole catchment with all the natural and cultural processes occurring therein. However, relationships between drainage and water flow are not always intuitive. Undrained peat lands have also been shown to produce flashy runoff (Holden and Burt, 2002) due to peat soils becoming waterlogged to the point that precipitation cannot infiltrate the soils and overland flow becomes rapid. Changes to the hydrological regime can affect stream habitats with low flows increasing the concentration of pollutants and increased flows leading to risk of wash out of gravel beds which are a primary habitat in upland streams.

The cascade effects of the increased channel density may result in enhanced hydrological connectivity and thus disturb habitats of low-order brown trout spawning streams. Wash out of gravel patches, increases in sedimentation of the gravel interstices, increased discolouration and acid flushes, all reduce brown trout populations through density independent factors. It may also be the case that increases in flow rate open up less suitable habitats upstream of the best habitats (which may have been denuded by the same flows) allowing spawning further up the stream network than would be accessible under natural drainage conditions, thus ensuring that spawning occurs at non-optimal locations. In order to visualise these cascades, it is important to illustrate the possible effects derived from the available literature. Figure 2.7 provides a management map of grips as they cascade through the scales of a catchment through primary, secondary, and tertiary responses until they ultimately effect brown trout populations.

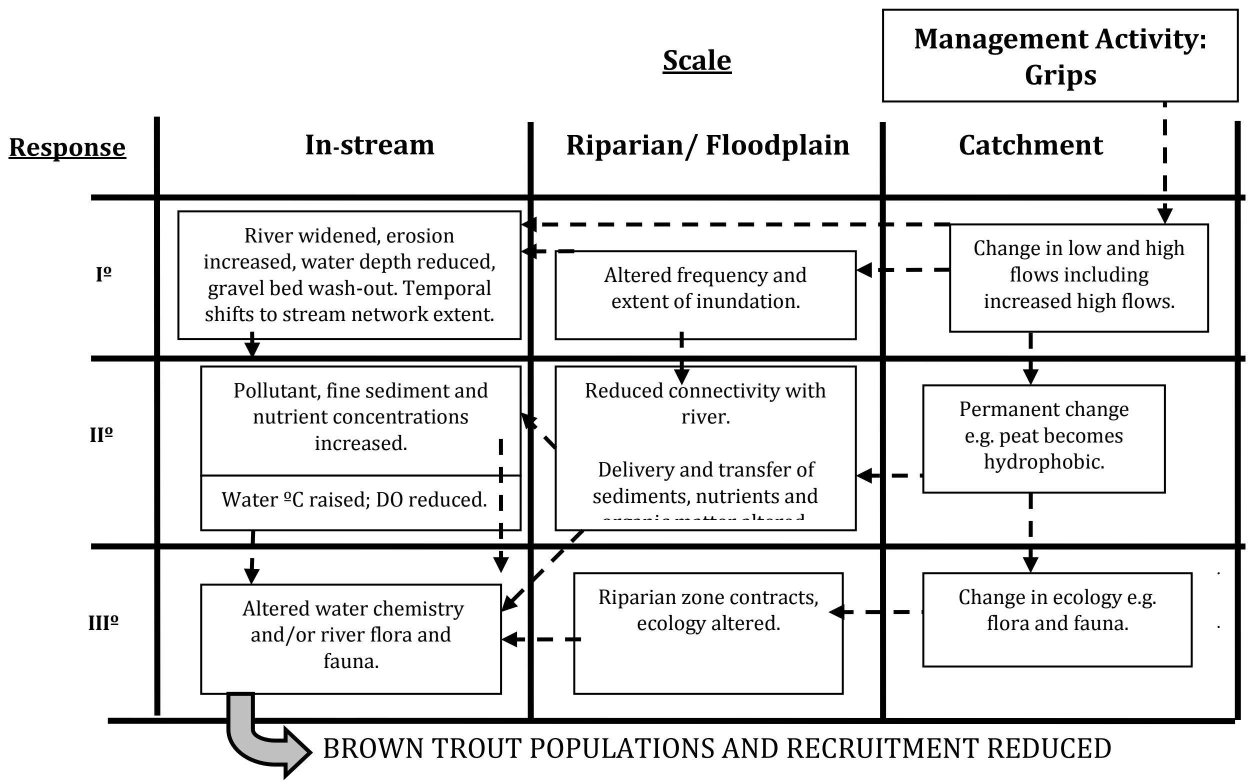


Fig 2.7: *Grips within the context of the three major scales within a catchment and how the consequences of such management can cascade through scales via primary, secondary and tertiary effects till brown trout are threatened from management that at first appears to be far removed from what occurs within watercourses.*

2.5.6 Slurry and overstocking

The hillsope scale of upland catchments is generally managed for livestock farming or forestry plantations. Livestock are present on moorlands but generally at lower stocking rates than on the floodplain. If over-stocking occurs at inappropriate locations then issues of diffuse pollution to watercourses may arise. However, diffuse pollution sources are notoriously difficult to pin down and tackle (Donaldson *et al.*, 2003). According to the Foundation for Water Research (2005) and Freedman (1995) agriculture accounts for approximately 50% of phosphate entering surface waters, and this proportion may increase depending on location and land use. The other major source of phosphate entering streams and rivers is sewage effluents from waste water treatment works. Bowes *et al.* (2005) and Jarvie *et al.* (2006) found that point source (specifically from waste water treatment works) accounted for the majority of phosphate entering lowland water courses. Whether this finding is replicated in upland systems is yet to be seen, though with smaller urban settlements it seems unlikely. Hooda *et al.* (1997) found that the highest concentrations of phosphate coming from agriculture are associated with dairy farming. Growth of herd size on dairy farms has occurred without a corresponding increase in farm size resulting in larger quantities of slurry being spread on the same area of land.

Cultural eutrophication of rivers may occur due to the input of anthropogenically derived nutrients to watercourses that alter community structure, species diversity and chemical composition of the water (Radovejić, 1999). Oligotrophic[23] waters are characterised by a low abundance of nutrients whilst eutrophic waters are rich in plant nutrients (Lawrence *et al.*, 1998). Whilst nitrate is generally the limiting factor in terrestrial systems, it is phosphates that limit primary productivity in freshwater systems (Filepelli, 2002). In oligotrophic conditions, algal growth is limited whilst under eutrophication algal blooms occur due in the most part to additions of phosphate (Radojević, 1999). There are three main forms of phosphate all of which can become biologically available; colloidal (organic), soluble and sediment attached which requires

[23] Oligotrophic applies to soils and water that are poor in nutrients and therefore have limited primary productivity, mesotrophic describes a moderate amount of nutrients whilst a eutrophic system is high in nutrients and has high primary productivity. Eutrophication as used here describes a process of moving a watercourse to a higher nutrient status due to human pressures including nutrient delivery from farmland and waste water treatment works.

reducing anoxic conditions to become biologically available (Lane, 2006). Colloidal phosphate is an anion that strongly binds to soil colloids and is thus not readily available to ecological systems. Soluble phosphate is immediately available to plants once in the watercourse (English Nature, 2000).

The cascade effects of farmland with high stocking rates and associated slurry can be seen in figure 2.8. The cascade follows the same pathways as figure 2.7 to highlight how activity across a catchment can impact streams and ultimately specific biota.

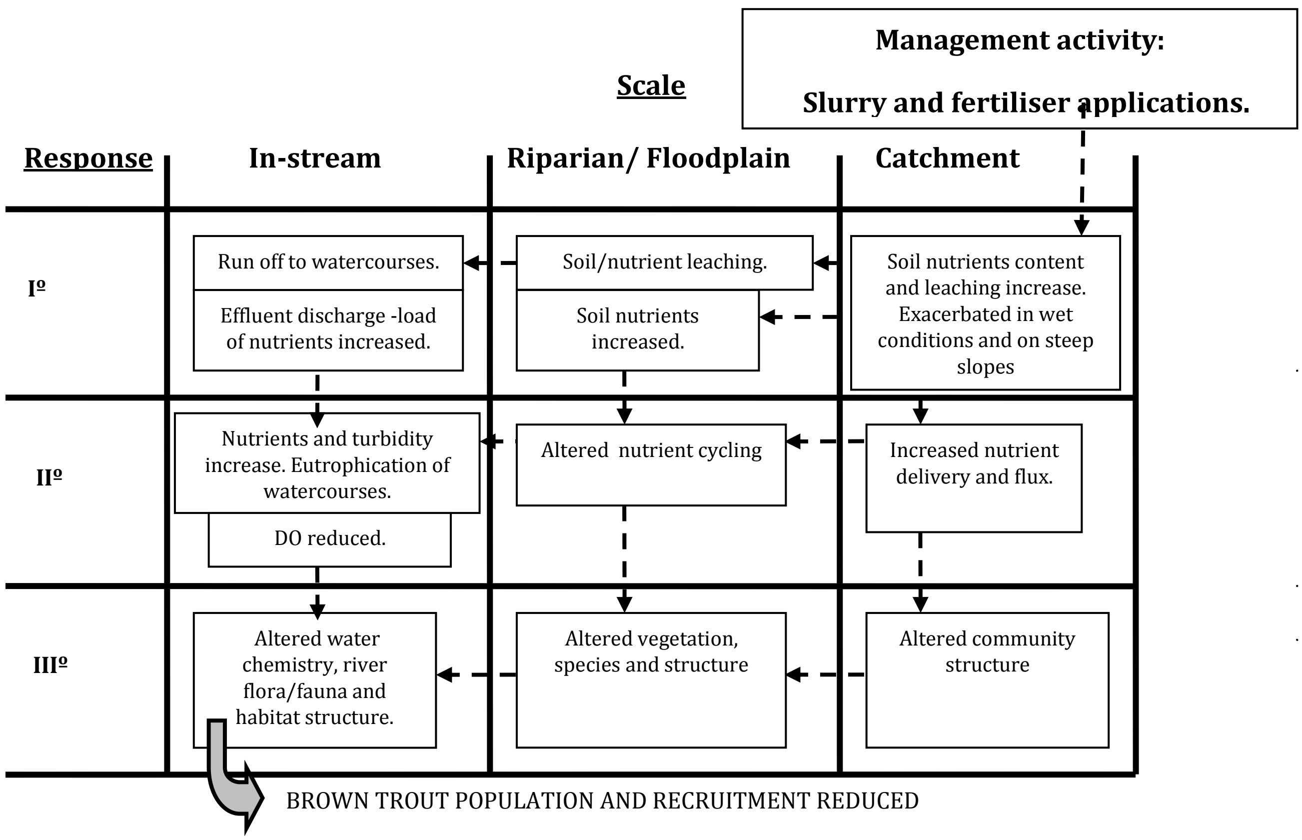


Figure 2.8: *slurry and fertilizer applications within the context of the three major scales within a catchment and how the consequences of such management can cascade through scales via primary, secondary and tertiary effects till brown trout are threatened from management that at first appears to be far removed from what occurs within watercourses.*

2.5.7 Overstocking and poaching

Agricultural practices have increased stream sediment load worldwide (Zimmerman *et al.*, 2003; Naismith *et al.*, 1996). Whilst fine sediment inputs to water courses are a result of natural processes, when these inputs become excessive they act as a pollutant (Waters 1995). Zimmerman *et al.*, (2003) found that lethal concentrations of fine sediment on fish could be reduced by up to 98% due to alterations in land use including the installation of riparian buffer strips, conservation tillage and the encouragement of a permanent vegetation cover.

Livestock farming not only results in enhanced nutrient supplies to soils, with leaching to water courses; it also increases erosion of land and delivery of fine sediments to rivers. For example, erosional resistance is reduced directly by grazing which can expose substrates more vulnerable to erosion (Trimble and Mendel, 1995). They further comment that cows can be important drivers of geomorphological change. Moreover in riparian zones, trampling and poaching can expose soils and erode river banks (Trimble and Mendel, 1995). Theurer *et al.* (1998) argue that livestock farming results in bank erosion through both trampling and poaching and the subsequent deterioration of the grass sward, and thus root depth, that is further exacerbated in wet conditions. Within upland rivers Theurer *et al.* (1998) identified problems associated with enhanced delivery of fine sediments including, '*accelerated stream bank degradation from livestock, major gullying of steep hillsides resulting from overgrazing by livestock and the introduction of grips.*' Connecting runoff compounds the problem by delivering higher quantities of sediments and nutrients.

Over-stocking of livestock compact soils and so reduces infiltration rates thus increasing surface runoff which in turn increases erosion of surface soils (Trimble and Mendel, 1995). These changes in hydrological processes can affect the concentration and delivery of known pollutants including phosphate and nitrate. Alterations of flow rates and types of flow (throughflow, overland flow, etc) can increase the transport of slurry and fertiliser applications from field to river. Lane *et al.* (2006, p.241) state that, '*certain areas are diffuse pollution hotspots, where high nutrient inputs and/or inappropriate land use generate a significant nutrient source that is also connected with a hydrological flow path to the drainage network.*' The pathway between pollution source and water course is most dependent on connectivity of field runoff to river systems. Zimmerman *et al.* (2003, p.94)) comment that, '*conversion of permanent vegetation to cultivated areas with bare soil greatly increases runoff and total sediment loss.*' Overstocking and poaching can result in similar effects especially if livestock have direct access to watercourses resulting in enhanced bank erosion. Walling (1999, p. 238) argues that, '*attempts to understand the linkages between land use, erosion and sediment yield should consider the overall sediment budget and the associated sources and sinks, rather than only sediment outputs*.'

This thesis will concentrate on diffuse pollution and sediment delivery from livestock farming in terms of identifying possible sources of erosion that connect to the stream network. As with grips, it is important to conceptualise the process cascades that result from livestock farming by illustrating the possible routes and effects as they pass through scales en route to entering watercourses. Figure 2.9 highlights how overstocking and poaching can cascade through a catchment with risk moving towards watercourses and ultimately degrading in-stream habitat quality.

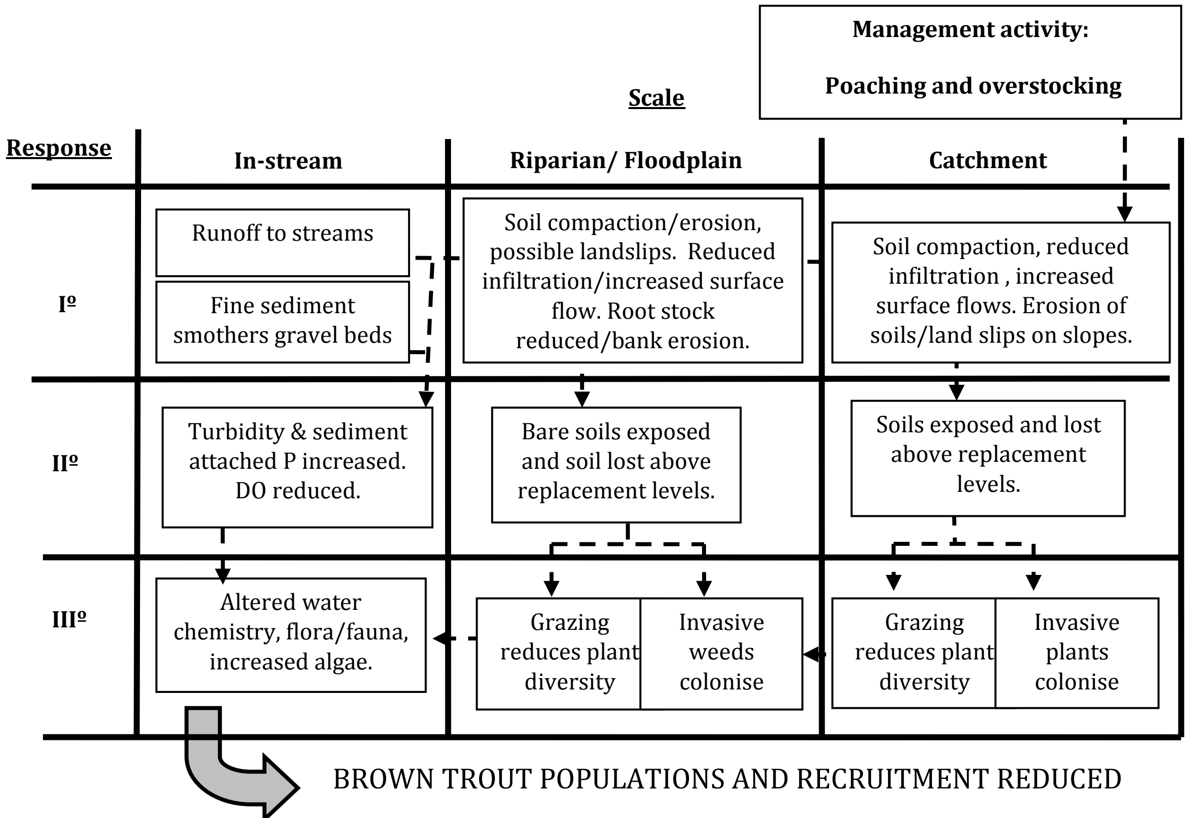


Figure 2.9: *Poaching and overstocking within the context of the three major scales within a catchment and how the consequences of such management can cascade through scales via primary, secondary and tertiary effects till brown trout are threatened from management that at first appears to be far removed from what occurs within watercourses.*

2.6 Scales of analysis

The inter-dependence of scale and ecological response has been implicit in research for decades. Carpenter (1928) recognised that the gradient of the river reach placed a positive control on brown trout populations. Hynes (1960) identified that erosive, high energy zones typified the trout zone of a river system. Nikolsky (1963) noted the inter-dependence between fish and its environment. Frost and Brown (1967) noted that trout streams were associated with hilly landscapes rather than mountainous locations and recognised that trout distribution is related primarily to topographic features including gradient, width and substrate. Mills (1971) remarked that river ecosystems were more reliant on nutrient delivery from surrounding land then lake systems. All these findings suggest that there is a large-scale template where trout are found which in turn requires a large-scale approach to investigation.

More recently the concept of scale has become explicit within hydrological and ecological research and it is becoming increasingly recognized. However, such a scale with its multiple and interacting processes (and competing stakeholder interests) is complex and dynamic (Burt, 2001). For example, with regards to upland drainage channels (grips), it is difficult to tease out whether it is the drainage channels or associated management practices, including higher stocking rates, afforestation, increased fertilizer rates and burning, that increase discoloration and altered chemistry of water (Holden *et al.*, 2004, Robinson 2006). In addition, management decisions often fail to recognise either the appropriate scales or the distorting nature of human impacts (Burt, 2001). Vaughan *et al.* (2009) make the point that the spatio-temporal variability of river systems has been impacted by human activities for over 7000 years. Dufour and Piégay (2009) argue that human populations are part of the river system although this does not distract from the impacts human development has on river systems across a variety of scales.

Bond and Lake (2003) comment that many restoration programmes concentrate on inappropriate scales and the false assumption that creating or improving habitat is key to improving the biotic conditions of streams. These efforts often fail due to poor consideration of numerous other factors operating at larger scales that continue to limit species despite localised improvements in the abiotic or biotic environment. Thus, it is important to incorporate a range of indicators that cross scales in order to identify whether impacts arise at the local habitat (reduced habitat structure), riparian (buffer strips, tree cover, stock access), floodplain (nutrient and sediment transfers) or catchment scale (changes to hydrology and sediment transfers) due to changes in land use and hydrology (Bunn *et al.*, 2010). This requires: 1) careful consideration of the factors to incorporate into investigations; and 2) consideration of interlinked scales. The latter includes local refugia, that species rely on during periods of environmental stress (Schosser, 1995), habitat quality including substrate composition and vegetation, barriers to migration (natural or anthropogenic), riparian and floodplain management and finally catchment scale processes. This multivariate approach is necessary if the impacts and processes important to a species are to be deciphered from the background noise inherent in natural systems.

The linkages between stream and terrestrial systems have been highlighted through research into resource subsidies that have shown the importance to stream biota of cross-ecosystem fluxes of matter and energy (Richardson *et al.*, 2010). Due to the increased edge to area ratio of small streams, allochthonous matter entering these streams is relatively more important to the local food webs (Richardson, 2010). This also suggests that less useful matter and pollutants will have the same kind of relationship in these smaller streams with increased inputs as a ratio to area. As brown trout fry inhabit small streams, a reduction of riparian vegetation combined with changes in land use can be expected to have disproportionate effects on their habitat and so reduce survivorship. Figure 2.10 (overleaf) highlights the importance of these transfers of matter and material revealing the linkages between freshwater and terrestrial ecosystems.

Harvey *et al.* (2008) identify a trend towards developing hierarchical approaches to understanding river systems that range from micro to macro scale considering links between large scale geomorphic processes and smaller scale habitats and their ecological components. The emphasis is often on habitat improvement (Wadeson and Rowntree, 1994; Poole, 2002; Brierley and Fryirs, 2005).

Downes (2010) suggests that information on the causality of stream degradation is poor and thus managers are not in a strong position to know which restoration method should be attempted. This is a problem for both resource apportionment and restoration success. Understanding and sorting between the effects of multiple impacts requires high levels of understanding in order to apply the correct remedies at the appropriate location and scale (Downes, 2010). Bunn *et al.* (2010) suggest that a range of indicators should be incorporated into river monitoring to help identify the important factors and the scale at which they operate. For example, do impacts on brown trout operate at the riparian (shading by vegetation) or reach scale (such as stock access), is it the effect of barriers downstream that govern populations (Pringle, 1997) or does the extent of land use at the catchment scale operate to suppress populations (Bunn *et al.*, 2010).

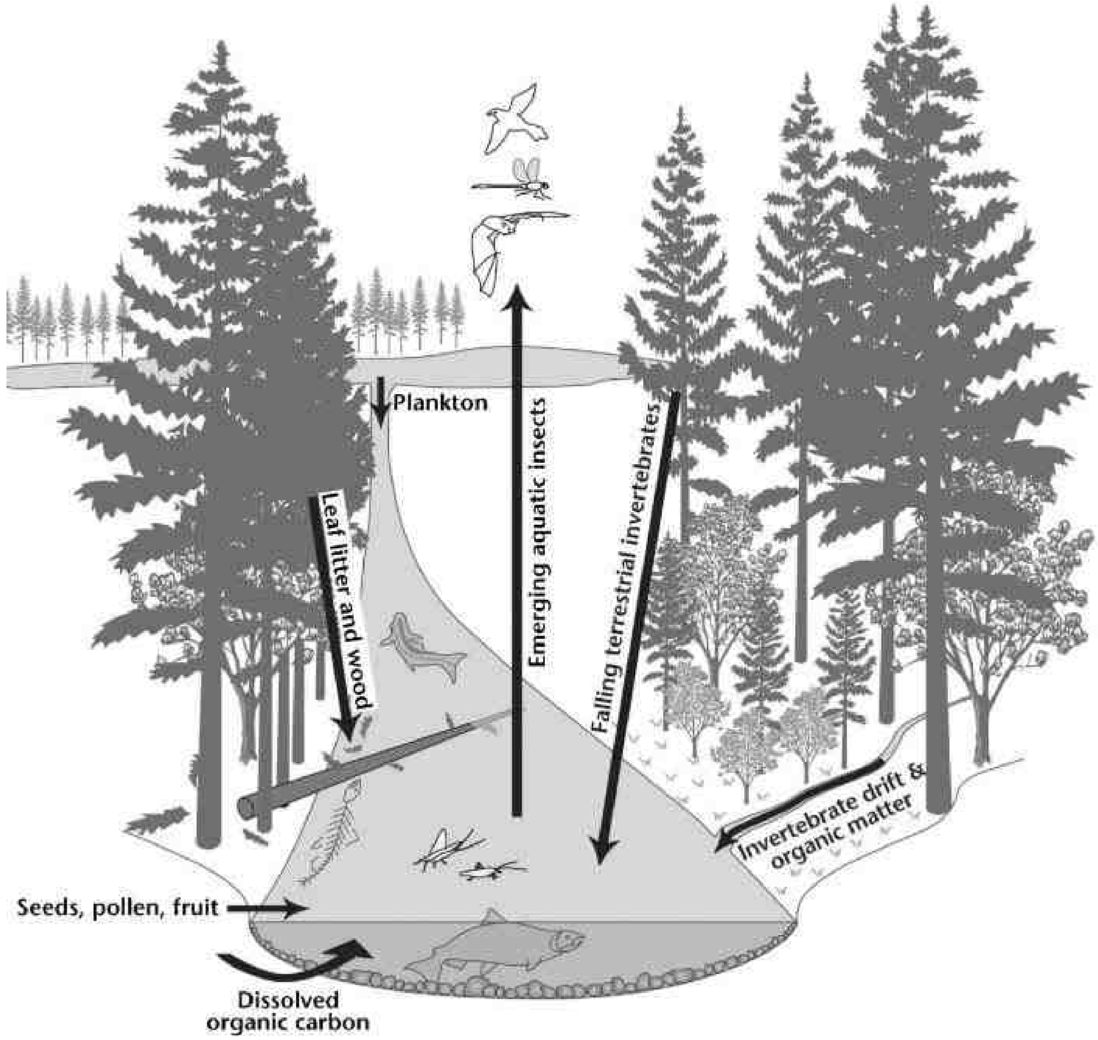


Figure 2.10: *An illustration the major flows of biological energy between stream and terrestrial systems and along a river corridor. Widths of arrows do not imply flux strength (Harvey et al., 2008)*

Much research into brown trout ecology has been carried out in laboratories under near laminar flow conditions allowing single variables to be manipulated and hence their effects on the species assessed. However, in riverscape conditions, laminar flow does not exist, except over designed gauges, whilst multiple pressures do occur resulting in interacting factors that create difficulties in developing complete knowledge of either the processes or impacts that reach through and between scales. Thus, it becomes more difficult to ascertain which factors are 1) limiting; 2) the most important factor suppressing populations; and 3) important limiting factors at specific life stages. Hence, reducing the impact from a single pressure in a given ecosystem may not improve the ecological condition of a river and all that may occur is a shifting of the proportional impact on to other limiting factors that, whilst considered of lesser importance, still suppress populations. This is of concern when restoration ecologists are faced with rivers enduring multiple and interacting pressures and goes some way, along with focusing on inappropriate scales, to accounting for the many examples of failed conservation.

This means that issues of multiple pressures are compounded, or confused, by issues of numerous interacting scales and processes. Thus, even when the important factors are deciphered, in order to carry out effective restoration, we need to know which scales and processes are controlling the impacts that emerge at the local habitat scale. Hydrological changes in watersheds have consistently been linked to changes in the composition, structure and function of aquatic systems (Ward and Stanford 1989; Richter *et al.*, 1996; Bunn and Arthington, 2002; Freeman and Marcinek 2006). And of course, other links exist between in-stream ecology and forest cover, reliable base flows, high quality cobble bed rivers, functioning wetland systems and the health and protection of aquatic biota and associated habitat (Kennen 1999; Ayers *et al.*, 2000; Kennen and Ayers, 2002). Such a myriad of interactions and connections pose complex and difficult questions to researchers when exploring aquatic ecosystems.

Palmer and Poff (1997) argue that stream ecology would benefit from the development of a new approach that views rivers as part of a multi-scale system. Poff and Allan (1995) argue that species can be explained in terms of their functional relationship to a variety of habitat features, which themselves can be described at diverse spatial scales and structured hierarchically (from microhabitat patch up to the watershed). Poff and Allan (1995) further argue that a predictive science of community ecology necessitates an understanding of underlying processes without becoming overly focused on the process detail. In order to engage in effective river restoration, these processes and the downstream ecological response must be incorporated into site assessment and outcome prediction. The emerging discipline of ecohydrology (discussed below), with its implicit focus on hydrological connectivity, scales and ecological response, may well close this gap and provide a conceptualisation of river ecosystems as an emergent response to catchment-scale processes. Approaches that aim towards a more holistic involvement with ecosystem exploration are what river restoration and practitioners require from scientific communities. Effort is now being directed into such endeavour.

Some researchers argue that abiotic variables should underpin theories of community structure. Orians (1980) comments that abiotic factors are minimally influenced by any co-evolved relationships between species and therefore habitats with similar abiotic conditions should contain species with similar attributes and adaptations and thus functional groups. Poff (1997) adds to this by arguing that species traits can only be predictive if they are placed within the context of their functional significance in relation to habitat controls. He further comments that for certain traits such predictive relationships are already known, for example substrate size and availability places a strong, and well understood, control on salmonid fish. A concentration on abiotic factors is only useful if such factors are viewed in terms of their filtering[24] abilities through which species pass at any given location (Tonn *et al.*, 1990). The presence or absence of a species is dependent upon their ability to adapt to, and so pass through, such selective landscape scale and site-specific filters which, as Poff (1997) explains, are simply habitat features that exist at a variety of scales. In order to pass through these filters species traits must be adequately adapted to enable them to match the filtering characteristics.

Hydrological variability, measured at the catchment scale, serves as one such filter on community composition. Such variability does not act through direct mortality but by influences on local habitat structure which can select against certain traits (Richards *et al.*, 1997). Whilst catchment land use may enable the prediction of local habitat (Roth *et al.*, 1996) and water quality (Hunsaker and Levene, 1995), such large-scale filters cannot explain all variability as they provide detail only on average fine-scale habitat conditions (Allen and Starr, 1982). For example, local-scale habitat features can buffer against such larger-scale filters through the provision of refugia such as hyporheic zones, undercuts, eddies and tree root structures (Townsend *et al.*, 1997). The buffering effect of such fine-scale features against the filtering effects of larger-scale processes suggests that to understand species compositions, multi-scale (and multiple pressure) analysis has to be adopted. But first, the large-scale processes must be deciphered in order to develop later understanding of how local buffers provide refugia against the overlying scale processes.

[24] Poff describes a landscape in terms of how it filters processes, and impacts that may otherwise be limiting to in-stream ecology. For example the presence of buffer strips may filter nutrients and so reduce the risk of eutrophication, or a thick grass sward may reduce surface flow and so fine sediment delivery.

2.7 Hydrological connectivity

Bracken and Croke (2007) argue that the notion of hydrological connectivity is a concept that can provide a sound theory for runoff generation and flood production. The concept describes the movement of water from one location within a catchment to another (Bracken and Croke, 2007). They describe a measure of hydrological connectivity termed the 'volume to breakthrough' which could help quantify hydrological connectivity. Bracken and Croke (2007, p. 1758) define this as, '*the accumulated runoff volume per unit width to be applied at a point before flow appears at a downstream point.*' Hydrological connectivity is affected by processes such as infiltration which in turn is effected by vegetation and soil type. Buffer strips may sever connections if the vegetation is coarse allowing runoff to infiltrate before reaching a watercourse. Here temporal aspects of hydrological connectivity come into play. During high-intensity rainfall events paths that are severed may well reconnect due to the increased runoff generated. However, in lowland UK buffer strips may act as rapid conduits for hydrological connectivity due to agricultural drainage (Burt and Pinay, 2005). Spatial and scale aspects of hydrological connectivity govern which plots of a catchment become connected. Hillslopes may readily produce connected runoff whereas at a catchment scale greater intensity rainfall events are required to create connected runoff and flood events (Bracken and Croke, 2007).

River ecosystems are influenced by a number of landscape processes that become connected through hydrological pathways (Reaney *et al.*, 2007) that may, or may not, themselves be affected by land management through drainage, soil compaction, deforestation and urbanisation. To result in an impact on a river habitat, a pollutant source arising from land use must be connected by a delivery agent to a water course. The most common connecting agent comes in the form of hydrological flow paths. Hydrological connectivity is a key driver for sediment, nutrient and ecosystem functions (Michaelides and Chappell, 2009) and place strong controls on aquatic habitat through space and time (Tischendorf and Fahrig, 2000). It is these connections running at a variety of scales, the land use impacts they connect with, and how they act as delivery agents of pollutants through time and space, that requires research in order to ascertain how processes and impact combine to diminish in-stream ecology.

Taylor *et al.* (1993, p. 571) define hydrological connectivity as, '*the degree to which hydrological connectivity facilitates or impedes movement between habitats.*' This definition relates hydrological connectivity to the ecological components of a system. However, in the focus of this research, it needs expanding to include how hydrology provides a pathway that connects pollution sources to stream networks combining the physical and ecological components of a catchment (Bracken and Croke, 2007). In river corridors, it is water that plays the key role of connecting habitats (Amoros and Bornette, 2002). Tetzlaff *et al.* (2008) argue that conceptualisation is a must to understanding, and predicting, catchment hydrology.

Hydrological connectivity has four key components that expand and contract through time dependent on local conditions: 1) lateral connectivity (e.g. river to floodplain, and vice versa); 2) vertical connectivity (e.g. surface to groundwater); 3) longitudinal connectivity (e.g. headwaters to estuary); and 4) temporal connectivity (e.g. changes in connectivity through time). These basic concepts are now accepted in the literature as the basics for understanding hydrological connectivity (Amoros and Bornette, 2002; Pringle, 2003; Tetzlaff *et al.*, 2007; Vaughan *et al.*, 2009). Whilst, for the purposes of conceptualisation, these distinct measures of connectivity are important, in reality hydrological connectivity exists as a continuum with tight (rainfall - runoff), loose (rainfall - groundwater) (Nadeau and Rains, 2007) and ecological linkages (longitudinal connectivity - salmonid migration).

Different stages of the brown trout life cycle respond to this continuum, in conjunction with habitat quality, as the triggers for dispersal and movement between life stages and it is therefore critical to maintain connectivity within the range that brown trout require (Nadeau and Rains, 2007). However, salmonid habitats have been fragmented by human interventions (weirs, altered hydrology, dams) which disconnect hydrological pathways thus impeding migration and dispersal (Gosset *et al.*, 2006; Rahel, 2007). There is now an urgency to reconnect habitats in order to ensure the viability of salmonid species and the economic benefits that accrue from their fisheries.

Connecting pathways within a catchment determine what pressures arising from land use are delivered to the stream network. Catchment characteristics and land use pressures, including drain density, stocking rates, soil type, geology and slope can all conspire to create synergistic effects at the lower reaches of the catchment. If artificial drainage density within a catchment is high, the effects of drought can be either mimicked or exacerbated, by producing a peat soil that is hydrophobic for example (Holden *et al.*, 2004). This results in aerobic conditions as air-filled porosity increases within the peat, thus producing a rapid increase in nutrient cycling (up to fifty times faster than anaerobic conditions, Holden *et al.*, 2006). This affects the water quality of runoff and ultimately affects the water in streams and rivers (Holden and Burt, 2002). Moreover the connectivity of runoff is changed.

In-stream ecology can be severely altered or depleted by changes in runoff rates and water quality. Low summer flows can result in an increased concentration of pollutants and increase light penetration producing ideal conditions for algal blooms. Low flows may also increase summer water temperatures resulting in lowered DO that may be worsened by night sag of DO as plants switch from photosynthesis to respiration. Moreover, water stress itself can be a limiting factor on brown trout. Drainage networks can also result in flashier runoff and produce sharp spikes in the hydrographs which may result in wash out of gravel beds forcing downstream migrations of fish and macro-invertebrates and increasing bank erosion. This highlights how a change in hydrological connectivity at the top end of a catchment can cascade down to impact watercourses.

Rivers are complex systems, characterised by multi-scale interacting processes that result in an '*integrated hierarchical set of subsystems, each of which exhibit some range of scale free behaviour*,' (Church, 2007). Hydrological connectivity links these subsystems either weakly or strongly with both slow and rapid change through time. The concepts of hydrological and habitat connectivity are integral to ecosystem functioning at a number of important scales. Hydrological connectivity between surface flow and river systems help switch on migration episodes of anadromous salmonid fish (Tetzlaff *et al.*, 2007) whereas habitat connectivity allows life cycles to complete providing pathways for fish to disperse between habitat cells (Tischendorf and Fahrig, 2000). This concept of connectivity works at smaller scales, allowing salmonid fry to move from gravel interstices to suitable territories or 2+ fish to move between spawning streams and the main river stem (Klemetson *et al.*, 2003). It is such complex life cycles on which hydrological and habitat connectivity pose major controls. The links between hydrology and ecology have been termed ecohydrology (Hannah *et al.*, 2004, p.1) which they define as, '*the study of the functional inter-relations between hydrology and the biota at the catchment scale.*'

Catchment processes, connectivity and ecological response are tightly interwoven so that misunderstanding of ecology may arise if research is carried out at fine spatial scales that fail to account for how populations, communities and ecosystems, develop through time (Tetzlaff, *et al.*, 2007). However, testing hydrologically connected catchment processes is beset with complexity due to 1) a lack of knowledge of scale (Lane *et al.*, 2008); 2) organisms being mobile and existing at a number of spatial scales (Lane *et al.*, 2008); and 3) poor collaboration between hydrologists and ecologists (Hannah *et al.*, 2004);. If ecohydrology is to flourish as a new discipline; it requires transdisciplinary work (Zalewski, 2000) so that, '*disciplinary boundaries are dissolved and new (hybrid) disciplines formed,*' (Lane *et al.*, 2006, p.240).

2.8 Good ecological status and river restoration

After disturbance events, ecosystems are stabilised by complex internal dynamism and feedback (Reynolds, 2002). However, Downes (2010) argue that there are no optimal conditions within biological populations or communities and such systems constantly respond to changing conditions imposed upon them from both biotic and abiotic components of the environment and the linkages between local systems and wider-scale processes. This poses difficulties when seeking evidence on ecological condition and then deciphering how far removed an ecosystem is from "natural", if indeed the concept of "natural" is meaningful in ecological terms. Thus, understanding how and where ecological restoration of riverscapes is best located within a catchment is beset with complexity borne from interacting processes that combine to 'muddy the waters' of ecological restoration. This should not stop restoration effort or the processes by which such effort is targeted. It may be that we can know enough to develop restoration targets and methods even if we can never fully understand all the detail or recreate some vague notion of natural. The aim of restoration ecology perhaps should be to gather enough information to direct effort to the most appropriate locations, and to accept that mistakes will be made and that knowledge advances from error: management must, therefore, be adaptive.

Indeed, new knowledge in the field of restoration ecology, and science in general, is often derived from failure. However, in a pragmatic, resource-limited, world, failure is not only financially costly it also poses costs in terms of confidence and acceptance amongst the wider stakeholder community. Moreover, environmental and ecological practitioners cannot work effectively with scientific communities and the tools they develop if they run a high risk of being refuted through future evidence acquisition that may well come from failure of activities directed from the 'best available' science at the time. Such failures may sit well within the scientific method but it is neither practical nor acceptable to use practitioners as an experimental base to complement scientific development (Lane *et al.*, 2006). Whilst failure cannot always be avoided, methods for restoration targeting require a degree of testing before they are passed onto practitioners. As Søndergaard and Jeppesen (2007) argue that one of the main challenges in restoration ecology is how to improve the physical condition of rivers in a cost-effective manner. Passing on untested tools would not be cost-effective or agreeable to the restorative process. Such a mistake could engender cynicism amongst stakeholder groups many of whom have competing views regarding catchment, land and river management. The issue of conflict will be addressed in Chapter 3 when the case study catchment is discussed.

Freshwater ecologists are now beginning to conceptualise river habitats as riverscapes embedded within catchments that function as linear, continuous and heterogeneous habitat patches (Schlosser 1991; Stanford 2006). From this conceptualisation of rivers as riverscapes, Sear *et al.* (2008) note that river management aims to develop sustainable management of water resources that are viewed as integral components of catchments. Understanding how connectivity places controls on ecosystems and habitat patches is required and research is beginning to address this issue. Moreover, research must aim to understand ecohydrology under the context of anthropogenic pressures (Vaughan *et al.*, 2009) including large-scale impacts such as climate change, meso-scale impacts such as altered runoff from hillslopes, and localised events such as bankside erosion due to intense land use from livestock (Theurer *et al.*, 1998; Trimble and Mendel, 1995). To understand riverscapes in a holistic sense river reach scales need to be understood in the context of upstream land management and processes (Lane *et al.*, 2008).

New developments in catchment-scale stream restoration programs across the world mark a move away from local reach restoration measures, often incorporating hard engineering, towards a more ecosystem-centred approach to river restoration (Hillman and Brierly, 2005). This appears to be part of the adoption of catchment-scale interventions which promotes community involvement and aims to create widely supported, achievable ecosystem and community outcomes (Hillman and Brierly, 2005). The move to catchment river management that views the river in holistic terms has, however, been slow (Hillman and Brierly, 2005) despite growing acceptance that local habitat scale interventions are generally ineffective.

It has been noted that restoration attempts often fail due to poor consideration of basic geomorphological controls at the catchment and reach scale (Kondolf, 2000). Without these spatial contexts, in which to inform management and restoration decisions, knowledge on the controlling mechanisms will be poor and restoration will be prone to failure despite perceived improvements at the local habitat scale (Boon, 1998). Indeed this has often been the case.

2.8.1 Conceptualising approaches towards restoration

A reduction of hydrological connectivity may hinder seasonal migrations of in-stream organisms resulting in reduced recruitment. Conversely, at larger spatial scales, humans can dramatically increase connectivity (Rahel, 2007), for example through changes in flood pulses. Flood pulses can enhance seasonal migrations of spawning salmonid fish. These changes in connectivity can result in altered migratory patterns, isolation of habitat patches, increased delivery of pollutants and threshold breaches in river ecosystems over a number of space and time scales. Lane *et al.* (1996) comment that research has often failed to consider hydrologically mediated transfers of material from the catchment to a riverscape that has also been altered through human land use change. This is despite research showing that hydrological connectivity drives transfers of nutrients and sediments. Moreover, as topography places strong controls on the catchment – stream linkage, land use impacts can only be fully accounted through an appreciation of their location within a landscape (Lane *et al.*, 1996). By not accounting for scale, connectivity and multiple impacts research fails to develop the correct information. Wissmar and Beschta (1994) argue that river restoration should aim to; 1) reconnect linkages between organisms and their environment 2) restore natural processes and the rate at which they occur, or, more pragmatically, 3) remove human pressures as far as is practical.

In order to plan restoration schemes, knowledge of the system, the landscape processes that impact upon it and organism responses are integral requirements. Heterogeneity in both space and time characterise riverscapes and this has to be a key consideration in designing sampling methods to develop required awareness of riverscapes and impacts (EPA, 1995). Moreover, results from one river system cannot be assumed to match the prevailing conditions in another. However, process concepts and cascades can be transferable as can methods for identifying the pertinent issues. White and Walker (1997) argue that every site has to be placed within its own spatial context. This is indeed true but whilst all systems differ to greater or lesser extents similarities also exist.

Fluxes of species, water, nutrients and weather systems all link communities and ecosystems with the wider landscape (Parker and Picket, 1997). However, in fragmented landscapes, where restoration sites are likely to have had their spatial context disrupted by human activity (agriculture, urban developments, quarrying, etc), it is even more integral to develop knowledge of ecohydrological processes and response at the catchment scale in order to develop the most efficient approach to restoration. Thus, by developing knowledge of catchment processes and the interactions with land use, it becomes possible to target locations having the greatest downstream impact (Lane *et al.*, 1996). Yet, too often management decisions are made with poor consideration of the changing state of the environment or the cumulative impact caused by human activities (Burt, 2001).

Habitat fragmentation creates difficulties for dispersal both in and out of a given habitat (White and Walker, 1997); hence the need to restore connections between sites to ensure viability of locations, species, communities and ecosystems through enhanced migration and linkages (Noss and Harris 1986), though some connections pose negative impacts on river systems, such as hydrological connections between pockets of soil erosion and the river network. Assumptions of species behaviour and habitat requirements must be accompanied by research to ensure conservation or restoration measures are able to target resources effectively. Moreover research can identify which habitats are important and therefore which should be put forward for restoration measures.

Ecosystem restoration has been defined as, '*the return of ecosystems to conditions that resemble their pre-disturbance state,*' (Wissmar and Beschta, 1998, p.571); '*a holistic process not achieved through the isolated manipulation of individual elements*,' (Cairns, 2006, p.1); bringing systems back to the stage where they have '*the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity and functional organisation comparable to that of the natural habitat of the region*' (Karr and Dudley, 1994, p.56). However, even if the pre-disturbance state of a river system was known, it is likely that under present ecological conditions it may not be possible to re-establish such idealised ecosystems (McDonald *et al.*, 2004). Under the prevailing human pressures placed on natural resources achieving restoration of ecosystems back to their pre-disturbance state appears unrealistic (Landers, 1997; McDonald *et al.*, 2004) as does catchment scale restoration (Boon,1998) with limited resources and issues of property rights.

River systems follow complex trajectories that, when placed under pressure, present new and often surprising emergent patterns which are difficult to drag back to states regarded as pristine (Dufour and Piégay, 2009). Therefore, more pragmatic approaches are required that reflect socio-political constraints placed on restoration effort (Dufour and Piégay, 2009). Such approaches involve objective based decision-making based on locations known to: 1) have an impact on rivers systems; and 2) have landowner permission for restoration effort. Such effort may include local-scale measures that ameliorate negative impacts such as gill planting, grip blocking and buffer strips (Landers, 1997). Moreover, as Lane *et al.* (2006) comment such measures not only disconnect pollution sources from rivers, they also have secondary ecosystem benefits. The challenge is to identify which locations matter most and so it is necessary to explore catchments to locate scale, and process, which matter. Catchment scale restoration requires approaches that can target numerous localised impacts, and restoration potential, at the appropriate scale in order to improve river ecology.

2.8.2 Hydrological approaches

In order to ensure that river ecosystems can be restored, Ehrenfeld and Toth, (1997) argue that the primary need may be to re-establish water flows. This is backed up by Archer and Newson (2002) who argue that hydrological disturbance is the dominant factor in the depletion of stream ecology. Giannoni *et al.* (2005, p. 567) state that, '*catchment response is strongly influenced by the dynamics of water flow movement on the hillslope*.'

Grips result in increased overland flow rates resulting in rapid runoff response to rainfall and possible flooding. They may also reduce summer base flows (Conway and Millar, 1960). This can result in wash-out of gravel beds and increased siltation, both of which place negative pressures on brown trout populations (Frost and Brown, 1967; Heywood and Walling, 2007; Theurer *et al.*, 1998; Shields *et al*, 2006). By reducing summer base flows, pollutants will become concentrated and habitat reduced. Moreover, reduced summer base flows can raise water temperature. Armstrong *et al.* (2003, p.159) state that as, '*fish are poikilotherms, many of their vital activities are triggered by temperature or have rates that are controlled by temperature.*' Therefore, a change in the temperature regime can have severe effects on brown trout biology. Richter *et al.* (1997) believe that, as most methods and models for deciding the correct in-stream flow regime have been reductionist, they have failed to assess fully the natural flow regime. They argue that a holistic approach is required in order to ascertain the flow regime in relation to natural variation and seasonality.

In terms of brown trout populations it is fragmentation and degradation of habitats through altered hydrology, barriers to migration (weirs, dams) and pollution episodes that hinder movement between life stages and are most apparent during spawning migrations and the early life stages (egg, alevin and fry). Richter *et al.* (1997) argue that a new approach is needed to quickly define initial, interim river management targets based on the natural flow regime that will serve as a starting point to begin restorative management efforts. Such an approach would enable improved management of catchments through both the identification of the issues and knowledge dissemination between researchers and river managers. However, ecological principles are just one of the key underpinning aspects of restoration ecology; a need for greater integration of approaches is required in order for hydrology and ecology to be explored as coupled concepts. Restoration ecology has to further advance the notion of cross-disciplinarity and integrate diverse disciplines including hydrology, geology, forestry, engineering, agricultural science, economics, sociology and geomorphology (Landers, 1997) chosen on a case by case basis. In order to achieve holistic ecological restoration cross-disciplinary working must also be integrated with the appropriate scale of interest at each site.

2.8.3 Single issue and multiple factor approaches to restoration

Whilst water flows are undoubtedly important, restoration effort cannot be conceptualised using this one factor. For example, the availability of spawning habitat is influenced by several factors including substrate size, stream width, barriers and temperature (Armstrong *et al.*, 2003). Survivorship of fry is further influenced by fine sediment inputs, nutrient loadings and surrounding land use (Theurer *et al.*, 1998; Armstrong *et al.*, 2003). As has been discussed earlier, pressures on salmonids are multiple, interconnected and cross scales. This suggests that spatially distributed multivariate studies are required to discern relative abundance of species being investigated, at appropriate scales and life stages, coupled with habitat variables in order to develop an understanding of the processes, factors and scales that matter.

River restoration is generally limited to working on small ‘bite sized chunks’ set within a catchment and involves work with farmers and landowners in order to move towards sensitive land management (Lane *et al.*, 2006). Such an approach aims to restore ecological function rather than claw systems back to their pre-disturbance state (Kondolf, 2000). It is within this context that river restoration is carried forward. Unfortunately restoration often fails to achieve its objectives due to a lack of consideration of basic geomorphological controls (Kondolf, 2000). Such failings are costly not only in terms of resources placed into any scheme but also in terms of future confidence in the credibility of restoration ecology. There are a number of reasons identified to explain why restoration fails including:

1) concentration on only charismatic species;
2) failure to address communities;

3) reactive restoration that intervenes when a population/community/system is beyond recovery (Nehlson, 1997); and the,
4) scale of restoration is too localised.

Nehlson (1997) argues that an ecosystem approach focusing on processes, habitats and functions addressing underlying causes rather than symptoms is the most effective approach. This is reinforced by NRC (1992) when they state that restoration is, '*a holistic process not achieved through the isolated manipulation of individual elements.*' Such an approach could help restore ecosystem types across a landscape which then helps to maintain populations of important species and the systems on which they rely (Noss and Cooperrider, 1994). This appears to be favourable in comparison to single-issue restoration objectives that may well appear to improve local habitat structure but fail due to a lack of consideration of upstream processes (Boon, 1998). For example, Pretty *et al* (2003) found that, after habitat and channel restoration, fish populations were still being limited by eutrophication. This highlights that single-issue approaches can create improvements in one area (e.g. the condition of the physical habitat) only to reveal that it is multiple issues that are impacting riverscapes and other factors continue to limit biota.

Salmonid populations exist in a patch dynamic that approximates a metapopulation (Rieman and Dunham, 2000). The spatio-temporal characteristics have been disturbed by human interventions to the stage that such effects are now the major control over the phenotypic responses of salmonids (Waples and Hendry, 2008). They further comment that population declines and local extinctions within salmonid species are generally driven by insensitive human interventions that result in reduced habitat quality and decline. Due to their status as species of high economic and recreational importance and their use as indicators of ecosystem health (Nehlson, 1997), their decline poses concerns on economic, social and ecosystem health grounds.

In response to the decline of salmonid populations, restoration effort should aim to restore populations back to abundant, self-replicating populations thought to have existed prior to severe human interference in catchments and riverscapes (Suding *et al.*, 2004). This requires awareness on all the limiting factors faced by the species coupled with an approach that accounts for appropriate scales and process cascades (Flodmark *et* *al.*, 2006) placing salmonid species firmly into the context of the ecosystem and the connections on which they depend.

2.8.4 Synthesis

Developing awareness of the non-linearity of natural systems has resulted in a broader perspective of landscapes within the field of ecology and restoration ecology appears to be following this theme. White and Walker (1997) argue that the, '*nature of the matrix surrounding sites, the nature of edges and boundaries, and the size, distribution, and isolation of the sites themselves,*' are the important considerations. Using self-sustainability as a measure of success (SER, 2004) means species requirements must be known. If this is not the case, restoration is likely to fail. Assumptions of species behaviour and habitat requirements must be research-led in order to ensure conservation and restoration is able to target resources effectively (Palmer *et al*, 1997).

Forman (1995) believes, '*restoration will fail if dispersal corridors are not put in place…we need critical threshold connectivity between the restoration site and regional pools.*' Successional processes, connections between habitats and ecosystems as well as dispersal dynamics are central to ecology and are therefore a necessity in restoration practice. For example, in a '*fragmented forested landscape, the primary goal may be the provision of additional habitat or re-establishing connectivity for particular target species, whereas in a modified river or wetland system, the primary need may be to re-establish water flows*' (Ehrenfeld and Toth, 1997) whilst in other locations reducing fine sediment inputs may be the key challenge (Theurer *et al.*, 1998).

The restoration of salmonid fisheries in upland river systems in the UK is considered a high priority. In order to re-establish a working oligotrophic river system, the wider catchment dynamics and possible future emergent systems arising from climate change, inter alia, cannot be ignored. However, it is often the case that pre-disturbance data on river systems is absent (Cowx, 2004). Restoration ecologists are in the position of having to interpret and piece together past events whilst predicting future ones. This lack of information leads Hobbs and Harris (2001) to argue that whilst measures of success should be based on food web complexity and symbiotic relationships, in reality, success has to be based on lesser measures such establishing re-introduced species, dispersal in and out of the system and regular surveys to monitor diversity and richness as it changes over time.

2.8.5 Implications

Restoration ecology is still in its infancy and the difficulties inherent in the transfer of knowledge from reductionism to holistic ecosystem-scale practice have offered only limited success. Assumptions, coupled with hope, are sometimes the best available approach highlighting the need for increasing focus on catchment scales. It is the role of research to reduce these uncertainties and allow restoration ecologists to target locations based on the best available science.

The basic requirements of restoration have often been ignored which has been at the expense of success. This has resulted in a wasteful approach to finite resources. Poppet *et al.* (1993) identified two approaches to ecological restoration; 1) strategy and 2) tactics. A strategy approach is comprehensive covering large scales and allocating resources across wide spatial areas whilst a tactic approach is local, immediate and cannot be considered holistic. Whilst strategy approaches, which place restoration in the context of large scale processes, are likely to be the most successful the reality is that restoration is often localised and reactionary (or tactical, Landers, 1997) and therefore fails to reach stated objectives. To combat such failures, effort to understand systems at scales meaningful to species, the process cascades and hydrological connectivity that link landscapes to riverscapes prior to restoration effort is required.

2.9 The role of remote sensing, GIS and modelling tools

Freshwater ecologists are aware that river ecosystems are poorly understood in catchment scale terms due to a paucity of methods and concepts applicable to these large, and connected, scales (Carbonneau *et al.*, 2009). The very nature of exploring large-scale, interconnected, systems requires novel approaches that weave together traditional exploration of systems with modern technological approaches that encompass remote sensing and GIS advances. This approach requires the meshing of fine-resolution, catchment-scale modelling tools with data gathered in the field. In order to do so requires new concepts to be devised that direct how large scale eco-hydrological research is moved forward.

Lane *et al.* (2006) note that diffuse pollution can be re-defined as a series of small point source events. This conceptualises a difficult-to-manage issue into a series of identifiable, and thus more manageable, point-source events. With contemporary modelling techniques providing resolution measured in metres, this definition provides a tool with which to view pollution sources, their pathways, and receiving waters. It brings a landscape scale issue down to a series of smaller problems even though they may be widely distributed across catchments. Once it is understand how these risky connect to riverscapes, management options can be identified. The idea is to explore river networks at the catchment scale whilst at the same time accounting for what occurs at the sub-field scale (Figure 2.11).

Modern techniques can assist in the process of collecting evidence at large scales and fine resolution. By utilising remote sensing, GIS data sets (e.g. digital elevation models, rainfall maps, land cover maps) and modelling techniques, it is possible to map and process landscape features in order to develop knowledge of the risks posed to rivers, link these risk locations to topographically-controlled surface flows and ultimately decipher where pressures emerge and ultimately impact on river biota. This allows concepts to be devised and followed across scales to their concluding effects on rivers. This improves the opportunity that restoration effort will be based on good science and thus improves the efficiency of the targeting of finite resources.

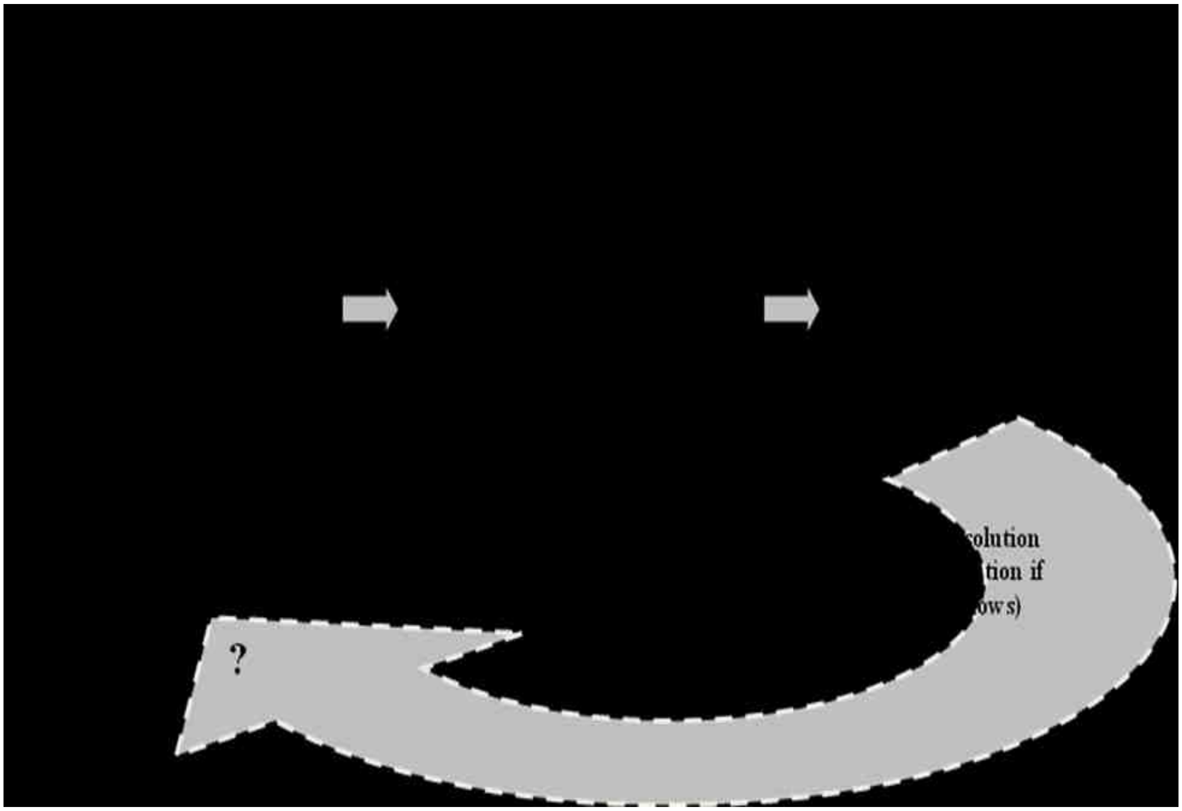


Fig. 2.11: *Catchments can be viewed as a range of land parcels each of which may have a risk probability attached based on landcover and surface flow. When the sum total of these risks is then scaled back to the catchment level it can be decided if indeed they degrade river health and which pose the greatest risk.*

Dugdale *et al.* (2005) record a number of benefits from remote sensing over traditional survey methods including:

1) rapid coverage of large geographic areas;
2) recording of attributes directly to GIS layers saving time;
3) data collection is possible in otherwise inaccessible sites;
4) land cover units can be viewed at a variety of scales;
5) spatially separated reaches can be viewed side by side;
6) a permanent record that can be revisited simply and quickly.

However, they also note a number of pitfalls such as variability in detail, accuracy and image quality that can be exacerbated due to on-ground issues such as shading. Remote sensing is poor at picking out detail of features such as bars and bank protection (unless resolution is high). Despite this, remote sensing and modelling at fine resolution over large geographic areas is becoming increasingly refined and utilised (Lane *et al.*, 2006;

Carbonneau *et al.*, 2009). This allows large areas to be assessed at interconnected scales that matter to river biota. Even so models have tended to concentrate on the small-scale and short-term where validation is most easily advanced (Kirkby, 1995).

Remote sensing and modelling provide the ability to identify which land units within a landscape are most likely to result in diffuse pollution of water courses without requiring on-ground surveys. This replacement of direct observation with remote observation has been termed ‘surveillant science’ (Lane *et al.*, 2006). It provides river managers with the ability to conduct investigations at the catchment-scale thus allowing processes that may limit brown trout to be investigated. Depending on the area being studied, these investigations can be carried out at sub-field scale down to 5 metre resolution. Reaney *et al.* (2007) comment that subtle topographic detail can place major controls on runoff generation and hydrological connectivity at scales <10m, these can now be captured. Heathwaite (2000) termed localised areas that place controls on diffuse pollution Critical Source Areas. The ability to pick these out of the catchment remotely and through modelling is a major advance for river management.

There must be notes of caution, however. Landers (1997) points out that GIS should not be conceptualised as the panacea to river management or restoration ecology and point out that ground truthing is still required in order to: 1) assess the quality of the GIS layers and outputs; and, 2) ascertain on-ground conditions that may have changed since the layers were created. Remote sensing, GIS and modelling techniques should be viewed as approaches that complement knowledge gleaned from direct observation.

2.10 Extending the peer review process

Following Healey (1997), Lane *et al.* (2006) argue that an extension of the peer review process is required in order to tie models such as SCIMAP into the local knowledge base. Healey (1997) terms this form of peer review ‘extended peer communities’ which are formed by people affected by the issue in question but are external to the traditional expert community (Lane *et al.*, 2006). This opens up links between scientists and restoration ecologists with local communities and land managers where the locations of concern exist. However, there are pitfalls to this extended form of interdisciplinarity. For example, models will always be inaccurate in certain circumstances and thus uncertainty, which is explicitly accepted by scientists, may become an armoury for cynicism for those who feel most threatened by modelled descriptions of the world such as those arising from SCIMAP (Lane *et al.*, 2006).

Lane *et al.* (2006) comment that research into diffuse pollution is yet to embrace these new approaches to validation and peer review. Without taking such steps, the gap between researchers and the subjects of research is maintained. And yet these subjects of diffuse pollution research are integral in the restorative process as they hold the property rights to land adjoining the watercourses that are the concern of restoration activity. If these communities are to be included in the process of research and validation they must: 1) have understanding of the scientific process, with all its uncertainties; 2) be included in the full process so as to gain this understanding; and 3) be approached to assist with validation of models and site identification for restoration. In turn, restoration ecologists must ensure they have knowledge of the farming process and the economic uncertainties that are embedded in land management. This two-way communication assists with negotiation and enables river restorers to approach farming communities with realistic expectations.

2.11 Legislative context

As with catchment science, legislation too crosses scales and boundaries. As knowledge of the interconnections between scales and habitats has developed, legislation has also become increasingly sophisticated, scaling up from small, localised protections to viewing species and habitats as set within a larger matrix of connections and impacts.

There has been a long history of water regulation in the UK. The first piece of water legislation passed in England was enacted in 1388 and prevented the dumping of animal waste, dung or litter into rivers. Further legislation followed including the River Pollution Prevention Act 1876 and the River Boards Act 1948. During the 1970s and 80s river management became institutionalised through the setting up of public water bodies. These were followed by the National Rivers Authority (1989) which was restructured to become the Environment Agency (1996) that has the responsibility of regulating river usage, monitoring riverscapes and enforcing pollution legislation (Lane *et al.*, 2008).

A move from national to supranational management of rivers has been apparent with the influence of the EU and international conferences that frame biodiversity as worthy of conservation effort on a global scale. The EU Freshwater Fish Directive (enacted in the UK under the Surface Waters Regulations 1997) identified standards for both salmonid and cyprinid fish (Table 2.2). Along with other EU Directives including the Urban Wastewater Treatment Directive (1991), Surface Water Abstraction Directive (1975), Groundwater Directive (2006) and the Habitats Directive (1992), a shift from national governance to international governance of freshwater habitats has been apparent.

More recently, there has been a major shift in legislative approach to river conservation moving from concentrating on species and their immediate surroundings to the development of holistic approaches that view rivers as firmly set within, and as responses to, catchment scale processes and pressures. The WFD encapsulates this new approach, and Holzarth (2002) argues that the WFD has the potential to encourage catchment-based governance to bring about successful management of water quality. This move up the ecological chain from concentrating on species, habitats, ecosystems to catchments recognises knowledge developed through hydrology and ecology showing how river habitats, and the species therein, are responses to a multitude of factors and processes that cross spatio-temporal scales and connect to factors within and between aquatic and terrestrial systems.

The WFD is based on the premise that water quality depends on what happens within a catchment and thus explicitly recognises that scale and connectivity pose major controls for riverscape habitats (Moss, 2008). Prior to the WFD, legislation viewed rivers as discrete, disconnected habitats (Moss, 2008). The aim of the Directive is to bring all surface and groundwaters throughout member states to “good ecological status” (Moss, 2008; Posen *et al.*, 2009; Saz-Salazar *et al.*, 2009). The WFD provides a legislative context for river restoration that marries advances in scientific knowledge with catchment-scale management of river basins.

Table 2.2: *Parameters and thresholds set for salmonid and cyprinid fish through the EU Freshwater Fish Directive.*

Parameter	Units	Salmonid Standard	Cyprinid Standard	Notes
Temperature	^{o}C	<21.5	<28.0	**Maximum at monitoring site**
	^{o}C	>10.0	>10.0	**Maximum for breeding season**
Dissolved Oxygen	mg/l	50% >9	50% >7	
pH	-	6-9	6-9	
Unionised Ammonia	mg/l	0.025	0.025	
Total Ammonia	**mg/l**	**1.0**	**1.0**	

2.12 Conclusion

Whilst all studies must be reductionist to some extent (due to the myriad of interacting processes in natural systems), many have oversimplified complex systems to the point that findings become difficult to scale up to catchment settings (Lane, 2003, Richter *et al.*, 1997). Developing knowledge regarding the impacts of land management on runoff is a research challenge that, if tackled successfully, will enable predictions of land management effects on river systems. A deeper understanding of the response of individual components of in-stream ecosystems will follow through the coupling of species requirements with factors that push an ecosystem in directions beyond threshold limits. By investigating natural systems using catchment-scale approaches, a variety of nested scales can be explored moving from the catchment down to the riverscape and reach scale following land use impacts from source to the receiving stream network. This concept of connectivity between hydrological reservoirs and pathways coupled with habitat connectivity will be central components of this research. It is through such studies, that explore systems holistically or at least in a less reductionist manner, that knowledge of processes, pathways, and response can be evaluated using multiple scales and connections that have meaning to species of interest.

The nature of ecosystems is rife with intricate connections and relationships played out at scales that stretch through spatial and temporal dimensions. Such complex, or highly complicated, interconnections and interactions, result in systems that are difficult to comprehend. Moreover no two ecosystems are exactly alike. Two adjacent streams can have very different underlying soils, geology, surrounding land use, chemistry, pH and ecology. This level of complexity hampers the transferability of research findings between low-order streams, rivers and catchments. Yet there will be similarities and methods of investigation can be transferable. Through simplifying the inputs and outputs of a system using easily followed models, a system may be understood enough in order to govern management decisions, restoration practice or the effects of changing, or intensifying, land use. Therefore, what is most required is a transferable model that allows a rapid assessment of a river system allowing for its uniqueness and land use types and intensity.

From this review a number of possible impacts on brown trout fry have been noted. These range in scale from localised habitat scale impacts (e.g. siltation of gravel beds) to catchment scale impacts (e.g. altered hydrology from upland drainage). The factors identified for later investigation must therefore take these scale differences into account. The broad categories highlighted in table 2.3 will be explored in relation to brown trout fry.

The next section will explore the methods to be employed in order to produce data on catchment scale processes, and how they may impact on river systems through hydrological connectivity.

Table 2.3: *showing fourteen hypothesis to be explored for this thesis split into catchment scale processes and those acting on smaller scales.*

	Catchment scale processes.		
1/ Diffuse source	A) Dairy farms with strong connectivity to water courses will be responsible for significant nutrient inputs. Frear (2006).	B) Dairy farms with strong connectivity to water courses will be responsible for significant nutrient inputs. Frear (2006).	D) BT populations will be reduced when fine sediment loads are high smothering spawning beds. Lane (2006).
2/ Water Stress	A) The presence of grips on moorland alters hydrology (low summer base flows and flashier conditions). Due to this the freshwater habitat is unable to support good BT populations. Robinson (2006).	B) BT move to spawning streams during October to November as a response to spate and high flow conditions. If flow rates are perturbed spawning behaviour will be disrupted. Frost and Brown (1967).	C) BT are non-viable in streams that run dry in summer conditions. Drying of streams is positively correlated to presence of grips on moorlands. Lane (2006).
3/ Water quality	A) Night sag of DO is prevalent due to algal blooms that flourish when high nutrient loads are emitted into rivers. BT will suffer exacerbated death rates under such conditions. Foulger (2006).	B) Low BT numbers will exist where there is a large upstream area of moorland. This measure is indicative of acid flushes. Frear (2006).	
	Smaller scale processes, not necessarily disconnected from catchment processes.		
4/ Ecology and habitat	A) Where BT recruitment is low or absent in historically good sites habitat conditions will be poor, e.g. no or over shading from riparian vegetation. Shilcock (2006).	B) Algal blooms are positively correlated with low shading and BT will not thrive in such places. Shilcock (2006).	C) Buffer strips and shading along riparian corridors will enhance the available habitat for BT and therefore aid recruitment. Populations of year class 0+ and above will be found in such habitats. Maltby (2006) Frear (2006).
	D) Brown trout populations will be highest where aquatic invertebrate populations typical of an upland river system flourish. Anderson (2006) Chalk (2006).		
	Synergistic impacts arising from connections within and between categories.		
5/ Synergistic effects	A) All the above are working in a synergistic manner in order to restrict BT populations and recruitment and no one factor in isolation is to blame for reduced populations.	B) The river system is undergoing multiple pressures which are individually, and in tandem, reducing BT populations at different spatial locations. Arc populations are thus unable to re-colonise regions of degraded habitat.	

Chapter 3: The upper Ure catchment case study

3.1 Introduction.

This chapter follows chapter two by setting the context of the case study catchment as part of the research. The first sections introduce the concept of locating the research in a case study catchment and the conflicts within and between stakeholder groups that exist within the catchment. After this the geography of the catchment is discussed covering the physical properties of the riverscape and how it connects to forms and patterns of land use. This leads into a discussion of the major subcatchments of the upper Ure system exploring their defining features. Climatic and hydrological data are discussed in terms of upland river systems, climatic trends, climate change and land use. After this brown trout and ecology within the River Ure system are described based on Environment Agency data and anecdotal evidence, this precedes a section covering the institutional framework of the catchment. Finally the rivers trust movement is introduced with a focus on the Yorkshire Dales Rivers Trust that was instrumental in funding this research.

3.2 Initial investigations of the case study catchment

Early investigations of the study catchment involved interviews with numerous stakeholders. These took the form of short unstructured interviews with the aim of developing awareness of any themes that came out of the discussions and any conflicts that were apparent. The interviewees ranged from EA fisheries scientists to local anglers and farmers. The interviewees were selected to include people with a good understanding of farming, angling, the upper Ure catchment, local fisheries and nature conservation in general. These early interviews helped develop skills required for later investigations with the local farming community whilst exploring the farm scale SCIMAP outputs.

The interviews were planned to inform the initial investigation which directed later research. It was considered important to form knowledge on a range of issues and concerns. This was accounted for by inviting interviewees from a range of stakeholder groups to take part. The interviews were informal and took place at locations chosen by the interviewee. This was considered important to ensure that the situations were convenient and allowed the interviewee to approach the process on an equal footing.

All the interviews began with basic questions that gathered information on the interviewee, there interest in the catchment and the condition of the river. Then the questions became more pertinent to the theme of the study. For example each interviewee was asked what they considered to be the major impacts on the river Ure that could be limiting brown trout stocks. The next stage of each interview was less structured. The interviewee was allowed to direct the pace and the issues discussed to allow them to impart what they considered to be the major issues. Importantly, each interviewee was given time to state which sectors they considered to be most damaging to the health of the river network, for example agriculture, Yorkshire Water treatment works, tourism or forestry.

Finally each interviewee was given the opportunity to suggest the direction of the research. For example should the research concentrate on brown trout, macro-invertebrates, farming or water quality sampling? Some took this a stage further by highlighting specific methodologies. Suggestions included utilising growth tiles within the water column to monitor algal growth at different locations, the best methods and locations for electrofishing, how best to sample and identify macro-invertebrates and what to investigate when collecting water samples.

3.2.1 Conflict and tension in the case study catchment

There appeared some clear lines of division within the catchment between vested interest groups. However conflicting opinion was not only present between groups but also within groups revealing that diversity exists in all interests within the catchment. Issues that were discussed included farming, forestry, fish populations, flooding and conservation. The following provides a summary of the interviews and the issues highlighted.

The major theme of the interviews was land use and how this may impact the river system. Most of the interviewees accepted that farming could impact on rivers, however, not all farmers believed this was a serious concern. For example, one interviewee commenting on downstream flooding issues asked the rhetorical question, ‘why should I help people in York?’ through changes in land management on his farm. The argument appeared to be firmly rested on the notion that people in York rarely assist farmers and appeared to display a prejudice towards urban populations. This ignores the concept of social responsibility and that all taxpayers subsidise upland (and lowland) farming through tax revenue directed into the Common Agricultural Policy and stewardship schemes. The single payment, Entry Level Scheme, Upland Entry Level Scheme, Higher Level Scheme and the classic stewardship schemes (e.g. Countryside Stewardship and Environmentally Sensitive Area Scheme) are all paid through tax revenue, the bulk of which is collected from urban populations. For example Agri-environment schemes buffer against the income losses from CAP reform (Acs *et al*, 2010) and ‘*the viability of upland farms often depends on core subsidy support (such as the Single Farm Payment) and on AES payments*,’ (Acs et al, 2009, p.2). These connections did not appear to prevent a general perception that farming was ‘under siege’. From many of the meetings it became apparent that there existed a general sense of suspicion amongst the farming community.

In general the farmers disliked ‘interference’ either from Defra, Natural England or from charities with a perceived environmental stance. Moreover some farmers displayed a strong dislike for the Yorkshire Dales National Park Authority. One even suggested there could be a violent confrontation if ‘they’ ever visited. He was unwilling to accept that the national park had any benefits to him or his family. This was despite members of his family finding employment with the authority. What was interesting was that all the farmers that showed reluctance towards ‘interference’ accepted the single payment and were either in the ELS, ELS/HLS or the earlier classic schemes. There did not appear to be any acceptance of contradiction by these strongly held beliefs and inclusion in stewardship schemes. Whilst these beliefs were prevalent within the catchment they were not universal. One farmer commented that the new stewardship schemes (ELS, UELS and HLS) were, ‘*well managed and a clear improvement on the classic schemes’*. He was very happy with his Natural England advisor and considered there assistance to be an asset. Another farmer from a neighbouring catchment accepted that interference was necessary by arguing that without these payments upland farming would not be viable.

There was a strong tendency to argue that issues of water pollution came from other sectors. Members of the farming community appeared to blame WWTWs, road runoff or forestry. During the initial stages of this study very little logging was taking place. However towards the end of the study period logging had become widespread. A chance encounter with one of the initial farmers interviewed highlighted this new concern in terms of diffuse pollution. He remarked that, '*after it rains the streams can run black.*' He also highlighted that road runoff, especially after the two severe winters, was directed immediately to watercourses and that this '*had to have an impact on the rivers.*'

Conflict was not confined to differences between vested interest groups. There were some clear lines of division within groups too. For example one fisheries scientist from the EA argued strongly that all hydroelectric schemes are either a physical barrier to fish movement or a behavioural barrier. The argument rested on the hydro-acoustic signal that could act to deter migratory fish. Yet the EA position statement reads, '*We support the use of sustainable renewable energy, including hydropower, to help meet UK and Welsh Assembly Government renewable energy and greenhouse gas reduction targets,*' (EA, 2009). The national policy aims to, '*to generate 15 per cent of our energy from a mix of renewable sources by 2020,*' (EA, 2009) highlighting that migratory fish are just one concern amongst many with regards river networks.

A second EA fisheries scientist was adamant that the Hawes Waste Water Treatment Works posed a significant risk to water quality in the upper Ure. This position was reinforced by numerous anglers in the locality including many members of the local Salmon and Trout Association who clearly believed this treatment works to be a major impact on water quality and thus fish stocks. Discussions with several farmers also resulted in this treatment works being highlighted as a risk. However amongst the farmers there was a clear reluctance to accept farming as a pollution concern. In contrast the EA farm advice officer disagreed with the position of his colleague (and the general position of the farming community) arguing that land use was the major concern within the catchment in terms of risk to the river network. He argued that the Waste Water

Treatment Works all worked within discharge permits and that pollution from farming was more pernicious and widespread.

However, on visiting Askrigg WWTW with the same Farm Advice Officer it did appear that the one of the two settlement tanks was not working correctly. Moreover eutrophication monitoring carried out by the EA at the request of the Ure Initiative group showed a wide range in the amount of orthophosphate in the final effluent of the Hawes WWTW (0.02 mg/l to 12.1 mg/l taken at different times on 11/05/2006). This water sampling was carried out on only a few occasions and the in river concentration downstream of the discharge pipe taken at the same times/date showed orthophosphate below 0.01mg/l and so within safe limits to salmonids. It was not surprising that a Yorkshire Water representative argued that without sufficient evidence the Hawes WWTW would not be upgraded to further strip phosphates from the final effluent. It was even less surprising to learn that YW would not offer financial support to assist the required monitoring.

In support of the argument that land use is the major concern within the catchment a number of farmers suggested that grips resulted in changes in flow dynamics with increasingly rapid spate events and sediment delivery to streams. It was interesting to note that the culprits here were perceived to be grouse moor owners and not the farming community. One gamekeeper from outside the catchment appeared to support this stance by stating that thousands of tones of sediment have been lost from moorlands due to gripping and as a result grouse broods often became trapped in the deeply eroded channels and subsequently died. To further highlight that there is rarely a full consensus one farmer argued that grips had improved his land and he would be reluctant to block them as he was now able to access the land with heavy machinery due to gripping. Another farmer argued with regards grip blocking, '*I wouldn't do it unless it was a 110% grant, what's in it for me otherwise?*' A member of ADAS (a farm and environment consultancy) staff commented that flow data in Swaledale (the neighbouring dale to the case study catchment) post grip blocking suggested that this restoration method had no impact on flow rates. He argued that, '*intensity of rainfall is changing due to climate change*,' and therefore it was hard to tease out the significant relationships.

The same member of ADAS staff suggested that there are numerous other issues that lack data. He highlighted septic tanks from isolated farms and campsites as a probable issue. Yorkshire Water was contacted in order to ascertain if they have any data on locations of septic tanks. Unfortunately they did not have this data available. A YDRT trustee confirmed this concern by suggesting that many properties and hamlets were situated on soakaways. Thornton Rust, on the south side of the catchment, was one example. Raw sewage could be identified here during periods of low flow that allowed waste to build up. A number of properties have been identified as having raw sewage leaching direct into the river network. One such location is utilised by white clawed crayfish at a location that appeared to lack suitable habitat; the raw sewage perhaps offering a viable food source to the species.

Numerous locations within the catchment appear to pose high risk of diffuse pollution. Many of these arise due to farming practice. One interviewee suggested, '*certain fields are easy to spread on and will have received massive loadings of slurry.'* This suggests that some fields are primed to provide greater risk than others. In terms of farming practice the fields that will receive the highest nutrient loadings will be the inbye meadows and pastures. Whilst these are not on the steepest land (which generates rapid runoff) they are located closest to the river network. The extent of damage on grasslands (the predominant land use in the catchment) depends on, '*rainfall, existing soil condition and the timing and density of grazing,*' (Cranfield University, 2002).

These early interviews provided a steer for the research and offered an insight into the social and economic concerns of the catchment, which in turn transferred risk to river networks. The conflicting views of stakeholder groups was interesting, though not always surprising. The interviews were generally led away from the interviewees sector in order to concentrate on issues that may arise from other interests within the catchment. The interviewees that offered a more holistic view were generally from those without a direct economic interest in the catchment, though even within these groups different opinions were apparent.

3.2.2 A photographic journey through the upper Ure catchment

In order to highlight some of the land use issues within the catchment a walk over survey was carried out. This involved walking the main river stem and a number of subcatchments to identify the possible issues following on from the interviews. The following images (figures 3.1 to3.11) highlight some of the possible sources of risk and the instream impacts that may arise from land use.

Figure 3.1: *The River Ure has a number of places where algal blooms can become severe during summer low flow periods. These include Yorebridge and Worton (pictured). Algal blooms are exacerbated by excessive nutrients (some of which will be sediment attached phosphates) and a lack of riparian trees to offer shading.*

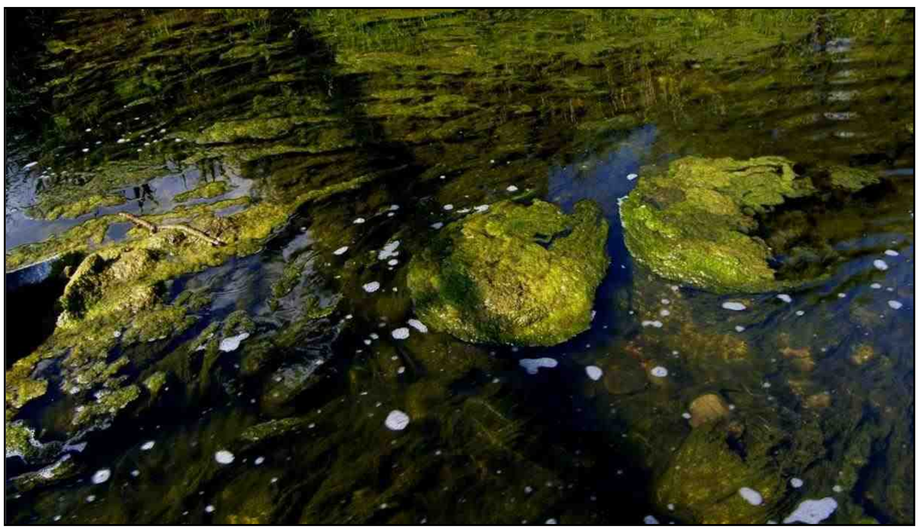


Figure 3.2: *Land management practice such as supplementary feeding can prime soils to erosion and enhance surface runoff. The practice of using supplementary feeders is widespread and poaching a common occurrence. Less risky practices include spreading feed across a wider area.*

Figure 3.3: *Small order streams that are open to livestock can deliver large amounts of sediment to the main river stem. Here cattle and sheep access has severely damaged a first order stream with the sediments being delivered to the main river stem during high flows.*

Figure 3.4: *Stock access to river banks increases the propensity for erosion as seen here on the inside of a meander near Hawes.*

Figure 3.5: *Where stock are excluded the riparian and emergent vegetation flourishes and erosion is less obvious. Here the vegetation allows the river to narrow and so replicates a more natural situation.*

Figure 3.6: *Different management techniques impacting the same stream highlight how changes in close vicinity can degrade or enhance environmental quality. In these images sheep have access to a stream (first two images) and where livestock are excluded in the adjacent field of the same farm (right.) the stream condition is clearly enhanced.*

Figure 3.7: *The top left image shows a location where stock are practically excluded from the stream whilst the top right image taken in the next field downstream shows an erosion nick where sheep have ready access. The bottom image (two further fields down) shows where constant trampling by dairy cattle can severely degrade soils and stream condition. Again the three images are taken on the same farm.*

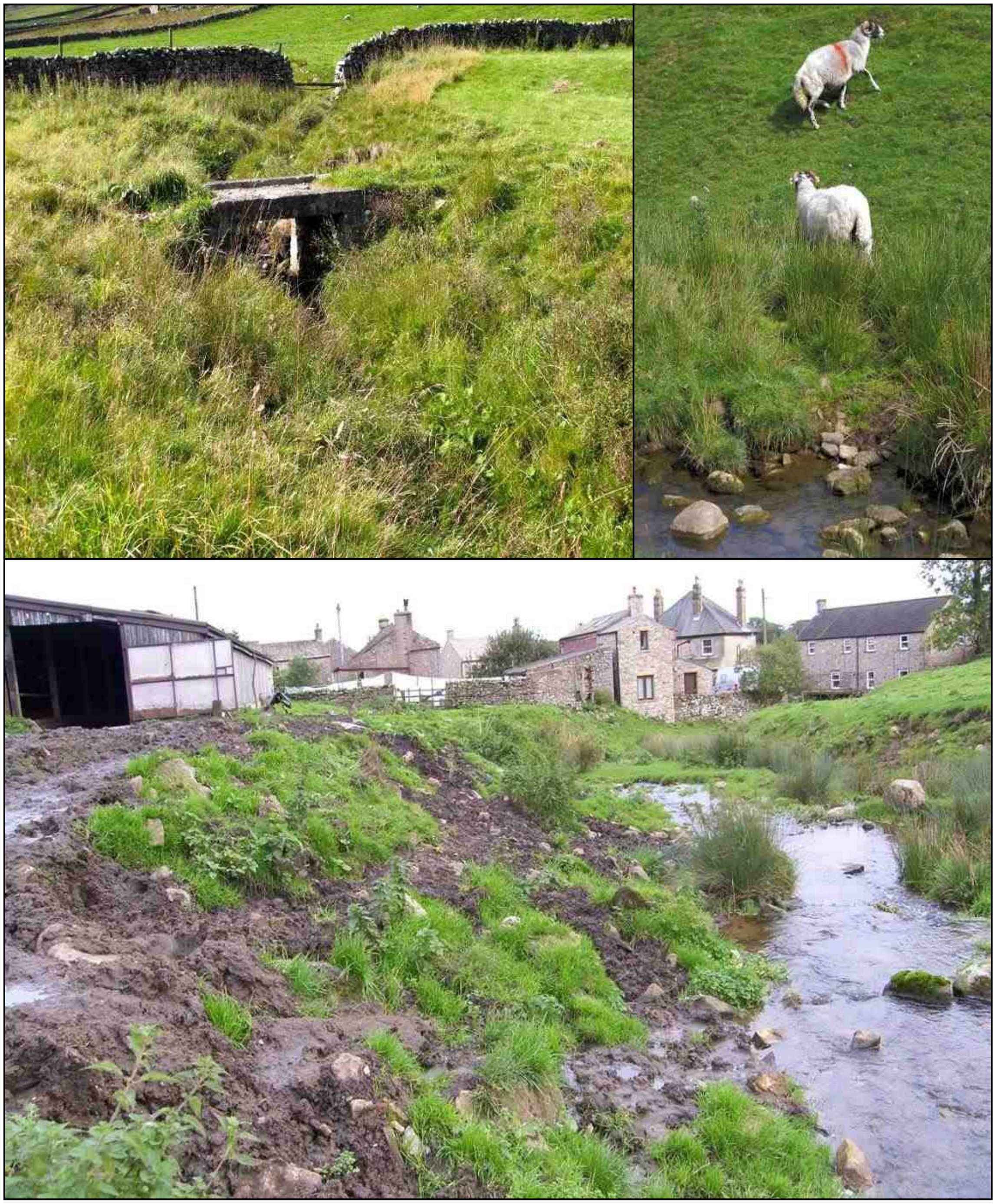


Figure 3.8: *Another example of how management on the same farm can prime soils for erosion. The top image shows where the stream is only open to sheep for part of the year whereas in the next field downstream (three bottom images) dairy cattle have ready access to the stream resulting in enhanced erosion and poaching.*

Figure 3.9: *Here sheep have ready access to the stream bank showing how stock access can severely degrade the river bank. The erosion is occurring on the inside of a meander. The opposite bank has no stock access and shows no sign of erosion even though it is the outside of the meander where the river has greater erosive power.*

Figure 3.10: *These images show how riparian management makes the difference between an eroding and intact bank. Where willows are allowed to grow the bank is stable, immediately downstream where the willow cover expires the bank begins to erode severely and this continues downstream for some distance.*

Figure 3.11: *Where erosion occurs farmers often try to shore up the bank using rubble. This is unsightly and only copes with the symptoms. Fencing out the stream and allowing willows and other tress to grow would reduce erosion and result in less land lost to the river.*

3.3 Justification of the case study approach

In order to research multivariate processes that impinge or enhance salmonid species, it is important to locate investigations within a catchment containing stocks possessing spatially variable recruitment, thus suggesting a number of limiting factors at work. Such variation of recruitment allows multiple factors to be investigated at appropriate scales including land use, in-stream habitat, water quality, fine sediment delivery, shading and flow rates. The scales range from river reach up to the catchment scale all of which are intrinsically linked by physical and hydrological processes, nutrient transfers and habitat connectivity. These factors have been identified as limiting salmonid populations (Bagliniere and Maisse, 1991; Elliot, 1994; Armstrong 2003). However, overly reductionist studies fail to highlight how these factors interact or which are the most important at any given spatial location. Importantly, by concentrating on single pressures subsequent management changes that reduce the pressure may not always result in population recovery if underlying pressures remain important.

By employing a case study approach a number of abstract concepts can be applied and tested in the field. Whereas laboratory experiments fail to account for the holistic nature of ecosystems that come with interacting, competing and symbiotic physical and ecological conditions a catchment-scale investigation is able to account for these linkages. Shader-Frechette and McCoy (2004) identified a number of reasons why a case study approach is preferable to laboratory based experiments. These include,

- No two ecosystems are exactly alike.
- General ecological theory is unable to account for all locations and interactions.
- It provides the ability to examine ecological relationships in real world situations.
- Such studies allow multi-factoral approaches at the relevant interacting scales.

Thus a case study approach offers a number of advantages to traditional experimental approaches, which hold all variables constant except for the two under investigation. In contrast a case study allows multiple pressures to be explored within the environment where they interact and the organisms on which they impact. This can provide a clearer understanding of which factors matter, where they matter and thus offer potential for identifying the most appropriate restoration methods, at the location and scale at which they operate. There are, however, a number of disadvantages. The complexity of natural settings is the most obvious. Adopting a case study approach may result in levels of complexity that are simply too difficult to decipher. The possibility of such an outcome reduces the attraction of a case study approach.

This study, whilst aiming to decipher limiting factors on brown trout (*Salmo tuttra*) as part of a generic investigation, is set within the upper Ure catchment, North Yorkshire, UK, (Figure 3.12). The study site runs from Aysgarth Falls up to the headwaters of the Ure. Aysgarth falls is considered a barrier to upstream migration of Salmon (*Salmo salar*) and the anadromous strain of brown trout (sea trout: *Salmo tuttra*). The catchment covers an area of 234km^{-2}, has a perimeter of 81km and an altitude range of 555m (lowest: 153m, highest: 708.3). Figure 3.13 shows an elevation model of the catchment.

The catchment is predominantly rural with agriculture and tourism the dominant industries. Land use is diverse including sheep, dairy, forestry, grouse moors and conservation. There is no arable within the case study area but crops of silage are commonly harvested for overwintering livestock. Figure 3.14 shows a land cover map of the catchment. The landcover map used is the CEH Landcover Map 2000, whilst this map may become outdated in some catchments here land use change is slow and a transect survey of the catchment showed an accurate fit with the data provided. Thus, the 2000 data was considered to offer an accurate representation of the landcover. It is interesting to note that since the data collection period a number of mature coniferous plantations were felled and anecdotal evidence offered suggested that this increased erosion and fine sediment delivery rates.

The presence of the Wensleydale Creamery at Hawes has ensured that dairy farming remains viable within the area. The upper dale has a population of 2602 (2001 census). Population density is therefore low at approximately 0.12 people/Ha with the majority located in the towns and villages of Hawes, Bainbridge, Askrigg and Aysgarth. Hawes is the main town of the upper dale and has a population of 1115 and in combination with Gayle (an adjoining village) and an adjacent caravan park covers an area of 0.33Km^{-2}. It has been estimated (Neale, 2008, pers. Comm.) that during the main tourist season the population of upper Wensleydale can double in number.

Figure 3.12: *the upper Ure catchment is located in the Yorkshire Dales National Park, North Yorkshire, within the Humber Basin. The Humber basin drain a fifth of the England covering a large geographical area. The main river stem of the upper Ure catchment has a length of approximately 32km*

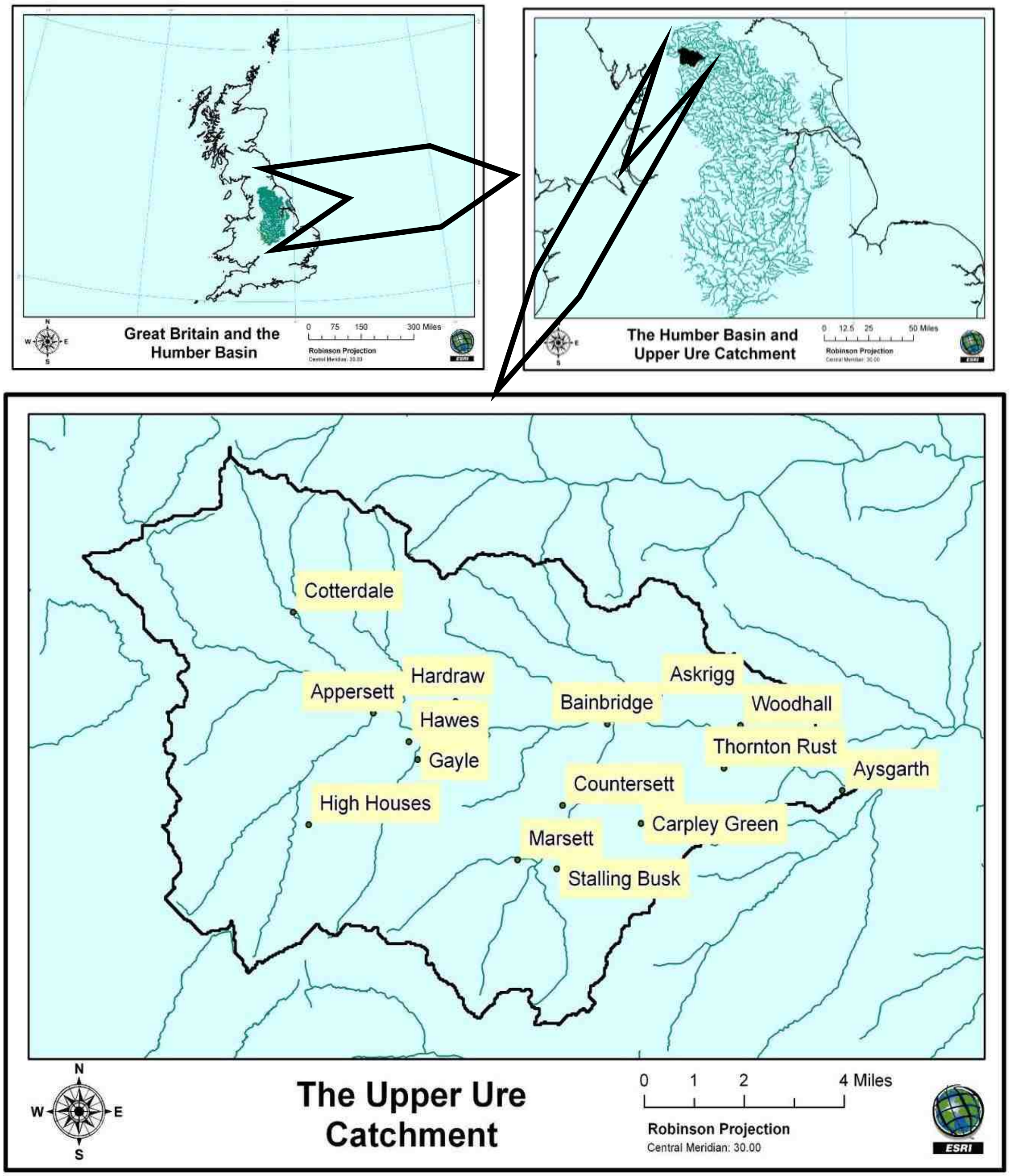


Figure 3.13: An e*levation map of the upper Ure catchment.*

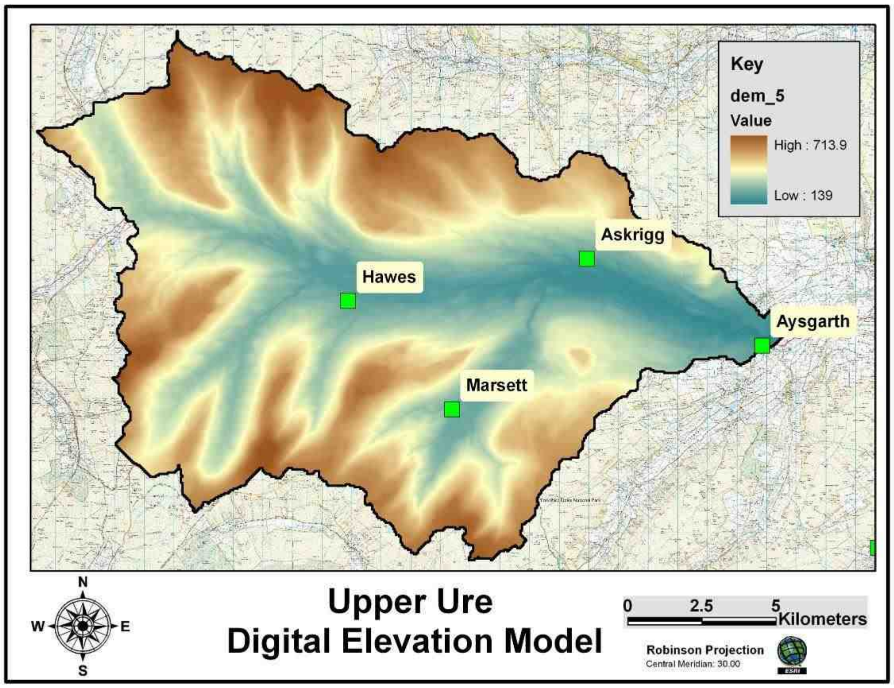
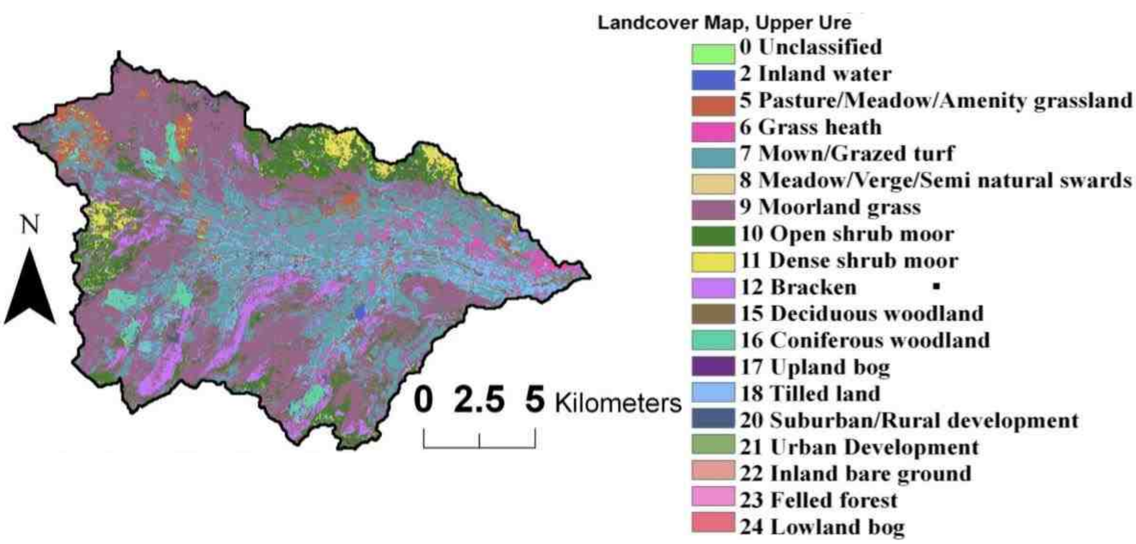


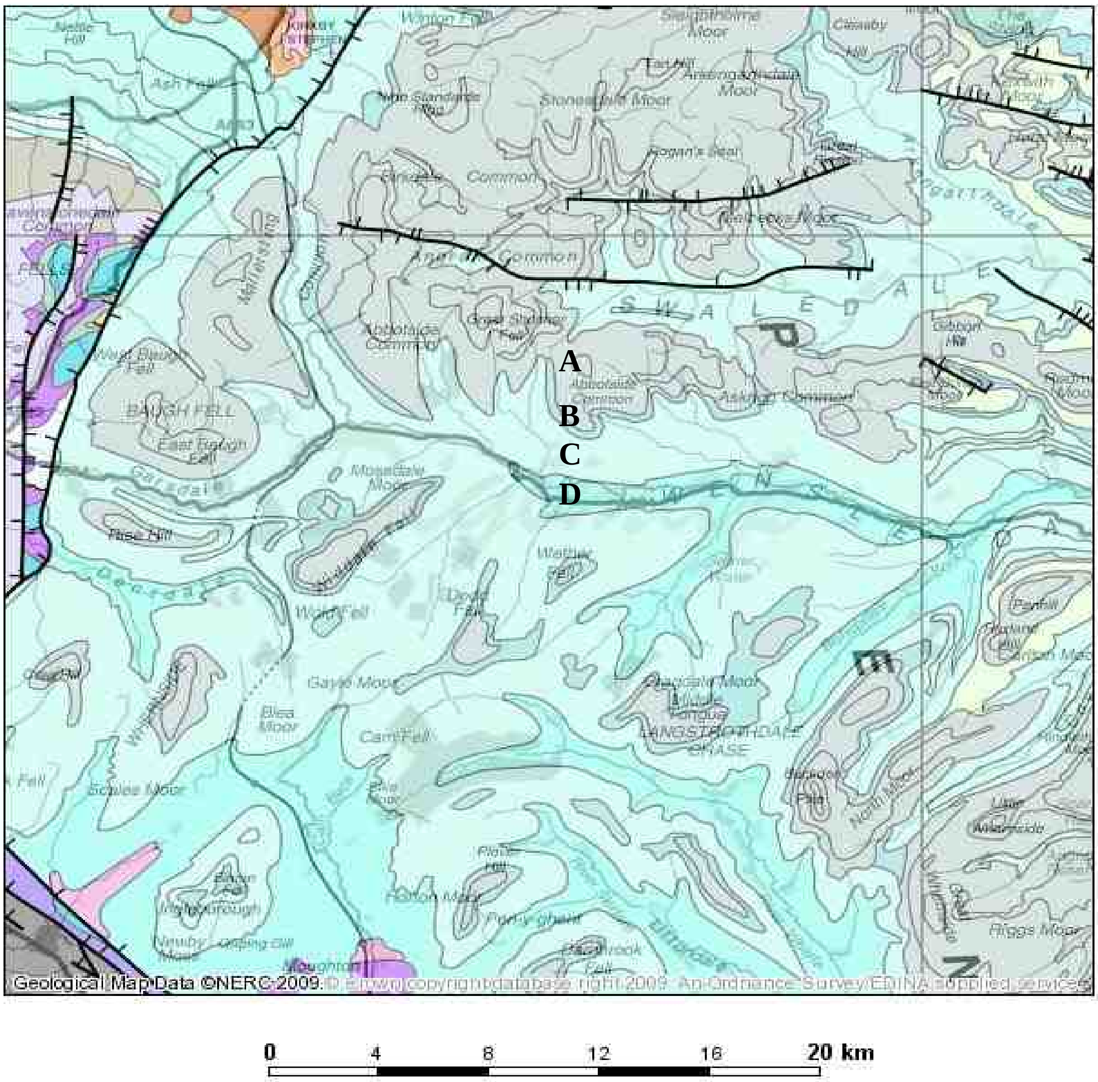
Figure 3.14: A *land cover map of the upper Ure, showing the major land cover types within the catchment.* (CEH, 2000)



The catchment is set within the Yorkshire Dales National Park which itself is set within the Pennine chain. The rock types are Carboniferous and composed of various limestone over-capped with sandstones and millstones (Figure 3.15). The lower elevations are composed of Great and Great Scar Limestone. The highest elevations are composed of the Stainmore foundation which is a series of sandstones, siltstones, mudstones, thin limestones and coals. All of these are within the Yoredale series. The landscape has been shaped by the Devensian glacial period (73000 to 10000 years B.P.). The dale is a broad U-shaped valley and is the widest dale within the National Park.

Whilst the physical geology was formed during the Carboniferous period (354 to 290 million years B.P.) the drift geology has been a more recent affair. The soils have developed since the ice retreated. The upper reaches of the dale are composed of peat soils of various depths underlain by clays. Farage *et al* (2009) suggest that these soils hold between 600 to 1325 g carbon m^2. The soils of the lower reaches are more varied and include loams and river alluvium with deposits of gravel and a number of glacial drumlins. Soil depth is lowest on the slopes and highest on the floodplains and in the upland peat horizons. Peat soils and moorland cover a large area within the catchment (Figure 3.16). The pH of the watercourses is generally neutral or above due to the buffering effects of the Limestone geology. However, under conditions of rapid rainfall, the pH can dip as acid flushes derived from peat horizons race through the catchment.

Figure 3.15: *The geology of the catchment.*



This map is drawn on the GB National Grid.

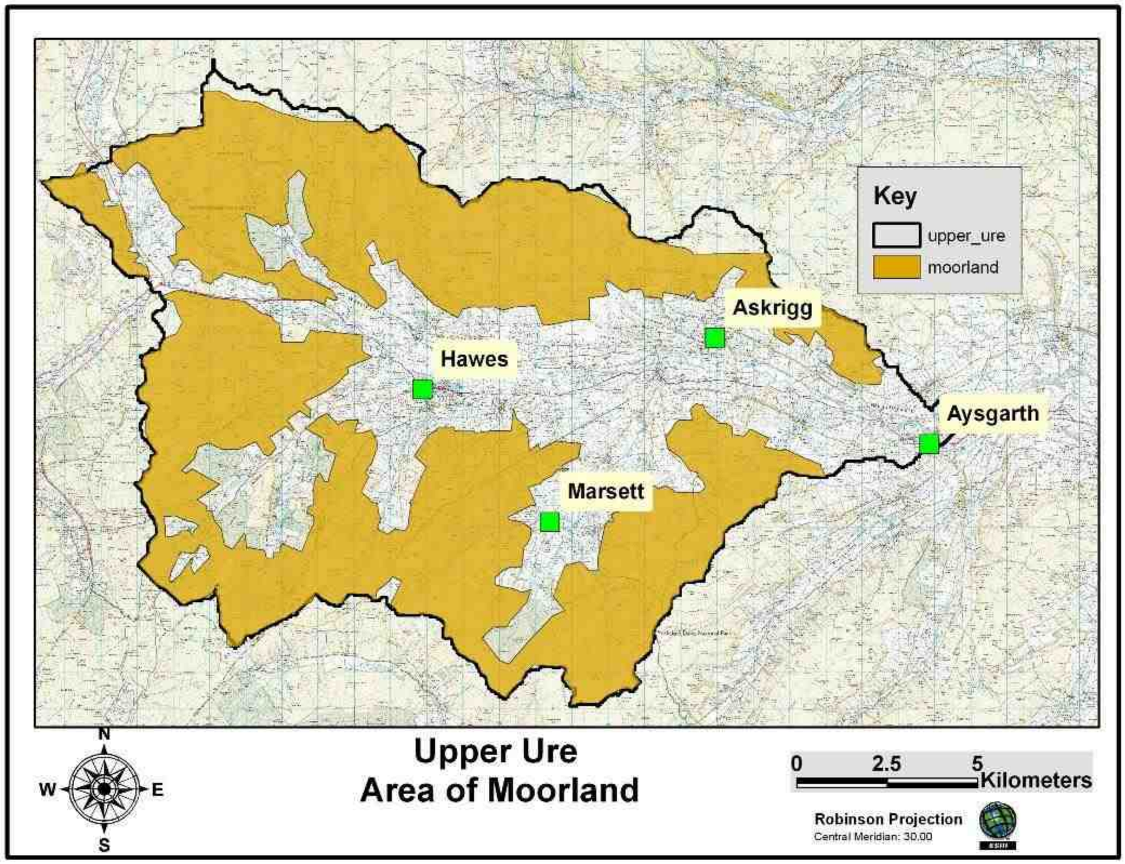
Heights (if given) are in metres above Newlyn datum.
The representation of a road, track or path is no evidence of a right of way.
The alignment of tunnels is approximate.

Reproduced using significant survey information from Ordnance Survey basic and derived scales digital data with the permission of the controller of Her Majesty's Stationery Office.

Produced at : 22:59:28 o'clock BST on 09/10/09.
Produced for : David Ian Higgins.

Key:
A Stainmore Foundation
B Great Limestone Member
C Alston Formation
D Great Scar Limestone Group

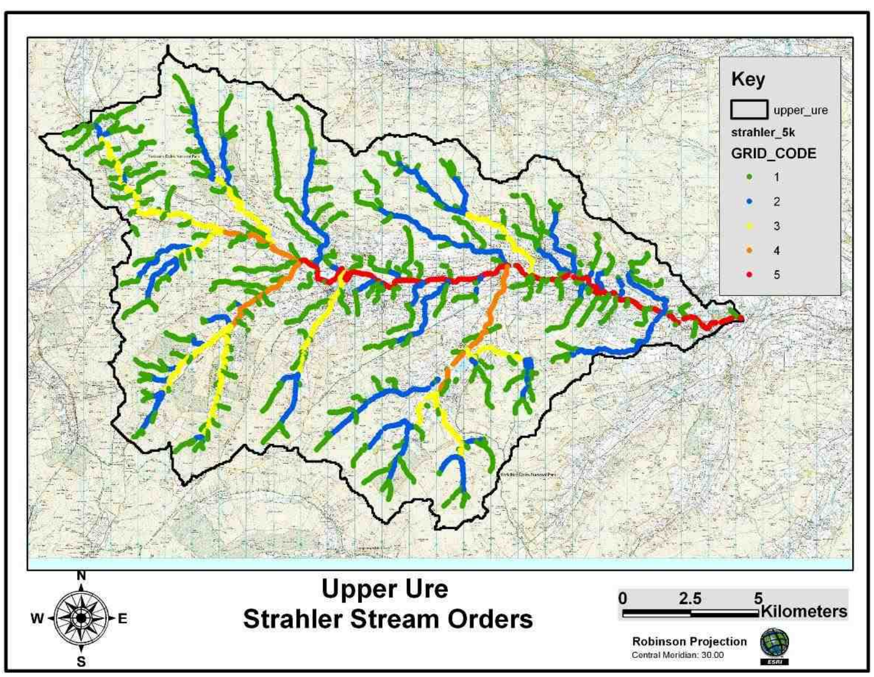
Figure 3.16: *the area of moorland within the upper Ure catchment covers a large proportion of the area (59% as opposed to 8% of a typical catchment).*



The river Ure flows from West to East and enters the tidal River Ouse upstream of York before reaching the North Sea at the Humber Estuary. It has been estimated that the Ure contributes 15% of the suspended sediment load to the Ouse (Walling *et al*, 1999). The headwaters rise near Lunds fell at the same watershed as the west-flowing river Eden. Like many upland rivers, the Ure is flashy with rapid spate events and low summer base flows. The ecology has evolved to cope with these extremes with the upper reaches holding high proportions of Plecoptera (stonefly), Trichoptera (caddis fly) and Ephemeroptera (mayfly, in particular Heptageniidae) with an increase in Gammarus (freshwater shrimp) and Simuliidae (black fly) species as the velocity reduces and the trophic status increases on the lower slopes. Apart from brown trout (the most important fishery species of the upper Ure) the upland sections of the river Ure hold grayling, bullhead, minnow, stone loach and eels. Bream, roach, perch and pike have been introduced into Semerwater, the only natural lake in the dale and the largest glacial lake in Yorkshire.

Below Appersett, the floodplain becomes wide and flat and contains an important area of wet meadow. There are a number of levées along this section which are regularly 'patched' by farmers. The main river has several sub-catchments the largest of which is Raydale the location of Semerwater. There are 5 Strahler stream orders on the upper Ure with the majority of the catchment being stream orders 1 and 2, (Figure 3.17). A number of natural barriers exist including Hardraw force and Mill Gill waterfalls, (Figure 3.18). Many of these are large and impassable to migratory fish. Several anthropogenic barriers exist within the catchment, though these are less of a barrier then the natural features, they include small weirs, fords and culverts (see figure 3.17). The sub-catchments are dynamic gravel bed stretches composed of tight v-shaped valleys with interlocking spurs and rapid responses to rainfall. The main river whilst dynamic is a more meandering affair and below Appersett and Hawes has gravel bars building up in zones of deposition and eroding banks particularly, but not exclusively, on the outer bends of the meanders.

Figure 3.17: *Strahler stream orders ranging from 1 to 5, in the case study catchment.*



The upper Ure is an upland oligotrophic stretch with issues of eutrophication arising from land management and Waste Water Treatment Works of which there are 6 in the catchment, the largest being at Hawes. Many of the houses and villages are 'off-grid' and require septic tanks. Data on the number or the impacts arising from poor maintenance of these is lacking. The Environment Agency has assessed most of the upper Ure to be of either good or moderate ecological standard (Figure 3.18). Important sub-catchments and streams are described below and basic data for each is shown in Table 3.1. Figure 3.19 shows these areas.

Figure 3.18: *The components used to assess the overall status for surface water bodies. Under the WFD a failure in any one aspect of these criteria will result in the river section being rated as less than good ecological status* (EA, 2009).

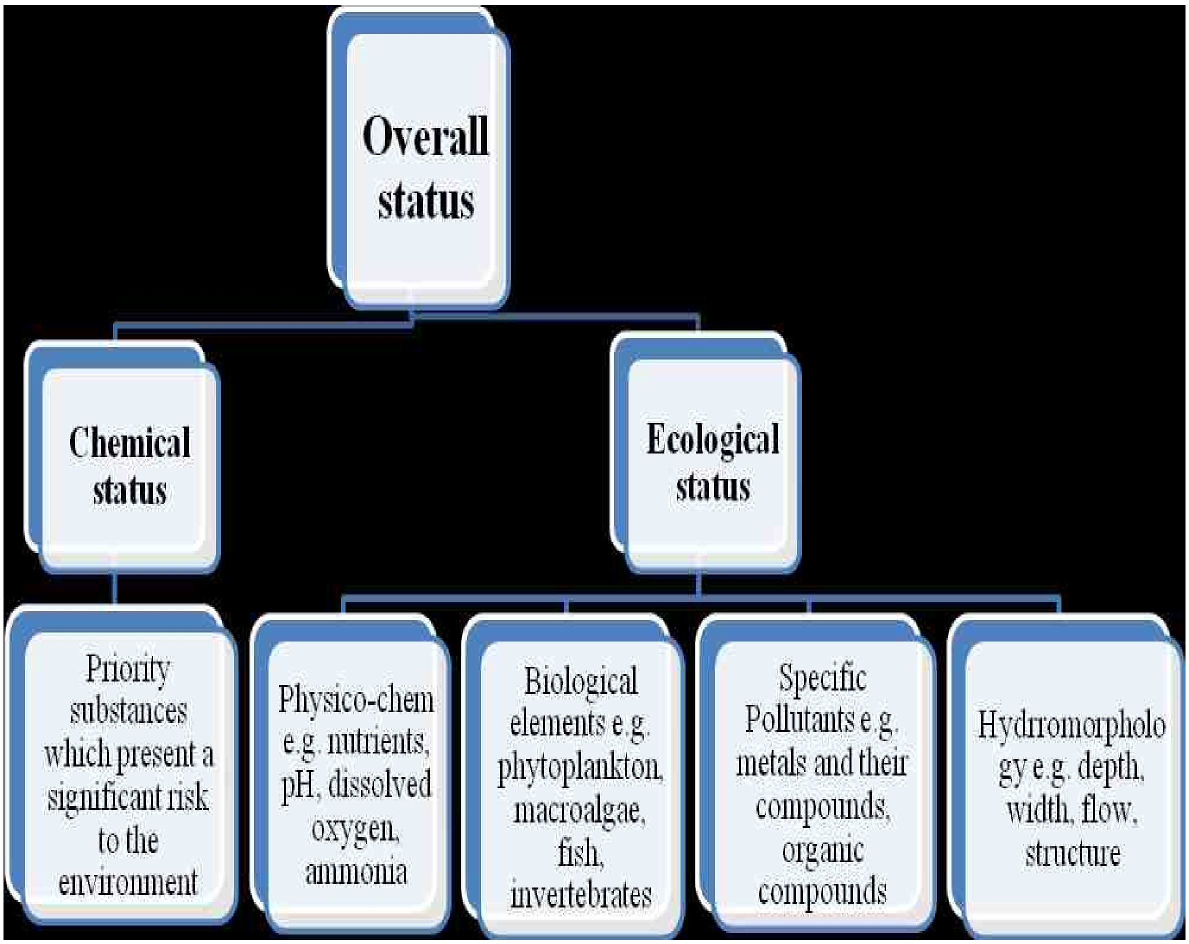
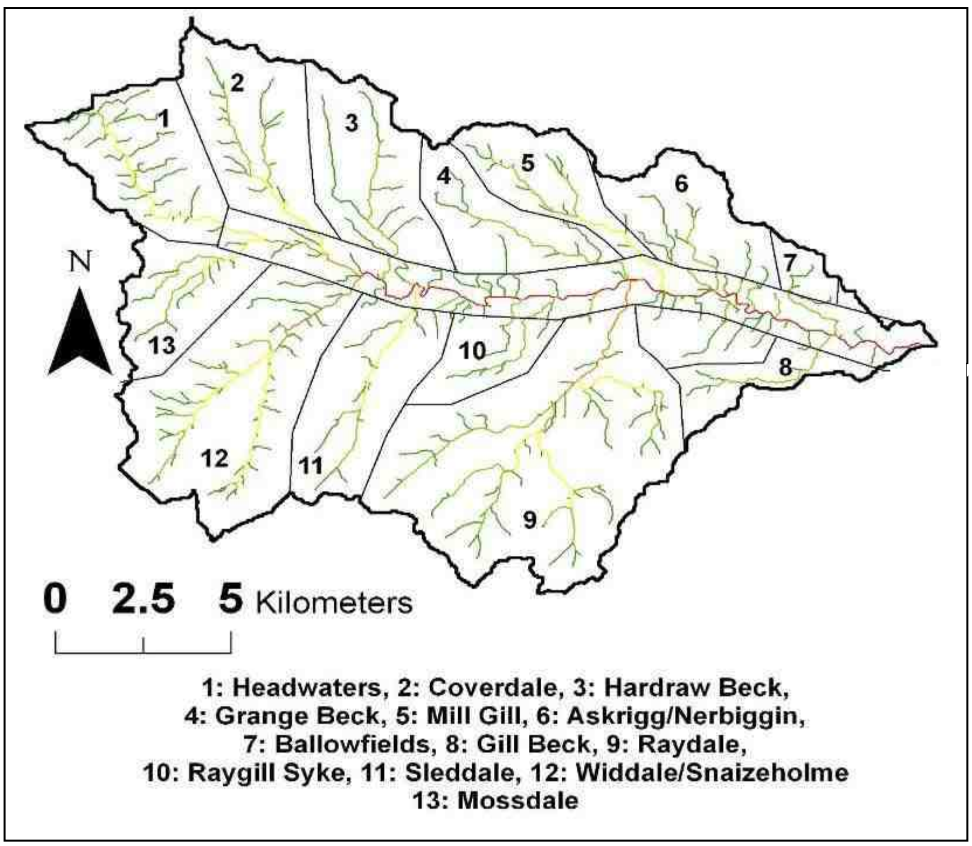


Table 3.1: *Basic data on area and altitude for the upper Ure catchment and its tributary catchments.*

Sub-catchment	Area Km^2	High altitude (m)	Low altitude (m)	Altitude range (m)
Upper Ure	234	708	153	555
Ure headwaters	20	666	263	403
Mossdale	11	646	262	384
Cotterdale	18.9	699	238	461
Widdale and Snaizeholme	36	667	228	439
Hardraw Beck (Fossdale)	17	706	226	480
Sleddale	17	668	226	442
Raygill Syke	6	598	215	383
Grange beck	11	665	204	461
Raydale	51	638	203	435
Mill Gill	15	658	198	460
Ballowfields	5	487	192	295
Gill Beck	7	495	189	306

Figure 3.19: *The locations of the sub-catchments of the upper Ure. The larger areas will be discussed below.*



3.4 The subcatchments of the upper Ure.

The main dale is composed of several subcatchments all of which differ in form and land management. Some of these differences are subtle, others are obvious. For example Raydale has a high proportion of coniferous plantations, grips and the only natural lake within the main catchment whereas Snaizeholme is ungripped, contains an artificial lake and a high proportion of native deciduous woodland. Other differences between the subcatchments include basin size and shape, extent of moorland, predominant land use and hydrological response. The following sections describe these subcatchments.

3.4.1 Ure headwaters.

This section of the catchment is composed of open and extensive rough grass and rush pastures running up to open moorland on the higher ground. Some more intensive fields provide two or three crops of silage each summer. The river here runs in the tight Mallerstang valley and is a dynamic gravel-bed stretch with numerous cascades and small waterfalls. The moorland is largely intact with very little drainage in comparison to other sections of the catchment. Land management is extensive grazing of sheep and beef running into open grouse moors on the higher elevations. Stream orders range from 1 to 3. Sections of forestry adjoin the river at Lunds. A number of these plantations have recently been felled. This area has been assessed to be of good ecological standard (EA, 2009).

3.4.2 Mossdale.

Mossdale is a small, narrow, sub-catchment with an area of floodplain close to the confluence with the main Ure where the grass fields are intensively managed. The dale rapidly opens up into wide areas of moorland on the upper reaches. Stream orders range from 1 to 3. Grazing is sheep and beef with grouse moor management on the upper sections. Large areas of the peat moor have been intensively drained with open channels (grips) during the 1970s and 80s. A recent landslip degraded this reach of river through the delivery of large quantities of fine sediments. It has been reported that the river ran discoloured down to Ripon when this occurred (Morland, 2006, *pers comm*.). Two large waterfalls pose impassable barriers to upstream migration of brown trout. Just above the confluence with the Ure a weir and culvert also pose a substantial, though lesser, barrier. This area has been assessed to be of good ecological standard (EA, 2009).

3.4.3 Cotterdale.

This catchment is well known as a shooting estate for grouse and pheasant and is managed with this as the primary aim. The lower reaches support sheep and beef and the fields here are intensively grazed. Stream orders increase to 3 and the streams fork into two gills that stretch into the moorland. There are numerous natural barriers on this stretch that all pose substantial barriers to upstream migration. Forestry above the village of Cotterdale covers a large portion of the catchment and some logging has taken place in recent years. There are a moderate number of grips in the catchment and this may have been compounded by a heather regeneration project. This involved extensive cutting of drains akin to grips but closed at each end and following contours so not directly connected to watercourses. This area has been assessed to be of good ecological standard (EA, 2009).

3.4.4 Widdale and Snaizeholme.

These two streams drain the second largest sub-catchment in the dale. The Environment Agency has long associated Widdale (Frear, 2007) with poor water quality though Snaizeholme, a small side catchment to Widdale, is less degraded. The lower reaches of Widdale Beck has been assessed to be of moderate ecological standard and the upper reaches, including Snaizeholme, have been graded as good ecological status (EA, 2009). Widdale is a long thin catchment with sheep, beef and conifer plantations as the dominant land uses. At the top of the catchment there are a number of low-density gripped areas. Widdale Beck is very flashy and joins the Ure at Appersett adding to regular overtopping of the bank along the floodplain. Snaizeholme is a cauldron-shaped valley which lacks moorland grips and contains a large area of conifer plantations along with the highest density of native woodland in the upper Ure catchment. Here land use is less intensive and includes beef and sheep. Snaizeholme Beck is the only stream with a flow gauge within the case study area. The gauge is recorded as an anthropogenic barrier to migration. However in high flows brown trout would navigate the gauge with little effort. A waterfall on Widdale Beck would also be navigable in high flows. Stream order increases to 5 at the lower reach, reflected in the relatively high discharge from this catchment.

3.4.5 Hardraw Beck.

This stream drains a long thin catchment that stretches up towards Great Shunner Fell the largest area of land above 600 m in the National Park. At the lower end, the stream falls over Hardraw Force the largest single drop waterfall in England and an obvious barrier to upstream migration which cuts the upper reaches of this sub-catchment off from brown trout populations in the main river. An area of mature semi-natural woodland surrounds the falls; the wood contains indicator species including bluebells and ransoms suggesting the woodland may be ancient. Above the falls a large caravan site is situated next to the stream. Further upstream, two more waterfalls add further barriers to trout movement compressing the available habitat of this stream to small resident stocks of brown trout. Land use is extensive with a large proportion of moorland given over to grouse and grazing. Despite the area of moorland there are few grips. Below Hardraw Force is the small village of Hardraw which has its own Waste Water Treatment Works. Hardraw Beck only reaches a 2nd order stream. This area has been assessed to be of good ecological standard (EA, 2009).

3.4.6 Sleddale.

This is another long thin sub-catchment that stretches from Hawes up towards the Ure/Wharfe watershed at Cam Lane. A number of waterfalls limit upstream migration. Land use is intensive with a high proportion of dairy farms. Poaching of the land is apparent in areas due to high stocking rates and high rainfall. This is most apparent in areas without fencing along the stream banks. Sheep and beef are also prevalent land uses. Whilst this stream stretches up towards extensive areas of moorland, gripping is virtually absent. The Beck becomes a 3rd order stream and flows through Gayle and Hawes discharging into the river Ure just upstream of the Waste Water Treatment Works' effluent discharge pipe. There are fewer plantations in this area but several small wooded areas and shelter belts dot the landscape and the riparian zone has strips of alder in the lower reaches. This area has been assessed to be of good ecological standard (EA, 2009).

3.4.7 Raygill Syke.

This is one of the smaller streams of the dale. This Beck barely stretches to the moorland line and flows over intensive farmland composed of dairy, beef and sheep production. In dry periods it dries along large reaches due to a number of sink holes. This stream suffers severe poaching and algal blooms exacerbated by low base flows and the whole system is obviously suffering with eutrophication. At the lower end a culvert provides a barrier to brown trout movement. This is a 2nd order stream at the confluence with the River Ure. This area has been assessed to be of moderate ecological standard (EA, 2009) and is obviously impacted by nutrient inputs.

3.4.8 Grange Beck.

Another narrow system that stretches up towards heavily gripped areas of moorland at Low Abbotside. An impassable waterfall delineates a gravel bed zone from the lower reaches that run over bed rock before discharging into the main Ure as a 2nd order stream. There are two further waterfalls and a weir upstream. There is little in the way of woodland with only short strips at the lower end of the catchment. The beck runs through a number of dairy and sheep enterprises. This area has been assessed to be in moderate ecological standard (EA, 2009).

3.4.9 Raydale.

This is the largest sub-catchment of upper Wensleydale and is in many ways a microcosm of the main dale. This area has all the land uses and pressures of the main catchment including a high number of dairy farms, sheep and beef, forestry and moorland. The top reaches of the catchment contain extensive and heavily gripped shooting estates that stretch along the Ure/Wharfe watershed. Three streams (Bardale, Raydale and Cragdale) drain the moors and these flow into Crooks Beck before entering Semerwater Lake. At the lake outlet, the locally reputed shortest river in England (the River Bain) flows down to the village of Bainbridge before joining the River Ure downstream of Bainbridge Waste Water Treatment Works.

Fleet Moss at the top end of the catchment is heavily hagged[25] and eroding and is considered to be the most degraded area of peat soil in the catchment and probably the national park. Semerwater suffers from eutrophication though evidence suggests that this is due to re-suspension of sediments rather than nutrient leaching from agricultural land. Re-suspension of sediments mixes nutrients into the water column that can become biologically available, in particular during anoxic conditions when the dissolution of sediment attached phosphates is enhanced (Mhamdi *et al*, 2009). Extensive wet meadows above the lake are designated as a Site of Special Scientific Interest. These locations have been assessed as in unfavourable condition by Natural England. Due to this a Catchment Sensitive Farming partnership has recently begun in conjunction with the Yorkshire Dales National Park Authority, Natural England and the Yorkshire Dales Rivers Trust.

A number of waterfalls make the top reaches of the catchment inaccessible to brown trout migration, although the majority of the dale, including good spawning sites, is open to brown trout stocks including those present in the main river Ure. It is likely that a number of this species use Semerwater for the adult stage of their life cycle and whilst it is unlikely that they reach the status of Ferox trout, it should be expected that some large individual fish are found in the lake. The Dale reaches a 4^{th} order system with Semerwater expanding and contracting rapidly in response to rainfall. Raydale has been assessed to be in moderate ecological standard (EA, 2009), although it only just failed reaching the good standard.

3.4.10 Mill Gill.

This stream stretches from Askrigg up to the Ure/Swale watershed. It is a tight valley with numerous gorge sections, high waterfalls revealing large sections of Yoredale strata and a large area of native woodland. At its headwaters on Abottside moor a large number of grips drain the peat soils. The stream is regularly discoloured and has a high velocity responding rapidly to rainfall. Several barriers prevent trout movement upstream. The gill is well wooded with native species and several indicator species

[25] Hagging describes banks of bare peat exposed by gully development. The exposed hags are prone to wind, water and animal erosion and can result in large fluxes of sediment transfer and thus severe degradation of moorland soils.

suggest some sections are ancient woodland. At the top end of the woodland a small larch plantation sits between the native woodland and open moorland. Land use includes dairy, beef and sheep with grouse moors on the highest ground. At the lower end the stream runs through the western edge of Askrigg. A small Waste Water Treatment Works discharges into the stream just before its confluence with the main Ure as a 3rd order stream. This area has been assessed to be in good ecological standard (EA, 2009).

3.4.11 Ballowfields.

This stream drains the smallest area of all the sub-catchments presented here. It also has the lowest altitude and the smallest altitude range. There are a number of barriers and historically this stretch has been polluted by heavy metals leaching from disused lead mines. Within a kilometre of the main river a large waterfall acts as a barrier preventing upstream migration. Above this fall, a holding pond had been used to provide water for past energy production. This poses a second barrier to fish migration. Above the pond the beck runs dry for most of the year and beyond this flows through extensive rush pasture. Further upstream, the stream runs underground and does not stretch up into the moorland. Land use is mostly sheep grazing, which is quite intensive above the first scar, and grouse moor at the higher elevations. There are no grips present in this area and the stream enters the main Ure as a 2nd order stream. This area has been assessed to be in moderate ecological standard (EA, 2009).

3.4.12 Gill Beck.

This is one of the smaller streams of the catchment with a low altitude and range in comparison to many of the sub-catchments further upstream. Land use at the lower end consists of sheep and dairy moving to beef and grouse moors on the upper reaches below Addleborough Hill. This stream runs dry in a number of sections due to sink holes. A dry waterfall acts as a barrier between the plateau of the moorland and the hillslope sections. Woodland is sparse but the moor and many of the pasture land is floristically rich. A number of base-rich flushes mix with peaty streams at the highest elevations. There are very few grips on the moor and heather is rife with low level grazing. As it passes through Thornton Rust there is anecdotal evidence that the water is polluted by poorly managed septic tanks. It enters the main Ure as a 2nd order stream. This stream has been assessed to be in moderate ecological standard (EA, 2009).

3.5 Land use of the wider dale

The catchment is largely rural with a high altitude and altitude range. Land use reflects the topography and supports a high number of dairy enterprises due to the Wensleydale creamery at Hawes. There are the typical range of conflicts and land use pressures of a UK upland region. These include livestock farming, extensive conifer plantations, large shooting estates, nature reserves and native woodlands. Large portions of the catchment are designated as Sites of Special Scientific Interest or have come under agri-environment schemes including Environmental Sensitive Area schemes, Countryside Stewardship, Woodland Grants and the more recent Entry and Higher Level schemes all administered by Natural England.

Yet despite all the interventions by way of stewardship schemes, there is still a recognised issue of nutrient delivery to watercourses. Lane *et al* (2006, p. 244) describe the location as, '*upland and piedmont landscapes where there is an acknowledged problem of in-stream eutrophication, believed to be related to phosphorus delivery. It has relatively shallow soils, relatively low levels of artificial under drainage and the predominance of low intensity pasture, which means that surface soil erosion and transport by overland flow is likely to be a major route by which phosphorus reaches the river system.*'

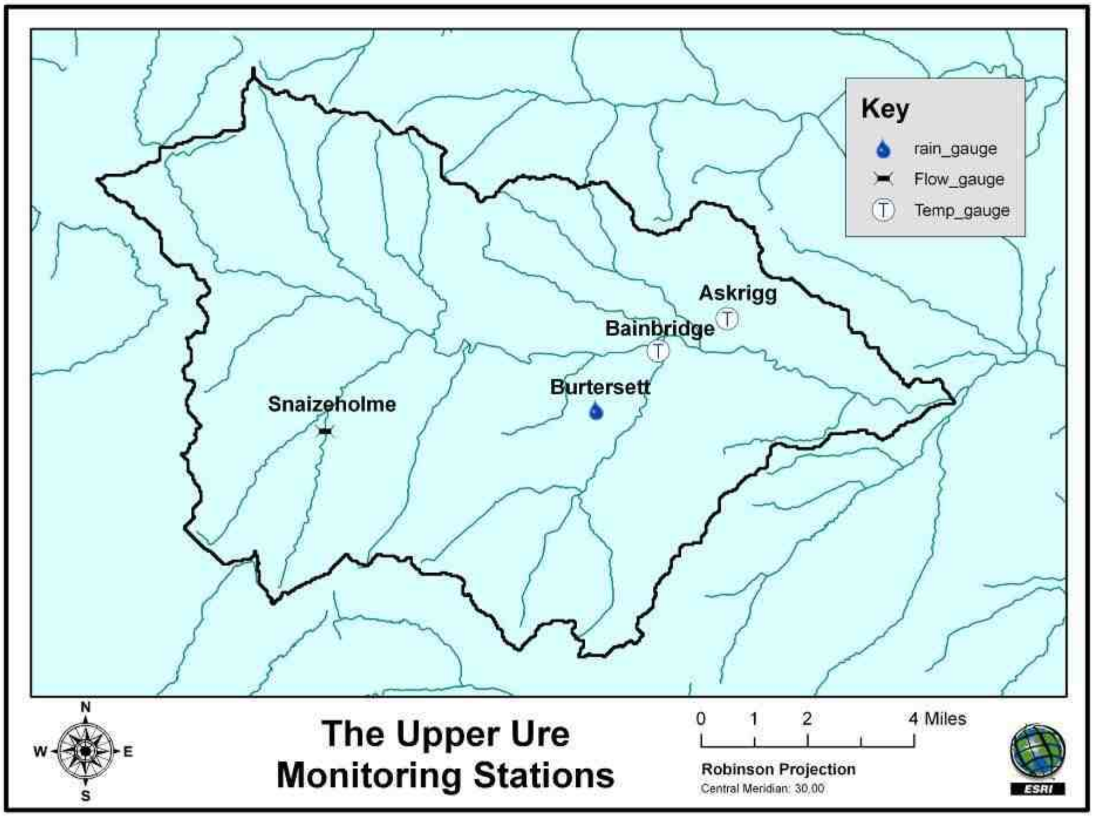
Whilst the levels of under-drainage of agricultural fields are low, the level of open drainage (gripping) on the moorland is high with especially high densities at High and Low Abbotside, Raydale and Mossdale. These are reputed to exacerbate both high and low flows, sediment delivery and water discolouration. To compound this grouse moor management involves rotational burning of heather to ensure a mixed stand and therefore habitat for grouse rearing. This results in areas of bare peat soils which are more prone to wind and water erosion.

3.6 Water sampling, flow and rainfall data

The British Atmospheric Data Centre offered an opportunity to explore medium to long term climatic data for the catchment. Temperature and precipitation data was downloaded for three sites in the upper Ure catchment, Askrigg, Bainbridge and Burtersett. The Bainbridge and Askrigg daily temperature records extended to a 17 year period whilst precipitation data for Askrigg extended between 1961 and 2008. There was no corresponding precipitation data for Bainbridge so data for nearby Burtersett was used. This extended between 1967 and 2000.

The upper Ure is typical of many UK upland catchments in that it is poorly gauged. The only flow gauge within the catchment is located at Snaizeholme. The data used here extended between 1972 and 2006. Unfortunately Snaizeholme is a sub-catchment with little or no upland drainage channels (grips) and therefore the data could not offer the possibility of exploring changes to flow dynamics from this land management style. Conversely, all the data offered the opportunity to explore climatic and flow dynamics in comparison to other upland catchments and under present climate change, either cyclical or anthropogenically induced. Figure 3.20 shows the locations of the gauges.

Figure 3.20: *the location of the gauges used for temperature, precipitation and flow data.*



There has been a plethora of research that has highlighted long term change in global climate (Solomon *et al*, 2007; Trenberth *et al*, 2007). These act out on regional scales in the form of shifting rainfall and temperature patterns. For example, during the1990s UK upland catchments documented record heavy rainfall events during winter months with an opposing summer trend (Burt and Ferranti, 2010; Burt and Holden 2010). In addition, Hulme and Jenkins (1998), noted a warming of 0.5°C in the UK Central Temperature Record (Manley, 1974; Parker *et al.*, 1992) over the course of the 20th century. Climate models corroborate the observed data suggesting increasing temperatures coupled with increasing winter rainfall contributing to higher annual totals despite decreasing summer rainfall (Christensen *et al*, 2007). Osborn *et al* (2000) defined a category 10 rainfall threshold as being the daily rainfall above which 10% of total rainfall occurs, Burt and Ferranti (2010) termed this the T10 threshold. Burt and Ferranti (2010) explored T10 events across a UK transect noting an increasing trend in T10 events during the winter months in upland sites.

3.6.1 Temperature data

Temperature data from BADC was transformed into seasonal and annual means. The mean was calculated from daily maxima and minima following the meteorological standard (Holden,2007). From this the monthly, seasonal and annual means were calculated. The seasonal analysis followed UK meteorological definitions taken from Burt and Holden, 2010: winter (December - February), Spring (March -May), Summer (June – August) and Autumn (September- November). Whilst the data sets are only medium term there was a clear trend with a one degree Celsius rise on the annual mean at both Bainbridge and Askrigg. Figure 3.21 shows the Askrigg annual and seasonal means, figure 3.22 shows the same for Bainbridge.

Despite being only medium term datasets there was a clear upward trend that mirrored trends noted elsewhere, though perhaps with a greater warming then noted in other catchments. These changes in temperature can be expected to alter hydrological response in the catchment through an increase in evapotranspiration and may impact on the high extent of organic peat soils. These may be converted from a carbon sink to a net carbon source placing a positive feedback on present climate change (Worrall *et al*, 2004; Burt and Holden 2010). To buffer against these pressures land management may have to become increasingly sensitive to environmental systems in order to safeguard water resource management and ecological systems.

Figure 3.21: *Temperature data at the Askrigg monitoring site showed an increasing trend in all seasons and the annual mean.*

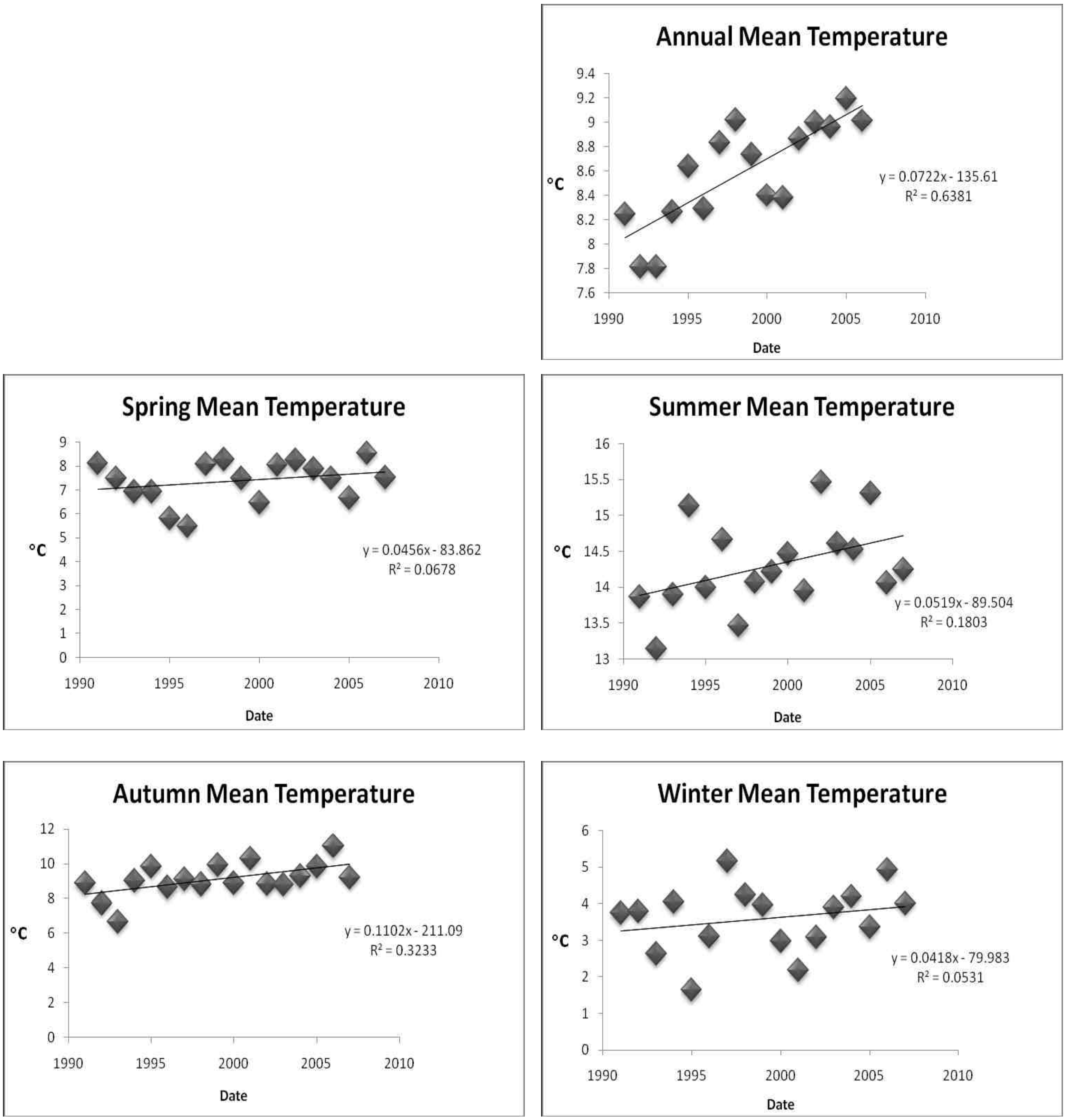
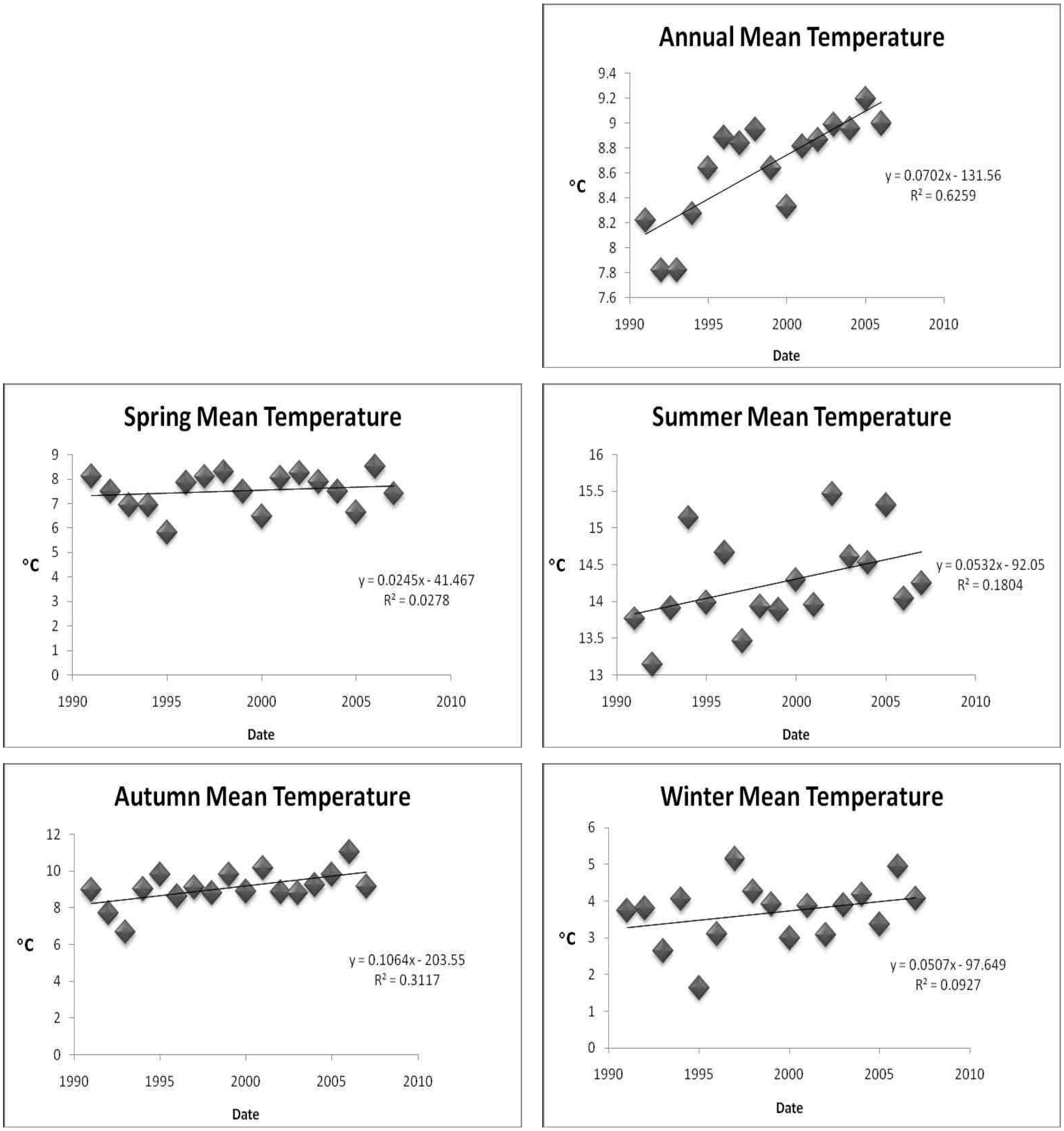


Figure 3.22: *Temperature data at the Bainbridge monitoring site also showed an increasing trend in all seasons and in the annual mean.*



3.6.2 Precipitation data

As with temperature data this dataset was transformed into daily,monthly, seasonal and annual means. The seasonal and annual means are exploredin figure 3.2.3 whilst the number of T10 events are displayed in figure 3.2.4. The patterns are clear. There is a slight increase in annual mean precipitation driven wholly by winter precipitation. Summer and Autumn means are decreaing whilst the spring mean remains static. Clearly this can impact on brown trout survival in the catchment. Reduced summer precipitaiton is likley to result in lower base flows reducing brown torut habitat and enhancing instream nutrient concentrations that place the river system at an increasing risk of eutrophicatrion. In addition reduced Autumn flows temper the spawning migration trigger whilst increasing winter flows put egg and fry populaitons at risk of wash out.

The trend in the T10 threshold mirror this with a slight annual increase in the number of events driven largely by winter T10 events. What is interesting to note here, in terms of the spawning trigger, is the strong downward trend in Autumn T10 events. This enhances the concern with the decreasing Autumn trend in mean precipitation, as brown trout require these high flow events to move upstreamin the river system in order to locate suitable spawning habitat in their natal streams. If river flows are reduced at this time of year the spawning fish may become trapped in unsuitable locations, such as below weirs or become concentrated for long periods in pools were competition may thin out the population. These noted trends confirm the findings of other studies carried out on longer data sets (Maraun *et al*, 2008; Burt and Ferranti, 2010) and should be of concern to hydrologists, ecologists, farmers and wider society.

Figure 3.23: *Mean precipitation at Burtersett shows a slight increase in the annual mean driven wholly by an increase in the winter mean.*

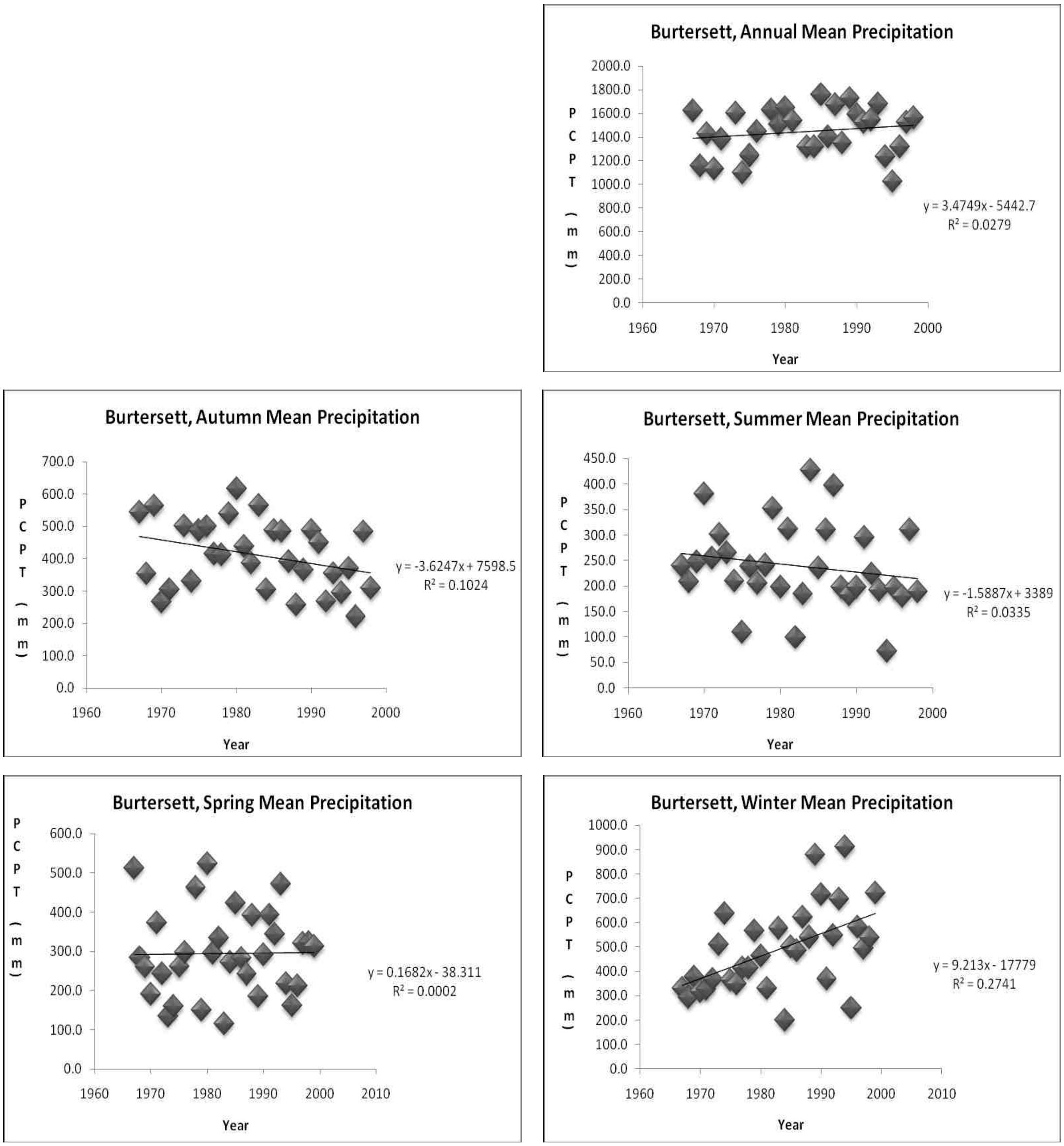
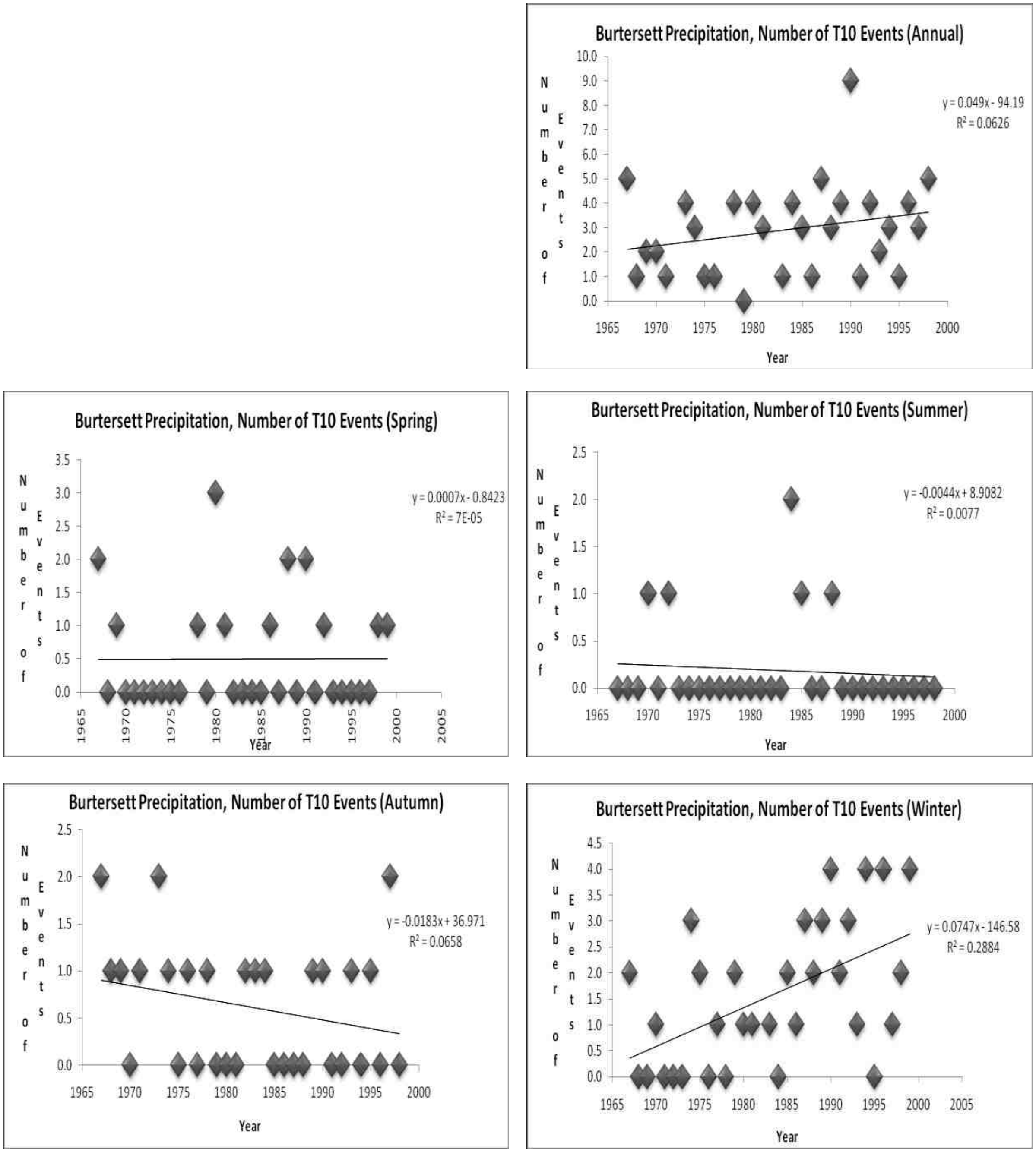


Figure 3.24: *The Burtersett precipitation T10 events mirror thenoted trend in seasonal and annual means with an increasing number of annual T10 events driven wholly by a winter increase inthese events.*



3.6.3 Snaizeholme flow data

Flow data from the Snaizeholme flow gauge was converted to daily, monthly, seasonal and annual means following the same conventions noted above. Here the seasonal and annual means are presented (3.25) along with the annual and seasonal 5 percentile flows (3.26). It has to be noted that the Snaizeholme flow gauge is located upstream of the other datasets used here and is the only flow gauge within the upper Ure catchment. It was unfortunate that the datasets were not gathered from close proximity allowing clearer comparisons to be made. Snaizeholme is a cauldron shaped subcatchment with steep hillslopes resuting in a rapid hydrological reponse. Therefore, it can be expected that flow dynamics here will not be the same as at Askrigg or Burtersett. However, whislt annual mean flow appears to have a static trend, winter flow does show an upwards trend with perhaps a slight decrease in Autumn flows. Thus, the flow response does mirror at least the winter findings from the prcipitation data.

The 5 percentile flows display stronger trends than the mean flows. Again the annual 5 percentile flow trend is fairly static but seasonal data show both decreasing and increasing trends that mirror the T10 rainfall events noted above. There is a clear increase in winter 5 percentile flows with decreases in both summer and autumn. Spring 5 percentile flow shows a static trend. This poses concerns for instream ecology, particularly with salmonid migratory triggers along with egg and fry survival. Again land management change and river restoration methods are the two tools that will allow these trends to be buffered against.

Figure 3.25: *The annual mean flow at Snaizeholme displays a static trend. However winter mean flow is increasing in contrast to autumn mean flows which are decereasing slightly.*

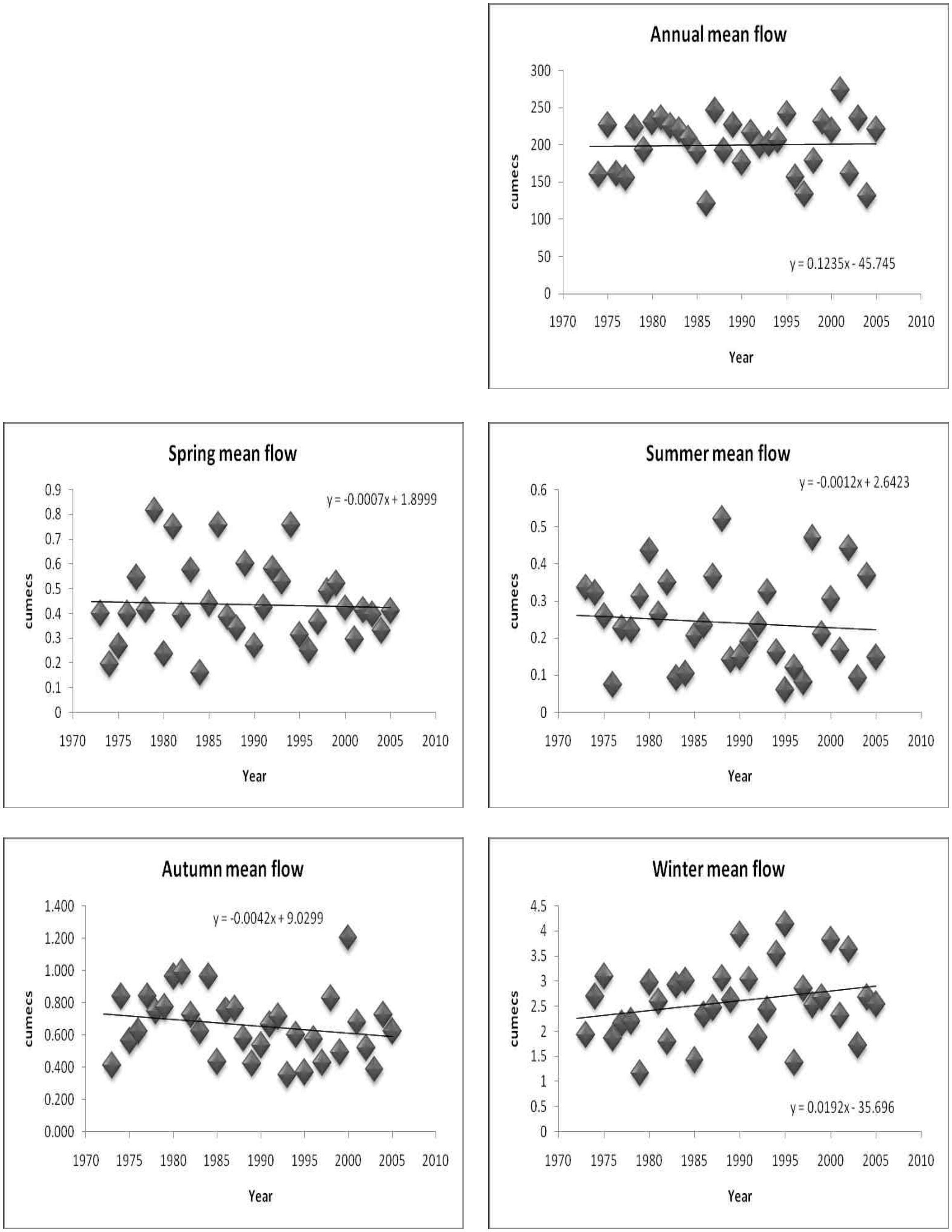
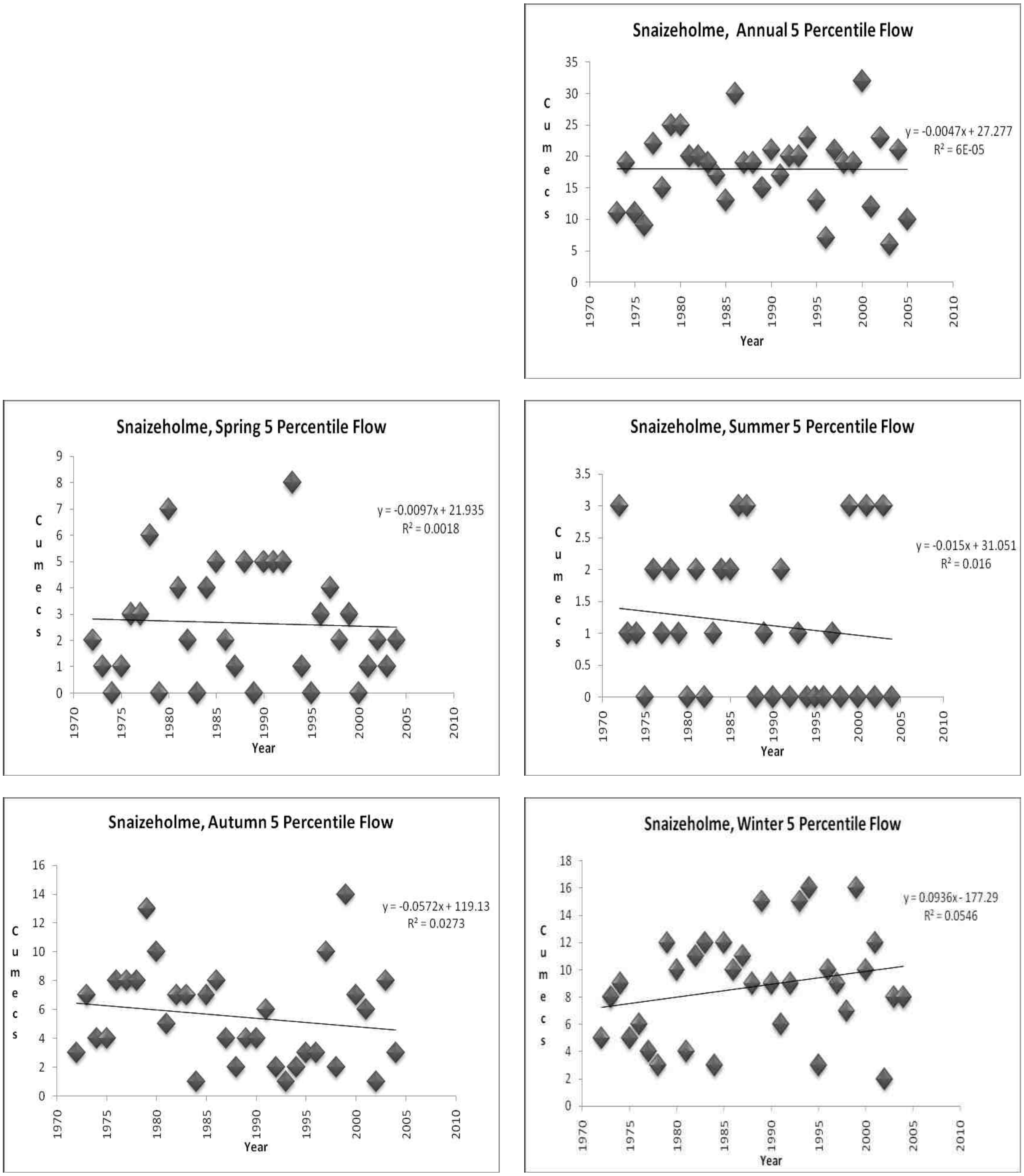


Figure 3.26: *Again the annual 5 percentile flow remains static but there are clear trends with the seasonal 5 percentile flows. Summer and autumn show decreasing trend whilst winter 5 percentile flows reveal an upward trend.*



3.6.4 Spatial and temporal water sampling

To increase knowledge of the case study catchment it was considered important to carry out a series of water samples in order to gather spatial and temporal data to inform on point source and diffuse pollution. Four samples were taken above and below Bainbrodge and Hawes WWTW discharge pipes over an eight week (Bainbridge) and sixteen week period (Hawes WWTW) during summer 2008 when concentration was likely to be at a peak due to low summer flows. The first sample was taken 10 metres above the discharge pipe,the second ten metres below, third 50metres below and finally 100 metres below Figures 3.27 and 3.28 show the results.The graphs display the threshold for salmonids and it is clear that only the sample taken ten metres below the discharge pipes breach the low level threshold. The Bainbridge WWTW serves a population of less than 500 whilst the Hawes WWTW serves a population of 1115, though during the peak tourist season the population equivelant doubles (Neale, 2008). Below the dicharge pipe from the Hawes WWTW phosphate levels peak however, the samples taken further downstream are lessofa concern. The resultsformthe Bainbridge WWTW show a similar pattern,though the peak directlybelow the discharge pipe are lower.

Figure 3.27: *the results from the water sampling above and below the Bainbridge WWTW.*

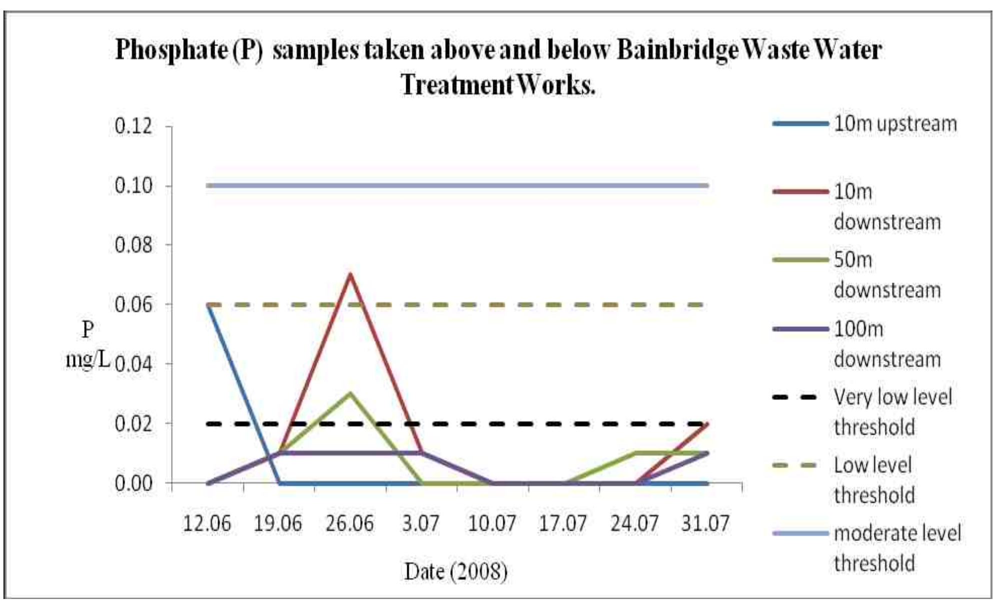
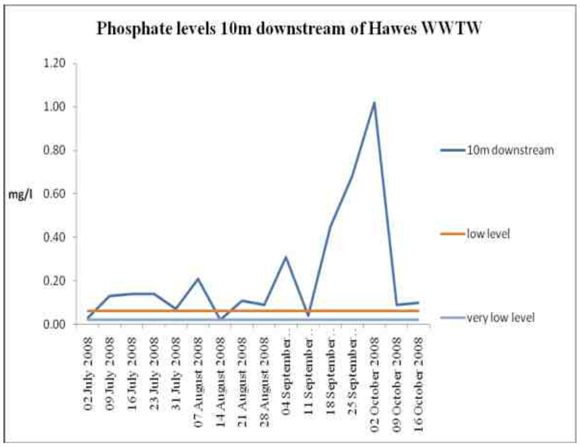
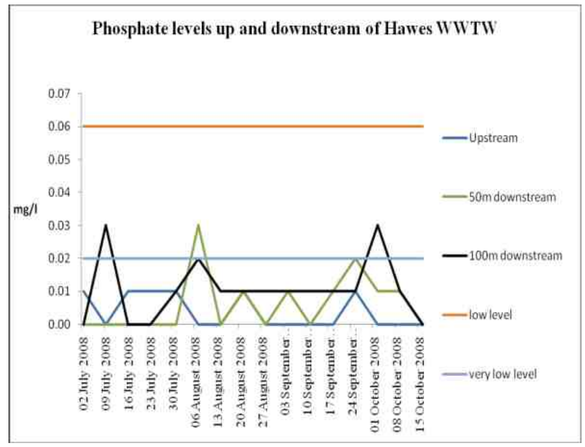


Figure 3.28: *the results from the water sampling above and below the Hawes WWTW. Here the spikes directly below the discharge pipe are substantially higher and willpose a threat to brown trout and other stream ecology. However the phosphate levels soon settle into a moreacceptable range at the sites further downstream.*

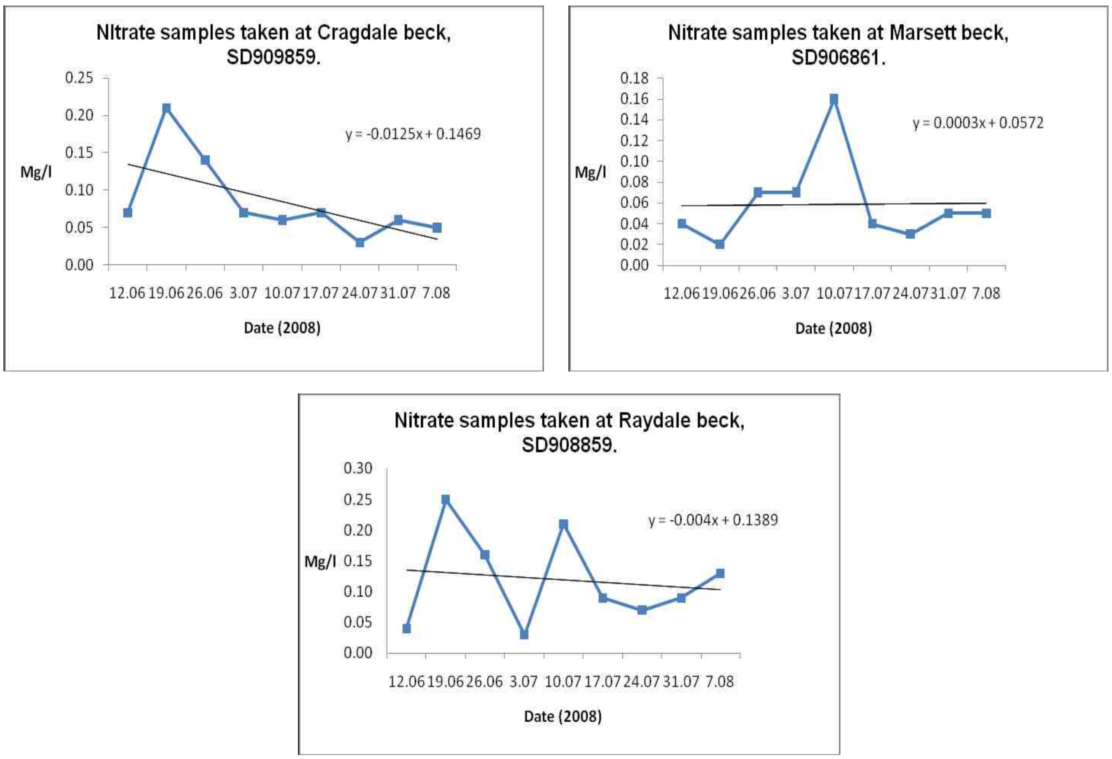




Two rapid water sample surveys were carried out across the catchment, when flow was moderate, in order to visualise nutrient levels across the Ure river network. What was interesting was that both phosphate and nitrate levels were either very low or below detectable limits. This was surprising as walkover surveys suggested that many farming methods posed a hgh risk of nutrient and sediment delievery to the watercourse. Due to this a short series of samples were taken in Raydale on the three feeder streams to Semerwater Lake during summer 2008. These streams were chosen as Semerwater Lake had previously been identified as having a sediment and nutrient problem arising from the surrounding agricultural practices, much of which is dairy farming. Again phosphate returns were all very low or below detectable levels. The nitrate returns can be seen in figure 3.29. Again these are relatively low and, as phosphate is the main limiting factor in freshwater systems, was not considered to be of major concrern.

These results provided a dichotomy between the walkover surveys and water samples. It was apparent from visual evidence that farming practice was risky at a number of locations whilst the water samples did not corroborate this finding. It may be that much of the phosphate is delivered attached to sediments, rapidly locked up in primary production or the sampling period missed the key events that deliver nutrients. A final possibility is that the percieved risk is simply not realised here. However, the EA have previously identiifed the subcatchment as having a nutrient delivery issue.

Figure 3.29: *Nitrate samples taken from the three feeder streams to Semerwater Lake were all low and not a major concern. This was a surprising result and did not marry with observations from the walkover surveys.*



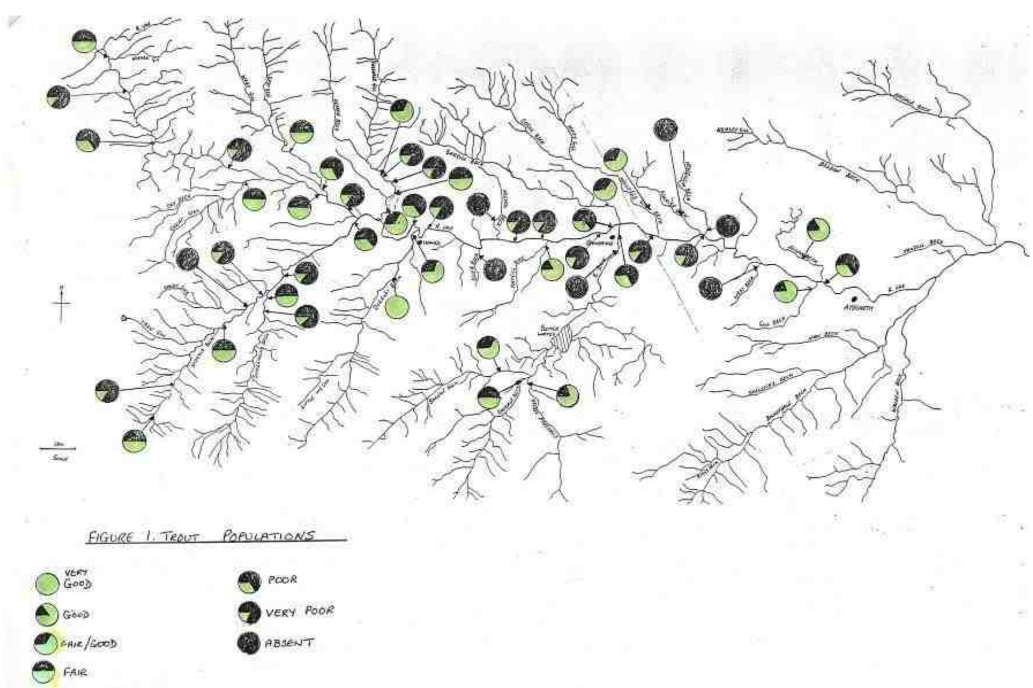
3.7 Salmonids in the upper Ure

Whilst Aysgarth Falls is considered a barrier to upstream migration, some anecdotal and photographic evidence suggests migratory fish do manage to navigate the falls. However, this occurs at such low density, as shown through electro-fishing surveys, that the upper Ure catchment can be considered to hold only a resident brown trout stock. Stocks of resident trout are so low that high re-stocking of fish has been pursued by a number of angling clubs. The Environment Agency has commissioned several reports on the state of the catchment including two eutrophication reports carried out by Atkins (2004) along with a number of fisheries reports and a programme of water quality monitoring.

The first Atkins report (2004) showed that the low trout densities throughout the catchment are, '*clearly unacceptable for an upland river of this type. The conditions in the river have in-fact become so poor that the river is now almost entirely dependent on stocking.*' The second Atkins report (2004) more succinctly comments that, '*populations within the Upper Ure are poor*.' Hopkins (1988) composed a map highlighting the conditions of brown trout stocks throughout the catchment (Figure 3.30). As can be seen the majority of sites are below the fair standard with 7 sites containing no trout at all and only one site achieving the very good standard. There have been no reports that map the relative population since this highlighting the paucity of information in the catchment.

The evidence to support the view that trout numbers are below acceptable standards comes from anecdotal evidence and semi-quantitative electro-fishing carried out by the Environment Agency. Local angling clubs, of which there are several, regularly stock the main river in response to these low numbers; however, they do not introduce stocks to the tributaries. Giles (2006) shows, using anecdotal evidence across the UK, that there is a steady decline in brown trout stocks in most rivers despite the diverse geographical regions.

Figure 3.30: *Diagram of brown trout populations showing stock condition at different locations across the catchment.* (Hopkins, 1988).



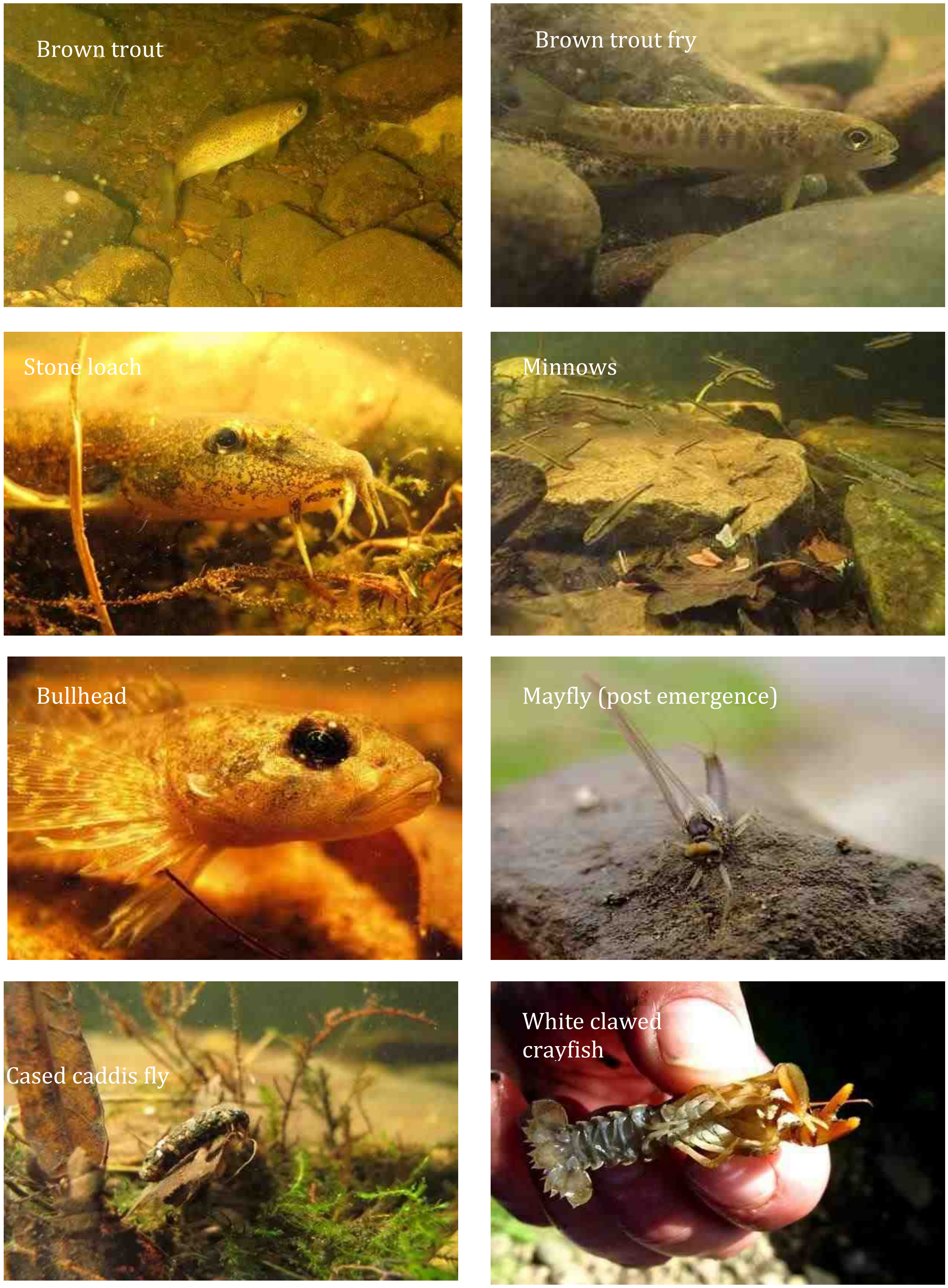
3.8 Ecology of the River Ure

The rivers and streams of the case study catchment are typical of many UK upland river networks with gravel beds set in steep sided hills dominated by glacial topography or v-shaped valleys. The waters are generally well oxygenated and contain a number of species that are associated with good water quality. Brown trout is one such species but others include macroinvertebrates such as stonefly (Plecoptera), mayfly (Ephemeroptera) and caddisfly (Trichoptera). Larger macroinvertebrates include white clawed crayfish (*Austropotamobius pallipes*) , this species is able to inhabit a diverse range of habitats including streams, rivers and lakes and have a specific requirement of high calcium waters (dissolved calcium content >5 mg/l).

Other fish species typical of the river system include stone loach (*Barbatula barbatula*), bullhead (*Cottus gobio*), and minnow (*Phoxinus phoxinus*). In the main river stem grayling (*Thymallus thymallus*) and pike (*Esox lucius*) can be found. Semerwater Lake in Raydale also contains bream (*Abramis brama*), perch (*Perca fluviatilis*) and roach (*Rutilus rutilus*). As adults brown trout position themselves at a high trophic level whereas the fry are preyed on by a number of aquatic species including bullhead and even adult brown trout.

The connections between terrestrial and aquatic ecosystems are two way. Brown trout will readily prey on terrestrial invertebrates that fall into streams and become trapped by surface tension. In contrast aquatic macroinvertebrates are taken by dipper (*Cinclus cinclus*), daubentons bat (*Myotis daubentonii:* as well as other bats), sand martin (*Riparia riparia*), swallow (*Hirundo rustica*) and grey wagtail (*Motacilla cinerea*). Fish species including brown trout are preyed on by kingfisher, red breasted merganser (*Mergus serrator*), grey heron (*Ardea cinerea*) and otter (*Lutra lutra*). These links between freshwater and terrestrial systems highlight ecological connectivity offering another reason (in addition to hydrological connectivity) why freshwater systems cannot be views in isolation of the wider landscape. Some species found in the catchment can be seen in figure 3.31.

Figure 3.31: *Some of the species found in the case study river system.*



3.9 Institutional Framework.

The landscape of the upper Ure is diverse and supports a number of important habitats, species, geological conditions and landscapes. This has resulted in a range of designations and overseeing institutions ranging from small NGOs to large governmental bodies. Due to this, the local institutional framework is complex and comprises of several tiers of influence. Table 3.2 shows the national and international laws that have bearing on the location. Table 3.3 lists the institutions that hold varying degrees of influence over the landscape and ecology.

Table 3.2: *National and international laws applying to the Upper Ure catchment*

Laws & Designations	Description
National Designations	
Wildlife and Countryside Act, 1981 Site of Special Scientific Interest (SSSI).	**The river Ure and its catchment supports a number of important species including Hen Harrier, Red Squirrel, Otter, white-clawed crayfish, bullhead, and the brown trout. A number of SSSI are designated for their geological interest with others designated for their botanical interest.**
The National Parks and Access to the Countryside Act 1949	**The full extent of the case study site lies within the Yorkshire Dales National Park.**
European Designations	
Directive 92/43/EEC, Conservation of Natural Habitats and of Wild Fauna and Flora. Special Area of Conservation (SAC)	**Extends the level of protection provided under the SSSI notification to include residual alluvial woodland.**
Directive 2000/60/EC, Water Framework Directive	**Under the Directive all European waters must achieve 'good ecological status' by 2015**

Table 3.3: *Institutional management framework within the upper Ure catchment*

Institution	Examples	Description
Central government	Department for the Environment, Food and Rural Affairs (DEFRA)	UK government dept responsible for rural development, the environment and the countryside.
Statutory public bodies	Environment Agency (EA) Natural England (NE)	Amongst other responsibilities the EA are the competent authority for delivering the Water Framework Directive. NE is responsible for maintaining SSSIs and SACs in favourable condition and manage agri-environment schemes.
Non-departmental public bodies (NDPBs)	Yorkshire Dales National Park Authority.	Manage and co-ordinate conservation efforts within designated and protected areas of the catchment. They rely on central government and the statutory public bodies for funding. Also responsible for public rights of way and planning.
Local government authorities	North Yorkshire County Council.	Responsible for refuse collection, highways and lighting. The authority has transferred planning authority to the national park.
Non-governmental organisations (NGOs)	Yorkshire Dales Rivers Trust; Yorkshire Wildlife Trust; Yorkshire Dales Millennium Trust; Campaign to Protect Rural England.	Cover a diverse range of environmental and conservation remits.

3.10 The rivers trust movement and the Yorkshire Dales Rivers Trust.

The rivers trust movement provides grassroots, bottom-up, community involvement in river conservation and restoration. There are now thirty-one river trusts covering the majority of river catchments in England and Wales. The movement developed from riparian, angling and river associations with a perceived need for local involvement in river conservation and restoration. The decline of Salmonid stocks has been a major driver in their formation. The Association of Rivers Trusts (ART) is the overseeing charitable body and states its aim as, '*to co-ordinate, represent and develop the aims and interests of the member trusts in the promotion of sustainable, holistic and integrated catchment management and sound environmental practices, recognising the wider economic benefits for local communities and the value of education.*' (ART 2008). They also state that river trusts are viewed as having, '*wet feet because they have the reputation of being doers, concentrating much of their effort on practical catchment, river and fishery improvement works on the ground.*'

The Yorkshire Dales Rivers Trust (YDRT) was established as a registered charity in 2004 as a response to a lack of concerted effort in river restoration at a local level. It covers a wide geographic region with an interest in the rivers Swale, Ure, Wharfe and Nidd from the headwater streams down to the Humber estuary. From its inception the trustees decided to carry out work based on three founding principles:

1. All work will be based on the best available science.
2. Work will begin at the upper reaches of a catchment before moving downstream.
3. Work will be carried out at the catchment scale in order to understand and respond to the processes that impact on river ecology.

However, much of the work is compromised by the availability of funding and the willingness of landowners and farmers to allow restoration work on their land. A second driver for founding the trust was the lack of effort towards engaging local communities, farmers and landowners in conservation and restoration. This was seen a prime barrier to cooperation and thus for improving the condition of the dales catchments. Without these partnerships, conservation and restoration had been piecemeal, inefficient and poorly supported. This, along with the founding principles, governs the approach of the Trust.

YDRT state, that '*upland farming is a notoriously difficult enterprise that provides small financial returns relative to the hours worked. It is essential that farming is maintained in the dales to ensure traditional forms of agriculture survive economic pressures but also to continue attracting visiting tourists. In the Dales land use has created conditions that provide habitat for numerous unique species,*' (YDRT, 2009). It is the recognition of the connections between land use, economic and physical processes and river ecology that provides the trust with a holistic view of catchment management. The trust has been instrumental in commissioning the research reported in this thesis.

3.11 Conclusion

By situating the research within a case study catchment multiple pressures on brown trout fry can be explored. This allows a variety of interacting scales to be accounted for and offers potential to identify the locations that require restoration and, of equal importance, which management options would be most suitable at a given location. The attraction of the upper Ure catchment is that it holds a resident brown trout stock with no, or little, recruitment from anadromous brown trout. This allows the research to focus on in-situ catchment factors that may be limiting recruitment without further complexity being added.

The upper Ure has long been considered to contain poor brown trout stocks. This suggests that there are limiting factors acting on the species. Moreover the land use varies between the sub-catchments and gripping of the moorlands is not equally distributed across the Dale. This allows differences in relative fry recruitment to be tested against a range of pressures, natural and human.

Chapter 4 will explore advances in remote sensing, GIS and modeling. This will be undertaken in order to capture data for later analysis against brown trout fry populations. In so doing it will also provide modeled fine sediment data that can be tested using, 1) human knowledge and 2) brown trout fry. This will meet objective 2 and prepare the data to meet objectives 3 and 4.

4.0 Methodology

The catchment review has offered a unique insight into an upland community and landscape. The lack of monitoring in the upper Ure catchment has been especially interesting and highlights the difficulties of working in data poor landscapes. For example, there is only one flow gauge within the catchment and this is situated on a sub-catchment that remains ungripped and so offers no information on how upland drainage has impacted flow rates. Moreover, there are no historic data on the extent of grips within the catchment, nor elsewhere in the dales. Natural England (and its forerunners English Nature and Rural Development Services) have only mapped grips on a land-holding basis only specifically for stewardship schemes. This provides piecemeal detail limited by confidentiality rule. In addition to this, there are no data on sedimentation, even though this appears to be a serious issue judging by the condition of the river banks.

Data from the Snaizeholme flow gauge show winter flows have an increasing trend. This includes maximum, 5^{th} percentile (Q5) and the mean daily average flow. Rain and temperature measurements from Askrigg, Bainbridge and Burtersett do not appear to show corresponding increases. The flows measured in spring, summer and autumn show a slight downward trend whilst the aggregated flow holds steady. The interesting aspect of the Snaizeholme data is that they come from a sub-catchment which has not been gripped with the majority of the land utilised for rough grazing, coniferous woodland and deciduous woodland. The reasons for this increasing winter flow could be many; however, it seems unlikely that Snaizeholme is suffering from severe soil compaction due to only extensive grazing within the dale. What it does suggest is that brown trout eggs and alevins within spawning redds may be at an enhanced risk of wash out. In addition, the slight downward trend shown in the other seasons could place greater pressure on fry populations. It is unfortunate that this is the only flow gauge within the catchment as this leaves no possibility of comparison with a gripped subcatchment within the study area.

Water sampling was carried out as part of the catchment review. The majority of the results show low phosphate levels with some spikes around waste water treatment works. These spikes tailed off after 100 metres and there were few away from treatment works that revealed any significant levels of nutrients. This is in spite of the condition of a number of farms that suggested there would be an issue with nutrient and fine sediment delivery (with the possibility of sediment-bound phosphate). It is possible that phosphate is taken up rapidly by algae or that water sampling missed any spikes of phosphate. It is equally possible that it is not an issue within the catchment, though the condition of a number of farms and fields in conjunction with stock access to watercourses and regular slurry spreading makes this unlikely.

These initial investigations reveal how the absence of data fuels conflict and allows differing opinions to maintain validity. Moreover opinions appear to be governed by preconditioned vested interests. In relation to river pollution farmers blame waste water treatment works or road runoff, environmental professionals seem predisposed to assign blame on diffuse impacts from farming, water companies highlight the lack of data to justify maintaining the status quo whilst forestry interests believe lines of brash can resolve sediment laden runoff during felling operations that encroach up to the river bank. In the meantime anglers have long blamed the waste water treatment works at Hawes for poor brown trout and grayling populations in the River Ure. Each of these interests is able to offer anecdotal, and perhaps accurate, evidence to support their claims. Chapter Two reveals that each claim is likely to have elements of truth. Upland rivers are generally impacted by multiple issues arising from a variety of sectors.

This work wishes to explore the catchment by utilising traditional and modern methods of investigation. In addition, there is an explicit attempt to cross scale and link impacts with the underlying processes that connect pollution sources to recipient streams. In so doing, it is felt that a need to ascertain how scale, processes and human interventions interact and combine to impact on river systems is important. This requires novel approaches to research that not only details a pollution source, pathway and recipient stream, but also explores the human processes involved and how these map out to either enhance or limit such impacts.

In an upland hill farm-dominated catchment agriculture has the greatest impression on the landscape in terms of visual impact, land use change away from a natural vegetation type and, dependent on the farming methods employed, the greatest potential for pollution impact. During the initial stages of this research, a number of people implicated agriculture as being a possible reason for the poor (or perceived poor) condition of the river Ure. This theme appeared in the majority of initial interviews and occasionally came through when discussing the issues with members of the agricultural community. This suggests that there is either a real issue with agriculture or its dominance within the catchment places a strong control on perception.

The walkover surveys of the catchment did suggest that agricultural practice was widely varied with a number of locations and farm types appearing to display disproportionate risk to the river. These risks included stock access to rivers, heavy poaching (livestock dominated soil erosion) around supplementary feeders and gateways, dairying and slurry spreading. In addition river banks within the catchment were noted to be generally unfenced and clearly eroding. Of the few locations where the river was fenced, the vegetation structure appeared to offer a strong buffer against erosion and the bank condition corroborated this.

The other land use type that interviewees gave high significance in terms of impacts on water quality was upland drainage. The general theme of criticism towards this land use type was the increasing flashiness of the river. Secondary to this, but also considered to be important was reduced base flows and increasing delivery of particulate matter to the river network. It was surprising to note that forestry got very few comments despite its high predominance on the hillslopes of the catchment. This could perhaps be explained by the lack of logging activity when these initial explorations of the catchment and interviews were carried out. Since then, logging activity has become widespread and with this has come an increasing concern. A number of people have suggested that sediment delivery to watercourses has increased in tandem with logging. One farmer suggested that the streams now ‘run black’ after rainfall.

These initial interviews coupled with the widely held perception that brown trout populations within the catchment have been in decline placed strong demands on this research in terms of the variables to be explored. Moreover, the literature supported many of the issues highlighted. This offered two sources of evidence suggesting there was a need to explore land use in terms of its impact on brown trout fry populations. A number of issues were excluded due to either a lack of evidence or due to location precluding them from directly impacting on recruitment. For example, the location of waste water treatment works meant that they were highly unlikely to impact on any spawning location. In addition, early discussions with Yorkshire Water (YW) and the EA offered little hope of discovering the condition, and often the location, of septic tanks. In some locations it was possible that poorly maintained septic tanks could be impacting on streams where recruitment took place but there was little hope of ascertaining if this was indeed the case, at least not at a catchment scale.

The methods employed here range from investigations at the reach scale to those at the catchment scale. Moreover, modelling was carried out at two spatial scales. The more common usage of SCIMAP has been employed by running the model at the catchment scale at a location with a known problem of eutrophication. Secondly, it was run at the farm scale to explore how accurate the risk categories are for the upper Ure catchment and to ascertain 1) the level of trust farmers have in the model; and 2) whether land management reduces or exacerbates the SCIMAP risk rating. This is necessary as SCIMAP only offers information on probabilities and so land management techniques can either enable the risk to become realised or hold it in check. Moreover, it offers two methods for validating the model, the first at the in-stream scale against what ecological components of ecosystems can tell us and, secondly, against information economic interests within the dale can offer.

Therefore, in order to capture the necessary level of complexity, numerous factors known to impact brown trout fry populations should be collected. As chapters 2 and 3 suggest the important factors are not simply within the organism's immediate habitat but stretch upstream and laterally into the terrestrial system, wherever land is hydrologically connected to a receiving watercourse or land management techniques are likely to impact. To do this takes careful planning in order to capture factors that matter at the appropriate scale so that the appropriate data can be employed for statistical testing against brown trout fry populations. The methods for data capture are described below, starting with the capture of brown trout fry data and expanding the collection out from the immediate habitat into the riparian zone, floodplain and finally to the catchment scale. It should be noted that many of the possible impacts transcended scale and therefore data collection had to account for this. For example, land use can be measured at the catchment, floodplain and riparian scales. The factors incorporated into the data collection can be seen in table 6.1 below.

The methodology described is placed into two broad categories. The first to be discussed are the more traditional field survey methods (4.1 to 4.5). The second section of this chapter discusses GIS, remote sensing and modelling methodologies employed to gather relevant data to help explain brown trout fry populations.

Table 4.1: *The data collected, scale at which each operates and method of capture*

Factor	Scale operating	Method of capture
Brown trout fry populations	Habitat	Electrofishing
Macroinvertebrate abundance	Habitat	Kick sampling
Macroinvertebrate richness	Habitat	Kick sampling
Simpsons diversity index (1/Total)	Habitat	Statistical analysis of kick sample result
Shannon's diversity index	Habitat	Statistical analysis of kick sample result
LIFE scores	Habitat	Statistical analysis of kick sample result
Obstructions upstream (<500m)	Reach	OS maps and field surveys
Obstructions downstream (<500m)	Reach	OS maps and field surveys
Obstructions upstream (<1km)	Reach	OS maps and field surveys
Obstructions downstream (<1km)	Reach	OS maps and field surveys
Survey area prone to drying	Habitat	Anecdotal evidence from National Park and Environment Agency staff and field observations
Stream prone to drying (d/s)	Reach	Anecdotal evidence from National Park and Environment Agency staff and field observations
Stream prone to drying (u/s)	Reach	Anecdotal evidence from National Park and Environment Agency staff and field observations
Bedrock	Habitat	Field surveys
Boulders and cobbles	Habitat	Field surveys
Pebbles and gravel	Habitat	Field surveys
Sand and silt	Habitat	Field surveys
Siltation	Habitat	Field surveys
River width (m)	Habitat	Field surveys
Pools present	Habitat	Field surveys
Algae (1: low 2: moderate 3:high)	habitat	Field surveys
Macrophytes	Habitat	Field surveys
Undercut	Habitat/riparian	Field surveys
Earthcliff	Habitat/riparian	Field surveys
Stock access	Habitat/riparian	Field surveys
Buffer	Riparian	Field surveys

Factor	Scale operating	Method of capture
Land use	Riparian / Floodplain / Catchment	Field surveys
Poached	Riparian	Field surveys
% shading	Riparian	Field surveys
Extent and location of upland drainage (grips)	Catchment	GIS and remote sensing
Upstream contributing area (km^2)	Catchment	GIS
Area of upstream moorland (km^2)	Catchment	GIS
Strahler stream order	Catchment	GIS
SCIMAP without grips	Catchment at fine resolution	Modeling and GIS
SCIMAP with grips in DEM and LCM	Catchment at fine resolution	Modeling, GIS and remote sensing
SCIMAP unweighted by land use	Catchment at fine resolution	Modeling and GIS
SCIMAP farm scale	Field scale at fine resolution	Modeling and GIS
Exploration of farm scale results	Field scale	Interviews and walk over surveys with appropriate farmers

4.1 Methods: Field data collection

As can be seen from Table 4.1, field work is an important aspect of this research. Whilst developments in GIS and modeling technology allow a number of observations to be made ex situ, these cannot offer data on in-stream organisms beyond offering suggestions on habitat quality. In addition, GIS layers such as the CEH landcover map may become outdated rapidly. For example, the riskiness in terms of fine sediment delivery is very different for intact woodland as opposed to felled areas. To capture such detail, field observations and surveys remain a necessity. The following sections describe the field data collection employed for this thesis.

4.1.1 Capturing spatially distributed brown trout fry data.

Two methods were considered for collecting brown trout data: 1) spotlighting and 2) electrofishing. In upland streams with a high proportion of riffle habitat that has regular episodes of discoloration from DOC, Hickley and Closs (2006) suggest that electrofishing is the most suitable method as it provides higher population estimates relative to spotlighting (which is more suited to clear water streams). Electrofishing involves stunning fish using an electric current to enable their easy capture with a hand net. There are number of methods appropriate to different conditions and scientific requirements including triple-pass quantitative survey methods, semi-quantitative single-pass surveys and spot sampling. Crozier and Kennedy (1994) developed a method of semi-quantitative electrofishing specifically for sampling 0+ salmonid species which involved 5-minute sampling of riffle habitats. This enables the same amount of fishing effort at numerous riffle habitat sites across a catchment, thus allowing rapid data collection on salmonid populations. Crozier and Kennedy (1994) recommend fishing downstream although Alabaster and Hartley (1962) found little difference in efficiency whether fishing in a downstream or upstream direction, if collecting fish in a hand net.

The Environment Agency employs a different method of semi-quantitative electrofishing which involves fishing a 50-metre stretch of stream using a single pass in combination with a number of triple-pass quantitative surveys. The triple pass method employs stop nets at the up and downstream extent of the survey site to prevent in- or out-migration of fish. This ensures that the single-pass fishing surveys provide a good percentage. The EA recommends that at least 60% catch of the total population is captured in a single run in order to make the sample as robust as such a method can allow; although the 60% efficiency value is arbitrary, it does attempt to set a level of acceptable efficiency. In other areas, researchers use a lower arbitrary measure of efficiency. For example, Kennedy and Strange (1981) suggested that 50% was the minimum efficiency required when electrofishing the river Bush (Northern Ireland). If fishing for 0+ salmonid fish, the EA utilise smooth direct current (as opposed to pulsed DC) as this results in a reduced stress response and lower spinal injuries amongst fish (Young and Schmetterling, 2004).

The electrofishing method used for this research followed the EA method for three reasons: 1) the EA (Lee 2007; Frear 2007) provided theoretical and practical training in the method; 2) the possibility of data sharing could add to the results; and 3) the EA will be charged with monitoring UK rivers in line with the requirements of the WFD. If this research is to have practical applications by river managers, it was considered important to employ the methodology they would use in-house.

The equipment for electrofishing was loaned from the EA (Lee 2007; Frear 2007) and consisted of a dedicated electrofishing hand-held generator (Honda: EU inverter 20i generator - unearthed), a control box (Electracatch: WFC7-96), a cathode, which remains submerged during surveys and generally at the upstream section of the survey site, and a single anode ring attached to an anode pole on a 50 m flex. The anode is swept through the water to ensure all microhabitats are sampled. One or two people (dependent on stream width) follow close to the anode operator and collect the fish as they appear. The fish are then transferred to a holding bucket for counting post-fishing. The small streams in which brown trout fry are generally found means a small anode ring was employed due to the proportion of boulders and cobbles on the stream bed. The voltage was set to 50V, as with such small streams the size of capture field is less important.

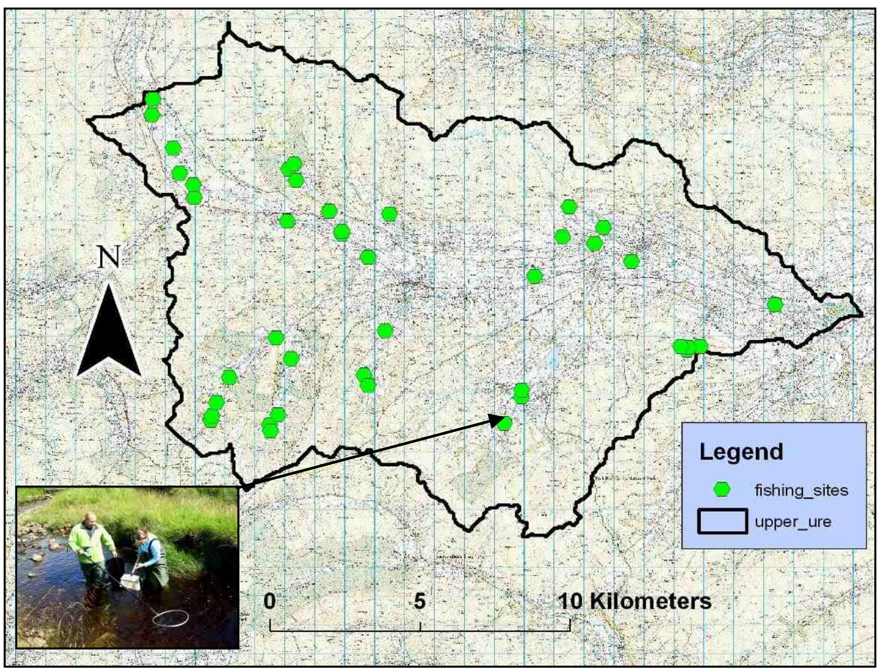
When collecting animals, employing a method that can cause trauma, there are a number of important ethical considerations to lessen the stress the animal endures. First, there are times when this method should not be employed. These include very low flow conditions and water temperatures above 16°C, as the fish will be in a pre-stressed state at these times. Secondly, they should not remain in holding buckets longer than required; this becomes increasingly important when air temperatures are high. If fish are being held for long periods, then the water should be replenished regularly. Triple-pass surveys should not be carried out during such periods as the fish caught in the first run would remain in the holding bucket for too long a period as, after each pass, there is a rest period of at least twenty minutes. This would not be acceptable without the ability to reoxygenate the water. Other important considerations are the safety of the field operators. In high flows, surveys should not be carried out. The surveys were generally carried out with a team of four people to ensure that at least one person remained on the bankside so that the control box could be disabled if required. When survey days were undertaken on streams narrow enough to warrant only one netter, the team would occasionally operate with only three people.

The survey sites were chosen following a number of considerations that included stream width and accessibility (both in practical terms and gaining permission from the relevant landowners). This process was carried out with assistance of Hannah Fawcett (Yorkshire Dales National Park conservation officer), Matt Neale (Yorkshire Dales National Park ranger for upper Wensleydale - upper Ure catchment) and Michael Briggs (Yorkshire Dales National Park access ranger for upper Wensleydale - upper Ure catchment). Their knowledge of the catchment and the landowners was invaluable in order to select sites and subsequently gain access to the sites. In total 49 sites were selected in order to provide good spatial coverage of the low-order streams of the upper Ure catchment (area: 234 km^2). Prior to electrofishing, each site was visited to ensure suitable brown trout fry habitat: adequate gravel in riffle-pool sequences with an absence of step-pool cascades. This stratifies the search to where brown trout fry exist, in order to gather information on the condition of the species at the appropriate life stage (Lane *et al*, 2008; Downes, 2010). Exploring any other type of habitat would offer little in terms of understanding recruitment. The only exceptions to this was immediately downstream of waterfalls where it was considered likely that physical barriers would encourage spawning in sub-optimal locations.

National Park staff and their team of volunteers assisted with the surveys during the period July to late September 2007 and 2008 by providing assistance with data gathering and accessing sites using a quad bike and trailer. Many of the sites would have been inaccessible without this assistance using the bankside equipment on loan from the EA. The surveys began late July during both seasons in order for the brown trout fry to have reached a size to enable their capture. Prior to this, the EA advised that they would be >5cm making capture rates inefficient. Thus the surveys would have fallen below the adequate capture efficiency. Crozier and Kennedy (1995) suggest that in late summer brown trout fry would be approximately 9.0 cm, however in the Ure catchment any fish caught below a length of 7.5cm was considered to be brown trout fry in line with EA observations (Frear 2007).

Crozier and Kennedy (1994) kept a count of 0+ fish seen but not captured (observations by all operatives) in order to develop a crude efficiency estimator. If efficiency was judged to fall below the arbitrary figure of 60%, the sample was discarded. They made the observation that catch efficiency generally fell when sampling in high-flow conditions. This crude method for estimating was followed during the surveys undertaken for this thesis and 9 samples had to be discarded taking the sample size down to 40. The locations of these 40 sample sites can be seen in figure 4.1 along with an image of electrofishing taking place on Raydale Beck, near Marsett. The locations are all low-order tributary streams or in the headwaters of the main river stem. Whilst this created difficulties with access, it did direct observation to locations most likely to be exploited as spawning sites by the species. The survey locations varied in type from small first-order streams to larger third-order streams. The surrounding land use varied as did the river habitat. The land use varied from high altitude moorland to improved meadows situated on small floodplains.

Figure 4.1: *the location of the electrofishing sampling sites.*



During each season, nine triple-pass electrofishing surveys using stop nets at the upper and lower reaches of the site were undertaken in order to test the efficiency of the single-run surveys. These were spatially distributed across the catchment in order to test a variety of subcatchments. The method for each of the three runs follows the same as with a single run, except with the addition of the stop nets. There was a wait of twenty minutes between runs and the fish from each were held in separate holding buckets, the largest of these was given to the fish caught on the first run and the water was replenished regularly to minimise the likelihood of stress from low dissolved oxygen levels. Images from the catchment including a number of electrofishing survey sites can be seen in figure 4.2 whilst the locations of the nine triple pass sites can be seen in figure 4.3. The trout data for 2007 and 2008 were transformed into trout fry/m^2 (channel width * survey length) and then an average of the two years was taken. Finally, the average fry density was ranked 1 to 41 in ascending order; rank 41 had the highest trout density.

Figure 4.2: *Locations within the upper Ure catchment. 1) Headwater streams at the upper limit of the catchment, 2) colloidal matter at electrofishing site close to the Moorcock Inn, 3) Erosion on the Grange Beck electrofishing site, 4) Mill Gill electrofishing site in winter, 5) brown trout at Ballowfields electrofishing site, 6) Aysgarth Falls, the downstream limit of the case study catchment, 7) Electrofishing site on Gill Beck, Thornton Rust Moor, 8) Cragdale Beck, Raydale, 9) Raydale Beck, a typical unfenced electrofishing site, 10) Duerley Beck, Sleddale, willow spiling along river bank to slow erosion close to an electrofishing site.*

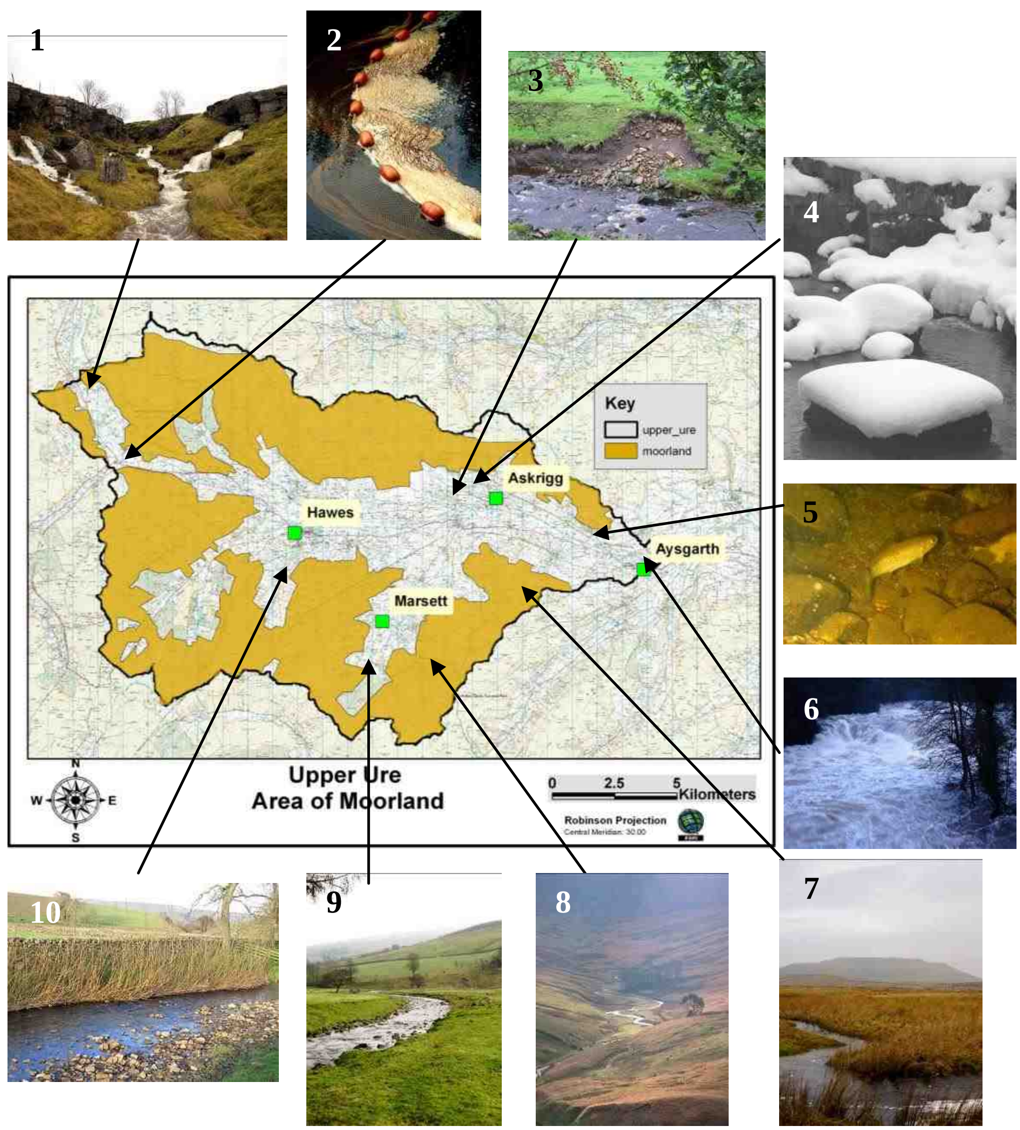
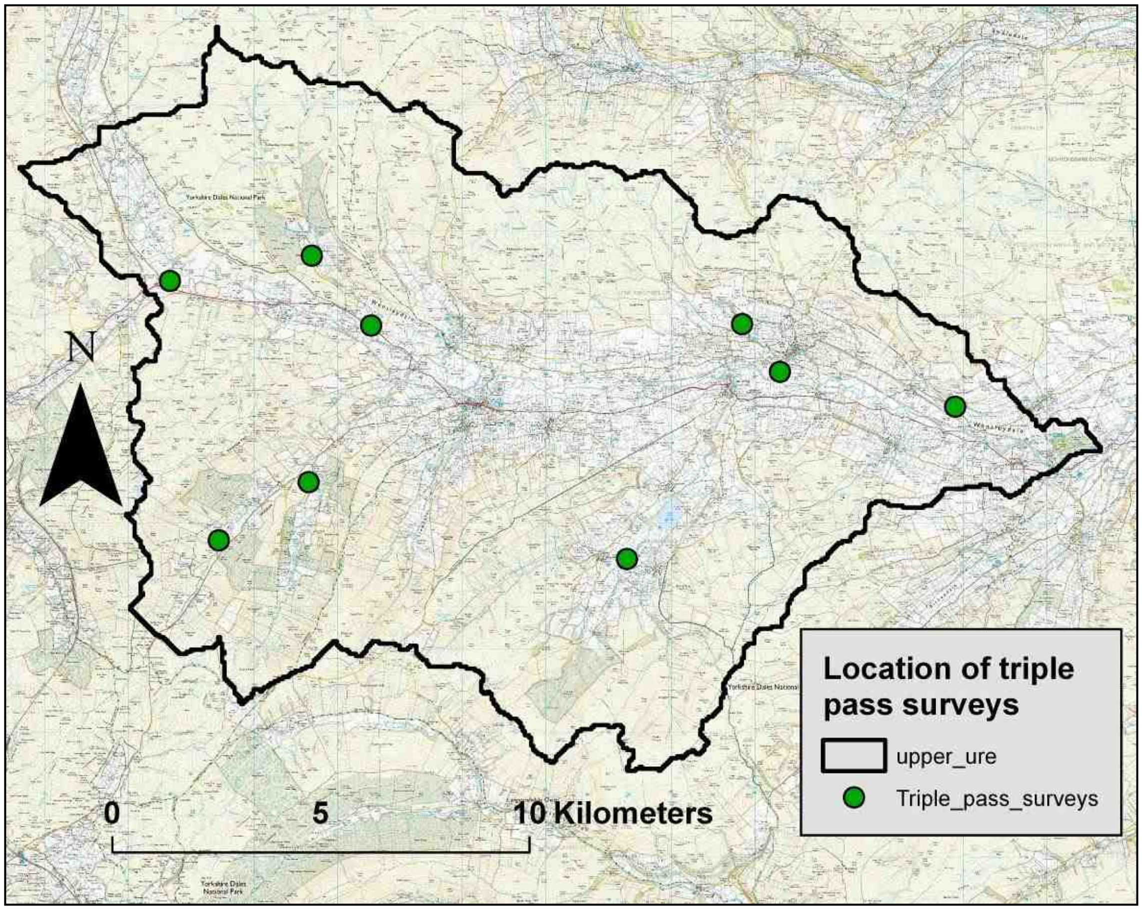
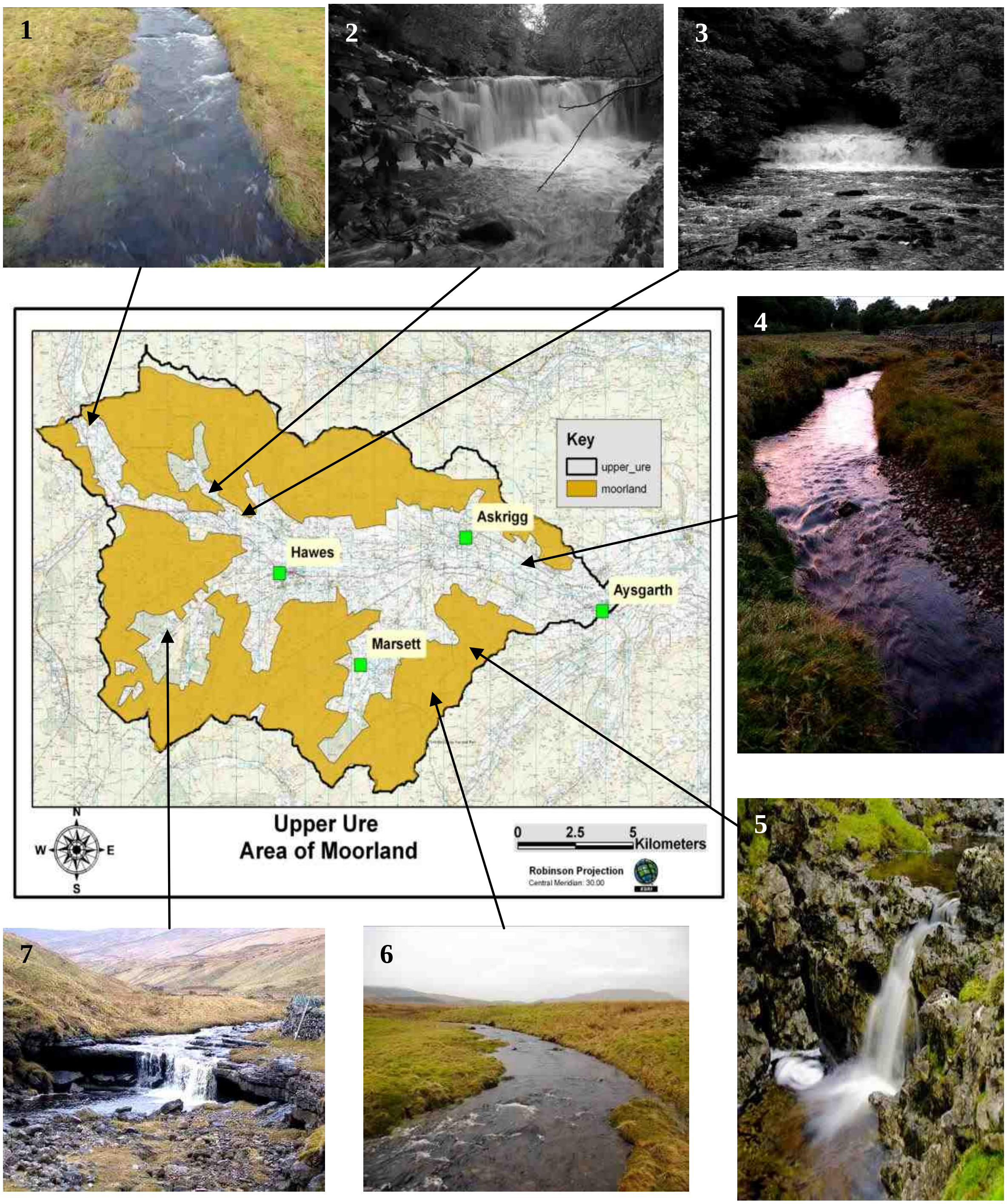


Figure 4.3: *the location of the triple pass electrofishing sites for 2007 and 2008.*



Prior to all electrofishing, surveys water temperature was taken along with specific conductivity, dissolved oxygen, pH, and turbidity. Temperature was important to assess whether the conditions were within a suitable temperature range to carry out the surveys. The other variables were taken to create a snapshot of the local conditions; however, these data were not used in later analysis since without a more complete time series, they would not provide adequate information in relation to brown trout fry. All the sites had gravel beds, or pockets of gravel that could be utilised for spawning but some were cut off from further upstream migration by waterfalls whilst others were open to further upstream migration (figure 4.4).

Figure 4.4: *Electrofishing locations within the upper Ure catchment with and without natural barriers. 1) Open stream close to the headwaters on the main Ure, 2) Waterfall on Cotter Beck, 3) Cotter Force, 4) Open stream, Eller Beck, Ballowfields, 5) Waterfall on Gill Beck, Thornton Rust Moor, 6) Open stream, Gill Beck, Thornton Rust Moor, 7) Waterfall on Bardale Beck, Raydale.*



4.1.2 Sampling macroinvertebrates.

At each of the electrofishing sites that gave a return of usable efficiency, a kick sample was taken to gather information on macroinvertebrate abundance and richness as prey items for brown trout. The kick sampling method is straightforward, standardised, and provides the ability to sample several sites in a day (Beagair and Lair 2007). However, due to the wide distribution of the electrofishing sample sites an average of four sites were sampled per day. As with the electrofishing method chosen, kick sampling is semi-quantitative but has the advantage that it is quick to access all the microhabitats of a survey location and can give relative information on macroinvertebrate communities spatially distributed across the same catchment.

Prior to carrying out the samples, practical training was provided by the EA (Axford, 2007) and a Field Studies Council course was accessed to ensure the correct method was followed. At each survey site the substrate was disturbed by kicking into the stream bed, in order to dislodge the macroinvertebrates, for a period of three minutes. A 1mm mesh hand net was held downstream so that the dislodged organisms would be carried by the current and trapped in the net. The sampling time was split between the microhabitats of each 50-metre sampling site in order to ensure a representative sample was gathered (figure 4.5). The organisms collected were preserved in 90% alcohol for later identification.

Figure 4.5: *Kick sampling for macronvertebrates. Whilst two people are shown here, often the samples taken were carried out alone. The nature of the streams made this relatively safe.*

Identification was carried out with a binocular microscope (Nikon SMZ 2B) using appropriate family keys for ephemoptera (mayfly), trichoptera (caddis fly) and plecoptera (stonefly) as well as generic keys for freshwater macroinvertebrates (Croft 1986; Edington and Hildrew 2004; Elliot 1983; Hynes 1977; Wallace *et al*, 1990). Identification was taken to family level in order to provide an overview of the groups present (abundance/richness) and to create other measures of richness including Shannon and Simpson diversity indices and LIFE scores.

4.1.3 Diversity indices

Both the Shannon's and Simpson's diversity indices were calculated from the family level data to develop information on evenness of macroinvertebrate diversity. The Simpson's index was calculated using:

$$D = \sum\left(n_i(n_i\text{-}1)/N(N\text{-}1)\right) \qquad 4.1$$

where D is dominance, n_i is the number of individuals in the *i*th species (or other taxonomic level) and N is the total number of individuals. In this format diversity decreases as D increases. To make the relationship intuitive, the index has been expressed here as 1/D which now shows diversity (or heterogeneity of community) rising in tandem with increasing values of the index. This index displays the probability of any two individuals from the same sample drawn at random from a community belonging to the same species, or taxonomic level of interest, (Stilling 1992).

The Shannon's index was the second diversity index to be calculated and is calculated using the formula:

$$H' = -\sum p_i \ln\ p_i \qquad 4.2$$

where p_i is the proportion of individuals from the *i*th species, or taxonomic level of interest.

The Simpson's index is biased towards dominance within the community whereas the Shannon's index is biased towards richness and evenness of the sample (Stilling 1992).

4.1.4 LIFE scores

As a surrogate for flow data, the Lotic-invertebrate Index for Flow Evaluation (LIFE) scores as developed by Extence *et al* (1999) were calculated for the macroinvertebrate results. These enable the macroinvertebrate community to be classified according to the prevailing flow conditions based on the average score per taxon, in this instance at the family level. Whilst the LIFE score categories were developed to be attached to taxa at the species level, it is possible to calculate these scores at the family level. However, there are notes of caution with this approach as many families of macroinvertebrates contain species associated with widely different prevailing flow regimes such as the baetidae family of the ephemeroptera order (mayflies) and the nemouridae family of the plecoptera order (stoneflies). Each taxon is assigned a LIFE score based on the flow regime with which it is associated coupled with an abundance rating according to scores shown in tables 4.3 and 4.4. In limestone catchments Extence *et al* (1999) found that it was summer flow variables that had the strongest influences on community structure. The LIFE scores for the upper Ure catchment reflect summer base flows and the 5th percentile high flow events showing that it is flow extremes that are most important to macroinvertebrate communities.

The LIFE score is calculated by assigning scores per taxon from the tables and the following formula:

$$\text{LIFE} = \sum fs/n \qquad 4.3$$

where $\sum fs$ is the sum of individual taxon flow scores for the whole sample and n is the number of taxa used to calculate $\sum fs$. The results show a higher score for taxa related to higher flow conditions.

Extence *et al* (1999) showed that on the river Ure the flow variable that places the greatest control on macroinvertebrate communities is the 5th percentile flow rate. What is interesting here is that this flow variable was shown to be increasing in winter (chapter 3, catchment review) with a slight decrease in the other seasons. However, with the trend aggregated into annual data, there was no trend evident. One off samples such as this will not offer any information on how such changes in flow may alter macroinvertebrate community structure but it is worth noting that community structure, may have been altered due to these changes in the 5th percentile flows.

Table 4.3 and 4.4: *The LIFE score calculation is made from scores derived from both abundance (table 5.2. left) and flow categories (table 5.3 right).*

category	Estimated abundance
A	1 – 9
B	10 – 99
C	100 -999
D	1000 – 9999
E	10000 +

Flow groups		Abundance categories			
		A	B	C	D/E
I	Rapid	9	10	11	12
II	Moderate/Fast	8	9	10	11
III	Slow/sluggish	7	7	7	7
IV	Flowing/standing	6	5	4	3
V	Standing	5	4	3	2
VI	Drought resistant	4	3	2	1

4.1.5 Habitat and riparian variables

A number of habitat variables were collected for each electrofishing site. These included riparian and land use condition. The full range of variables is shown in table 4.5. These were usually collected either prior to or after electrofishing surveys but occasionally on the same day. The variables were measured across the full 50-metre stretch of the survey sites. Substrate was assessed as percent cover of sediment type split into four categories (Table 4.4): bedrock (>4096 mm), boulders and cobles grouped together (64-4095 mm), gravel (2-63mm) and finally fine sediments (sand/silt, 0.0039 - 2 mm). Silt is defined using the Udden-Wentworth classification using grain size of between 0.0039 and 0.063 mm (Naden *et al*, 2000) though a broader definition for siltation is followed here to account for fine sediments including sand particles along with silt and clay. All particles within this size range can quickly fill interstitial pore space of the bed load and so reduce inter gravel flows (Shackle *et al*, 1999). However, nutrients do not attach to sand particles in the same manner as they do to clay particles. Due to this, sand has a physical impact only, whereas finer particles can have a chemical imprint too and be responsible for nutrient enrichment, at least in part.

Bed material was assessed using qualitative methods adapted from the River Habitat Survey method (Environment Agency, 2005): particle size was assigned into one of the four categories by visual inspection combined with measurements using calipers when uncertainty arose.

Table 4.5: *substrate types and size fractions.*

Sediment/bed load type	Particle size (mm)	Method of capture
Bedrock	>4096	Field surveys
Boulders and cobbles	64 – 4096	Field surveys
Gravel	2 – 64	Field surveys
Sand and silt	0.0039 - 2	Field surveys

River width was taken as wetted perimeter measured at three locations within the sampling site (15, 30 and 45 metres); the mean of the three was taken as the river width. Siltation was assigned a measure of 0 when absent and 1 if assessed to be present. If present the percentage of sand/silt had to be >10% and the fine sediments had to have deposited within a matrix of coarser sediments or have smothered the surface of a gravel bed.

The number of pools present within the sample site was recorded. The presence/absence of undercuts and earthcliffs[26] were recorded as 0 (absent) or 1 (present). In-stream algae production was recorded as one of three categories: 1) low levels; 2) moderate levels; and 3) high levels. The extent of algae production in the tributaries was generally lower than in the main river stem which could be quite severe (figure 4.6). The presence or absence of emergent macrophytes was recorded but extent was not assessed; this was generally due poor coverage when present making categories meaningless beyond this. Surrounding land use was recorded in one of five categories: 1) improved grassland, 2) semi-improved grassland, 3) wet meadow, 4) broadleaf woodland and 5) coniferous woodland. Stock access and poaching were both recorded as either present or absent.

[26] In this context an earthcliff is an exposed bank revealing a bare soil surface. In the streams sampled these were not large but may be significant in terms of fine sediment delivery.

Buffer strips were recorded as either 0 (no buffer strip), 1 (buffer strip on one bank) or 2 (buffer strip on both banks). Shading was taken as a percentage of tree cover over a 100-metre length (50 metres on both banks). Shading is important in suppressing photosynthesizing organisms. Hutchins *et al* (2010) found that light levels were more important in encouraging algal growth than nutrients suggesting the main limiting factor in streams is light availability with a secondary limiting factor being phosphate levels. This suggests that locations with greatest percent of shading would be those with the lowest algal growth. Table 4.6 shows the variables and the method for data collection.

Figure 4.6: *At some locations on the main river filamentous green algae is rife during the summer months. This image is taken at Worton Bridge near Askrigg (see figure 4.4 for location of Askrigg).*

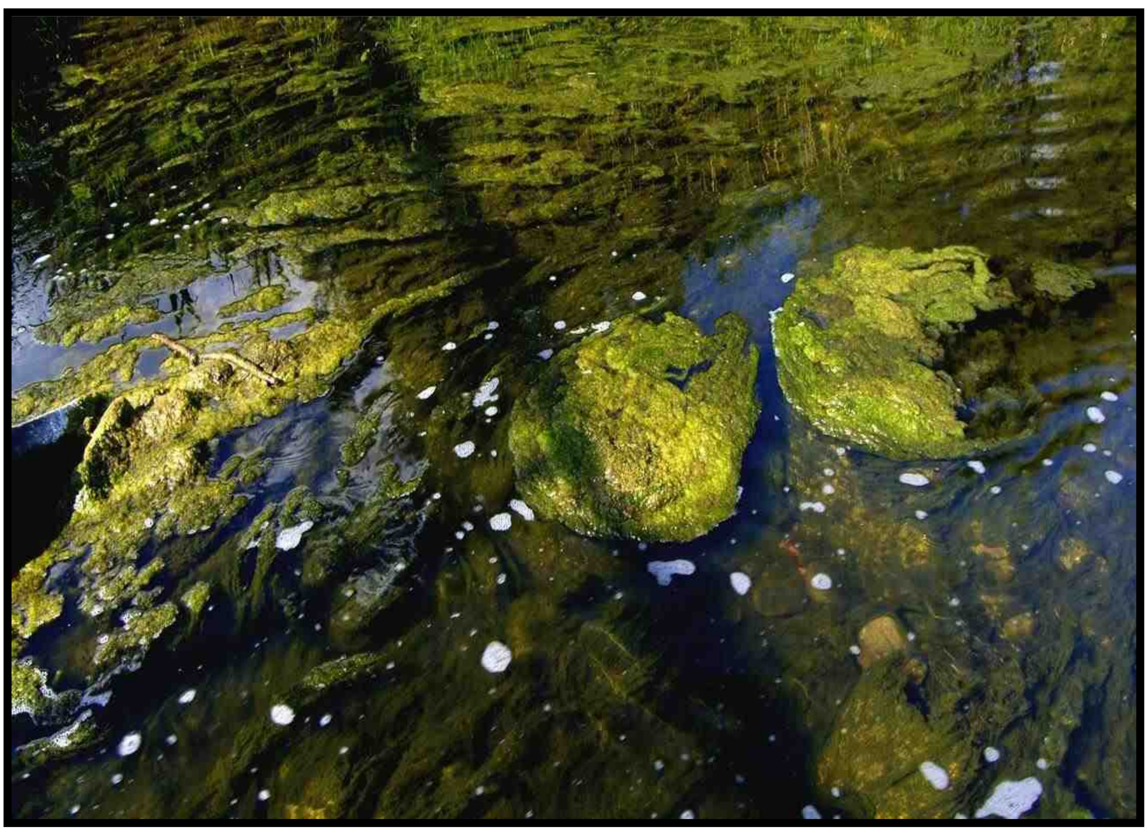


Table 4.6: *The habitat variables collected, the scale at which they operate and the method of capture.*

Factor	Scale operating	Method of capture
Bedrock	Habitat	Field surveys
Boulders and cobbles	Habitat	Field surveys
Pebbles and gravel	Habitat	Field surveys
Sand and silt	Habitat	Field surveys
Siltation	Habitat	Field surveys
River width (m)	Habitat	Field surveys
Pools present	Habitat	Field surveys
Algae (1: low 2:moderate 3:high)	habitat	Field surveys
Macrophytes	Habitat	Field surveys
Undercut	Habitat/riparian	Field surveys
Earthcliff	Habitat/riparian	Field surveys
Stock access	Habitat/riparian	Field surveys
Buffer	Riparian	Field surveys
Land use	Floodplain/catchment	Field surveys
Poached	Riparian	Field surveys
% shading	Riparian	Field surveys

The presence of obstructions to migration (natural or anthropogenic) were recorded within 500 metres and 1 kilometre of the sampling site using OS maps in ArcGIS coupled with field surveys (see table 4.7). In general obstructions were natural but in five locations anthropogenic barriers did exist.

Table 4.7: *Obstructions were identified by OS maps and checked during field surveys to verify they acted as significant barriers.*

Factor	Scale operating	Method of capture
Obstructions upstream (<500m)	Reach	OS maps and field surveys
Obstructions downstream (<500m)	Reach	OS maps and field surveys
Obstructions upstream (<1km)	Reach	OS maps and field surveys
Obstructions downstream (<1km)	Reach	OS maps and field surveys

Drying of streams was recorded in one of three ways: 1) the sampling site being prone to drying; 2) the stream is prone to drying upstream; or 3) the stream is prone to drying downstream (within 1 kilometre). This was recorded through either direct observation or by anecdotal evidence supplied by National Park and EA staff (see table 4.8).

Table 4.8: *Anecdotal evidence was needed to identify which streams were prone to drying. This came from respected EA and YDNPA staff.*

Factor	Scale operating	Method of capture
Survey area prone to drying	Habitat	Anecdotal evidence from National Park and Environment Agency staff and field observations
Stream prone to drying (d/s)	Reach	Anecdotal evidence from National Park and Environment Agency staff and field observations
Stream prone to drying (u/s)	Reach	Anecdotal evidence from National Park and Environment Agency staff and field observations

4.2 GIS, Remote Sensing and Modelling Methodologies

The following sections of chapter 4 will explore how land use across large spatial scales can be described using GIS, remote sensing and modelling technologies. Initially this will be through GIS which will be employed to provide information such as the extent of specific landcover types, upstream contributing areas and stream ordering. Then remote sensing will be expanded specifically to capture risky land management types at the catchment scale. After this the SCIMAP fine sediment model will be explored. This research uses the SCIMAP model in the manner it was initially developed i.e. to model risk at a catchment scale to provide information on the locations delivering fine sediment to streams. It does this by offering relative information on which subcatchments are most likely to be delivering risk disproportionately to the river network. This will be carried out using a Digital Elevation Model (DEM), an erodibility map derived from the CEH Landcover map (2000) and a rainfall map. All of these data sources are available throughout England and Wales. After SCIMAP has been run at the catchment scale remote sensing will be employed to capture other forms of risky land management (in this instance grips) prior to being added to the SCIMAP model. Thus the model will be used in three ways at the catchment scale; 1) weighted by land use, 2) unweighted by land use and 3) weighted by land use with the remotely sensed grips added to the model.

The ability to use remote sensing, GIS and fine sediment modelling in order to elucidate catchment processes and how catchment hydrology and land use combine to deliver impacts to rivers are important advances in the scientific toolbox. The work at the catchment scale allows river systems to be explored in their full spatial context, however finer scales can offer important information. To investigate SCIMAPs ability to describe tighter scales the model will be adapted to the farm scale in order to explore accuracy in offering information at the subcatchment level.

The outputs from these farm-scale runs will be explored with the relevant landowners to ascertain 1) how land managers view the model and 2) the accuracy of the model outputs at this 5m scale by incorporating land managers expertise into the validation process. Thus, this research aims to use local knowledge to extend the peer review community to assist with validating modelled outputs as described above. By doing so the model can be validated in two ways, the catchment scale modelling can be assessed against ecological components of the river system, in this case brown trout fry populations, whilst the farm scale modelling can be assessed against human knowledge of the land. These methods will be used to complement the field data gathered on the ecology and habitat of possible brown trout spawning sites.

4.2.1 Using GIS to explore catchments.

It was considered important, in order to situate each sampling site into its spatial context, to calculate a number of variables including upstream contributing area and upstream area of moorland (table 4.9). In relation to moorland area, this would provide a surrogate for pH in terms of which locations would be most likely to encounter acid flushes during periods of high rainfall (Jutila *et al.* 2003). The next step was to calculate stream orders. It was expected that brown trout fry would be found in the lower order streams of the catchment.

Table 4.9: *The catchment-scale factors captured and manipulated through GIS, remote sensing and modeling techniques are highlighted here.*

Factor	Scale operating	Method of capture
Upstream contributing area (km^2)	Catchment	GIS
Area of upstream moorland (km^2)	Catchment	GIS
Strahler stream order	Catchment	GIS
SCIMAP without grips	Catchment	Modeling and GIS
SCIMAP with grips in DEM and LCM	Catchment	Modeling, GIS and remote sensing
SCIMAP unweighted by land use	Catchment	Modeling and GIS

4.2.2 Calculating upstream contributing areas.

The upstream contributing area for each sampling site was calculated in SAGA GIS using the same method for cutting out the extent of the upper Ure catchment (see step A, appendix 1) up to importing the ASCII layer into ArcGIS. Each of the new topography layers for the sampling sites created was then converted into a polygon shape file. The first step was to convert the topography layer for each sampling site into a layer of two values 0 and 1 using the 'Is Null' function in 'Spatial Analyst Tools'. This sets the extent required to 0 and the surrounding area to 1. To reverse these values, the raster calculator was utilized to calculate '1 –IsNull'. Then in 'Spatial Analyst' 'Options' was opened and in the 'General' tab the 'Analysis mask' drop-down menu was set to the original topography layer to be converted to a polygon shape file. Under the 'Extent' tab, the same topography layer was selected and in the 'Cell Size' tab 'Maximum of Inputs' was selected in the 'Analysis cell size' window. Using raster the calculator the layer was set to the same extent as the original topography layer with a constant value of 1. This was then converted into a polygon shape file by opening the 'Conversion Tools' menu in 'Arc Toolbox', opening the 'Conversion Tools' menu and then 'From Raster' and selecting 'Raster to Polygon'. In the newly opened dialogue box the Input raster was 'Calculation2', the field set to 'Value' and the 'Output polygon features' linked to a file and given the required name. This process was followed for each of the 41 sampling sites.

To calculate the area of the created polygon files, the ‘Spatial Statistics Tools’ menu in ‘Arc Toolbox’ was opened and then ‘Utilities’. Here the ‘Calculate Areas’ was opened. From the ‘Input Features Class’ drop-down menu the newly created polygon was entered and a folder linked to in the ‘Output Feature Class’ and a file name entered. The attribute table of the new layer was opened to show the area in m^2 which was then converted to Km^2.This process was followed for each of the 41 sample sites.

4.2.3 Upstream area of moorland.

With the upstream contributing area for each of the sample sites calculated, it was possible to ascertain the area of upstream moorland in ArcGIS. This was carried out using the moorland shape file for the upper Ure catchment provided by the EA in combination with the topography layer calculated for each of the upstream areas in SAGA GIS. In ‘Spatial Analyst’ ‘Options’ was chosen and the moorland polygon shape file was set as the ‘Analysis Mask’ and under the ‘Extent’ tab the moorland file was again selected. In the ‘Cell Size’ tab ‘Maximum of Inputs’ was selected in the ‘Analysis cell size’ window. The topography layer of the upstream area for one of the sampling sites was entered in the raster calculator and then ‘Evaluate’ chosen. This cut the topography layer down to the extent which intersected with the moorland shape file. This layer was then converted into a polygon shape file and the area calculated following the same method as above. Each of the sample sites was processed in the same manner until the upstream moorland area was calculated for all 41 locations.

4.2.4 Strahler stream orders

The DEM for the upper Ure calculated in SAGA GIS was imported into ArcGIS (ASCII to raster function) and the pits filled (Spatial analyst, hydrology, fill sinks). Then the ‘Flow Direction’ and ‘Flow Accumulation’ were calculated (both located in ‘Spatial Analyst Tools’ and ‘Hydrology’). The symbology tab was opened for the flow accumulation layer. In the left hand workspace of the new dialogue box the option ‘Classified’ was highlighted and then at the far right of the box the ‘Classify’ tab was opened to show the ‘Classification’ dialogue box. Here the number of classes was set to two and the method to manual. In the lower right hand box the top break value was changed to 10000 (to set the limit for channel formation).

The next step was to reclassify the layer. Under 'Spatial Analyst' the 'Reclassify' function was opened. In the 'Reclassify' dialogue box the 'Flow Accumulation' layer was entered into the 'Input Raster' row, the 'Reclass field' was set to value and in the 'Reclassification' table new values of 0 and 1 were set respectively under the 'New Values' column. The output raster was exported as 'Channels_10k'. The final step was then to calculate the Strahler stream orders under 'Stream Order' in 'Spatial Analyst Tools' and 'Hydrology'. The input raster was the newly created 'Channels_10k' and the earlier calculated 'Flow Direction' was entered into the 'Input flow direction raster' row. The 'Method of stream ordering' was set to Strahler. The final stage was to assign the appropriate stream order to each electrofishing site. The electrofishing sites shape file was imported and overlaid onto the stream order layer. Finally the Strahler stream order was recorded for each site.

4.2.5 SCIMAP fine sediment modelling

Landscape processes can have a strong influence on in-stream ecology (Lane, 2008) through a series of process cascades that both transcend scale and are scale dependent. Therefore, explaining how upstream land use coupled with hydrological connectivity affects in-stream ecology is an important consideration when it comes to river restoration (Lane, 2008). Past landscape research has often focused on abiotic metrics of land use and management practices (Reaney *et al*, 2011). As river systems and their ecological components respond to a combination of biotic and abiotic processes there is a need to develop research that addresses the full suite of issues that impact rivers. Carrying out such research is complex due to the nature of the scale interactions and the multiple impacts that arise in agricultural catchments. Moreover, organisms may be mobile and thus linking populations with land use becomes increasingly complex. In order to circumvent these issues, careful selection of a bioindicator is essential to ensure that only those with short ranges and limited dispersal abilities are selected (Lane, 2008). The case for brown trout fry as a bioindicator was set out in chapter 3.

Prior to assessing in-stream organism populations against multiple possible impacts, it is important to develop awareness of the large-scale processes that occur in a catchment and link these to spatial patches where their impact most likely emerges. In order to do this, a number of important considerations need to be assessed. Initially, there is a need to develop knowledge of catchments at a variety of interacting scales ranging from the full catchment, through the subfield scale that capture CSAs down to the local in-stream scale where impacts map out onto in-stream ecological processes. Without modern GIS, remote sensing and modelling tools such objectives may be prohibited by a lack of resources or restricted access to sections of rivers.

In recent years there has been development of a number of modelling tools that offer detail on catchment processes and in-stream ecosystem response. These range from quantitative modelling tools that attempt to describe fluxes of water, sediments or nutrients through a catchment to less complex models that seek to identify locations most probably impacting river systems. The latter approach provides risk-based prioritisation of CSAs within a catchment and has been developed from transfer-function models. Such models approach the issue through risk-based identification of land parcels in contrast to quantifying volumes of stores and fluxes of matter (Lane *et al*, 2006). Modern advances in computer modelling and processing power make it possible to assess the processes operating at the catchment scale whilst also accounting for the finer sub-field resolution thus accounting for process cascades that impact river ecosystems (Mollot and Bilby, 2008). In order to model processes like sedimentation, oxygen uptake, mixing and biochemical decomposition substantial assumptions are required (Cenbrowicz *et al*, 1978) and thus the validity of computer application is reliant on the quality of the initial data (Russell *et al*, 1997). This is a concern when modelling land use as the GIS data sets can become quickly outdated.

Most catchment models aim to follow pathways of diffuse pollution to the end point where impacts occur. Such models rely on data availability to calibrate the model and validate the results. However it is often the case that high resolution data is unavailable and so less complex models become increasingly suitable (Cembrowicz *et al*, 1978). Moreover, taking an inverse modelling approach allows known impacts to be modelled back to the locations in the landscape most likely to be the CSAs. This enables catchments to be modelled from the location of known local-scale problems within the context of the catchment (Dugdale, 2007). The Sensitive Catchment Integrated Modelling and Analysis Platform (SCIMAP) is one such modelling tool. SCIMAP is designed to capture the catchment scale whilst also accounting for the sub-field scale through fine resolution modelling which can be carried out down to 5m using the NEXTMAP DEM (www.intermap.com/elevation-data). This approach allows local, sub-field, hydrological pathways to be followed as they connect CSAs to the stream network (Reaney *et al*, 2010) and so the scales at which these processes matter can be captured and routed through a catchment allowing a probability based risk map that runs from a known impact back through and up to the catchment scale.

The SCIMAP model is based on three sources of data: 1) a Digital Elevation Model, 2) a land cover map converted into a risk of erosion map based on land cover types, and 3) a rainfall map. The NEXTMAP DEM has a resolution of 5m and a vertical precision of ± 1.5m (Intermap, 2003) though it has been reported to be even more accurate in upland catchments, such as the upper Ure, where it has a vertical precision of ± 0.897m (Reid et al, 2007). The NEXTMAP DEM covers England and Wales and was developed using interferometric synthetic aperture radar (IFSAR) technology (Intermap, 2003). The DEM has vegetation and buildings digitally removed to leave only the underlying terrain (Intermap, 2003). The NEXTAMP DEM is an order of magnitude improvement on earlier topographic data such as Ordnance Survey landform PANORAMA 50m DEM. More recent DEM data derived from LiDAR (Light Detection And Ranging) remote sensing has a spatial resolution of 1m and a vertical precision of 1mm (Vaze and Teng, 2005). However LiDAR data was not considered here due to incomplete coverage of the upper Ure catchment. Moreover, the processing capabilities required would outweigh the benefit of such fine-scale resolution.

The LCM data for this thesis is the Centre for Ecology and Hydrology (CEH) Landcover map 2000 and is used as a proxy for agricultural type and other types of land cover including forestry and urban areas. The map estimates land cover at 25m resolution and thus must be re-sampled to 5m to coincide with the DEM resolution using the nearest neighbour algorithm in ArcGIS. The LCM is then converted into an erodibility map by assigning each land cover type at any location a risk value based on the probability of generation of erosion parameterised by expert knowledge (Lane *et al*, 2006). The LCM is synoptic and highly interpolated and thus it most likely misrepresents land cover type (Lane *et al*, 2006); however, it is the best dataset available due to confidentiality of the agricultural census. Therefore, the LCM is considered to provide an adequate surrogate in the absence of other options. Moreover, land use is slow to change in upland regions that are limited by both climate and topography. The assumptions used for converting the land cover map into are an erodibility map is based on the likelihood of erosion occurring and are based on the following assumptions about erodibility:

1) negligible or zero under woodland cover
2) slightly higher on moorland
3) higher again for extensive pasture
4) still higher under intensive or improved pasture
5) significantly higher still for any land use (e.g. arable) where there is a risk of the land being left as bare soil for part of the year.

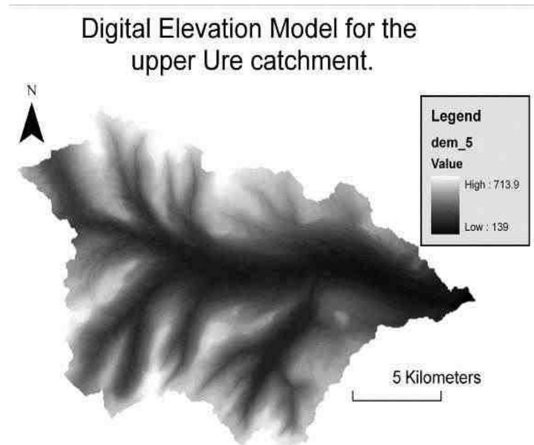
Thus the risk loadings devised using expert knowledge are (Lane *et al*, 2006):

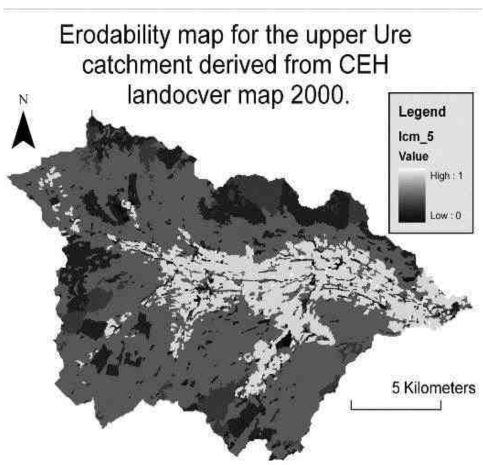
- Horticulture 1
- Arable 1
- Grassland 0.1
- Improved Grassland 0.2
- Heath / peat / bog 0.05
- Woodland 0.00
- Urban 0.00

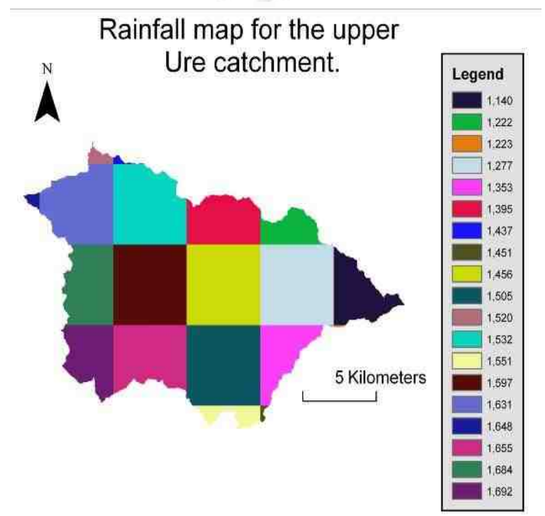
In the study catchment the land use has generally remained unchanged for a number of years and the CEH 2000 landcover map provides a good fit to the on ground reality. However since field work and data collection ended many tracts of coniferous plantation have been felled creating a riskier situation in terms of fine sediment delivery then was the case prior to logging. It is possible to recode the landcover map to take account of such changes and the addition of upland drainage channels to the landcover map here will highlight this.

The final data source is a spatial rainfall map which is derived from the UK Met Office average rainfall dataset covering 1996 – 2000 (Reaney *et al*, 2010). As SCIMAP requires all data to be set to the same extent and resolution, this dataset was also re-sampled from a very coarse 5km resolution down to 5m, again to match the resolution of the DEM. The three primary datasets for the upper Ure are shown in Figure 4.7.

Figure 4.7: *Clockwise from right, 1) Digital Elevation Model showing the altitude range of the upper Ure, 2) the erodability map weighted 0 to 1 and 3) the rainfall map. The lowest section of the study catchment is at Aysgarth Falls on the right whilst the Ure headwaters are just below the N*

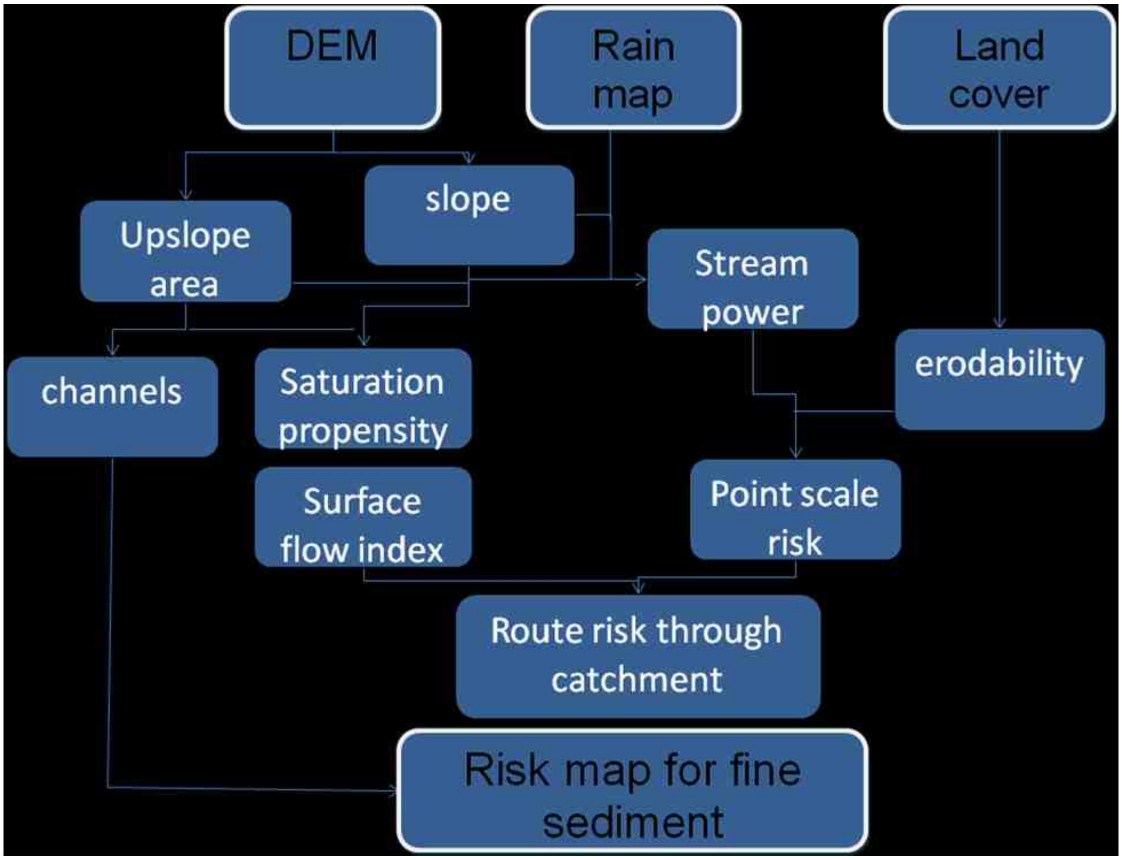






The SCIMAP model was developed collaboratively by Lancaster and Durham universities through interdisciplinary working specifically to bridge gaps between curiosity-driven research and the need for practical tools to enhance the process of river restoration. The ultimate aim is to develop a policy-relevant model that further enables the science of catchment management by identifying locations that are most likely to be degrading river quality (Lane *et al*, 2006). The model is '*based upon a conception of catchments as organising entities; catchments can be conceptualised as a set of flow paths that accumulate distributed sources of possible contaminants from across the landscape into receiving waters where, for surface waters, diffuse pollution may become visible either to routine monitoring through the occurrence of unwanted water quality problems (e.g. algal blooms)*' (Lane *et al*, 2006, p.243). The data flow that SCIMAP follows can be seen in Figure 4.8.

Figure 4.8: *A schematic representation of the SCIMAP model, the top three boxes indicate the initial model inputs.* (Adapted from



As an upland river system that is known to be impacted by eutrophication and fine sediment delivery, the upper Ure catchment lends itself well to the SCIMAP approach, in particular, because research shows that overland flow and shallow sub-surface flow are primary pathways for delivery for these agents into river systems (Walling *et al*, 2002). These forms of pathway in a landscape dominate by meadow and permanent pasture where sheep, beef and dairy are prevalent suggests that fine sediment is more likely to be diffuse pollution issue then herbicides or high levels of chemical fertiliser for example.

The first stage in running SCIMAP is to set each data set to the same resolution and extent. In order to do this the erodibility and rainfall maps were re-sampled to 5m resolution to match the DEM. These initial data preparation stages were carried out in ArcGIS. Each of the primary data sets were cut to the same extent in Spatial Analyst by setting the extent in 'options' before cutting out using Raster Calculator. The land cover map was converted to an erosion map by Dr. Reaney (2009) at Durham University prior to use in this thesis in line with the weightings shown above. This was carried out in ArcGIS using the ‘reclasss’ function within the Spatial Analyst toolbox resulting in a map of relative erosion risk across the catchment weighted 0 to 1 (low to high risk). With the three datasets cut to the same extent and at matching 5m resolution they were all exported as ASCII files from ArcGIS using the Raster to ASCII function ready for importing into SAGA GIS where the SCIMAP platform sits. The three datasets were imported into SAGA GIS using the ‘Import ESRI Arc/Info Grid’ function in Import/Export –Grids. The full SCIMAP process can be seen in appendix 1.The next section will explore SCIMAP outputs to describe the model assumptions and process.

4.2.6 Exploring SCIMAP assumptions and outputs

Reaney *et al* (2011) explain that SCIMAP offers a fresh approach to modelling that allows fine scale representation of the landscape to be explored at a catchment scale. This enables sub-field scale erosion locations to be upscaled to the catchment to identify which are most likely to matter in terms of fine sediment delivery. The basic principle of SCIMAP is if an erosion source is connected to a watercourse by surface flow it provides a diffuse pollution concern. In addition by making a whole catchment comparison it also highlights which catchments are likely to be delivering disproportionate amount of risk. From this it targeting of finite resources is enhanced. The model was developed to contain the most basic information on processes that allows a sufficient exploration of a catchment (Reaney *et al*, 2011; Lane 2009, pers comm.). In particular the two most important processes are erosion and delivery offering information on the likelihood of eroded material reaching a river system.

The model is based on: 1) risk generation for the material that can be eroded (pg); 2) probability of hydrological connection (pc); 3) the combination of (1) and (2) to identify a pollution pathway (pgc); 4) routing of the pathways to ascertain the risk loading (Lj); and 5) transformation of risk loading to risk concentration (Reaney *et al*, 2011). Thus SCIMAP identifies where risk accumulates at a greater rate than dilution. The following will discuss (1) to (5) in turn.

(1) SCIMAP determines fine sediment risk to watercourses through an exploration of the energy required for erosion (hydrological risk: p_i^{H}) and resistance to erosion. In the SCIMAP model this is based on landcover type which is provided a risk weighting (p_i^{lc}). The generation of risk is understood by:

$$p_i^{R} = p_i^{H} \cdot p_i^{lc}$$

The energy required for erosion is positively correlated to the upstream contributing area (A_i) and the local slope (β_i). This is represented by a stream power index (Ω_i) described by:

$$\Omega_i = A_i \tan \beta_i$$

Estimation of p_i^{lc} was developed through expert knowledge and relates to the erodability of landcover types highlighted above (horticulture/arable 1.....Woodland/Urban 0). Reaney *et al* (2011) argue that the specific focus on landcover over soil can be justified due to landcover being generally correlated with soil type and so erodability (see also: Abaci and Papanicolauo, 2009). Figure 4.9 shows the SCIMAP output that describes the relative risk of erosion of the upper Ure catchment.

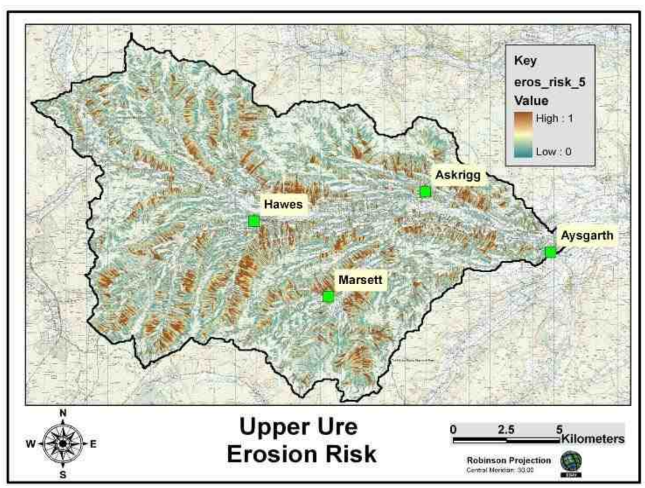


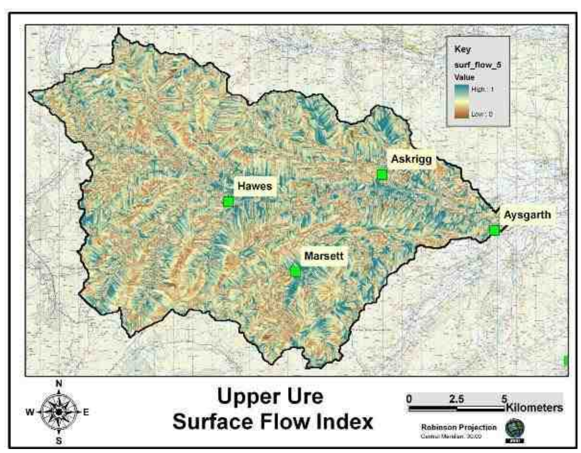
Figure 4.9: *Fine sediment risk is dependent on the propensity to erode here given by an erosion risk loading based on landcover type, rainfall and slope provided by the initial data inputs shown in figure 4.8.*

(2) The probability of hydrological connection is based on the notion that at any point in time there will either be a connection or not (Reaney *et al*, 2011). When the scale of inquiry is expanded there become a series of connected and disconnected erosion sources that respond to slope, upstream area and rainfall patterns. SCIMAP accounts for

these temporal patterns by assuming there will be a spatial pattern of connection strengths across a landscape. It is this connection strength, or likelihood to connect given the prevailing rainfall conditions, that SCIMAP explores. In terms of material carried to rivers by surface flow there has to be a complete flow path from erosion source to river. Where a flow path becomes disconnected prior to reaching a recipient stream then eroded sediments will also become disconnected. Integral to SCIMAP is a treatment of these distributed connections/disconnections through a network index. The network index (Lane *et al*, 2009) identifies the weak points along a connection pathway and identifies these as the controlling factors governing hydrological connectivity of the upslope flow path. These weak points are simply the low values across a topographic wetness index (Beven and Kirby, 1979). Lane *et al* (2009) highlight that the network index is the measure of the likelihood of vertical or lateral flow, lateral flow serves to connect whilst vertical flow disconnects.

These points of connection and disconnection cannot be static. As the landscape wets up a greater number of connections occur, when it dries a greater number of disconnections occur. However any point with a high wetness index has a greater propensity towards connection then one with a low network index. Reaney *et al* (2011) assume a linear duration of connection between the largest (points that are always connected at location i,) and smallest 5% (points that are always disconnected at location i,) of the network index (figure 4.10).

Figure 4.10: *The surface flow (or network) index reveals which surface flow paths are most likely to be connected. This is integral to SCIMAP and reveals important information on hydrological connectivity that is required for understanding fine sediment delivery. The high numbers show the most likely location of strong hydrological connectivity.*



(3) The locational risk is a combination of (1) and (2) and describes the risk of fine sediment delivery to a river network, i.e. there is an erosion source that is hydrologically

connected along a flow path which is governed by a high network index (figure 4.11). This is understood as [illegible]:

[illegible]

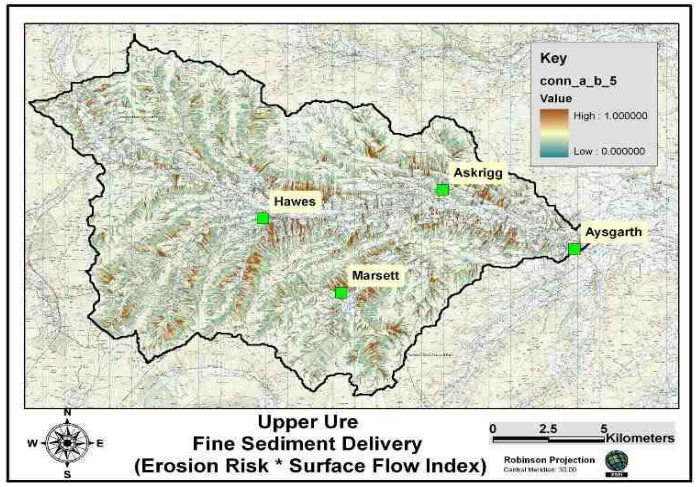


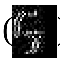
Figure 4.11: *SCIMAP determines fine sediment risk to watercourses through an assessment of likelihood for erosion and hydrological connectivity across surface flow paths. This map shows surface flow as defined by the network index multiplied by erosion risk to show where CSAs are most likely to occur.*

(4) The risk is now routed through the landscape. Surface flow is assumed to be topographically driven. Risk at any given point is considered as the sum of all risk locations in the upslope contributing area. This in itself creates a number of issues in chalk and limestone regions where surface flow occurs in conjunction with lateral and subterranean flow pathways. However during rainfall events surface flow is generated in limestone regions offering the potential for SCIMAP to be helpful for river restoration effort.

From this risk routing the risk loading ([illegible]) is calculated with j being the sum of the upslope contributing area. [illegible] increases monotonically (always increasing and never decreasing or always decreasing and never increasing) as a function of the distance down the drainage network (Reaney *et al*, 2011):

[illegible]

This treatment of the risk loading does not account for dilution. For example a high loading from a location with a small upstream contributing area should be considered to have a greater impact and conversely risk may be lost through dilution. Furthermore

SCIMAP does not take into account the loss of risk through deposition. This is for two reasons 1) there is an assumption that deposition in comparison to delivery is small (this is a common assumption, see: Naden and Cooper, 1999) and 2) SCIMAP is often focused on gravel bed rivers. Owens *et al* (2008) showed that in the River Tweed only 4% of fine sediment was deposited as bed load. Though disposition significantly increases once past the transition between gravel bed to sand bed rivers (Collins and walling, 2007). Reaney *et al* (2011) dealt with the dilution effect through scaling the loading by the upslope contributing resulting in a risk loading per unit area ():

Where a_i is the cell size and r_i is the rainfall weighting factor (Reaney *et al*, 2011). This now offers a treatment of rainfall variation between subcatchments and such variation will increase as a function of basin size. Reaney *et al* (2011, p. 1021) explain that this, '*is represented by weighting upslope contributing areas by the amount of upstream contributed precipitation, using temporal averages*.' This final risk map for the Ure will be presented in chapter 5.

(5) SCIMAP assumes that hydrological connectivity along surface pathways in conjunction with a high erosion risk equates to a diffuse pollution issue. However as Reaney *et al* (2011) highlight there are a number of uncertainties inherent within the model. These include:

1. The determination of the hydrological risk of erosion
2. The relationship between landcover and erodability
3. The relationship between topographic data uncertainty and the network index
4. The scaling between the network index and the delivery index
5. The impacts of topographic uncertainty on flow paths and thus flow and risk accumulation

6. The straightforward manner in which the risk loading is transformed into risk concentration using rainfall weighted upslope contributing area (from: Reaney *et al*, 2011)

Whilst the model offers information on risk how organisms respond to that risk is another source of conflict. For example atlantic salmon may well respond to fine sediment delivery in a completely different manner to brown trout. In addition chironomidae may well flourish where fine sediment enters a watercourse whereas mayfly and stonefly will be negatively impacted on. These different responses require some level of value judgement and often these relate to perceptions of how river ecosystems should behave in certain zones in combination with which organisms have the most economic potential. Here atlantic salmon would win out over brown trout. However in the upper Ure system atlantic salmon are generally excluded by natural barriers.

The manner of all models is to offer simplistic estimates of real world situations and there will always be concerns that they can never fully equate to the complexities of natural systems. However modeling does offer an approximation. Whether that approximation is close enough to reality to offer real insight requires testing. In this work SCIMAP will be utilised at two scales. Here at the catchment scale and in the next section at the farm scale. This offers opportunity to test the model against freshwater organisms and also to farmers' perception and knowledge. The model offers two outputs at the final stage. First the surface flow index multiplied by erosion risk will be assessed at the farm scale and secondly the instream risk concentration will be assessed against brown trout fry populations in conjunction with data collected on habitat condition, surrounding land use and other catchment scale factors.

4.2.7 Capturing risky land management through remote sensing

The CEH Land cover map 2000 captures land management types such as woodland cover, moorland, grassland and arable fields. However, there are other types of land management that occur within these land cover types that may well add risk to rivers through alterations of catchment hydrology and diffuse pollution delivery. Remote sensing offers opportunities to capture these otherwise difficult to view management types. Upland drainage channels on peatlands (grips) are one such management activity that has been shown to add risk to the stream network. To date there has been little information on the coverage and densities of these drainage channels at a catchment scale. Much of the information has been through mapping at the land holding scale for Natural England's Environment Schemes (previously Rural Development Services) or for research purposes which are generally at the sub-catchment scale.

In order to account for these open drains in a catchment-scale assessment of fine sediment impacts, they were first mapped utilising remote sensing techniques. This was carried out in ArcGIS from aerial photographs supplied by the Environment Agency (see Figure 4.12). The aerial photographs were supplied in 5 km grids and spatially referenced to the British National Grid. They gave full coverage of the upper Ure catchment in 5 km grids. The grips were overdrawn as a polyline shape file opened in ArcCatalog and also spatially referenced to the British National Grid. In order to create the file, the photographs were viewed at 0.3 km scale using the 'create new feature' function in the editor toolbar with snapping set to the edges (see Figure 4.13). The photographs were examined systematically concentrating on the hillslopes and moorlands in order to capture the grip at the locations were they exist. Other examples of employing remote sensing and GIS will be explored in Chapter 5 when multiple impacts and processes will be examined in terms of the limiting factors on brown trout fry populations.

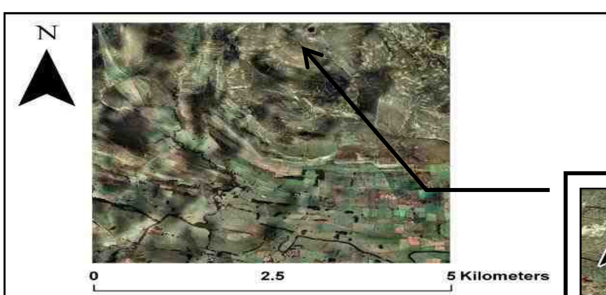


Figure 4.12: *The grip map created by overdrawing from aerial photographs supplied by the EA. The 5 km scale photographs (left) were zoomed in to 0.3 km in order to locate the grip lines.*

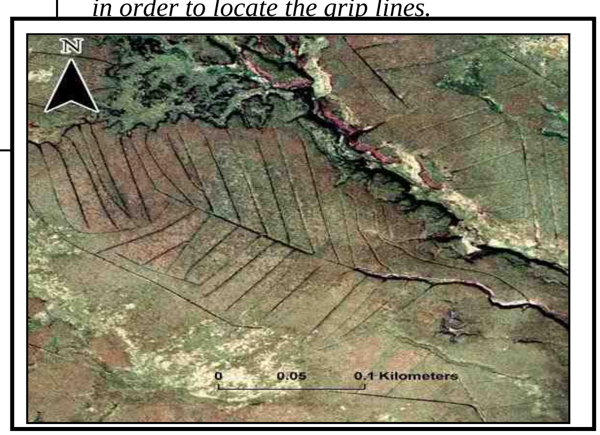
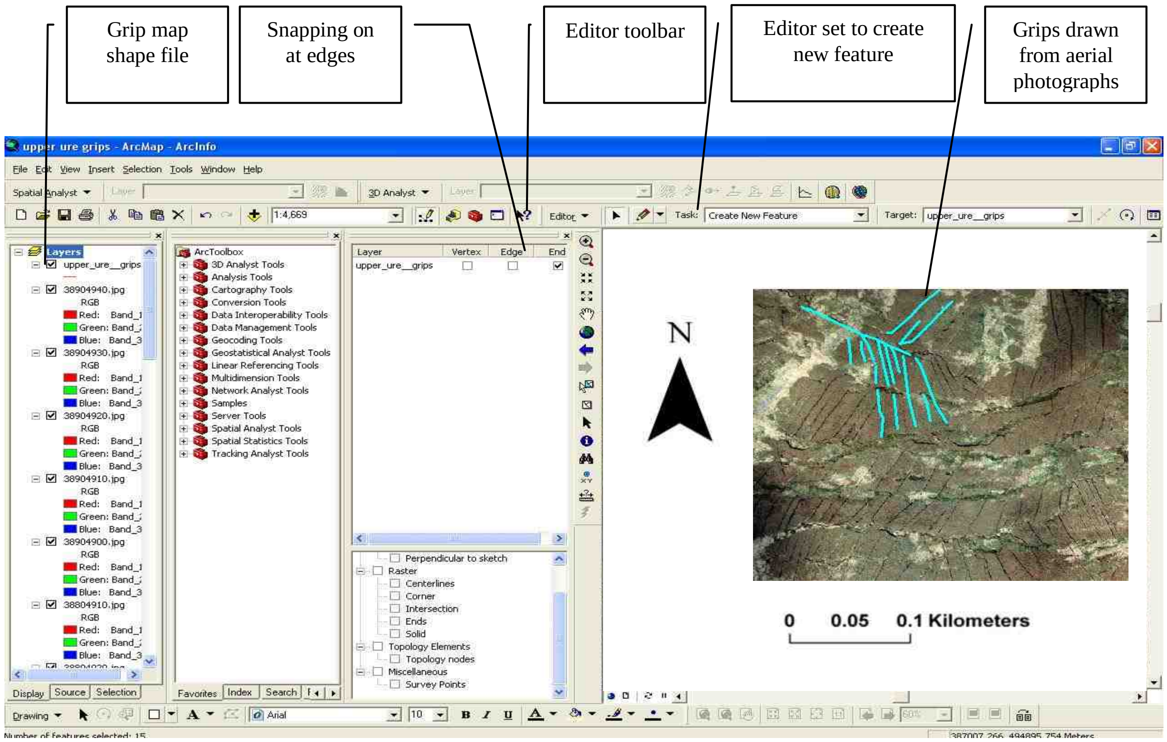


Figure 4.13*: The grip map was created in ArcGIS by opening the shape file in the editor toolbar which was set to 'create new feature' with the grip file set as the target. Snapping was set to the edges to create connected grips as they exist on the ground.*



4.2.8 Accounting for upland drainage in SCIMAP

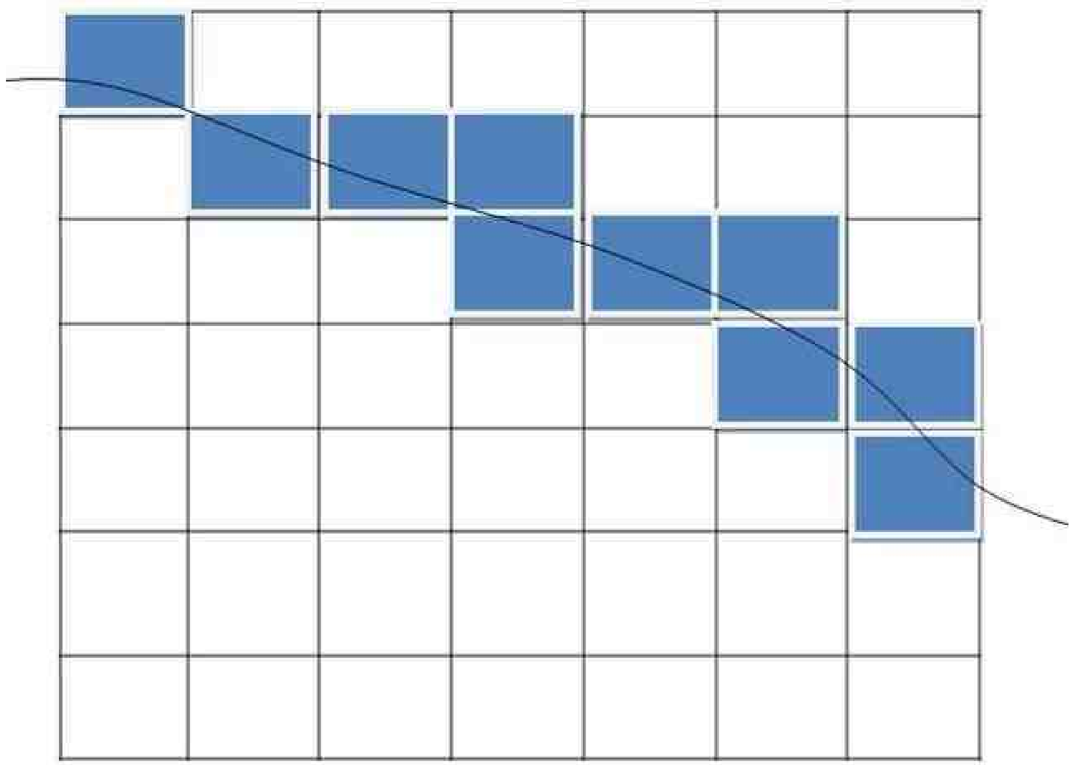
It is possible to include other land management methods that may add risk to the landscape by adding them to the DEM and erodibility map layer and then recoding the risk values to adjust the model in order to account for the on-ground situation. The evidence highlighted above and in Chapter 3 suggests that the effects of upland drainage should cascade through the catchment to affect in-stream ecology at the local habitat scale. This extra risk may arise due to changes in flow rates but it is the alterations in delivery of POM and other fine sediments that are likely to become more prone to erosion that is of interest here. In order to identify how peatland drainage alters the SCIMAP risk category, the grip lines were added to both the DEM (to account for modifications to the drainage network) and the erodibility map in order to upgrade the risk category where these lines cross peatlands. This was carried out in ArcGIS and finalised in SAGA GIS.

The grip map developed in section 4.8.2 overlapped the upper Ure boundary so the first stage was to ensure that only the grips located within the upper Ure catchment were selected. This was carried out by opening the 'selection' menu and choosing 'select by location.’ The grip map was selected as the layer to select features from and the upper Ure catchment was selected as the layer where the grips had to be contained within by ‘are contained by’ option from the drop down menu. The newly created layer was made permanent in the selection tab of the ‘table of contents’ window.

As stated earlier, in order for SCIMAP to work, all layers must be of the same extent and resolution. In order to do this the grips had to be converted to a raster file by expanding the ‘Conversion Tools’ and opening the ‘To Raster’ menu and double clicking the ‘Feature to Raster’ option. In the ‘Feature to Raster’ box the grip layer was the input feature, the field was ‘Id’ and the output feature was given the name Grips_5. The final choice was output cell size and 5m was selected in order to place the grips in the same resolution as the other SCIMAP layers. At this resolution the grips do not replicate the on-ground situation as they become far wider in the model then any of the drainage channels identified on the walkover surveys. However, after this process they are in a format that can provide a relative risk map based on land cover and topography and so it was considered acceptable to run the model based on these coarse assumptions.

The next step was to add the newly created grip raster to the DEM and then the LCM layers. In order to do this, the grips were converted to a value of 0 and the remainder of the layer was given the value of 1 by using the ‘IsNull’ function found in ‘Spatial Analyst’, ‘Math’, ‘Logical’. This created a layer that gave the value of 0 to the grips and a 1 value to the background. This was then added to the DEM using the ‘Plus’ function in ‘Spatial Analayst’. This then raised the DEM by 1 metre except for the location where the line of grips crossed the DEM, which remained the same, thus producing a layer where the grips where reduced by 1 metre in comparison to the rest of the landscape. With each step the assumptions become coarser as when adding the grips to the DEM, for example, the line of each drainage channel becomes a stepped cascade (Figure 4.14). Again, as the SCIMAP model does not aim to quantify the movement of matter but simply provides statements on relative risk, this was considered to be acceptable. As each of the grip lines followed the same assumptions, it could be expected that topographic and land cover risk would be the primary drivers providing a qualitative statement on which sets of grips are the more risky across the catchment, and how this alters the catchment wide risk categories.

Figure 4.14: *When adding grips to the DEM it represents them by converting every square the grip touches into the drainage channel. The black line shows the routing of the grip on the ground, and therefore the route of runoff within the channel, whilst the blue boxes show how this is altered into a stepped cascade as each grid cell the grip line touches is converted into the grip within the DEM layer.*



Once this was complete, the grips were added to the LCM. The newly created 'IsNull' layer had to be reversed by opening the 'Raster Calculator' from the 'Spatial Analyst' menu bar and performing the calculation 1 – [Isnull_featu1] which then gave a value of 1 to the grip lines and 0 to the remainder of the grid. The layer was then made permanent and given the name Grips_Ure5. This layer was then added to the LCM map in 'Raster Calculator' with the calculation [Grips_Ure5] + [LCM_5]. The newly created layer now had a value ranging between 0 and 1.3 as the grip lines crossed a land type with a risk value of 0.3 (whilst the grips had a risk loading of 1). This was converted to a risk loading of 1 in SAGA GIS after the adjusted DEM and LCM were exported as ASCII files for opening into SAGA GIS along with the rainfall map (appendix 1).

The files were then loaded into SAGA as detailed in section 4.8.2. Prior to running the SCIMAP module, the erodibility map had to be converted into an erosion risk range of 0 to 1. This was carried out by opening the 'Change Grid Values' function in the 'Grid – Tools' menu of the workspace. The grid system working under was entered into the

Grid system and the LCM erodibility map was entered into the ‘>> Grid’ and ‘< Changed Grid’ boxes. Under the options choices the ‘Replace Condition’ was set to ‘Low value < grid value < high value’ from the drop down menu. ‘The Lookup Table’ was opened and in the second row the replace value column was altered to 1. This then gave the erosion risk probability range of 0 to 1 as required. Figure 4.15 shows the erodibility map with the grips added.

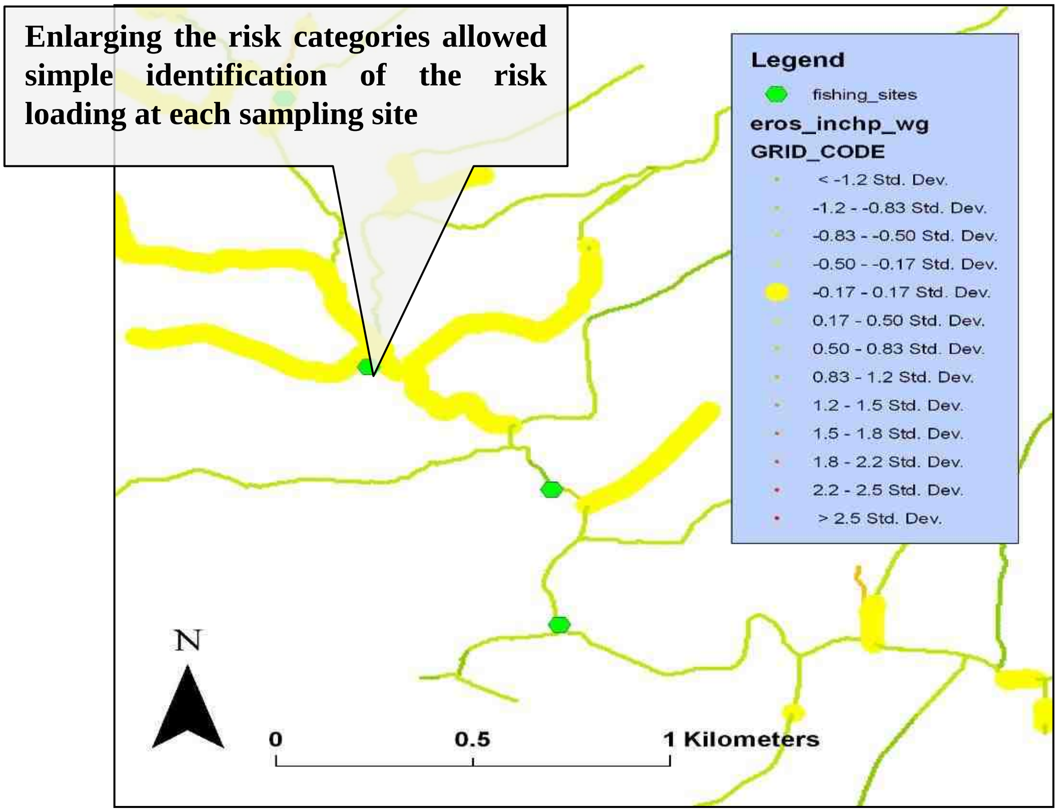
The next stage was to run SCIMAP and export the appropriate outputs to ArcGIS as described in (appendix 1). The visualisation process in ArcGIS again followed the process described in appendix 1 (step C) for the ‘Erosion Risk in Channels Conc.’ and the ‘surface flow * erosion risk’ outputs.

4.2.9 Loading the SCIMAP risk categories on to the electrofishing sites

SCIMAP was run once more on the upper Ure catchment following the method in Appendix 1. However, on this occasion the LCM was replaced by a constant grid calculated in SAGA GIS using the ‘Create Constant Grid’ function under the ‘Grid – Tools’ menu in the workspace. On opening the dialogue box the extent of the DEM and RAIN map was entered into the Grid system window. The base grid was set as the DEM. Then SCIMAP was run as in appendix 1 but only the ‘Erosion Risk in Channels concn’ output was exported. On importing to ArcGIS the layer was converted into a point shape and manipulated as in appendix 1.

The ‘Erosion Risk in Channels concn.’ layer from the SCIMAP outputs with and without grips was also imported into ArcGIS along with the shape file map for the electrofishing sites. Each of the SCIMAP outputs were taken in turn and the risk categories enlarged with the value recorded against individual sampling sites wherever they coincided ranging from 1 (the lowest risk category) to 13 (the highest risk category). This was carried out for all three in-stream SCIMAP outputs (without grips, with grips and unweighted by land use). This can be seen in figure 4.15.

Figure 4.15: *The instream SCIMAP output risk categories were enlarged individually as seen below to ascertain the risk loading for each electrofishing site.*



4.2.10 SCIMAP Modelling at the Farm Scale

Models that provide information on risk within the landscape are useful tools for restorers of natural systems, but their assumptions must be evaluated if they are to be trusted as management tools. Lane *et al* (2006) argue that in data poor, ungauged, catchments testing of models require new approaches. They suggest that: 1) ecological data; and 2) local knowledge; can be used to this end. The reason for developing the SCIMAP model was to provide a management tool with the aim of leading river managers to the locations that have a high likelihood of delivering fine sediment to watercourses. The model highlights land parcels that come with a high erosion risk weighting and then multiplies these locations with a surface flow index to ascertain where CSAs are connected to watercourses and thus delivery of fine sediment that may occur along the surface flow pathway. The assumptions are simplistic, landcover is assigned a risk value (see section: 4.8.2) and surface flow is derived based on lumped rainfall data coupled with topographic data. One of the objectives of the SCIMAP model development was to provide enough information to complete the task and so avoiding over complication by having an analysis with too many parameters.

There are a number of reasons for incorporating local knowledge into the process of model testing. First, as the community under scrutiny from this form of scientific research, farmers have a vested interest in its reliability. Second the model is unable to pick up the level of detail that a land manager sees every day as part of their working life. Third the model cannot account for a number of aspects of land management, such as under-drainage, grazing systems and other methods that can either enhance or reduce erosion risk and so it is necessary to understand the extent to which these impact upon application of SCIMAP in particular situations. Lockertz and Anderson (1990) suggest that farmers have the ability to offer important perspectives and insights to research. In addition to this they argue that many methods of sustainable agriculture were developed through innovation from agricultural communities. This suggests that erosion is a process farmers will avoid and remedy when possible. Whilst the loss of soil from land to water may be perceived as a problem in different ways it remains a concern for ecologists and farmers. Farmers have good reasons for keeping soil in their fields and will adapt management practices to this end (Romig *et al*, 1995).

Forms of participatory methods, that actively seek farmer involvement as part of research and model testing, have become increasingly common (Lane *et al*, 2006). This is important in order to co-evolve understanding of environmental and farming systems. Whilst the knowledge held by farmers has not been formalised in the same way as the scientific process it remains invaluable if it can be captured. Sandor *et al* (2006) comments that knowledge of soil management within agricultural communities has been transferred orally, generation to generation, and so has had many decades, even centuries to evolve to, and adapt with, natural systems. Berry (2002) calls this form of information ‘preserving knowledge’ and believes that in healthy communities it is persistent and adaptive. This traditional form of information transfer is not static; it builds on the past and adapts to changing climatic patterns and, through being situated at a local scale, has become keenly tuned to subtle changes of the environment (Sandor *et al*, 2006).

Despite the knowledge and awareness of sound land management at the farm scale their remains concerns of fine sediment transfer that impact on components of in-stream ecology. Evans (2006) believes this arises from differences in perception between scientists (e.g. ecohydrologists and restoration ecologists) and farmers. There is likely to be a lack of awareness amongst farmers on the impacts to biodiversity arising from the transfer of materiel from land to stream (Evans, 2006). This in itself poses a problem as 'not knowing' invariably means 'not acting'. Although in recent years the shift from policies that pushed production to policies that aim to reduce pressures on environmental systems provides, in combination with payments for environmental modes of management, a good driver for adapting farm methods. Thus if SCIMAP can be validated at the farm scale it has the ability to become a supporting tool for reducing sediment transfers from land to water.

This Chapter will explore local knowledge amongst the farming and land owner community of the upper Ure catchment with a view to evaluating the model outputs at the farm-scale, in so doing it will meet objective 3 described in chapter 1. This part of the model output shows the routing of fine sediment, vie surface flow, across the land towards the channel network. It is a fundamental piece of the SCIMAP model; if this fails to capture the delivery of fine sediment then the in channel assessment of risk becomes less likely to match reality also. Nine farms were visited to explore the model output with the farmer. The farms were selected with assistance from Matt Neale, the National Park area ranger, and through previous visits. They covered the major agricultural types in the Dale, dairy, sheep and beef and covered a range of landcovers from moorland to meadow land. The visits took the form of a brief explanation of the model, semi-structured interviews to ascertain farm management practices and first impressions of the models accuracy and finally a walkover survey of the locations highlighted by the model with the farmer. The next section gives an overview of the results.

4.2.11 SCIMAP at the farm scale

In the absence of raw data for validation of catchment models such as SCIMAP, Lane *et al* (2006) suggest that new modes of validation must be sought. They identify two such possibilities: 1) validating using specially-collected ecological data; and 2) using local landowner knowledge as a form of extended peer review. They also suggest that scientists should be embedded into local communities to bridge gaps in both understanding and awareness of the scientific and farming methods. This thesis has employed all three approaches; the methods to validate the model by tapping into local knowledge will be described here.

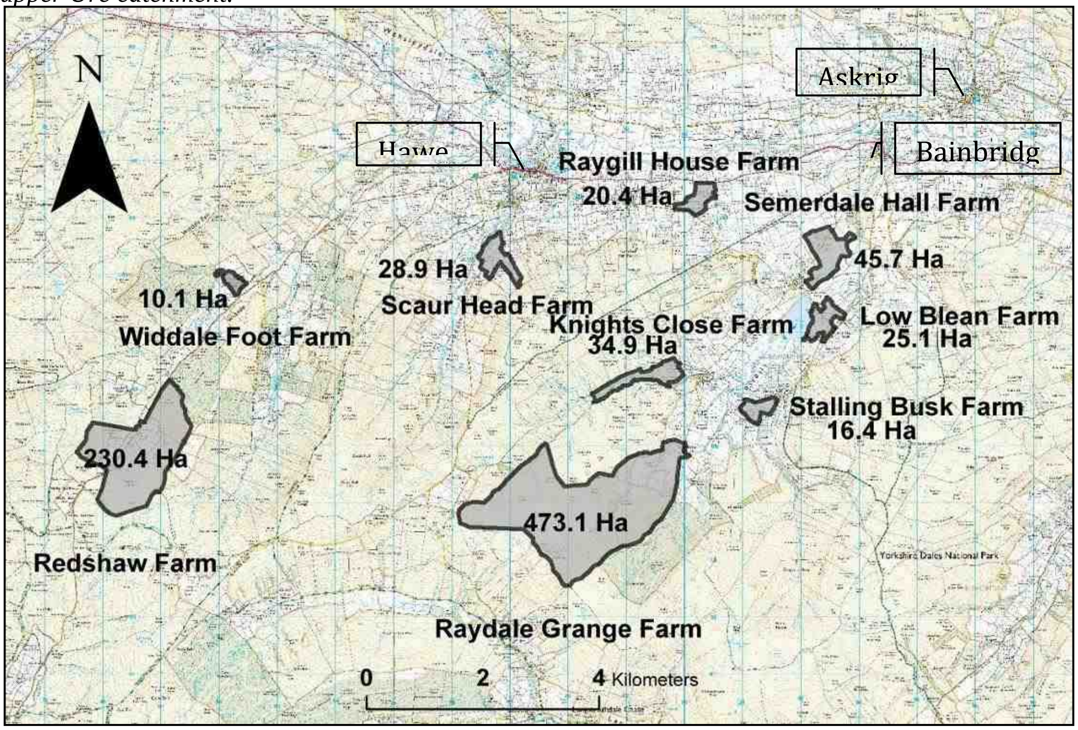
By becoming embedded into the local community, access to land has been made more acceptable to farmers and landowners. By having researchers accessible and therefore not remote at all, trust has been developed. This has been enhanced by good contacts within the park through 1) national park staff and 2) landowning trustees of YDRT and 3) through farming networks after initial contact. This has allowed attempts at validation of the SCIMAP model through the observations of land managers on their own 'patch' and on their own terms i.e. at times suited to their work patterns, on their own land holdings and with their full permission. This access has been invaluable throughout the research and in particular when aiming to validate a catchment model through local expertise.

However, the process has not been straightforward. For example, many land holdings within the upper Ure catchment are spread over a number of small to medium sized land parcels with the larger land holdings typically grouse moors. This has created difficulties when choosing which farms and land parcels to include in this analysis due to issues with locating land parcels of a suitable area and ensuring a mix of management styles. A cut off area of 10 ha was chosen with land parcels below this size excluded from this validation approach. The farms were identified with the assistance of the National Park ranger for Upper Wensleydale, Matt Neale. They land parcels were chosen to provide a variety of sizes and farm types to include dairying, beef cattle and sheep extending from in-bye land to open enclosed rough grazing bordering the moorland line. One land parcel extended into the peatland regions and was the largest land parcel chosen at 473.1 ha. The smallest land parcel was just over 10Ha and could be considered a hobby farm with extensive grazing of hebridean sheep.

Once the land parcels were identified to provide good coverage of the land types of the upper dale (intensive pasture, rough extensive grazing, coniferous woodland and moorland), the landowners were approached. With their permission granted, shape files of the nine land parcels were drawn in ArcGIS (Figure 4.17). These provided the template for cutting out the LCM, DEM and rainfall maps in the same manner as in appendix 1. The raw data for each of the land parcels was cut to the same extent and re-sampled to 5m resolution prior to being exported as ASCII files for importation and processing in SAGA GIS. The same methodology was followed as with appendix 1 to produce risk maps at the farm scale. However, only the connectivity index (surface flow index * erosion risk) for each farm was exported from SAGA GIS for visualisation processing in ArcGIS.

The validation method utilised here is only to test the performance of the model in predicting fine sediment delivery from the land to the stream network. It was considered more appropriate to interview members of the agricultural community only about the land they manage and not approach subject matter on which they likely have less expertise such as how the risk routes through the catchment and into the stream network. Validation of in-stream fine sediment (Erosion Risk in Channels Conc.), which identifies the streams most likely to be delivering disproportionate amounts of fine sediment into the river network in comparison to their upstream area (or streams where the rate of accumulation of fine sediment/risk is greater than the rate of dilution), will be considered separately by assessing this against brown trout fry populations, with and without grips added to the LCM and DEM, along with other, and multiple, impacts on the species.

Figure 4.17: *The land holdings for running SCIMAP at the farm scale are shown in the map below. They hectarage arnged from 10.1 Ha to 473.1 Ha and covered the typical land cover types of the upper Ure catchment.*



4.2.12 Testing SCIMAP

Discovering the weighting between different processes (hydrology, diffuse pollution, habitat variables) that impact on in-stream ecology is only one aspect in the process of river restoration. Incorporating local expertise into the peer review process is important as land managers will understand the land they work in a manner a model is unlikely to replicate (Lane *et al*, 2006). By including local communities in the scientific process relationships between scientific and farming communities are developed and these may assist with future negotiations regarding river restoration. There is always the danger that uncertainties inherent in the scientific method, coupled with the inevitable inaccuracies in the model outputs, may fuel cynicism.

As a piece of embedded scientific research, it was important to develop links and incorporate local knowledge in the validation process, in recognition that such expertise has often arisen through generations of hands on management, thus providing knowledge of the landscape both past and present (Lane *et al*, 2006). Such knowledge is difficult for a researcher to develop over a three-year research project. The farm-scale SCIMAP outputs were thus investigated with the assistance of the relevant land managers in the form of semi-structured interviews and walk-over surveys. As part of this it was felt important to investigate the farm type and the views of the land manager regarding the general ecological condition of the river. From informal discussion with numerous members of the local agricultural community, it had become apparent that farmers had developed a 'siege' position whereby they considered themselves to be under constant, and possibly unjustified, scrutiny regarding the effects of diffuse pollution. Thus for many farmers the default position appeared to be one of mistrust and suspicion. When this is combined with the poor economic situation for many upland farms and the number of crises (BSE, Foot and Mouth) that have affected farming over the previous two decades, a default position of mistrust seems unsurprising.

It must be highlighted that SCIMAP is not a value judgement on selected farmers who happen to have CSAs that connect to the river crossing their land. The model is simply a process of identifying where land parcels are situated that could be targeted for management change due in part (or wholly) to topography and surface flow combining to connect CSAs to watercourses. By seeking to validate such models with those that have the property rights to the land, this enables discussions not only on model parameters but also on how land can be realistically managed in the future to reduce such impacts.

There are some essential protocols for this form of research. When seeking to interview and publish details arising from the interviewing process, the interviewee must be working from a position of informed consent. This in itself may give rise to issues of agenda forming. For example, under informed consent, the interviewee may wish to mislead the interviewer or over-analyse their actions so that the information they impart is closer to how they ought to behave rather than how they behave in reality (Bogdan and Biklen, 1998). In order to overcome this pitfall, the interviews were followed by walk-over surveys with the land managers in order to view the locations that SCIMAP suggests are risky (along with locations the model suggests are less risky). The methods utilised here are semi-structured interviews that cover four key areas: 1) on-farm management and production; 2) perceptions of diffuse pollution within the agricultural community; 3) a simple description of how the SCIMAP model processes information; and 4) validation of the SCIMAP outputs bespoke to each land holding.

There are a number of considerations to be followed when carrying out qualitative methods such as interviewing. Denzin (2001) states that it is important to ensure that the language used during interviews is acceptable, and understandable, to both interviewee and interviewer (Cassell, 2005). This helps prevent opposing interpretations between protagonists. It may be that misinterpretation and assumption forming will always be an issue with the interview process making the need for development of communication skills by the interviewer essential (Cassell, 2005). Widdison (2005 p. 247) argues that '*to be successful, the interviews (need) to produce … honest and frank answers to what (may) be sensitive issues. To achieve this, the interview (should begin) with questions that (are) non-threatening or sensitive, aiming to put the interviewee at ease so that he/she (will) be more willing to answer more sensitive questions later on…When interviewing members of the farming community, it (is) also important that the interviewer demonstrates empathy with farmers and their perceptions of regulations and guidelines whilst maintaining a 'neutral' stance.*' In order to do this, Widdison began interviews with simple situational questions based on farm size and production adapting these questions from questionnaire answers given some time prior to the interviews. Widdison (2005, p 248) discovered that, '*there are barriers and mistrust between 'lay people', 'politicians' and 'scientists'. Each group often believes that the others have something to hide, or deliberately use language that can be interpreted in different ways. In particular, farmers are suspicious of scientific models as they do not always understand the methodology, calculations or even the language used in their interpretation.*' As part of the interviews, the SCIMAP model was explained and a copy of the output with a written explanation was given to each farmer.

Rapley (2001, p. 319) observes that interview extracts, '*should always be presented in the context in which they occurred, with the question that prompted the talk as well as the talk that follows being offered. In this way, readers can view how the talk is co-constructed in the course of the research and, thereby, judge the reliability of the analysis.*' This approach creates transparency of method but it should never compromise confidentiality which was at the forefront of the process. Whilst all the farmers stated that they would be happy to be named in the thesis this was not considered appropriate and anonymity was maintained. This approach has been followed here and a number of extra fail safes have been incorporated. For example all the farmers were given the opportunity to read and comment on their interview and survey write up in order to ensure 1) accuracy, 2) confidentiality and 3) avoidance of misrepresentation. Only one farmer reported concerns and this was with grammar as opposed to content.

Each interview followed a set format:

1) Introductions
2) Questions on farm enterprise
3) Exploration of farmer understanding and thoughts on water pollution
4) Introduction to SCIMAP with an explanation of what the model describes
5) Farmer exploration of the modeled outputs
6) A walkover survey in conjunction with the farmer

This allowed the model to be tested against their initial views and in the field whilst viewing the different parcels of land perceived by the model to be risky or otherwise. The notion of connected erosion was described both in the initial interview and during the walkover surveys. This is a difficult concept to describe as intuitively erosion of banks or standing water on fields became topics of conversation regardless of whether these were eroding or depositing. Careful note was taken of comments that highlighted conflict between stakeholders within the catchment to see if these followed the same concerns highlighted during the catchment review. In addition the degree to which SCIMAP was accurate on different intensity land parcels was noted. There was an initial wish to discuss education levels to explore if this provided any information on levels of mistrust. However this was not followed as it was felt this could create suspicion and appear to be a value judgement. In addition the sample size was not considered large enough to provide pertinent information. However inclusion in environmental schemes was noted as part of the questions on the farm enterprise as this could offer information on environmental awareness and land management.

4.3 Statistical testing of the data

Statistical testing of the data was employed to ascertain which of the variables related and offered a significant explanation of the variance in the brown trout fry data. Pearson’s correlation analysis was carried out twice, once on the data excluding brown trout fry and then again with the inclusion of brown trout fry. After this a stepwise regression was carried out confined only to the data that showed a significant correlation with rank average brown trout fry populations. Those that returned as significant from this then had further stepwise regressions carried out incorporating the data that had returned significant correlations with the variable of interest (e.g. siltation, stock access to streams, etc.). A second stage of stepwise regression was carried out incorporating all the data against rank average brown trout fry. This was to ascertain whether there was a close match to the initial run. The variables that returned as significant (at $P = 0.01$ or 0.05) from this run underwent a secondary stepwise regression analysis with the inclusion of the data that each had displayed significant correlations with from the Pearson's correlation analysis. Each of the catchment-scale SCIMAP runs ($SCIMAP_U$, $SCIMAP_L$ and $SCIMAP_G$) were included in this analysis in order to test whether they offered an explanation of in-stream biota. This offered the opportunity to ascertain if SCIMAP at the catchment-scale could be validated when tested against biological components of an ecosystem that operates at tighter scales but is still expected to respond to overarching processes.

This process offered the opportunity to ascertain the variables that placed controls on brown trout fry populations. Out of the statistical tests the factors that had a limiting effect on fry populations could be identified allowing an assessment of which factors were important and the identification of which mitigation and restoration measures would be the most suitable to lessen their impact. The identification of the appropriate restoration methods was assisted by the literature review undertaken in Chapter two. Thus, the final analysis allows the identification of the controls on fry populations, the linkages and the scale at which they operate. Ultimately this process meets objective four and meets the aim of the thesis.

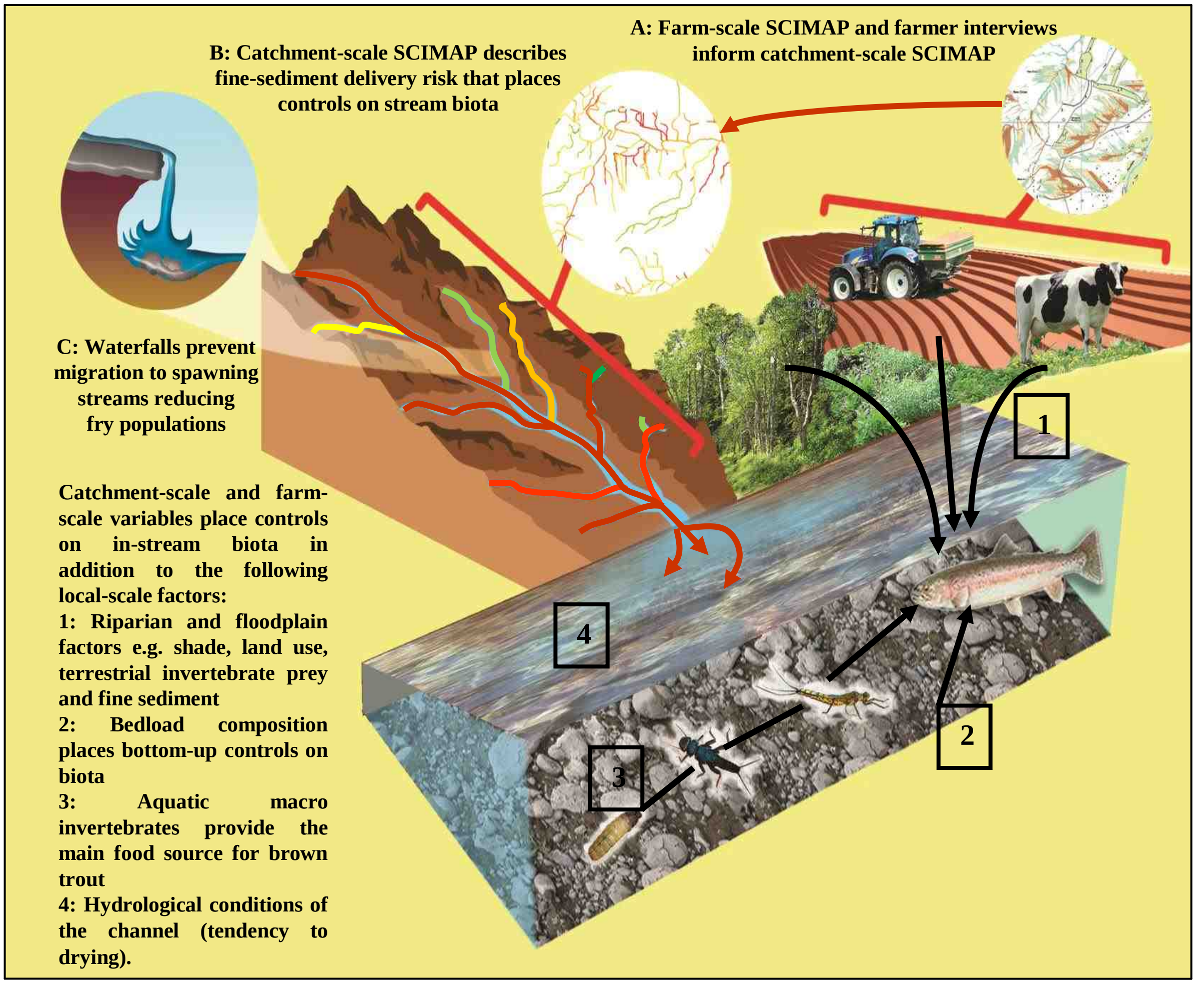
4.4 Preparing to explain the controls on brown trout fry populations

The methods have been devised to explore the linkages between scales and factors that may be placing controls on in-stream biota. Figure ?? describes how these methods link together to inform the investigation into relative brown trout fry populations. There are three distinct scales. 1) the catchment-scale, 2) the field-scale and, 3) the in-stream scale. The farm-scale exploration can be further split into two related areas, 1) the field/floodplain and, 2) the riparian zone, this distinction has been made here. Many of the fields within the catchment are managed up to the river bank however, at some locations there are clear differences in management between the field and the river-adjacent zone.

The SCIMAP model has been run at two scales. The first (A) covers the field and farm-scale and explores the accuracy of the model when describing connected erosion sources. At this scale land managers' knowledge of their holding has been examined in order to understand where the model deviates from field-scale hydrological processes. The second scale (B) assesses landscape-scale hydrological connectivity in particular which streams are delivering disproportionate amounts of fine-sediment into the river network weighed against their upstream contributing area. This catchment-scale exploration investigates the impact of the modelled index of risk delivery on brown trout fry populations. Both these modeled scales inter-relate. The farm-scale exploration has been carried out to understand whether the terrestrial output of the model can be validated in order to offer confidence of the catchment-scale output. In turn the catchment-scale model has been linked to the in-stream-scale.

A central theme of the thesis is to examine the connections between scales and to identify where these connections may place controls on in-stream ecosystems. Thus, other factors within the catchment have to be incorporated into the research in addition to the modelling approach. These factors include riparian management, the intensity of the surrounding land-use, barriers to dispersal and migration situated along the river network (the most apparent of which are waterfalls) and a number of in-stream variables (substrate and ecological components of the system). Furthermore, GIS and remote sensing have been included in order to gather information on difficult to capture variables. These include upstream area of moorland, Strahler stream order and forms of risky land management that are otherwise hard to incorporate into research (in this case the number and extent of upland drainage channels). Figure 4.18 describes the links and highlights the direction of the connections which may be placing controls on brown trout fry.

Figure 4.18: *The links between the methods and data collection.*



The following chapter will explore the meaning behind these results in more detail. It is important to highlight the information gleaned in this chapter and decipher this in the context of the background information gathered on the case study catchment (explored in Chapter 3) and the existing literature discussed in Chapter 2. Importantly, the results need to be assessed to explore how they offer insights into catchment processes and how human interventions skew natural processes to either enhance or negate impacts on river ecosystems. The SCIMAP results will be discussed in terms of how these inform the model development and thus offer potential for exploring catchments from remote locations to enhance our knowledge of catchment science. The perceived disparity in the model's effectiveness at different operating scales will be further developed. This will involve an assessment of the farmer's value judgements when it comes to a difficult and contentious form of surveillant science (Lane *et al*, 2006).

The process cascades introduced in Chapter 2 will be reintroduced to assess how effective these are at offering a simple visualisation of linkages and scale. Importantly, the initial aims and objectives stated in Chapter 1 will be discussed in the context of these results. In so doing, the information gathered here will be used to identify areas where further research is required, or where answers to the research questions failed to materialise. The discussions will be used as a basis to reflect and explore how this research could have been improved to offer a final learning outcome.

5.0 Results

5.1 Introduction

Earlier chapters have set the scene by introducing the aims and objectives and exploring the scientific literature. Chapters 1 and 2 contextualised the work, Chapter 3 developed the case study theme and described the case study catchment and Chapter 4 explained and justified the GIS, remote sensing and modelling approach alongside the more traditional forms of data collection that are central to this work. An attempt was made to validate the modelling approach through the local knowledge base of farmers prior to expanding the SCIMAP model to the catchment scale. The preliminary results will be approached here prior to a full discussion in the following chapter.

This chapter will expand on these earlier sections by displaying the results of the methods prior to exploring brown trout fry in the context of multiple pressures on their populations in chapter 6. This will incorporate the earlier data collection, much of which was gathered remotely, coupled with the SCIMAP modelling work and finally a number of habitat and land use variables, which operate at different spatial scales. The data collection presented has been developed through GIS and more traditional methods in order to capture information that would be expected to place strong controls, either positively or negatively, on brown trout fry survivorship. One of the reasons for this is to highlight that traditional and novel methods for data collection are not mutually exclusive. Indeed, it will be argued that, in order to improve river ecology, such differing methods can complement each other when aiming to understand multiple stressors.

The data encompass multiple impacts and scales, biotic and biotic factors and human interventions, including the detail gathered through SCIMAP in the previous chapter. A number of statistical tests will be examined to develop the data collection in order to elucidate the important relationships. This will begin with a basic correlation matrix prior to expanding the tests to include stepwise regression of the important relationships to decipher a level of weighting between the major controls on brown trout fry populations. Then further stepwise regression will be used to understand the background relationships that govern these controls in order to delve beneath and decipher the linkages and scale effects within the catchment.

It is important in ecological restoration terms to understand the limiting factors on brown trout fry and at which scale these operate. Without this information, future resources targetted at improving the ecological condition for brown trout may well be inefficient, or worse, ineffective. The two scale approaches used for exploring SCIMAP will be used to develop awareness of the risk within the catchment and to link human management in-stream ecological processes.

The following describes the results of the data collection, GIS and SCIMAP modeling. It is presented in four sections. The first (5.2) presents the results of the field data collection, the second (5.3) describes the farm scale SCIMAP work and farmer interviews the third (5.4) provides the GIS and catchment scale SCIMAP modeling result whilst the fourth (5.5) describes the statistical analysis employed to elucidate the relationships between the data forms.

5.2 Field data

The field data were collected in order to gather information on habitat and catchment conditions that may be placing controls on brown trout recruitment and fry survival. The following tables (5.1 to 5.5) show the results of this process. The tables are presented to provide information in related sets, for examples whether the stream is prone to drying at or close to the survey sites, the proximity of up and downstream barriers and substrate type. These variables range in form from very localised controls (e.g. gravel type) moving towards catchment-scale concerns that may be impacting on recruitment from what at first appear to be remote locations (e.g. surrounding land use). They will be presented here in order of proximity to the survey sites beginning with the closest and widening the investigation out into the surrounding land uses.

Table 5.1: *Bedload composition and fine sediment pressures. Data are presented here as % composition and severity of siltation.*

Site	Grid ref	Bedrock (%)	Boulders (%)	Cobbles (%)	Pebbles (%)	Gravel (%)	Sand (%)	Silt/clay (%)	Siltation 0: no issues 1: minor 2: moderate 3: severe
Ballowfields	SD994890	0	5	10	40	40	5	0	0
Cotterdale	SD832938	20	25	25	15	15	0	0	0
Cotterdale	SD833939	20	40	15	15	5	5	0	0
River Ure	SD839916	0	30	20	25	15	10	0	0
Cotterdale	SD834934	20	40	15	15	5	5	0	0
Cotter Force	SD849916	0	20	40	20	10	10	0	0
Widdale (L/S)	SD805850	0	25	30	35	10	0	0	1
Widdale (R/S)	SD805850	0	20	30	30	20	0	0	1
Widdale	SD805852	0	0	30	45	20	5	0	0
Widdale	SD811865	0	20	30	30	25	5	0	0
Widdale	SD812866	0	30	40	15	15	0	0	0
Widdale	SD827879	0	5	10	30	40	10	5	1
Widdale	SD857907	0	10	25	35	20	5	5	1
Sleddale	SD863881	10	5	10	20	40	10	5	1
Sleddale	SD864858	0	25	25	30	15	5	0	0
Sleddale	SD856866	0	35	35	15	15	0	0	0
Raygill	SD913900	0	0	15	30	35	10	10	1
Mossgill Ford	SD830919	0	15	40	25	15	5	0	0
Snaizeholme	SD832872	0	5	15	25	35	10	10	2
Snaizeholme	SD827853	0	20	15	25	40	5	0	0
Snaizeholme	SD825849	0	5	10	10	60	15	0	1
Snaizeholme	SD825847	0	10	10	30	40	5	5	1
Mill Gill	SD914942	10	15	40	25	10	0	0	0
Mill Gill	SD936917	30	20	30	10	10	0	0	0
Grange Beck	SD923914	0	10	25	35	25	5	0	0
Grange Beck	SD933912	0	15	25	25	20	10	5	1
Strands	SD865921	0	20	30	25	20	5	0	0
Paddock Beck	SD946905	0	10	25	30	30	5	0	0
Raydale,	SD904849	0	10	15	35	35	5	0	0
Raydale	SD909862	0	0	30	40	30	0	0	0
Raydale	SD909859	0	10	25	10	50	5	0	0
River Ure	SD786962	0	30	30	25	15	0	0	0
River Ure	SD785956	0	10	20	30	30	10	0	0
Ure, Lunds	SD792945	0	10	20	30	40	5	0	0
River Ure	SD799932	0	5	0	30	30	20	15	2
River Ure	SD799928	20	10	25	30	10	5	0	0
Cotterdale	SD845923	5	30	25	20	20	0	0	0
Thornton Rust	SD969876	0	15	30	25	25	5	0	0
Thornton Rust	SD964875	0	0	0	45	45	10	0	0
Thornton Rust	SD965876	0	0	35	55	10	0	0	0

Table 5.2: *The local conditions and dimensions of each survey site. A number of possible limiting factors are also displayed including algae and earthcliffs which may act to deliver fine sediment either to the survey site or to downstream locations.*

Site	Grid ref	River width (m)	Pools present	Algae: 1: low 2: moderate, 3: high.	Emergent macrophytes	Undercut of bank	Earthcliff
Ballowfields	SD994890	2	4	1	0	1	0
Cotterdale	SD832938	5	2	1	0	1	0
Cotterdale	SD833939	4	1	1	0	1	1
River Ure	SD839916	9	2	2	0	1	0
Cotterdale	SD834934	7	1	1	0	1	1
Cotter Force	SD849916	7.5	1	1	0	0	0
Widdale (L/S)	SD805850	0.25	1	1	0	0	0
Widdale (R/S)	SD805850	0.25	1	1	0	0	0
Widdale	SD805852	1	1	1	0	0	0
Widdale	SD811865	3.5	1	1	0	1	0
Widdale	SD812866	3.5	1	1	0	1	0
Widdale	SD827879	3	1	1	0	1	0
Widdale	SD857907	9	0	2	1	0	0
Sleddale	SD863881	3	2	1	0	1	0
Sleddale	SD864858	3	2	1	0	1	0
Sleddale	SD856866	1	1	1	0	0	0
Raygill	SD913900	1.5	2	3	0	1	1
Mossgill Ford	SD830919	3.5	1	1	0	1	0
Snaizeholme	SD832872	3.5	1	2	0	1	1
Snaizeholme	SD827853	3	1	1	0	1	0
Snaizeholme	SD825849	0.3	1	1	0	0	0
Snaizeholme	SD825847	3	2	1	0	1	0
Mill Gill	SD914942	5	1	1	0	0	0
Mill Gill	SD936917	4	2	1	0	1	0
Grange Beck	SD923914	3.5	2	1	0	0	0
Grange Beck	SD933912	3.5	2	1	0	1	0
Strands	SD865921	3.5	1	1	0	1	0
Paddock Beck	SD946905	3	1	1	0	0	0
Raydale,	SD904849	6	1	1	0	1	0
Raydale	SD909862	3.5	2	1	0	0	1
Raydale	SD909859	6	1	1	0	1	0
River Ure	SD786962	0.5	1	1	0	0	0
River Ure	SD785956	1.5	1	1	0	1	1
Ure, Lunds	SD792945	3.5	1	1	0	1	0
River Ure	SD799932	2.5	3	1	0	1	1
River Ure	SD799928	3.5	2	2	0	1	0
Cotterdale	SD845923	6	1	1	0	1	0
Thornton Rust	SD969876	3	2	1	0	1	0
Thornton Rust	SD964875	0.5	0	0	0	0	0
Thornton Rust	SD965876	2.5	1	1	0	0	0

The habitat variables for the sample sites displayed a range of results with very different substrate compositions although, as expected, the main substrate is coarse sediment with fewer fine fractions, although at a number of sites there were issues with siltation.

Table 5.3: *Likelihood of stream drying out in relation to study site location. Local in-stream flows are an important factor for brown trout fry. Where streams are prone to drying recruitment may be intermittent. Equally, downstream drying will act as a barrier and upstream migration may cut off upstream spawning locations. However the nature of these spate rivers suggests that drying will occur during periods of low rainfall and so the limiting factor would be expected to act in the summer months when fry have emerged from the spawning beds.*

Site	Grid ref	Survey area prone to drying	Stream prone to drying d/s	Stream prone to drying u/s
Ballowfields	SD994890	0	0	0
Cotterdale	SD832938	0	0	0
Cotterdale	SD833939	0	0	0
River Ure	SD839916	0	0	0
Cotterdale	SD834934	0	0	0
Cotter Force	SD849916	0	0	0
Widdale (L/S)	SD805850	0	0	1
Widdale (R/S)	SD805850	0	0	1
Widdale	SD805852	0	0	1
Widdale	SD811865	0	0	0
Widdale	SD812866	0	0	0
Widdale	SD827879	0	0	0
Widdale	SD857907	0	0	0
Sleddale	SD863881	0	0	0
Sleddale	SD864858	0	0	0
Sleddale	SD856866	1	0	1
Raygill	SD913900	0	0	1
Mossgill Ford	SD830919	0	0	0
Snaizeholme	SD832872	0	0	0
Snaizeholme	SD827853	0	0	0
Snaizeholme	SD825849	0	1	0
Snaizeholme	SD825847	0	0	0
Mill Gill	SD914942	0	0	0
Mill Gill	SD936917	0	0	0
Grange Beck	SD923914	0	0	0
Grange Beck	SD933912	0	0	0
Strands	SD865921	0	0	0
Paddock Beck	SD946905	0	0	0
Raydale,	SD904849	0	0	0
Raydale	SD909862	0	0	0
Raydale	SD909859	0	0	0
River Ure	SD786962	1	0	1
River Ure	SD785956	0	0	1
Ure, Lunds	SD792945	0	0	0
River Ure	SD799932	0	0	0
River Ure	SD799928	0	0	0
Cotterdale	SD845923	0	0	0
Thornton Rust	SD969876	0	0	0
Thornton Rust	SD964875	0	0	0
Thornton Rust	SD965876	0	0	1

Table 5.4: *Obstructions in the vicinity of the study site. Obstructions are generally in the form of natural features however at two sites a culverted ford and a weir act as barriers to upstream migration.*

Site	Grid ref	Obstructions upstream, 500m	Obstructions downstream, 500m	Obstructions upstream, 1km	Obstructions downstream, 1km
Ballowfields	SD994890	0	0	0	0
Cotterdale	SD832938	0	0	0	0
Cotterdale	SD833939	0	0	1	0
River Ure	SD839916	0	0	0	0
Cotterdale	SD834934	0	1	0	1
Cotter Force	SD849916	1	0	1	0
Widdale (L/S)	SD805850	1	1	1	0
Widdale (R/S)	SD805850	1	1	1	0
Widdale	SD805852	1	0	0	0
Widdale	SD811865	0	1	0	0
Widdale	SD812866	0	0	1	1
Widdale	SD827879	0	0	0	1
Widdale	SD857907	1	0	0	0
Sleddale	SD863881	0	1	1	0
Sleddale	SD864858	1	0	1	0
Sleddale	SD856866	1	0	1	0
Raygill	SD913900	1	0	1	0
Mossgill Ford	SD830919	0	1	1	0
Snaizeholme	SD832872	0	1	0	0
Snaizeholme	SD827853	1	0	1	0
Snaizeholme	SD825849	1	1	1	0
Snaizeholme	SD825847	1	0	1	0
Mill Gill	SD914942	1	1	1	1
Mill Gill	SD936917	0	1	0	1
Grange Beck	SD923914	1	0	1	1
Grange Beck	SD933912	0	1	1	0
Strands	SD865921	0	0	1	1
Paddock Beck	SD946905	0	0	0	0
Raydale,	SD904849	0	0	0	0
Raydale	SD909862	0	0	0	0
Raydale	SD909859	0	0	1	0
River Ure	SD786962	1	0	0	0
River Ure	SD785956	0	0	0	0
Ure, Lunds	SD792945	0	0	0	1
River Ure	SD799932	0	1	0	0
River Ure	SD799928	0	0	1	0
Cotterdale	SD845923	1	0	0	1
Thornton Rust	SD969876	1	0	0	1
Thornton Rust	SD964875	0	1	1	0
Thornton Rust	SD965876	1	0	1	0

Table: 5.5: *Floodplain and surrounding catchment land use.*

Site	Grid ref	Stock access	% shade	Poached soils	Buffer strips 0 – no buffer strip 1 – on one bank, 2 – on both banks	Land use 1 imp grassland, 2 unimproved grassland 3 wet meadow 4 broadleaf woodland 5 coniferous woodland
Ballowfields	SD994890	0	50	0	0	1
Cotterdale	SD832938	0	10	0	2	2
Cotterdale	SD833939	0	40	0	2	1
River Ure	SD839916	1	15	0	1	1
Cotterdale	SD834934	1	0	0	2	4
Cotter Force	SD849916	1	30	0	2	4
Widdale (L/S)	SD805850	1	0	0	0	1
Widdale (R/S)	SD805850	1	0	0	0	1
Widdale	SD805852	0	15	0	2	2
Widdale	SD811865	1	10	0	0	1
Widdale	SD812866	1	5	0	1	1
Widdale	SD827879	1	0	1	0	1
Widdale	SD857907	0	10	0	1	1
Sleddale	SD863881	1	40	1	0	1
Sleddale	SD864858	1	0	0	0	2
Sleddale	SD856866	1	0	0	0	2
Raygill	SD913900	1	15	1	0	1
Mossgill Ford	SD830919	0	20	0	1	1
Snaizeholme	SD832872	1	0	1	0	1
Snaizeholme	SD827853	1	0	0	0	2
Snaizeholme	SD825849	1	0	0	0	2
Snaizeholme	SD825847	1	0	0	0	2
Mill Gill	SD914942	1	20	0	0	1
Mill Gill	SD936917	1	0	0	0	2
Grange Beck	SD923914	0	35	0	2	2
Grange Beck	SD933912	0	50	0	2	4
Strands	SD865921	1	15	0	1	1
Paddock Beck	SD946905	0	60	0	2	3
Raydale,	SD904849	1	0	0	0	1
Raydale	SD909862	0	80	0	2	3
Raydale	SD909859	1	15	0	1	3
River Ure	SD786962	1	0	0	0	1
River Ure	SD785956	1	0	0	0	1
Ure, Lunds	SD792945	0	10	0	1	5
River Ure	SD799932	1	0	0	0	1
River Ure	SD799928	1	0	0	1	1
Cotterdale	SD845923	0	15	0	1	2
Thornton Rust	SD969876	1	0	0	0	2
Thornton Rust	SD964875	1	0	0	0	2
Thornton Rust	SD965876	1	0	0	0	2

5.2.1 Brown trout fry

The electrofishing surveys were spatially distributed through low-order streams in the catchment to capture appropriate spawning sites, i.e. gravel beds at riffle locations. The nature of the survey meant that each site could capture other habitat types within the 50-metre run, some of which have been presented in the above tables.

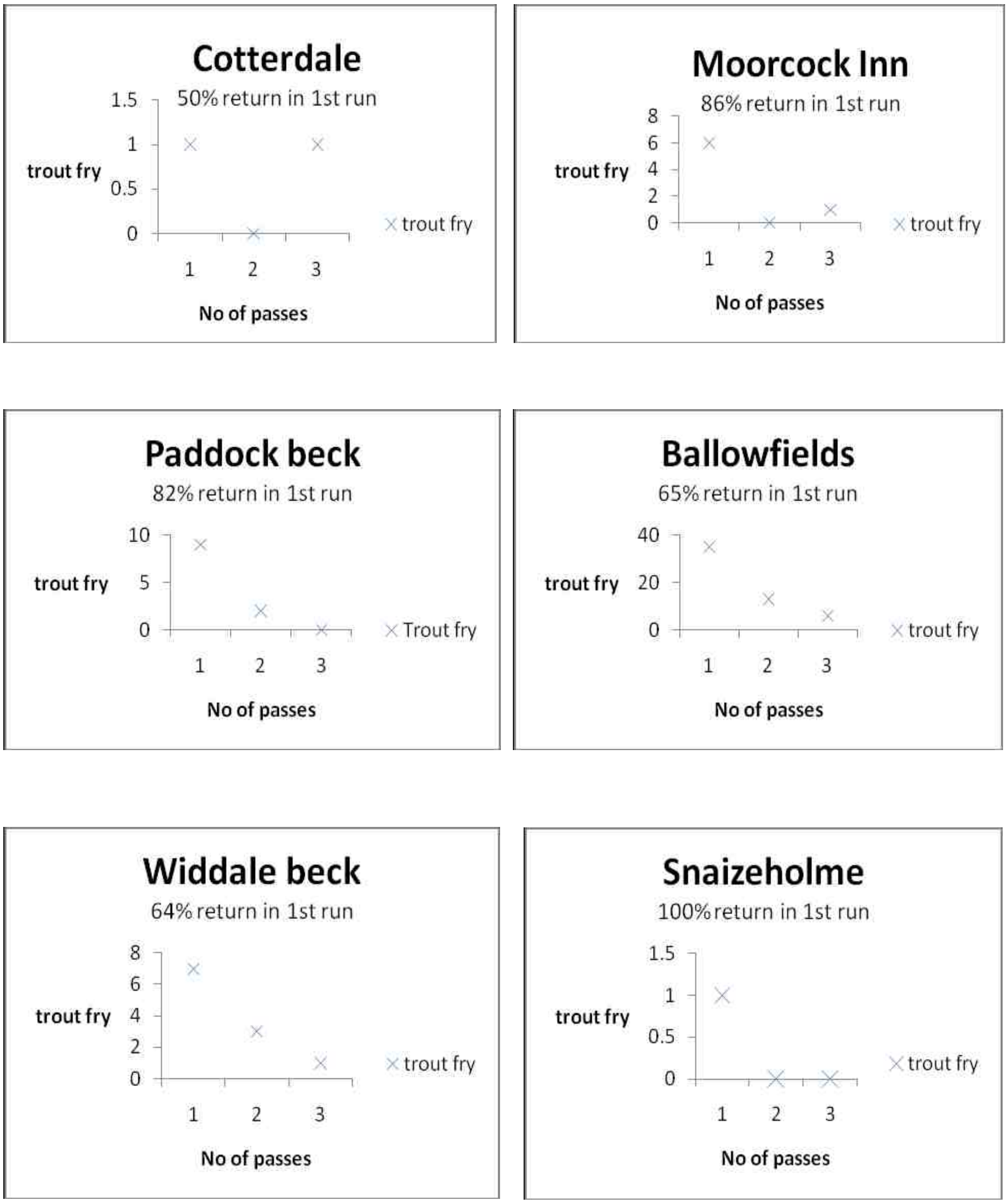
There was a wide distribution in brown trout fry populations with 20 zero returns in 2007 and 19 zero returns in 2008. Fourteen sites showed a zero return in both years. Only five surveys returned a count above 5 in 2007. In 2008 8 sites gave a return above 5. The highest count in both years was collected at the same site (Thornton Rust; SD9690 8765) with 50 fry caught in 2007 and 31 in 2008. The second highest return was at Ballowfields. These sites shared some common factors; however, the Thornton Rust site was located on moorland whilst Ballowfields was just above the floodplain of the main river and drained an area that had been a site of major mining activity in the 19th and 20th centuries. Indeed, the surrounding land had been designated as a Local Nature Reserve due in part to the number of rare metalliferous plants that had responded to the heavy metal concentrations of the water and soils.

The average brown trout fry count in 2007 was 2.63; in 2008 the average count was 2.83. The triple pass surveys that gave a return all resulted in a catch efficiency >60% on the first run. Only one fell below this level reaching 50% efficiency on the first run. Table 5.6 displays the results of the single pass electrofishing runs with the fry counts at each site, the average of both years and finally the rank average fry density. The table has been sorted by rank average fry density. The site names and grid references correspond with Tables 5.1 - 5.5.

Table 5.6: *shows the sampling sites with the electrofishing results sorted by rank average brown trout fry numbers. Due to the number of 0 and low returns this form of data presentation allowed statistical testing.*

Electrofishing site	Grid ref	Brown Trout fry 2007	Brown Trout fry 2008	Average Fry Density	Rank Average Fry Density
Cotterdale	SD834934	0	0	0	7.5
Widdale (l/s)	SD805850	0	0	0	7.5
Widdale (r/s)	SD805850	0	0	0	7.5
Widdale,	SD81866	3	0	0	7.5
Sleddale,	SD856866	0	0	0	7.5
Raygill Syke	SD913900	0	0	0	7.5
Snaizeholme	SD832872	0	0	0	7.5
Mill Gill	SD936917	0	0	0	7.5
Raydale,	SD904849	0	0	0	7.5
Raydale	SD909862	0	0	0	7.5
River Ure	SD786962	0	0	0	7.5
River Ure	SD792945	0	0	0	7.5
River Ure	SD799928	0	0	0	7.5
Thornton Rust	SD965876	0	0	0	7.5
Cotter Force	SD849916	1	0	0.001	15.5
Widdale Beck	SD857907	0	0	0.001	15.5
River Ure	SD839916	1	0	0.0011	17
Cotterdale	SD845923	0	1	0.00165	18
Mill Gill	SD914942	0	1	0.002	19
Sleddale	SD863881	0	1	0.00335	20
River Ure	SD799932	1	0	0.004	21
Cotterdale	SD832938	0	3	0.006	22
Raydale	SD909859	2	3	0.0085	23
Widdale	SD827879	0	3	0.01	24.5
Sleddale	SD864858	0	3	0.01	24.5
Grange Beck	SD923914	2	3	0.0135	26
Cotterdale	SD833939	0	7	0.0175	27
River Ure	SD785956	1	2	0.0198	28
Grange Beck	SD933912	7	1	0.02285	29
Snaizeholme	SD827853	3	4	0.0235	30
Strands	SD865921	0	9	0.025714	31
Widdale	SD811865	9	1	0.0285	32.5
Mossgill ford	SD830919	0	10	0.0285	32.5
Widdale,	SD805852	0	3	0.03	34
Paddock Beck	SD946905	9	5	0.0465	35
Ballowfields Bridge	SD994890	2	8	0.05	36
Snaizeholme	SD825847	11	8	0.063	37
Thornton Rust,	SD964875	3	5	0.16	38
Snaizeholme	SD825849	3	4	0.2335	39
Thornton Rust	SD969876	50	31	0.268	40

Figure 5.1: *Triple-pass electro-fishing results. Many of the triple pass surveys returned a 0 record on all three passes and so have not been included here. As can be seen, the majority that gave a return had >60% trout fry catch on the first run giving confidence in the survey method and volunteer teams. These triple-pass surveys were carried out during the w/c 23/07/2007 with further triple-pass surveys carried out during the 2008 season.*



5.2.2 Macroinvertebrates and diversity indices

At each electrofishing site the macroinvertebrate community was sampled. The sample was removed from site and examined at a later date. Identification was generally taken to family level or, in the case of Coleoptera, to the level of order. As expected, there was a high dominance of mayfly (Ephemeroptera) in the majority of samples with stone fly (Plecoptera) and caddisfly (Trichoptera) being well represented. The families identified across the catchment can be seen in table 5.7.

The flow types associated with LIFE scores can be seen in table 5.8 along with the results of the Simpson's diversity index and Shannon's diversity index. Higher stream flows are associated with higher LIFE scores; this will be discussed further in the next chapter. The Simpson's diversity index expressed as 1/D shows diversity increasing as the value increases and so the lower values show a higher degree of homogeneity, or dominance, in the sample. The Shannon's diversity index provides a measure of richness of the community and the spread between the taxa, or evenness. One sample shows no data. This site was a small calcareous flush flowing into Gill Beck (Thornton Rust moor), where three brown trout fry were caught in 2007 and 5 in 2008. The stream was not sampled for macroinvertebrates as the habitat was of a nature that this level of disturbance would be inappropriate and damaging. Table 5.9 summarises these results.

Table 5.7: *The macroinvertebrate families represented in the samples*

Order	Amphipoda	Coleoptera	Diptera	Ephemeroptera	Neuroptera: sub-order: Megaloptera	Plecoptera	Trichoptera
Family	*Gammaridae*		*Tipulidae*	*Heptageniidae*	*Sialidae*	*Taeniopterygidae*	*Philopotamidae*
			Chironomidae	*Leptophlebiidae*		*Chloroperlidae*	*Psychomyiidae*
			Simuliidae	*Ephemerellidae*		*Leuctridae*	*Rhyacophilidae*
				Baetidae		*Nemouridae*	*Polycentropodidae*
						Perlidae	*Hydropsychidae*
						Perlodidae	*Limnephilidae*

Table 5.8: *Abundance categories and associated taxa characteristics and water velocity* (Extence *et al*, 1999)

Flow groups	Abundance categories				Flow characteristics of Taxa	Associated velocity
	A	B	C	D/E		
I Rapid	9	10	11	12	Taxa primarily associated with rapid flows	Typically \100 cm s^{-1}
II Moderate/Fast	8	9	10	11	Taxa primarily associated with moderate to fast flows	Typically 20–100 cm s^{-1}
III Slow/sluggish	7	7	7	7	Taxa primarily associated with slow or sluggish flows	Typically B20 cm s^{-1}
IV Flowing/standing	6	5	4	3	Taxa primarily associated with flowing (usually slow) and standing waters	
V Standing	5	4	3	2	Taxa primarily associated with standing waters	
VI Drought resistant	4	3	2	1	Taxa frequently associated with drying or drought impacted sites	

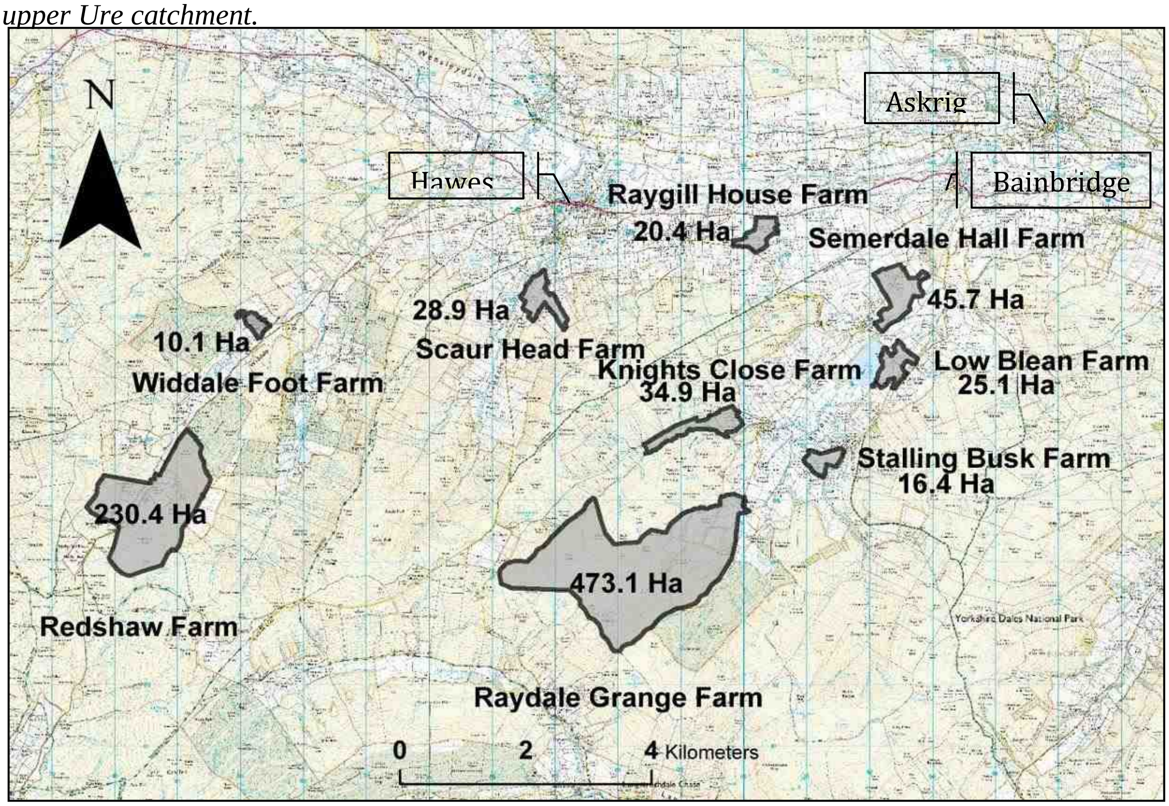
Table 5.9: *Results from the macroinvertebrate sampling and diversity indices*

Electrofishing / survey site	Grid ref	Abundance	Richness	Simpson's Diversity index, 1/D	Shannon's Diversity Index	LIFE scores
Ballowfields	SD994890	548	12	8.330578512	2.243	7.67
Cotterdale	SD832938	244	10	2.735627941	1.34	8.7
Cotterdale	SD833939	229	11	3.13171785	1.419	8.5
River Ure	SD839916	63	11	2.971883614	1.437	8.3
Cotterdale	SD834934	260	10	2.160965278	1.227	9
Cotter Force	SD849916	60	10	4.317073171	1.705	8.75
Widdale (L/S)	SD805850	169	13	6.924878049	2.113	8.41
Widdale (R/S)	SD805850	144	12	5.580487805	1.986	8.45
Widdale	SD805852	169	11	3.759533898	1.676	8.33
Widdale	SD811865	336	13	2.370782257	1.413	8.5
Widdale	SD812866	215	12	2.913500507	1.369	8
Widdale	SD827879	346	14	2.603148988	1.259	8.23
Widdale	SD857907	195	14	2.93301287	1.5313	8.5
Sleddale	SD863881	250	12	3.626776975	1.585	8.5
Sleddale	SD864858	510	13	3.657124341	1.647	9.33
Sleddale	SD856866	470	14	2.494567924	1.3508	8.75
Raygill	SD913900	175	12	6.221904373	2.0535	8.2
Mossgill Ford	SD830919	193	9	3.556238004	1.546	8.63
Snaizeholme	SD832872	232	14	2.367346939	1.317	8
Snaizeholme	SD827853	208	12	2.689319176	1.343	8.67
Snaizeholme	SD825849	277	17	6.924878049	2.113	8.83
Snaizeholme	SD825847	124	13	7.844449005	2.305	8.36
Mill Gill	SD914942	128	6	5.907048799	2.068	8.67
Mill Gill	SD936917	95	12	5.405569007	1.936	7.33
Grange Beck	SD923914	171	13	4.026315789	1.806	9.67
Grange Beck	SD933912	155	13	4.45502053	1.7567	8.18
Strands	SD865921	200	10	3.193196406	1.4984	8.88
Paddock Beck	SD946905	74	13	5.10560113	1.954	8.27
Raydale,	SD904849	275	13	3.356913671	1.635	8.58
Raydale	SD909862	172	11	3.860855868	1.695	8.67
Raydale	SD909859	204	12	3.158810069	1.462	8.27
River Ure	SD786962	173	10	3.117437722	0.645	8.33
River Ure	SD785956	344	12	3.085403483	1.474	8.78
Ure, Lunds	SD792945	374	8	2.105372774	1.005	8.38
River Ure	SD799932	248	10	1.480512226	0.521	8.12
River Ure	SD799928	299	9	3.217375605	1.443	8.12
Cotterdale	SD845923	116	8	1.916013438	1.141	8.85
Thornton Rust	SD969876	246	16	5.756446991	1.312	8.27
Thornton Rust	SD964875			No Data		
Thornton Rust	SD965876	370	10	3.032517436	1.449	8.9

5.3 Farm-scale SCIMAP case examples

The farms selected to assist with testing the model ranged in both size and land use. The least intensive was a small-holding that has very low stocking rates and is managed as a hobby farm at an economic loss. In contrast, the most intensive is run as a sheep and dairy enterprise where the meadows are managed to provide up to three crops of silage each year. This is a very intensive farm operation for this geographical location. All the holdings have watercourses crossing their land; however, only the erosion * hydrological connection was modelled[27]. This was to test, using farmer knowledge and walkover surveys, if the underlying assumptions contained in the SCIMAP model held for this location. The farms are shown in Figure 5.2. The farm visits are presented here in chronological order of visit.

Figure 5.2: *The land holdings for running SCIMAP at the farm scale are shown in the map below. They hectarage arnged from 10.1 Ha to 473.1 Ha and covered the typical land cover types of the upper Ure catchment.*



[27] The SCIMAP model outputs include an index of erosion risk for each land parcel in the catchment and a surface flow index which describes hydrological connectivity. The output from the model used here is where an erosion source is connected to a watercourse by surface flow.

5.3.1: Widdale Foot farm

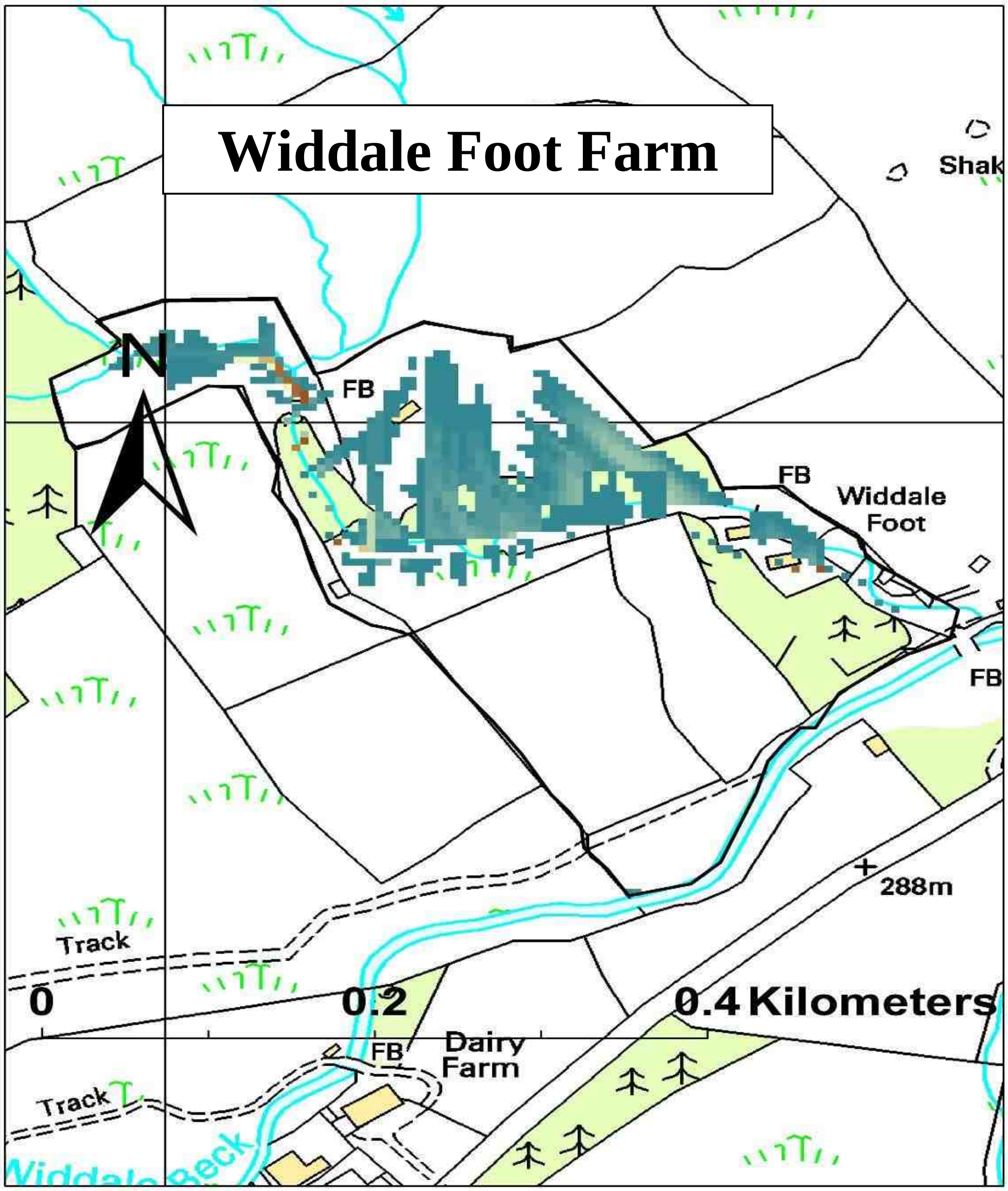
This farm is the smallest of the land parcels modelled at the farm scale. Indeed, this is one of the smaller land holdings in the catchment stretching to just over 10 ha contained within one land parcel. This is fairly unusual in a location where many farms are composed of numerous distinct land parcels. The holding is managed as a small-scale Hebredean sheep enterprise comprising 45 animals including lambs and two tups. The farm is managed by a retired husband and wife (F1) who run the enterprise at an economic loss with many of the sheep being sold to wildlife trusts and other rare breed farms. The farm is situated in Widdale close to Appersett in the upper reaches of the catchment. There are two streams that run through the holding which have been fenced-out and planted with a native tree mix which complements pre-existing mature trees. This management reduces any issues of poaching and helps reduce the risk of fine-sediment delivery as the well-vegetated buffer zone acts to increase friction allowing sediments to settle out prior to reaching the streams. The farm is managed well in terms of biodiversity and environmental considerations. Initially, the land was entered into the Environmentally Sensitive Area Scheme (ESA) for ten years and is now in the Entry Level Scheme (ELS) with a view to entering the new Upland Entry Level Scheme (UELS). It has been under the present ownership for 18 years and prior to this time was part of a larger land holding which had been managed as a highly intensive piggery for over twenty years. During this earlier incarnation the farm had been implicated by the Environment Agency as one of the major causes of the low brown trout populations in Widdale Beck (Frear, 1997). Such was its notoriety that it is still mentioned amongst long-term EA fisheries scientists as one of the reasons why Widdale Beck produces low brown trout stocks.

The present management is far more sensitive to the environment and all of the locations that the SCIMAP outputs identified as being a high risk of fine-sediment delivery are presently excluded from livestock access. Each of these locations was visited with F1 and there was no obvious sign of erosion. F1 commented that '*erosion is not a concern on the farm*' and was clearly proud of his land which was both floristically rich and held rare species including red squirrel with possible sightings of pine marten. He also pointed out a number of pools in the becks that contained trout parr and other locations where the EA had gravel-seeded the channel to encourage trout spawning, although F1 commented that, '*in high flows you can hear the boulders move along the stream bed...I think the gravel wouldn't have lasted long.*'

On visiting the locations that SCIMAP outputs identified as being high risks for sediment delivery (Figure 5.3), it was apparent that the ongoing management of the land holding negated this risk: the land manager had apparently adapted his land use accordingly to ensure risk was minimised. A number of the high-risk locations were contained within fenced-out sections and others were in fields that were so extensively grazed that the grass sward would undoubtedly bind the soils well and slow runoff which would most likely ensure that the risk was not realised. F1 commented that he '*does not use heavy farm machinery and only uses a quad bike to reach the higher elevations on the steep slopes*' so even around farm gates there were no obvious signs of bare soil. Sensitive land management at locations highlighted as being a high risk of fine sediment delivery included low stocking rates, buffer strips and extensive tree planting.

When asked if he considered the risk to be reduced by his management style, F1 agreed this could be the case, though he did add that much of the flow is underground and the majority of surface flow occurs in fields that the model had not highlighted with water springing out of underground channels, '*like artesian wells*.' On viewing these fields, it appeared that these 'artesian wells' were most probably damaged underdrains. The location of these subsoil channels (which were all within the meadow land) suggested that the more productive land has been extensively under-drained, although F1 did not have knowledge of when this might have been done, if this were indeed the case. It was interesting that even in these areas there was no sign of bare soil or erosion but this was most probably due to a combination of low stocking rates and sound environmental management. The walkover surveys discussed in Chapter 3 highlighted that high stocking rates result in poaching and soil erosion on all farms within the catchment. The visit to Widdale Foot offered an interesting contrast to the predominant management which is needed to provide an economic return.

Figure 5.3: *SCIMAP results for Widdale Foot Farm. This is based on a surface flow index multiplied by the erosion risk of each land parcel (5*5 m^2). Here min-max is presented showing low risk of fine sediment delivery as light blues to high risk, brown. The locations with no data are where fine sediment delivery does not occur according to the model's assumptions.*



5.3.2: Raygill House farm

Raygill House farm is purely a dairy enterprise with both Ayrshire and Friesian cattle. The farm is managed by husband and wife partnership (F2) and stretches to approximately 90 ha over two land parcels close to Burtersett. The top moor is kept free from stock and is within the Countryside Stewardship Scheme. An application will be made to Natural England for inclusion in the Higher Level Scheme when the CSS agreement runs its course. Two small streams run across the land, Hunger Hill Syke and Raygill Syke. Parts of Raygill Syke have been fenced to exclude stock and planted up with trees, both native and non-native. Hunger Hill Syke is not fenced-out and stock has ready access to the channel. On the visit, this stream was dry after a prolonged period without rain. It seems probable that there is a water sink upstream of the farm as below the holding the stream was noted to be flowing, albeit slowly.

After viewing the model outputs and having the details explained, F2 stated that he would describe it as '*a mile out and not representing the situation on the farm.*' He continued to explain that the areas where he would have expected risk of erosion to have been highlighted would be on the steeper slopes and in a number of wet fields that the model had not picked up. At this point it was reiterated that the outputs were a combination of erosion risk and surface flow which meant that soil erosion was only picked up by the model if it was connected to a channel. In fields that hold water there was a likelihood that sediments would settle out and steep slopes are only highlighted if they are deemed to directly connect to a watercourse. Many of the steep slopes perhaps ran into less steep fields where hydrological connectivity was severed. Even so F2 was not convinced of its accuracy, maintaining that, '*it did not provide a good fit with reality*.'

On a walkover survey with F2, it became apparent that neither bare soil nor poaching was a concern on the land with a lush grass sward covering the fields. This was despite being a fairly intensive operation. F2 explained that the farm utilised electric fencing to manage stock movement thus excluding stock from areas on a rotational basis, and as and when required, to ensure no piece of land was put under too much stock traffic as '*it's not in my interest to lose soil.*' He further described how the stock was managed and explained that cattle would be housed during prolonged wet periods as well as the more typical period of overwintering indoors. Indeed, the 2009/10 winter had involved an 8-month period of housing the cattle due to the extreme cold weather of the experienced in the catchment which resulted in a delayed growing season.

On visiting a number of locations that SCIMAP highlighted as high risk, there was no erosion or bare soil apparent. F2 described how runoff that followed the most apparent high-risk location, according to SCIMAP, was gathered into a natural sink hole and he '*had not known a time when the drain couldn't cope with the amount of runoff even in severe downpours.*' After two very wet years it seems that the drain is able to convey surface water from the farm; F2 was not sure where it re-emerged. He considered the drain to be a natural feature of the limestone landscape though he did point out that all of the in-bye land, including the steeper slopes, had been extensively under-drained using a V-shaped system with a topping stone, using limestone or sandstone, or rectangular in shape but using the same materials (figure 5.4). This combination of natural limestone sinks and under-drainage could explain the difference between the model outputs and the on-ground situation. When asked if this may be the case, F2 was of the opinion that '*the model was simply wrong and doesn't fit with what actually happens.*' However, observation did suggest that in the absence of the sink hole surface runoff would indeed flow in the pattern SCIMAP suggested and clearly, if this were the case, it would connect with Hunger Hill Syke.

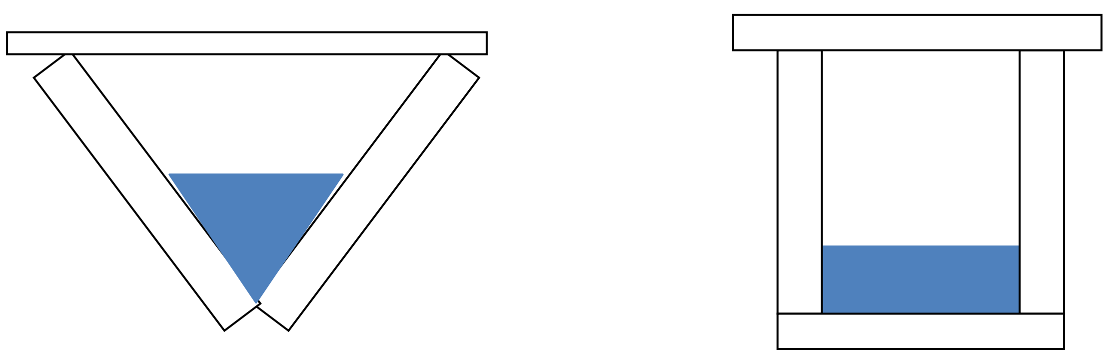
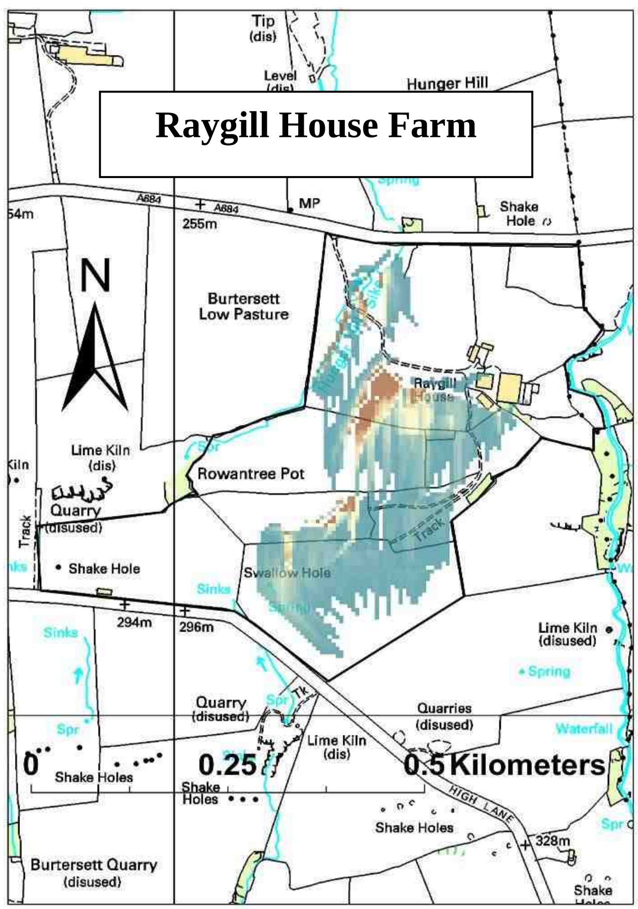


Figure 5.4: *the drains of meadowland in the upper Ure catchment are designed in natural stone in either a v–shape or rectangular design. All the meadowland is extensively under-drained and pasture land has been drained using open drain systems similar to small grips.*

A second location where SCIMAP may have given a good fit with reality was along the banks of Hunger Hill Syke. As noted above, this stream was unfenced and stock had ready access to the banks and channel bed; indeed, they congregated in this location during the visit. Some bare soil was noticeable here and surface runoff would have resulted in fine-sediment delivery at a number of locations, though the grass sward away from the immediate riparian was generally good.

It may be that the combination of under-drainage, natural drainage and careful stock management (on the whole) resulted in fine-sediment delivery generated by surface flow not being an issue on the farm and that under less sensitive management, the SCIMAP outputs would fit with observed erosional processes. However, when asked if such a scenario would provide this result, F2 remained unconvinced. There was clearly an issue with bank erosion on Hunger Hill Syke due to stock access and in high flows this may pose a significant stress on downstream ecosystems, at least if this was combined with other sources of fine sediment from downslope farms. Figure 5.5 shows some of the SCIMAP outputs for the farm.

Figure 5.5: *SCIMAP results for Raygill House Farm. The outputs are based on the same scale and form as with figure 5.3.*



5.3.3 Low Blean Farm

Low Blean Farm is managed as a family enterprise (F3) in conjunction with a second farm, Leas House Farm, near Askrigg. Low Blean Farm is located in Raydale and runs down to the shores of Semerwater Lake. The farm is a mixed livestock enterprise with 20 dairy cattle, sheep and beef. In total, between the two farms, there are approximately 100 cattle including calves. Presently, the farm is not in a stewardship scheme but there is an application being prepared for entry into the Upland Entry Level and Higher Level Schemes. The soil in many of the in-bye fields had recently been tested and as a result less chemical fertiliser had been applied to the soil and a dose of lime had been spread to raise the pH. F3 explained that the farm had been in possession of the family since the 1940s although it had been leased out for a number of years prior to the family taking direct control of the enterprise. Two streams cross the land, Little Ing Sike and a second unnamed stream. Both of these flow into Semerwater Lake which the farm adjoins. A third stream has been wholly culverted and does not show on present OS maps. However, it is shown on maps prior to 1940, suggesting that this stream was probably modified sometime in the 1940s.

The model was explained, concentrating on how the data are collected and how the SCIMAP output at the farm scale is a combination of erosion risk and a surface flow index and so describes where fine sediment may be delivered to watercourses. After the visit to F2, a more detailed explanation and description of the model was thought necessary as part of the initial discussion. On first viewing the SCIMAP outputs, F3 highlighted a number of areas where the outputs suggested fine sediment was delivered to watercourses and stated that, '*if anything, it deposits there.*' He also noted a few locations where SCIMAP had not highlighted risk and expressed surprise as these were '*very wet fields that often have standing water on them.*' It was suggested that this may result in sediments depositing out as opposed to connecting to the lake.

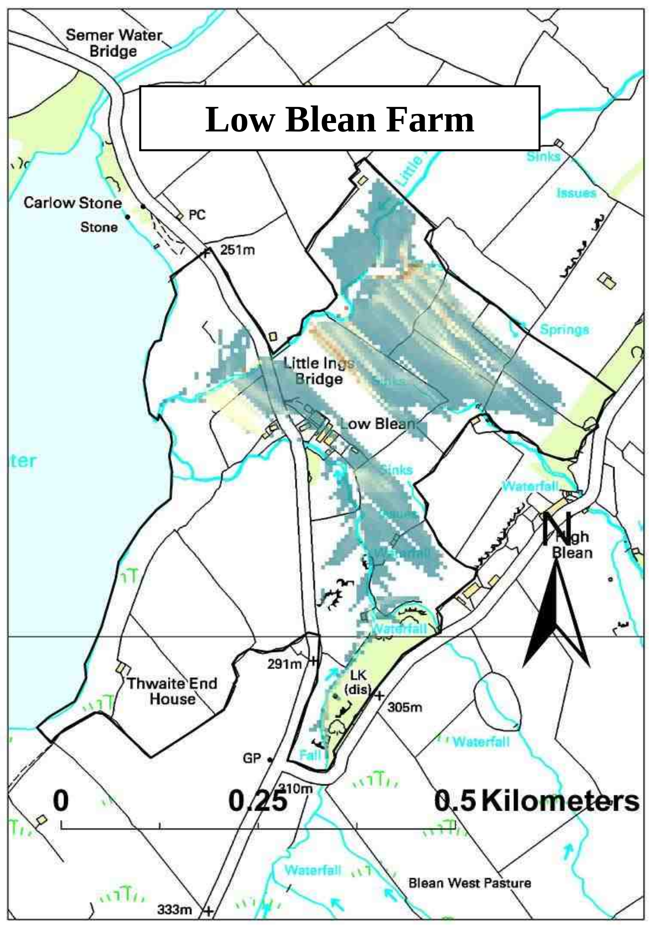
After viewing the map, a walkover survey was carried out concentrating on locations that SCIMAP suggested may deliver sediments. The first location visited was the location of the culverted stream, highlighted in Figure 5.4. The line of the culvert followed a location where SCIMAP suggested surface flow delivered sediments and F3 confirmed that, '*when the culvert becomes blocked with debris, the water does follow* *the locations shown on the map (SCIMAP output).*' In addition it would clearly connect directly to Little Ings Sike. This suggested that, at least in this location, SCIMAP had correctly described where surface flow and sediment delivery could be an issue in periods of high rainfall, but only on the occasions when the culvert was unable to transmit the flow. It is these fine details of land management that SCIMAP cannot capture, although there is a sense in which the risk is always there, it just needs the culvert to become blocked for it to be realised. Another location adjacent to this was also known to transmit surface flow when rainfall was high. These two locations are two of the higher-risk land patches on the farm according to SCIMAP and so a level of validation was attained.

The next field visited was immediately to the south of this location. This was another interesting location for revealing fine-scale nuances with limestone hydrology and land management practice. Here, a natural sink collects surface water from a spring. This then resurfaces and has been diverted into a drain to flow approximately 50 metres north to join a second drain. When the water began to resurface at this end point, F3 redirected the flow to join another patch of surface flow from a spring to the west that had previously been channelised. From here it flowed into a small stream. Interestingly, this ensured that the stream maintained reasonable flow year round and during the visit brown trout were noticeable in one of the pools. In addition to these natural drains and the culverted stream, F3 pointed out that most, if not all, of the fields had been under-drained in the past stating that, '*you can see them when its frosty.*'

The next field north followed the small unnamed stream. SCIMAP had highlighted some locations close to the channel that may deliver sediment, though these were in a relatively low risk category. It did appear that surface flow would follow the route suggested by the model as the topography followed a concave slope down to the stream and appeared to be an obvious flow path. In the final location visited, the stream had been fenced-out to provide a buffer strip on both banks which would act to sever the sediment transfer route. F3 explained that '*this had been carried out under the Raydale Project and the bank side had improved since the fence had been put in.*' The Raydale Project was a catchment-scale restoration project managed by Deborah Millward of the Yorkshire Dales Rivers Trust.

There were very few areas of bare soil within the fields although a number of gateways clearly suffered with erosion due to stock passage and heavy machinery. These seemed likely to be a source of fine sediment reaching the streams. In addition, the weeping wall slurry system appeared to pose a point source pollution risk. Presently, the model is unable to pick up enhanced erosion risks, such as at gateways, though it would be possible to use aerial photos and map them in the same manner the grips were mapped in Chapter 4. Then, an arbitrary area could be chosen and recoded to a higher-risk category (probably the highest risk loading as with the grips). In this way SCIMAP could account for such features, although this would be time-consuming and perhaps a better option is to use the model as it stands and assume that all gateways within the area of interest will enhance risk. SCIMAP was not developed to offer information on infrastructure issues. Figure 5.5 highlights he SCIMAP outputs for the holding.

Figure 5.6: *SCIMAP results for Low Blean Farm. The outputs are based on the same scale and form as with figure 5.3.*



5.3.4 School House Farm

School House Farm is situated in Raydale at Stalling Busk. It has been managed by the same family (F4) since the 1930s and owned by them in the early 1970s. It is presently managed as a beef and sheep enterprise which is supplemented, and possibly exceeded, by a successful pickles and preserves business. The farm consists of meadow, pasture and rough grazing over two land parcels. A number of streams cross the land. The main stream is Cragdale Water but several smaller springs and streams flow across the holding. Many of these follow subterranean routes for part of their course highlighting the high proportion of subsurface flow in this limestone-dominated region. F4 manages the farm enterprise and has entered the in-bye land into the Environmentally Sensitive Area Scheme with the higher ground entered into the Entry Level Scheme. Future plans are to amalgamate these into the Upland Entry Level and Higher Level Schemes.

Prior to discussing SCIMAP, F4 explained a number of interesting aspects of the local hydrology. These included work carried out by the local council which resulted in a shift in water flow in a high elevation first-order stream. The work diverted water which flowed down a public right of way into the stream. This resulted in rapid erosion of the stream which took several years to find equilibrium. Another example of how human management can disrupt flow paths was removal of a large boulder from a second order stream bed which significantly altered the stream dynamics. The stream bed became severely scoured and the channel widened to the point that a bridge has had to be erected where previously it was possible to drive a quad bike over the channel bed. These nuances of local hydrology reveal how small alterations can result in unexpected changes. F4 highlighted these examples to express the dynamic nature of upland dales streams.

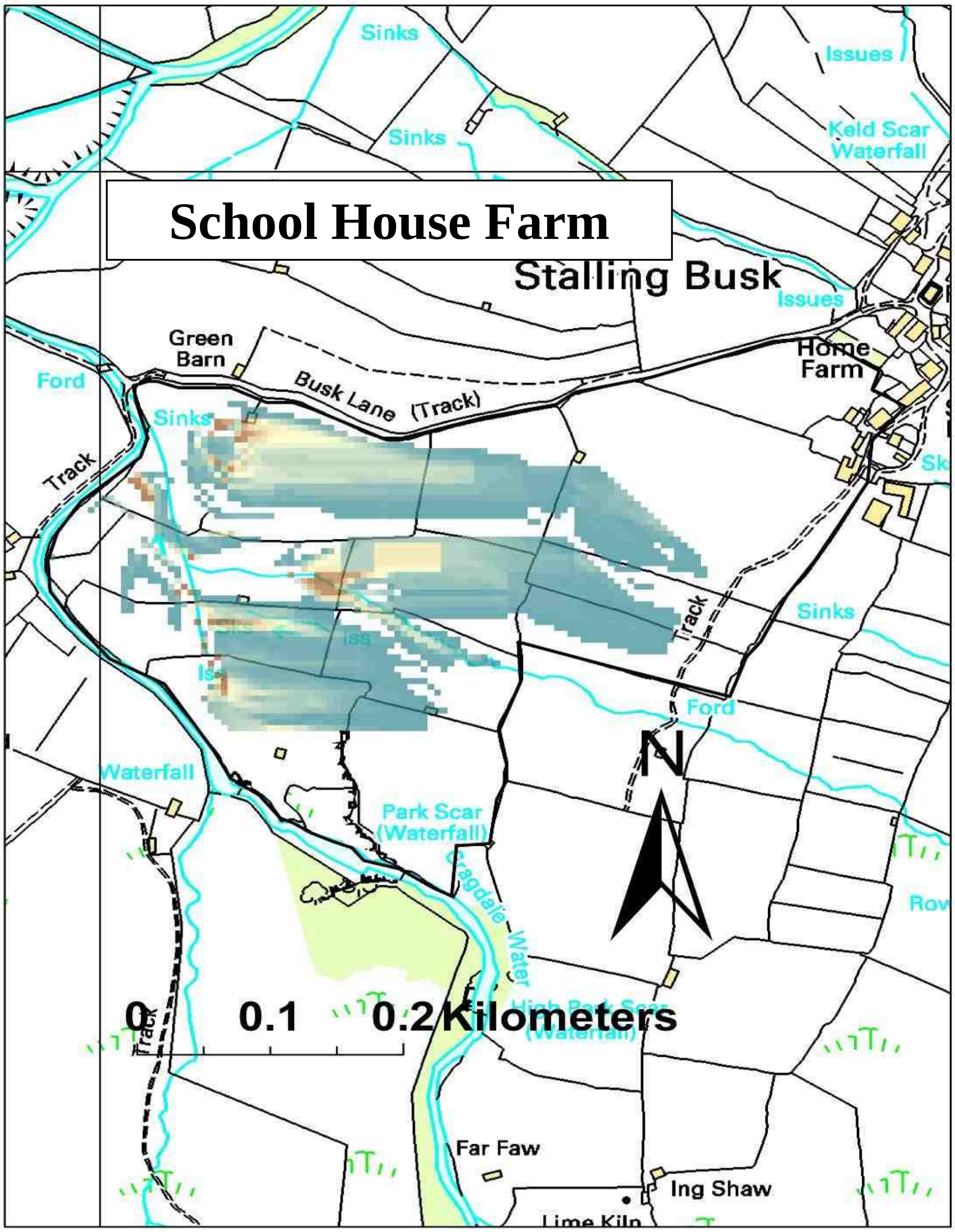
On viewing the SCIMAP outputs and having the model explained, F4 suggested that in a number of locations the model highlighted as risky, the issue with erosion was not due to land cover and surface flow but caused '*by rabbits that have become extensive on the holding causing bare soil that is washed from the fields when it rains*.' He also noted that some of the '*locations shown as high risk were not as risky as some of the lower-risk fields*'. An example of this was at locations where F4 suggested should be reversed in order '*to fit with the situation he recognised on the ground.*' This could be due to confusion between erosion and connected erosion. Whereas erosion is simple to identify, connected erosion is less visible in the landscape and often occurs during heavy rainfall events when few people are present to witness the issue.

Whilst walking the land, it became apparent that much of the water had been diverted into culverts and that the fields had been extensively under-drained as F4 had described. At one location a land drain had collapsed resulting in surface flow. On repairing the drain, F4 noted '*that three further drains entered from the upslope direction in a herringbone fashion*'. There are no data to reveal the location or extent of the subsoil drains though conversations with all the land managers provided strong anecdotal evidence that much of the lower in-bye, or meadow, land in the dale has been extensively under-drained. This level of drainage results in a situation where surface flow does not behave as the assumptions implicit in the model suggest, as highlighted by all the farmers interviewed. However, at a number of locations it was simple to visualise how surface flow would follow the route the model proposed in the absence of drainage. This was especially noticeable at locations where steep slopes are strongly connected to watercourses. How fine sediment reacts to these alterations to the local hydrology is an interesting complication that will be discussed in the next chapter.

In a number of fields on this holding, it appears that strong drivers of erosion are gateways, moles and rabbits. There are several locations where these appeared to be the primary cause for exposing bare soil. Whilst these should be a concern, they are being controlled across the catchment, though perhaps with less effort then is required. On the whole the grassland management appeared sound and a number of field corners had been given over to native tree plantations. Whilst Cragdale Water was well buffered from the farmland where it crossed the holding, many of the smaller streams are exposed to livestock. This undoubtedly adds nutrients and most likely offers a primary route for fine-sediment delivery, although there did not appear to be significant levels of erosion occurring within the land parcel that had been modelled.

F4 has for some years taken an environmentally sensitive approach to land management and was very proud of the wildlife on his land. As part of stewardship schemes he had created a number of circular walks. His desire for people to have access to his holding reflected the pride he held in the work he has carried out at the location. The SCIMAP results can be seen in figure 5.7.

Figure 5.7: *SCIMAP results for School House Farm. The outputs are based on the same scale and form as with figure 5.3.*



5.3.5: Redshaw Farm

Redshaw farm was the second largest land holding modelled (230 ha). The farm is located at the top of Widdale close to the Ure/Ribble watershed. The holding is managed for sheep with a smaller beef enterprise by F5 who has taken on the role of local coordinator for the newly created Dales Farming Network. The majority of the land is rough grazing with a small number of meadows surrounding the farm buildings. The soils are 'acidic and require good management.' In recognition, stocking rates are low and the farm has entered into Entry Level and Higher Level Schemes with meadow restoration being a target prescription. Due to this, only a few fields receive fertiliser with small applications of slurry being the main source of nutrients. Soil tests have shown that the soils '*are low in everything which means I can only take one cut of silage a year*.' A number of fields on this holding have never provided a silage crop and as a result these locations are floristically rich with globe and cuckoo flowers being prevalent. There are a few grips on the land '*but the majority have infilled and revegetated without intervention and the remainder will be blocked during the course of the agreements'*. There are a number of headwater 1st and 2nd order streams that run across the holding.

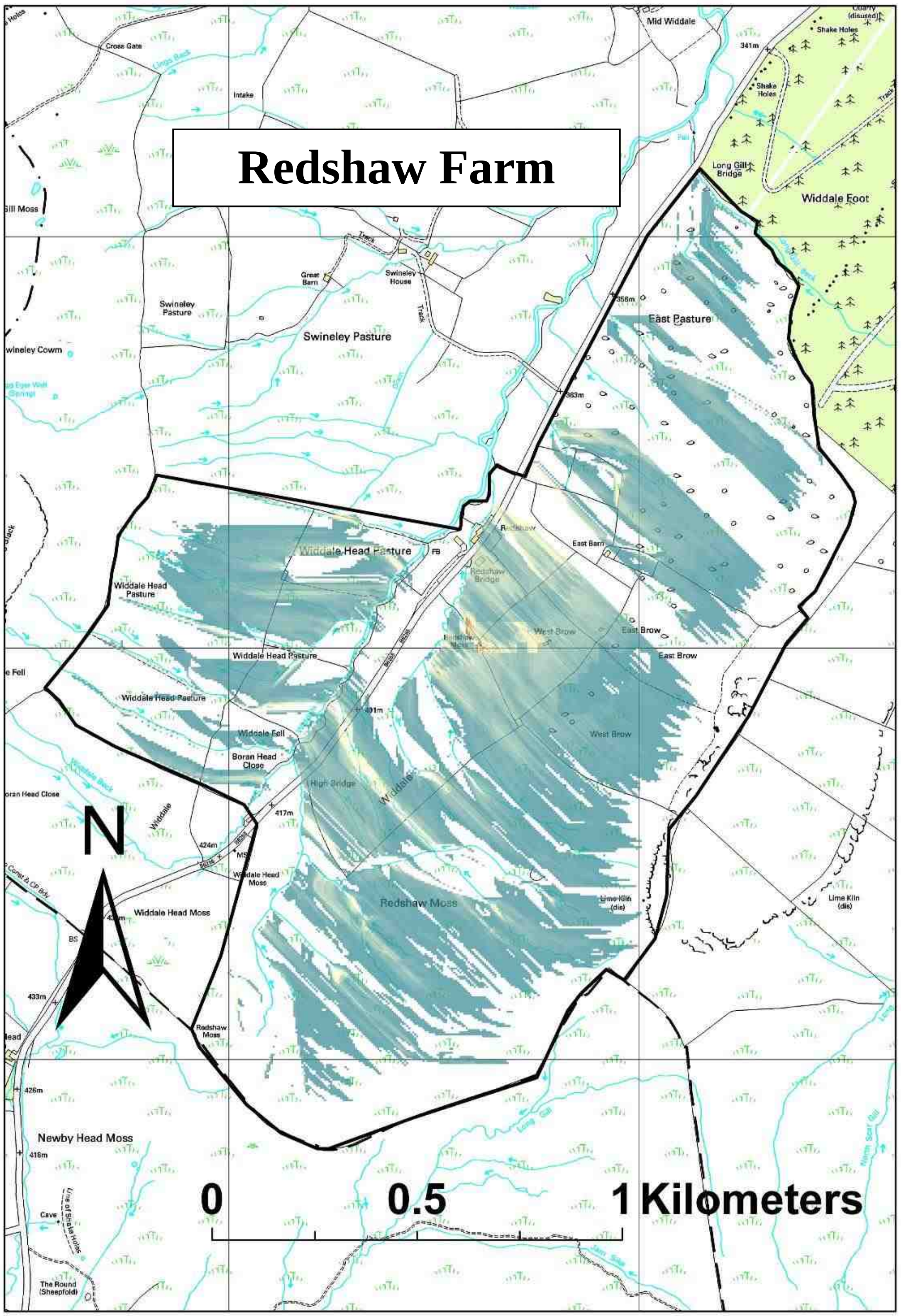
On describing and discussing the SCIMAP outputs, F5 suggested that the areas highlighted as high risk are on steep slopes but there is a '*thick matt of grass that reduces erosion and runoff rates.*' The area where erosion was a concern on his land was where hagging had developed on the peat soils in the rough grazing land parcels. These areas were '*now in the heather restoration option of HLS and the issue was being addressed.*' A number of small gills had been fenced-out and planted with native tree species to encourage black grouse. A secondary benefit of this is to provide a good buffer along the riparian zones which will reduce issues of fine-sediment delivery.

Two concerns of F5 was road runoff and sediment delivery from a plantation adjacent to his holding that had just begun to be felled. With regard the road runoff, he commented that '*the amount of salt that went down the becks this winter was shocking*,' and concerning the plantation he stated that post-logging the '*becks were a disgrace and on occasion ran black*' where sediment-laden water joined Widdale Beck. At these times it was clear to see as the two types of water ran '*side by side for some distance before mixing*.'

During the walkover survey, it became apparent that many of the old underground drains had begun to collapse. F5 suggested that these were one of the major pathways for sediment delivery on his land. Moreover, when repairing them, he was obliged under the Higher Level Scheme to make repairs in the traditional manner despite the fact that plastic drains would reduce sediment delivery through the drainage network, although he was quick to point out that the new schemes are better managed than the classic schemes and Natural England have been flexible and approachable. What this did highlight was a conflict between traditional methods of land management and resource protection, both at the field scale and catchment scale as fine sediment is routed towards river networks.

The locations that SCIMAP highlights as the greatest risk on the farm did not appear to be responsible for sediment delivery due in the most part to the land management style adopted. These had clearly developed in ways that countered soil loss. The grass sward was thick; stocking rates low and several riparian zones had been fenced out and planted with native trees. F5 did suggest that '*the model is right but the farming system changes the result.*' Whilst SCIMAP processes the relative risk of the spatial extent modelled the risk weighting at this farm was no higher than 0.3 which at the catchment scale suggests a low risk overall and F5 conferred with this by stating that he would view his land '*as a very low risk (due to) the farming system we have which negates sediment runoff.*' Figure 5.7 shows the SCIMAP output for this holding.

Figure 5.8: *SCIMAP results for Redshaw Farm. The outputs are based on the same scale and form as with figure 5.3.*



5.3.6: Town Head farm

Town Head Farm is owned and managed as a family enterprise (F6). The present incumbents are the third generation to manage the holding. This family continuity is not atypical of this form of upland farming. The holding sits on the outskirts of Askrigg and the farm enterprise is composed of dairy (80 head) and sheep (300 head). In contrast to many dairy farms in the dale, the majority of the milk is not sold to the Wensleydale creamery but directly to domestic and commercial premises in Wensleydale and Swaledale. The farm is presently in the Entry Level Scheme and will transfer to the new Upland Entry Level Scheme in the near future. The Higher Level Scheme does not appear to be an option for the farm, due in part to being a dairy enterprise which is regarded as risky in terms of diffuse pollution with a reduced floral diversity due to the increased intensity of the meadows. Despite this, F6 has carried out several small native tree plantations and plans to fence out some stream banks to exclude cattle. The family have a keen interest in the biodiversity on his land and regard their management in time frames that stretch to future generations.

The meadows provide one cut of silage or haylage[28] each year, with two cuts taken from a small portion of the land in some years when yields have been low across the holding. Fertiliser use is low with ten tonnes of 25:5 applied to 120 ha of land each year. The rest of the soil nutrient needs are supplied by slurry spreading; unusually within the catchment, F5 has at least fifteen weeks of slurry storage. F6 could not remember when the soil was last tested for nutrients and pH but this is due to be carried out in the near future as part of a Catchment Sensitive Farming Partnership. As the slurry store is large, spreading can be timed to optimise efficiency and reduce leaching to watercourses. Cattle are housed for approximately five months over winter. The higher rough grazing allotments do not receive fertiliser. In the past this higher area were the source of Askrigg's drinking water supply and is still used for this purpose on occasion.

On being asked if he had identified any issues of sediment erosion on his land, F5 replied that, '*in wet months at the back end of the year the cows can make a mess on the way to the milking parlour*.' On viewing the SCIMAP outputs, F5 pointed out one area

[28] Haylage is a semi-wilted grass wrapped in a bag. It is considered to have a lower water fraction than silage.

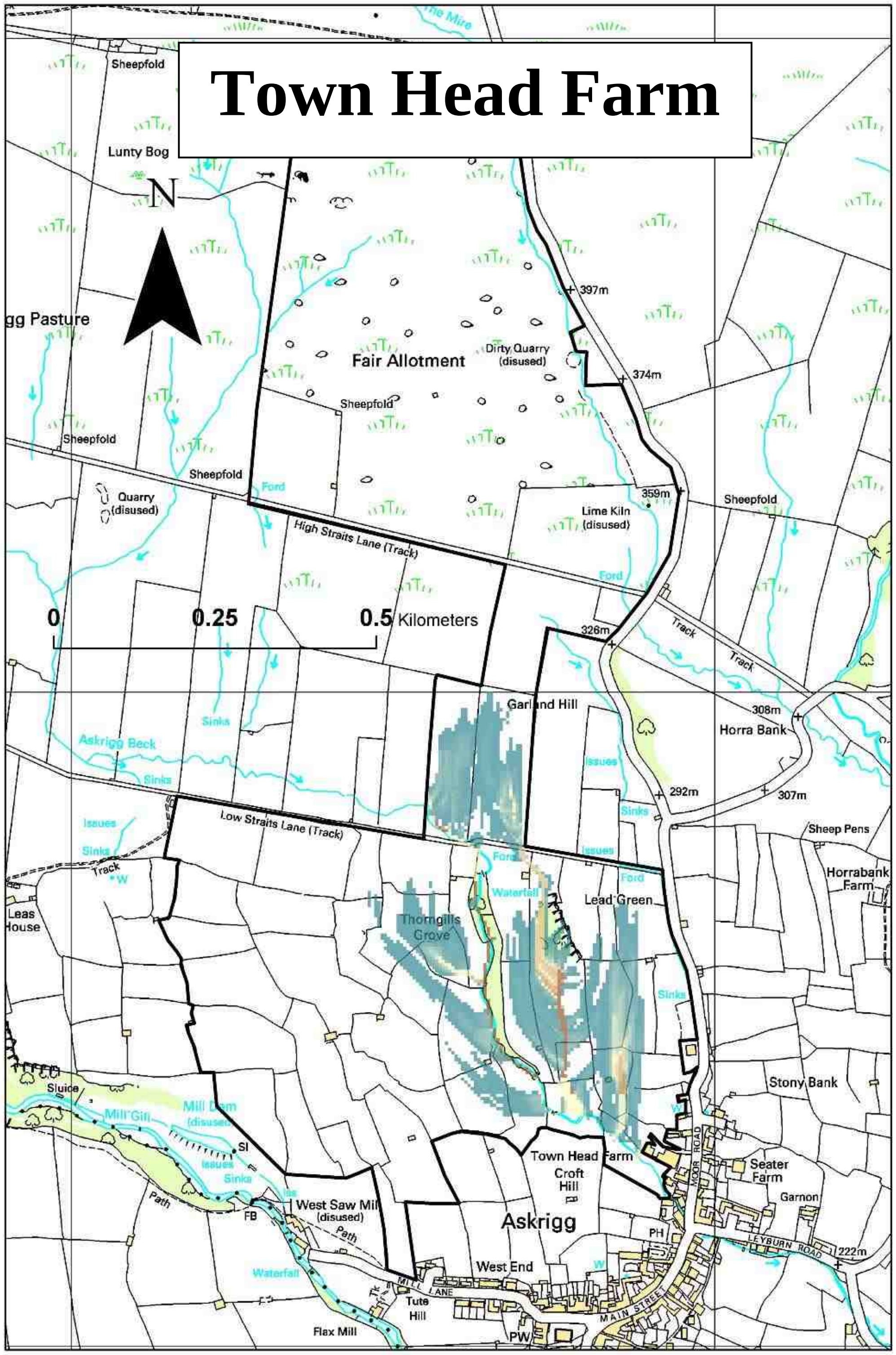
which had been identified as a possible risky location and noted that a large drain transports the water there reducing surface flow. As at other farms, this suggests that farmers are keenly aware of where surface flow could cause concerns and highlights how they can adapt management to alleviate these issue. During the visit several water sinks and locations of upwelling were noted, again highlighting that water flow is often through drains and natural sinks. F6 suggested out that, '*all the meadows are under-drained*' confirming findings on earlier farms. He corroborated that these drains were of the traditional stone culvert style and thought they could possibly act as alternative sediment transfer routes.

On walking the land, F6 pointed out several other locations that had been drained and culverted to remove water from the surface and pointed out that, '*there is a good grass sward across the farm*,' that will reduce runoff. In some fields the cattle grazing was carefully managed by deploying electric fencing on a rotational basis within the land parcel to reduce pressure. This '*allows the grass sward to remain healthy and reduces trampling*.'

The locations that appeared to have the greatest pressure are where cattle pass to the milking parlour and gateways. F6 also mentioned that post-gripping of the higher peat soils erosion increased severely. A number of locations that SCIMAP identified as high risk looked obvious contenders for surface flow but past management had reduced this risk by putting in subsoil drains and managing the water to alleviate the problems.

Figure 5.9 shows the SCIMAP results from the work at Town Head Farm with F6. It is interesting to see how the main body of risk is located on the inbye land close to the farm buildings. This fits with observation but there are subtle differences between observation and the model outputs. These will be expanded in the next chapter.

Figure 5.9: *SCIMAP results for Town Head Farm. The outputs are based on the same scale and form as with figure 5.3.*



5.3.7 Raydale Grange farm

Raydale Grange Farm is managed in conjunction with the neighbouring farm by the same family (F7) close to the valley head enclosing the source of Raydale Beck. This is the largest of the farms visited and the one with the fewest high risk locations identified by SCIMAP both at the farm and catchment scale. It has a large proportion of moorland which probably accounts for this due to this land cover type being rated as a low risk. Despite this, the farm contains Fleet Moss, one of the most heavily eroded areas of moorland in the dale, and probably within the National Park. The peat here is heavily hagged and eroded though the erosion is likely to be wind driven perhaps primed by grips which are easily noticeable from aerial photos. However, F7 did not think the erosion was due to gripping as it was eroding '*long before we put the grips in and they (the grips) are filling up on their own.*' Despite this, the grips do direct water into this heavily eroded area before their form disappears amongst the peat hags suggesting that they have been active in the past.

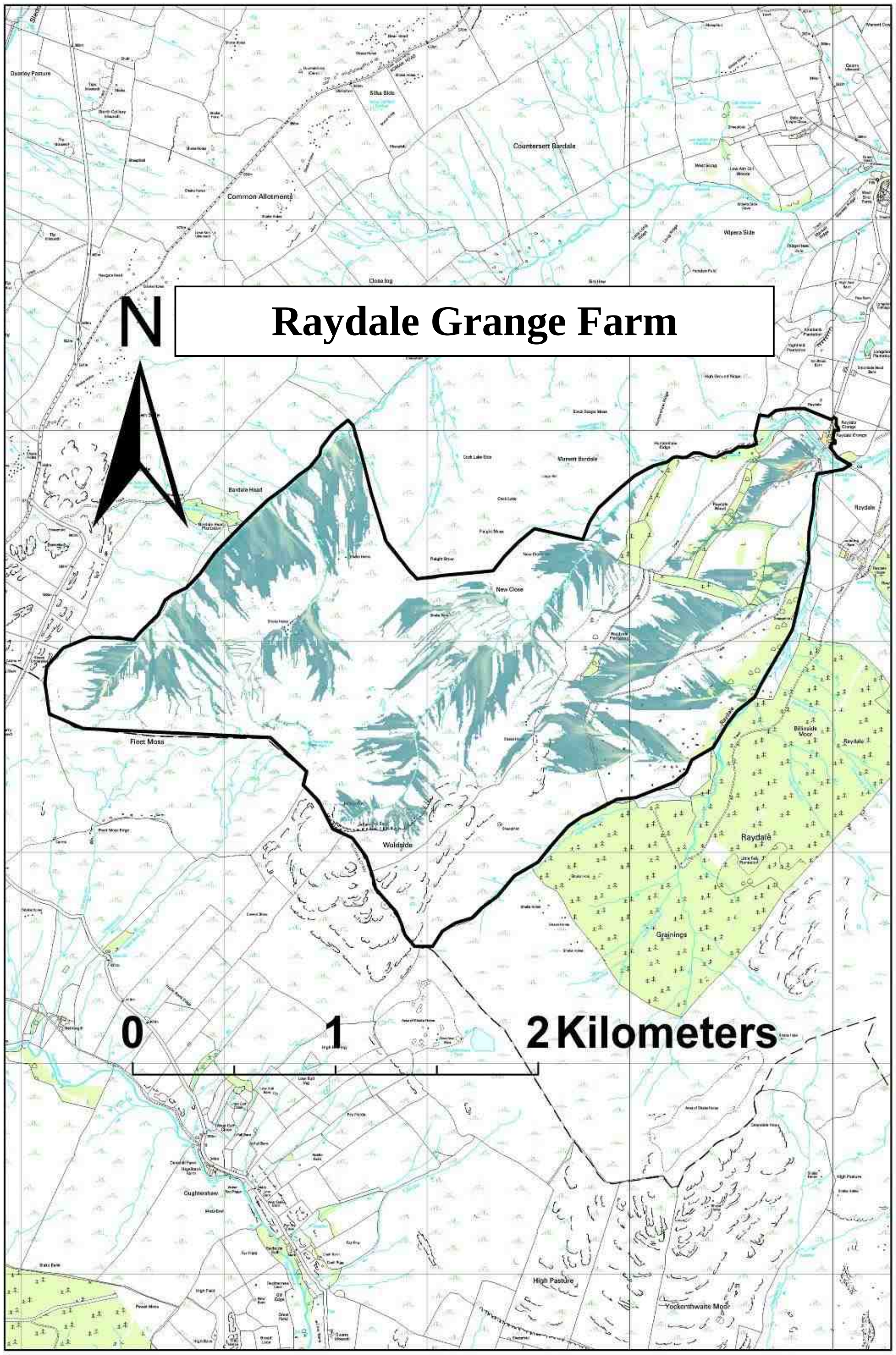
The farm covers 628 ha over three land parcels (though less then this was modelled). It is a sheep and dairy enterprise with 57 milking cows, 70 heifers and calves and approximately 1000 sheep; whilst this seems a large number of livestock, the stocking rate is only 0.26 livestock units per ha. The farm is in the Entry Level Scheme with some of the higher ground still within the classic Countryside Stewardship Scheme. There is presently an application being made to enter the Upland Entry Level and Higher Level Schemes. The meadows are managed to provide two or three cuts of silage (dependent on annual weather conditions) and receive 250kg/ha of nitrogen with applications of slurry spread three times over the winter with a further application after the first cut. This is an intensive management regime in comparison to other farms in the catchment. The soils of the main silage fields have been tested as part of the Yorkshire Dales Rivers Trust Raydale project. Cattle are housed from October to May and there is little sign of poaching on the in-bye land. However, higher up in the rough grazing locations there is notable erosion surrounding supplementary feeders.

On first viewing the SCIMAP outputs, F7 expressed surprise at the locations it had identified as high risk. He mentioned that the worse location for erosion was New Close Gill where undercutting by the stream had resulted in a number of landslides and '*new* *close stream can get black*' after rainfall. This kind of undercutting erosion at locations with interlocking spurs is not something that is represented in SCIMAP. F7 also suggested the intensity of rainfall has become more extreme in recent years adding to issues of erosion.

The model only returned one small strip on a hillslope close to the farm house as high risk and this location was visited first. The topography suggested that runoff would follow the route that SCIMAP identified as the major delivery pathway, though F7 suggested that '*there was rarely visible surface flow down the slope*.' He commented that all the meadows would be under-drained and pointed out an additional two surface drains, although these may have been streams that had been deepened and straightened. These collected much of the surface flow and highlights once more how active management has responded to the need to remove water from the fields to enhance farming. At the time of the visit, the field was being utilised for grazing and there were no signs of poaching.

A number of shelter belts have been planted on the lower meadows and whilst they were not specifically for reducing water flow, the location of at least three of these would help to sever the delivery route for fine sediment. Moreover, interception would reduce runoff across the hillslope. Overall, it was possible to see that, although surface flow would theoretically follow the routes that SCIMAP identified, several management techniques, especially surface and under-drainage, negated this to a large extent. F7 explained that '*the drains would be of the traditional stone 'culvert' style'* and these would perhaps offer a secondary delivery pathway. Figure 5.10 highlights a number of areas on the farm where management has reduced hydrological connectivity.

Figure 5.10: *SCIMAP results for Raydale Grange Farm. The outputs are based on the same scale and form as with figure 5.3.*



5.3.8 Semerdale Hall farm

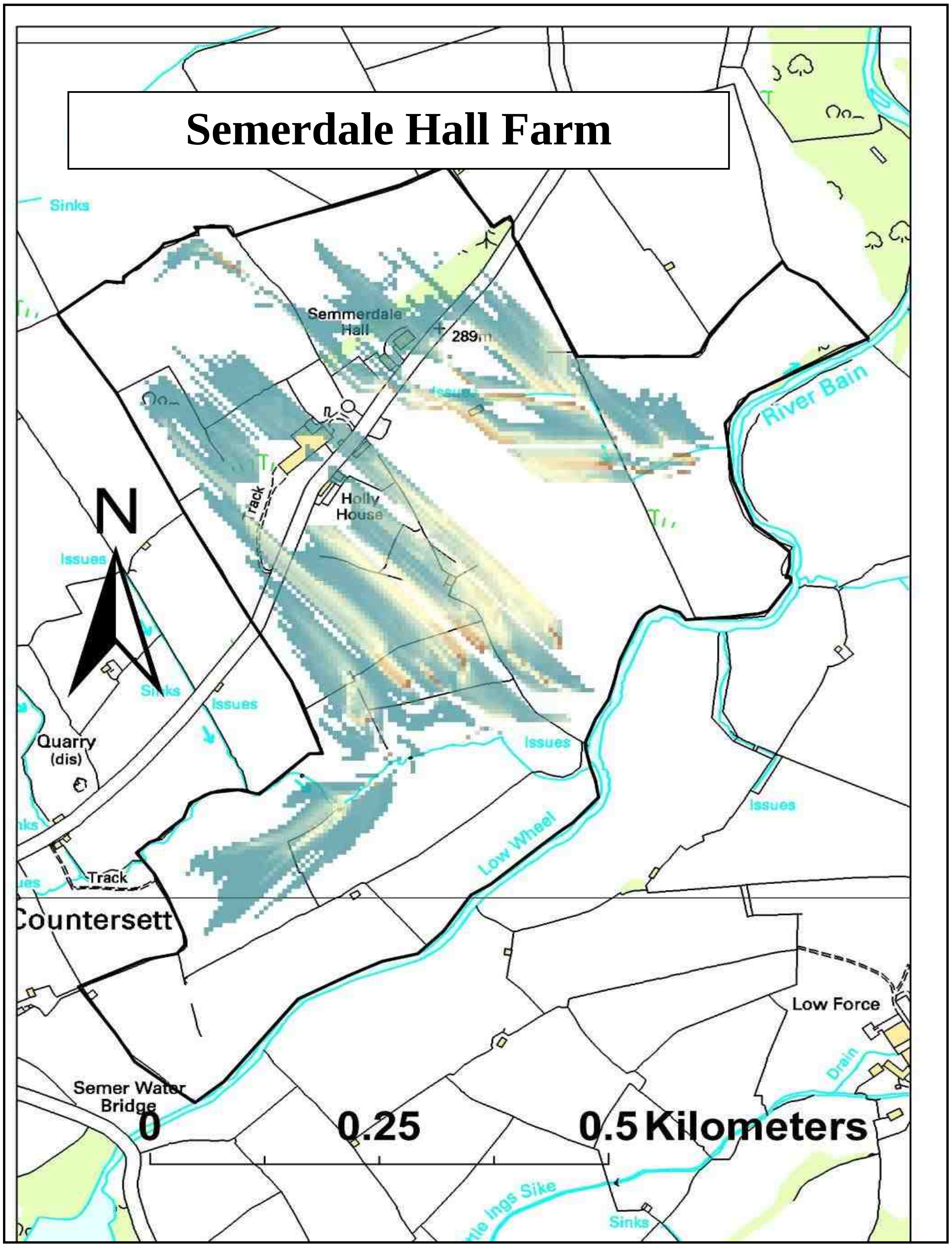
Semerdale Hall Farm is located in Raydale and adjoins the river Bain below the outtake of Semerwater Lake. The farm is owned by F8 with help from his father. The holding is run as a dairy and sheep enterprise with 70 milking Holsteins with 40 followers (young cattle) and 300 sheep with followers. The majority of the farm is within the land parcel close to the river Bain but there is another parcel at the high end of the valley on the flanks of Wether Fell. The meadow land is extensively under-drained and F8 suggests that '*this would be the case throughout the dales.*' This confirms findings on all the other farms. He also commented that they are of the same stone 'culvert' design as on other farms but when they fail he replaces them with plastic piping which would stop the drains acting as sediment transfer conduits. In contrast to the other farmers, F8 said that '*he would know where most of the drains are laid'*. F5 (Redshaw farm) mentioned that as he was in the Higher Level Scheme he was encouraged to repair drains using traditional methods. This suggests that Natural England accept a trade-off between traditional skills and resource protection as they administer their stewardship schemes. The Semerdale Hall Farm is in the Entry Level Scheme and is keen to join the Upland Entry Level Scheme shortly. An application to join the Higher Level Scheme was unsuccessful, possibly due to a combination of high intensity farming and dairying.

The meadow land is managed to provide three cuts of silage each year and receives 20:10:10 fertiliser with additional nutrients coming from slurry. The livestock are carefully managed on the land with fields being partitioned for grazing with electric fencing. In the largest field, grazing can run for up to 3 weeks but generally livestock are rotated between fields every 3 to 5 days. As part of the Raydale project (administered by the Yorkshire Dales Rivers Trust), F8 has fenced out approximately 1 km of river bank with the buffer strip being planted with native trees. This extensive buffer strip links two pre-existing woodland areas creating a good wildlife corridor and providing a management practice that may reduce sediment delivery. As part of this project, drinking bays were installed but the fields also contain gravity-fed drinking troughs which help reduce the pressure on the river bank. In two places F8 mentioned that poaching had become a severe problem (particularly during the foot and mouth crisis) but the management of these fields has been altered and the problem consequently remedied.

On viewing the SCIMAP outputs and being asked if the model accorded with his experience, F8 replied, '*not exactly, no*.' He went on to explain that the locations '*above the road on the north side of the holding appeared to be reasonably accurate (with regards surface flow) but the water is picked up by drains once it reaches the road and is redirected away from the lower sections of the farm*.' On the lower fields, he noted that some locations were also reasonably accurate. At other locations water is transferred from the land to river via surface and subsurface drains meaning that SCIMAP would be unable to capture the routing of this modified hydrological system. F8 did mention that the river overtops and stands deeply on a number of the lower fields and wondered '*if this would be a route for sediment to leave the land.*' although it was agreed that it was more likely to deposit sediment and so would be a net provider of soil to the farm.

On walking the lower meadows with F8, it became apparent that water was being redirected and that the delivery pathways SCIMAP identifies would likely be inaccurate. It was also apparent that water would have indeed followed the pathways highlighted in an unmodified system. This confirmed the situation with all of the visits. What remains to be understood is if the drains themselves still provide a delivery pathway for fine sediment. It seems quite likely that they do and on an earlier visit a farmer had stated that this was the case (F5, Redshaw Farm). To confirm this would require sediment monitoring in these drains as a comparison with plastic drains. Figure 5.11 highlights a number of the areas where management reduces risk of fine sediment delivery on the farm.

Figure 5.11: *SCIMAP results for Semerdale Hall Farm. The outputs are based on the same scale and form as with figure 5.3.*



5.4 Catchment-scale SCIMAP modelling, GIS and remote sensing

The SCIMAP model was run at the catchment scale on three occasions. First, unweighted by land cover ($SCIMAP_U$); second, weighted by land cover ($SCIMAP_L$); and third, weighted by land cover with grips added to the DEM and LCM ($SCIMAP_G$). The results from each of the SCIMAP runs show some differences in risk apportionment. When unweighted by land cover, the risk has a more uniform spread in contrast to weighted by land cover. $SCIMAP_L$ shows that the risk is greater when in-bye meadow and pasture land are provided a higher risk than moorland and woodland as befits the likelihood of delivering fine sediments based on proximity to river networks and the increased intensity of land management at these locations. However, when grips are added to the land cover map and coded as high risk to reflect the enhanced sediment delivery from these open drains, there is a shift of risk towards the upper locations of the catchment. Whilst this is only obvious under close scrutiny, this increased risk occurs close to low-order streams where brown trout recruitment is most obvious.

Figure 5.12 shows the SCIMAP catchment-scale results. The map is based on two outputs: 1) the risk of a recipient stream in terms of fine sediment; and, 2) the likelihood of each $5m^2$ land parcel delivering fine sediment. This is based on a surface flow index multiplied by the erosion risk of each land parcel. The colours that follow the stream network in figure 5.12 are indicating indicate the risk categories for fine sediment concentration/delivery from that stream. It has been calculated as a standard deviation around the mean going from green (low risk) to red (high risk) with light orange being the mean for the catchment. To use the model in the field, a red stream needs to be identified (i.e. one that is delivering a disproportionate amount of fine sediment compared to its upstream area) and then the underlying model output surrounding the red stream (light blues to brown) can be viewed as the locations most likely to be delivering fine sediment to the river (i.e. there is a source, a pathway and a recipient stream). These are the places to direct surveys in order to assess whether buffer strips, contour and gill planting (or any other management measure) are required to sever connectivity, reduce erosion risk to control the original erosion issue or sever the delivery pathway. It is important to note that SCIMAP does not provide definitive answers but assists with targetting across broad spatial scales by assigning a risk probability framework to a landscape.

Figure 5.12: catchment-scale *SCIMAP results for the upper Ure case study. The outputs show the in-stream risk and field scale risk of fine sediment delivery.*

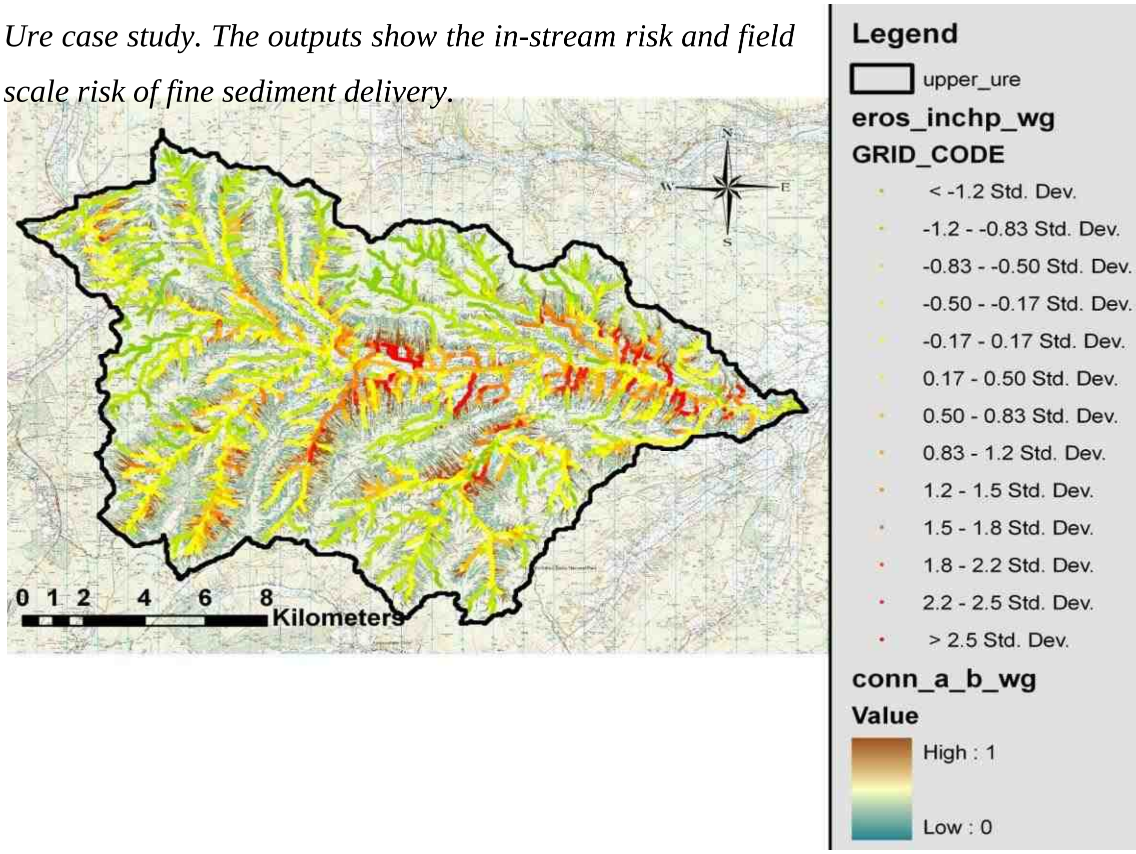


Figure 5.13 shows the in-channel risk of fine sediment delivery for recipient watercourses throughout the catchment. These maps cover SCIMAP$_U$, SCIMAP$_L$ and SCIMAP$_G$ to highlight the differences in the three runs. Much of the difference is of risk being restricted to the floodplain when weighted by land cover and towards the upper reaches of the catchment when grips are added. The maps highlight how risk can be viewed in relation to on the assumptions made beforehand. Here, the model that is expected to most reflect the upper Ure catchment is SCIMAP$_G$ as it most closely represents the landscape in terms of risk derived from land management. However, the grips that are added to the DEM and LCM are a uniform 5 metres width and 1 metre depth. This does not reflect the reality. Initially, the grips were cut according to the Cuthbertson Drainage Plough which cut the grips to a uniform 20cm wide and 50cm deep. Despite this initial cut, the grip networks have been dynamic and now offer spatio-temporal differences dependent on location in the catchment, location in the grip network, peat soil type, land use intensity and time since cutting. Hence some grips have filled in, some remain static whilst others have become severely eroded and are greater

than 2 metres deep and 5 metres wide. This level of detail cannot be captured in the model.

The changes in risk arising from the different runs reveal how land cover can redistribute risk across a catchment. It is interesting to note that $SCIMAP_G$ adds risk to low-order streams at higher elevations within the catchment. These are often the locations where brown trout recruitment occurs.

In order to identify which of the model outputs did indeed “best” reflect the catchment, it was necessary to assign the SCIMAP risk code for each of the runs against the appropriate survey site. In order to do this the ‘Erosion Risk in Channels concn.’ layer from the SCIMAP outputs was imported into ArcGIS along with the shape file map for the electrofishing sites. Each of the SCIMAP outputs were taken in turn and the risk categories enlarged with the value recorded against individual sampling sites, wherever they coincided, ranging from 1 (the lowest risk category) to 13 (the highest risk category). This was carried out for all three in-stream SCIMAP outputs (without grips, with grips and unweighted by land use). A depiction of this process can be seen in figure 5.14, the results are shown in table 5.10 and discussed in chapter 6 (section 6.3.2).

Figure 5.13: *The SCIMAP in-stream outputs reveal likely risk of a stream delivering fine sediments based on the surrounding landcover, slope and rainfall. This offers potential for conservation bodies to explore a catchment systematically based on the model description of risk. The outputs here show SCIMAP*$_U$ *(map A), SCIMAP*$_L$ *(map B) and SCIMAP*$_G$ *(map C).*

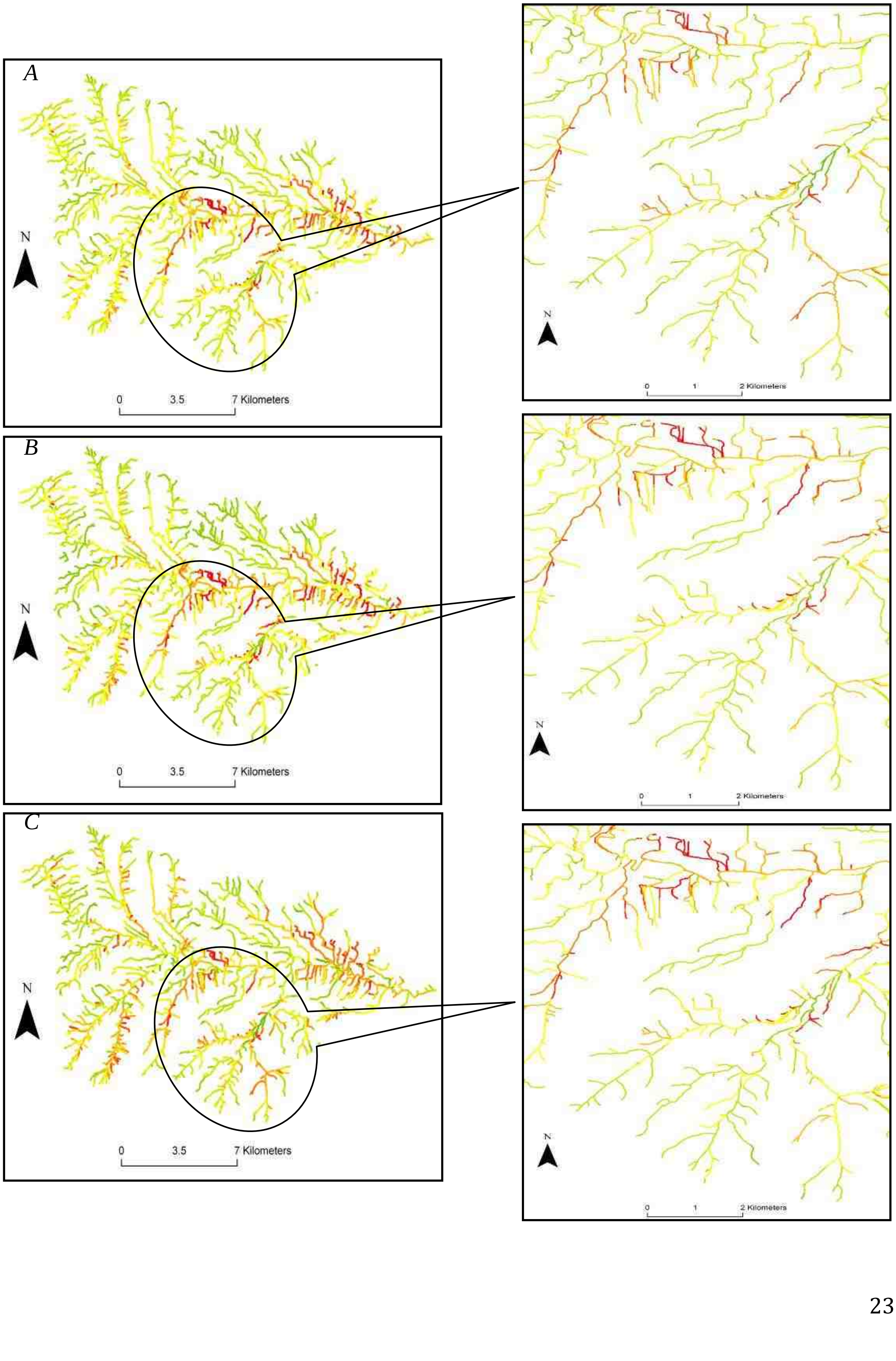
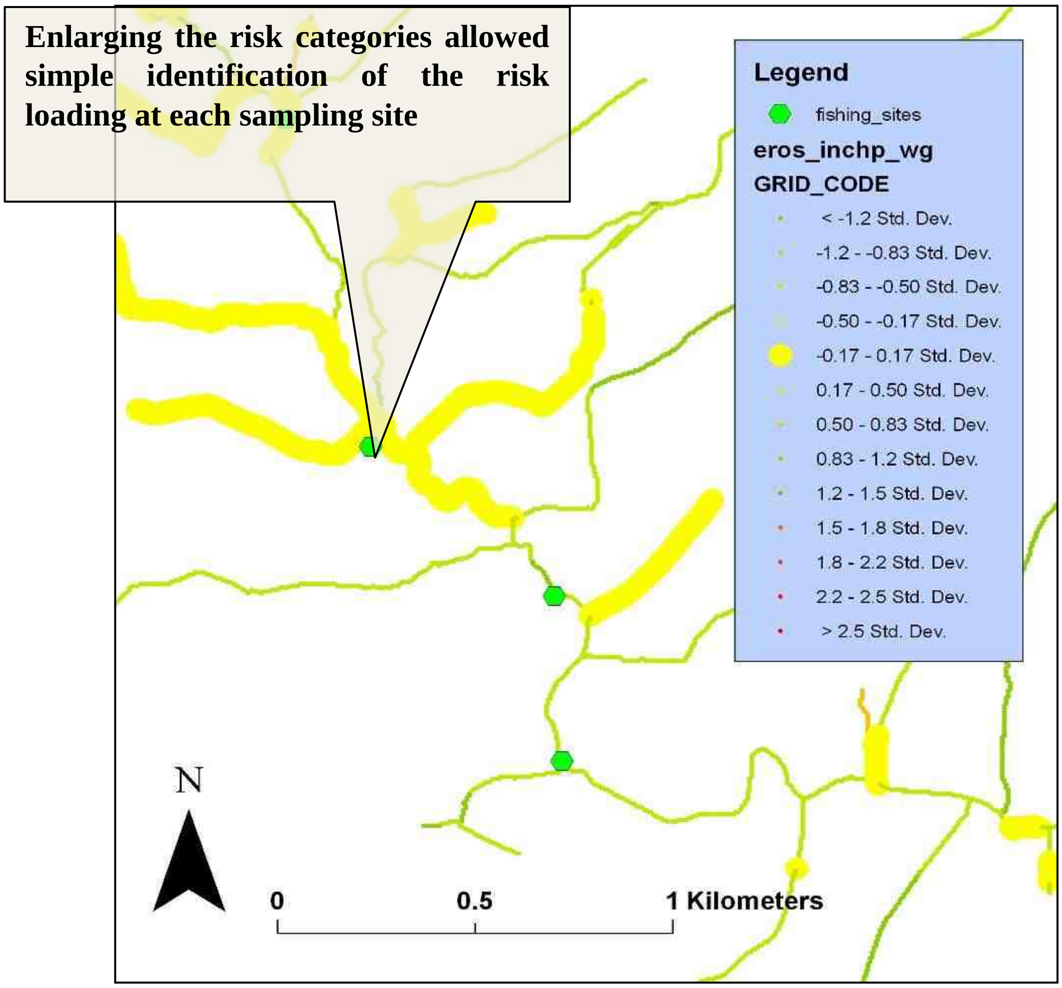


Figure 5.14: *The in-stream SCIMAP output risk categories were enlarged individually, as seen below, in order to ascertain the risk loading for each electrofishing site of the three different SCIMAP runs.*



5.4.1 GIS, remote sensing and SCIMAP model results

At a number of sites, the loadings between $SCIMAP_{LandG}$ with and without grips were similar with many sites showing the same risk loading. The greatest risk loading was 12 and the lowest 1; both these came from $SCIMAP_{U}$. The results are shown in Table 5.10 and the units for the remaining GIS and remotely sensed variables are also presented in the same table.

Table 5.10: *Results of the catchment scale analysis derived from GIS, remote sensing and modeling.*

Electrofishing site	Grid ref	Upstream contributing area Km^2	Area of upstream moorland Km^2	Strahler stream order	$SCIMAP_L$	$SCIMAP_G$	$SCIMAP_U$
Ballowfields	SD994890	4.42	2.36	2	8	8	4
Cotterdale	SD832938	7.16	6.2	1	5	5	6
Cotterdale	SD833939	5.1	4.19	2	5	5	6
River Ure	SD839916	19.9	10.64	4	5	5	6
Cotterdale	SD834934	12.74	10.45	3	4	4	10
Cotter Force	SD849916	18.8	14.46	3	5	5	10
Widdale (L/S)	SD805850	0.46	0.46	1	4	5	8
Widdale (R/S)	SD805850	0.39	0.39	1	4	5	7
Widdale	SD805852	2.53	2.05	2	4	6	7
Widdale	SD811865	11.77	7.15	3	4	5	8
Widdale	SD812866	8.33	5.84	3	4	5	8
Widdale	SD827879	12.62	7.6	3	5	5	7
Widdale	SD857907	35.54	21.41	4	5	5	1
Sleddale	SD863881	12.99	7.94	3	5	6	12
Sleddale	SD864858	6.81	5.62	2	5	5	8
Sleddale	SD856866	2.18	2.15	2	6	5	9
Raygill	SD913900	6.31	3.3	2	6	6	7
Mossgill Ford	SD830919	11.42	10.49	3	5	6	6
Snaizeholme	SD832872	10.73	6.4	3	6	5	3
Snaizeholme	SD827853	4.95	4.73	3	6	5	10
Snaizeholme	SD825849	1.11	0	1	6	5	12
Snaizeholme	SD825847	2.63	2.63	2	5	5	8
Mill Gill	SD914942	11.41	5.5	2	4	5	8
Mill Gill,	SD936917	5.94	5.5	3	4	6	8
Grange Beck	SD923914	5.44	5.44	2	4	6	5
Grange Beck	SD933912	11.12	7.81	2	5	6	5
Strands	SD865921	11.91	10.85	2	6	5	8
Paddock Beck	SD946905	14.66	11.16	2	4	6	8
Raydale,	SD904849	9.41	5.77	3	5	5	6
Raydale	SD909862	8.71	8.22	4	5	7	8
Raydale	SD909859	9.04	8.8	3	6	7	8
River Ure	SD786962	1.4	1.24	1	5	5	5
River Ure	SD785956	4.58	3.65	2	5	5	5
Ure, Lunds	SD792945	8.6	5.67	3	7	5	6
River Ure	SD799932	11.17	7.5	3	6	6	2
River Ure	SD799928	12.17	7.85	3	4	4	5
Cotterdale	SD845923	14.48	11.56	2	5	5	10
Thornton Rust	SD969876	3.82	3.38	2	5	5	8
Thornton Rust	SD964875	0.34	0.34	1	9	9	10
Thornton Rust	SD965876	3.22	3.22	2	6	5	8

5.5 Statistical analysis

Once the data collection and modelling was complete, it was important to explore the data to develop an understanding of which relationships were significant. This process allows the important factors to be discriminated from the background noise which is inherent in all systems. The initial step was to run correlation analysis (Table 5.11) to understand where the positive and negative relationships exist and identify the strength of these relationships. It was interesting to note a number of strong correlations between habitat-scale variables and catchment-scale processes. In particular, there exist strong correlations between substrate types and SCIMAP loadings. Streams that are prone to drying show a strong ($p<0.01$) negative correlation with both upstream contributing area and upstream area of moorland. Macroinvertebrate richness showed a negative correlation ($p<0.05$) with $SCIMAP_{LandG}$ but not $SCIMAP_{U}$ highlighting the need to include land cover risk categories. Buffer strips showed positive correlations with both upstream contributing area ($p<0.05$) and upstream area of moorland ($p<0.01$). This suggests that buffer strips are more likely to be part of land management practice on larger streams where land management takes advantage of the wider floodplains; however; there was no correlation with stream order.

Another interesting positive correlation was between poaching and siltation ($p<0.01$) suggesting that fine sediment inputs, and subsequent deposition, can be derived directly from the adjacent land use. Siltation did not correlate significantly with any of the catchment-scale factors (e.g. upstream area, $SCIMAP_{LandG}$). Unsurprisingly, it did show a negative correlation with the presence of buffer strips ($p<0.05$). Sand and silt substrate type had a number of strong correlations (positive $p<0.01$ with siltation, in-stream pools, algae, macrophytes; negatively $p<0.05$ with poaching); although these finer substrate fractions did not show any correlations with SCIMAP or any other catchment-scale processes. This was surprising and suggests that adjacent land use of the immediate riparian zone may be a stronger factor in fine-sediment delivery and deposition than first thought. The percentage shade of the sample sites only showed two correlations: 1) positively with SCIMAP risk loading with grips ($p<0.01$) and 2) negatively with stock access ($p<0.01$). The negative relationship with stock access is the more intuitive of the two.

Table 5.11: *Pearson's correlation matrix showing the significant relationships between the data. See appendix 1.*

	1 Macro-invertebrate abundance	2 Macro-invertebrate richness	3 Simpsons diversity index	4 Shannon's diversity index	5 LIFE scores	6 Obstructions <500m upstream	7 Obstructions <0.5km d/s	8 Obstructions <1km u/s	9 Obstructions <1km d/s	10 Survey area prone to drying	11 Stream prone to drying d/s	12 Stream prone to drying u/s	substrate				17 siltation	18 Channel width	19 In-stream pools present	20 Algae growth	21 Macrophytes present	22 undercuts	23 Earthcliff on bank	24 Stock access	25 Buffer strip	26 Surrounding land use	27 Poaching of soils	28 % shading/tree cover	29 Upstream area	30 Upstream area of moorland	31 Strahler stream order	32 SCIMAP $_L$	33 SCIMAP $_G$	34 SCIMAP $_U$
													13 bedrock	14 boulders and cobbles	15 pebbles and gravel	16 sand and silt																		
1																																		
2	*																																	
3		**																																
4		**	**																															
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	1 Macro-invertebrate abundance	2 Macro-invertebrate richness	3 Simpsons diversity index	4 Shannon's diversity index	5 LIFE scores	6 Obstructions <500m upstream	7 Obstructions <0.5km d/s	8 Obstructions <1km u/s	9 Obstructions <1km d/s	10 Survey area prone to drying	11 Stream prone to drying d/s	12 Stream prone to drying u/s	substrate				17 siltation	18 Channel width	19 In-stream pools present	20 Algae growth	21 Macrophytes present	22 undercuts	23 Earthcliff on bank	24 Stock access	25 Buffer strip	26 Surrounding land use	27 Poaching of soils	28 % shading/tree cover	29 Upstream area	30 Upstream area of moorland	31 Strahler stream order	32 SCIMAP risk category	33 SCIMAP risk category (grips)	34 SCIMAP unweighted
													13 bedrock	14 boulders and cobbles	15 pebbles and gravel	16 sand and silt																		
24																																		
25																	*	**						**										
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33		*			**									**	**					*								**				**		
34								*			*									*				*										

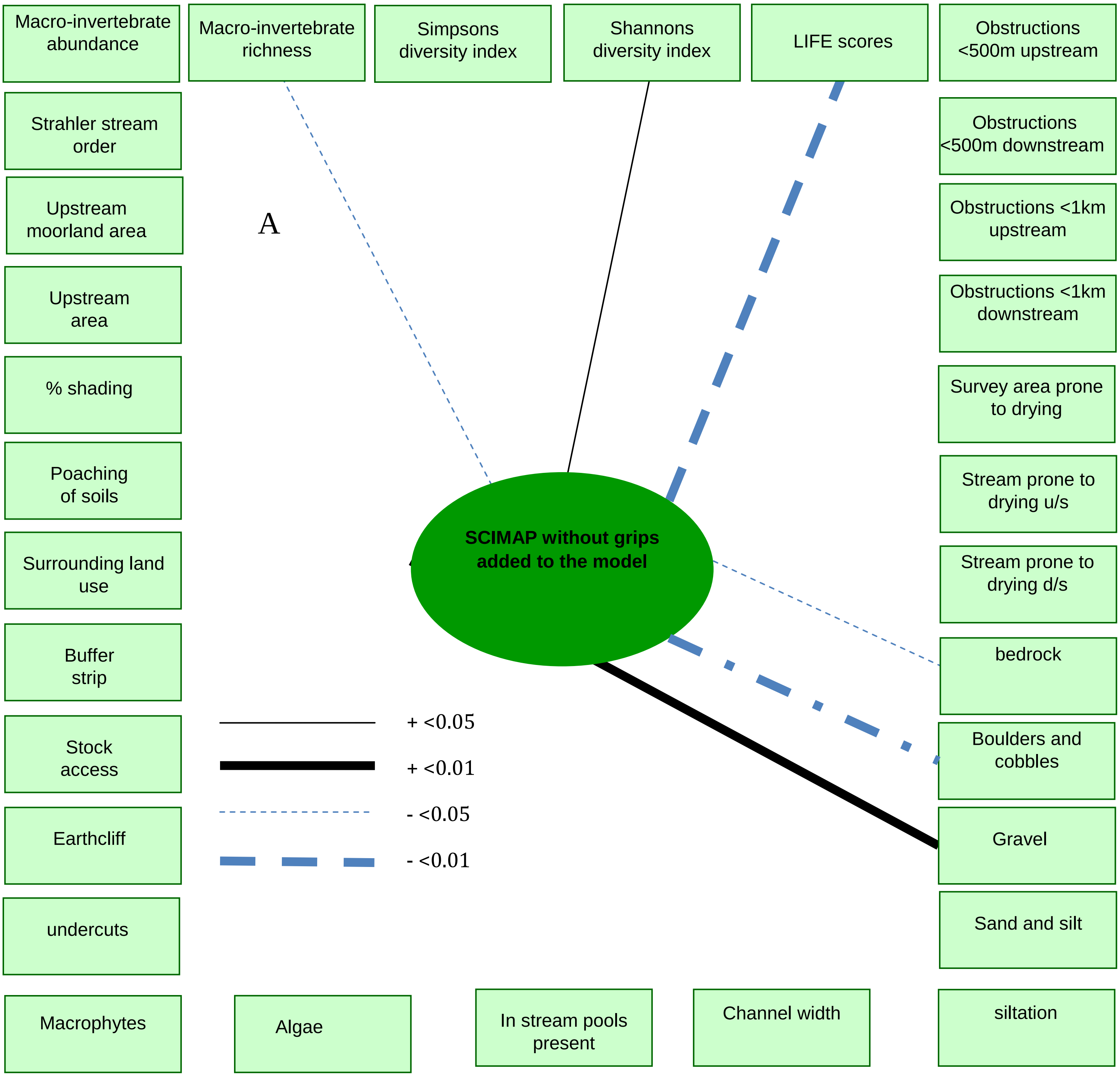
A second correlation analysis was carried out to ascertain which factors correlated with brown trout fry (rank average density). Rank average brown trout fry was used due to avoid the high level of 0 returns during the electrofishing surveys which skew the relationships. In addition, log transformation was not possible for the same reasons. Moreover, the two sites that gave high returns added to the complexities of the data and so ranking the data appeared to offer the most suitable approach to the analysis. The results of the correlation analysis can be seen in table 5.12. A number of interesting aspects are highlighted here, for example the absence of correlations between any of the macroinvertebrate measures or with percent shading. This suggests that factors other than prey availability are driving brown trout fry abundance. It was interesting to note that, whilst SCIMAP did not correlate with fine sediment or siltation, it does reveal itself as a positive relationship with brown trout fry in the case of $SCIMAP_G$.

Table 5.12: *Pearson correlations between rank average brown trout fry populations and the data variables collected*

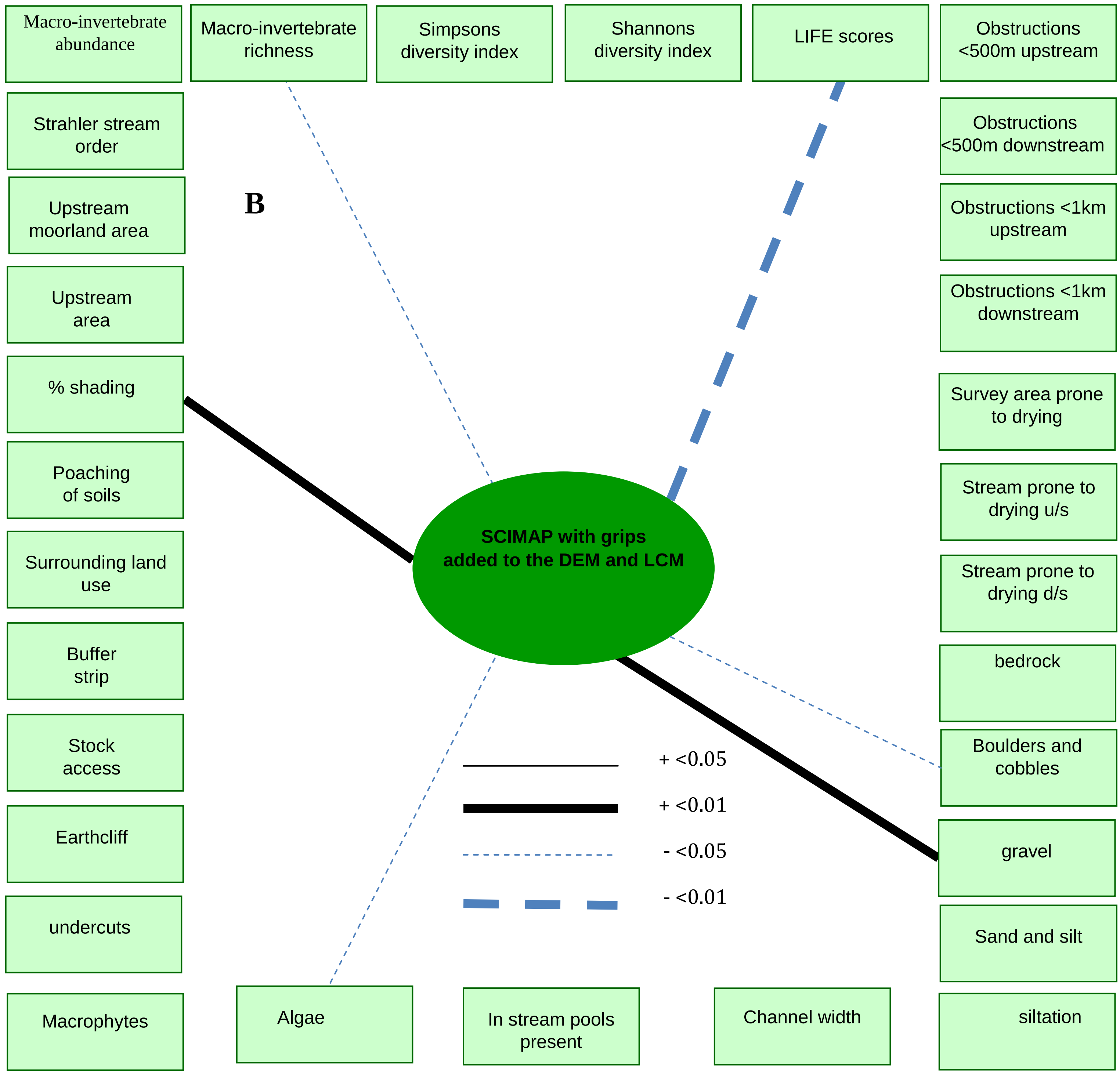
Factor	Rank average brown trout fry significance level and direction of relationship	
	0.05	**0.01**
Strahler stream order	-0.312 0.047	
$SCIMAP_G$	0.315 0.045	
Boulders and cobbles	-0.322 0.040	
Gravel		0.607 0.000
Sand and silt	0.306 0.051	
Algae		-0.564 0.000
Stream area prone to drying		-0.504 0.001

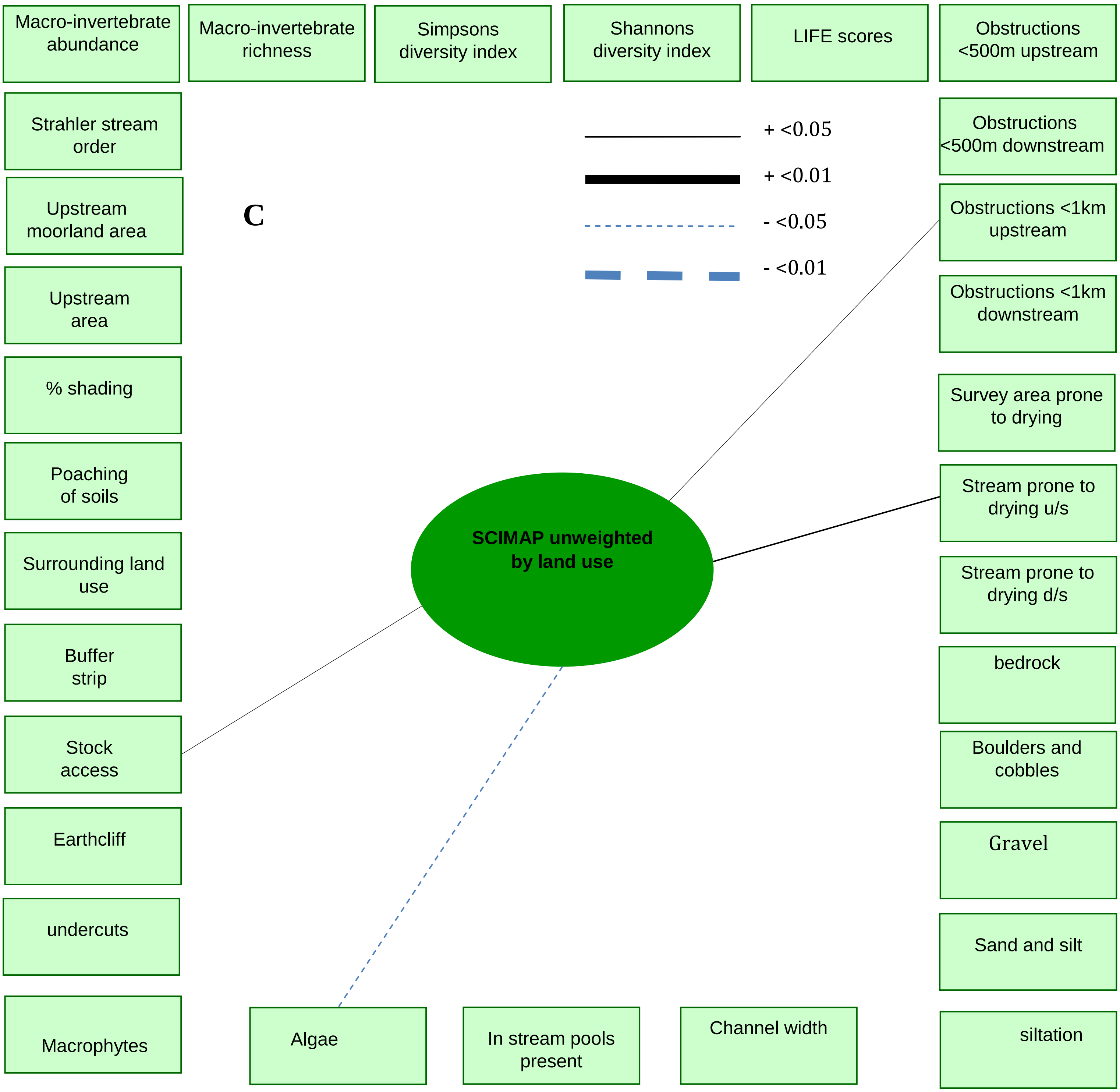
SCIMAP relationships with the other variables were interesting to note, in particular how the correlations change depending on which SCIMAP risk loading was used. $SCIMAP_U$ showed the least, and the weakest, correlations. SCIMAP with and without grips showed the same number of correlations (and the same number of $P<0.01$ and $P<0.05$), although they correlated with some of the same factors they did also display some difference. These relationships are visualised in figure 5.15 (A to C) on the following three pages. Correlations < 0.05 level of significance has not been included in the diagrams.

Figure 5.15 (overleaf, pages 242, 243 and 244)**:** *The correlations between the three SCIMAP runs and the other variables to see where the important relationships exist, their strength and direction. A: SCIMAP without grips added to the model, B: SCIMAP with grips added to the model and C: SCIMAP unweighted by land use.* (After Burt, 2010)



Macro-invertebrate abundance
Macro-invertebrate richness
Simpsons diversity index
Shannons diversity index
LIFE scores
Obstructions <500m upstream
Strahler stream order
Obstructions <500m downstream
Upstream moorland area
A
Obstructions <1km upstream
Upstream area
Obstructions <1km downstream
% shading
Survey area prone to drying
Poaching of soils
Stream prone to drying u/s
SCIMAP without grips added to the model
Surrounding land use
Stream prone to drying d/s
Buffer strip
bedrock
+ <0.05
Stock access
+ <0.01
Boulders and cobbles
- <0.05
Earthcliff
Gravel
- <0.01
undercuts
Sand and silt
Macrophytes
Algae
In stream pools present
Channel width
siltation


Macro-invertebrate abundance
Macro-invertebrate richness
Simpsons diversity index
Shannons diversity index
LIFE scores
Obstructions <500m upstream
Strahler stream order
Obstructions <500m downstream
B
Upstream moorland area
Obstructions <1km upstream
Upstream area
Obstructions <1km downstream
% shading
Survey area prone to drying
Poaching of soils
Stream prone to drying u/s
SCIMAP with grips added to the DEM and LCM
Surrounding land use
Stream prone to drying d/s
Buffer strip
bedrock
Stock access
+ <0.05
Boulders and cobbles
+ <0.01
Earthcliff
- <0.05
gravel
- <0.01
undercuts
Sand and silt
Macrophytes
Algae
In stream pools present
Channel width
siltation


Macro-invertebrate abundance
Macro-invertebrate richness
Simpsons diversity index
Shannons diversity index
LIFE scores
Obstructions <500m upstream
Strahler stream order
+ <0.05
+ <0.01
- <0.05
- <0.01
Obstructions <500m downstream
Upstream moorland area
C
Obstructions <1km upstream
Upstream area
Obstructions <1km downstream
% shading
Survey area prone to drying
Poaching of soils
Stream prone to drying u/s
SCIMAP unweighted by land use
Surrounding land use
Stream prone to drying d/s
Buffer strip
bedrock
Stock access
Boulders and cobbles
Earthcliff
Gravel
undercuts
Sand and silt
Macrophytes
Algae
In stream pools present
Channel width
siltation

The correlations between brown trout fry and the other factors are highlighted in the proceeding scatterplots (figures 5.16 to 5.19). These show the direction of the relationship and describe clearly where the significant relationships exist. Some of these relationships are not intuitive. For example, it was thought that the presence of boulders and cobbles would increase survival due to the refugia they present from environmental variables and through increasing the available territorial positions within the stream. Yet it appears that above a low percentage the presence of boulders and cobbles create unsuitable habitat for brown trout recruitment. This could be a function of the size of the fish in a resident stock. In populations with a high propensity for smolting and utilising the life cycle of sea trout there is a sexual dimorphism with more hen fish becoming migratory sea trout. These fish are able to use larger fraction of gravel to create a red. They also produce larger in eggs in greater numbers. In such locations the available gravel is likely to be of a large fraction and only large hen fish would be able to scrape a redd. Moreover, the flow rates after rainfall events may be too high to allow egg to fry survival in such locations.

Figure 5.16: *Rank average brown trout against the various substrate types. These highlight the increasing presence of finer fractions correspond with an increase in spawning gravels.*

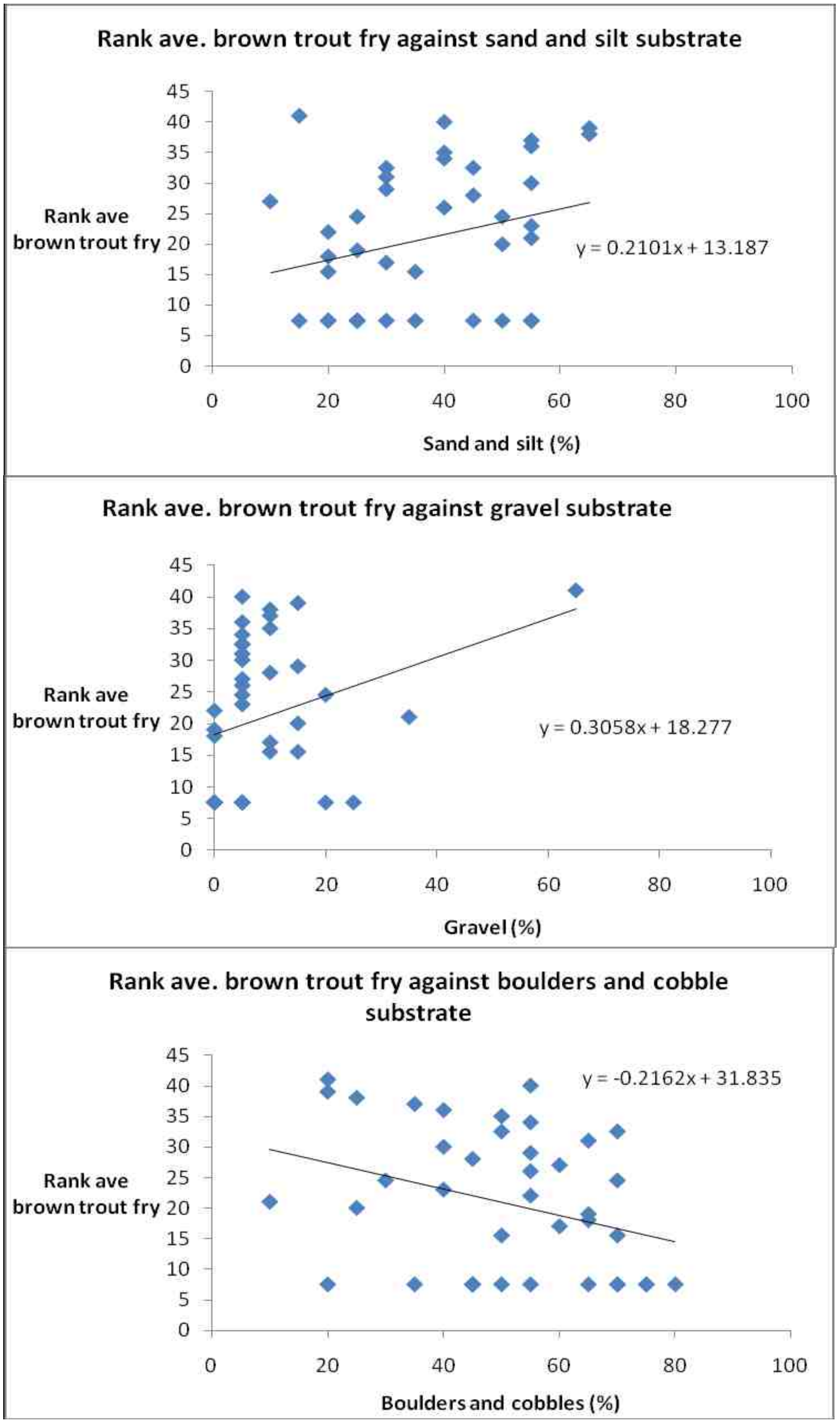


Figure 5.17: *Rank average brown trout against the three catchment-scale SCIMAP runs. It appears that SCIMAP$_G$ displays the best fit with fry populations.*

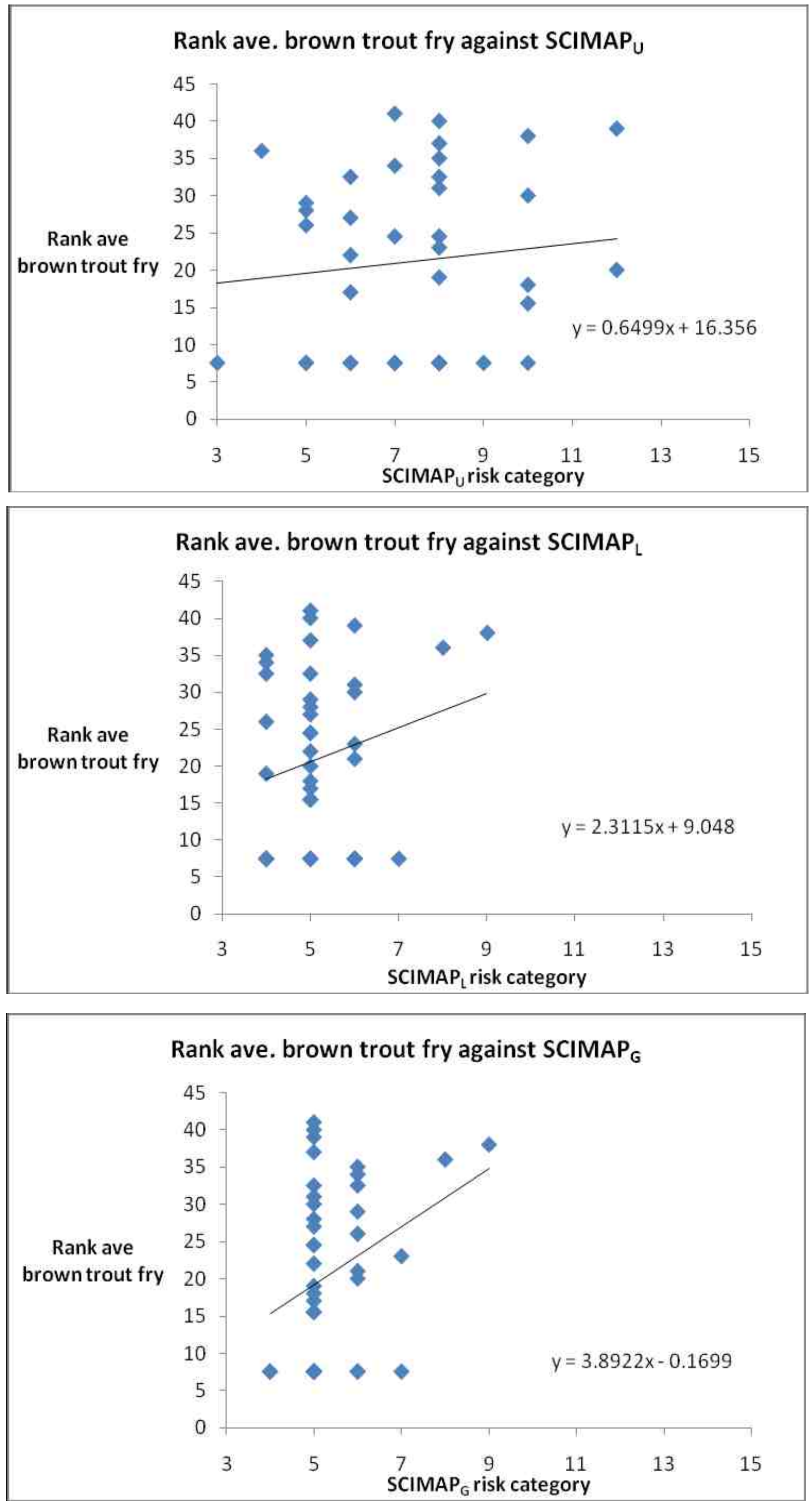
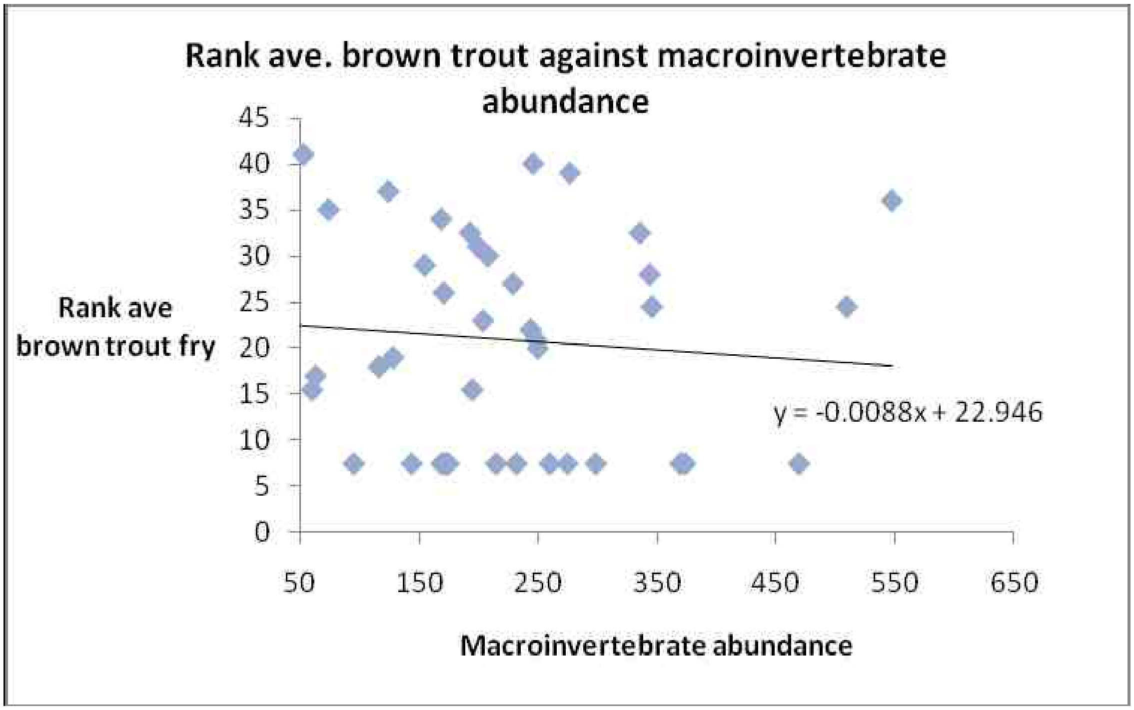


Figure 5.18: *Rank average brown trout against macroinvertebrate data.*



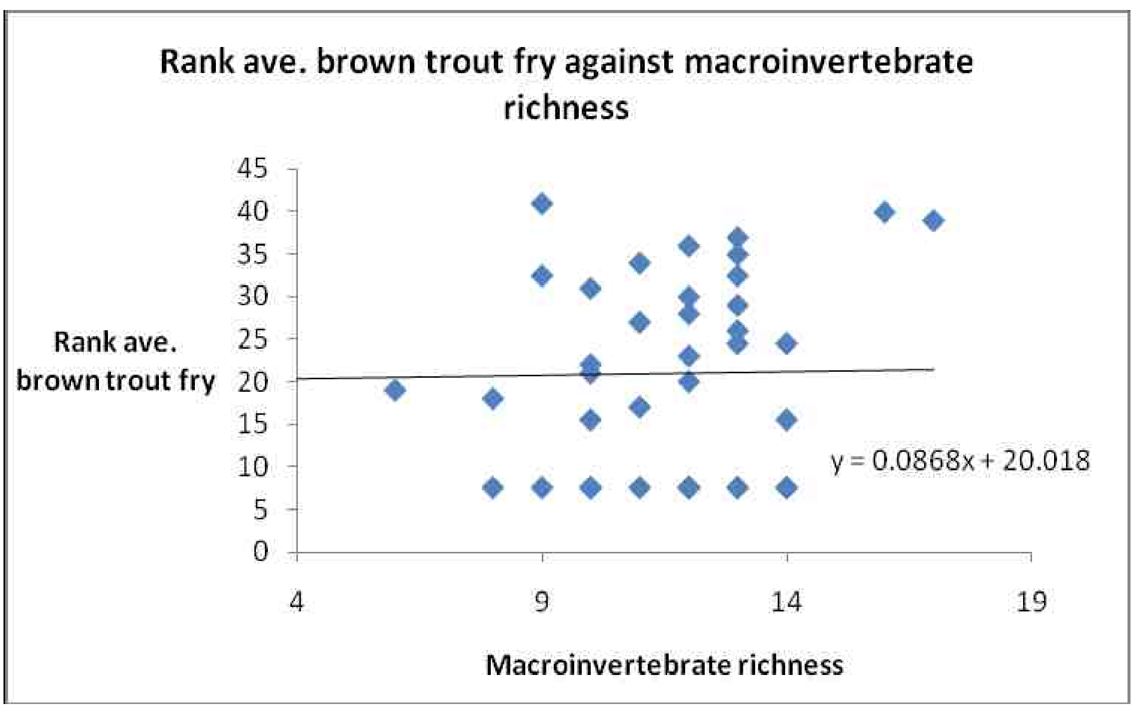
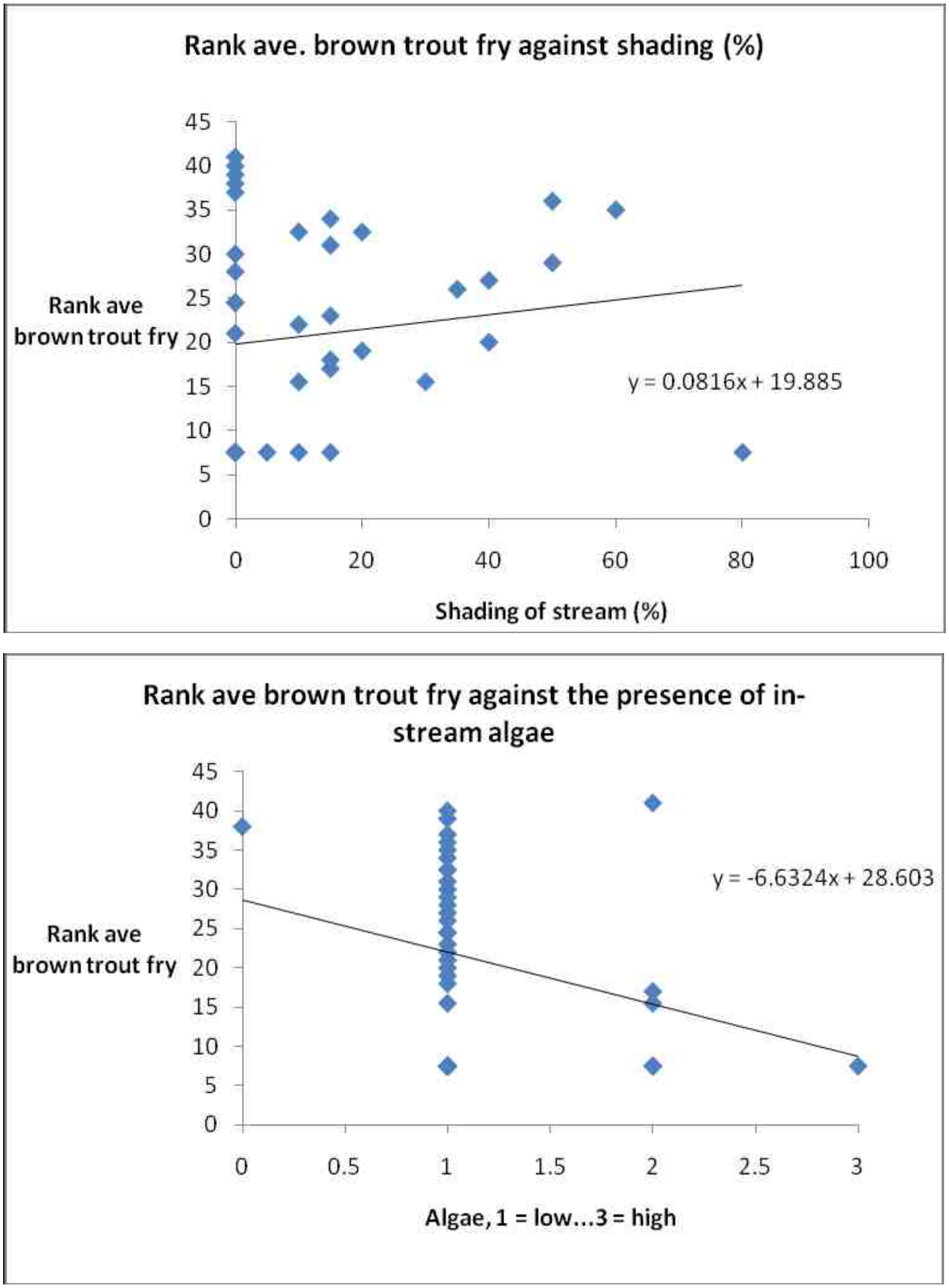


Figure 5.19: *Rank average brown trout against percent shading of the stream and algal presence within the channel.*



5.5.1 Multiple regression analysis of brown trout populations and physical variables

The brown trout bivariate correlation and regression analyses showed that in general it is the physical nature of the habitat and catchment that are the controlling factors on recruitment. The strongest correlations ($P<0.01$) arose from within the immediate habitat. These are substrate composition, algal growth and whether the stream area was prone to drying. The significant correlations at $P<0.05$ are Strahler stream order, presence of boulders and cobbles as well as $SCIMAP_G$. To decipher the weighting behind each of these relationships, stepwise multiple regression was taken for rank average brown trout fry against the variables revealed to be significant in table 5.13. Two variables returned as significant at the <0.01 level and two at the <0.1 level. The results are shown below:

Table 5.13: Stepwise regression of the significant correlations against rank average brown trout fry

step	1	2	3	4	5	6
constant	31.83	44.78	36.10	30.68	31.51	37.84
Boulders and cobbles	-0.216	-0.271	-0.098			
T-value	-2.12	-2.76	-0.79			
P-value	0.040	0.009	0.433			
Algae		-8.9	-12.3	-12.6	-12.6	-10.5
T-value		-2.47	-3.26	-3.38	-3.43	-2.81
P-value		0.018	0.002	0.002	0.001	0.008
Sand and silt			0.44	0.54	0.50	0.44
T-value			2.18	3.55	3.32	2.93
P-value			0.036	0.001	0.002	0.006
Stream prone to drying					-11.4	-14.7
T-value					-1.58	-2.02
P-value					0.123	0.051
Strahler stream order						-3.4
T-value						-1.77
P-value						0.085
S	11.3	10.6	10.1	10.1	9.86	9.59
R-Sq	10.37	22.76	31.52	30.35	34.74	39.98
R-Sq(adj)	8.07	18.69	25.96	26.69	29.45	33.31
Mallows C-p	15.8	10.5	7.4	6.0	5.5	4.4

The regression equation for these relationships is:

Rank average fry density = 37.8 – 10.5 (algae) + 0.44 (sand/silt) – 14.7 (stream prone to drying) – 3.41 (Strahler stream order)

This offered information on the direct relationships with the rank average brown trout fry populations. The stepwise regression revealed that a number of relationships are significant. These include algae (negative relationship) and substrate composition (boulders and cobbles negative relationship, sand and silt positive relationship). The substrate composition shows an interesting result as it displays that sand and silt has a positive relationship with brown trout fry whilst larger fractions of the bedload are negative. This appears to be a function of percentage composition closer to that which brown trout will utilise for spawning. A large proportion of boulders and cobbles would suggest higher flows than a large proportion of sand and silt. This could explain the relationships in that brown trout would show a preference for medium-sized fractions of substrate when spawning. Yet gravel displays a strong correlation with boulders and cobbles and not the smaller fractions. The precise controls linking substrate grain size and trout numbers needs more detailed examination therefore and perhaps a more detailed analysis of the broadly "gravel" grain sizes; this is discussed further below. The presence of algae displays a negative relationship with brown trout fry. Algae can be viewed as an indicator of nutrient enrichment and reduced dissolved oxygen levels. Both of these impacts are known to limit brown trout recruitment.

Beneath this initial level of analysis there are other levels of order that require exploration and therefore stepwise regression was run for each of the variables that correlated with trout fry in order to explore which relationships linked to each of these in turn (tables 5.14 to 5.18). This was carried out to enhance the knowledge of the impacts and the underlying relationships for each.

1) Stepwise regression for Algae shows:

Table 5.14: Stepwise regression of significant correlations against algae						
step	1	2	3	4	5	6
constant	0.9827	1.8565	1.8559	1.7850	1.7866	1.7441
Sand and silt	0.0184	0.0187	0.0147	0.0052		
T-value	3.17	3.40	2.68	0.73		
P-value	0.003	0.002	0.011	0.470		
$SCIMAP_G$		-0.161	-0.163	-0.144	-0.139	-0.137
T-value		-2.37	-2.54	-2.31	-2.25	-2.26
P-value		0.023	0.015	0.027	0.030	0.030
Poaching of soils			0.50	0.65	0.73	0.65
T-value			2.34	2.97	3.74	3.25
P-value			0.025	0.005	0.001	0.003
Macrophytes				0.73	0.91	0.83
T-value				1.99	3.37	3.06
P-value				0.054	0.002	0.004
Earthcliff						0.23
T-value						1.51
P-value						0.140
S	0.431	0.408	0.386	0.371	0.369	0.363
R-Sq	20.47	30.74	39.65	45.63	44.83	48.12
R-Sq(adj)	18.43	27.10	34.75	39.59	40.35	42.35
Mallows C-p	15.4	10.7	6.8	4.9	3.4	3.2

The regressions equation for this relationship is:

Algae = 1.79 – 0.139 ($SCIMAP_G$) + 0727 (poaching of soils) + 0.908 (emergent macrophytes)

This was an interesting regression as it showed that algae displayed a relationship with $SCIMAP_G$. This helps validate the model in this catchment; since algae correlates negatively with brown trout and $SCIMAP_G$ correlates negatively with algae, this suggest a positive link, albeit indirect, between connectivity and trout The delivery of fine sediments from peat soils will result in POC and other nutrients which would be expected to provide a source of phosphate and nitrate for algal growth. However, on the low order streams sampled this risk did not appear to be realised. Poaching of soils perhaps offers a similar nutrient function in the form of sediment attached phosphate that may become disassociated during low dissolved oxygen conditions, for example during a night sag of oxygen due to algal respiration. This would offer a positive feedback route that exacerbates algal growth and thus reduces brown trout fry viability.

This is counter to the effect seen in Table 5.13. The final relationship is with the presence of macrophytes. Fine sediment delivery form grips and poaching of surrounding soils will provide the substrate required for these plants to become established.

2) Stepwise regression for sand/silt substrate shows:

Table 5.15: Stepwise regression of significant correlations against sand and silt

step	1	2	3	4
constant	30.33	26.73	22.69	18.32
Boulders and cobbles T-value P-value	-0.447 -5.57 0.000	-0.382 -6.04 0.000	-0.325 -4.96 0.000	-0.266 -3.88 0.000
Emergent macrophytes T-value P-value		26.6 5.26 0.000	25.5 5.26 0.000	22.5 4.63 0.000
Earthcliff T-value P-value			6.3 2.22 0.032	6.3 2.31 0.027
Siltation T-value P-value				5.4 2.12 0.041
S R-Sq R-Sq(adj) Mallows C-p	8.88 44.29 42.86 38.6	6.85 67.74 66.04 8.8	6.52 71.54 69.23 5.6	6.23 74.71 71.90 3.3

The regressions equation for this relationship is:

Sand and Silt = 18.3 – 0.266 (boulders and cobbles) + 22.5 (emergent macrophytes) + 6.26 (earthcliff) + 5.4 (siltation)

Unsurprisingly, this shows that sand and silt as a substrate type has a negative relationship with boulders and cobbles and a positive one with macrophytes. These relationships have been touched on in the above paragraphs. Importantly, this regression shows that the presence of earthcliffs provides a source of fine sediment. Much of the river Ure system has clear signs of earthcliffs that are clearly driven by stock access and the lack of buffer strips. This suggests that surrounding land use may outweigh catchment-scale delivery mechanisms in terms of fine sediment within this upland catchment. The river Ure catchment has higher intensity land uses in comparison to neighbouring catchments. This is most likely due to two reasons. First, the catchment has a wide a floodplain suitable for meadow and pasture which encourages higher stocking rates. Secondly, the presence of the Wensleydale Creamery encourages a high proportion of dairying within the catchment. In neighbouring catchments there are now very few dairy herds.

3) Stepwise regression for the stream area prone to drying shows:

Table 5.16: Stepwise regression of significant correlations against stream area prone to drying

step	1	2
constant	0.0000	-0.1549
Boulders and cobbles	0.222	0.198
T-value	2.95	2.65
P-value	0.005	0.012
Stream prone to drying upstream		0.0032
T-value		1.79
P-value		0.082
S	0.200	0.194
R-Sq	18.23	24.59
R-Sq(adj)	16.14	20.62
Mallows C-p	3.8	2.6

The regression equation for this relationship is:

Stream are prone to drying = - 0.155 + 0.198 (stream prone to drying upstream) + 0.00320 (boulders and cobbles)

Whilst this variable was not significant in the stepwise regression for rank average brown trout fry; however, it did add 4% to the variance explained and so stepwise regression was run to ascertain the underlying relationships here. This factor may indeed become significant in the future if the trends in precipitation displayed in chapter 3 continue. Reduced summer rainfall may drive an increasing issue with drying streams within the catchment. The relationship here suggests that boulders and cobbles are important and is probably a function of location within the catchment. Presently the streams prone to drying (either at the survey site or up and downstream) are few. However, all but one are located at the upper reaches of the catchment where changes in hydrological connectivity respond rapidly to changing patterns of rainfall and all the sites have small upstream contributing areas. This appears to be an issue that could come to be a serious control on brown trout fry in the future. Restoration to ameliorate such an impact would require changes to the physical condition of the streams with pools and refugia needed to allow survival during periods with little rainfall.

4) Stepwise regression for Strahler stream order shows:

Table 5.17: Stepwise regression of significant correlations against Strahler stream order

step	1	2	3
constant	1.580	1.854	1.723
Upstream contributing area (Km2) T-value P-value	0.090 6.00 0.000	0.082 5.68 0.000	0.053 2.23 0.032
Obstructions upstream (<500m) T-value P-value		-0.47 -2.49 0.017	-0.45 -2.43 0.020
River width (m) T-value P-value			0.107 1.49 0.144
S R-Sq R-Sq(adj) Mallows C-p	0.624 48.00 46.67 5.3	0.586 55.31 52.96 1.4	0.577 57.85 54.44 1.3

The regressions equation for this relationship is:

Strahler stream order = 1.85 + 0.0817 (upstream contributing area – km2) – 0.471 (obstructions upstream<500m)

As with the streams showing a propensity to drying, Strahler stream order did not show up as significant (Table 5.13). However, it did add 5% to the variance explained and indicates that resident brown trout stocks favour low-order streams. The results here show that as both upstream contributing area and river width increase so does the Strahler stream order. This is a simple function of increasing discharge and is as expected. Obstructions upstream are more prevalent on low-order streams, again as expected.

$SCIMP_G$ being the only SCIMAP run at the catchment-scale that displayed significance with brown trout fry populations was also tested using a stepwise regression.

5) Stepwise regression for $SCIMAP_G$ shows:

Table 5.18: Stepwise regression of significant correlations against $SCIMAP_G$			
step	1	2	3
constant	9.062	9.057	7.933
LIFE scores T-value P-value	-0.438 -5.20 0.000	-.0480 -7.60 0.000	-0.421 -7.14 0.000
% shading T-value P-value		0.0255 5.69 0.000	0.0241 5.98 0.000
Gravel T-value P-value			0.0178 3.29 0.002
S R-Sq R-Sq(adj) Mallows C-p	0.739 40.97 39.46 46.9	0.550 68.14 66.46 10.3	0.490 75.36 73.37 2.0

The regression equation is:

$SCIMAP_G$ = 7.93 - 0.421 LIFE scores + 0.0241 % shade + 0.0178 gravel

Here the LIFE scores suggest that as the $SCIMAP_G$ risk category increases the LIFE score decreases. Many of the organisms that have high LIFE scores (e.g. Plecoptera – stonefly; Heptagiinadea - mayfly) also require high dissolved oxygen and ≥ gravel bedload fractions over the other substrate fractions. High $SCIMAP_G$ probably controls LIFE scores rather than vice versa, indicating that very well connected sites in gripped low-order basins are not conducive to these organisms. Sites with low $SCIMAP_G$ scores would therefore seem more conducive to brown trout. Whilst gravel shows a positive relationship, this is likely to be an artefact of location; fine sediments are certainly present in storm runoff but are unable to settle out at the river bed surface due to the high flow velocities at these locations.

These stepwise regressions highlight that beneath the causal relationships placing controls on brown trout fry populations there other subsets of information influencing that may too be placing controls on stream biota. Thus, to develop knowledge, an exploration of in-stream ecology must be able to delve beneath the apparent relationships to identify the full suite of impacts and relationships within these ecosystems. This second suite of information is of interest as it begins to develop clearer knowledge how land use and larger--scale impacts drive through the catchment to impact at a local setting. This perhaps offers restorers of the ecosystems the chance to widen investigations beyond the reach scale to place each location in its full spatial setting. This catchment-scale reach will be further explored in the next section where all the collected data sets are tested against rank average brown trout fry to highlight if any changes in the significant relationships occur.

5.5.2 Incorporating all variables to describe brown trout fry populations

Table 5.19 (overleaf) displays the results of a stepwise regression testing all the gathered data against rank average brown trout fry populations.

Table 5.19: Stepwise regression of all collected data against rank average brown trout fry

Step	1	2	3	4	5	6	7	8	9	10
constant	31.83	44.78	36.10	30.68	24.76	22.67	26.20	19.16	26.05	26.01
Boulders & cobbles	-0.216	-0.271	-0.098							
T-value	-2.12	-2.76	-0.79							
P-value	0.040	0.009	0.433							
Algae		-8.9	-12.3	-12.6	-14.1	-13.5	-13.5	-12.4	-13.7	-12.5
T-value		-2.47	-3.26	-3.38	-3.90	-3.88	-3.97	-3.66	-4.10	-3.73
P-value		0.018	0.002	0.002	0.000	0.000	0.000	0.001	0.000	0.001
Sand and silt			0.44	0.54	0.55	0.73	0.75	0.74	0.78	0.88
T-value			2.18	3.55	3.83	4.41	4.62	4.66	5.05	5.46
P-value			0.036	0.001	0.000	0.000	0.000	0.000	0.000	0.000
Simpson's					1.95	2.54	2.48	2.52	2.55	2.32
T-value					2.24	2.85	2.85	2.98	3.11	2.88
P-value					0.031	0.007	0.007	0.005	0.004	0.007
Siltation						-8.3	-7.6	-9.0	-10.1	-10.5
T-value						-1.97	-1.84	-2.20	-2.52	-2.69
P-value						0.056	0.075	0.035	0.017	0.011
Stock access							-5.2	-5.3	-6.9	-7.0
T-value							-1.66	-1.75	-2.23	-2.35
P-value							0.106	0.090	0.033	0.025
Gravel								0.166	0.172	0.172
T-value								1.74	1.87	1.91
P-value								0.090	0.071	0.065
Land use									-2.6	-2.5
T-value									-1.82	-1.84
P-value									0.077	0.076
Earthcliff										-6.7
T-value										-1.72
P-value										0.095
S	11.3	10.6	10.1	10.1	9.56	9.21	8.99	8.74	8.45	8.21
R-Sq	10.37	22.76	31.52	30.35	38.64	44.63	48.68	52.89	57.21	60.83
R-Sq(adj)	8.07	18.69	25.96	26.69	33.67	38.48	41.35	44.57	48.13	51.03
Mallows C-p	4.6	0.8	-1.2	-2.7	-4.6	-5.3	-5.2	-5.2	-5.2	-4.8

The regression equation is:

Rank average brown trout fry = 26.2 - 13.5 algae + 0.749 sand and silt + 2.48 Simpsons (1/total) - 7.59 siltation - 5.23 stock access

Whilst this shows a high degree of agreement with the earlier analyses (e.g. algae, sand and silt) there are some notable differences. This run suggests that stock access and siltation are important as negative controls on brown trout recruitment. This corroborates work done by others (Armstrong *et al*, 2003; Klemetsen *et al*, 2003; Theurer *et al*, 1998; Elliot, 1994). Here Simpson's diversity index displays a strong positive control on brown trout recruitment showing that species diversity within the macroinvertebrate community can be seen as an important factor for brown trout and again fits with earlier findings (Skoglund and Barlaup, 2006; Armstrong *et al*, 2003; Klemetsen *et al*, 2003; Elliot, 1994). This is intuitive and it was surprising when this did not display a significant relationship in the correlation analysis, but is probably a result of collinearity between independent variables. It is interesting too that gravel appears as a positive control on brown trout numbers. Statistically, sand/silt appears a stronger control but this may not necessarily indicate causation. As noted above, further work on substrate grain size analysis to better discriminate between categories, might be helpful in future studies.

The results that displayed significance were processed through a stepwise regression the results are below.

1) Stepwise regression for Simpsons diversity index shows:

Table 5.20: *Stepwise regression of significant correlations against Simpsons diversity index*			
step	1	2	3
constant	1.9670	0.8496	0.4032
Shannons diversity index	1.30	1.24	1.14
T-value	3.70	3.91	3.93
P-value	0.001	0.000	0.000
In-stream pools		0.84	0.88
T-value		3.21	3.72
P-value		0.003	0.001
Obstructions upstream (<500m)			1.21
T-value			3.04
P-value			0.004
S	1.54	1.38	1.25
R-Sq	26.03	41.84	53.44
R-Sq(adj)	24.13	38.77	49.66
Mallows C-p	21.9	11.3	4.1

The regression equation is

Simpsons diversity index (1/D) = 0.403 + 1.14 Shannons diversity index + 1.21 Obstructions upstream, (<500m) + 0.884 pools present

This further highlights that habitat diversity results in increased diversity amongst ecological communities. Here the results highlight that evenness is requires structure and diversity.

2) Stepwise regression for Stock access to streams shows:

Table 5.21: *Stepwise regression of significant correlations against stock access to streams*			
step	1	2	3
constant	0.9516	0.9931	0.5889
Buffer strips	-0.371	-0.245	-0.231
T-value	-5.55	-3.24	-3.31
P-value	0.000	0.003	0.002
Percent shading		-0.0091	-0.0094
T-value		-2.85	-3.19
P-value		0.007	0.003
$SCIMAP_U$			0.056
T-value			2.82
P-value			0.008
S	0.349	0.321	0.295
R-Sq	44.13	53.95	62.10
R-Sq(adj)	42.70	51.53	59.03
Mallows C-p	17.5	10.0	4.0

The regression equation is

Stock access to streams = 0.589 - 0.231 buffer - 0.00939 % shade + 0.0558 $SCIMAP_U$

These results display is reducer wherever buffer strips exist. The relationship with shading of the stream is intuitive as the greater bankside structure the less opportunity livestock have to access the bank and the stream. $SCIMAP_U$ provides a surface flow index but this is most likely an artifact of location (i.e. floodplains where farming is more intensive). The surface flow has been well managed in these locations and in many locations now follows artificial sub-surface routes.

3) Stepwise regression for siltation:

Table 5.22: *Stepwise regression of significant correlations against siltation*					
step	1	2	3	4	5
constant	0.08618	-0.01541	-0.71211	-0.61514	-0.63945
Sand and silt	0.0232	0.0226	0.0224	0.0194	0.0188
T-value	4.59	4.88	5.52	4.80	4.78
P-value	0.000	0.000	0.000	0.000	0.000
D/S obstructions (<500m)		0.336	0.378	0.344	0.327
T-value		2.90	3.71	3.53	3.44
P-value		0.006	0.001	0.001	0.002
Macroinvertebrate richness			0.061	0.052	0.038
T-value			3.55	3.14	2.10
P-value			0.001	0.003	0.043
Poaching of soils				0.38	0.42
T-value				2.31	2.64
P-value				0.027	0.012
Simpsons index (1/D)					0.051
T-value					1.83
P-value					0.076
S	0.376	0.345	0.302	0.285	0.276
R-Sq	35.03	46.83	60.34	65.46	68.48
R-Sq(adj)	33.36	44.04	57.13	61.62	63.98
Mallows C-p	31.2	20.8	8.6	5.3	4.1

The regression equation is

Siltation = - 0.615 + 0.0194 sand and silt + 0.344 Obstructions downstream, (<500m) + 0.0520 macroinvertebrate richness + 0.378 poaching of soils

Here the clear relationships with sand and silt and poaching of soils are as expected. The relationship with macroinvertebrate richness again suggests that habitat diversity offers makes the community structure more robust. Downstream obstructions can be explained through location, there are many waterfalls on the low order streams of the catchment so it can be expected that these relationships appear despite being more likely an artifact of location.

These results will be discussed further in the following chapter.

Chapter 6 Explaining brown trout fry populations

6.1 Introduction

Whilst aquatic organisms are good bio-indicators of river health, these components of an ecosystem are unable to provide a complete picture of condition. Monitoring only biotic components of a river system poses a number of concerns as:

1) The chosen group may be insensitive to the dominant stressors;
2) There may be a time lag between disturbance and biotic response;
3) Monitoring biota may highlight a change has occurred but miss the underlying cause of the change (Norris *et al*, 2007).

Due to these issues, it is important to not only monitor biotic components but also the physical environment. This needs to incorporate the pertinent land use to ensure that the full suite of controls on ecological condition is represented in the data collection (Norris *et al*, 2007). River ecosystems and processes are a response to a continuum of the prevailing climate, geological condition, topography and human influences that all operate over a range of spatial and temporal scales (Macklin *et al*, 2009). Thus, research has to be broad enough to account for these at the appropriate scales, which in terms of river processes is now widely accepted as being at the catchment scale (Burt and Pinay, 2005).

The history of human land use displays rapidly increasing over-bank sedimentation on to floodplains that acts in tandem with advances in farming technology; Macklin *et al*, (2009) note that marked increases in catchment erosion always occurs during periods of land use intensification. Thus, there has been a significant increase in this process of erosion, delivery and conveyance of sediments for the past two hundred years with sharp increases in the process that neatly couple with advances in agricultural technology and intensity. Such advances nearly always prime soil for erosion. In terms of overbank sedimentation, there must be sources of connected soil erosion upstream of the impact location; this too is the case with in-stream sedimentation. Identifying and reducing the impact of these sources is one of the many imperatives in river restoration.

This rising sedimentation rate in tandem with growing agricultural land use is just one example of how human developments can impact river ecosystems. Brisbois *et al* (2008) concluded that agriculture led to increased nutrient delivery, large fluctuations in dissolved oxygen, increased turbidity, high chlorophyll *a* content, as well as altered macroinvertebrate populations and communities in river networks. Hence, a restoration policy that tackles single issues is likely to miss the multiple impact nature of river degradation, even though some restoration methods may work to reduce more than one impact. It appears that there is no single overarching policy for managing biodiversity at the river or floodplain scale (Looy *et al*, 2006). The river continuum concept (Vannote *et al*, 1980) suggests that river systems experience gradual change in their dynamics and ecosystems on a downstream gradient (figure 6.1). They comment that (p.130), '*the structural and functional characteristics of stream communities are adapted to conform to the most probable position or mean state of the physical system*.' However, it has since been argued that disorder in the river continuum arising from different land uses and intensity, subcatchment condition and geological variation can add disorder (or at least disrupt discontinuities) to a river network which disrupts the gradual change suggested by the RCC (Looy *et al,* 2006: Romanuk *et al*, 2006: Statzner and Higler, 1985). Indeed, some changes act in what appears to be an exponential manner. For example, particulate phosphorus concentration has been shown to increase 20-fold within the lowland reaches of the river Swale (an adjoining catchment to the Ure) whereas there is only a doubling of the concentration in the transitional zone (Bowes *et al*, 2003).

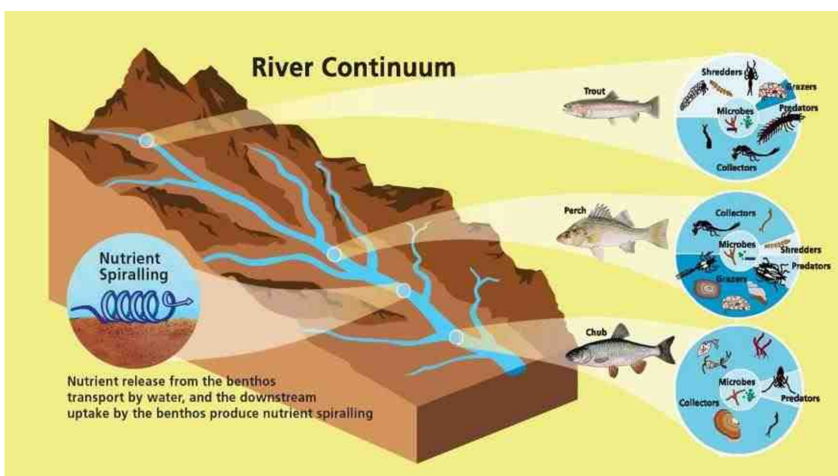


Figure 6.1: The River Continuum Concept suggests that rivers undergo gradual downstream changes in their dynamics and ecosystems. However, there is evidence that disorder is prevalent in rivers due to land use, subcatchment condition and other factors that disrupt gradual change.

This notion of disorder in river systems is perhaps most keenly felt in catchments that are dominated by agriculture. In such locations changes in management between farms, or even fields, can have immediate and significant impact on the river system. Thus, it is important to understand how these impacts interact, their operating scales (riparian farmland to full catchment), the spatial distribution throughout a river system, and how these impacts expand or contract through time (e.g. poorly managed daily milking operations through to seasonal alterations in rainfall and thus hydrological connectivity). This research has explored an upland catchment in terms of the linkages between stream biota, catchment processes and agricultural practices.

6.1.1 Catchment cascades

Looy *et al* (2006) argue that minimal levels of disorder represent river reaches that are fairly independent of (i.e. well separated from) upstream processes such as transfers of energy, material and propagules. In contrast, they remark that in high-level disorder reaches connection with upstream reaches and other components of the riverscape are important factors. Thus, increased hydrological connectivity alone could be said to enhance disorder. It is in these high-disorder reaches where Looy *et al* (2006) suggest restoration potential is maximised and the emphasis at such locations should be on repairing processes. Thus, it is important to appreciate both local and cascading impacts arising from land and river usage in catchments (Jakeman *et al*, 1999). Such an appreciation will allow judgements to be made on where each location sits on the disorder 'scale' and so how much influence the catchment possesses over each location along the river.

The results presented in chapter 5 suggest that cascades are important. However, adjacent land uses can become the dominant control if agricultural intensity is high. Poff (1997) comments that increased habitat structure allows refugia from the impacts that flush through systems, thus allowing survival through periods of increasing stress. In a similar manner, high-intensity adjacent land use can perhaps mask other stressors that may be operating at overarching scales. Addressing localised impacts may enhance the habitat and ecology of the river reach; however, it may also reveal other stressors that come to the fore after 'improvements' are made. So, even where adjacent land use is the major stressor, an appreciation of other, process-driven, controls on river ecology is required to ensure the system is understood adequately in order that multiple stressors can be addressed. In addition, each impact is likely to have a series of causal relationships that must be accounted for if restoration is to work. This highlights the need to be aware of process cascades which are important whether they act at the local or catchment scale.

In recognition of this, organisations such as the Yorkshire Dales Rivers Trust are embarking on a number of catchment-scale projects that aim to understand the more significant pressures relative to all other locations within the landscape. The process of identifying these pressures requires a mix of traditional surveying methods and more modern techniques that include remote sensing, GIS methodologies and modeling techniques. These rapidly developing methodologies add detail to traditional knowledge development allowing informed decisions to be made about ecological condition and restoration techniques. Small charities such as rivers trusts are now being offered the opportunity to embark on restoration projects due to grants from charitable trusts and government agencies (e.g. Defra, Natural England and the Environment Agency) that have the EU WFD as a primary driver.

However, to ensure that resources are targetted optimally, there needs to be the capacity to provide rapid assessment of catchment condition to provide a targetted approach to restoration that will sit well with grant-giving agencies and farming communities who, without sound evidence, may become cynical. Moreover, these new process-driven projects allow a new and better informed period of river restoration that will likely provide improved opportunity of meeting targets for improving ecological condition of rivers. Whilst some authors argue that the impact of land use change on the hydrological regime of a catchment cannot be generalised (Ott and Uhlenbrook, 2004), the methodologies utilised to explore a catchment can follow the same approach. Thus, templates for catchment investigations can be developed allowing a good start towards identifying the offending variables when it comes to depleted river health. Such a template could provide policy drivers with the tools to achieve river restoration at the transnational scale required.

The catchment cascade diagrams offer a quick visual of the possible routes that impacts may take through a catchment. They describe process driven impacts and whilst they only offer suggestions on where the routings may occur they do provide a good start when creating a plan for investigations as they provide a focus. In this way they can help explorations in a catchment.

6.1.2 Revisiting the aim and objectives of this research.

The aim of this research was to combine advances in remote sensing, Geographical Information Systems (GIS), catchment-scale modelling and ecological survey techniques with current awareness of salmonid species, specifically brown trout fry populations, to develop an effective approach to the ecological restoration of habitat through the prioritisation of location and management options. The case study chosen was the upper Ure catchment upstream of Aysgarth Falls which is believed to pose an effective barrier to both anadromous forms of brown trout and Atlantic salmon. This allows a resident brown trout fishery to be explored without the added complications of migratory sea trout or salmon entering the system and perhaps skewing population numbers and breeding success within the catchment. However, there have been reports of Atlantic salmon, and possibly sea trout, jumping the lower falls at Aysgarth which pose the greater barrier of this series of three falls. These reports suggest that this only occurs during very high flows and in such low numbers that populations of salmonids within the catchment remain a largely resident stock. Moreover, these long-range migratory forms would most likely utilise larger gravels in higher-order streams. Thus, there is a reasonable level of confidence that recruitment at locations surveyed arise from only resident stocks. The objectives of this work provided the necessary steps in order to achieve the aim. These are listed below:

Objective 1: To review and synthesise in-stream, riparian and catchment scale controls on salmonid habitat, focusing on brown trout fry populations, in order to formulate a set of hypothesis for further investigation.

Objective 2: To employ advances in remote sensing, GIS and modelling to explore land use risk at the catchment scale that links to the in-stream habitat scale, in particular the risk of fine sediment delivery from the wider catchment.

Objective 3: To identify qualitative methods in data-poor catchments for testing model predictions and to employ the experience of agricultural communities in testing these predictions.

Objective 4: To use the data acquired under 2 to investigate hypothesis formulated in 1 to test which impacts on brown trout fry populations are important and to discuss the results in the context of model testing and ecological restoration.

Objective 1 was achieved in Chapter 2, objective 2 in Chapter 4 and objective 3 in Chapter 5 and will be further discussed here. Objective 4 was introduced in Chapter 5 through the presentation of the results and will be expanded in this chapter as the implications arising from the results are discussed. This will complete the research and offer insights into how data poor catchments can be explored in terms of identifying the impacts and implementing restoration methods.

6.2 SCIMAP farm scale

The farm-scale run of the SCIMAP model offered a possible validation method of the outputs. It was utilised in this research to test the performance of the model in predicting fine-sediment delivery from land to the stream network. To avoid complications in the process, it was considered more appropriate to interview members of the agricultural community about the land they manage only and not approach subject matter on which they may have less expertise or experience, for example, how fine sediment routes through the catchment and into the stream network. Validation of in-stream fine sediment, which identifies the streams most likely to be delivering disproportionate amounts of fine sediment into the river network in comparison to their upstream area (or streams where the rate of accumulation of fine sediment/risk is greater than the rate of dilution), will be considered in the next section by assessing the three model outputs and later against brown trout fry populations along with other, multiple, impacts on the species. The following discusses the results of this initial step in the SCIMAP validation process within the catchment.

Running SCIMAP at the field scale, and working with the farming community to assess the fit with actual processes, was considered important in order to ascertain how well SCIMAP represented the hydrology and erosion potential of the catchment. In addition, there are secondary reasons for engaging with the farming community in this manner including bringing in new forms of peer review into the scientific process. Lane *et al* (2006) offer the possibility that by extending the peer review process out to non-traditional reviewers there will be the risk of vested interests misleading the scientific community. This may become a more appealing option if there is a perceived threat to their specific sector. In addition, misleading comments may arise from confusion between the interviewer and interviewee. These are difficult issues to avoid. Without a high level of expertise in interview techniques, those who choose to mislead will be able to offer misinformation that can be taken on trust. The possibility that this occurred during the farm-scale SCIMAP modelling and subsequent farmer interviews has to be considered. However, all the farmers where known to the interviewer through working relationships that had been built over the previous six years and each farmer was carefully chosen in conjunction with a National Park Ranger who considered these farmers to be honest and forthright. Indeed, during the interviews there was no sense that any deliberate attempt to mislead was taking place and each farmer appeared to be acting in an open and honest manner. This was to the point that contentious issues were often discussed and, on at least two occasions, these discussions delved into farm practice that could be considered as sub-optimal in terms of possible environmental impact.

Despite this confidence in the interviewee’s motives there was a sense that difficult concepts led to a degree of confusion. This was to be expected, as even though the farmers were clearly very knowledgeable and aware of the land they manage, and the farming systems of the dales in general, some concepts were beyond their experience in terms of how surface flow connects erosion sources. This may because, 1) SCIMAP is time-integrated with predictions meant to be valid over many decades, or, 2) SCIMAP combines erosion with connection. This can result in counterintuitive results due to some steeply sloping fields not being flagged as risky simply because they are not connected to a watercourse by surface flow, whereas less steep land may be highly risky according to SCIMAP as there is a delivery pathway. This makes validation through local knowledge difficult as it is simple to see areas of erosion but more difficult to see where these connect.

Any confusing aspects broached in the initial interviews were dealt with during the walkover surveys and so there was an impression that accurate information was gathered from each farmer. This became more apparent when the local nuances in hydrology were discussed. All the farmers understood water movement across their holdings, displaying a deep knowledge of the sources, sinks and routes for the local hydrological flow paths. This offered confidence in the process of information gathering during these interviews and walkovers. This confidence was further enhanced when F5 provided an insight into the manner of the under-drainage in the meadows and pasture fields and how these would likely offer an alternative pathway for fine sediment delivery. F5 suggested that some of these subtle, and not so subtle, re-routing processes would likely result in fine sediment being delivered at similar locations identified in the farm-scale SCIMAP modelling. Research supports the assertion that subsurface routes may be preferential pathways for fine-sediment delivery (Deasy *et al*, 2008). This offers a level of validation despite farm management having apparently negated the risks in terms of erosion sources connected by surface flow at a number of locations. The fact that these drains may act as alternate routings was due to the nature of the traditional drainage method of the Yorkshire Dales, and other upland catchments. An interesting observation was that in the Higher Level Scheme these drains had to be repaired in the traditional manner despite plastic drainage blocking this possible sediment route. There appears to be a trade-off here between traditional methods and resource protection.

Despite the information gathered at this scale, the results of SCIMAP at the farm-scale were, in general, inconclusive. This was due to a number of reasons. Initially, there was confusion as to what constituted connected sediment sources. To begin with, discussions were directed towards locations where water stood on the field or where rivers and lakes expanded out on to the floodplain extending lateral connectivity during flood events. The concept of connected sediment sources was quite difficult to explain, in part because it is a hard concept to visualise due to locations where, 1) water extended onto the floodplain and, 2) obvious locations of erosion occurred. These were quickly identified by the farmers as locations of fine sediment delivery. This confusion was easier to clear during the walkovers where dips and slopes in the topography of the farm, which would transport water directly to a watercourse, could be linked to landcover types.

In many cases SCIMAP had indeed located the obvious routes where connected runoff should occur. Unfortunately this was further confused. The first reason for this appeared to be the nature of farming in the catchment. Many of the farmers had responded to this risk by redirecting water through culverts or via under drainage that subsequently transported water across the holding through sub-surface routes. In addition to the farmer's response to the local hydrology, careful management of grazing regimes occurred on many of the holdings. This allowed a good grass sward to develop at locations of high risk as directed by SCIMAP findings, thus reducing the likelihood of CSAs. The final confusion occurred due to the nature of the limestone geology. A number of the routes SCIMAP had identified where devoid of significant runoff due to being directed through limestone sink holes. Again these carried water across the holding via subsurface routing.

Despite the distortions between the model and the farm landscape, there where locations that appeared to have been accurately identified by SCIMAP. Moreover, at times when subsurface routes became blocked, and so runoff was redirected towards surface flow, the patterns did occur as SCIMAP suggested. At farms where stock management was less careful, severe poaching was visible at points in the landscape identified as risky. In addition, where the exact location of risk had not been realised due to land management techniques, there were often other locations of high risk in close proximity. This suggested that fine sediment would indeed be delivered to a watercourse via similar, or very close, routes to those identified in the model. Moreover, as was stated by F5 and confirmed by other farmers, there was a risk that fine sediment delivery was simply rerouted through under drainage of the meadows and pastures. F5 highlighted that eroded sediments would escape into the drainage network and thus be transported rapidly to a watercourse. Both these issues suggested that fine sediment would be delivered at, or close to, the locations suggested. However, this assertion is difficult to confirm. What was obvious was that as a working model SCIMAP had picked up the routes and landcover types that should conspire to connect fine sediment to stream networks. In addition it appeared that routes existed that would allow the risk to be delivered to the stream network in close proximity to those that the model highlighted as high risk.

Figure 6.2 displays the SCIMAP farm-scale maps first presented in chapter 5. Here the maps will be used to identify where the outputs did, or failed to, confirm the models accuracy. Locations 1 and 2 on map A highlight two locations where the model did not conform to the landscape. Location 1 was identified as high risk. Here the farm management had fenced out the river and a well wooded buffer zone was in place. Whilst the model identified this as high risk, this was not realised due to this management. However, it did appear that the outputs would have been correct under less sensitive management styles. Indeed, this farm had a history of poor management under previous ownership and during this period the stream was identified as in poor condition by the Environment Agency for many years (Frear, 1997). Location 2 displayed an area of low risk that had been exacerbated by an upwelling, either from a spring or damaged underdrains (the farmer was unsure). This resulted in enhanced risk not identified by the model, though SCIMAP is unable to account for such fine-scale management and issues due to the synoptic nature of the landcover map. In this case the model could be said to have directed investigations to some of the correct locations but management had negated the risk in one location whilst it had been heightened in another.

Locations 3 and 4 on map B are two interesting sites. Location 3 adjoins a small stream. Here the high risk has been realised due to stock access resulting in severe poaching of the bank sides. It is clear that at this location surface flow will deliver large amounts of fine sediment into a small first order stream. Location 4 displays a large area where the model has identified connected erosion sources. However, a sink hole (the farmer believes this is a natural feature of the limestone geology) at point 4 carries all the surface flow and this has sink has not failed to do so even in the most severe rainfall events.

Location 5 on map C shows an area where water has been redirected via under drainage so that it fails to follow the topographic controls. Interestingly this diversion has failed on several occasions and water has had to be redirected on more than one occasion. The water is now directed into a small second order stream and this appears to have increased its viability as a spawning stream. Location 6 is at an old lime kiln. Here the land slopes in a concave hollow towards the stream and the model appears to have predicted the risk correctly.

Location 7 on map D is where the farmer identified with the findings. However, he did suggest that a location in the adjoining field was at a greater risk. This suggests that the model is not picking up the finer nuances of the landscape, or that the farmer has perceived the situation incorrectly. The farmer at this location is well known for his environmental awareness and the interview highlighted his knowledge of fine-scale hydrological processes. He commented on two locations outside of the modelled area where simple management had resulted in severe impacts. The first was where a boulder was removed from the stream bed and subsequent vertical erosion of the bed resulted in the creation of a deep pool. The second was where the local authority redirected drain water from a bridleway into a first order stream with the unexpected consequences that the stream became over widened to the point where he has to drive upstream to get across on his quad bike. These anecdotes suggested that he had a good understanding of the landscape and how management can have large and unexpected consequences. Location 8 highlights where under drainage has failed and surface water now follows the routing suggested by SCIMAP to a large extent. This shows how SCIMAP would describe the landscape well in the absence of these rerouting methods.

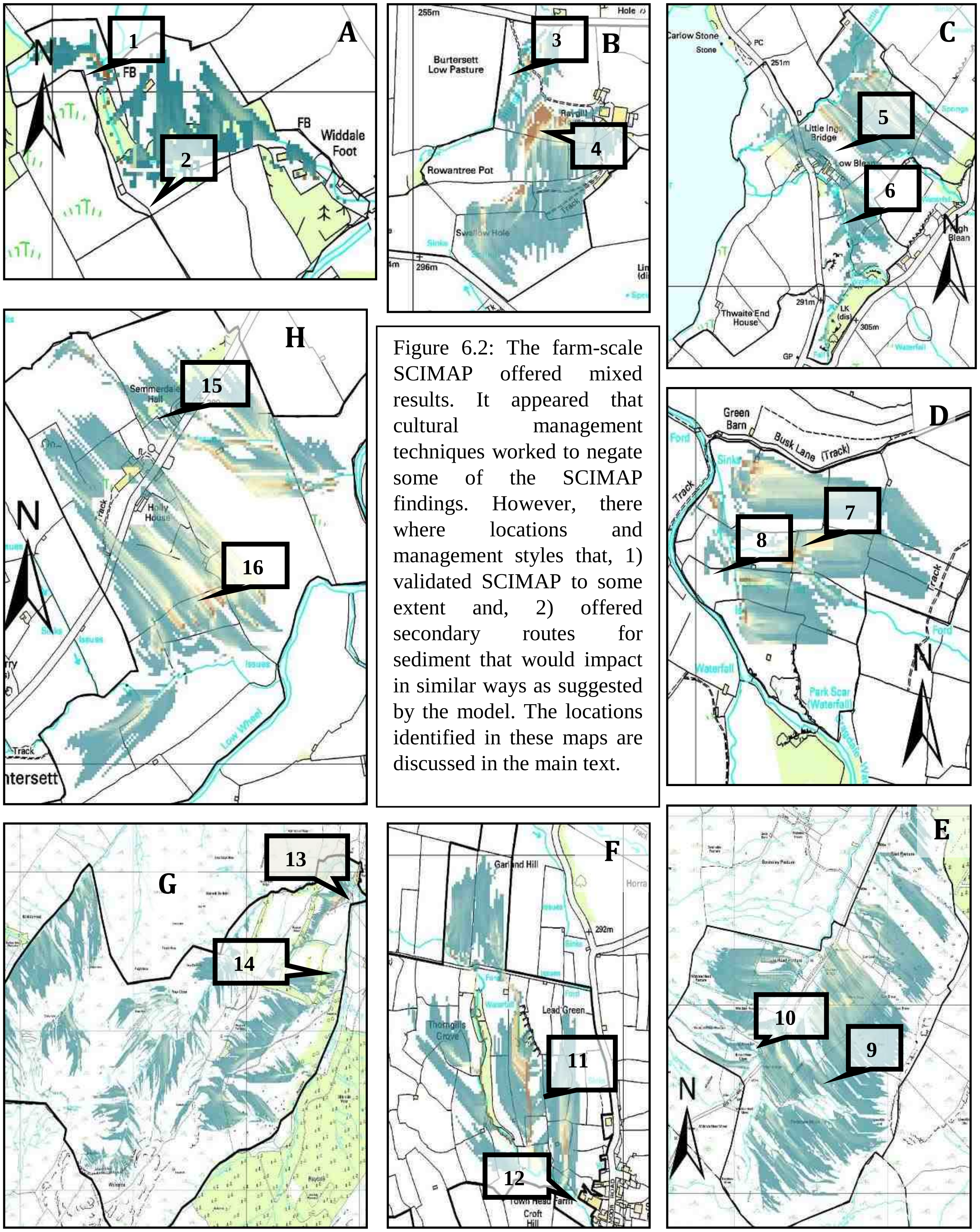


Figure 6.2: The farm-scale SCIMAP offered mixed results. It appeared that cultural management techniques worked to negate some of the SCIMAP findings. However, there where locations and management styles that, 1) validated SCIMAP to some extent and, 2) offered secondary routes for sediment that would impact in similar ways as suggested by the model. The locations identified in these maps are discussed in the main text.

The farm displayed on map E had little in the way of severe erosion and this is highlighted by the near absence of high-risk zones in the SCIMAP outputs. However, there were some issues. Location 9 is where under drainage carries flow in the traditional drains shown in figure 5.4. The farmer believes these act as conduits, or erosion pathways, for fine sediment simply rerouting the risk across the holding and emerging into the stream network at similar locations suggested by SCIMAP. Location10 is an area of high risk and the field at this location is utilised for the two farm pigs. The land here is poached, and is the only obvious erosion on the holding. The field is close to a stream and provides good evidence that land management can indeed be the difference between realised and non-realised risk.

Location 11 on map F shows a steeply sloping section of land running down towards a first-order stream. This is perceived as high risk in the model but this risk is only realised at a few locations along this route as most of the stream is well wooded with livestock excluded by fencing. Location 12 is an area where risk is clear but not revealed through SCIMAP. Again, this shows how management results in risk distribution. This location is used as a resting area after milking twice per day. The field and stream bank are severely poached and during rainfall events the stream becomes highly sediment laden. This situation occurs during relatively low rainfall periods showing that risk can be severely heightened through heavy livestock footfall.

Map G shows the largest holding visited. Most of the land is rough grazing or open moor. Because of this the in-bye land is perhaps more important than at other farms. Location 13 is on steeply sloping land directed towards a second order stream. During the visit there was a high level of sheep grazing within this field but despite this there was no obvious sign of erosion. The pathway for runoff was clear and well identified by SCIMAP; however, it appeared to be directed into an open drain (or heavily managed stream) which acted to transport surface flow. The farmer did not believe there was a high risk form this field. The field directly below appeared to be of greater concern in terms of connected sediment sources. Location 14 was on shallow gradient land but was used as a fairly intensive pasture. Here stock has direct access to the stream. The obvious points of erosion were around gateways but it appeared that erosion pathways would follow those revealed by SCIMAP.

Point 15 on map H shows a location north-west of the road that splits the holding. The farmer identified these locations as accurate. However, runoff is then directed into drains and the lower fields are all under drained and so a close fit with SCIMAP does not appear south-east of the road. The drains on this farm are of the more modern plastic type; the farmer is not in the Higher Level Scheme and so can replace the drains according to methods that fit with his land management. Due to this, the secondary routing for fine-sediments observed on other farms is unlikely to occur. The high-risk location present in the field identified at location 16 did not appear to be realised. The farm had a good grass sward and stock was managed by rotation and electric fencing. Despite this being one of the more intensive farms visited the risk did seem to be reduced by management techniques. In, addition the river downslope of these locations has recently been fenced out offering a good buffer zone against fine sediments.

From these interviews and walkover surveys with the landowners, it was not possible to show that SCIMAP's predictions always matched those seen in the field. This could be due to a number of reasons. First, it has to be considered that the model is not correct. However, work in other locations has suggested that the model does provide a good management tool fitting well with the on-ground situation (Reaney *et al*, 2010; Dugdale, 2007). The second possibility is that in limestone regions with subsurface flow, through potholes and underwater streams, surface flow does not occur in every location as expected; this results in a mismatch between reality (based on surface topography) and the model outputs. This seemed to be the case at the majority of the farms (though not in all fields), where culverts and sink holes carried water away from the locations SCIMAP identified. It should be noted that when the drains, sinks and culverts overflowed at Low Blean Farm during periods of high rainfall, surface flow did follow the routes SCIMAP displayed. In addition to the issues of culverts and sink holes, much of the in-bye land has been extensively under-drained. SCIMAP has already been shown to be less accurate in areas with chalk geology due to vertical flow directly into aquifers (Lane, 2008) possibly suggesting that it will be less accurate in locations with other types of limestone geology too. Yet locations with limestone geology do generate surface flow during rainfall events and it would be expected that the model would be able to predict the direction of flow on enough occasions to provide an overview of the surface pathways and thus fine sediment delivery.

F5 suggested that at Redshaw farm the drains themselves act as conduits for sediment transfers and that this was worsened by moles utilising the drains as proxy tunnels extending the network with side tunnels that act as soil macropores. This would suggest that, whilst SCIMAP is not able to pick up under-drainage, fields that it does highlight as risky would continue to be risky though the pathway would be via the drainage network and not follow the exact route SCIMAP suggests. If this is the case, then SCIMAP outputs at the in-channel level would still be generally accurate even though it is unable to capture the exact delivery route. The design of the drains is similar across much of the catchment and so, if this is the case, it would be expected to carry for all locations that have been under-drained. In many ways the subsurface flow through natural potholes and cave systems in the limestone geology could act in a similar manner by redirecting the route of sediment delivery.

The third possibility is that the model is not appropriate to be run at the farm scale and works with greater accuracy across larger spatial scales where, on average, there could be a greater degree of fit with the catchment surface hydrology and erosion. Anecdotal evidence from officers working on Catchment Sensitive Farming schemes has suggested that at the catchment scale, farmers in Nidderdale recognise the outputs as marrying reasonably well with reality.

The fourth possibility is the one Lane *et al* (2006) suggest may occur. This is that landowners are suspicious of the model and so attempt to mislead in order to refute the models claims. As noted above, this did not appear to be the case. The farmers provided quick and thoughtful responses and showed a high degree of awareness of their land and the hydrological processes that govern surface and subsurface flow. Moreover, they appeared willing to engage and showed an interest in the work and how it related to their land. Not one avoided visiting locations that the model highlighted and several extended the visit by looking at land they considered to be risky which SCIMAP had not captured.

The final possibility is the most likely. This relates to the ways in which land managers have already adapted their land use management practices in response to the kinds of risks that SCIMAP might identify, but without needing a model to tell them they need to do this. Farmers in the upper Ure catchment seem to have developed management methods to reduce surface flow, and so erosion, through active learning over generations. This form of knowledge may be a cultural phenomenon associated with the traditional characteristics of dales farming (Lane, 2010). It is a valuable source of information and should not be ignored but it is also difficult to capture in a model because it requires detailed knowledge of the management methods being adopted at each land holding. In the case of the Ure, it may be that these management methods are sophisticated because this is an agriculturally-marginal system, with shallow, nutrient poor soils. Farmers have thus evolved sound land conservation practices precisely because of the sensitivity and fragility of the resource. Again, this suggests that the model is perhaps more appropriate at the catchment scale than at the farm scale.

Whilst a definitive conclusion is hard to achieve from these visits, it is apparent that hydrological pathways have been extensively modified. This undoubtedly complicates the ability of SCIMAP to capture the exact pathway. The results emphasise what SCIMAP was originally designed for: a screening tool to prioritise where to look first in sensitive agricultural catchments (Lane *et al.*, 2006). When starting to look more closely at the upper Ure, many of the risks identified by SCIMAP were not being realised and this appears to be due to land managers evolving their land use practices to mitigate against these risks. Thus, SCIMAP needs to be described as a model that maps where risk could be and not where risk is. Such complications in validating the model at this scale do pose some awkward questions. However, there were enough locations that conformed to the model, and others that would perhaps provide the same in-stream result through altered pathways, that it was felt prudent to explore the model at the catchment-scale. The results of this exploration will be discussed next.

6.3 SCIMAP performance at the catchment scale

The SCIMAP fine sediment model was developed to offer a tool for river managers that would allow them to embed each river reach and land parcel into the context of the whole catchment. The notion was to provide guidance on where the most likely places to be delivering fine sediment to a river network would occur. This form of investigation begins the search for the more risky locations allowing a targetted and systematic search of a riverscape in terms of exploring fine sediment delivery. The model is not devised to give quantitative information but it does provide a qualitative framework that displays a risk probability. Thus, it cannot be expected to be accurate on all occasions. Nuances within the landscape, different farmers approach to land management and land management change since the CEH landcover map 2000 may result in a less accurate risk assignment (Lane *et al*, 2006). Issues with local-scale farm management could be clearly seen in the previous section highlighting how farm management can distort the findings. A model can only be as accurate as the information and assumptions inherent in the inputs allow. Therefore, to a large extent the raw data determines the results. If these inputs poorly reflect the catchment then the outputs will offer a similarly poor reflection of the land and riverscape.

For this work SCIMAP was utilised in four ways. First the farm-scale modelling allowed an exploration of the delivery index to ascertain its accuracy and whether farmer knowledge can offer insights into the model outputs. The results from this model run were inconclusive. However, it was felt that some degree of accuracy could be assigned to the model and some forms of management (e.g. under-drainage of in-bye land) may simply re-route risk across a land holding so that surface runoff still reaches the river network at similar locations described by the model. An interesting outcome from this stage of the model testing was that farmers' inherent knowledge of the land allowed them to engage in adaptive management practice at a number of locations. These forms of management included regular stock rotations, electric fencing to manage grazing and stocking rates. The effectiveness of these measures appeared to depend on the farmer more than the topography of the landscape. In addition to this, erosion around gateways and by rabbit burrowing appeared to offer the higher levels of erosion risk, though these were not always connected to a watercourse and cannot be accounted for in the model.

This section will discuss the three catchment-scale SCIMAP runs ($SCIMAP_U$, $SCIMAP_L$ and $SCIMAP_G$). The results from these catchment-scale runs will look specifically at the erosion risk in channel. This output describes the relative chance of a river reach delivering risk in terms of their upstream contributing area. Those assigned higher risk categories are deemed to be delivering disproportionate levels of risk relative to their upstream contributing area. Another way to conceptualise this is that the riskier locations are picking up fine sediment at a greater rate than they pick up dilution.

$SCIMAP_U$ was the first run at this scale. Here the land cover map was assigned no risk rating. The second run was weighted by land cover based on the CEH map with risk loadings ranging between 0 and 1. The final run was weighted by landcover in the same manner with the inclusion of the remotely mapped grip network added to the LCM and DEM. The addition of the grips to the LCM allowed this high-risk land management type to be assigned the highest risk rating (1) in recognition of the increased threat of erosion represented by these upland drainage channels. Prior to running the model, it was expected that the third run ($SCIMAP_G$) would offer the best fit with the on-ground situation as it most closely represented the landscape in terms of erosion and delivery to streams from different land cover types. However, the assignment of risk was based on expert opinion which can be fallible despite the level of knowledge of the practitioner and the intuitive nature of the risk loadings. Moreover, at the scale and resolution of the land cover map it was not possible to account for localised differences that could skew the model away from the reality of the catchment.

Any practitioner running the model is expected to have incomplete knowledge of the system. Hence it was considered important that the model should be tested against components of the in-stream ecology to ascertain if the outputs mapped onto known populations and whether it could offer any pertinent information on the success of these populations. This follows from the farm-scale modelling that explored hydrological connectivity and the ability of water to offer a delivery pathway from an erosion source to a recipient stream.

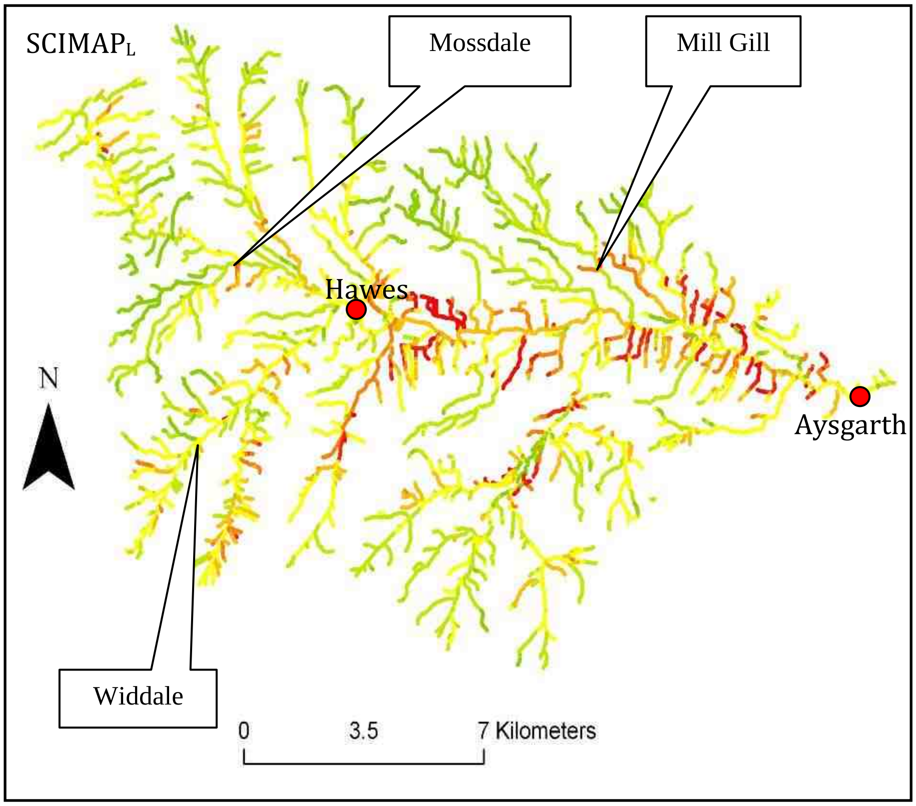
6.3.1 Catchment concerns

Running SCIMAP in a limestone-dominated catchment could be considered problematic. In such systems subterranean flow through sink holes and cave systems occur over wide geographical areas. This undoubtedly reduces the likelihood of surface and shallow sub-surface flow that are required for SCIMAP predictions to represent the hydrology of the catchment. The farm-scale model did indeed highlight how such sink holes and networks of under-drainage within meadows and pastures could provide a significant conduit for runoff. These concerns arising from the geology and land management could reduce the model’s accuracy in identifying the locations of highest risk and thus the worth of the model as a tool for river managers. However, the results at the farm scale did offer some potential for the model despite many farmers displaying a cultural response to the risk inherent in the landscape arising from the nature of farming in an upland catchment. However, fine sediment is generally delivered during heavy rainfall events. During such events surface flow is more likely to dominate as rainfall rates begin to exceed the capacity of sink holes and drains. The case study chapter identified that the number of T10 events were increasing within the upper Ure catchment and this change was dominated by an increase in winter T10 events. To compound this finding, winter is the period when soils are more likely to be eroded as the vegetation cover is poor and heavy machinery, used to spread slurry for example, churns up soils priming them for mobilisation. The farmer interviews and walkovers did highlight a number of areas where the capacity of sink holes, drains and culverts could be exceeded during high rainfall events. To compound this, many of these drains could become blocked by woody debris resulting in surface flow prior to the capacity of these pathways being breached.

6.3.2 Catchment-scale SCIMAP modelling

Despite these concerns, the three catchment-scale runs of the model highlighted how land cover was important in assigning risk across the catchment. When unweighted by landcover, the dominant controls on erosion and fine sediment delivery risk arise from rainfall coupled with the DEM. In this version of the model the risk of fine sediment delivery is more evenly spread across the catchment. In contrast, $SCIMAP_L$ contracts the risk towards the floodplain where the higher intensity land use is situated, whilst $SCIMAP_G$ draws some of the risk towards the upper zones of the catchment where the higher density grip networks occur. However, these differences are subtle and the maps have to be scrutinised in order to identify the locations where risk has been relocated between the model runs. By mapping the risk categories onto the survey locations the risk was easier to visualise. For example at the Mossdale electrofishing survey site (figure 6.3) the risk categories were 6, 5 and 6, at the survey location at the upper section of Widdale the risks were 8, 4 and 5 and at Mill Gill (Askrigg) they were 8, 4 and 6 ($SCIMAP_U$, $SCIMAP_L$ and $SCIMAP_G$ respectively). This highlights how land cover can spread risk across a catchment and how the model outputs change dependent on the assumptions made at the early stages. $SCIMAP_G$ also offered the possibility of coupling remotely sensed data with the model allowing high risk land uses to be captured in the raw data despite not being present in the original format.

Figure 6.3: *Locations were differences in the SCIMAP risk categories between SCIMAP$_U$, SCIMAP$_L$ and SCIMAP$_G$ are clear.*



It was interesting to note the significant correlations from each of the model runs. SCIMAP$_U$ displayed only four significant relationships, none of these at P = <0.01. SCIMAP$_L$ and SCIMAP$_G$ both showed 6 significant relationships both having three at P = <0.01. Interestingly only SCIMAP$_G$ displayed a correlation with brown trout fry populations. This was at the P = <0.05 level (table ??, ch.6). Whilst this run of the model was expected to have the best fit with catchment processes, it was surprising to find that the relationship was positive. The risk category of SCIMAP$_G$ that attached to the electrofishing sites ranged from 4 to 9. This suggests that relatively high levels of hydrological connectivity are required in order for successful brown trout recruitment. The finding is not related to gripping, more that the SCIMAP$_G$ model emphasises locations conducive to recruitment in the headwaters. The positive correlation suggests that gripping has not been a significant factor in limiting recruitment, despite fear over erosion and fine-sediment delivery.

One possible reason for this is that whilst the risk is being delivered via these routings, the impact of fine-sediment delivery is unlikely to be realised until further down the system. In addition, brown trout respond to a flow trigger (generally the receding limb of a spate event) and so reasonably strong hydrological connectivity is more likely to provide the flows required to trigger spawning migratory behaviour. Clearly the nature of the grip network poses an enhanced risk of sediment delivery to the stream network. This is apparent through simple observation of the erosion that has occurred on peat soils since the grip networks were ploughed. Yet, whilst the risk is delivered through these networks into low-order streams, it is perhaps likely that the risk will not be realised till much further down the system, beyond the case study catchment. The nature of these low-order streams displays highly turbulent flow on steeply sloping land. Sediments are most likely to settle out where the channel gradient is low and deeper pools become common in the system. This may result in the ecological impact of grips being most readily felt some distance from the grip networks creating a disjunction between cause and effect. This poses a research challenge in terms of survey and experimental design.

As $SCIMAP_G$ was the only run of the model to offer a correlation with brown trout fry, it can be assumed that this version does indeed offer the best representation of fine sediment delivery pathways within the catchment at least in terms of stream biota. However, this delivery matrix of fine sediment does not appear to display a limiting control on recruitment as would at first be expected. Whilst other results do not seem to corroborate with this assertion, this may indicate different scales of control. Brown trout fry populations showed a strong positive correlation (<0.01) with the presence of gravel as expected but also with sand and silt. Additionally, there is a negative correlation (<0.05) with the proportion of boulders and cobbles as bedload. For a species that has a close relationship with substrate type and composition, strong correlations with bedload are of no surprise. Poff (1997) suggests that the presence of refugia, such as boulders, offers a structure to the habitat that enhances survival. Indeed a number of EA fisheries scientists have suggested that boulders within the bedload offers increased territories for fry to exploit (Frear 2007; Lee 2008). Yet there appears to be a threshold level which likely reflects flow conditions and gradient of the stream. Whilst recruitment occurs in low-order streams, gravel is required in higher proportions and so there seems to be a cut off point to the flow rates above which recruitment becomes less likely. These are the locations where boulders and cobbles begin to dominate in the upper reaches of low order streams. Thus, whilst the presence of grips does not seem to be limiting recruitment, the $SCIMAP_G$ configuration does seem to emphasise locations where recruitment is strong; on top of this, other correlations emphasise in-channel controls especially in terms of substrate. One implication is that further work is needed on the SCIMAP model to ascertain exactly which configurations best accord with runoff production, and why.

An additional consideration is whether the grips remain actively eroding. These artificial channels may now have reached equilibrium with the prevailing climatic conditions and thus sediment delivery, which clearly increased post-cutting, may now have settled so that delivery is within the assimilatory capacity of the in-stream ecology in this catchment. Finally, the ecology must have responded to this alteration in sediment delivery (most probably downstream of many of the survey sites) and thus be suppressed to the point where the baselines we now recognise are shadows of past communities or wholly different in composition. This shifting baseline between generations may occur due to a paucity of historical data. Where the data are available, there is evidence that gripping severely depleted salmonid populations through fine sediment delivery (Theurer *et al*, 1998; Stewart and Lance, 1983; Stewart, 1963).

From this exploration of SCIMAP it appears that it does offers a reasonable level of accuracy when the riskiest land uses are represented in the input data ($SCIMAP_G$). This is despite the inherent concerns of running the model in a limestone-dominated system. However, the results from the stepwise regression analysis still need to be explored. Prior to this, the results from the field surveys, electrofishing and macroinvertebrate investigations will be discussed in the next section.

6.4 Catchment and river characteristics

The correlation analysis provided some basic detail that enabled the catchment to be characterised as a typical dales river system. For example, there was a negative correlation between stream order and obstructions (mostly waterfalls in this catchment) (<500m $P = 0.007$; <1km $P = 0.042$) highlighting that, as stream order rises, there is less likelihood of an obstruction of this sort. Also, for higher stream order, there is reduced likelihood of the stream area drying upstream of the survey locations ($P = 0.001$) whilst the channel width increases ($P = 0.000$). Positive relationships exist between upstream contributing area ($P = 0.000$) and upstream area of moorland ($P = 0.000$). Increasing channel width shows a strong positive relationships with Strahler stream order, upstream contributing area and upstream area of moorland (all at $P = 0.000$) and as the river widens there is a reduced propensity for drying close to the survey locations ($P = 0.000$). Stream width also correlated positively with the presence of buffer strips suggesting that the agricultural community is more likely to fence the stream and so excluding livestock as the river widens ($P = 0.004$), though this is likely due to livestock farming being more prevalent on the lower elevation floodplains.

The proportion of sand and silt as substrate type increases with the number of pools present within survey sections ($P = 0.001$) whilst boulders and cobbles decline ($P = 0.027$). Algae increases where siltation occurs ($P = 0.018$), where earthcliffs are located on the river bank ($P = 0.018$), due to poaching of soils by livestock ($P = 0.006$) and due to the presence of sand and silt ($p = 0.003$). At the same time, emergent macrophytes increase in tandem with algae ($P = 0.008$) though this may be a function of substrate needs rather than nutrient delivery increasing with fine sediment proportion. Additionally, algae increases positively in correlation with upstream contributing area ($P = 0.034$) and shows positive correlations with both $SCIMAP_G$ and $SCIMAP_U$ ($P = 0.048$ and 0.016 respectively). The stronger relationship with $SCIMAP_U$ and the relationship with upstream area suggest that hydrological connectivity is driving an increase in fine sediment as the stream order increases, although the presence of earthcliffs also correlated positively with in-stream sand and silt proportions ($P = 0.001$) and negatively with boulders and cobbles ($P = 0.007$). In contrast to the above assertion, this suggests that localised land use may also be a strong driver of fine substrate composition within the bedload. Perhaps scale is important here: whilst some locations have strong drivers arising from local land use, others require catchment-scale processes to drive sediment composition. Moreover, the delivery of fine sediment from surrounding land use requires strong hydrological connectivity. $SCIMAP_U$ is in effect an index of surface flow and this may explain the relationship with in-stream fine sediment better.

None of the substrate types correlated with the macroinvertebrate indices whilst siltation did. This suggests that qualitative methods reliant on the judgement of the surveyor may not be very precise. There is always the risk that assessments based on judgement provide results that have wide deviations at, and between, survey sites. This risk is extended when different surveyors are offering information across a catchment as is often the case with government agencies. The macroinvertebrate results will be discussed further below. Boulders and cobbles correlated with the propensity of the stream to dry at the survey locations ($P = 0.037$) although this again is a qualitative measure and was reliant on external sources (EA and YDNPA staff). Whilst there is no reason to question the information given, there was no further detail provided. For example, did the stream dry up regularly or very rarely? None of the streams at the survey locations have dried during the previous five years.

Very strong correlations existed between substrate types. Gravel was negatively correlated with bedrock ($P = 0.001$) as well as boulders and cobbles ($P = 0.000$). Boulders and cobbles displayed negative correlations with siltation ($P = 0.002$) along with sand and silt ($P = 0.000$). Gravel showed strong positive relationships with both $SCIMAP_L$ and $SCIMAP_G$ ($P = 0.000$ and 0.001) respectively suggesting that fine sediment delivery suppresses the proportion of gravel within the channel.

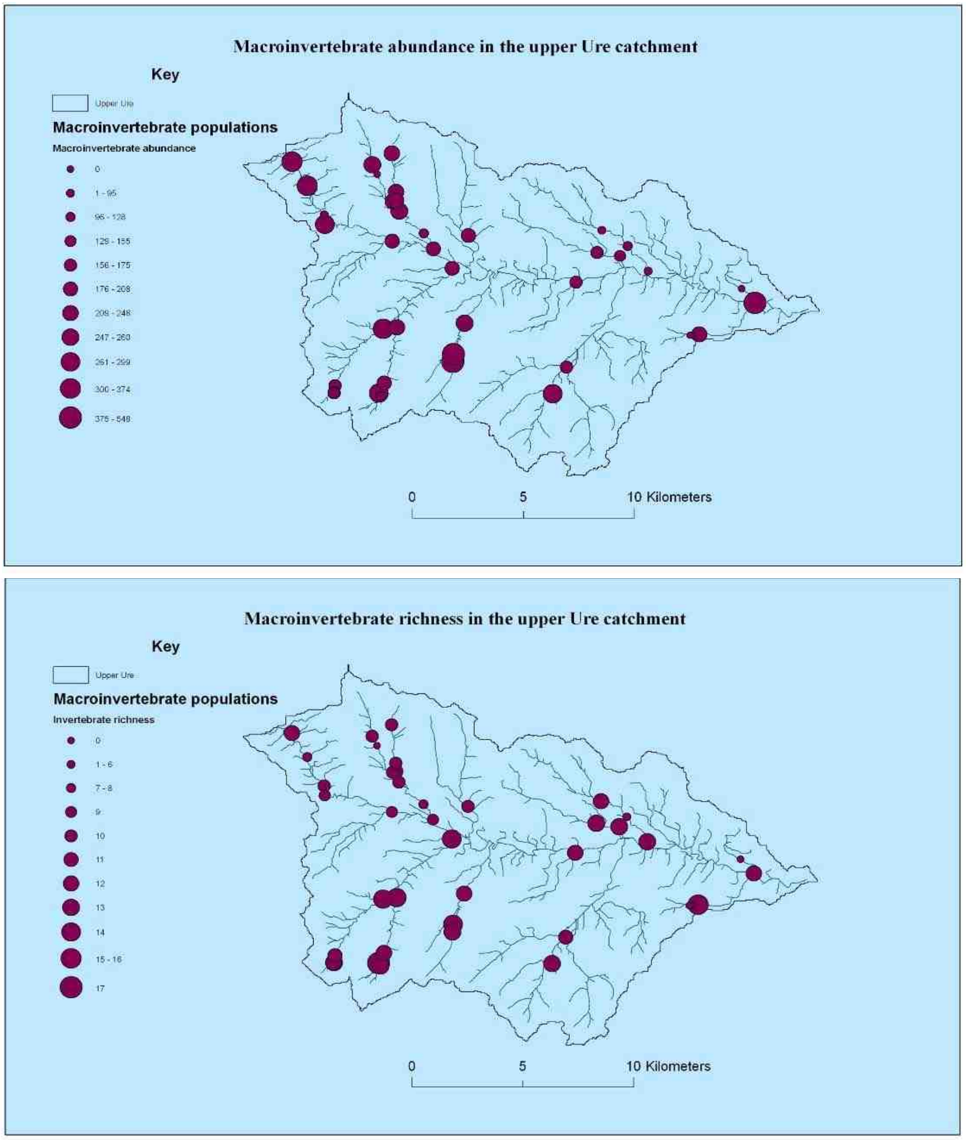
The propensity of the steam to drying at or around the survey locations showed very few correlations. Furthermore, there can be little confidence in this measure due to its very low return in the correlation analysis and the nature of how the information was gathered. Thus, it will not be considered further despite that it can have severe negative impacts on brown trout populations as habitat contracts during times of drying (Lake, 2003; Mathews and Marsh-Mathews, 2003; Bell et al, 2000). Another measure that showed very few correlations was the presence of barriers. In this catchment, barriers were largely in the form of natural waterfall features; however, there were a number of unnatural barriers including weirs, culverts and fords. Obstructions within 500m upstream correlated positively with undercuts of the bank ($P = 0.001$) most likely due to the increasing flow rate and turbulence which acts to increase undercutting, but this may just be dependent on the general location within the basin.

Land use within the catchment displayed very few significant correlations but did provide some interesting relationships. For example, the presence of buffer strips effectively reduces stock access (P = 0.000) and increases shading of the stream (P = 0.000) whilst the percent shade along the riparian zone decreases where stock have free access to the stream (P = 0.000). Buffer strips are more likely to occur where land use is extensive (P =0.001) and with increasing upstream area and area of upstream moorland (P = 0.029 and 0.002 respectively). The percentage of shading of the river from the riparian zone also appears to increase as the $SCIMAP_G$ risk category increases (P = 0.004). All these correlations probably indicate general position within the basin and indicate the most likely locations for buffer zone location. It is interesting that buffer strips do not correlate with intensive land use; this perhaps deserves greater attention.

6.5 Macroinvertebrate communities

The results showed a large range in macroinvertebrate abundance (60 to 548) and richness (8 to 17 families recorded). There were no significant relationships with abundance against the other variables, apart from the obvious relationships with the other macroinvertebrate scores. Macroinvertebrate richness did display some significant relationships; notably both $SCIMAP_L$ and $SCIMAP_G$ showed significant negative relationships (P = 0.037 and P = 0.033 respectively) suggesting that land cover has a negative impact on aquatic invertebrates by reducing diversity at the family level if not overall abundance, although this reduced richness did not appear to impact brown trout fry populations (figure 6.4). Propensity for the stream to dry downstream of the survey site showed a positive significant relationship with richness (P = 0.039), although, since very few streams were known to dry up, that this is likely to have very little meaning at the catchment scale. Siltation was the final positive relationship (P = 0.034), but as the proportion of fine sediments was never above 35%, this may well add to the habitat structure and thus allow higher diversity over the survey length. The following will describe the indices calculated from these two scores.

Figure 6.4: *Macroinvertebrate abundance (top map) and richness (bottom) at each of the survey locations.*



6.5.1 LIFE scores

LIFE scores were calculated as the macroinvertebrate community response to the fifth percentile flow (Extence *et al*, 1999). On a spate river it is to be expected that the extreme flows will sort the ecological community, excluding those species that are unable either cope with such flows, or fail to find refugia as the extreme peak flows pass through the system. The scores themselves offer no value judgement on nutrient status, or other chemical qualities, but assist with an understanding of the dominate flow type in the absence of gauging stations. Thus, LIFE scores offer some context to the river processes and how ecological communities respond to the high, rapid flows that frequent this catchment. This explains the Ure macroinvertebrate communities being composed largely of mayfly (ephemeroptera) families including heptageniidae (commonly termed rockclingers), baetidae, caddisfly (trichoptera) and stonefly (plecoptera) that can cope with rapid flow types but also require turbulent waters to ensure dissolved oxygen remains high. These spate flows pass risk on to brown trout eggs, alevins and fry which may respond by being washed out from the gravel redds if the spike arrives rapidly or reducing time for fry to locate adequate refugia. A rapid rising flood limb reduces response time for these members of the ecosystem. In addition, if the habitat structure offers little in the form of refugia, as is often the case where riparian buffer zones have been lost, then wash-out of brown trout at early life stages is likely to become increasingly common. Thus, in a river system whose ecological community is dominated by spate flow-adapted species, those unable to adapt will become diminished.

The upper Ure has only one flow gauge within the catchment at Snaizeholme. The data from this gauge have shown an increase in the winter fifth percentile winter flows with a decrease in summer and autumn fifth percentile flows. This could impact the in-stream communities in a number of ways. During the winter the macroinvertebrate and brown trout egg stages are the most prevalent. These are at risk from numerous processes that act to disturb gravel substrates. An increase in winter flows and corresponding decreases in summer and autumn could result in a decrease of brown trout eggs and alevins which emerge in late winter. In addition, an increase in the predominance of low flow adapted macroinvertebrates could occur. This alteration in the macroinvertebrate community may be coupled with decreasing populations as the egg and early life stages become increasingly prone to drift response due to winter spate events. A reduction in the autumn flows could reduce the spawning migration signal needed to the move to natal streams. Another possible outcome of these changing flow patterns would be a greater propensity for algal growth in summer as a response to a reduction in the fifth percentile flows. High summer flows help flush the system of nutrients, fine sediments, algal growth, recharge the water with dissolved oxygen, reduce water temperature and wash finer sediment fractions from gravel beds.

LIFE scores showed a positive correlation with the presence of boulders and cobbles (P = 0.05). This confirms the relationship with high flow streams that are dominated by larger bedload fractions. They were also strong negative correlations with $SCIMAP_L$ and $SCIMAP_G$ suggesting that land use impacts in the form of fine sediment delivery may have some impact. However, it is worth mentioning that fine sediment risk is more likely to be realised in the lower reaches of the system where there is a greater opportunity of it settling out although if fine sediment is infiltrating the gravel matrix then it can be expected to impact on those species that require the interstitial spaces. Brown trout do scrape out the gravel matrix during spawning and this will mobilise fine sediments restoring the pore spaces which in turn allows intergravel flows to increase. These are required by the egg and alevin stage to ensure that oxygen is replenished and excreta are removed from the vicinity (Armstrong *et al*, 2003). Finally, there was a strong positive relationship with the Shannons diversity index; this will be discussed in the following section.

6.5.2 The Simpsons and Shannons diversity indices

Simpson’s index gives the probability of any two individuals from the same sample drawn at random from a community belonging to the same species, or taxonomic level of interest, (Stilling 1992). The Simpson’s index is biased towards dominance within the community whereas the Shannon’s index is biased towards richness and evenness of the sample (Stilling 1992). The Simpson’s diversity index has been expressed as 1/D showing diversity increasing as the value increases and so the lower values show a higher degree of homogeneity, or dominance, in the sample. This explains the strong positive correlation (P = 0.001) between these two indices. Both show increasing diversity as the value increases, thus dominance within the community is displayed as low values in the Simpsons diversity index. The Shannons index had two other correlations, firstly with LIFE scores suggesting evenness of the community is attained in rapid spate situations where flow transports nutrients rapidly downstream where nutrient-tolerant species may become dominant. Correlations with the Simpsons diversity index support this assertion. As Strahler stream order increases, this index decreases (P = 0.04) showing that in higher-order stream there in a propensity for the community to be less evenly spread as dominance becomes more common. Increasing upstream area of moorland also diminishes evenness (P = 0.044). An increasing number of pools in the fifty-metre survey section also display a positive correlation with the Shannons index showing that greater habitat diversity results in greater evenness amongst macroinvertebrate communities. Upstream obstructions provided a similar correlation (P = 0.017) but this is perhaps more an artifact of location then a controlling effect on the evenness of the community. The final correlation with the Simpsons diversity index was the least expected. Siltation appears to increase evenness (P = 0.048) although the proportion of these small fractions was never greater than 35% and generally much lower , suggesting that fine sediments increased the spread of species by adding microhabitat diversity.

There were no significant relationships between any of these indices and brown trout fry populations. This suggests that food supply is not a limiting factor on population and thus at some locations recruitment is being limited below the carrying capacity of the environment by other factors. This is highlighted by some of the results. Paddock Beck (SD9460090500) was ranked 35 out of 40 in terms of the average fry density over two years yet this location returned the third lowest macroinvertebrate abundance score (75). In contrast, a site in Sleddale (SD8566086660) had the second highest macroinvertebrate score (470) but both years of electrofishing provided a zero return for brown trout fry. The failure of a pattern to emerge between prey and trout fry certainly points to other limiting controls on the population. These will be discussed later.

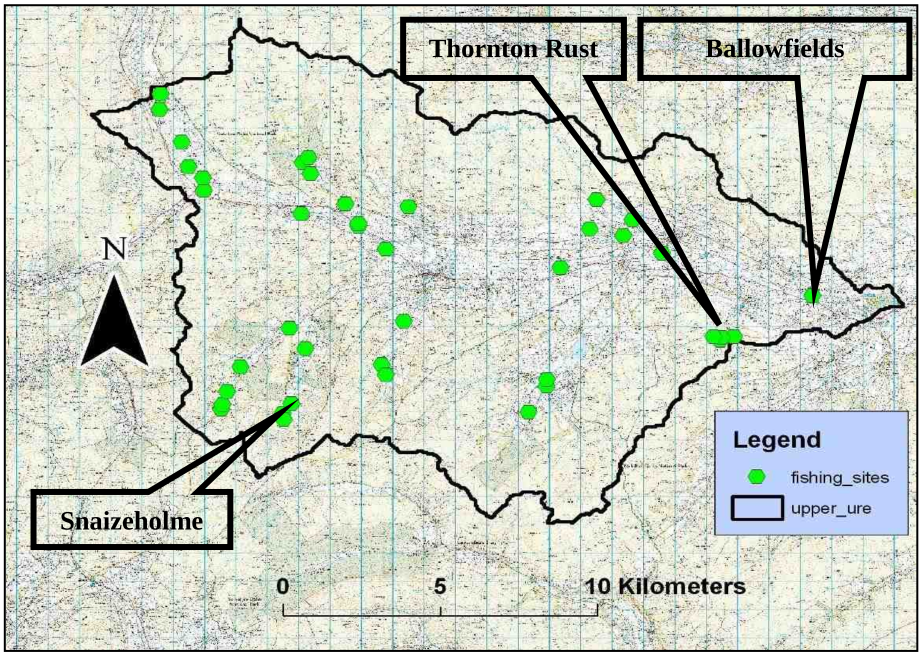
6.6 Spatial distribution of brown trout

Salmomid species are notoriously difficult when it comes to understanding their populations. This is due to annual and cyclical fluctuations in recruitment (Cowx, 2010). Therefore, sampling will always return a number of low and even zero returns. However, the number of sites that offered a zero return in the case study catchment was unexpected. Sampling had been specifically stratified to incorporate the most appropriate brown trout spawning habitat at all electrofishing sites. Moreover, the range in fry populations was especially large (0 to 0.268 fish m^{-2}) with two sites (located at the downstream end of the catchment) offering good returns and the majority of others showing low numbers or no recorded population.

The difficulties with electrofishing (e.g. sampling over a full season, different flow conditions, reliance on a number of operatives) are unlikely to account for this. Whilst reliance on volunteers, many of whom had limited experience of the survey method, was necessary, each survey had experienced operatives present. Lost fish were recorded to offer a marker of efficiency and triple-pass surveys were undertaken to measure the semi-quantitative method against a depletion method. The results from the single-pass surveys stood up well to this scrutiny. The climatic conditions of the survey seasons, and the preceding spawning season, were notable for their high rainfall and this may explain some of the poor returns. However, the sites where good returns were recorded were subject to similar climatic conditions. Whilst the upper dale is more likely to have received greater rainfall, and thus high flow rates, it seems unlikely that this alone could explain the range in recruitment success.

The few sites with relatively good recruitment did have some specific and similar qualities. All had small upstream areas with a high proportion of moorland in comparison to meadow or pasture. Shading was present, in the form of trees, rush/grass riparian cover on a small stream or due to the nature of the valley (steep sided v-shape form); however, none of the streams were wholly shaded. A significant natural barrier to migration was located upstream of all these survey sites but perhaps most importantly livestock were either excluded or the stocking rate sufficiently low to be of little importance, certainly in comparison to locations with intensive dairy operations. The substrate was dominated by gravel with little in the way of finer fractions and the stream banks were intact with no signs of an earthcliff. None of the streams had been widened through poaching and a good flow was maintained in a well defined stream bed. Whilst none of these properties are unique in the catchment, they did provide unusual less common setting in comparison to the majority of other sites (figure 6.5).

Figure 6.5: *Locations within the catchment where trout fry populations were good.*



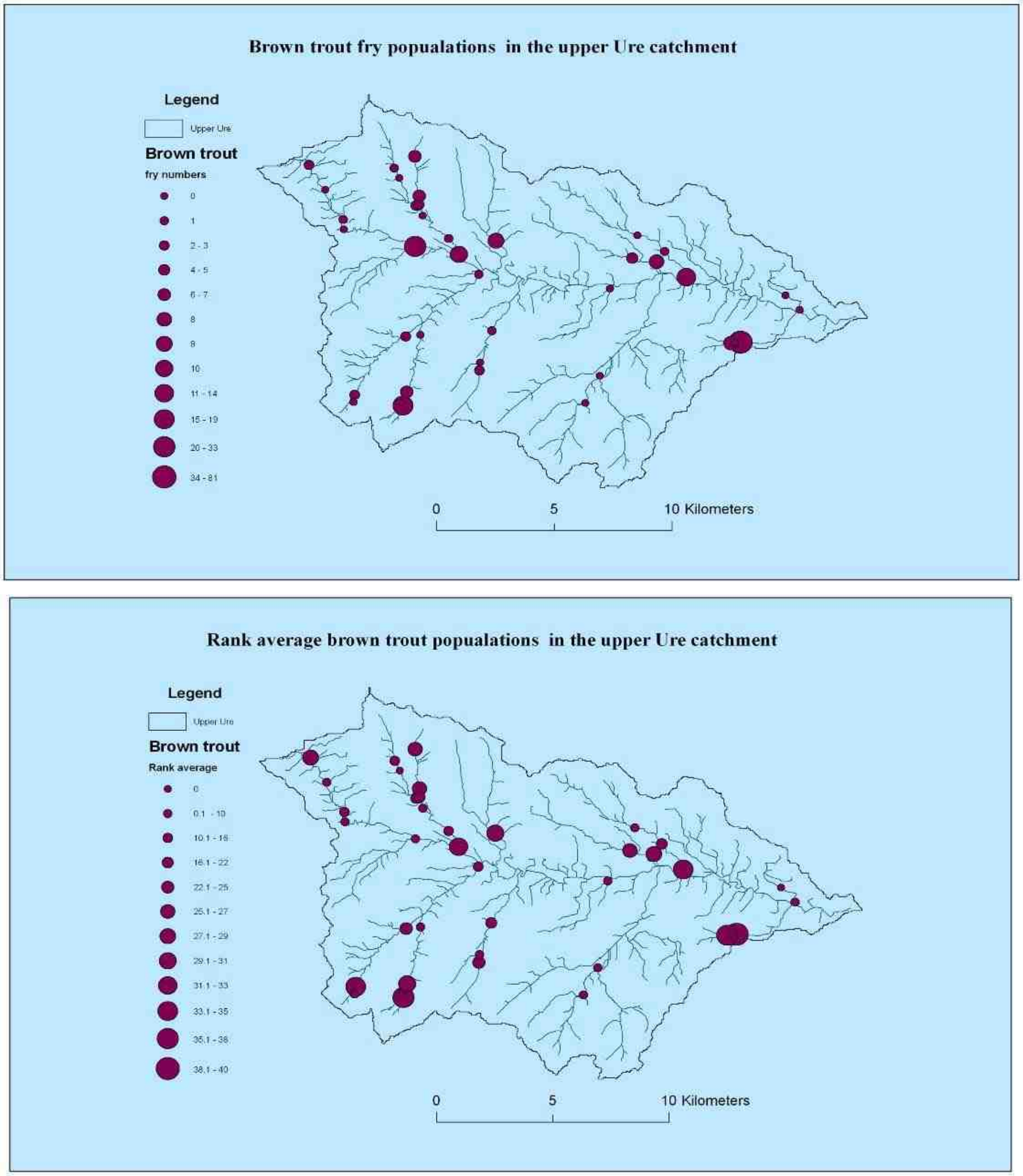
In contrast to these "good" sites, the majority provided a mix of zero returns or very low densities. Figure 3.30 in Chapter 3 highlights that this situation of poor recruitment is a pernicious problem in the upper Ure catchment and a number of reasons have been offered for this ranging from diffuse sediment and phosphate pollution along with physical and morphological concern (EA, 2011). These findings are confirmed by anecdotal evidence provided by anglers within the catchment. In some locations stocks of trout and grayling are so low that according to Waldman (2010), 'last season...involved so many blanks that the catching of a single fish became a notable event.' Angling has long been supplemented by stocking within the main river stem, yet despite this, stocks are still considered to be low. The addition of stocked brown trout may add to the breeding pool but this does not appear to have translated into increased recruitment within the upper Ure catchment. Angling clubs downstream of the case study have argued that stocked fish provide the main pool for recruitment (Anderson, 2011). This debate has come to the fore recently with an impending EA ban on stocking with diploid fish whilst triploid[29] stocking will remain permissible. Anderson (2010)

[29] Diploid fish are fertile with the ability to develop into fully breeding adults whilst triploid fish are sterile having been subjected to high pressure shock during the egg stage.

argued that when one club moved to triploid stocking, recruitment tailed off rapidly. In terms of this work, it would not matter whether recruitment was through stocked or native trout as recruitment success is the major interest. It would not matter whether it was recruitment from native or stocked fish so long as detail could be gleaned about the nature of the habitat and thus the controls on fry populations.

Whilst poor recruitment has created concerns during this work, it has highlighted that fish stocks in the system are poor and replacement is highly limited throughout the catchment. The lack of long-term data set at the most appropriate recruitment locations (low-order streams, riffle habitats) also created difficulties. The rapid gathering of spatially distributed brown trout fry data was required to counter this issue. The results offered an insight into an upland brown trout fishery (figure 6.6). Downes (2010) argues that investigations of stream biota should be targetted to where populations are known to exist and locations with no members of the target species should be ignored. She contends that such sites may be avoided by the species for long-standing natural habitat reasons and not due to human influenced controls. Taking this approach within the case study catchment would have been difficult due to the lack of a long-term data set on where recruitment does or does not occur. Moreover, in a human-dominated system, it can be expected that all locations will have a degree of human interference and thus a zero return has an increased chance of arising from human interference and impacts. Furthermore, the gathering of physical, habitat and ecological data across a number of scales allows human and non-human derived controls to be assessed. What remains is to decipher the results in terms of why the recruitment pattern is so poor throughout and importantly, what can be done to mitigate against human-dominated controls that appear to be derived from land use impacts. The next section will begin this exploration of the results.

Figure 6.6: *The brown trout fry populations in the upper Ure catchment (top map) and the rank average populations (bottom map) based on density of population. As can be seen due tot he area of the stream the highest populations are not always the most dense.*



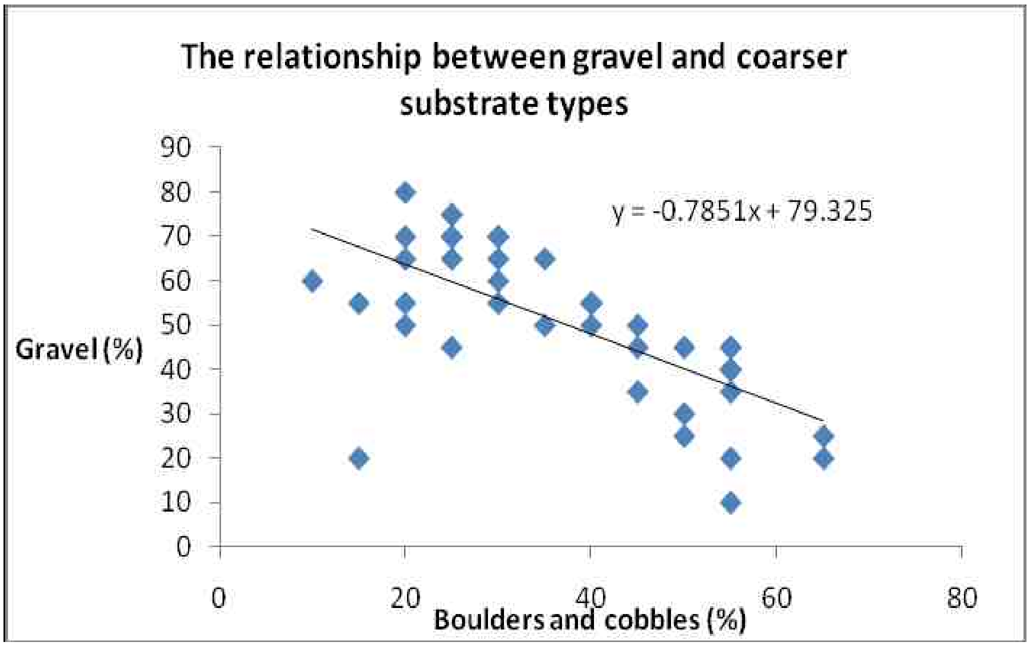
6.7 Exploring the causal factors of brown trout fry distribution

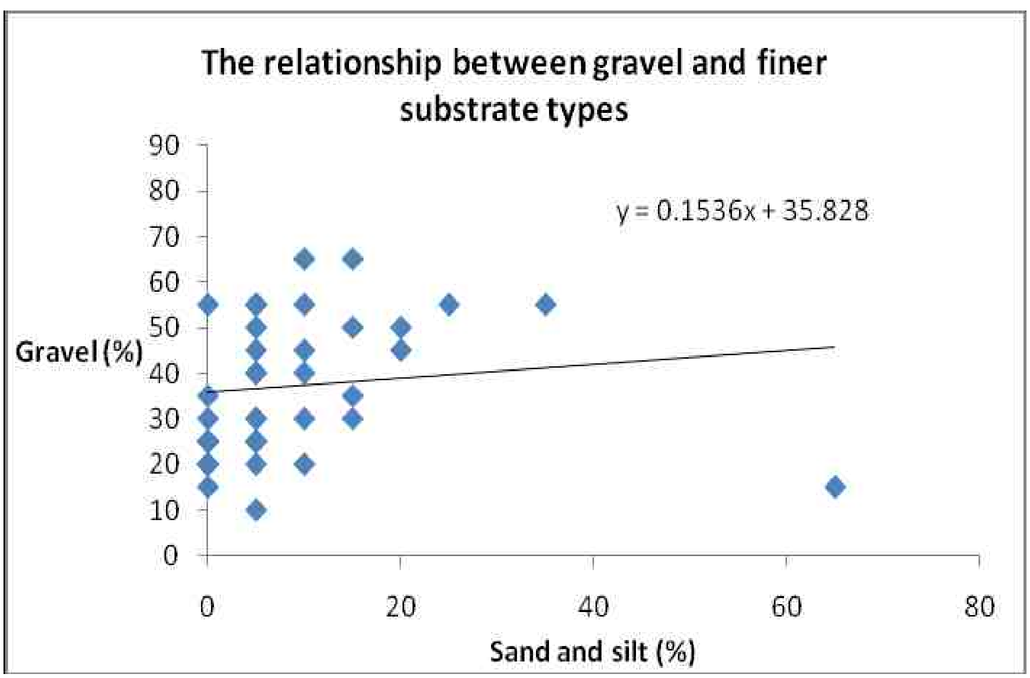
The aim of this research was to combine advances in remote sensing, Geographical Information Systems (GIS), catchment-scale modelling and ecological survey techniques with current awareness of salmonid species, specifically brown trout fry populations, to develop an effective approach to the ecological restoration of habitat through the prioritisation of location and management options. The methods employed reflected this and they enabled difficult to capture variables such as land uses (e.g. grips) and catchment dynamics (e.g. upstream area of moorland) to be included. This allowed the statistical testing of a range of processes and factors that may be placing limiting controls on brown trout fry populations. This section will explore the results presented in Chapter Five in terms of how the variables impact on brown trout recruitment to explore which factors are indeed important controls. The final section will then provide an overview of a catchment-scale restoration plan and summarise the work.

The regression analysis offered information on the multiple strands of evidence gathered in terms of how they may affect river biota. The initial run of this statistical method explored the relationship between brown trout fry populations (rank average over two years) against the variables that displayed a significant correlation with these populations. The results showed that the presence of algae within the stream channel displayed a negative relationship with fry populations (r^2 adj. 8.07, P = 0.008) whilst increasing proportions of sand and silt as a substrate type displayed a positive relationship (r^2 adj. 7.27, P = 0.006). This gave a first indication of what may be placing controls fry populations. However, these relationships have to be qualified. First sand and silt proportions were always below forty percent so, despite the presence of these fractions, the survey sites displayed enough gravel to allow brown trout recruitment, and paradoxically positive correlations involving the sand and silt fraction are simply a stronger indication of suitable substrate than gravel in the correction analysis. Gravel does appear in some stepwise regression models. Strong recruitment therefore seems to be found in medium-sized sediments (gravels with some sand and silt), neither too coarse (upland, extreme flow conditions) nor too fine (fine sediments unsuitable for spawning), see figure 6.8, overleaf.

The presence of algae within the channel may be a response to a number of criteria. First nutrient loadings (phosphate is generally accepted to be the limiting nutrient in freshwater systems) in upland oligotrophic streams have to be supplemented by external sources (e.g. diffuse agricultural sources), the stream has to be subject to solar energy and so heavy shading will inhibit photosynthesis. Thus, two possible methods exist for countering algal growth, first suppress the source and second suppress light infiltration. Both of these would work at specific locations; however, the latter would still allow the nutrients to be transported downstream and impact lower sections of the river network.

Figure 6.8: *There is a clear relationship between substrate types. Gravel is less dominant where coarser fractions exist and becomes more abundant where sand and silt become more common.*





To understand the factors acting beneath these relationships, stepwise regression was run on algae and sand and silt incorporating just the factors that they displayed significant correlations with. The result showed that algae displayed a negative relationship with $SCIMAP_G$ (R^2 adj. 10.27, P = 0.030) and positive relationships with soil poaching by livestock (r^2 adj. 8.91, P = 0.003) and the presence of emergent macrophytes (r^2 adj. 5.98, P = 0.004). It appears that in low-order streams the channel is a conduit for risk as described by $SCIMAP_G$, which is most likely realised further downstream. Thus $SCIMAP_G$ at the scale of low-order streams does not display a negative impact. However, further research that extends the exploration down to the main floodplain of the river Ure, even perhaps downstream of the case study catchment, may elucidate where the risk is realised. A second explanation is that $SCIMAP_G$ offers a description of hydrological connectivity and is thus suggestive of rapid responses in flow during rainfall events. Thus, at locations where the $SCIMAP_G$ index is high, flow rates may well be inhibiting algal growth.

Poaching of bank-side soils is a well understood impact on upland streams where fine sediments can be rapidly delivered to rivers. Moreover, cattle accessing bankside habitats often enter the channel and can add direct nutrient sources through faecal matter. This has been witnessed in the case study catchment on numerous occasions showing that the river is viewed as a source of water for stock and perhaps offers a secondary function of providing a place for cattle to cool during the summer months. Regular access to the stream in this way exacerbates soil poaching and encourages the mobilization of poached soils to the stream network (figure 6.9). This form of stock management is a clear risk and encourages algal growth within the channel.

The presence of algae within the channel may have a number of impacts on water and habitat quality (figure 6.10. Thick algal growth can smother the surface of the stream bed creating night sag of dissolved oxygen as photosynthesis stops and only respiration continues. When the algae die back, the decaying matter can result in a high BOD which further reduces oxygen levels and smothers gravel beds. Spate events can reset the system by washing out the algae and replenishing dissolved oxygen; unfortunately the long-term trend of fifth percentile summer flows shows a reduction and so there is an increased likelihood that algae will remain established for longer periods of time.

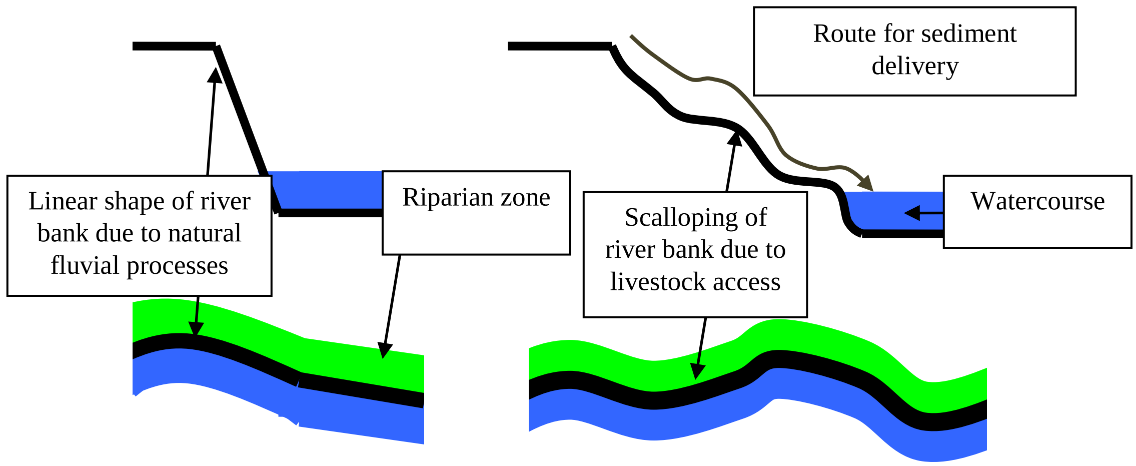
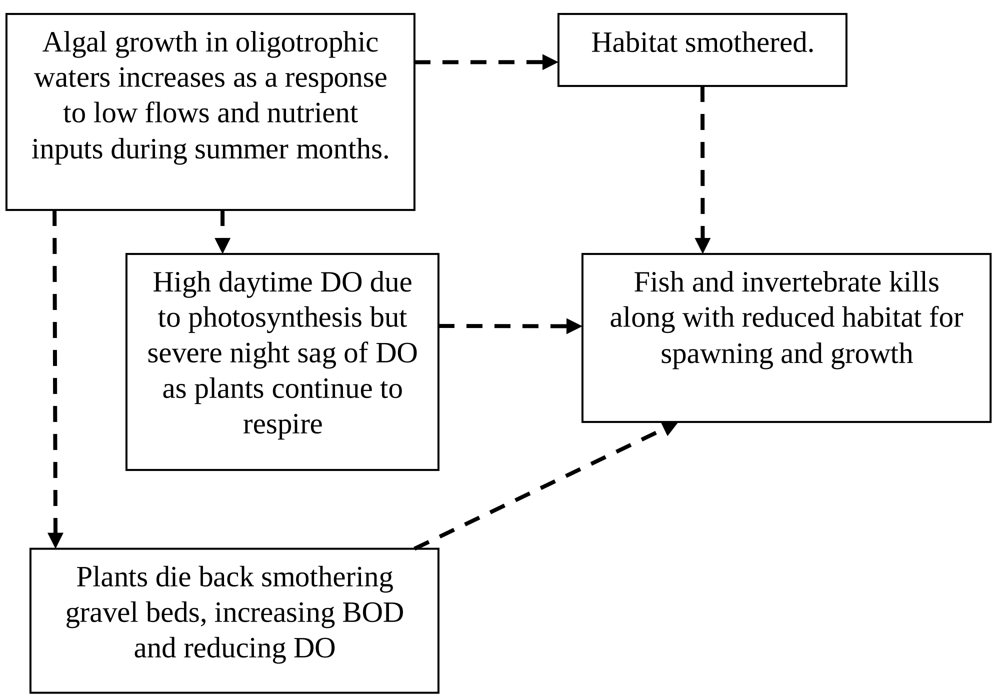


Fig. 6.9: Profile and aerial view of a) natural bank erosion from fluvial processes that may be exacerbated due to poor binding from riparian vegetation reduced through overstocking and b) bank erosion through overstocking and trampling from livestock showing the resulting irregular scalloping effect.

Fig 6.10: Algal growth can in itself be a major contributor to BT reductions within upland streams and rivers. Algal growth can have several deleterious knock-on effects to the local ecology within an oligotrophic system. The issue is only touched on in the preceding management diagrams, figs 2 to 5.



Sand silt fractions showed four significant relationships. There was a negative relationship with boulders and cobbles (r^2 adj. 42.86, P = 0.000), a positive relationships with emergent macrophytes (r^2 adj. 23.18, P = 0.000), earthcliffs on the stream bank (r^2 adj. 3.17, P = 0.027) and finally with siltation (r^2 adj. 2.66, P = 0.041). The relationships here are intuitive. Processes that deliver fine sediments (e.g. earthcliff collapse) are expected to place a positive control on in-stream fine sediment fractions. Larsen *et al.* (2009) found that sedimentation of gravel beds was directly linked to eroding banks within 500m upstream. When they increased the scale of inquiry, they discovered that bank erosion was negatively correlated with riparian and catchment woodland extent. The upper Ure catchment is dominated by agriculture and grouse moors with a low proportion of woodland. Therefore, a strong argument exists to incorporate woodland planting upstream of collapsing banks in conjunction with stream side management to stabilise river banks. As stated in Chapter Five, emergent macrophytes at these locations will be taking advantage of the increased fine sediment proportions and so this relationship is unlikely to be a controlling factor. In extreme cases, increased fine sediment delivery within the channel leads to siltation of gravel pore spaces (Grumiaux *et al,* 1998) whilst the high proportions of boulders and cobbles offers a descriptor of high flows that are the most likely to mobilise fine sediments preventing significant infiltration. Thus, the presence of the larger bedload fractions can be taken as an indicator of low fine sediment fractions. However, where there is a higher sand and silt fraction, it does not seem high enough to limit brown trout recruitment and the positive correlation indicates a mixed substrate suitable for the trout.

The stream area prone to drying was not further explored here despite a stepwise regression having been performed. This was due to the difficulties with the data, as discussed above. Stream order will also be passed over in this chapter. The stepwise regression against brown trout was not significant despite the species utilising low-order streams when spawning. This is probably due to the wide variation in the trout data.

The $SCIMAP_G$ (table 5.18) stepwise regression is highly significant. As the $SCIMAP_G$ risk weighting is increased, LIFE scores are reduced suggesting that these highly adapted macroinvertebrate species respond poorly to risk passing through, or perhaps settling in, the system. This adds some validity to the SCIMAP model in that it shows some significant relationships between aquatic biota and the model predictions. It is more difficult to explain the positive relationship returned between $SCIMAP_G$ and percent shading except that at some of these locations extensive coniferous plantations did offer some shade of the river bank. This suggests that the relationship is only an artefact of location. The apparent relationship between $SCIMAP_G$ and the presence of gravel is perhaps a response to these highly connected zones encouraging the delivery of gravels from land to water.

This initial stage of the analysis, in combination with the relevant literature, suggests there is something of a hierarchy in terms of impacts and restoration methods at the survey sites (figure 6.11). The restorative methods highlighted in figure 6.11 can be employed rapidly at appropriate locations once landowner agreement has been achieved. However, ensuring that the correct analysis has been carried out is essential. The next section explores stepwise regression of all the variables collected against rank average brown trout fry to explore how these results compare to the analysis presented above that incorporates only the significant correlations.

Figure 6.11: *Impacts on stream biota, the causse and possible restoration method.*

<u>Impact</u> → **<u>Underlying cause</u>** → **<u>Restoration methods</u>**

- 1: Algae → Soil poaching by livestock
  - → Buffer the stream bank from live stock
  - → Increase upstream tree cover
  - → Reduce nutrients from upstream & adjacent farms
- 2: Sand and silt → Earthcliffs on bank
  - → Buffer the stream bank from live stock
  - → Increase upstream tree cover
- 2: Sand and silt → Siltation processes
  - → Gravel cleaning
  - → Creation of spawning side channels
  - → Woodland clearance to be buffered at stream sides
  - → Sediment traps on farms

6.7.1 Explaining brown trout fry populations in relation to the complete dataset

It was felt prudent to run an additional stepwise regression analysis on the complete dataset against brown trout fry populations to see how this fitted with the results discussed above. The results from this showed five significant relationships that explained 41.35% of the variation (table 5.19). There was some fit with the earlier runs of the analysis with algae displaying a negative relationship with brown trout (r^2 adj. 12.39, P = 0.001) whilst sand and silt returned a positive relationship (r^2 adj. 7.27, P = 0.000) as before.

In addition, three other significant relationships were returned. The Simpsons diversity index displayed a positive relationship (r^2 adj. 5.98, P = 0.007) suggesting that as evenness within the community increases so do brown trout fry populations (this indices was expressed as 1/D). This provided the first indication that prey resource places a control on trout fry populations. As macroinvertbrate abundance did not reveal a significant relationship, it can be concluded that it is diversity of prey types that offers increased recruitment success. Thus, measures to enhance the habitat mosaic would be the most appropriate measures when exploring restoration measures (Jong *et al*, 1997). Siltation returned a negative relationship (r^2 adj. 4.81, P = 0.011). Again, this offers the first indication from this work that increasing fine sediments do indeed decrease brown trout fry viability in line with the extensive literature (e.g. Armstrong *et al*, 2003; Theurer *et al*, 1998). Finally, stock access displayed a negative relationship (r^2 adj. 2.87, P = 0.025). In conjunction with siltation and the discussion in the previous section, this highlights that stock access to streams rapidly passes risk to some forms of aquatic biota. The risk posed by stock access can include disruption to the physical habitat of the stream, fine sediment delivery through bank erosion or direct nutrient delivery. This is perceived as a major concern in the upper Ure catchment and so it was reassuring that the results offered further evidence that this is indeed an issue.

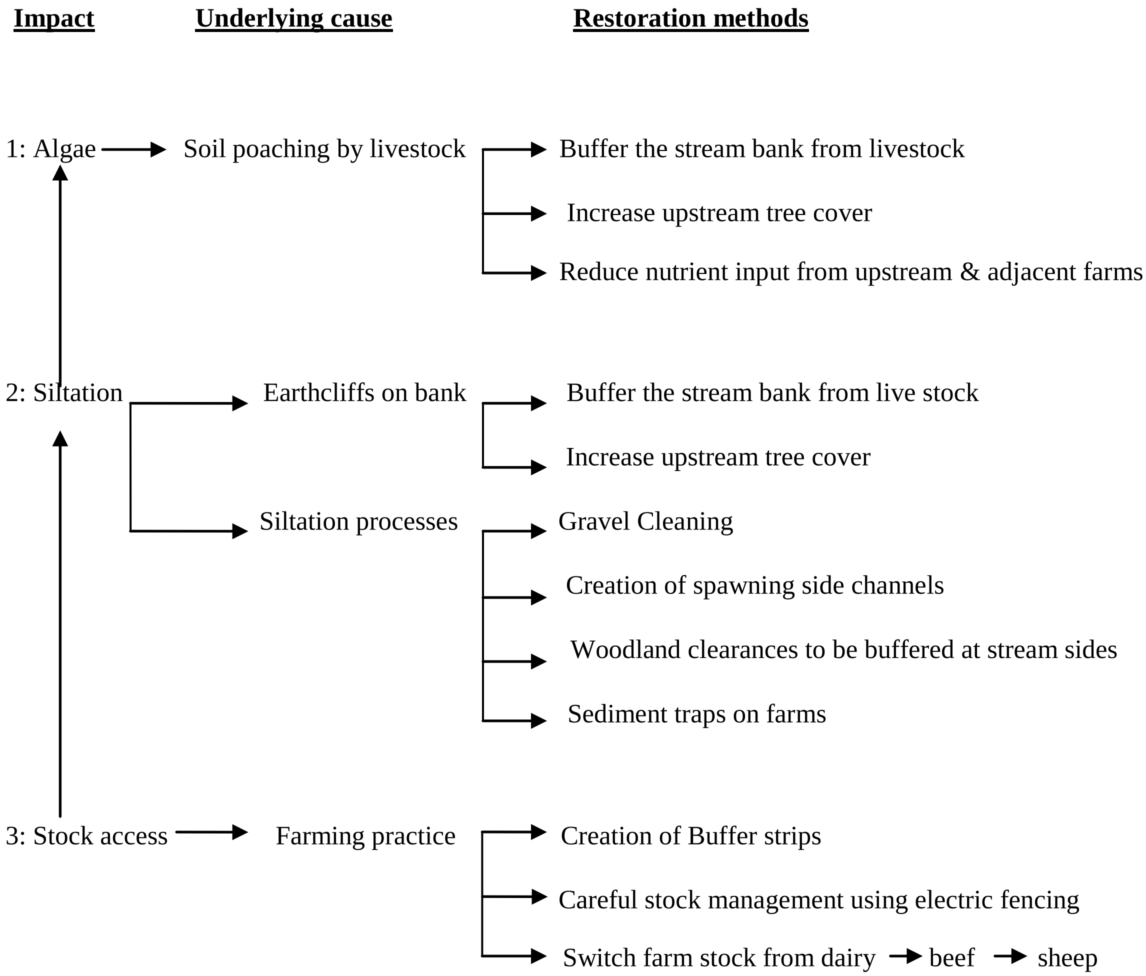
In the same manner as the earlier stepwise regressions, the variables that came back as significant had stepwise regressions carried out in order to assess the underlying drivers and/or relationships. The first run was against Simpson's diversity index. On the whole the relationships here were intuitive; however, no land-use impacts were apparent. The positive relationship with the Shannon's index (r^2 adj. 24.13, P = 0.000) is most probably due to the Simpsons index being presented as 1/D which shows evenness on a rising scale in a similar manner as the Shannon's index. The presence of pools in the survey reach appears to add diversity to the habitat and thus an increased range of microhabitats. Due to this the positive relationship (r^2 adj. 14.64, P = 0.001) from this run was to be expected. The positive relationship with upstream obstructions (r^2 adj. 10.79, P = 0.004) are a little less intuitive though this may simply be an artefact of location. These upland streams are known to be highly diverse due to a range of flow types, numerous microhabitats, high dissolved oxygen content and a varied substrate. Thus, the highest diversity indices may be expected to be derived from locations on low-order streams where waterfalls are increasingly prevalent.

The second run was against stock access to streams. This revealed three significant relationships. There was a negative relationship with the presence of buffer strips (r^2 adj.42.70, P = 0.002) and percent shading of the river (r^2 adj. 8.83, P = 0.003). These two variables are inter-related in that shading generally increases where buffer strips exist and so fencing out the river would not only be an effective measure for reducing stock access to watercourses it would, given time, increase the shading effect of the stream. The creation of buffer strips can be expected to be one of the better methods for ensuring stock access to streams is prohibited. The positive relationship with $SCIMAP_U$ (r^2 adj. 7.50, P = 0.008) at first appears unusual as this SCIMAP run in effect only displays a surface flow index. However, due to the difference in the model outputs and the actual situation within the catchment, the surface flow index is not necessarily a measure of surface wetness. Lane *et al* (2006) suggested that the catchment had a low level of underdrainage; however, investigations of SCIMAP at the farm scale enhanced the understanding of land management in the catchment. All of the meadows and pastures have been extensively underdrained; thus surface-flow indices fail to deliver accurate information of hydrological pathways. The index suggests that the greatest connectivity (averaged over time) occurs in close proximity to the stream network. However, land management techniques have modified the landscape thus ensuring that farming is viable at these locations. Due to this, the index of surface flow provided by $SCIMAP_U$ may be a coincidental relationship with stock access due only to location.

The final statistical run was against siltation. This returned four significant relationships, some of which have already been discussed above. For example the positive relationship with sand and silt fractions within the substrate (r^2 adj. 33.36, P = 0.000) is simply due to higher fractions of fine substrate occurring at locations impacted by siltation. In contrast, the positive return with poaching of soils (r^2 adj. 4.49, P = 0.012) by livestock can be considered to be one of the controlling factors on the siltation process. Macroinvertebrate richness displayed a positive relationship against siltation (r^2 adj. 13.09) again suggesting that diverse substrate composition adds to the microhabitat mosaic which in turn appears to enhance macroinvertebrate richness/evenness. Downstream obstructions (<500m) displayed a positive relationship here (r^2 adj. 10.78, P = 0.002). Previous explanations of this relationship most likely hold. These low-order streams have a high proportion of waterfalls along their length and so it can be expected that many of the variables will display significant relationships with their presence.

This second analysis provides evidence that supports the earlier stepwise regressions but also adds some detail to the thesis in that it provides better evidence of land management impacts on freshwater biota. Moreover, it provides the first indication that prey species dynamics can place a controlling factor on fry populations. As dominance within a community is often the result of some level of impact, either due to pollution sources or habitat degradation, evenness can be taken as an indication that water and habitat quality are relatively high. Moreover, in this instance it appears to provide an indicator that there is a good habitat mosaic which in turn not only offers increased prey sources for brown trout fry but also increased locations that can be utilized as refuge when external pressures flush through the system. Due to this second analysis, the diagram presented in figure 6.12 can be added to in order to provide increasing knowledge of impacts, underlying causation and possible restorative measures.

Figure 6.12: *When the second set of stepwise regression results are incorporated the impacts increase, though the causes and restoration methods remain largely the same with, for example, the addition of enhancing instream structure.*



In addition, there are a number of in-stream factors that aquatic organisms typical of upland streams prefer. Adding structure to the stream appears to enhance macroinvertebrate communities, for example. Research suggests that brown trout respond positively to improved habitat structure arising from large woody debris, tree roots and a dappled shading effect from riparian tree cover (Armstrong et al, 2003: Crisp, 2000; Greenberg and Dahl, 1998; Poff, 1997; Allan, 1995). The results attained from these investigations seem to support this. For example, buffer strips appear to improve habitat, although here the major benefit of this measure would be to keep stock excluded from watercourses. Indeed, the strongest negative controls on brown trout fry populations appear to arise from land use practices in this catchment (algae, stock access and siltation), despite many of the sites surveyed being at higher altitudes than the most intensive farming practices in the catchment.

6.8 Developing restoration plans

There are a number of sub-optimal land management practices taking place in the upper Ure catchment. These include spreading of slurry in wet conditions or directly on snow, intensive stocking rates of dairy and other stock, high-density gripping of peatlands, a paucity of buffer strips and woodland cover (which could be either along riparian zones or acting as surface flow-mitigating shelter belts). Many of these decisions are directly driven by economic considerations. For example, many farms lack the infrastructure to store slurry and thus spreading is governed not by weather patterns but by storage issues. These issues attached to farming practices are not unique and transmit risk to rivers in the form of nutrients, sediments and physical habitat degradation.

The results here suggest there is a cascade of sediment and nutrient delivery to the river network that begins at the field-scale and culminates with in-stream algal growth during summer months. This is the period when brown trout fry have established territories and begun exogenous feeding. It also describes one of the more vulnerable stages in the brown trout life cycle. Thus, targetting of restoration effort would be most efficient at locations where fry populations are clearly suppressed. These locations show that spawning is viable but due to environmental conditions egg to fry survival is poor. Once investigations have identified the limiting factors, such locations offer the quickest restoration opportunities. After this, restoration effort can be broadened out into apparently good spawning habitat where there has been no evidence of fry populations.

This work highlights that farm management can be one of the more pernicious controls on salmonid recruitment. The upper Ure catchment is an unusual catchment within the Yorkshire Dales National Park in that it maintains high numbers of dairy farms. These are riskier operations than beef or sheep enterprises. However, at the upper locations of the catchment surveyed, the major land management types are sheep, beef and grouse moor. At these locations there are very few buffer zones and stock have ready access to the stream. So, despite the more damaging farming methods confined largely downstream (though not exclusively), the streams at these upland locations are still being impacted by insensitive farming methods. Adding structure to these locations through tree planting along the riparian zone, fencing out the river from stock, adding large woody debris to the channel or by increasing upstream tree coverage can improve conditions by reducing sediment and nutrient inputs and offering an increased habitat mosaic.

Most river trusts have the capacity to carry out these investigations or are rapidly increasing capacity. With the recent announcement of a Defra £110 million fund for river improvements (ART, 2011) wide concern over river ecology has been recognised as a high-priority conservation issue by policy makers. This new phase in river management has been driven by the WFD. In terms of the upper Ure catchment, five of the water bodies are considered to be of "moderate ecological status" whilst seven are considered to be meeting "good ecological status" having been assessed in line with the WFD. However, the information used to make these assessments has often been based on expert opinion alone. Only two of the failing water bodies (main river Ure between Duerley Beck at Hawes and Aysgarth and the Raydale subcatchment) have any level of detail attached to allow a sound assessment of present and future condition.

Unfortunately this situation is not uncommon. Many other river systems have poor levels of detail attached and so the information imperative is not consigned to the upper Ure system. Without rapid work to fill these knowledge gaps, bringing UK rivers (and those in other member states) up to good ecological status will be inefficient and beset with failure. Thus, models such as SCIMAP are much needed tools that will allow restoration ecologists, government agencies and conservation charities, such as the rivers trusts, to direct investigations in data-poor catchments. The Defra fund for work on rivers is specifically targetted to bring failing water bodies up to good ecological status, providing a unique opportunity to direct restoration work to the country's poorest rated water bodies. However, work must be intelligently devised so that it is informed (and informs) and based on catchment-scale knowledge of river systems allowing the poorer quality sections to be targeted and quick win-wins to be identified.

This work shows that SCIMAP is able to identify the most risky location but that nuances in the landscape, high proportions of subsurface flow and differences in land management style can reduce the model's accuracy. These issues are understood and the model needs to be used in conjunction with wider investigations to ensure that SCIMAP acts only as a guide. In combination with data collection in the field, and through remote sensing and GIS methods, smart approaches to investigation can rapidly develop detail on river networks.

These data can then be processed through stepwise regressions against biotic components of the river system (e.g. fish or macroinvertebrates) to highlight what the limiting factors are and where they exist. Many rivers trusts already possess detail on the biological nature of the river system and habitat. Thus, the first stage of the investigations may already be complete. The final stage is to match restoration method with known problems. For example, buffer strips reduce poaching and siltation (Feyen *et al*, 2000) whilst increasing shading, in-stream woody debris adds structure to the habitat creating new habitats which improve biological diversity of all three measures (genetic, species and ecosystem) whilst willow spiling supports eroding banks and changes an erosion issue to a deposition process behind the stakes. This restores the bank-side and may act to narrow the river channel increasing depth which may at as a buffer against drought conditions. In addition, this method increases shading, adds structure to the bank foot habitat and provides a salmonid food source in the form of terrestrial invertebrates.

Hydrological connectivity is a catchment scale concept that acts to link riparian, floodplain and catchment controls to each river reach. Whilst it may not be possible to achieve wholesale restoration of catchments, it is possible to analyse diffuse pollution at the catchment scale to help identify the locations which have disproportionate impact on the stream networks. This helps with targetting and also provides the best locations for restoration measures, subject to negotiation with the appropriate stakeholders, in particular the landowning and agricultural communities. Thus, catchment-scale investigations offers the best chance of improving the ecology of river systems over the largest geographic areas and in this way the worst locations can be married with the best restoration method.

6.9 Thesis summary

Chapter 1 introduced the aim and objectives of the thesis. It outlined the justification for the research and how it fits with present policy and scientific advances concerning rivers and their catchments. It provided a summary structure of the thesis structure introducing the general themes.

Chapter 2 explored the themes briefly introduced in Chapter 1, setting the context and developing the concepts of the research. It outlined the current awareness of salmonid ecology, focusing on brown trout and ultimately on brown trout fry. This provided detail on a number of limiting factors that impact on the species at this critical life stage. It explored these impacts in terms of biotic, abiotic and then human factors that suppress, or enhance populations. Three process cascades were mapped to conceptualise how land management can cascade through a catchment, and ultimately impact ecological components of rivers. These highlighted how there can be a spatial mismatch between cause and impact.

The chapter then introduced a number of important concepts and themes that implicitly run through this thesis. These included hydrological connectivity and restoration ecology. Hydrological connectivity is an integral aspect of riverscapes; it has a number of dimensions including vertical, horizontal, longitudinal and temporal. These link habitats in time and space and are essential components for understanding river ecosystems. Indeed, restoration effort has often failed due to not considering such large-scale detail. In discussing restoration ecology, there appeared an apparent mismatch between the scales of catchment processes and the scales of restoration effort. Often restoration is stuck between pragmatic reality, governed by finances and property right issues, and idealised notions of restoring ecosystems back to some pre-disturbance state. Whilst it is important to consider scale and process when restoring habitats, it is not possible, under present socio-economic conditions, to carry out wholesale restoration of catchments to some idealised baseline condition. Even if we knew what pre-disturbance means, ecosystems are not linear systems that can be see-sawed back and forth to suit current thinking. The chapter finished with a brief consideration of remote sensing, GIS and modelling capabilities that can be used for site identification and mapping risk in locations otherwise inaccessible and ended with a short exploration of the legislative context of catchments, rivers and ecology at the national and international level. Chapter 2 provided the foundation for meeting Objective Two

Chapter 3 expanded on Chapter 2 by discussing the case study setting of this thesis. It provided a justification for a case study approach in comparison to conventional experimental approaches. It also discussed some of the disadvantages of setting research in complex systems were variables cannot be manipulated. The chapter introduced the sub-catchments of the upper Ure and discussed them in terms of land use and probable pressures on the riverscape. After this it gave a broader view of land use and current awareness on the condition of brown trout populations in the catchment. The next section explored the localised institutional framework at the national down to county level. The Yorkshire Dales Rivers Trust was introduced as a local NGO with a keen interest in river restoration and who have been central in commissioning this research. This chapter provided information on the state of the local brown trout stocks which helped meet objective two of the thesis.

Chapter 4 focused on Objective 2 by explored novel approaches to catchment investigation. Attention focused on the possibilities that remote sensing and GIS offer in terms of catchment exploration and data manipulation. The notion of extending the peer review community to encompass local knowledge was explored, in terms of how model outputs could be tested against farmers understanding of the land they manage. This provided a more inclusive approach to scientific investigation, and importantly, opens dialogue between those aiming to restore river systems and the owners of the land where restoration effort has to take place.

Remote sensing, GIS and modelling methods used in this thesis were described. Capturing the extent and number of upland drainage channels was carried out through remote sensing in ArcGIS using aerial photographs. After this, the SCIMAP model was explained in terms of how it identifies possible risk as a combination of CSAs that are connected to a watercourse. The process for running SCIMAP at the catchment-scale was described. SCIMAP was run on three separate occasions: 1) $SCIMAP_U$, 2) $SCIMAP_L$ and 3) $SCIMAP_G$. The results showed how risk is redistributed when land management risk is incorporated into the model and shifted upstream when grips are included. The statistical methods were introduced here.

Chapter 5 presented the results of the work and continued with the theme of extending the peer review process. SCIMAP was modelled on eight land holdings in the catchment. Validation of the model could not be attained through the visits for a number of reasons: 1) the meadows were more extensively under-drained than expected, 2) natural sink holes shifted surface flow underground, and, 3) differences in land management practice. The results did show that the model would offer reasonable detail in the absence of such nuances. From the results it did appear that farmers had responded inherently, over generations, to risk by adapting management that reduced soil to water loss.

Chapter 6 discussed the results and explored their implications within the case study catchment and further afield. The results showed that $SCIMAP_G$ provided significant explanation even though it did not explain a high percentage of the variance. As $SCIMAP_G$ most closely resembled the landscape, this result was heartening. However, the relationship was positive which was surprising. This is probably due to collinearity between variables so that gripping indicates flow conditions conducive to brown trout whilst the sand-silt fraction indicates a flow regime more conducive to gravels and finer material as opposed to boulders and cobbles. Other important factors to arise were stock access and poaching of soils and other land use factors impacted on brown trout fry populations. In-stream habitat is highly connected to the surrounding land and so responds rapidly to pressures arising from intensification of land use. Identifying which impacts matter to ecological components of river ecosystems is complex due to the multi-factor and multi-scale complexity of catchments. Although, here it seems likely that as upstream contributing area is low, the impacts from adjacent habitats are more keenly felt than the catchment-scale impacts. Finally, a brief discussion on restoration of habitats identified which methods would be most suitable and where these should be employed.

The overall aim was achieved by incorporating a variety of in-field and remote methodologies into an investigation of brown trout fry populations. The research identified the most keenly felt impacts and where these are located. Moreover, it has offered suggestions on which restoration methods would best remedy the impacts identified. In this way, the work here could be used at other locations. Most rivers trusts and other conservation charities have the capacity to carry out similar investigations. Brown trout fry could be substituted for different organisms, or community, of interest and from this it would be a simple process to identify which variables to capture through the data acquisition process. The SCIMAP model is freely available to charitable organisations and The Association of Rivers Trusts (the overseeing body of the local river trusts) has a good GIS unit that would assist when difficulties arise. Stepwise regression could be carried out in Excel with the inclusion of a data analysis add-in and from this an understanding of the limiting controls on biotic components of communities could be developed. The work here shows that this is viable and that under WFD funding, financial constraints are less an issue when such work is expected to lead to restorative action along river systems.

This work shows that catchment-scale investigations can be quickly employed in order to identify the more pernicious controls on in-stream ecological units, where these issues occur and what methods may reduce their impact. This provides a useful tool as we move into the active stage of WFD work where restoration measures are beginning to be employed by agencies and charities, often in partnership. The suggestion here is that catchment-scale investigation, and restoration, offers the best chance of directing resources efficiently.

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8.0 APPENDIX 1: Pearsons Correlations

	Macroinv_ab	Macroinv_rich	simpsons, 1/	Shannon's
Macroin_rich	0.353			
	0.024			
Simpsons, 1/	-0.022	0.414		
	0.891	0.007		
Shannon's	-0.013	0.190	0.510	
	0.938	0.235	0.001	
LIFE scores	0.327	0.607	0.260	0.329
	0.037	0.000	0.101	0.036
Obstructions	-0.100	0.164	0.371	0.113
	0.532	0.305	0.017	0.482
Obstructions	-0.177	-0.115	0.024	-0.011
	0.267	0.472	0.881	0.944
Obstructions	-0.118	-0.123	0.213	0.302
	0.463	0.445	0.182	0.055
Obstructions	-0.112	-0.128	-0.032	-0.160
	0.487	0.423	0.841	0.316
survey area	0.189	0.056	-0.134	-0.146
	0.237	0.730	0.402	0.363
stream prone	0.073	0.323	0.279	0.156
	0.651	0.039	0.077	0.329

stream prone	0.033	0.024	0.165	0.141
	0.838	0.880	0.302	0.379
bedrock	-0.047	-0.193	-0.092	0.048
	0.771	0.226	0.567	0.764
boulbers and	0.030	0.004	0.004	0.131
	0.854	0.979	0.982	0.415
gravel	0.161	0.092	0.016	-0.130
	0.316	0.568	0.923	0.418
sand and sil	-0.196	0.011	0.042	-0.064
	0.220	0.947	0.792	0.692
siltation	-0.135	0.332	0.310	0.163
	0.400	0.034	0.048	0.309
River width	-0.178	-0.025	-0.256	0.019
	0.264	0.875	0.106	0.905
pools presen	0.150	0.109	0.429	0.063
	0.348	0.497	0.005	0.698
algae: 1: lo	-0.062	0.244	0.182	0.230
	0.702	0.125	0.256	0.148
macrophytes	-0.187	0.015	0.033	0.080
	0.242	0.926	0.838	0.619
undercut	0.233	0.082	-0.123	-0.148

	0.142	0.610	0.443	0.355
earthcliff	-0.034	-0.034	-0.112	-0.091
	0.834	0.833	0.486	0.571
stock access	0.012	0.054	0.006	-0.146
	0.939	0.737	0.970	0.362
buffer	-0.270	-0.092	-0.183	0.122
	0.087	0.567	0.252	0.448
land use	-0.093	-0.107	-0.080	-0.018
	0.563	0.504	0.618	0.911
poached	0.079	0.199	-0.026	0.056
	0.625	0.212	0.872	0.728
% shade	-0.120	0.008	0.193	0.284
	0.455	0.961	0.226	0.072
upstream con	-0.132	0.042	-0.277	0.019
	0.409	0.793	0.080	0.905
Area_upst_moor	-0.166	0.012	-0.316	0.007
	0.301	0.940	0.044	0.968
Strahler str	0.069	0.131	-0.322	-0.046
	0.670	0.416	0.040	0.773
$SCIMAP_L$	0.220	-0.327	-0.175	-0.273
	0.167	0.037	0.275	0.084

$SCIMAP_G$	-0.139	-0.334	0.032	-0.065
	0.388	0.033	0.845	0.686
$SCIMAP_U$	-0.143	-0.086	0.084	0.110
	0.372	0.592	0.604	0.492

	LIFE scores	Obstructions	Obstructions	Obstructions
Obstructions	0.231			
	0.147			
Obstructions	-0.260	-0.180		
	0.101	0.259		
Obstructions	-0.065	0.329	0.108	
	0.687	0.036	0.503	
Obstructions	0.092	0.019	-0.058	-0.100
	0.569	0.906	0.720	0.535
survey area	0.046	0.256	-0.154	-0.017
	0.776	0.106	0.335	0.918
stream prone	0.065	0.179	0.232	0.147
	0.684	0.264	0.144	0.359
stream prone	0.072	0.481	-0.108	0.138
	0.654	0.001	0.501	0.388
bedrock	0.033	-0.285	0.147	-0.090
	0.838	0.071	0.360	0.578

boulbers and	0.307	0.236	-0.126	0.106
	0.051	0.138	0.432	0.511
gravel	-0.290	-0.146	0.040	-0.109
	0.066	0.361	0.803	0.498
sand and sil	-0.108	0.020	0.042	0.060
	0.501	0.901	0.795	0.711
siltation	0.020	0.187	0.368	0.168
	0.900	0.242	0.018	0.294
River width	0.238	-0.217	-0.182	-0.265
	0.135	0.172	0.254	0.094
pools presen	0.157	-0.054	-0.108	-0.039
	0.325	0.739	0.501	0.809
algae: 1: lo	0.288	0.142	-0.211	-0.023
	0.068	0.375	0.185	0.888
macrophytes	-0.011	0.256	-0.154	-0.017
	0.945	0.106	0.335	0.918
undercut	0.118	-0.503	-0.062	-0.153
	0.462	0.001	0.700	0.338
earthcliff	0.048	-0.188	0.061	-0.160
	0.768	0.240	0.703	0.319
stock access	-0.125	0.137	0.208	0.262
	0.436	0.393	0.192	0.098

buffer	0.196	-0.232	-0.229	-0.089
	0.219	0.144	0.150	0.579
land use	0.016	-0.024	-0.075	-0.034
	0.919	0.884	0.639	0.833
poached	-0.007	-0.125	0.129	-0.024
	0.964	0.435	0.421	0.881
% shade	0.116	-0.270	-0.102	-0.039
	0.471	0.088	0.525	0.807
upstream con	0.197	-0.220	-0.072	-0.289
	0.218	0.167	0.655	0.067
Area_upst_Moor	0.242	-0.242	-0.125	-0.242
	0.127	0.128	0.437	0.127
Strahler str	0.205	-0.416	-0.089	-0.319
	0.198	0.007	0.579	0.042
$SCIMAP_L$	-0.564	-0.140	-0.059	0.056
	0.000	0.383	0.713	0.726
$SCIMAP_G$	-0.640	-0.257	0.128	0.018
	0.000	0.105	0.425	0.912
$SCIMAP_U$	-0.104	0.197	0.137	0.373
	0.518	0.216	0.393	0.016

	Obstructions	survey area	stream prone	stream prone
survey area	-0.137			
	0.393			
stream prone	-0.096	-0.036		
	0.552	0.824		
stream prone	-0.188	0.427	-0.084	
	0.239	0.005	0.602	
bedrock	0.204	-0.102	-0.071	-0.239
	0.201	0.526	0.658	0.132
boulbers and	0.075	0.326	-0.276	0.184
	0.639	0.037	0.081	0.250
gravel	-0.273	-0.185	0.294	-0.177
	0.084	0.247	0.062	0.270
sand and sil	0.081	-0.174	0.083	0.126
	0.615	0.278	0.606	0.431
siltation	-0.148	-0.146	0.246	0.177
	0.357	0.363	0.121	0.269
River width	0.152	-0.278	-0.227	-0.563
	0.341	0.078	0.153	0.000
pools presen	0.078	-0.120	-0.084	0.003
	0.629	0.455	0.602	0.983
algae: 1: lo	-0.071	-0.070	-0.049	0.210

	0.658	0.662	0.761	0.187
macrophytes	0.118	-0.051	-0.036	0.153
	0.461	0.750	0.824	0.338
undercut	0.204	-0.314	-0.220	-0.364
	0.201	0.045	0.168	0.019
earthcliff	-0.020	-0.111	-0.078	0.185
	0.900	0.488	0.629	0.247
stock access	0.027	0.146	0.102	0.212
	0.869	0.363	0.527	0.184
buffer	0.051	-0.183	-0.128	-0.284
	0.751	0.252	0.426	0.072
land use	0.170	-0.068	0.030	-0.188
	0.287	0.673	0.851	0.239
poached	-0.014	-0.074	-0.052	0.024
	0.933	0.644	0.747	0.881
% shade	-0.143	-0.160	-0.112	-0.284
	0.371	0.317	0.486	0.072
upstream con	0.035	-0.233	-0.179	-0.479
	0.829	0.143	0.263	0.002
Area_upst_moor	0.084	-0.239	-0.230	-0.509
	0.600	0.133	0.148	0.001

Strahler str	0.016	-0.226	-0.251	-0.494
	0.921	0.156	0.113	0.001
$SCIMAP_L$	-0.198	0.069	0.121	-0.084
	0.215	0.668	0.449	0.602
$SCIMAP_G$	-0.225	-0.106	-0.074	-0.123
	0.158	0.510	0.646	0.445
$SCIMAP_U$	0.151	-0.014	0.329	-0.033
	0.347	0.930	0.036	0.836

	bedrock	boulbers and	gravel	sand and sil
boulbers and	0.070			
	0.663			
gravel	-0.504	-0.679		
	0.001	0.000		
sand and sil	-0.218	-0.665	0.119	
	0.171	0.000	0.457	
siltation	-0.228	-0.470	0.210	0.592
	0.152	0.002	0.187	0.000
River width	0.263	0.146	-0.245	-0.109
	0.097	0.362	0.122	0.496
pools presen	0.067	-0.345	-0.019	0.482
	0.676	0.027	0.907	0.001

algae: 1: lo	0.009	-0.227	-0.097	0.452
	0.953	0.154	0.544	0.003
macrophytes	-0.102	-0.198	-0.185	0.607
	0.526	0.214	0.247	0.000
undercut	0.265	-0.293	0.003	0.242
	0.094	0.063	0.988	0.127
earthcliff	0.133	-0.413	0.010	0.497
	0.406	0.007	0.950	0.001
stock access	-0.082	-0.135	0.059	0.170
	0.612	0.399	0.716	0.288
buffer	0.275	0.254	-0.299	-0.194
	0.082	0.109	0.058	0.223
land use	0.031	0.015	0.028	-0.059
	0.847	0.925	0.861	0.712
poached	-0.053	-0.431	0.282	0.314
	0.741	0.005	0.074	0.045
% shade	-0.027	0.024	0.072	-0.107
	0.867	0.880	0.656	0.504
upstream con	0.058	0.020	-0.118	0.074
	0.717	0.902	0.462	0.646
Area_upst_moor	0.098	0.097	-0.181	0.012
	0.541	0.546	0.256	0.941

Strahler str	0.035	-0.095	0.144	-0.049
	0.830	0.555	0.370	0.762
$SCIMAP_L$	-0.306	-0.413	0.559	0.141
	0.052	0.007	0.000	0.378
$SCIMAP_G$	-0.241	-0.319	0.497	0.022
	0.129	0.042	0.001	0.892
$SCIMAP_U$	0.086	0.045	0.036	-0.201
	0.591	0.780	0.822	0.207

	siltation	River width	pools presen	algae: 1: lo
River width	-0.238			
	0.134			
pools presen	0.177	-0.134		
	0.269	0.402		
algae: 1: lo	0.369	0.204	0.273	
	0.018	0.201	0.085	
macrophytes	0.352	0.193	0.153	0.410
	0.024	0.228	0.338	0.008
undercut	0.011	0.236	0.382	0.223
	0.945	0.137	0.014	0.160
earthcliff	0.224	-0.065	0.259	0.369
	0.158	0.687	0.102	0.018

stock access	0.178	-0.163	-0.047	0.086
	0.265	0.308	0.769	0.593
buffer	-0.323	0.435	-0.103	-0.060
	0.040	0.004	0.521	0.708
land use	-0.193	0.197	-0.043	-0.245
	0.227	0.217	0.789	0.122
poached	0.511	-0.100	0.024	0.421
	0.001	0.533	0.881	0.006
% shade	-0.136	0.149	0.213	-0.072
	0.397	0.351	0.182	0.653
upstream con	0.044	0.814	-0.195	0.332
	0.783	0.000	0.222	0.034
Area_upst_moor	-0.076	0.822	-0.200	0.220
	0.636	0.000	0.210	0.166
Strahler str	-0.133	0.667	-0.075	0.242
	0.406	0.000	0.642	0.127
$SCIMAP_L$	-0.002	-0.204	0.053	-0.145
	0.988	0.200	0.744	0.366
SCIMAPG	-0.072	-0.233	0.160	-0.310
	0.653	0.143	0.317	0.048
$SCIMAP_U$	-0.155	-0.139	-0.210	-0.374

	0.332	0.387	0.187	0.016

	macrophytes	undercut	earthcliff	stock access
undercut	-0.076			
	0.638			
earthcliff	0.174	0.225		
	0.276	0.158		
stock access	-0.103	0.102	0.046	
	0.521	0.526	0.774	
buffer	-0.044	-0.112	0.055	-0.664
	0.784	0.484	0.731	0.000
land use	-0.068	-0.088	-0.027	-0.281
	0.673	0.586	0.869	0.075
poached	-0.074	0.237	0.253	0.212
	0.644	0.136	0.111	0.184
% shade	-0.102	-0.157	0.082	-0.642
	0.527	0.328	0.610	0.000
upstream con	0.363	0.101	-0.059	-0.223
	0.020	0.528	0.712	0.162
Area_upst_moor	0.308	0.123	-0.045	-0.288
	0.050	0.444	0.778	0.068
Strahler str	0.043	0.291	0.092	-0.057

	0.792	0.065	0.565	0.722
SCIMAP$_{L}$	-0.036	0.066	0.036	0.002
	0.824	0.681	0.822	0.988
SCIMAP$_{G}$	-0.106	-0.156	-0.034	-0.270
	0.510	0.329	0.835	0.087
SCIMAP$_{U}$	-0.305	-0.131	-0.242	0.316
	0.052	0.414	0.127	0.044

	buffer	land use	poached	% shade
land use, 1	0.508			
	0.001			
poached	-0.266	-0.260		
	0.093	0.100		
% shade	0.584	0.266	0.002	
	0.000	0.092	0.992	
upstream con	0.340	0.035	0.109	0.225
	0.029	0.828	0.499	0.158
Area_upst_moor	0.473	0.166	0.012	0.287
	0.002	0.299	0.940	0.069
Strahler str	0.205	0.078	0.159	0.208
	0.199	0.630	0.320	0.192
SCIMAP$_{L}$	-0.294	0.097	0.100	-0.030

	0.062	0.547	0.533	0.852
$SCIMAP_G$	0.005	0.064	0.021	0.444
	0.977	0.690	0.895	0.004
SCIMAP	-0.064	0.279	0.015	-0.009
	0.693	0.077	0.928	0.955

	upstream con	area_upst_moor	Strahler str	$SCIMAP_L$
area_upst_moor	0.954			
	0.000			
Strahler str	0.693	0.679		
	0.000	0.000		
$SCIMAP_L$	-0.183	-0.204	-0.117	
	0.252	0.202	0.465	
$SCIMAP_G$	-0.184	-0.156	-0.066	0.576
	0.249	0.330	0.681	0.000
$SCIMAP_U$	-0.288	-0.206	-0.199	-0.010
	0.068	0.195	0.213	0.951

	$SCIMAP_G$
$SCIMAP_U$	0.004
	0.980