DURHAM UNIVERSITY

THE EFFECTS OF LAND MANAGEMENT AND PREDICTED CLIMATE CHANGE ON HYDROLOGICAL CONNECTIVITY AND DIFFUSE FINE SEDIMENT POLLUTION RISK WITHIN THE RIVER EDEN CATCHMENT.

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Declaration

I confirm that no part of the material presented in this thesis has previously been submitted by me or any other persons 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.

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Abstract

There is a growing recognition that future management of the water quality in UK rivers will depend upon an improved understanding of the effects of projected climate change on catchment systems. Until recently, little attention has been given to the secondary effects that climate change may have. However, it is now becoming clear that successful management will depend upon research into factors beyond the primary changes in soil moisture and river flows. One area of particular concern is the way climate change may alter patterns of diffuse pollution of fine sediment, with associated impacts on river flora and fauna.

If the UK is going to meet the stringent targets laid out in the EU Water Framework Directive, then urgent management of diffuse pollution is required. In 2012 only 28% of water bodies met their ecological potential or good status and 67% of river water bodies cite diffuse pollution as a key pressure which is preventing improvement and the achievement of good ecological status (Environment Agency, 2012). For management solutions to be cost-effective, they need to be targeted at the key problem areas within a catchment. This research uses the River Eden catchment in Cumbria as a test catchment and applies a hydrological simulation model, risk mapping framework and risk filter to the area in order to determine current connectivity and diffuse pollution trends. From this toolkit, projections of the future patterns of risk are calculated.

The SCIMAP based toolkit predicted that the fine-sediment erosion risk varies spatially across the River Eden catchment. Locations deemed to be most at risk of causing a fine-sediment pollution issue are in the lower reaches of the catchment where intensive arable farming is found. When risks were modelled temporally, variations depending upon vegetation cover and average monthly rainfall were found. It was noted that the presence of autumn-sown crops could reduce risk over a year whilst spring-sown crops are likely to increase fine-sediment erosion risks.

Several conclusions are drawn from this research: 1) it has been shown that the SCIMAP framework is an effective way of identifying critical source areas of diffuse pollution and could prove an invaluable tool to environmental managers; 2) the important role that autumn-sown crops can play in minimising erosion risk has been shown to be applicable in the River Eden catchment and the best way to incorporate this into crop cycles highlighted; 3) through the use of projected climate change data and a hydrological simulation model, it has been shown that the location of critical source areas are likely to change as a result of projected climate

change and associated variability in rural land management. This highlights the need for continuous catchment-wide monitoring and management of hydrological connectivity and associated diffuse pollution risks.

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Chapter 1: Introduction to Research

1.1 Background to Research

There is a growing recognition that future management of the water quality in UK rivers will depend upon an improved understanding of the effects of projected climate change on catchment systems. Until recently, little attention has been given to the secondary effects that climate change may have. However, it is now becoming clear that successful management will depend upon research into factors beyond the primary changes in soil moisture and river flows. Many studies have focused on the projected impacts that climate change and changes in land use may have on runoff and the associated implications for flooding (e.g. O'Connell et al., 2004; Raven et al., 2010). Conversely, few researchers have attempted to study the additional effects that changes to hydrological dynamics within catchments may have on channel morphology and sediment delivery rates, both of which are explicitly linked to land use and climate (Raven et al., 2010).

One potential secondary effect of projected climate change that is of particular concern is the way variations in climate may alter patterns of diffuse fine sediment pollution, with associated impacts on flora and fauna (Heathwaite, 2002). If the UK is going to meet the stringent targets laid out in the EU Water Framework Directive that all waterways must achieve good chemical and ecological status by 2015, then urgent management of diffuse fine sediment pollution is required. In 2012 only 28% of water bodies met their ecological potential or good status and 67% of river water bodies cite diffuse pollution as a key pressure which is preventing improvement and the achievement of good ecological status (Environment Agency, 2012). Lane et al. (2006) discuss the necessity of prioritising locations based upon their relative risk in order to ensure that finite resources are targeted on areas most at risk hence ensuring the maximum environmental improvement for the invested money.

Successful management of fine sediment risk is dependent upon addressing the effects that changes in spatial and temporal connectivity will have on pollution risk (Reaney et al., 2011; Pringle, 2003). Determined predominantly by topography and rainfall patterns, the pattern and strength of the hydrological connectivity controls overland and subsurface flows and hence the export of fine sediment to receiving waters (Bracken & Croke, 2007). In order to simulate current diffuse pollution dynamics, these processes have been studied using a 'source, mobilisation, pathway and delivery' framework. It has been predicted that connectivity and associated pathways may alter in response to projected climate change driven variations in the intensity and duration of rainfall, thus meaning the mobilisation and transport of material will also be altered (Reaney et al., 2011). This is likely to be particularly evident in catchments where surface topography is the primary driver of connection. Topographic variability which controls hydrological connectivity occurs at the small scale (less than 10m) and, as a result of this, catchment-scale models that are capable of capturing small scale processes are required (Lane et al., 2006).

Connection pathways and fine sediment pollution risk are closely linked to land management practices at both the catchment and the farm scale. Agriculture, in the broadest sense of the word, is thought to be responsible for the vast majority of diffuse pollution found across the UK (Withers & Lord, 2002). Soil loss from agricultural land has increased substantially over the last half a century and accelerated erosion has led to soil losses in some areas of 20 t ha^{-1} yr^{-1} (Merrington et al., 2002). As a result of this export of diffuse pollution, suspended sediment yields in some waterways across the UK have doubled.

Traditional, environmental simulation models (e.g. SHETRAN; Nasr et al., 2006), have been used to simulate patterns of diffuse pollution risk, but have treated hydrological connectivity too simply, or not at all,, meaning many of the complex interactions and feedbacks that drive the export processes are not captured. For example, SHETRANS is often run at a $1km^2$ grid resolution, hence connectivity patterns are lost within the sub-grid representation. Similarly, many models fail to recognise the importance of delivery processes within a catchment and the potential for disconnection of flow pathways before the reach the receiving waters (Lane et al., 2006). As a result, risks are often overestimated as not all material will be exported to a waterway. Reaney et al. (2011) suggest a move away from traditional modelling methods and instead work in an inverse fashion using models set within a minimal information requirement framework that predict risk relatively as opposed to absolutely. One example of this is the SCIMAP framework, used in this research to conceptualise and assess the relationships between hydrological connectivity and diffuse fine sediment pollution. This change in modelling approach enables a balance to be achieved between the representation of the pathways / connectivity and the source risks. By minimising process representation complexity, it will also mean that the typical problems of a lack of data (e.g. boundary conditions, knowledge of model parameters) may be reduced provided suitable calibration data are available (Reaney et al., 2009).

By using diffuse pollution risk analysis coupled to climate change projections, this research will focus on the effects that projected climate change is likely to have on fine sediment pollution in a test catchment in Northern England and consider how rural land management can be optimised to mitigate such risks. Due to the significance of the problem, mitigation of diffuse fine sediment pollution may require a comprehensive rethink of land management policy.

1.2 Research Aims & Objectives

The aim of this research is to develop a framework to "investigate the effects of land management and projected climate change on hydrological connectivity and the associated impacts on fine sediment pollution risk within the River Eden Catchment". In order to achieve this aim the following research objectives have been established.

1) ***Conceptualise and assess the interactions between projected climate change, hydrological connectivity and diffuse pollution risk within the River Eden Catchment using the SCIMAP framework.***

To target management resources on locations that pose the greatest risk of being fine sediment pollution sources under the changes in climate projected for the 21st century, an overall understanding of the current diffuse pollution dynamics of catchments is required. Areas of high connection risk, when combined with fine sediment source areas become critical source areas of pollution (Heathwaite, 1999.) Using SCIMAP, a recently developed diffuse pollution mapping framework (Lane et al., 2006) the location of these critical source areas within the study catchment will be identified as part of this first research aim.

The initial stages of the SCIMAP framework will enable the identification within the study catchment of areas with 1) the highest erodibility risk, 2) the highest erosivity risk and 3) the highest probability of becoming hydrologically connected to a receiving waterway. Areas of high connection risk, when combined with polluting areas, become critical source areas for diffuse pollution (Lane et al., 2006). To establish the location of these potentially polluting areas, each point within the catchment will then be assigned a risk value signalling the likelihood of the generation of fine sediment risk based on a landscape parameter, in this case land cover (Lane et al., 2006).

By determining the current connectivity and diffuse pollution dynamics within the study catchment this objective will outline the analysis approach necessary for subsequent research objectives.

2) To assess what a changing pattern of hydrological connectivity means for changing fine sediment diffuse pollution risk.

It is a hypothesis of this research that patterns of hydrological connectivity will alter in response to the projected changes in the intensity, duration and pattern of rainfall (Reaney et al., 2007; Bracken & Croke, 2007). Recent research predicts changes in the size of hydrologically well-connected areas and the number and frequency of connection-disconnection cycles (Reaney et al., 2009). UKCP09 projections of precipitation and temperature will be combined with CRUM3, a hydrological model which can predict changes in hydrological connectivity (Lane et al., 2009). A relationship can be created between these model predictions and the Network Index (Lane et al., 2009). This new relationship can be used within SCIMAP to assess the potential impact of changing climate on fine sediment diffuse pollution risk export.

These potential changes in the patterns of hydrological connectivity have the potential to alter the diffuse pollution dynamics of catchments across the UK (Reaney et al., 2011). If catchment scale management is going to successfully combat the problem of diffuse fine sediment then finite management resources need to be targeted on locations most at risk of being critical source areas (Heathwaite, 1999) both for the current climate and the projected future climates. This objective therefore seeks to determine possible future changes to the location of critical source areas of fine sediment pollution within the study catchment. This work will therefore ensure that future catchment management is targeted at the most at risk locations.

3) To assess how changing patterns of land management may affect fine sediment diffuse pollution risk.

Connection pathways and diffuse pollution risk are closely linked to management practices at both the catchment and the field scale (Walling & Quine, 1991). Fine sediment erosion risks associated with various crops will be investigated using a simple risk filter (Temporal Risk Integration Process, TRIP, model), developed for this thesis, which calculates a time integrated risk by analysing the temporal patterns of vegetation cover and precipitation on annual and inter-annual time scales.

The TRIP model, which has been created as part of this research, works on the basis that fine sediment erosion risk within a catchment is a product of the erosive force of the rainfall, the protection to the soil surface offered by vegetation cover and the natural erodibility of the soil. TRIP is able to capture both temporal variability in fine sediment erosion risk as a result of changing crop cover and farming practices whilst at the same time having the capacity to capture the effects that future rural land management and projected climate change may have on risk levels.

From this approach suitable land management measures that could be implemented so as to mitigate against this risk can be highlighted. The suitability of these practices will be discussed, jointly considering the economic and social implications alongside environmental factors.

1.3 Structure of Thesis

An initial discussion of the literature surrounding hydrological connectivity, fine sediment pollution and diffuse pollution modelling is in Chapter Two of this thesis. This is followed by an introduction to the test catchment used in this research in Chapter Three. In Chapter Four the current climate dynamics of the River Eden catchment are discussed and an overview of projected climate changes in the area is presented.

In order to simulate current fine sediment pollution dynamics within the test catchment two modelling frameworks are used within this research. One of these frameworks, SCIMAP, has already been created and successfully applied to several catchments across the UK (Lane et al., 2006). The second framework, TRIP, has been created in this research. The methods used to apply both of these frameworks to the study catchment are discussed in Chapter Five.

Results from the SCIMAP modelling framework and a discussion of current connectivity dynamics within the River Eden catchment are presented in Chapter Six. From this, this research uses a new approach based on the analysis of aerial photographs to calculate percentage ground cover at sites within the study catchment and the results of this are discussed in Chapter Seven. Based upon the results obtained from the processing of the aerial photographs annual fine sediment erosion risk dynamics across the River Eden catchment are simulated using the newly developed TRIP model and a discussion of the results of this can be found in Chapter Eight. This thesis concludes with an overview of the results of this research and recommendations for future research.

Chapter 2: Diffuse Fine Sediment Pollution

This chapter will explore the nature of fine sediment pollution and the processes that drive the transfer of material from source areas to receiving waterways (Figure 2.1). A short introduction into how these processes are simulated within hydrological modelling is given and the potential impacts that both climate change and land management may have on fine sediment risk are considered. This chapter concludes by discussing techniques that could potentially be used to mitigate against the risks associated with fine sediment pollution.

2.1 What is Diffuse Fine Sediment Pollution?

Diffuse pollution is defined as pollution which cannot be attributed to a single, identifiable source and hence it is often referred to as non-point source pollution (Munafo et al., 2005). It has previously been believed that point source pollution is the greatest threat to water quality in the UK (Environment Agency, 2010). However, it has recently been acknowledged that a decrease in levels of point source pollution, as a result of targeted management, has not been mirrored by improvements in water quality thus highlighting the importance of managing diffusive pollution (Heathwaite, 2002). Lane et al. (2008) suggested that diffuse pollution may not be as diffuse as once thought and that in fact diffuse pollution is composed of individual point sources that connect to form a risk. It is therefore important to highlight areas within the landscape where losses are most significant and prioritise efforts to reduce these losses before individual source areas have the opportunity to connect forming a greater risk (Heathwaite et al., 2005).

Unlike point source pollution, diffuse pollution involves processes that are small in magnitude but distributed over large spatial areas which makes identifying critical source areas difficult (Carpenter et al., 2008). In order to make waterways clean again, we need to be able to determine the source of the material, the methods by which it is mobilised and how it is transported to a receiving waterway. This source-mobilisation-pathway-delivery concept (Figure 2.2) is now widely accepted. However, there is a major gap in both our understanding of processes and available data at the scale required when it comes to the delivery component (Heathwaite et al., 2005; Brazier et al., 2005). The assessment of delivery is a problem that has been neglected in the past largely as a result of the processes of mobilisation and transport being mechanistically modelled (PEDAL Report, 2003). In-channel interactions between sediment transfer and morphological change are fundamental to catchment fine sediment dynamics along with river ecology and flood risk (Raven et al., 2010); hence, the success of future management of diffuse fine sediment pollution risk is likely to rely on improved understanding of sediment delivery dynamics.

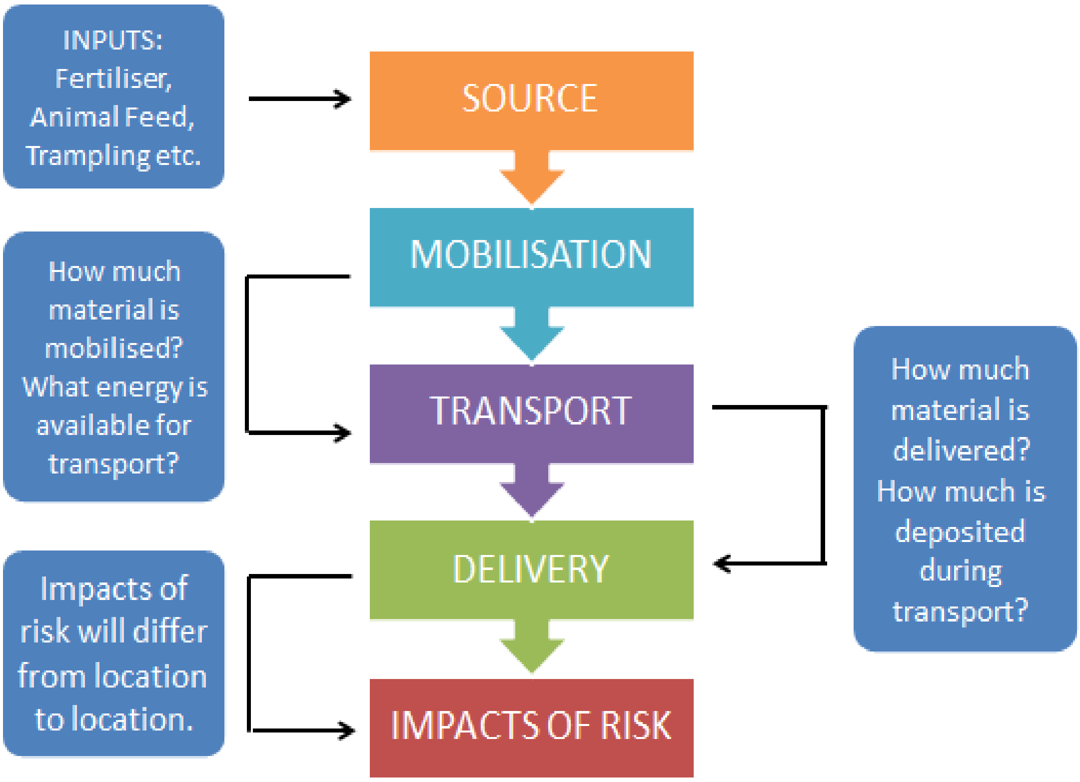


Figure 2.2: The conceptual source-mobilisation-delivery model.

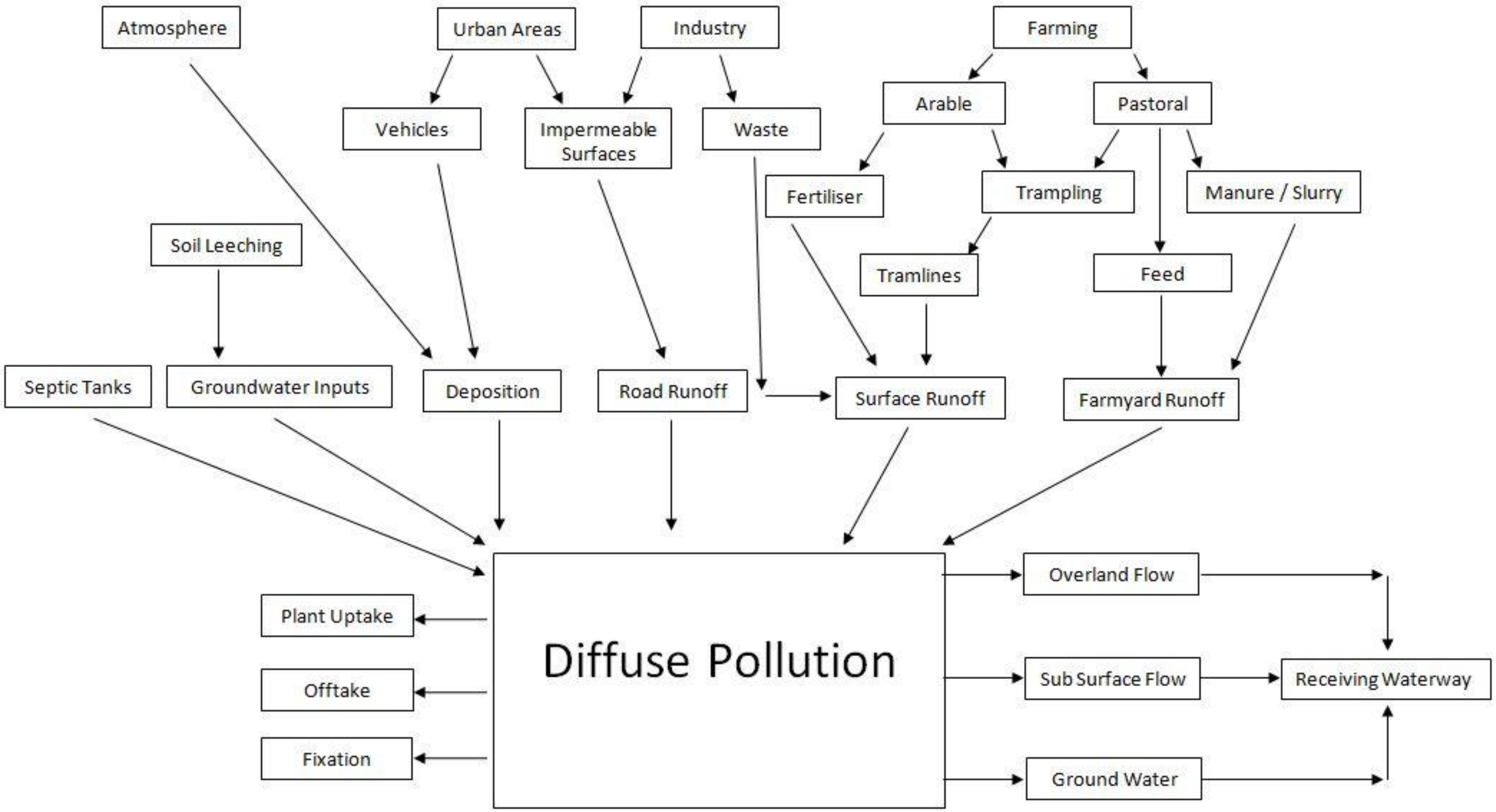


Figure 2.1: A conceptual model of diffuse pollution.

For an area to be a diffuse pollution risk, two things must be occurring. Firstly, the landscape must be producing pollution that has the potential to become a hazard; and secondly it must be possible for that risk to be exported to a waterway. Parts of a landscape generating diffuse pollution which are hydrologically well-connected are classified as critical source areas (CSAs: Reaney et al., 2009). Heathwaite et al. (2005) define a CSA as having "a significant source of pollution input and flow from land that is directly connected to receiving waterways". Due to the complexity of the hydrological system, the identification of CSAs is a difficult process and when CSAs are finally identified, the problem has often become so severe the chance of successful mitigation is slim (Heathwaite et al., 2005). One of the key objectives of this research is to identify the location of CSAs within the study catchment using modelling techniques to enable us to understand the current and potential future diffuse pollution dynamics.

2.2 The Problem of Fine Sediment Pollution

Since 1940 levels of fine sediment pollution in catchments across the UK have been increasing with some research estimating that levels may have increased by an order of magnitude or more (Walling & Quine, 1991). Such an increase may have detrimental effects on salmonids that require freshwater gravels for spawning (e.g. Soulsby et al., 2001). Suspended sediment can suffocate incubating eggs and fry and reduce the abundance of insect larvae, a common food source for many fish (Hynes, 1973). Past research has shown that high levels of suspended sediment, exceeding 200-300 mg/l, can cause fish mortality if lasting for longer than a few days (Soulsby et al., 2001).

Fine sediment pollution does not only affect the biological components of a river system. Increases in sediment input within a river channel may disrupt the channel equilibrium and lead to aggradation if the supply of material into the channel is greater than that being transported out (Raven et al., 2010). Aggradation, if allowed to continue over a prolonged period, will raise the river bed thus decreasing channel capacity and heightening the risk of overbank flooding (Knighton, 1987). In modified channels the finest particles of sediment can block the workings of dams and reservoirs which are logistically very difficult to clean (Shapiro, 2001). In addition to degrading the ecological status of many river systems, fine sediment pollution can have significant social and economic costs through polluting drinking waters and degrading fisheries (Environment Agency, 2010).

Particles of the most insidious toxic pollutants, such as heavy metals and bioaccumulative chemical compounds, can combine with particles of fine sediment and be transported, during storm events, to receiving waterways by overland flow (Younger, 1997). In the freshwater ecosystems found around the UK very few species are able to tolerate high levels of such insidious pollutants and hence the number of dead zones in waterways may increase substantially.

Combined with the increase in fine sediment, there has also been an over-application of phosphate and nitrate rich fertilisers to farmland in recent decades (Brady & Weil, 1999) and nutrient accumulation is now having adverse effects on the ecological health of waterways (Merrington et al., 2002). Both phosphorus and nitrate can quickly become limiting and have detrimental effects on flora and fauna both in the channel and on surrounding banks (Hey, 2000). Increased phosphorus (P) and Nitrogen (N) levels can lead to algal blooms and the onset of eutrophication (Ferguson et al., 1996). As well as resulting in water anoxia, these blooms may cause changes to ecosystem function thus leading to the dominance of one species (Merrington et al., 2002). Freshwater ecosystems are delicately balanced and such an increase in the number of one species could disrupt the entire food web. The transfer of phosphorus occurs predominantly alongside sediment (Maybeck, 1982) hence the issues of fine sediment pollution and nutrient pollution are often intrinsically linked.

Figure 2.3: Fine sediment pollutants entering a waterway from an incoming tributary. Source: Eden Rivers Trust (2010).

2.3 Sources of Fine Sediment Pollution

Whilst there are no global figures, it is probable that agriculture is responsible for much of the global supply of sediment to rivers (Iowa State University, 2003) and estimates suggested the benefits of dealing with agricultural pollution to be in excess of £250 million per year in 2002 (Environment Agency, 2002). Over the last 50-60 years, soil loss from agricultural land has increased dramatically with some research estimating that levels may have increased by an order of magnitude or more (Walling and Quine, 1991). A typically sustainable soil loss rate for a UK field is between 0.1 – 0.5 t ha^{-1} yr^{-1} which equates to a layer of soil between 0.01 and 0.05mm deep (Morgan, 1979). In recent decades, however, accelerated erosion has led to soil losses in some areas reaching 20 t ha^{-1} yr^{-1} and as a result of this suspended sediment yields in many waterways are likely to have doubled (Merrington et al., 2002).

Increasing rates of soil erosion have often been attributed to the post 1940 intensification of agriculture for several reasons. Firstly, large proportions of rough grassland were ploughed in the early 1950s to be reseeded to create extra grazing land (Hester & Harrison, 1996). Whilst increasing productivity, this ploughing was responsible for loosening sediment that was previously too compact to be eroded and creating tramlines for material to be transported out of the catchment. Secondly, over the same time period arable crops have spread to steeper slopes, are grown in larger fields and cultivation with heavy machinery has become standard practice (Robinson et al., 2000). Furthermore, over time crop rotation systems have declined in favour of monocultures and levels of organic matter within soils has decreased affecting aggregate stability.

Along with changes to arable farming practice significant developments within the dairy farming sector may also have contributed to the observed increase in fine sediment erosion levels. A drive to increase British food production led to a rapid increase in the number of livestock and on average, British farms went from having 4 cattle per hectare in 1950 to more than 15 cattle per hectare in 1980 (Heathwaite & Burt, 1991). The associated trampling released large amounts material for transport. Similarly, cattle degrade stream banks and thus are often responsible for forcing sediment directly into waterways (Heathwaite & Burt, 1991).

Asides from agriculture the other main source of fine sediment pollution is urban areas. Where surface runoff is not connected to water treatment works, pollutants may be deposited on impermeable surfaces before being washed into waterways. Similarly, urban construction work can lead to sudden increases in fine sediment levels as the initial stages of projects typically involve clearing the original vegetation exposing fine material to erosion.

2.4 Erosion of Material

The transfer of sediment begins when an impact detaches particles from the surface and disperses the material (Wild, 1993). Soil erosion by rainfall and overland flow (OLF) are recognised as the main pathways through which material is lost to channels (Haygarth et al., 2005). The likelihood of rainsplash being able to detach particles depends upon several factors including: rainfall intensity, surface slope, soil resistance and vegetation cover (Iowa State University, 2003). Annually, the potential for soil detachment may vary, hence an understanding of the frequency with which pollutant delivery pathways operate is therefore needed before targeted mitigation can be introduced. If climate change predictions are correct and the intensity of rainstorms changes in future decades, then this may affect the ease at which sediment can be detached and made available for transport.

All soils exert some degree of resistance to erosive powers and hence when attempting to quantify potential rates of transport it is necessary to know the local soil resistance for the area in question. This inherent resistance of soil to erosional processes is generally well recognised and has been the focus of several pieces of research (Bracken, 2010). Soil erodibility and resistance is a complex phenomenon (Bryan, 2000) and a range of factors can influence the overall susceptibility of an area of soil to erosion. Soil properties that control soil erodibility include; shear strength, soil aggregation, crusting levels and sealing (Bracken, 2010). Research by Poesen and Savat (1981) found moisture content and levels of organic matter to also be important.

Shear strength of soil is a measure of its cohesiveness (Morgan, 1979). The effect that shear stress will have on the soils response to rainfall impact is minimal, however it is a useful measure of the potential for mass movement and hence should be carefully considered in any soil erodibility calculation (Morgan, 1979).

Particle size is one of the most important controlling factors of soil erodibility. Large particles are resistant to transport, except in very large events, as a result of the greater force needed to entrain them. At the opposite end of the spectrum very fine particles are resistant to detachment as they have a strong degree of cohesiveness. The most erodible particles are therefore those of medium size, the silts and fine sands, and research has shown soils with a silt content of 40-60% to be the most erodible (Wild, 1993). It has also been suggested (e.g. Evans, 1980) that varying clay content can also affect erodibility and it has been found that soils with a restricted clay content are very susceptible to erosion. Soils with higher clay content have stronger bonds between their colloids and hence are more stable (Evans, 1980). Stony soils are less liable to erosion for two reasons: 1) the soil is protected by the stones and 2) infiltration rates are increased as water can flow into soils around the stone edges.

Crusting levels also exercise a degree of control over soil erodibility. An increase in the level of crusting within soils will often result in a marked decrease in infiltration rates and will offer soils a degree of protection from erosion by rainsplash impacts (Bracken, 2010). Crusts, formed by the compaction of soils, will interact differently with rainsplash impact depending upon their individual characteristics, and hence a blanket policy on the effects of crusting on soil erosion cannot be applied. Whereas findings by Bracken (2010) have suggested that crusting may offer soils protection from rainsplash impact other research (e.g. Zhu et al., 1997) has found that crusting of soils may lead to increased rates of Hortonian overland flow, heightening the exposure of soils to erosion (Ward & Robinson, 2000).

The erodibility of soils cannot be assessed just by considering the surface properties of soils and instead it is important to determine how the whole of the upper soil profile may affect risk (Bryan, 2000). The strength of soils particles varies considerably depending upon where in the soil column they are located as a result of the affect other particles can have on bonding soils together (Bracken, 2010). For example soils at the surface may be affected by frost action which can bind particles together or force particles apart. This type of action may however have little effect on particles beneath the surface. Soils below ploughing level are often more compact and hence less erodible (Evans, 1980).

2.5 Material Transfer and Hydrological Connectivity

Once particles have been eroded at a point, for that point to then export risk via surface flow, every point along the flow path to the channel must be capable of transferring the risk whether in a single event or as a result of a series of events (Lane et al., 2006). If this chain of connectivity is broken, the risk will be captured and disconnected (Hooke, 2003). The delivery of sediment from a hillslope to a receiving waterway is therefore critically dependent on the extent to which sediment sources can connect with a receiving river network (Brunsden & Thornes, 1979). In the past only wind erosion was thought to be responsible for the transfer of large amounts of fine sediment from sources to waterways across the UK; however, in the past

30 years there has been an increased acknowledgment of the role water plays in fine sediment erosion particularly in intensively farmed arable areas. (Evans, 1980). The extent to which material can be transported across a landscape by water is dependent upon the degree of connectivity between the sediment source and receiving waterway.

The term "connectivity" is referred to widely in hydrological literature and connectivity has become an increasingly important term in hydrology in recent years (Bracken & Croke, 2007). Despite this however, a certain degree of ambiguity is associated with the concept as a result of the different meaning it has to different researchers (Michaelides & Chappell, 2009). Thus, although the term is widely recognised as important, it lacks an agreed definition (Michaelides & Chappell, 2009). Generally speaking, hydrological connectivity can be referred to as "the transfer of water from one part of the landscape to another and the related physical movement of matter through the catchment" (Ward & Robinson, 2000). In a subsequent review of hydrological connectivity it was suggested that the incorporation of time and spatial position into any definition was important and hence connectivity should be defined as a way of "describing all the former and subsequent positions, and times, associated with the movement of water or sediment passing through a point in the landscape" (Bracken & Croke, 2007). The same review also developed the concept of hydrological connectivity to include two separate components; static and dynamic. This separates the two features that drive patterns of connection; the physical landscape and temporal conditions. Only when a comprehensive understanding of both of these components and the interactions between them has been developed can hydrological connectivity be understood (Bracken & Croke, 2007).

Catchment scale hydrological connectivity has been found to be controlled by four factors: climate, hillslope runoff potential or slope, soil characteristics and buffering (Bracken & Croke, 2007). As with many other hydrological processes climate is the primary control on hydrological connectivity. In humid regions, areas usually experience long periods of low intensity rainfall where soil water can be gradually recharged. At times when heavy rain occurs on the already saturated land both infiltration-excess overland flow and saturation-excess runoff occurs. This combination of events often leads to the transfer of large amounts of material. In semi-arid environments however, where high intensity rainfall is much more common, infiltration-excess runoff regularly occurs and hence hydrological connectivity in such areas is limited by topography as opposed to rainfall levels. Increasing rain intensity is usually matched by an increase in raindrop particle size, which reduces the soils capacity for infiltration. Larger drop sizes fill pore spaces and increase surface compaction thus increasing the likelihood of infiltration excess runoff later on during the storm event (Tackett & Pearson, 1965).

Slope characteristics act as a control on hydrological connection for several reasons. Firstly, in areas of higher slopes water flows faster over the soil surface thus meaning there is less time for soils to absorb water and hence infiltration rates are reduced (Liu & Singh, 2004). This decreases the chances of saturation excess overland flow occurring but at the same time increases the chance of infiltration excess runoff. In upland systems, with steep valley slopes, coupling between sediment sources and nearby channels tends to be high. In comparison in lowland systems, hillslopes and channels are often further apart hence source areas of pollution are often disconnected (Hooke, 2003).

Shape of slope is also a key feature in determining rates of connection. Slopes with a concave profile and a large number of surface depressions are more prone to saturation due to their low connectivity. In times of heavy rain it is therefore likely that these areas will become highly active (Talebi et al., 2008). Consideration of slope length is necessary when trying to create an accurate representation of catchment connectivity. Research has shown that runoff per unit area decreases as a function of slope (Lal, 1997), and this is particularly important regarding rates of infiltration excess runoff. In dry catchments precipitation falling on longer slopes will interact with the surface for longer periods of time and hence, although infiltration excess runoff may be produced initially, along the flow path there is a greater chance of infiltration occurring into the dry soils. It can therefore be said that distance from the channel is one of the most important factors determining rates of connected infiltration-excess overland flow (Bracken & Croke, 2007).

The degree of gullying on a hillslope is a significant controlling factor when assessing the degree of fine sediment and water being transferred from a hillslope. A gully is normally defined as a deep channel on a hillside, generally cut by running water (Kirkby & Bracken, 2009) and is usually characterised by a concentrated flow of water which is responsible for removing large amounts of upland soil. Typically gullies form in steep upland areas where overland flow or repeated mass movements bring a regular supply of water and material into the gully (Kirkby & Bracken, 2009). In upland areas gullies are often responsible for the transfer of large amounts of sediment from hillslopes to receiving waterways. Once a gully has been formed it is very difficult to effectively fill it and prevent its return, it has therefore been suggested that preventative measures are required to manage the issue gullies pose in upland areas (Kirkby & Bracken, 2009).

Antecedent conditions are an important consideration when attempting to determine rates of connection (Macrae et al., 2010). In catchments where hydrological response is driven by saturation-excess overland flow, high intensity rain falling on a dry catchment may have little impact as water will be able to infiltrate. However, a storm of similar intensity on a catchment that has already been "wetted up" is likely to produce large amounts of runoff (McGuire & McDonnell, 2010). Such patterns of hydrological response may be very sensitive to climate change particularly if precipitation intensities change.

Factors driving patterns of connectivity change according to the scale of study being investigated (Ali & Roy, 2010). For example at the hillslope scale vegetation cover and slope are likely to be main controlling factors (Wainwright & Parsons, 2002) whilst at the smaller plot scale soil moisture content and soil structure will be of fundamental importance (Sole-Benet et al., 1997). When investigating patterns of hydrological connectivity it is therefore important to pay careful consideration to scale and ensure that study methods are tailored to the scale being studied. Similarly, when attempting to introduce mitigation techniques to alter patterns of connection it is again important to ensure the impacts of these are considered at the right scale.

It has been shown that not all areas contribute equally to observed water quality problems and the delivery of diffuse pollutants to waterways (Heathwaite, 2010). Hence, for management to be targeted at the most appropriate points the identification of CSAs is critical (Heathwaite, 2010). In order to accurately identify the location of CSAs, processes must be modelled at the catchment scale whilst also capturing factors that are operating at the smaller, sub-10m scale (Lane et al., 2006). Research has shown that pollution dynamics are dependent upon small scale connectivity processes (Burt et al., 1999; Quinton & Catt, 2004) however these need to be combined with catchment scale processes to enable a holistic assessment of required management.

2.6 Modelling Diffuse Pollution

A model is an abstraction of reality that represents a complex system in the simplest way possible (Wainwright & Mulligan, 2004). Traditionally, diffuse pollution models have been introduced in an attempt to develop our understanding of processes that form the environment around us (Wainwright & Mulligan, 2004). Within both geomorphology and hydrology, *processes* are rarely observable features (Richards, 1990); however, *effects* and *outcomes* are and it is these that have traditionally been modelled. Models can thus be used to evaluate whether we are able to reproduce the effects and outcomes of environmental systems based upon our current knowledge of processes.

In recent decades a shift has been observed within environmental modelling and at present the context for modelling focuses much more on concerns relating to human-induced climate change and land degradation (Wainwright & Mulligan, 2004). These application-driven models can form the basis of management policy at local, regional and national level and are thus viewed as a highly influential tool for environmental management.

Several models have been developed which aim to identify locations within a catchment most at risk of being a source of diffuse pollution. However, the extent to which any of these models accurately capture the delivery of fine sediment and nutrients to waterways is variable (Dugdale, 2007, *unpublished PhD thesis*) as modelling the transfer and delivery of fine sediment is notoriously difficult (Heathwaite et al., 2005). Delivery processes are often over-simplified through the use of a distance-decay equation, meaning the dynamic connectivity within a catchment is overlooked (Munafo et al., 2005). Similarly, even if models are successfully developed to simulate sediment delivery, they are often at too coarser scale and so the small-scale details which drive many hydrological processes are lost (Lane et al., 2006). Some models have attempted to use resolutions of more than 1 km^2 despite typical control factors for connectivity occurring at scales of less than 0.002 km^2 (Lane et al., 2009).

The reasons for the use of such coarse resolution within many of the models available to hydrologists are twofold. Firstly, limitations in data often mean that, despite recent advances in data collection, the data needed for a model to run at a finer resolution are not available (Heathwaite, 2002). Secondly, even if required data are available at an appropriate scale the volume of information needed makes modelling a large catchment an extremely time consuming and computing intensive process and results can be difficult to verify.

Herein, this chapter introduces the various types of models available for use when studying fine sediment transfer and associated diffuse fine sediment pollution risk. This chapter concludes by assessing the relative methods of mitigating against fine sediment erosion risk.

2.6.1 Empirical Models

Empirical models describe observed behaviour between variables within a system based on observations alone and without considering process in any way (Abbott & Refsgaard, 1996).

Empirical models are usually simple and make few assumptions about the relationships between variables. This results in them having high predictive power but low explanatory depth as their outputs are limited to the specific location where data was collected (Wainwright & Mulligan, 2004).

Empirical models are usually sub-divided into two broad categories: statistical models and export coefficient models.

2.6.1.1 Statistical Models

Statistical models explore the relationships between data that are available for different variables within the catchment; for example they may investigate the relationship between land cover and soils type or total precipitation and phosphorus levels (Anderson et al., 2005). The application of statistical models is limited both spatially and temporally as a result of them relying on available data from a single study site. This means they cannot be applied to other catchments or applied to time scales that were not included in the original statistical analysis (Heathwaite, 2002). Statistical models are thus of limited use from a management perspective.

2.6.1.2 Export Coefficient Models

The earliest attempts to model diffuse pollution dynamics began in the 1970s and since then export coefficient models have been proposed as a potential management tool for predicting the export of pollutants from agricultural catchments (Johnes, 1996). Such models have low input requirements and simple structures and thus it is widely believed that they are highly suitable for catchment scale mitigation work (Worrall & Burt, 1999).

The majority of export coefficient models work on an annual time step and thus it is possible to combine them with land cover data in an attempt to determine how land use change may affect pollution dynamics within a catchment (DEFRA, 2002). Each land use is assigned an export coefficient (E_i) which expresses the rate at which N, P or fine sediment is exported from an area into the river basin.

Agriculture within the UK is highly diverse and hence a single export coefficient value for each crop is far too insensitive for the huge spatial heterogeneity in land cover that is found within a normal UK river catchment. In the early 1990s several studies attempted to derive export coefficients based on field data and expert judgement for each type of agricultural land cover found within a catchment (Johnes, 1996). Once export coefficient values had been determined, these were combined with discharge data to predict mean annual concentrations of phosphorus, nitrogen and fine sediment at a given point. This method is an accurate way of differentiating between levels of pollutant loading from year to year (Heathwaite and Johnes, 1996).

Unfortunately, the export coefficient approach to modelling is unable to distinguish between different species of phosphorus and cannot account for varying concentrations of nutrients or sediment within a yearly time step. Models also need a comprehensive amount of reanalysis to be able to simulate dissolved material transfer as opposed to particulate transfer and vice versa (Lindenschmidt et al., 2004). On the other hand, the simple structure of the models and the fact they can be incorporated into readily available spreadsheet software, makes them an extremely useful management tool (Wainwright & Mulligan, 2004). Furthermore, several of the models have been applied at a variety of scales, meaning they can be tested within an uncertainty framework in order to determine their overall accuracy (Murdoch et al., 2005).

One such example of an export coefficient model is the Phosphorus Index (P-Index), developed in the United States of America (Heathwaite et al., 2003). The purpose of the Phosphorus Index is to assess the various landforms and management practices for potential risk of phosphorus movement to water bodies (NRCS, 2011). The ranking of Phosphorus Index identifies sites where the risk of phosphorus movement is relatively higher than at other sites (Heathwaite et al., 2003). It also can be used to identify the critical parameters of soil, topography, and management that most influence the movement of material (NRCS, 2011). Using these parameters, the Index can help in the selection of management alternatives that could minimise the risks associated with high levels of phosphorus transport. Currently, however, local conditions cannot be included in the P-Index and hence it does not capture individual, unique processes operating at different locations (NRCS, 2011).

2.6.2 Conceptual Models

Multi-layered models, based on the combination of export coefficient models and statistical models, are the most modern conceptual models. Two of the most developed current multi-layered models are the Phosphorus Indicators Tool (PIT) (Heathwaite et al., 2003) and the Phosphorus and Sediment Yield Characterisation in Catchments model (PSYCHIC; Davison et al., 2008).

2.6.2.1 PIT Model

The PIT model is a GIS-based, distributed, catchment scale phosphorus model that operates on an average annual time step (Heathwaite et al., 2003). PIT is divided into three modelling layers or stages. The first layer estimates the amount of available phosphorus from manure and fertiliser application based on land use and soil type. The potential loss of phosphorus from manure is calculated by multiplying the total numbers of each animal with a specific coefficient which has been calculated for each animal (Heathwaite et al., 2003).

Once potential losses have been established, the model then goes on to determine how much of this phosphorus can be transferred away from its source. Within this second layer of PIT four flow pathways are considered; subsurface flow, surface flow, vertical flow and topsoil lateral flow (Liu et al., 2005). Properties controlling the transfer of P include soil structure, climate and slope.

Now that PIT has estimated rates of phosphorus transfer, it is necessary for the model to try and simulate delivery pathways which would transport phosphorus from source areas to nearby waterways (Liu et al., 2005). The key controls for P delivery considered in PIT are: the degree of sediment retention within fields and ditches; the extent of artificial drainage; and, the distribution of routes of high connectivity which may increase the efficiency of the transport of sediment and associated P to watercourses. Included in layer 3 are features such as farm tracks and roads that may facilitate the rapid delivery of transferred P to water (Heathwaite et al., 2003). The model accounts for three pathways via which diffuse P is known to be delivered to watercourses: surface flow, drain flow and percolation to groundwater (Liu et al., 2005). Using the delivery estimates from layer three of the framework and combining them with indexes of connectivity between land and surface water, a risk index is created highlighting areas within the catchment that are at particular risk of being critical source areas of phosphorus pollution.

A simplified version of the PIT model has been used in the River Basin Characterisation assessment as part of the Water Framework Directive to help identify the pressures on surface waters from diffuse P sources (Environment Agency, 2010). Similarly, PIT has also been recommended for inclusion in the Environment Agency toolkit: 'Environmental Effects of Agriculture and Land Use'.

2.6.2.2 PSYCHIC Model

Recent changes in environmental legislation have made diffuse pollution a priority concern and there is an urgent need for an appropriate risk assessment and decision making tool to be developed (Heathwaite, 2002; Lane et al., 2006). To facilitate the cost-effective targeting of a range of land management options to mitigate fine sediment and phosphorus loss, a prototype catchment-based decision support tool called PSYCHIC was developed and applied to a number of catchments in the UK with well-known siltation and eutrophication problems.

Nationally-available data-sets on climate, soil types and their characteristics, slope, river drainage density, agricultural census information (land use and livestock numbers), crop and manure phosphorus inputs and human population at the 1 km^2 scale are needed to successfully run PSYCHIC (Davison et al., 2008). The model quantifies the loads of suspended sediment, particulate and dissolved phosphorus in runoff from agricultural land along different surface and sub-surface hydrological pathways and assesses their contribution relative to point sources (Stromqvist et al., 2008). The model is based on the concept of the source-mobilisation-delivery pathway (Haygarth et al., 1999).

PSYCHIC adopts a process-based modelling approach to ensure sensitivity to land management practices that have a big influence on the mobilisation and delivery of fine sediment and phosphorus. The model takes account of climate, landscape and land management factors, and utilises current knowledge on the processes that drive sediment transport to predict the spatial and temporal distribution of flow and pollutant loads on a monthly time step at both catchment and field scales (Stromqvist et al., 2008).

PSYCHIC simulates several key processes by which both fine sediment and phosphorus are mobilised at the plot scale and transported via surface pathways and drains to watercourses. The fraction of the mobilised material delivered to rivers is determined by individual connectivity coefficients for each of the main transport pathways (Stromqvist et al., 2008). PSYCHIC then assesses the impact of land management on rates of pollutant transport (Davidson et al., 2008). This enables the end-user to identify and prioritise where control measures need to be adopted, and to help justify their adoption in line with policy mechanisms (e.g. catchment sensitive farming) to reduce diffuse agricultural pollution. PSYCHIC is both a catchment and farm management tool.

A number of factors are likely to constrain the performance of PSYCHIC when attempting to predict diffuse pollution dynamics within catchments. By choosing to use climate data as an input PSYCHICs ability to capture shorter-term behaviour is likely to be constrained (Stromqvist et al., 2008). Furthermore, there is a large degree of inherent uncertainty associated with taking short-term measurements of phosphorus and fine sediment loads in this fashion. Potential errors include issues with sampling frequency, load estimations, the impact of extreme events and problems with determining base flow index values (Walling et al., 2002). Since the input data are statistical in nature (1 km^2), uncertainty will increase as the area to which the model is applied decreases. PSYCHIC is therefore currently best suited to characterising longer-term catchment response; however, allowing the model to respond to weather (rather than climate) is a priority for future model development.

Furthermore, PSYCHIC does not represent channel-bed sediment storage or consider the effects of pollutant remobilisation and the significant role this can play in the transfer of phosphorus (Jarvie et al., 2005). Similarly, the model does not incorporate the effects of bank erosion which can significantly increase levels of fine sediment and associated pollutants within waterways. It has been planned that future work will attempt to incorporate these in-channel processes into the model framework to enable a more accurate prediction of load to be obtained (Stromqvist et al., 2008).

Because of the large number factors and processes that are included within multi-layered models often over 100 coefficients are needed for the model to run (Liu et al., 2005). This, combined with a lack of data for certain components, may limit the overall success of multi-layered models when applied to many catchments.

2.6.3 Physically Based Models

Physical based models attempt to represent all processes operating within the source-mobilisation-delivery model of diffuse pollution as comprehensively and accurately as possible (Kreuger et al., 2007). Such models should be derived from established physical principles and produce results that are consistent with field observations (Beven & Feyen, 2002); however, it is often difficult for models to achieve both of these goals and most end up doing one but not the other (Wainwright & Mulligan, 2004). Most physical models have to include some degree of empirical generalisation to fill the gaps where processes are just not well enough understood to be reliably modelled (Abbott & Refsgaard, 1996).

Physical based models tend to have good explanatory depth due to the large number of processes that are considered within them and the interactions between processes that are accounted for. However, they are often characterised by low predictive power and they often do not agree with field based observations (Wainwright & Mulligan, 2004). This demonstrates a poor understanding of the processes and physics operating within the system (Beven, 1989) and as a result models are often calibrated against field observations. Physically based transfer models are usually designed for specific purposes and are set to be applied at a certain scale (Kreuger et al., 2007). As a result, it is extremely difficult to accurately apply a physically based transfer model to a problem or location that it was not initially designed for.

Many of these models attempt to include several of the complex processes controlling fine sediment mobilisation and transport including soil evapotranspiration, infiltration rates and vegetation properties (Lewis & McGechan, 2002). Unfortunately, many of these properties are extremely difficult to accurately represent within a modelling framework and the reliability of many models such as DAYCENT and MACRO has been called into question as a result (Lewis & McGechan, 2002).

There are numerous examples of physically-based diffuse models that have been introduced both within the UK and further afield (e.g. SHETRAN, SHE, MACRO). SHETRAN is a physically based, spatially distributed modelling system which allows the spatial variation of the hydrological features within a catchment to be represented numerically (Ewen et al., 2000). Based on SHE, SHETRAN can simulate both vertical and lateral flows along with sediment transport and contaminant migration at a range of spatial scales from a single field plot to a large river basin (Nasr et al., 2006). The model requires minimal calibration as inputs are based on physical properties; however, the data requirements of the model are large and much of the required information is not directly available. Furthermore, the scale is too coarse (1km) so small-scale processes are excluded from analysis.

Beven (1989) discusses the issues of data availability related to physically based models, especially those which focus on pollutant transfer. He concludes that, as the models are data and parameter dependent, there is often a lack of suitable data to run the model and thus they are inappropriate to use in studies attempting to identify critical source areas of diffuse pollution.

2.6.4 The Relative Risk Based Approach

The changing focus of hydrological modelling stems largely from research into the proposed effects of climate change. Changes in precipitation and temperature have been predicted across the UK and this may result in alterations to patterns of connectivity and CSA location (Reaney et al., 2011). A large amount of research has shown that sediment dynamics are dependent upon small-scale processes (Burt et al., 1999; Quinton & Catt, 2004); however, before mitigation options can be considered these need to be combined with catchment-scale processes to enable a holistic assessment of required management to be carried out.

The most developed approach to identifying and modelling the location of CSAs is the use of a relative risk based approach (Lane et al., 2006). Risk based analysis of diffuse pollution risk is a well-established idea and such an approach has already proved to be very effective in diffuse pollution modelling (e.g. Johnes, 1996; Munafo et al., 2005). The primary assumption here is that the amount of material that is exported from a land unit can be traced to the properties of that land unit (e.g. physical attributes like slope and soil type) and how it is managed (e.g. levels of fertiliser application).

Early use of this type of model focused on determining the rate of export of fine sediment and nutrients from areas of particular land cover, but gave little attention to the delivery processes of this material from its source to a receiving waterway. It has now become recognised however that the incorporation of the delivery component into a risk-based analysis is important if a modelling framework is going to successfully identify areas of high risk across spatial scales (Reaney et al., 2011).

One example of a new relative risk based approach is the SCIMAP model, which assesses the relative risk of diffuse fine sediment pollution in catchments within a probabilistic framework. Within such a framework sources of risk are thought to be distributed across a river catchment. Landscape attributes may combine with human attributes to make particular areas within the catchment a greater risk than others (Reaney et al., 2011). The basis of the analysis is the joint consideration of the probability of a unit of land producing a risk and then of that risk reaching the drainage network (Lane et al., 2006). Hydrologically well-connected and risky land uses should be the prime focus of management activities, and hence the result of SCIMAP is a method for determining where finite management resources should be concentrated in order to achieve optimum environmental protection (Lane et al., 2006). The theory behind the

SCIMAP based framework can broadly be split into three categories; risk generation, risk delivery and risk loading.

2.6.4.1 SCIMAP Risk Generation

The first step in the SCIMAP framework is to evaluate the risk that contaminants may be generated and exported from a given point in the landscape (Heathwaite et al., 2005). In order to calculate the likelihood of an area producing a diffuse pollution risk, each point within the catchment is assigned an erosion risk generation value for the pollutant in question (e.g. fine sediment) based upon several topographic controls including slope and upslope contributing area. This initial calculation of erosion risk is then combined with a land cover weighting value to get an overall risk generation value. Each area of a landscape is categorised into one of six land cover groupings. Each group is assigned a risk value, parameterised by expert judgement (Lane et al., 2006), which are scaled between 0 (low risk of erosion) and 1 (high risk of erosion). The land cover weightings used within SCIMAP in this research are shown in Table 2.1.

Land Cover	Assigned Risk Value
Arable	1.0
Intensively Managed Grassland	0.30
Extensively Managed Grassland	0.15
Peat and Bog	0.10
Heath and Bracken	0.05
Woodland	0.05

Table 2.1: Default Land Cover Risk Weightings Used In The SCIMAP Framework

Once overall risk values for each point in a landscape have been determined, these are combined with the risk of there being sufficient energy to erode material to establish the likelihood of the transport of material occurring. SCIMAP assumes energy available for erosion is a function of the upslope contributing area of any part of the landscape. By combining stream power, as calculated from the upslope contributing area, and topography, an estimate of the potential energy available to transport material is obtained.

$$\boldsymbol{Pi^g = Pi^h \,.\, Pi^e}$$

Equation 2.1: SCIMAP Risk Generation Parameter Equation

Where Pi^g is the risk generation parameter, Pi^h is the risk of there being sufficient energy to erode the material and Pi^e is the risk of that material actually being erodible.

2.6.4.2 SCIMAP Risk Delivery & Loading

Once each point within the catchment has a risk weighting attached, it is necessary to determine each point's delivery index, in other words the chance of the risk from a location becoming connected to a waterway. The risk of contaminant delivery is determined based upon catchment topography and hydrological connectivity along flow paths (Lane et al., 2004). Connectivity is conceptualised as a series of points, each one of which can be envisaged as being in either a connected or disconnected state at any one time (Reaney et al., 2007; Lane et al., 2009). The degree of connectivity between the points will determine whether a risk is exported from its source to the channel or whether the risk will become disconnected and captured within the landscape (Reaney et al., 2011).

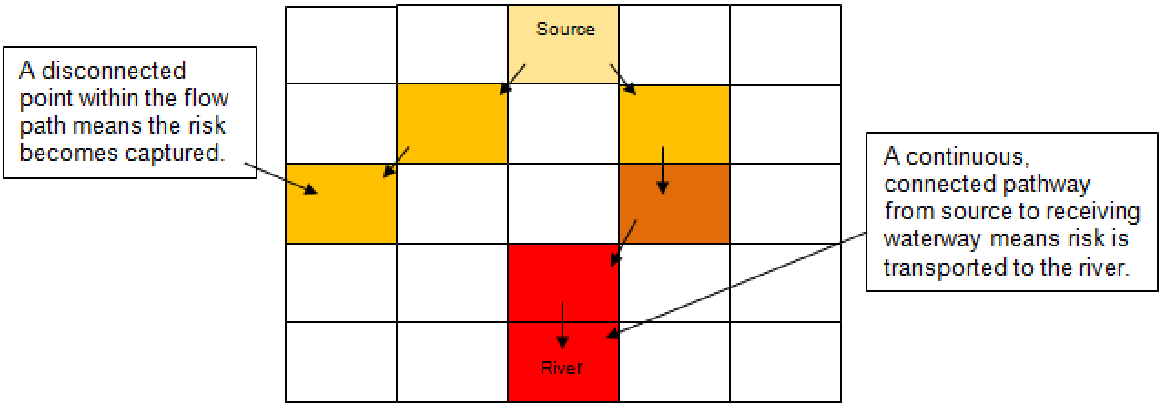


Figure 2.4: Connected and Unconnected Flow Paths

After initial calculations to determine upslope contributing areas, flow paths are then calculated using the deterministic flow routing algorithm (Wilson et al., 2008). This uses a nearest neighbour approach to define the relationship between a central cell and its eight contiguous neighbours. The algorithm works on the assumption that all water will flow from the central cell, down the steepest gradient into the next cell (Wainwright & Mulligan, 2004).

In order to calculate the delivery index, SCIMAP first calculates the topographic wetness index (equation 2.2; Kirkby, 1975, Beven & Kirkby, 1979) which is a measure of propensity to saturation and overland flow generation.

$$\mathbf{k} = \ln\left({}^{a}/_{\tan\beta}\right)$$

Equation 2.2: Topographic Wetness Index

Where *k* is the topographic wetness index, *β* represents the local topographic slope and *a* represents the rainfall weighted upslope contributing area.

Not all areas of saturation will connect or deliver material to the channel during a storm event, or indeed multiple storm events (Lane et al., 2004), and the connectivity of many areas to waterways depends upon the connection of a particular critical cell along the flow path. For a point in the landscape to export risk in surface flow, every other point along the flow path to the channel must be capable of transporting the risk. If an area downslope is not fully saturated, then water will infiltrate and the material it is transporting will be deposited. Research by Walling et al. (2002) found that only very small amounts of the total sediment mobilised is likely to reach the river channel. It is therefore assumed that the less likely a continuous hydrological flow path is, the greater the likelihood of onslope deposition occurring. To account for this onslope deposition and disruption to connection, SCIMAP uses a modified version of the topographic index, the network index. The network index is based upon the lowest value of the topographic index along a given flow path from the point of interest in the drainage network (Lane et al., 2004) and thus represents the average relative risk of connection at a particular location through time. During rainfall events, a greater number of points within the catchment will become connected and thus an increased amount of overland flow will be produced. Points within the landscape with a higher network index value are therefore likely to be connected for a longer period of time than those areas with a lower network index (Reaney et al., 2009). Locations with a low value of the network index are assumed to have a particularly dry cell along their flow path (Lane et al., 2004*)*, and hence are less likely to hydrologically connect, whether through shallow subsurface flow or surface flow. They do not connect as regularly and thus pose less of a pollution risk.

The final stage of the SCIMAP framework is to accumulate the connected pollutant sources along flow paths and integrate them through to the drainage network to establish the delivery "loading" at each point. The loading risk, or risk concentration, is deemed to be the sum of all upstream contributing risks and by dividing it by the upslope contributing area dilution processes can be accounted for (Lane et al., 2006).

After reviewing the available types of diffuse pollution model it can be seen that the relative risk based method is the most appropriate way of identifying areas within a catchment most at risk of being source areas of diffuse fine sediment pollution. The use of a relative-risk approach ensures that data requirements are minimal and in the framework chosen for this research all required data are readily available. Furthermore, the risk-based framework can effectively capture the potential effects that changing land management practices and predicted future climate changes may have on patterns of hydrological connectivity and associated risks of diffuse pollution. The structure of the SCIMAP framework means it is suitable to be combined with the TRIP framework developed in this research to assess fine sediment erosion risks over an annual period.

As mentioned previously when working with hydrological models it is important to ensure that the model used is appropriate to the scale of the study. Processes controlling patterns of diffuse fine sediment pollution and the location of critical source areas operate on very small scales (Lane et al., 2006), hence to accurately predict the location of CSAs the selected model needs to be able to capture these processes. A relative-risk approach, such as SCIMAP, which can work across scales from catchment-scale to field-scale, is therefore most appropriate. Furthermore, when attempting to implement management strategies to combat the risks associated with diffuse fine sediment pollution, projects are introduced at varying scales depending on the institutions involved and funding available. Research therefore needs to be able to identify “risky” locations at catchment, farm, field and sub-field scale to enable targeted management to be introduced. An empirical model, where results are limited to the specific location where data was collected (Wainwright & Mulligan, 2004) would therefore be inappropriate.

When attempting to identify areas that have the potential to become critical source areas of diffuse pollution it is important to recognise that a lumped model, which considers a catchment as one whole entity, would not be suitable due to its failure to capture the unique pathways of hydrological connectivity across the catchment. This is another reason why the more distributed relative-risk based approach offered by the SCIMAP framework is used in this research.

2.7 Mitigating Against Fine Sediment Risk

Once CSAs have been identified, appropriate mitigation methods need to be carefully selected. Deasy et al. (2009) showed on-farm management can significantly reduce the loss of fine sediment at the hillslope, farm and field scale by targeting nutrient availability (source methods), the timing of practices (timing methods) and nutrient delivery (transport methods). Possible mitigation suggestions include early crop sowing, restricting livestock access, establishing buffer zones and reducing ploughing (Cherry et al., 2008).

Currently, 56% of arable land in the UK is intensively ploughed (DEFRA, 2010) which detaches soil particles making them susceptible to entrainment (Hester & Harrison, 1996). If farms were to revert to a policy of reduced tillage, where at least 30% of crop residue is left on the field surface, then the loss potential would be significantly reduced (Boardman, 1992). Unfortunately, at present only 40% of farms use the reduced tillage method (DEFRA, 2010).

Furthermore, changing ploughing direction could also be a successful way of reducing loss (Deasy et al., 2009). Ploughing across contours instead of down them would stop the creation of furrows which efficiently channel material into waterways (Silgram et al., 2006). Similarly, it has been suggested that ceasing to create tramlines may assist in reducing sediment erosion (Walling & Quine, 1991). These pathways, usually formed by the wheels of heavy machinery, compact soil and the furrows act as pathways for OLF (Burt & Slattery, 2005). Previous studies have shown that the presence of tramlines can increase runoff during a storm from 0.4mm to 8.4mm (Silgrim et al., 2006) and sediment loads can increase from 21 kg ha^{-1} to 400kg ha^{-1}. Earlier research by Withers and Jarvis (1998) acknowledged that stopping the creation of tramlines is not viable and suggested a more realistic option may be to block tramlines at the end of the growing season.

Figure 2.5: Tramlines in a field of young wheat. Tramlines can act as a pathway for overland flow transporting fine sediment off arable land. This image also shows the disconnection of a flow pathway at the end of the tramline. Source: Eden Rivers Trust (2010).

In areas where land is predominantly used as pasture, farmers have been encouraged to keep cattle away from waterways. By erecting fences, the risk of bank erosion and the chance of waste ending up in the waterway are minimised. In the River Eden Catchment, where 76% of land is agricultural, this could be an excellent way of reducing fine sediment pollution.

If the risk of fine sediment pollution cannot be controlled by in-field measures, then edge of field measures, those that halt material delivery, will need to be applied (Deasy et al., 2009).

One of the most widely used approaches is the introduction of buffer zones. A buffer allows runoff to be attenuated before reaching waterways, via infiltration, adsorption and deposition (Cherry et al., 2008) and can be planted to grass or trees which, as well as disconnecting OLF, provide habitats (Norris, 1993). Research shows buffer zones are instrumental in water quality improvements for both surface and subsurface flow (Gilliam, 1994) as a result of the dual effect that a buffer can have; solid material can be physically retained whilst vegetation removes nutrients deeper within soils. Although proven to be effective at removing sediment from surface flow, whether buffer strips can effectively reduce concentrations in sub surface flow is debatable (Blackwell et al., 1999). Water needs to be in the buffer long enough for the deposition of material to occur, but during storm flows this may not be the case (Harris & Forster, 1997). Furthermore, if the water table is very low, the subsurface flow and the buffer zone may not interact at all. The implementation of a buffer zone is likely to be most effective in a higher order tributary where a floodplain has been able to form; in lower order tributaries steep slopes may directly border the channel thus meaning there is little opportunity for fine sediment to be deposited before reaching the waterway. Despite the conflicting evidence over the success of buffer strips, what is clear is that continuous active management of buffer zones is required to ensure they remain effective over long periods of time (Burt, 2001).

The most researched alternative to buffer zones is the use of a settling or retention basin. This is a basin, linked to areas of surface runoff, where water is routed to (Laws, 2000). As runoff flows into the basin, its speed is reduced, thus leading to the deposition of suspended sediment (Laws, 2000). In practice, settling basins may have little impact on fine sediment pollution as, as with buffer zones, the success of the basin is governed by retention times. It has been shown that a residence time of anything less than two hours fails to remove more than negligible amounts of material (Shepherd & Chambers, 2007) and as retention time during a storm is rarely more than an hour settling basins are often ineffective.

Instead of focusing on *managing* fine sediment diffuse pollution recent plans have concentrated more on *mitigating against* the issue. Work by Merrington et al. (2002) suggests that farmers could work at controlling the temporal patterns dictating levels of soil erosion and OLF. By using a simple *risk = vegetation ground cover x rainfall* risk filter the impact of altering the timing of planting and harvesting of crops can be assessed. As part of this research a filter based on this simple formula has been created. The Temporal Risk Integrated Platform (TRIP) has been designed to be a user friendly risk generator which can capture fine sediment erosion risk at a given point within a catchment. By working on the basis that fine sediment erosion risk for a location within a catchment is a product of the erosive force of the rainfall, the protection of the soil surface by vegetation cover and the erodibility of the soil, TRIP is able to simulate the temporal issues associated with fine sediment pollution.

TRIP is particularly useful for showing the impact of soils being left bare for long periods of the year, particularly over the winter months, when heavier rainfall is likely to result in increased chances of erosion. In areas where large proportions of the ground are covered by crops for most of the year less rainfall will hit the soil whilst travelling at terminal velocity (Ward and Robinson, 2000). Hence, the erosive power of the rainfall is less and thus the risk of erosion of top soil is reduced. Where fields are left bare for large parts of the year, rainfall will not be intercepted by vegetation and any particles hitting the soil will have high erosive powers.

If levels of precipitation and/or rainfall intensity are altered as a result of projected global climate change, the associated variations in runoff dynamics could dramatically change fine sediment erosion risk. These changes in risk can be accurately modelled using the newly created TRIP framework by combining the analysis of aerial photographs to assess percentage ground cover with readily available monthly rainfall data. From this possible mitigation options to minimise erosion risk, for each individual location within a catchment can be suggested.

2.8 The Cost of Fine Sediment Pollution

The most recent estimates suggest the benefits of dealing with diffuse agricultural pollution to be in excess of £250 million per year (Environment Agency, 2002). Currently, the UK spends £120 million removing pesticides from drinking water and a further £55 million removing fine sediment (DEFRA, 2002). Monitoring water supplies and providing advice and guidance to farmers on the management of pesticides and nutrients costs a further £11 million (DEFRA, 2002). Such a huge financial burden has meant that minimising the risk of diffuse pollution has become a key priority for those involved in environmental policy and management.

2.9 Conclusions

It can be seen that successful management of fine sediment pollution requires detailed study into the current pollution dynamics of catchments along with an investigation into the potential effects that climate change and future land management may have on risks. Fine sediment pollution is at the centre of the Water Framework Directive (WFD) the Habitats Directive and Urban Waste Water Treatment Directive. The forthcoming deadline that all water bodies must achieve good ecological status by 2015 has propelled the problem of increasing levels of fine sediment entering waterways to the top of many research agendas. In order for suitable mitigation methods to be deployed, there is a refreshed need for the quantification of changing sediment risks to assess what management methods are the most ecologically and economically effective. Further legislative commitments such as the Nitrate Directive (91/676/EEC, OJ L 375, 31.12.1991) and the Habitats and Birds Directives (92/43/EEC, OJ L 206, 22.7.1992) only serve to increase the need further.

Chapter 3: The River Eden Catchment

The River Eden Catchment was chosen as the main test application site for this research for several reasons. Firstly, the topographic characteristics of the catchment make it well suited to the SCIMAP modelling framework. Secondly, the large spatial area of the catchment combined with the variety of rural land uses makes it an ideal location to pinpoint critical source areas of diffuse pollution and examine possible management strategies for the future. Thirdly, the River Eden catchment is one of three selected catchments used in the DEFRA Demonstration Test Catchment project and so it is hoped that, once complete, this research can be implemented in the other catchments in the initiative. Lastly, predicted climate change has the potential to dramatically change the hydrological dynamics of the catchment hence the impacts of projected climate change can be directly assessed.

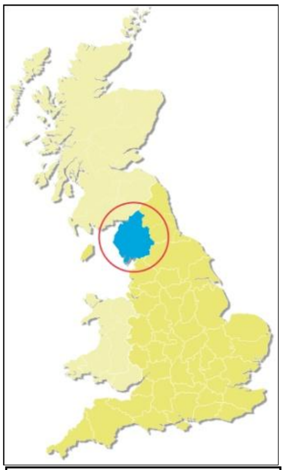
3.1 Catchment Overview

The River Eden flows for 145 kilometres from its beginnings high in the Howgill Fells through the Eden valley before entering the Irish Sea on the Solway Firth at Rockcliffe. The total area of the catchment is 2288km^2 and represents a third of the land area of the county of Cumbria. Major tributaries include the Rivers Irthing, Caldew, Eamont and Lowther.

The River Eden catchment includes several natural lakes, tarns and reservoirs. It is fringed to the south by the Yorkshire Dales National Park, to the east by the North Pennines Area of Outstanding Natural Beauty, and to the west by the fells of the Lake District National Park. The total population of the catchment, as recorded in the 2001 census, was found to be 145,000; the majority of whom live in or around the towns of Carlisle and Penrith. Population density is low with about 0.2 persons per hectare. Figure 3.1 shows the boundaries of the Eden catchment, the main water courses and the location of major urban areas.

The entire length of the River Eden has been designated a Site of Special Scientific Interest (SSSI) as it represents a rare, pristine example of a semi-natural river system from source to sea . The catchment is also home to 16 Special Areas of Conservation (SACs), 2 Special Protection Areas (SPAs), 2 Ramsar Sites, 2 designated Areas of Outstanding Natural Beauty (AONBs) and 11 National Nature Reserves (NNRs) as well as having 454km^2 of the Lake District National Park within its boundaries (Environment Agency, 2006). Several sites are internationally important habitats for species including salmon, bullhead, otter, white-clawed crayfish and water crowfoot (Environment Agency, 2006). The catchment is also a designated UNESCO HELP Basin. The project is attempting to deliver social, economic and environmental benefits to stakeholders through research towards the sustainable and appropriate use of water.

The River Eden catchment was designated as a Catchment Sensitive Farming priority catchment in 2006. The project is a joint initiative between the Environment Agency and Natural England, funded by Defra and the Rural Development Programme for England, which delivers practical solutions and targeted support to enable farmers and land managers to take voluntary action to reduce diffuse water pollution from agriculture to protect water bodies and the environment (Natural England, 2011). Target areas within the catchment include Dacre Beck, the River Lyvennet and the main channel of the River Eamont.



Location of Cumbria within the UK.

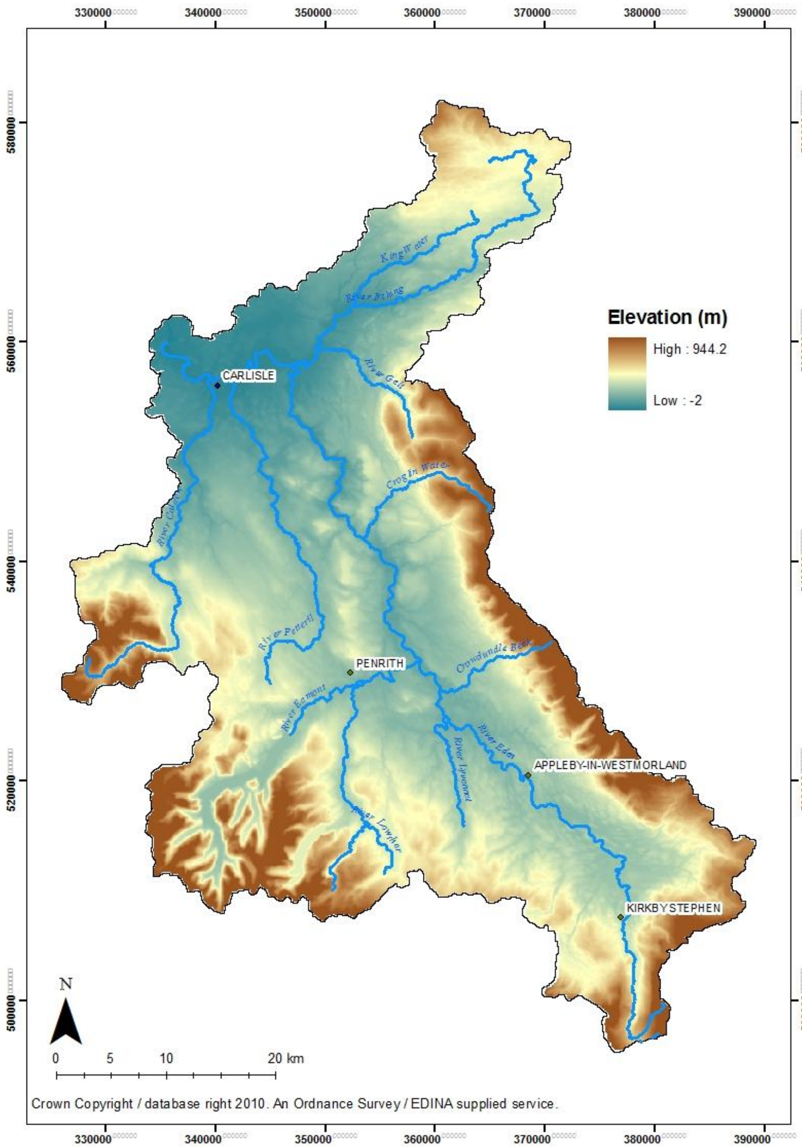


Figure 3.1: Digital Elevation Model of the River Eden Catchment.

Figure 3.2: The River Eden at Temple Sowerby. Source: Eden Rivers Trust, 2010

Figure 3.3: The upper reaches of the River Eden in the Mallerstang Valley. Source: Eden Rivers Trust, 2010.

3.2 Institutional Management Framework

Managing a catchment as vast as the River Eden is a complex and often difficult task that requires cooperation between various organisations. The Department for the Environment, Food and Rural Affairs (Defra) is responsible for policy and regulations on the environment and these regulations are managed on a day to day basis by the Environment Agency.

Other groups who have a key role in the management of the environment within the River Eden catchment include local and district authority groups, water companies and several voluntary organisations. A description of the roles these organisations play is given in Table 3.1.

Institution	Department or Agency	Description
Central Government	Defra	Government department responsible for policy and regulations on the environment, food and rural affairs.
Statutory Public Body	Environment Agency	Executive non-departmental public body. Principal aims are to protect and improve the environment, and to promote sustainable development
	Natural England	Executive non-departmental public body. Advises government. Provides practical advice on how best to safeguard England's natural environment for the benefit of wildlife and people.
District Authority	Cumbria County Council	An elected authority which provides a range of services for local people, visitors and the business community.
	Lake District National Park Authority	Encourages people to enjoy and understand the area. Assists in the management of long-term development and planning. Attempts to balance the needs of local people with the needs of the local environment. Funded by Defra.
Local Authority	Eden District Council	Local council with responsibility for services including; planning, environmental health and tourism.
Water Company	United Utilities	United Utilities owns and operates the water network in north west England. Responsible for managing the catchment to collect and store water in reservoirs before treating it and delivering it to homes and industry across the region.
National Non-Governmental Organisations	RSPB	Charity group who work to protect birds and the environments they live in across the entire catchment area.
	Wildlife Trusts	Manage nature reserves and advise farmers on land management.
Local Organisations	Eden Rivers Trust	Aims to conserve, protect and improve the environment of the River Eden, its tributaries and the wildlife associated with them.

Table 3.1: Environmental groups involved in the management of the River Eden Catchment.

3.3 Topography

As shown in Figure 3.1 elevation varies considerably across the River Eden catchment. Upstream of Kirkby Stephen, the Eden channel and surrounding area is characterised by steep slopes and surrounding fells. These promote rapid runoff during times of heavy rain where water is quickly routed over the surface into nearby waterways before reaching the main river channel further downstream. The fells at the top of the catchment reach as high as 690m but these quickly decrease in size and by Kirkby Stephen the highest fells are, on average, only 160m high. At this point the steep tributaries disappear and the valley begins to widen out. Downstream from Kirkby Stephen the river continues to fall at a fairly constant rate as it winds through Appleby-in-Westmorland and Penrith towards its exit onto the Solway Firth. Throughout this part of its course the river forms wide floodplains and washlands which are important in providing storage capacity during high flow events (Environment Agency, 2006).

3.4 Geology

The Vale of Eden lies on the northeast of the Lake District massive and in the upstream areas of the catchment the majority of the bedrock is composed of Millstone Grit. As the main channel flows north-west, significant differences in the geology to the east and west can be seen. The areas to the east of the main channel become dominated by Carboniferous limestones whereas areas to the west have larger amounts of sandstone and mudstone (Allen et al., 2010).

Geology plays a key role in the response of a catchment to rainfall events. In the south of the catchment steep tributaries, underlain by impermeable sandstone, promote rapid surface runoff causing river levels to peak quickly after rain events. To the west of the catchment metamorphic rocks from the Borrowdale Volcanic Group and Skiddaw Slates dominate the bedrock geology. Usually, this type of geology would be associated with rapid overland flow; however, large lakes in the area (Haweswater & Ullswater) dampen the potential for runoff (Allen et al., 2010).

Glacial drift covers over 75% of the bedrock geology in the catchment with thick deposits of boulder clay and alluvium having been left by retreating ice (Allen et al., 2010). Much of the till has been moulded into hummocky drumlin fields and evidence of these dominates the quaternary cover. Along the main channel of the River Eden extensive deposits of sand and gravel left during glacial outwash events can be observed.

Most of the catchment has some drift cover, although the high fells of the Lake District have little or no cover, promoting faster runoff. The thickest deposits are found in the central areas of the Eden valley, overlying the Penrith Sandstones and Eden Shales (Allen et al., 2010). These clays may be as thick as 20m locally and can insulate the underlying geology from the surface, limiting groundwater recharge and restricting hydraulic contact between aquifers and surface waters (Environment Agency, 2006). Alluvium cover is extensive in the areas around Carlisle and north of Penrith and some areas of peat can be found in the north east of the catchment, but generally not at elevations higher than 200-300m (Environment Agency, 2006). Peat may be able to retain moisture from rainfall hence slowing down the release of surface runoff after a rain event. However, if waterlogged, it can soon become a semi-impervious bed and rapid run-off will begin to occur.

Groundwater contributions to river flow within the River Eden catchment are low and account for less than 10% of flow (Butcher et al., 2003). Across the catchment the baseflow index, a measure of the proportion of the river runoff derived from stored sources, increases from 0.26 at Kirkby Stephen to 0.5 at Carlisle. The more permeable the rock, superficial deposits and soils in a catchment, the higher the baseflow (Ward & Robinson, 2000). Such variations in the base flow index highlight catchment-wide variability in surface permeability.

3.5 Catchment Hydrology

The exposure of the North West of England to westerly maritime air masses and the presence of extensive areas of high ground make it one of the wettest parts of the UK (Met Office, 2011). Rainfall in the River Eden catchment predominantly arrives from areas further to the west of the UK (Fowler et al., 2008) and due to such a high range of relief across the catchment, an enhanced orographic effect causes the volumes of rainwater received to differ substantially between lowland and upland areas. In the upland areas of the catchment, average annual rainfall exceeds 2800mm (Figure 3.4). However, downstream in the valley bottoms around Carlisle this figure is much lower and the area rarely receives more than 750mm of rainfall annually (Environment Agency, 2006). The partial rain shadow effect of the lowland valley combined with good soils provides an excellent environment for dairy and mixed farming in the central areas of the catchment. In recent decades, farmers have invested heavily in land drainage in an attempt to make land in the wetter areas of the catchment more productive.

Peak daily total rainfall records tend to be greatest in areas of high elevation. Similarly, peak rainfall intensities are also highest in the upland areas of the catchment (Mayes et al., 2006). Research by Pattison et al. (2008) showed the River Eamont, which drains from the Lake District, receives the highest total annual rainfall, with EA records showing an average of 1768mm falling per year for the period between 1977 and 2007.

Snowfall within the River Eden catchment is normally confined to between November and April, but upland areas may have occasional snowfalls in October or May. The number of days with snow falling can be as high as 60 days per year over the highest grounds, falling to just 20-30 days in lower-lying parts of the catchment.

The combination of heavy rainfall, hard volcanic rock, thin soils and steep gradients in the upper parts of the catchment promotes large amounts of overland runoff and a rapid river response to rainfall events as shown by Figure 3.5. Research has shown that at Kirkby Stephen, the River Eden responds to rainfall just 1.5 hours after the event and rain falling on the top peaks on Helvellyn, the highest point in the catchment, will reach Ullswater Reservoir in less than 2 hours (Environment Agency, 2006).

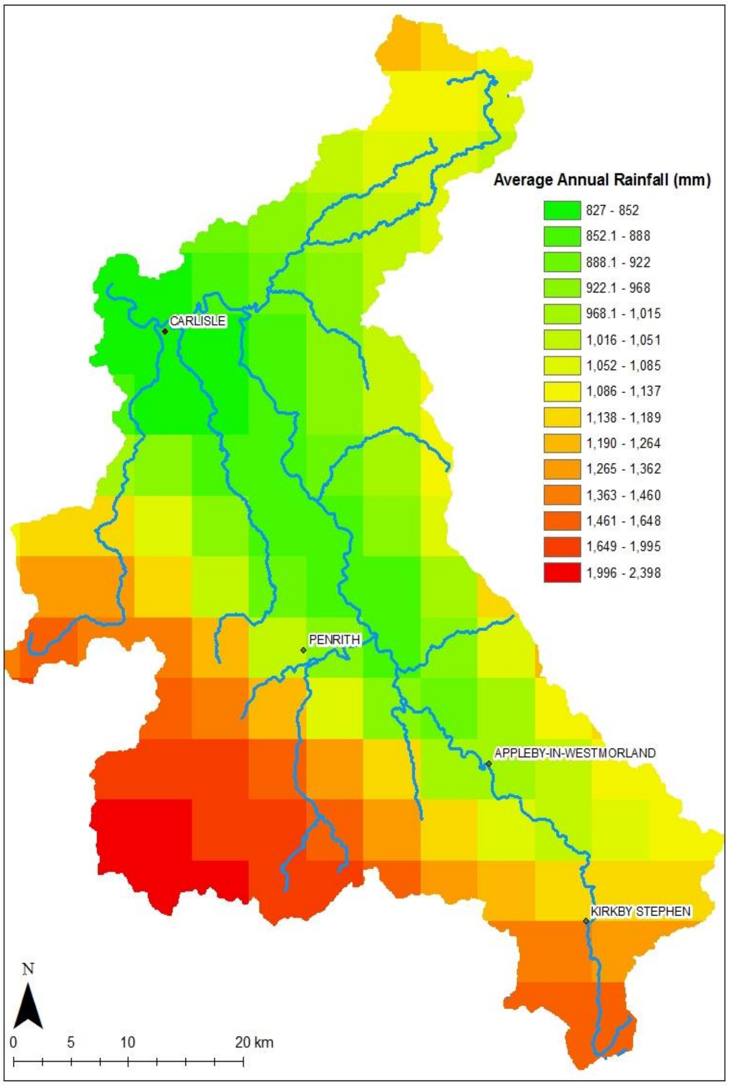


Figure 3.4: Average Annual Precipitation (mm) in the River Eden Catchment. Source: Perry & Hollis (2005).

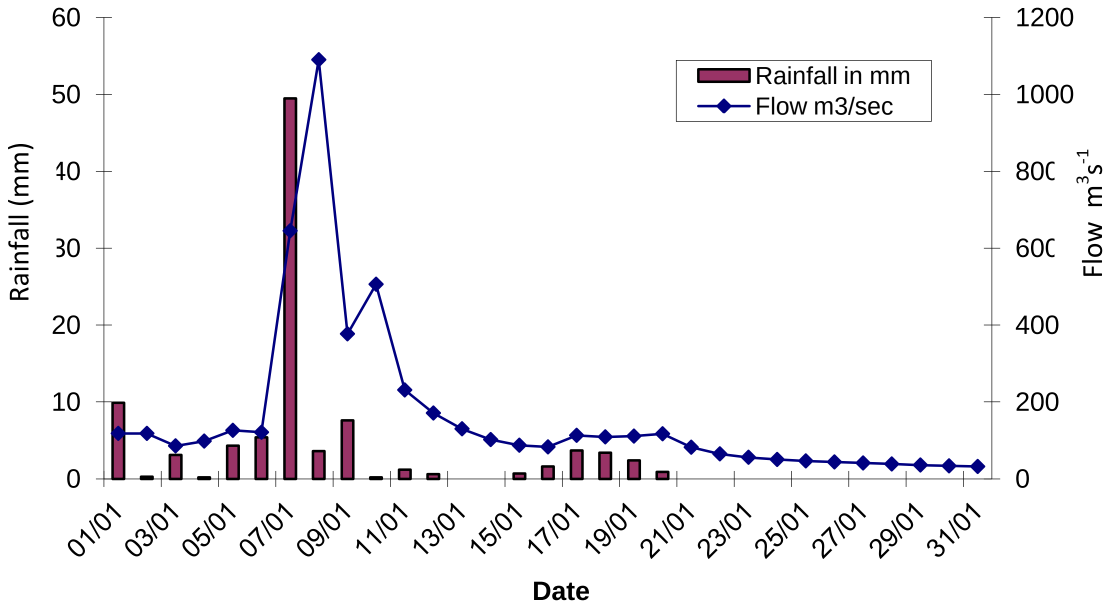


Figure 3.5: Hydrograph for a Storm Event at Carlisle, January 2005. (Source: Geographical Association, 2010)

3.6 Land Cover and Land Use

95% of the Eden catchment is classified as being agricultural or rural (CEH, 2010) and a large part of the non-agricultural land is Wark Forest, the southernmost point of Kielder Forest, in the northeast of the catchment. Only 1% of the catchment is classified as urban (Environment Agency, 2010). Tourism is the second largest industry within the Eden catchment after agriculture and accounts for 18% of total economic output (Reaney et al., 2010).

In the upland areas of the catchment the predominant agricultural sector is livestock with most farms using land for sheep and cattle grazing. During the winter months sheep will generally move from the upland, hilly areas down to the lowlands, this is known as sheep stratification. Although this process has been occurring for centuries in recent decades the number of sheep being moved off the uplands during winter has increased considerably. This is largely a result of farmers being paid, as part of environmental management and stewardship schemes, to winter sheep away in the valleys to enable fragile upland habitats to recover. Some farms within these upland areas may grow small amounts of arable crops and use the resulting crop as fodder. However, climate and soil type makes managing arable land here extremely challenging.

In the lowlands, where climate and soil is more favourable, dairy production, mixed grazing and some arable cropping can be found. Consequently, issues relating to agriculture and its intensification are often cited as the cause of environmental degradation within the catchment.

Several areas of the Eden valley, particularly those close to the main stem of the River Eden, have extremely productive soils with significant areas of Grade 2 agricultural land. Farms in these areas are generally all capable of having every field cultivated and most are in rotations of rye grass, cereals and maize.

During the past decade a considerable amount of consolidation has occurred within the dairy sector with many smaller farms making way for larger units where herds of over 300 cows are common. The majority of these cattle are housed in modern cubicle facilities whilst their manure is stored in slurry tanks during the winter prior to spreading in late spring. The introduction of Nitrate Vulnerable Zones within the catchment has led to many dairy farms having to invest large amounts of capital in providing suitable manure storage facilities.

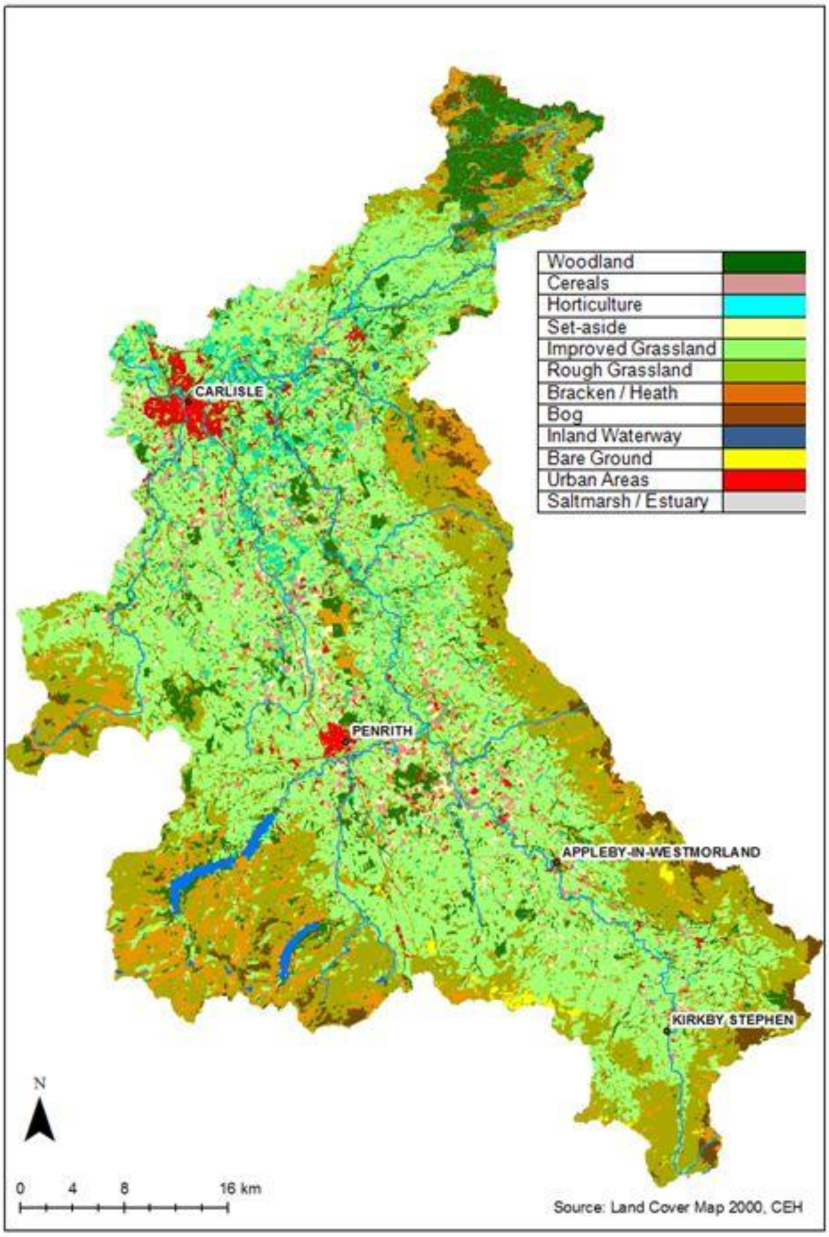


Figure 3.6: Land Cover in the Eden Catchment as classified by the Centre for Ecology & Hydrology. (CEH, 2000).

3.7 Conclusions

The River Eden in Cumbria, North West England and its surrounding catchment will be used as the application test catchment within this research. The principle water courses within the catchment include the Rivers Eden, Eamont, Irthing, Petteril and Caldew, and the total catchment area is approximately 2400km^2. As discussed above there is a great degree of variation in the topographical, geological and climatic conditions across the catchment and together these factors control the catchments hydrological responses.

Appropriate management of the River Eden catchment relies on careful cooperation between several government organisations along with local councils and charity organisations. In an attempt to protect and enhance the ecological significance of the catchment much of the catchment has been designated as a Site of Special Scientific Interest (SSSI) and a further thirty per cent of the area lies within Areas of Outstanding Natural Beauty (AONB.) A significant part of the study area lies within the Lake District National Park.

Chapter 4: Current and Projected Climate Dynamics

As discussed in Chapter 2, the recent changing focus of hydrological modelling stems largely from research into the proposed effects of climate change. This chapter explores the potential effects of projected climate change on the hydrology of the UK in future decades. It considers how UK climate may change in the future and explores the likely implications that this may have on the study catchment used in this research; the River Eden catchment, Cumbria.

4.1 Overview of Predicted UK Climate Change

Climate is not constant and varies over time scales ranging from millions of years, to glacial-interglacial cycles, centuries and decades. It is difficult to draw a clear distinction between natural climate variability and climate change. Generally, climate change can be interpreted as being the result of a change in the variables forcing climate, for example increases in greenhouse gas emissions, and in recent years the term has become synonymous with global warming (Arnell & Reynard, 2000). Climate variability, on the other hand, reflects the inherent fluctuations within climate without the presence of any forcing variables (Arnell & Reynard, 2000). Climate variability includes fluctuations in global circulation patterns and inter-annual variation in the North Atlantic Oscillation.

In recent decades the importance placed on understanding current climate dynamics has increased considerably and investigations into the causes of climatic variation have become an intense area of research across various scientific disciplines (Grace, 2005). The present concern is that global warming is now occurring at an unprecedented rate. Warming over the past century has been linked to anthropogenic changes in atmosphere composition, through the "greenhouse effect". The natural greenhouse effect refers to the ability of several gases within the earth's atmosphere, including $C0_2$, CH_4 and N_20, to absorb thermal radiation from the earth's surface. Without this effect temperatures would be substantially cooler than present and much of the water across the planet would be ice (Grace, 2005). As a result of human activity the levels of these gases in the atmosphere, along with CFCs, are now increasing and strong correlation between increasing gas levels and temperature have been observed (Grace, 2005; Henson, 2006).

The effects of global climate change are already being seen across the UK. Average temperatures have already begun to rise and eleven of the twelve years between 1995-2006 rank among the warmest fifteen years in the instrumental record of global surface temperature since 1850 (IPCC, 2007). The Intergovernmental Panel on Climate Change (IPCC) has predicted that in forthcoming decades the occurrence of hot, dry summers will increase, whilst the number of extremely cold winters will decrease (IPCC, 2007). As a result of this, average annual temperatures across the UK are likely to increase by between 0.9 and 2.4°C by 2050 (IPCC, 2007). The UK Climate Impacts Programme has indicated that temperatures will continue to increase beyond the 2050s and most areas can expect to be 2°C to 3.5°C warmer by 2080. In some locations the amount of warming could even exceed 5°C (Environment Agency, 2006). Such increases in temperature could have detrimental impacts on agricultural productivity along with human health, water resources and water quality (Henson, 2006).

Climate trends over the past 30 years show winters have already become wetter and more rain is falling in storm events (Osborn et al., 2000; Burt & Ferranti, 2010). Based on a low-emissions scenario a 10% - 20% increase in total annual rainfall, dependent upon location in the UK, has been forecast (Ekstrom et al., 2005). For a high-emissions scenario, the range increases to between 15% and 35% (UKCP, 2009). These increases in winter rainfall levels, predominantly from frontal storms, are likely to contribute to higher annual rainfall totals (Burt & Ferranti, 2010). Fowler and Kilsby (2002) predict increases in rainfall will probably result in increased river flows of up to 20%.

Overall increases in rainfall will be accompanied by an increase in the number of heavy winter storm events. In the northwest of Britain, rain events which currently occur on average every two years are predicted to become annual occurrences (Hulme et al., 2002). The intensity of these storms is also forecast to increase meaning greater amounts of rain will fall on catchments over shorter periods of time (Fowler & Kilsby, 2002). Such an increase in storminess, combined with generally wetter winters, could result in increased flood risk for many places across the country (Environment Agency, 2006) and research by Cameron (2006) has shown that under all of the various UKCP02 climate scenarios flood magnitudes will increase. This is particularly the case for the north of England where the biggest increase in the number of winter storms has been predicted (Osborn et al., 2000).

Whilst levels of winter rainfall are expected to increase, precipitation during the summer months is likely to decline (Fowler et al., 2007). Model results based on a low-emissions scenario suggest the UK could become up to 35% drier whilst predictions based on a high-emissions scenario suggest an even greater decline in rainfall levels which would see the UK become 50% drier than present by the 2080s (UKCP, 2009). A reduction in total summer rainfall will not necessarily mean intense summer rainfall events will become less frequent and it has been suggested that increased atmospheric temperatures may actually result in an increase in the number of convective storms (Lane, 2008).

The reduction in total summer rainfall, combined with warmer summer temperatures, may lead to increased levels of evaporation thus putting greater pressure on water resources. Higher temperatures are likely to lead to increased demand for water along with higher peak demands, both of which would require increased financial investment in water resources and infrastructure (Glynn, 2005). Herrington (1996) predicts an increase in average demand for domestic purposes of almost 5% by 2020. Furthermore, higher summer temperatures would enhance the demand for agricultural water abstraction for irrigation. The combination of increased agricultural and personal demand means that it is widely anticipated that droughts, such as those seen during the summer of 2010, will become more common (Fowler et al., 2008).

An overview of the likely effects of climate change in the UK is given in Table 4.1. It should be noted that in different catchments and management units, the magnitude and extent of the impacts listed below will differ, and not all impacts will necessarily occur in every area or catchment (Arnell & Reynard, 2000).

The severity of all projected climate impacts will largely depend on the amount of greenhouse gases released into the atmosphere from now on, and how the climate system responds. Although the basic principles behind global climate change and global warming are relatively simple, knowing how the world's climate systems will react to these changes and the potential feedback mechanisms that could be caused is far less certain (Arnell & Reynard, 2000). Much research has suggested that preventing the effects of anthropogenic climate change is now not possible (e.g. Hulme & Turnpenny, 2004; Henson, 2006); however, management and mitigation if implemented correctly could help to manage some of these potential impacts.

Phenomenon and direction of trend	Examples of major projected impacts by sector			
	Agriculture, forestry and ecosystems	**Water resources**	**Human health**	**Industry, settlement and society**
Over most land areas, fewer cold days and nights, warmer and more frequent hot days and nights	Increased yields in colder environments; increased insect outbreaks	Effects on water resources relying on snowmelt; effects on some water supplies, less snowfall	Reduced human mortality from decreased cold exposure	Reduced demand for heating; increased demand for cooling; declining air quality in cities; reduced transport disruption due to snow, ice.
Warm spells/heat waves. Frequency increases over most land areas	Reduced yields in warmer regions due to heat stress; increased danger of wildfire	Increased water demand; increased abstraction demands from reservoirs; water quality problems, e.g. algal blooms	Increased risk of heat-related mortality.	Changes to type of agricultural production; commodities grown for export change; impacts on the elderly, young and poor
Heavy precipitation events. Frequency increases over most areas	Damage to crops; agricultural runoff; soil erosion, inability to cultivate land due to waterlogging of soils	Adverse effects on quality of surface and groundwater; contamination of water supply; water scarcity may be relieved. Possible flash flood events.	Increased risk of deaths, injuries and infectious, respiratory and skin diseases	Disruption of settlements, commerce, transport and societies due to flooding: pressures on urban and rural infrastructures; loss of property
Area affected by drought increases	Land degradation; lower yields/crop damage and failure; increased livestock deaths; increased risk of wildfire. Disruption to water ecosystems.	More widespread water stress. Adverse effects on quality of surface and groundwater.	Increased risk of water shortage; increased risk of water- borne diseases	Water shortage for settlements, industry and societies; reduced hydropower generation potentials; potential for population migration
Increased incidence of extreme high sea level	Salinisation of irrigation water, estuaries and fresh-water systems	Decreased fresh-water availability due to saltwater intrusion – disruption to ecosystems.	Forced migration from coastal areas – associated health risks.	Costs of coastal protection rise; potential for movement of populations and infrastructure

Table 4.1: Projected future climate change impacts across the UK. Source: Adapted by author from IPCC 2007 Report.

4.2 Current Climate Dynamics in the River Eden Catchment

Currently climate across the River Eden Catchment varies considerably, mainly as a result of the effect the Lake District and Pennine Hills have. Long-term precipitation data from the Environment Agency and average annual temperature records obtainable from the Met Office show a strong degree of variability between upland and lowland areas. In the upland areas of the catchment, average annual rainfall exceeds 2800mm, however downstream in the lower valleys this figure is much lower and areas near the city of Carlisle rarely receive more than 750mm of rainfall annually (Met Office, 2011).

Periods of prolonged rainfall can lead to widespread flooding, especially in winter and early spring when soils are usually near saturation. In January 2005 widespread flooding at Carlisle, considered to be the worst to affect the city since 1882, resulted in water overtopping many river defences and flooding of almost 3,000 properties (Environment Agency, 2006). Similar, unprecedented floods in November 2009 had serious adverse effects on properties and businesses across the catchment, including in the town of Appleby-in-Westmorland. During this period, 227 mm of rainfall were recorded at the Environment Agency's rainfall gauging station at Shap in a single 72 hour period (Environment Agency, 2006).

Average annual air temperature at Carlisle is currently 9.5°C. At Penrith, which is further upstream in the catchment, temperatures are slightly cooler on average and annual average air temperature for the last decade is approximately 9.1°C. Temperatures continue to fall when moving further up into the catchment headwater areas. This is a result of the increase in elevation combined with the strong influence that high winds have on the exposed upland moors. Recent estimates from the Met Office suggest a decrease of 0.5 °C for every 100 metre increase in altitude (Met Office, 2011). In December 2010, during a particularly long cold period, the Meteorological Office weather station at Newton Rigg recorded temperatures as low as -9.0°C (Met Office, 2011). In the more rural and exposed parts of the catchment it is thought temperatures will have fallen even further.

The warmest month within the catchment ever recorded occurred in the summer of 1995 when temperatures in August averaged 24.5°C. This is closely followed by July 2006 when temperatures averaging 24.0°C were recorded over the course of the month. Thunderstorms are most likely to occur during these particularly warm summer months and on 8-10 days per year thunder is usually heard across the catchment (Met Office, 2011).

A degree of spatial variability in temperature can be identified across the catchment. Western areas of the catchment are generally cooler compared to areas nearer the coast where average annual temperature is between 9°C and 10°C. Despite lower temperatures occurring in the western region of the catchment these areas are still warmer than the central Lake

District generally where higher elevations mean average annual temperature rarely manages to average more than 5°C (Met Office, 2011).

The River Eden catchment and surrounding areas of the Lake District and Pennine Hills are not renowned for high levels of sunshine. In general, sunshine duration decreases with increasing altitude and distance from the coast and such trends can be seen here. In the higher Pennine regions, upstream of Kirkby Stephen, 1200 hours of sunshine are recorded on average per year. Further downstream, at Carlisle and at the coast where the main river channel enters the Solway of Firth, sunshine occurs on average more regularly and records show 1500 hours of sunshine per year are typical (Met Office, 2011).

Water levels within the catchment are known to vary considerably, especially in winter months, as a result of snow melt in the surrounding uplands. The occurrence of snow is linked closely with temperature, with falls rarely occurring if the temperature is higher than 4 °C. For snow to lie for any length of time, the temperature normally has to be lower than this. Over most of the area, snowfall is normally confined to the months from November to April, but upland areas may have occasional falls in October and May. Snow rarely lies on low ground outside the period from November to March but over higher ground lying snow can also occur in October and as late as May. On average snow falls on 20-30 days in the lower lying parts of the catchment but can occur on as many as 60 days over the highest ground (Met Office, 2011). An average increase of about 5 days of snow falling per year per 100 m increase in altitude has been found to be typical.

Snowfall is highly variable from year to year. For example, at Carlisle there were 26 days with snow lying in the unusually cold winter of 1962/63 but no snow at all was recorded during the winter of 1963/64. Generally, snow does not exceed depths of 15cm at low altitudes but during extreme events depths of 30-60cm are possible as a result of drifting. Over the winter of 2010 snow was present over much of the catchment for several weeks due to a period of particularly cold weather. Ground frost can be observed on 75 to 130 days on the upland areas of the catchment.

The River Eden Catchment lies in one of the most exposed areas of the UK. The strongest winds are associated with the passage of deep areas of low pressure close to or across the UK, the frequency and strength of which are greatest in the winter months when mean speeds and gusts are strongest.

4.3 Projected Climate Change in the River Eden Catchment

The potential impacts of projected climate change are likely to have a profound effect on the hydrology and associated diffuse fine sediment pollution dynamics of the River Eden catchment. Catchments have a non-uniform and non-linear response to rainfall events caused by the spatial variabilities in hydrological processes due to complex geological, pedalogical and management histories (Cerda, 1995). As a result, it is predicted that future climate change has the potential to alter the hydrological dynamics of catchments at both field, farm and catchment scale (Reaney et al., 2009). Recent research has suggested there will be changes in the size of hydrologically connected areas, the duration of connection-disconnection cycles and the number of connection-disconnection cycles. The combination of these three changes will alter channel dynamics, the export of non-point source pollutants, the potential for high flows and the location of critical source areas.

Research conducted by the UK Climate Predictions Group along with work undertaken by the Environment Agency suggests mean levels of winter precipitation in the northwest of the UK during the 2020s will increase by around 6%, whereas summer precipitation may fall by as much as 8%. By the 2050s it is likely that these seasonal differences in rainfall levels will have become even more apparent, with estimates suggesting that winter precipitation could increase by as much as 13% whilst summer levels drop by nearly 18%. This pattern is predicted to continue into the 2080s when winter levels could increase by 16% and summer levels fall by 22% (UKCP, 2009). It has been shown by Lane et al. (2008) that the impacts of future climate change and the redistribution of rainfall inter-annually is likely to increase rates of fine sediment delivery to waterways. Not only will this be a result of greater sediment generation but also because connectivity between sediment sources and the river network is likely to increase (Lane et al., 2009).

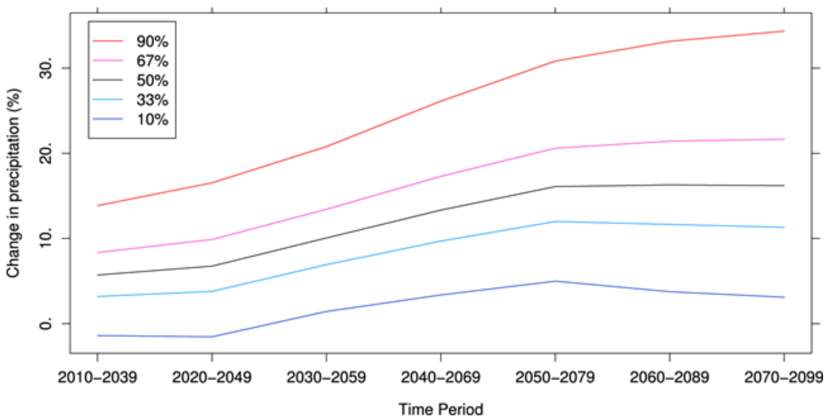


Figure 4.1: Projected future changes in winter precipitation levels for the north west of UK based on medium emissions scenario. Each line corresponds to a probability level for the given time period. Source: UKCP09 (Available at: http://ukclimateprojections.defra.gov.uk/content/view/1538/499/)

Such changes would have dramatic impacts on the catchment; with an increase in winter flood events likely, whilst summer months would become characterised by water shortages and droughts. Much of the River Eden catchment relies on surface water resources because the local geology results in groundwater storage potential of around only 15% (Allen et al., 2010). Research by Fowler et al. (2007) found that the river system is likely to be able to meet abstraction demands for most of the year; however, a significant investment in abstraction equipment will be needed so water can be obtained from lake and borehole sources instead of just the river channel. Simulation runs, based on current catchment conditions, have indicated that future predicted climate change combined with high peak demands will likely result in there being occasions when the system is unable to meet demand and the system becomes compromised (Fowler et al., 2007).

Longer periods of low flows in the majority of the region's waterways during the summer months would reduce the quality of river water and available habitat space (Williams et al., 2000). Nutrient inputs may stimulate greater biological activity under low-flow conditions whilst pollutants will be diluted into lower flows thus increasing concentrations. This could create several eutrophic zones within a channel that fewer species would be able to survive in (Ferguson et al., 1996). At the same time, the drying out of channel banks would leave material more susceptible to erosion and thus increases in levels of fine sediment in waterways are likely. A rise in mean average air temperature is likely to be accompanied by a similar increase in water temperatures. Warmer water can hold less oxygen, so dissolved oxygen levels will fall, and the potential for oxygen-consuming algal blooms will increase further (Webb, 1992).

The intensity of rainfall within the River Eden catchment is also expected to be altered as a result of climate change and by the end of the 21st century, a 10-30% increase in the magnitude of UK rainfall events up to a 50-year return period has been predicted (Burt & Ferranti., 2010). The more intense a rain storm is, the greater powers of erosivity rain droplets have (Bracken & Croke, 2007). When these high-intensity droplets hit an unprotected surface particle detachment is likely, making material available for transport (Nearing, 2001).

High-intensity rainfall events pose an increased risk of flash flooding and overland flow especially during the cold, winter months when soils are frozen and infiltration is prevented from occurring. Such increases in overland flow can lead to the transport of soil particles and pollutants into waterways thus having a detrimental effect on water quality (Merrington et al., 2002). An increase in levels of surface flow would likely be mirrored by an increase in fine sediment transport as water moving across the surface entrains loose soil particles and transports them to nearby waterways. During the winter months, especially on agricultural land where soils are often left bare without any protection from vegetation, this could become a particularly severe issue.

A second dominant pathway of influence by climate change on soil erosion rates is through changes in plant biomass. This is a complex process in which climate change directly affects levels of plant biomass which then affect levels of surface runoff (Pruski and Nearing, 2002). For example, increases in atmospheric carbon dioxide concentrations will increase production rates of certain plants, creating a larger ground cover of plants. This increase in ground cover will protect the underlying soils from rainsplash impact and wind erosion, reducing the amount of material susceptible to detachment. On the other hand, increases in rainfall levels, combined with predicted increases in air temperatures, will cause faster rates of plant decomposition and hence a larger amount of material may be left on the surface ready to be entrained (Pruski and Nearing, 2002).

An even more complex idea to think about when considering the possible effects that changing precipitation patterns may have on fine sediment pollution risk is the potential for changes in land use and land management which may be necessary to accommodate for changes in climate (Nearing et al., 2004). For example, a reduction in spring rainfall would result in delayed germination of spring-sown crops thus leaving soil bare and hence more susceptible to erosion for longer (Nearing et al., 2004). Without appropriate land management an increase in fine sediment erosion during winter months is therefore likely (Nearing et al., 2004). However, at the same time increased summer temperatures and decreased summer rainfall would mean soils would be drier come the autumn months. Hence, heavy autumn rainfall would be more likely to infiltrate as opposed to immediately becoming overland flow.

Changes to average temperatures and rainfall levels are likely to drive an increase in the total amount of land used for arable activities. Wilson (1989) suggested that the total amount of land classed as arable in the UK may increase by as much as 28% as a suitable climate for crop growing is created in more areas. As growing seasons become longer, upland areas of improved grassland are likely to be replaced with cereals (Boardman et al., 1990). Although providing protective cover to soils for large parts of the year, cereal crops regularly cause high erosion rates because of the bare periods associated with the preparation, drilling and early growth phases on new crops. Combining this issue with the drying out of peatland in catchments, and the ongoing loss of areas of natural heather, will greatly increase the risk of soil erosion and associated fine sediment pollution (Arnett, 1980).

Along with predicted variations in the amount of crops grown within the UK, it has also been suggested that the type of cereals grown across the UK will alter, driven by changes in climate and economics. Research by Boardman et al. (1990) found that autumn-sown cereal crops will decline, as a result of predicted wetter winter weather, and be replaced by more erodible spring-sown crops such as maize. Studies have shown that erosion rates of maize, particularly when planted on sandy loam soils, can be three times greater than winter cover crops (Boardman et al., 1990). In the River Eden catchment a decline in the number of winter crops is already being observed and this trend is likely to continue in future decades.

4.4 Evidence of Projected Climate Change in the River Eden Catchment

To assess whether projected variations in climate can be identified within the test catchment used in this research long and short-term records of precipitations were analysed. Variations in precipitation levels were investigated at century, decadal and annual scales and these findings are presented below. Spatial variation in rainfall across the catchment was also analysed.

4.4.1 Long-term Analysis of Rainfall Patterns

Both the UK Meteorological Office and Environmental Agency hold long-term rainfall records for several measuring gauges within the River Eden catchment. The most complete of these records is for Appleby Castle (Station Number: 12936), where daily rainfall totals have been recorded since 1891. Data are complete from 1891 to 2000 with the exception of September and October 1963 which were infilled from a nearby station (Appleby Bongate, Number: 12937). Data from April 2000 to February 2005 are from Appleby Mill Hill (Station Number: 30005). Both stations are within 2 km of the castle, hence in this instance infilling the data set in this way was deemed to be acceptable (Burt & Ferranti, 2010). Of course there must always be some doubt about this method and it would have been preferable to have records from a single site.

The average monthly precipitation levels for Appleby are shown in Figure 4.2 along with the standard error of the mean. There is relatively low inter-annual variability of annual rainfall totals with the coefficient of variation being 0.5.

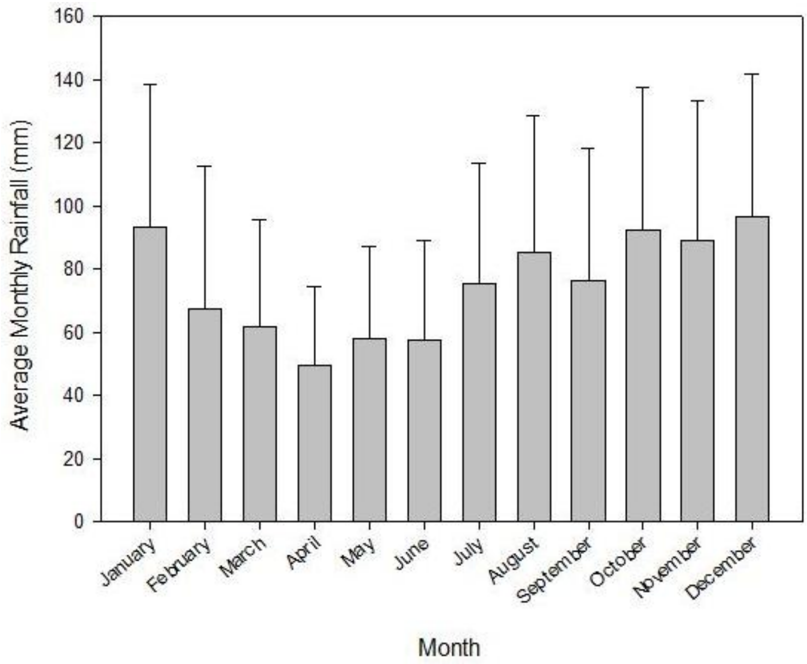


Figure 4.2: Average monthly rainfall recorded at Appleby 1891-2010.

The climate at Appleby shows strong seasonal trends with maximum rainfalls occurring in October and December. Such high amounts of precipitation in these months are likely to be a result of the occurrence of convectional storms. Minimum rainfall occurs in April and May.

In order to establish an accurate representation of the long-term climate trends operating within the River Eden Catchment it is important to determine whether the climate has been showing stable trends over a prolonged period. In order to investigate this, published long-term monthly climate trends were broken down as part of this research into ten periods, each representing one decade. For this analysis decades were taken to run from year 1 to year 0; for example the years 1961 to 1970 were assigned to the 1960s. This follows the standard convention set out by the World Meteorological Society where climate averages are for the period 1961 -1990.

A simple t- test was used to analyse any significant differences between the decadal datasets. When comparing the first winter 30-year trend (1901-1931) with the last (1981-2011) the test returned a value of 0.05, which indicates that there are not significant differences in levels of total precipitation at Appleby on a decadal scale over the time period studied. A Mann-Whitney rank sum test confirmed these results.

Long-term autumn, summer and spring rainfall were also analysed to determine whether total amounts of rainfall during these seasons at Appleby has changed since 1891. Seasons were defined using UKMO convention: winter is December (of the previous year) to February, spring is March to May, summer is June to August, and autumn is September to November (Burt & Ferranti, 2010). Similar results to those found for changing winter trends were observed for the remaining three seasons. T-test results show there has not been a significant change in patterns of rainfall at Appleby over the period of time studied here.

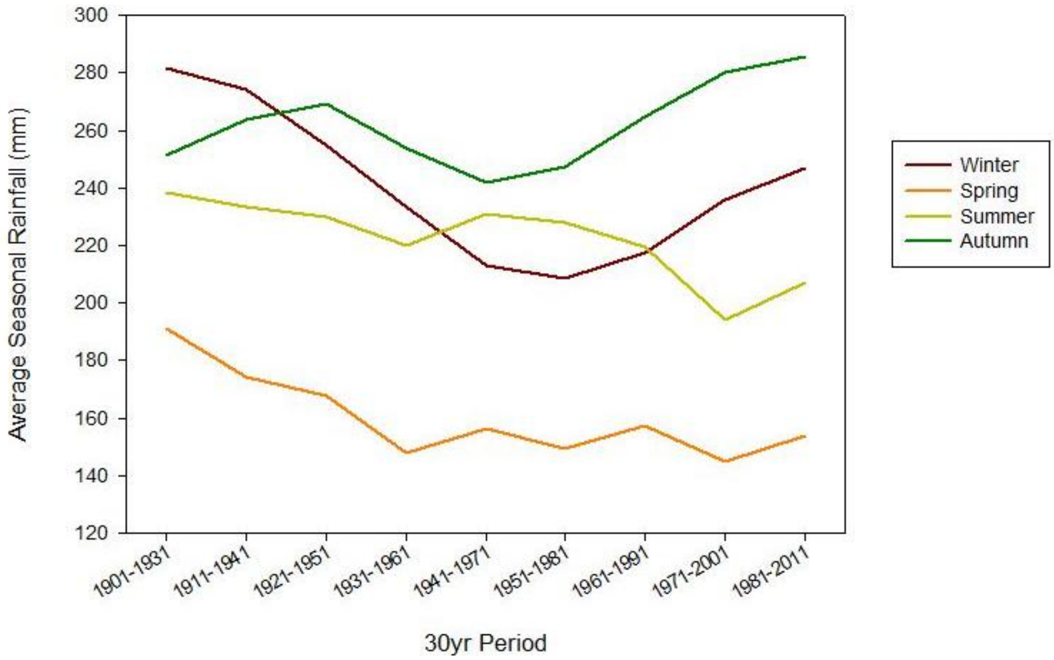


Figure 4.3: 30-year average seasonal rainfall patterns recorded at Appleby 1891-2010.

Over the period of the record studied at Appleby 57% of days experienced rainfall and 3% of days had more than 15mm of rain. Days with more than this amount of rain were, for the purpose of this research, classified as storm days. The probability distribution function for the depth of rainfall on rain days can be seen in Figure 4.4.

Of the rain days that occurred across the time period studied most days received only a very small amount of rain with 75% of days receiving less than 3mm and 83% less than 5mm. However, despite the majority of days having very light rainfall, on less than 1% of days more than 40mm of rainfall fell at Appleby and the maximum daily total recorded for the site is 97.8mm which occurred during a storm event in July 1983.

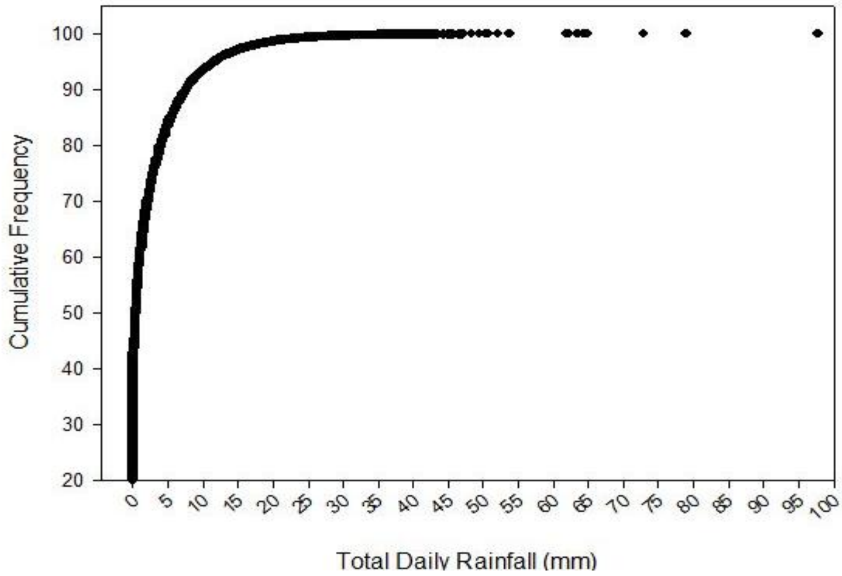


Figure 4.4: Probability distribution curve for daily rainfall totals at Appleby 1891-2010.

4.4.2 Shorter-Term Variability

Although there has been little long-term change in climate trends over the study period, analysis of the rainfall record from Appleby highlights several changes in climate over the last 50 years. When total annual rainfall per year is plotted on a simple line graph (Figure 4.5) it can be observed that annual rainfall is decreasing on average and has been since the middle of the 1960s. Similarly, it can be seen that since the beginning of the 21st Century there is greater variability between total rainfalls from one year to the next. This can be identified in Figure 4.5 by the increasingly spiky nature of the graph towards the end of the time series.

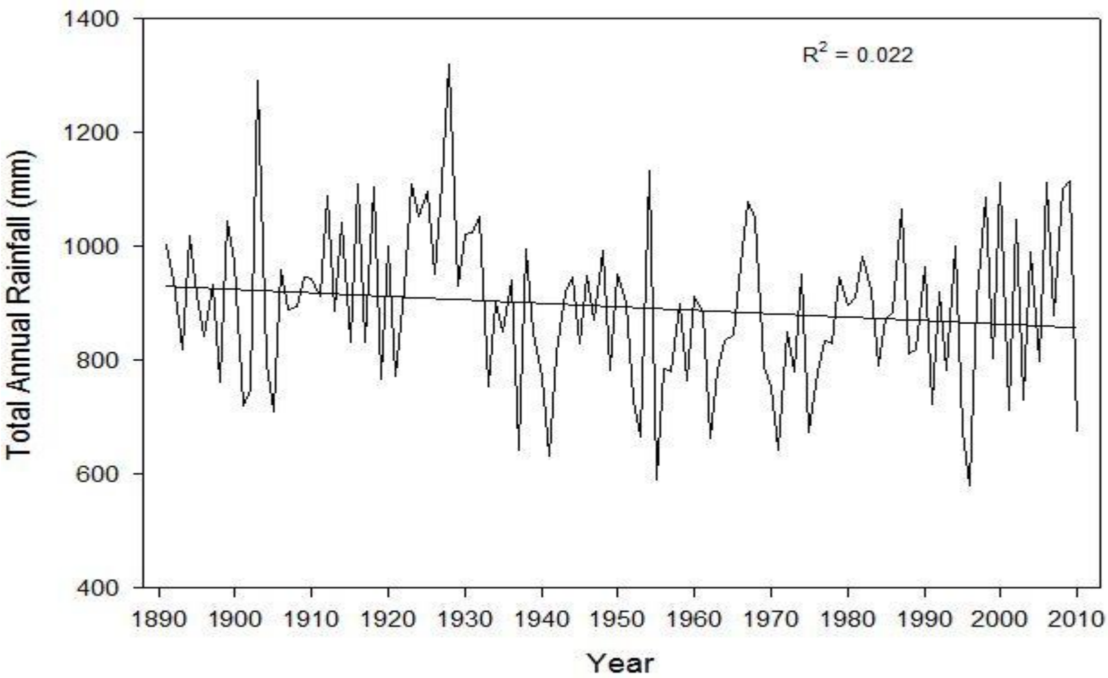


Figure 4.5: Average annual rainfall for Appleby 1891-2010

4.4.2.1 Seasonal Variability

As discussed previously in this chapter, it is widely accepted that winter months are likely to become wetter in future years as a result of predicted climate change whilst summer months will become drier. A strong degree of seasonal variability can already be identified within the catchment by studying the long-term daily rainfall record at Appleby. Over 150 mm more rain regularly falls in the wettest season of a year compared to the driest. Autumn is usually the wettest season at Appleby when over 250mm of rain falls on average. During the 120 year period that records are available for the Appleby monitoring station, autumn has been the wettest season in 74 years. The high levels of rainfall occurring in autumn are in marked contrast to spring, the driest season, when just 160mm of rain occurs on average. The driest spring on record occurred in 1984 when just 62mm of rain fell over the course of three months at Appleby.

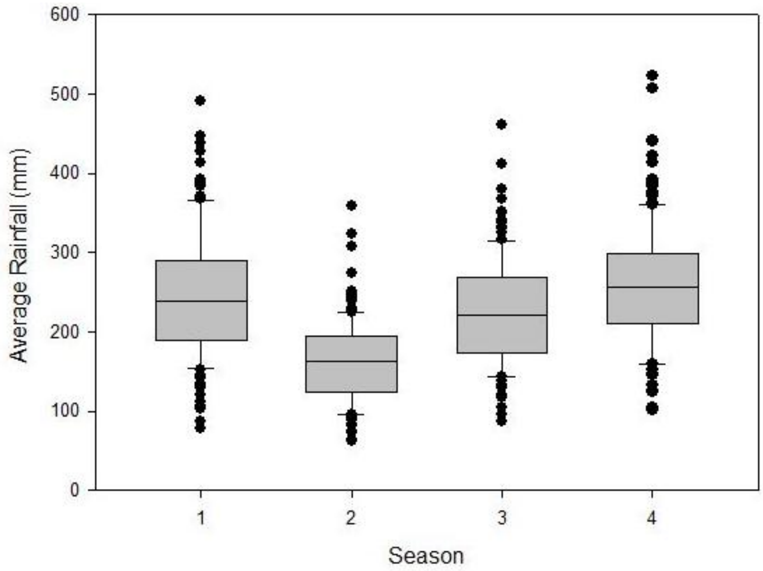


Figure 4.6: Seasonal variability in rainfall totals at Appleby 1891-2010. (Season 1 = Spring)

Strong increases in winter rainfall in the 1980s and 1990s can be observed for the Appleby gauge and this supports the finding of Malby et al. (2007) who noted an increase in winter rainfall across the north-west of England from the 1970s to 1990s. Data shows the highest average decadal winter totals for Appleby occurred in the 1990s and that between the 1940s and 1990s there was a strong increase in the ratio of winter-to-summer rainfall.

Analysis of patterns of summer rainfall for the Appleby gauge support predictions that summer rainfall totals are currently declining (Figure 4.7). Since 1990 some exceptionally dry summer periods have been observed, most notably in 2005 when less than 85mm of rain fell throughout the summer months. Levels of summer rainfall within the catchment recovered somewhat in the 2000s however levels are still much lower than those recorded in the 1960s.

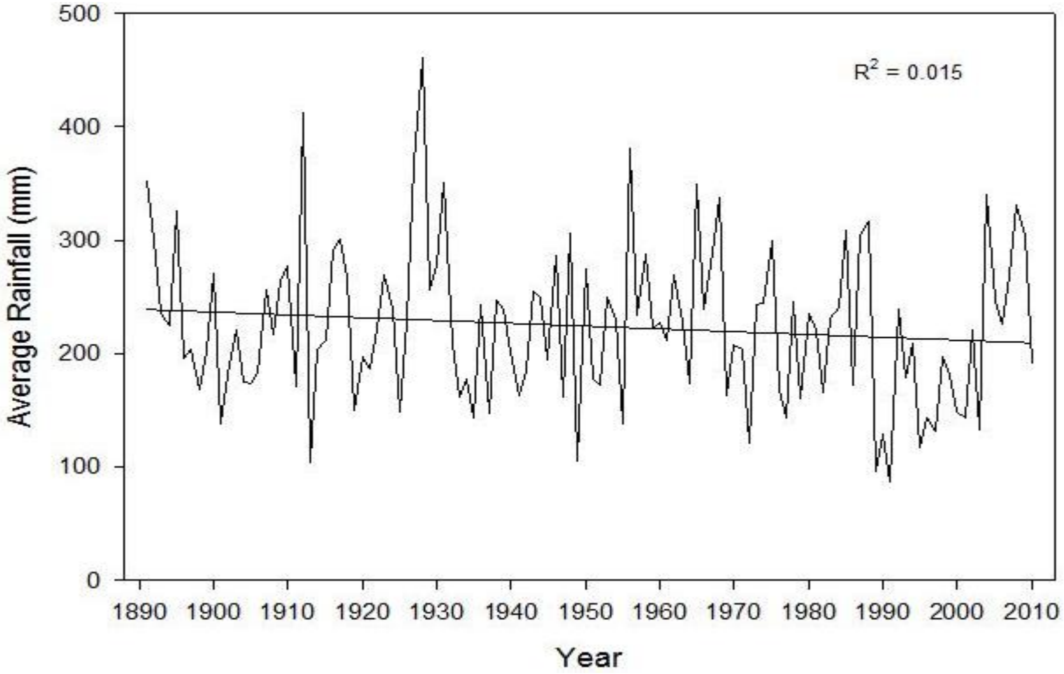


Figure 4.7: Changing patterns of average summer rainfall at Appleby 1891-2010.

4.4.2.2 Storm Events

The long time series of data available for the Appleby measuring gauge allows the frequency and inter annual variability of storm events to be considered in a longer historical context than in most other studies. When these annual trends are subdivided into seasonal totals an increase in the frequency of winter storm events combined with a reduction in the frequency of summer storm events can be observed.

Historically, data recorded at the Appleby gauge shows a dominance of storm events in the summer months throughout the period from the 1940s to the 1980s. It has only been in more recent decades, since the late 1980s that intense winter storm events have outnumbered summer storms. During the period from the 1970s to the 1990s a doubling in the number of intense winter events at Appleby can be observed.

Although the 1990s recorded the most winter storm events at Appleby, this period saw the fewest number of summer storm events and thus this shift in seasonal patterns dramatically increased the winter: summer storm event ratio in the 1990s. Although the 1990s saw the highest number of winter storm events (12) ever recorded at Appleby totals of 9 or 10 storms per year were common prior to the 1940s. It has been argued that the middle of the 20th Century was a “flood poor” period (Lane, 2008) and data recorded at Appleby in the 1940s and 1950s appears to support this theory. In both decades only 3 winter storm events were recorded.

The shift from summer to winter rainfall seems to have reversed to some extent in the 2000s as more summer storm events and slightly fewer winter events have occurred. Results from the present decade, although these are currently incomplete, suggest a reduction in the winter: summer ratio of storm events across the River Eden catchment. It has been suggested therefore (e.g. Osborn et al., 2000) that variation in the frequency of summer storm events is a result of inter-decadal variability rather than a representation of a long-term change in trend.

4.4.2.3 Rain Days

Since 1990 there has been a marked decline in the number of days during the year when rain is recorded at Appleby as shown by Figure 4.8. Using the method suggested by Burt & Ferranti (2010) rain days were classified in this research as any day where rainfall exceeded 0.25mm and all data below this level was excluded. For a very long daily rainfall series, only a very small fraction of total rainfall is thereby excluded.

Over the period of the record obtained from Appleby 57 % of days experienced rainfall. Throughout the 1990s the number of rain days recorded at Appleby has been declining thus highlighting the widespread shift towards more frequent incidence of, and greater contribution from, heavy falls of rain.

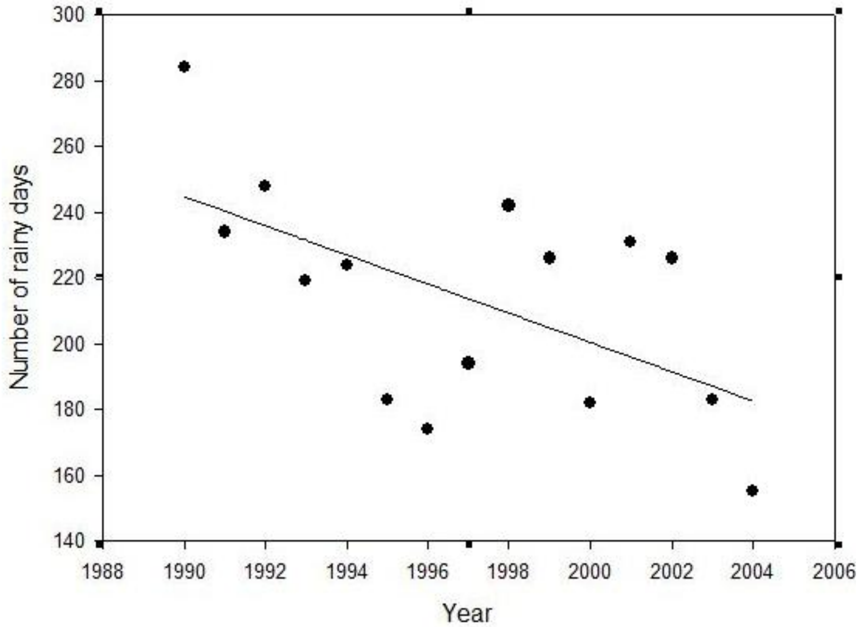


Figure 4.8: The declining number of "rain days" (where rainfall exceeded 0.25mm) at Appleby from 1988-2005.

4.4.3 Spatial Variability

Further analysis of the Appleby record, alongside records from Broadfield House and Kirkby Stephen, was carried out to determine the extent of spatial variability in rainfall levels across the River Eden Catchment. As shown previously in Figure 3.4, rainfall levels are known to vary considerably across the catchment and hence it is important to capture this degree of spatial variability when modelling catchment hydrological connectivity.

Figure 4.9 shows average monthly rainfall for the three locations studied across the catchment. Kirkby Stephen, the most southerly gauging station used in this research, receives 909mm of rain on average per year. This is in comparison to Appleby, further downstream along the main River Eden channel, which receives 894mm on average and Broadfield House where 810mm can be expected on average. A pattern such as this is to be expected when moving downstream along the course of a river valley such as the Eden and these spatial variations are similar to those shown in the Meteorological Office 5km griddled annual average rainfall maps (Perry & Hollis, 2005).

Similarities in patterns of monthly precipitation can be identified between Broadfield House and Appleby. However at Kirkby Stephen, which is further up into the headlands of the River Eden catchment, patterns of average rainfall across the year are considerably different. Average monthly rainfall over the winter months here is much higher than at either of the other two locations and summer rainfall levels, particularly in August, are much lower. There is also a greater degree of variation between the winter and summer season. At Appleby 78% more rain falls in the wettest month of the year (December) compared to the driest month (April). At Kirkby Stephen however there is a huge 300% difference between rainfall levels in the wettest month (January) and the driest (August).

Strong increases in winter rainfall in the 1980s and 1990s are observed at the Appleby gauge however this effect is not seen at the lowland gauge at Broadfield House. These findings support the work of Malby et al. (2007) who recorded an increase in rainfall across the region during the 1980s and 1990s, with greatest increases being found at higher elevations.

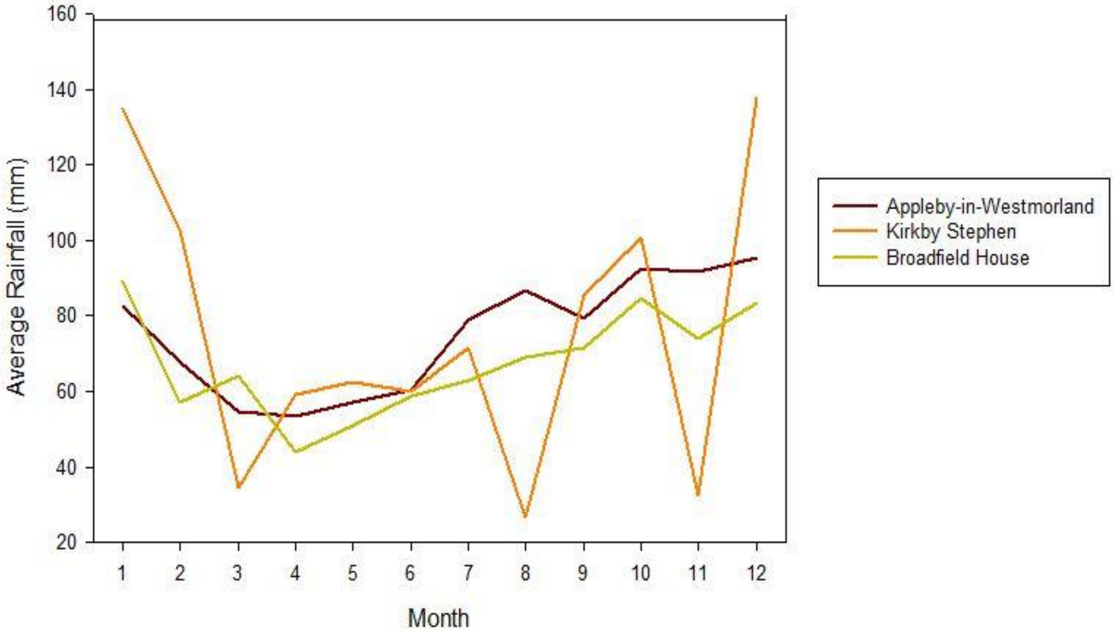


Figure 4.9: Spatial variation in rainfall totals across the River Eden catchment. This graph shows average monthly rainfall at three locations across the River Eden catchment, 1972-2006.

4.5 Conclusions

It can be seen that predicted future climate change has the potential to dramatically change the hydrological dynamics of the River Eden catchment. To ensure these changes do not have detrimental effects on waterways across the catchment appropriate management and mitigation needs to be carefully implemented. As it has been shown that the effects of climate change are unlikely to be evenly distributed across the catchment, each location needs to be individually assessed and appropriate management options selected.

It can be seen that average annual rainfall across the River Eden catchment varies considerably at both temporal and spatial scales. Analysis of the long-term Appleby rainfall record has enabled us to see that total annual precipitation, particularly during the summer months, is decreasing markedly. In contrast to this, average winter precipitation levels are continuing to rise and the contrast between winter and summer averages are becoming more apparent. Furthermore, the number of rain days is decreasing however when rain does occur more is falling over the course of a day. As this research is focused on the potential effects of global climate change on hydrological connectivity and associated diffuse pollution risk the fact that climate induced changes to rainfall levels can be identified at Appleby show the River Eden Catchment to be an appropriate test catchment for use in this research.

Spatially it has been shown that there is significant difference in total annual average rainfall between various locations in the River Eden catchment with areas in the uplands, such as Kirkby Stephen, receiving more rain on average than areas further downstream. When attempting to accurately represent rainfall trends across the River Eden catchment it is therefore important to ensure that these spatial trends are captured. Hence, in the SCIMAP framework used in latter parts of this research, a 1 km^2 gridded dataset is used as opposed to a single average annual rainfall value for the entire catchment.

Long-term analysis of the rainfall record from Appleby has shown that, despite the recent changing trends in seasonal rainfall, there is little significant variation in total annual rainfall from 1891-2010. This therefore shows that using an annually averaged rainfall dataset will be an appropriate method to simulate long-term rainfall characteristics within the test catchment.

Despite the evidence for anthropogenic global climate change it should be noted that climate change is not the only factor controlling climate within the study catchment and, as it is difficult to draw a clear distinction between natural climate variability and climate change, careful research to accurately identify the controlling factors behind observed variations is required.

Chapter 5: Methodology

This chapter introduces the two modelling frameworks used in this research, SCIMAP and TRIP, and the methodologies required to apply them to the River Eden catchment. Firstly, the data needed to apply SCIMAP to a test catchment and the pre-processing requirements of the model are explained (section 5.1). In the latter parts of this chapter, the methodology behind the Temporal Risk Integrated Platform (TRIP) which was created as part of this research is discussed and an explanation given as to how this can be combined with SCIMAP to incorporate temporal patterns of risk into diffuse pollution modelling (section 5.2).

5.1 Data Requirements & Acquisition

In order for SCIMAP to compute the relative risk of an area producing a diffuse fine sediment pollution risk three layers of data are required, a digital elevation model, a land cover map and a map of the time integrated rainfall pattern (Lane et al., 2009).

5.1.1 Digital Elevation Model

A Digital Elevation Model (DEM), a representation of the topography of an area, is needed in order for slope gradient and topographic attributes of the catchment such as flow routing pathways and slope to be calculated. The resolution of the DEM is extremely important as several variables including upslope contributing area and the topographic wetness index have been shown to be sensitive to resolution changes (Elsheikh & Guercio, 1997). Coarsening of DEM resolution usually results in reduced predictive capabilities of models and a decline in the reliability of model outputs (Elsheikh & Guercio, 1997). Recent estimates suggest the processes controlling the distribution of runoff and rainfall transport may be conditioned by local, often sub-field hydrology occurring at spatial scales in the order of less than 10m (Lane et al., 2004), thus highlighting the importance of having a DEM at a resolution which will capture these effects.

In this research the topographic data used were the Interferometric Synthetic Aperture Radar data produced by Intermap. These data sets are more commonly known as NEXTMAP data and are supplied at a resolution of 5 metres with a stated vertical precision of +/- 1m. The Intermap DEM was selected for use in this research due to it being suitable to apply to a catchment which incorporates areas of both high and low relief. Lidar data, at a resolution of between 1 and 2 metres, is unfortunately not currently available for the entire River Eden catchment and thus could not be used in this study.

5.1.2 Time Integrated Rainfall

The second data set that SCIMAP requires is a catchment-wide layer of time-integrated rainfall. This dataset is needed to calculate the dilution potential of runoff; the higher the ratio of water to pollution, the lower the concentration of pollutant and thus the lower the risk (Lane et al., 2006). As SCIMAP is focused on investigating spatial distributions of risk, it therefore requires a long-term, time integrated dataset of spatial rainfall distribution. The data set selected for use in this research was the 1961-2000 baseline, 5km griddled data of mean annual precipitation which is available from the UK Meteorological Office as part of the UK Climate Impacts Programme (Perry and Hollis, 2005). The format of the grid text files is the same as that used by the majority of GIS programmes and so files can be imported for analysis without any modifications being necessary.

5.1.3. Land Use Classifications

The combination of DEM and rainfall data enables SCIMAP to assign values of risk to areas based on the likelihood of them being hydrologically connected to nearby waterways. In order to calculate the potential of a unit of land producing a diffuse pollutant risk, a third layer of data is required: land use. In this research the Land Cover Map 2000 (LCM2000), supplied by the Centre for Ecology and Hydrology was used.

LCM2000 categorises land into one of forty-four broad habitat classifications as developed under the UK Biodiversity Action Plan. These are designed to represent the major surface types across the whole of the UK. Each location is assigned to one category depending upon what the land cover was at the time of the survey. Areas that are used for several different activities throughout the course of a year, or over a period of several years, may therefore not be accurately represented as the map relates to the cover at a certain time.

Data are derived from a computer classification of satellite scenes, obtained mainly from Landsat. The satellite sensors record spectral reflectance's from the Earth's surface, on a grid of approximately 25 m x 25 m cells, with each land area then being shown as a polygon. Research has shown that pollution dynamics are dependent upon small scale processes (Burt et al., 1999; Quinton & Catt, 2004) and in order to accurately identify the location of CSAs, processes must be modelled at the catchment scale whilst also capturing factors operating at the smaller, sub 10m scale (Lane et al., 2006). The resampling method used within the SCIMAP framework is thought to be an appropriate method of capturing land use as many areas will have the same land use over a far larger spatial area. Furthermore, as data from the agricultural census remains confidential at anything finer than catchment scale use of the LCM data is the most viable option for this research (Reaney et al., 2011).

Map accuracy of LCM2000 is estimated to be in the order of 80-85% (CEH, 2010); however the year of capture was 1999 and thus it is highly possible that land use may have altered between the year of capture and present. CEH has recently carried out a calibration of LCM2000 by comparing it with the CS2000 field survey. Generally the structural patterns shown on the LCM and those observed on the ground were similar. Where classes were found to differ, they usually show confusion between similar types: bracken and acid grass, rough and semi-natural grass, conifers and broadleaved trees (CEH, 2010). Although the differences in classes usually only occur between similar types this could potentially still have an effect on the risk estimations produced by SCIMAP.

5.2 Model Structure

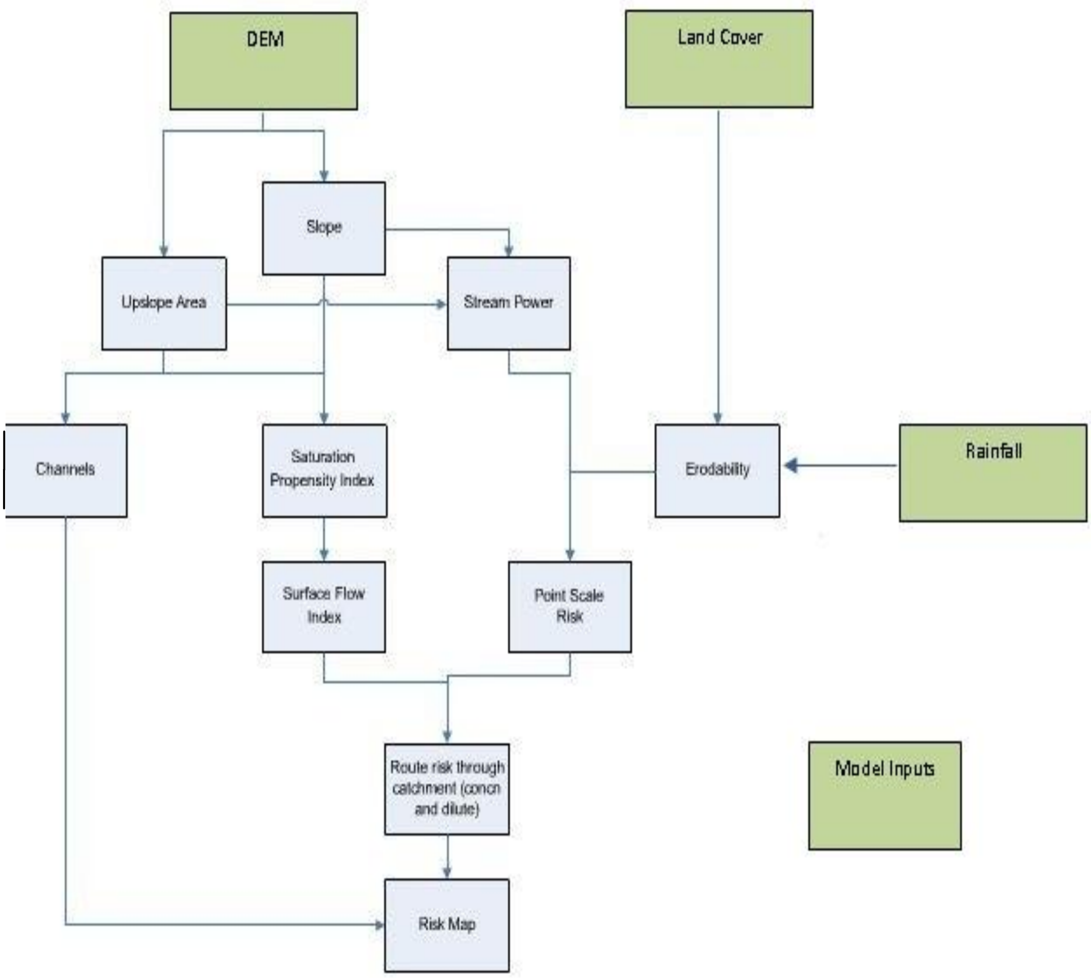


Figure 5.1: The SCIMAP model framework showing inputs and outputs. Source: Modified by Author. Original image available at www.scimap.org.uk.

5.3 Data Pre-Processing

Before the topographic data can be used within the already formulated SCIMAP framework, several stages of pre-processing are required in order to get the data into a suitable format for use. All initial stages of processing of the three data layers use the ERSI ArcGIS suite. ArcMap, the main component of the ArcGIS suite, is used to view, edit, create and analyse the geospatial data.

Firstly, it is necessary to fill the pits and sinks within the DEM surface to enable calculations of upslope contributing area (UCA) to be obtained. If the potential for dilution within the catchment is going to be modelled, then an understanding of flow pathways and the amount of water being delivered to an area from upslope is required. The Planchon-Darboux (Planchon & Darboux, 2001) algorithm is used to fill the depressions in the DEM and this new DEM is then used to obtain accurate flow routing pathways and upslope contributing areas. It should be noted at this point that later on in the modelling process, when calculating catchment slope, the original (non-filled) DEM is used. This ensures the important role that surface depressions and natural sinks play as areas of water storage and disconnection can still be considered (Reaney et al., 2011).

Once this initial stage of pre-processing has occurred, it is then necessary to clip the DEM to the catchment outline, meaning that streams and topography outside of the catchment boundary are not included in the analysis. A catchment buffer of 20 metres is applied to ensure that the watershed line is defined within the subsequent analysis.

Land cover and rainfall data are then imported into ArcMap along with the pre-processed DEM. At this point it is necessary to get the rainfall (5km resolution) and land cover data (30m resolution) into the same resolution as the DEM (10m). Ten metre resolution was deemed optimal for this project as it allows hydrological processes to be accurately represented whilst remaining at a sensible enough size so it can be handled by the computer hardware and software without any issues. Annually-averaged rainfall varies with altitude, slope aspect and distance from the coast in complex ways. It is beyond the scope of this project to produce detailed predicted maps of rainfall amounts and hence the rainfall data were resampled to 10m with a nearest neighbour algorithm to minimise the number of additional assumptions that are added to the model.

The SCIMAP framework is coded in the C++ language and has been integrated into a custom-made module in the System for Automated Geographical Analysis (SAGA) platform. SAGA, a freeware GIS framework, has excellent grid handling capabilities and comes with several key hydrological processing applications built into the program. Once pre-processing of the data has been completed, the three layer files are imported into SAGA GIS software. The SCIMAP module, built into the SAGA framework, then uses the initial three layers to produce a selection of new layers calculated using the topographic, land cover and rainfall inputs which include slope, stream power and erodibility risk. These layers are then exported back into the ESRI ArcGIS suite to manipulate the data for display purposes. Interpretation and viewing of the final SCIMAP risk maps is easier if these are overlaid onto Ordnance Survey 1:25,000 map tiles. Tiles for all areas of the UK are available from the Edina Digimap service and these were imported into ArcMap for viewing.

Traditional model calibration on the outputs produced by SCIMAP is not required as the model does not attempt to predict absolute pollutant delivery or account for temporal variations in risk. The overall performance of SCIMAP has, in the past, been principally assessed according to its ability to accurately explain the spatial variance in the distribution of salmonids (Reaney et al., 2011). Details of initial validation between results obtained from this study and past work using the SCIMAP methodology can be found in Chapter 6.

5.4 Process Limitations of the SCIMAP Framework

There is a significant amount of inherent uncertainty attached to any model prediction (Wainwright & Mulligan, 2004) and SCIMAP is no exception to this. Furthermore, the methodology behind the SCIMAP framework has several potential limitations attached to it. These are discussed below.

To calculate Network Index values across the catchment SCIMAP uses a deterministic flow routing equation, D8 (O'Callaghan & Mark, 1984) since overland flow will organise into a single flowpath at spatial scales below the 10 metre DEM resolution. It has been suggested that the D8 method oversimplifies water transport by turning flow routes into single straight lines. It is possible that using a multiple flow path approach, such as the the D-infinity method, would have been more accurate as it creates a dynamic representation of flow direction and avoids the unrealistic straight line pathways produced by the D8 model (Reaney et al, 2009) and this is an option that has now been built into the SCIMAP framework. However, use of a multiple flow path approach leads to the development of distribution functions of connection potential and hence requires a significant reformulation of the SCIMAP theoretical approach and was beyond the scope of this research.

SCIMAP cannot currently simulate non-topographic controls on connectivity which are not represented in the DEM, for example the presence of structural features such as hedgerows or roads which have been suggested to be import connectivity controls (e.g. Buttle, 2002). The impacts of such features upon hydrological connectivity, and hence the routing of risk across the landscape, is one of the key questions that emerges from the framework. In catchments where non-topographic controls on connectivity are evident, it is important to consider the accuracy of SCIMAP's risk estimations carefully.

SCIMAP assumes that for material for reach a waterway every cell between source and outlet must be capable of transporting overland flow (Reaney et al., 2011). The model thus assumes that the topographic wetness index can be used to describe the propensity of a given area to generate saturation-excess overland flow and works on the basis that the higher the network index, the greater the propensity for overland flow generation. Although this may be true, problems arise when attempting to map network index values into the duration of connection cycles. SCIMAP currently assumes that areas in the top 5% of the network index are always connected, those in the lowest 5% are never connected and those in-between have a linear relationship.

A forth limitation of the SCIMAP framework is that it does not account for groundwater inputs or storage. Away from the steep, headwaters of catchments most water that infiltrates through the layers of topsoil will percolate to the underlying groundwater. From here water will flow through the zone of saturation and eventually reach the main stream channels (Ward & Robinson, 2005). Although fine sediment will not be transferred via ground water systems, groundwater inputs into waterways may help in diluting levels of pollutants and hence risk may be less than first thought.

The framework does not differentiate between areas of different geology or soil type, both of which may significantly alter connectivity dynamics (Ward & Robinson, 2005). Research has shown that even very subtle differences in soil composition can quickly lead to variations in levels of infiltration and overland flow. On sandy soils, which are known to have large pore spaces, greater amounts of infiltration are likely to occur hence lower levels of surface runoff and erosion can be expected (Foth, 1990). If the area in question is a particularly critical hotspot of pollution, as result of other factors such as it being on a steep slope or having a high number of rills and gullies, then higher rates of infiltration may not be observed. However, there is a significant relationship between the soil type and the land cover and hence some implicit details on the spatial pattern of soil types is captured within SCIMAP. Soil type information could be incorporated into the approach through the development of a combined land cover and soil type risk layer. However, in this application, this approach has not been taken.

Particle size is also important when considering erodibility of soils and in sandy areas, which have larger particles, the risk of erosion is lower as more energy is required within overland flow to move particles (Foth, 1990). Variations in soil properties occur across landscapes and fields at fine spatial scales. Although such variations may be important to connectivity dynamics we currently do not have access to maps of these properties and thus the SCIMAP framework attempts to construct connectivity pathways based on the best available data.

The simple manner in which risk loading is transferred into risk concentration using the upslope contributing area has been highlighted as an area of concern within the framework (Reaney et al., 2011). SCIMAP works on the basis that the risk at any given point is the accumulation of all upstream risks that feed into that point. However, it is important to consider the effects of deposition and re-mobilisation within channels on top of this. As it is impossible to quantify this accurately across a landscape, deposition is assumed to be relatively small and hence no correction is made for loss of risk.

The standard SCIMAP framework is unable to account for temporal variations in risk due to the framework being based on a land cover map that represents only one point in time. SCIMAP assumes that land use stays constant over the course of a year. However, in reality this is clearly not necessarily the case as crop cover will vary between seasons. Hence, to overcome this weakness with the SCIMAP approach and to enable an assessment of monthly variations in risk, a temporally integrated framework has been created as part of this research. This is discussed in greater detail in Section 5.5.

The SCIMAP framework aims to make the best possible use of current available datasets. Once initial risk mapping has been formulated using the SCIMAP approach this can then be used to target field work at areas thought to be most at risk of fine sediment pollution and a more in-depth analysis of risk at these locations can be carried out. Information found as part of the field work process can then be incorporated back into the SCIMAP framework to help improve the accuracy of its outputs.

5.5 The Temporal Risk Integrated Platform (TRIP)

The Temporal Risk Integrated Platform (TRIP) is a user-friendly risk generator designed to capture fine sediment erosion risk at a given spatial point within a catchment. The TRIP model works on the basis that fine sediment erosion risk within a catchment is a product of the erosive force of the rainfall, the protection to the soil surface offered by vegetation cover and the natural erodibility of the soil. In areas where large proportions of the ground are covered by crops for long periods of the year, less rainfall will hit the soil whilst travelling at terminal velocity. Therefore the erosive power of rainfall is less and the risk of erosion of topsoil is reduced. Where fields are left bare for large parts of the year, rainfall will not be intercepted by vegetation and any particles hitting the soil will have high erosive power. TRIP is able to capture both temporal variability in fine sediment erosion risk as a result of changing crop cover and farming practices whilst at the same time having the capacity to simulate the effects that future rural land management and projected climate change may have on risk levels.

5.5.1 The TRIP Framework: Pre-processing

The first stage of developing the TRIP framework required research into the five key variables that control fine sediment erosion risk at any given location within a catchment. The variables that are incorporated into the TRIP framework are: 1, average monthly rainfall, 2, rainfall occurring in runoff-driving events, 3, percentage monthly crop cover, 4, baseflow index and 5, soil resistance. In order to obtain reliable estimates of these five controlling factors, several stages of pre-processing within the framework were required before the final risk filter could be produced. The methodologies used in this research to determine values for each of these factors are described below.

5.5.1.1 Average Monthly Rainfall

In order to determine the erosivity risk at points within the catchment daily precipitation records, obtainable from the UK Met Office via The British Atmospheric Data Centre (BADC), were collected for Appleby for the period 1891-2010. Data from BADC is a single measurement of total rainfall (mm) for each 24 hour period. Once daily precipitation records had been gathered these were re-sorted into monthly groupings and average monthly precipitation for each location was determined. Values for average rainfall for each month were then inputted into the first section of the TRIP risk framework.

5.5.1.2 Runoff-driving Storms

Using an average monthly rainfall value to calculate erosion risk would result in an inaccurate representation of risk as not all rainfall will result in fine sediment erosion. Instead, in order to determine the erosive power of rainfall, precipitation needs to be split into two categories: 1) intense rainfall falling in storm events, which is likely to be erosive and 2) less intense rainfall. To distinguish between the two, daily rainfall records as opposed to monthly averages are required. The longest of these daily records, held by the Meteorological Office, is for Appleby-Castle (monitoring station number: 12936) where daily precipitation totals have been recorded since 1891 (Burt & Ferranti, 2010).

It has been found that fine sediment erosion is related to two types of rainfall event: 1) short intense storms and 2) prolonged storms of slow intensity that saturate the soils. It is difficult to separate the effects of these two types of rainfall (Morgan, 1979) and hence for the purpose of this study a single threshold above which it was assumed particulate transfer would occur, was used. When creating the TRIP framework it was decided that the precipitation threshold would be set at 15mm. This threshold has been widely suggested as being the point at which runoff and rilling begin on arable land (Burt & Horton, 2007; Osborn et al, 2000; Speirs and Frost, 1985). Although seemingly relatively low, enthusiasm for extreme events must not blind us to the fact that thresholds at which runoff and fine sediment transfer begin *are* low especially on bare land. There are many examples of serious erosion occurring as a result of relatively modest rainfall amounts and hence it is important to ensure the threshold is set at an appropriate level so as to capture this.

Further support for the theory that fine sediment transport will increase considerably once rainfall has surpassed a 15mm threshold was provided by daily rainfall and turbidity measurements taken from a gauging station established as part of the EdenDTC project at Sewborwens Farm, Newton Rigg, Cumbria. Turbidity is a measure of the decrease in the transparency of a solution due to the presence of sediment particles, coloured organic matter and the water itself which causes incident light to be scattered, reflected and attenuated (Ziegler, 2002). It is the most widely used surrogate for measuring suspended sediment concentration (Gray and Gartner, 2009). Similar relationships can be observed in data collected from Broadway Foot monitoring station in the River Rye catchment, North Yorkshire (Perks, 2012, *unpublished PhD thesis*).

Using rainfall data collected at the Appleby measuring station since 1891 (Burt & Ferranti, 2011) the number of rain days where total rainfall in a 24 hour period exceeded 15mm were calculated. The percentage of rain days that these storm events accounted for on an annual basis was then determined. To assess the effect that these storm events have at varying times of the year it was necessary to establish the number of events within each calendar month. Using a filter command in Microsoft Office Excel 2007, the average number of storm events in each calendar month where rainfall exceeded 15mm was calculated. From this the percentage of annual storms occurring in that month was determined which enabled the calculation of the total amount of rain falling in storm events within each calendar month.

5.5.1.3 Monthly Percentage Crop Ground Cover

To establish the degree of protection that soils have from rainfall, the percentage of crop cover throughout the course of the year for various different crops was investigated using a novel approach based on the collection of aerial photographs from a remote controlled helicopter. Aerial photographs of a variety of crops were collected using a Microdrones MD4-200 remote controlled helicopter with a Pentax Optio A40 12 megapixel digital camera attached (Figure 5.2a and Figure 5.2b). The digital camera selected for use has a built in shake reduction function meaning the camera can effectively reduce shaking and any associated blurring when taking still images. The fast shutter speed of the camera means the risk of blurring is further reduced. Ensuring images would be of a sufficient quality to enable them to be processed to assess patterns of vegetation cover was one of the key considerations when planning this research and the Pentax Optio A40 was selected because of its proven track record of producing high quality, clear images.

Each of the images collected was taken at a height of approximately 50 metres and images were collected on days when skies were mostly bright and clear to ensure the quality of the image was the best possible. Aerial photographs were taken at the beginning of each calendar month to assess how crop cover changed over the course of a year.

Figure 5.2a: Microdrone MD4-200 helicopter.

Figure 5.2b: 12 megapixel digital camera

These pictures were then analysed, using MATLAB programming software to determine the percentage of the ground covered with vegetation. MATLAB is a computer programming language which allows the user to create individual programmes suited to their needs. By producing a set of new code within the MATLAB software (Appendix 2) it was possible to manipulate the collected images to obtain values for areas of the picture that were vegetated and areas that were not.

Firstly, an image, in JPEG format, is loaded into the MATLAB user interface. The image is then resampled based upon coordinates showing areas of vegetation and bare ground that are input into the code sequence created as part of this research. Markers, showing the noted areas of vegetation and bare ground are then displayed on the image before a nearest neighbour classification is run and a colour label assigned to each pixel of the original image. The label matrix is then used to separate objects in the original image by colour in order to obtain total pixel numbers for vegetated and non-vegetated ground. By calculating the overall number of pixels per image the percentage of crop covered ground and bare ground can be determined. A copy of the MATLAB coding created for use in this research can be found in Appendix 1.

The EdenDTC project, in partnership with the University of Cumbria, has an access agreement arranged with Sewborwens Farm at Newton Rigg Campus near Penrith. Therefore aerial photographs were gathered from this location. Land use around Sewborwens Farm and the Newton Rigg campus is similar to land use at Appleby and other test sites. Hence, the images gathered from Sewborwens Farm and the calculations of percentage crop cover based on them were deemed to be an accurate representation of crop cover and conditions at the other test sites around the River Eden catchment.

To assess the accuracy of the approach created here using the Microdrone and image processing code, field surveys were also carried out on the days when aerial photographs were carried to manually calculate percentage vegetation cover. A 30m tape measure was used to mark out a sample area in each field. Ground cover, at 10cm intervals, was then recorded for the 30-metre section. To ensure consistency, the same person was responsible for recording ground cover for the entire 30-metre sample area, and for a point to be classed as having crop cover a stem or fully developed leaf had to be present directly at the point. This process was repeated twice in each of the study fields and aerial photographs were collected directly above the study sites so comparisons between the field surveys and MATLAB processing results could be obtained.

5.5.1.4 Soil Resistance

All soils exert some degree of resistance to erosive powers and hence, when modelling potential rates of fine sediment erosion, it is necessary to be able to quantify this. The Universal Soil Loss Equation κ factor was used to determine the resistance of soil types across the study catchment. The κ factor is a measure of the susceptibility of soil particles to detachment and transport by rainfall and runoff. The factor reflects the fact that different soils erode at different rates depending upon their characteristics (Kirkby & Morgan, 1980). Texture is the principal factor affecting κ, but structure, organic matter and permeability also contribute (USLE, 2011).

Measurements of soil resistance were taken by the USLE at 23 sites across North America. From these initial measurements values of κ for other soil types were inferred and approximated by comparing the characteristics of soils with the characteristics of the measured soils (Morgan, 1979). Although direct measurements would be a more reliable way of establishing erodibility, these require considerable time and are costly to perform at the landscape scale at the resolution required by SCIMAP. Hence, the approximated κ values for each soil type obtained by the USLE study were deemed to be acceptable for use in this research.

As discussed in Chapter 2.4 there is a large degree of variability when considering the erodibility of soils. Soils with high silt content are the most erodible of all soils. They are easily detached, tend to crust and produce high rates of runoff; κ values for these soils tend to be greater than 0.4. Conversely, soils with high clay content have low κ values, about 0.05 to 0.15, because they are resistant to detachment (Morgan, 2005).

Within this research κ values for the test locations were determined based upon soil type and structure information obtained from the National Soil Map of England and Wales (NATMAP), created by the National Soils Research Institute at Cranfield University. NATMAP is a 1:250,000 scale map showing soil types as classified into one of 297 distinct soil associations (LANDIS, 2011). These soil associations are then sub-divided into various soil series.

For each study site within the River Eden Catchment the dominant soil type was determined by analysing a 1:250,000 NATMAP vector data set with the catchment outline and GPS points showing study locations imposed on top. In order to represent the resistance to erosion that soils have, the in-situ erosion risk as calculated in the previous stage of the framework, is multiplied by one minus the resistance factor.

Although a κ factor was selected in this research to represent soils in their natural condition, past management or the misuse of soil by intensive cropping could increase a soil's erodibility. The κ factor may therefore need to be increased if the soil structure has been destroyed or organic content has been depleted.

Textural Class	κ Value
Clay	0.22
Clay Loam	0.30
Coarse Sandy Loam	0.07
Fine Sand	0.08
Fine Sandy Loam	0.18
Heavy Clay	0.17
Loam	0.30
Loamy Fine Sand	0.11
Loamy Sand	0.04
Loamy Very Fine Sand	0.39
Sand	0.02
Sandy Clay Loam	0.20
Sandy Loam	0.13
Silt Loam	0.38
Silty Clay	0.26
Silty Clay Loam	0.32
Very Fine Sand	0.43
Very Fine Sandy Loam	0.35

Table 5.1: κ values for various soil types as classified in the Universal Soil Loss Equation.

5.5.1.5 Base Flow Index

Base flow index (BFI) is defined as being the amount of river flow which is derived from groundwater sources rather than surface runoff. BFI is usually scaled between 0 (0%) and 1 (100%). The more permeable the rock, superficial deposits and soils in a catchment, the higher the baseflow (Ward & Robinson, 2005). Hence, areas that have a greater baseflow index will have lower fine sediment transport risk. In a study such as this, it is necessary to incorporate estimates of baseflow into the modelling framework to represent rainfall which is likely to percolate into groundwater stores rather than becoming overland flow. This incorporation of the BFI into TRIP partly addresses one of the listed limitations with SCIMAP through the use of spatial soil information.

For each study site within the River Eden Catchment, the dominant soil type was determined by analysing a 1:250,000 HOST vector data set with the catchment outline and GPS points showing study locations imposed on top. Baseflow index values were inferred as a method of categorising whether areas would have high or low permeability. The HOST classification is based on a number of conceptual models that describe dominant pathways of water movement through the soil and, where appropriate, substrate. Once these values had been established, the total risk value is multiplied by one minus the baseflow index in order to reflect the amount of water that will percolate into groundwater as opposed to becoming overland flow.

5.6 Model Programming

Once all necessary data required to run the TRIP framework had been assembled, they were then made into a Microsoft Office Excel Spreadsheet format where all necessary calculations could be run (Figure 5.3).

1. Firstly, average monthly rainfall was divided into two categories; rain that falls in storm events of above 15mm and rain that falls in events of less than 15mm. From this the percentage of total annual storm events occurring in each calendar month was calculated by dividing the amount of rain falling in events of more than 15mm each month by the sum of all rainfall falling in events greater than 15mm over the course of the entire year.

2. Once the percentage of storms has been established for each calendar month this is multiplied by the average crop cover for that month to give an estimate of fine

sediment erosion risk. Months which have the greatest storm levels combined with lowest crop covers thus have the highest fine sediment erosion risks.

3. Now that fine sediment erosion has been predicted for each month these are added together and divided by 12 (total number of months) to determine the average overall risk for the location on an annual basis.

4. The last stage of the risk assessment process is to incorporate the two constants that represent soil resistance and baseflow index. The overall annual fine sediment erosion risk is therefore multiplied by previously determined values for these two factors.

5.7 Simulating Crop Cycles

As discussed in Chapter 5.2.1.3, average crop cover over the course of the year for several crops commonly grown across the UK was estimated using aerial photography. Within the River Eden catchment, a variety of crops are grown, each of which is planted and harvested at different times of the year and grows at a different rate. The amount of time arable land is bare and the growth rates of crops are both known to effect fine sediment erosion risk and hence it was important to ensure that harvesting and sowing times were accurately represented in the TRIP framework. Based on information obtained from the Farming Handbook (SAC, 2010), the following dates for sowing and harvesting were used in this research.

Crop	Planted	Harvested
Winter Wheat	October	August
Winter Barley	September	July / August
Oilseed Rape	August	July
Grass	Various	Various
Spring Barley	March / April	September
Maize	May	September / October

Table 5.2: Normal sowing and harvesting dates for a selection of crops grown in the UK. Source: Farming Handbook, SAC (2010).

Fine-sediment erosion risk at a point will be affected, not only by the crop cover of the year in question, but also by past land use and land cover. For this reason it was important to reproduce land cover cycles that are an accurate representation of farming practice within the River Eden catchment. Six typical land use cycles, based upon information from The Eden Rivers Trust (Cleasby, per. comm.) and the Farming Handbook (SAC, 2010), were initially inputted into the TRIP framework (Table 5.3). Commonly, crops planted around the UK follow a five-year cycle and so the same length cycle was used here.

Simulation	Year 1	Year 2	Year 3	Year 4	Year 5
1	Oilseed Rape	Winter Wheat	Winter Barley	Winter Barley	Oilseed Rape
2	Winter Wheat	Winter Wheat	Winter Barley	Grass	Grass
3	Grass	Grass	Grass	Grass	Grass
4	Grass	Grass	Grass	Winter Wheat	Winter Barley
5	Maize	Maize	Maize	Maize	Maize
6	Spring Barley	Spring Barley	Spring Barley	Spring Barley	Spring Barley

Table 5.3: Typical crop cycles used across the UK.

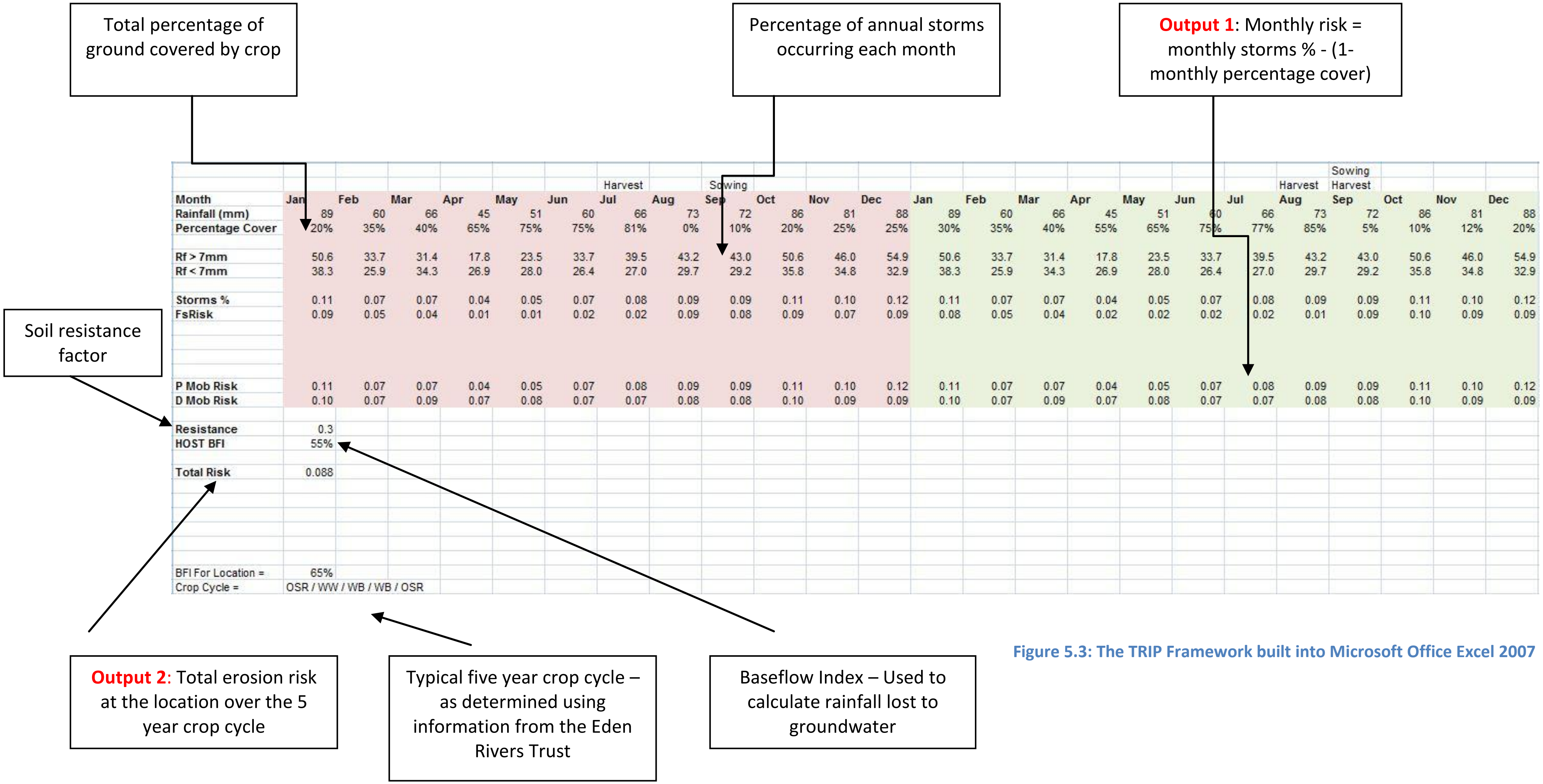
Total risk values were obtained for each of the standard crop cycles in order to determine which of the current common cropping cycles is likely to be causing the largest fine-sediment erosion risk and which is likely to result in the lowest risk. Once predicted levels of fine-sediment erosion risk had been found for each of the crop cycles, the optimum cycle which minimises risk to the greatest extent, taking into consideration social, economic and environmental factors that all impact upon management and policy, was identified.

Once risk values had been obtained for each of the original crop cycles alterations to the cycles were made to incorporate possible future land use change. This included reducing the amount of autumn-sown crops grown in an area, changing the type of crop grown and varying the 5 year crop cycles to predict the effect that a crop could have on fine sediment risk in the years after it has been harvested.

5.8 Conclusions

Fine-sediment pollution risk is closely linked to hydrological connectivity and land management at both the field, farm and catchment scale. This chapter has introduced an established framework, SCIMAP, which can be used to identify areas within a test catchment that are currently at high risk of being critical source areas of diffuse pollution. Once an initial understanding of the current hydrological connectivity pathways and critical source areas of diffuse pollution within the test catchment have been identified, these are combined with a newly created temporal risk framework to assess how fine sediment pollution risks may vary at a given location over a given time period.

Along with being able to assess fine-sediment erosion risk on a temporal scale, the TRIP framework has been designed to be able to identify current risks of fine-sediment erosion associated with various crops which are widely grown across the study catchment. Similarly, the framework enables analysis of the effects that five-year crop cycles have on fine-sediment erosion risk and how farm management, in the form of alterations to these cycles, could help to mitigate against erosion risk. Because of the flexible nature of the TRIP framework the effects that changing patterns of hydrological connectivity, as a result of climate change, may have on diffuse pollution can be simulated and assessed.



							Harvest		Sowing			
Month	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Rainfall (mm)	89	60	66	45	51	60	66	73	72	86	81	88
Percentage Cover	20%	35%	40%	65%	75%	75%	81%	0%	10%	20%	25%	25%
Rf > 7mm	50.6	33.7	31.4	17.8	23.5	33.7	39.5	43.2	43.0	50.6	46.0	54.9
Rf < 7mm	38.3	25.9	34.3	26.9	28.0	26.4	27.0	29.7	29.2	35.8	34.8	32.9
Storms %	0.11	0.07	0.07	0.04	0.05	0.07	0.08	0.09	0.09	0.11	0.10	0.12
FsRisk	0.09	0.05	0.04	0.01	0.01	0.02	0.02	0.09	0.08	0.09	0.07	0.09
P Mob Risk	0.11	0.07	0.07	0.04	0.05	0.07	0.08	0.09	0.09	0.11	0.10	0.12
D Mob Risk	0.10	0.07	0.09	0.07	0.08	0.07	0.07	0.08	0.08	0.10	0.09	0.09
Resistance	0.3											
HOST BFI	55%											
Total Risk	0.088											
BFI For Location =	65%											
Crop Cycle =	OSR / WW / WB / WB / OSR											

									Sowing			
								Harvest	Harvest			
Month	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Rainfall (mm)	89	60	66	45	51	60	66	73	72	86	81	88
Percentage Cover	30%	35%	40%	55%	65%	75%	77%	85%	5%	10%	12%	20%
Rf > 7mm	50.6	33.7	31.4	17.8	23.5	33.7	39.5	43.2	43.0	50.6	46.0	54.9
Rf < 7mm	38.3	25.9	34.3	26.9	28.0	26.4	27.0	29.7	29.2	35.8	34.8	32.9
Storms %	0.11	0.07	0.07	0.04	0.05	0.07	0.08	0.09	0.09	0.11	0.10	0.12
FsRisk	0.08	0.05	0.04	0.02	0.02	0.02	0.02	0.01	0.09	0.10	0.09	0.09
P Mob Risk	0.11	0.07	0.07	0.04	0.05	0.07	0.08	0.09	0.09	0.11	0.10	0.12
D Mob Risk	0.10	0.07	0.09	0.07	0.08	0.07	0.07	0.08	0.08	0.10	0.09	0.09

Figure 5.3: The TRIP Framework built into Microsoft Office Excel 2007

Chapter 6: Application of SCIMAP to the River Eden Catchment

The results of applying the SCIMAP model to the River Eden catchment are presented in this chapter. The purpose of this application was to enable an understanding of the current hydrological connectivity dynamics and associated fine sediment erosion risks within the catchment to be developed.

Initially the intermediate layers of the SCIMAP modelling process are analysed independently of each other, this allows areas susceptible to soil erosion and locations with highly erosive rainfall to be identified. At the same time SCIMAP is able to determine, based upon Network Index values, the likelihood of hydrological connection at each point across the landscape. Once these three intermediate modelling layers have been combined together SCIMAP makes predictions, based on topographic controls, of in-situ fine sediment risk at each location across the catchment. By combining this risk with land cover risk weightings the important role played by different surface covers in increasing or decreasing erosion risk can be identified and areas of particularly high risk highlighted. The last stage of the modelling process, before an overall risk output can be calculated, is to incorporate the effects of dilution. The results of each of these stages of the modelling process are presented and discussed below.

6.1 Energy Available for Soil Erosion

As discussed in Chapter 2, in order for a location to be a fine-sediment pollution risk, the area must be both at high risk of erosion and also be well connected to the surrounding area so material can be transported away from its source. Figure 6.1 shows the predictions of the amount of energy available to erode the soil surface across the River Eden Catchment based on a locations topography and average annual rainfall. This layer is produced as part of the SCIMAP modelling process and highlights areas where the likelihood of soil erosion occurring is high. The amount of available energy is calculated based upon stream power index:

$$p = a.\tan(b)$$

Equation 6.1: Stream Power Index

where p is the stream power, a is the rainfall weighted upslope contributing area and b is the local slope gradient. This index therefore captures how much water is available to do erosive work and the likely overflow flow velocity and hence the amount of energy available.

Erosion risk is scaled between 0 (low erosion; green) to 1 (high erosion; red). This data layer represents the potential for erosion based solely on the topography and rainfall, no consideration is given to vegetation cover or land use, this is added to the model later. This data layer can be seen as a map of where in the catchment is most sensitive to the land management and hence where in the catchment is most sensitive to changes in management practices.

Figure 6.1 shows that areas with the highest amounts of energy available for soil erosion can be found in the upland areas of the Pennines and Lake District Hills. These regions experience very high levels of annual precipitation, as discussed in chapter 3, and are characterised by their steep slopes. Rainfall is therefore channelled off the hillslope very quickly with higher erosive potential. Although this area is known to be well vegetated this is not considered at this stage of the modelling process hence the upland areas of the Lake District and Pennine hills may not have the highest in-situ erosion risk but they do have the highest erosive potential. Further downstream, in the valley bottoms of the catchment, gentler slopes mean water is not routed off the landscape as fast and has a lower erosive potential. The longer slope profiles identifiable in these areas ensure there is an increased chance of infiltration occurring between the area where runoff begins and the receiving waterway. Currently, SCIMAP does not account for geological controls on infiltration; hence this possibility cannot be incorporated explicitly into the modelling framework. However, the geological effects are captured later in the processing due to the correlation between soil type and land cover.

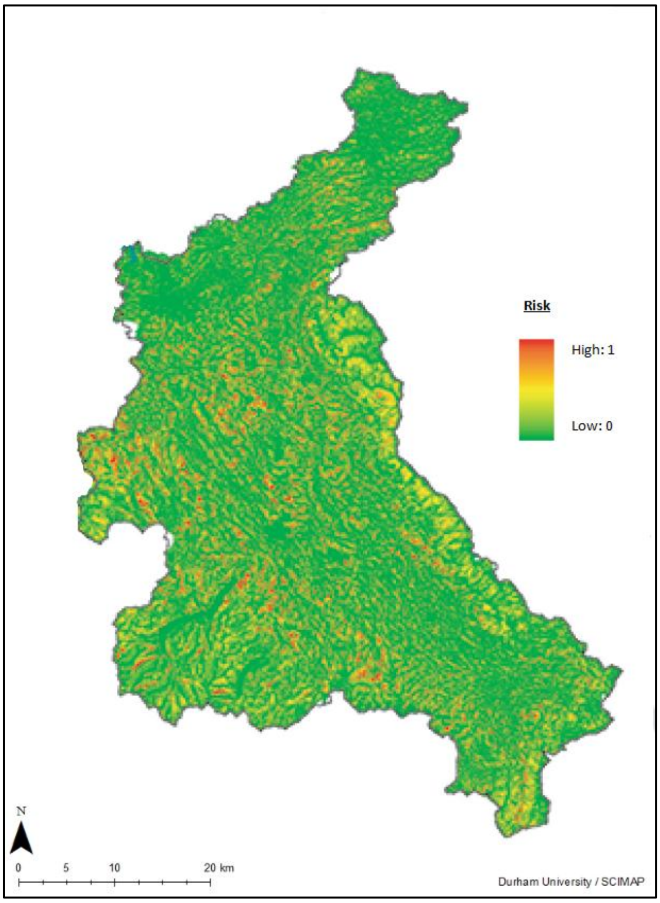


Figure 6.1: Stream Power index in the River Eden catchment. This is a representation of the total amount of energy available at a location which can be used to drive soil erosion.

6.2 Surface Hydrological Connectivity with the Network Index

As discussed in Chapter 2, areas within a catchment only become critical source areas of diffuse pollution if the pollution source can be connected to a receiving waterway. Figure 6.2 shows a Network Index map for the River Eden catchment. This is a representation of relative surface connectivity risk, highlighting areas where the likelihood of connection between the point and the river channels are high. For each point in the landscape, the probability of continuous flow to the river channel network is assessed. This map is achieved through the prediction of the spatial pattern of soil moisture and hence the susceptibly of each point in the landscape to generate saturated overland flow and the flow pathway between the point and the river channel. The Network Index is the lowest value of the wetness index along the flow pathway and hence is the catchment wetness required for a point to generate overland flow and for that water to be able to reach the river channels along a set of points that are also generating, and hence transmitting, overland flow. The Network Index values are taken as relative within the catchment and rescaled to give a probability of connection between 0 (low connection; red) and 1 (high connection; blue). This rescaling assumes that the most connected 5% are always connected and the least connected 5% of the landscape never connect. Between these two points, it is assumed that there is a linear scaling. The details of this re-mapping of the Network Index to connection probability are refined in Chapter 8 where the potential impacts of climate change on the connectivity are investigated.

The Network index shows areas in the wide, gently sloping landscapes of the western side of the catchment have the highest rates of surface connectivity. The concave nature of the slopes in this region makes them prone to saturation, hence during storms events they quickly become active and saturation excess runoff is a common occurrence. Similarly, the long, gently sloping profile of many of the valley sides in the region allows a greater accumulation of surface runoff from their large upslope contributing areas.

In the steeper Pennine hills to the eastern edges of the catchment and in the south-west where the catchment borders the Lakeland fells areas are much less connected. Rainfall falling here will have little time to infiltrate due to the high speeds at which it will flow away laterally under gravity, hence propensity to saturation excess runoff is lower. Although this is not accounted for within the Wetness Index approach it is possible, at times of heavy rain, that these areas may produce large amounts of surface runoff via infiltration-excess overland flow.

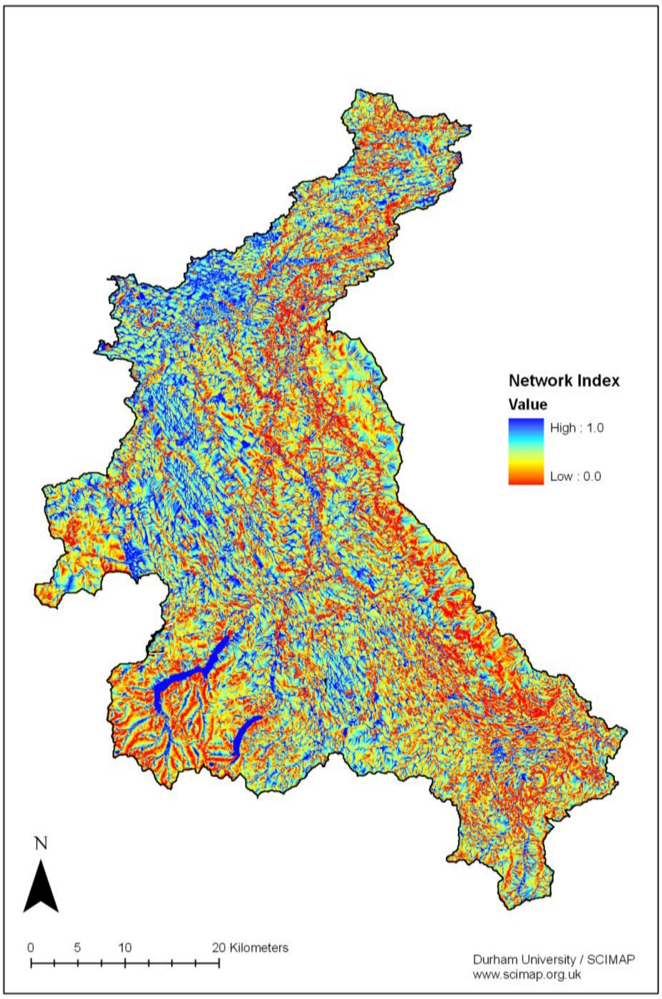


Figure 6.2: Network Index values for the River Eden Catchment. The Network Index is used as a representation of surface connectivity risk.

6.3 Diffuse Fine Sediment Erosion Risk with Land Cover Weightings

The initial results obtained from the SCIMAP framework which have been discussed so far in this chapter have considered fine-sediment erosion risk based purely on topographic factors. When land cover weightings are added into the SCIMAP framework, and erosion risks reanalysed, the location of areas most at risk of fine-sediment pollution may change. Each land cover classification was scaled between 0 (low risk) to 1 (high risk) based upon its potential for producing soil erosion. Before land cover weightings are applied the areas deemed to be most at risk are those in the highest headwater parts of the catchment where high risks of connectivity combine with high rates of energy available for erosion. After the integration of land cover weightings into the SCIMAP modelling process, the focus for concern shifts to the central lowland parts of the main River Eden valley.

Land in the upper parts of the catchment is predominantly used as rough pasture and large segments are covered with shrub and bracken. This low-intensity activity means land is not ploughed or trampled by cattle and is rarely disturbed. Such land covers therefore have a much smaller risk of erosion associated with them and in the SCIMAP framework they are assigned lower risk values. In contrast, land in the lowland parts of the River Eden valley is predominantly improved grassland or used to grow arable crops meaning it is regularly ploughed, left bare for large proportions of the year and will be intensively managed. Such activity is likely to result in increased levels of soil erosion and so these locations are assigned higher land use risk weightings within SCIMAP (Table 5.1).

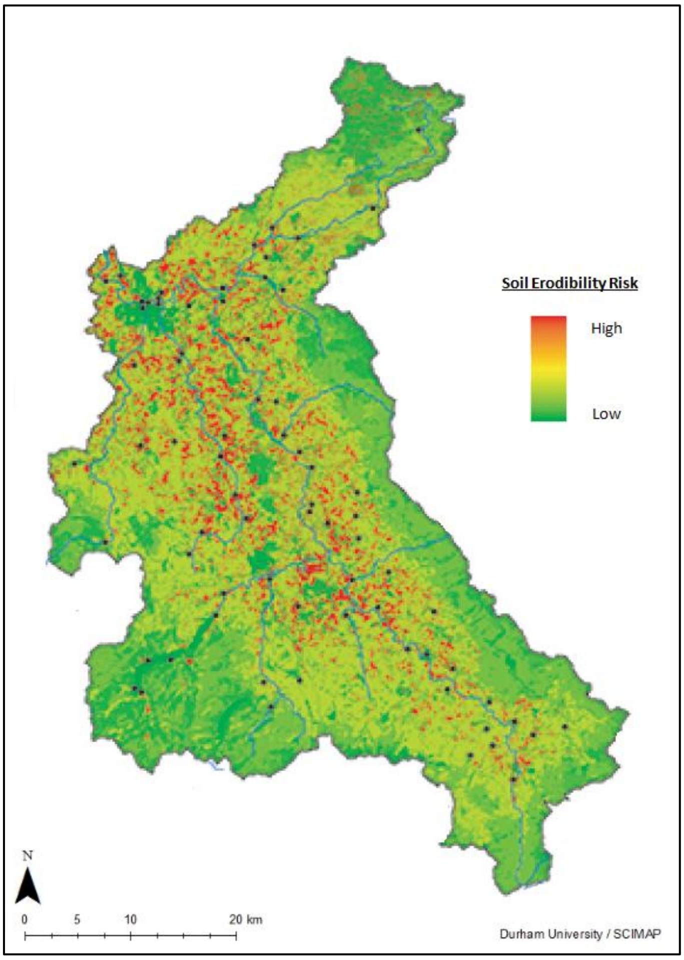


Figure 6.3: In-situ fine sediment risk based on land cover weightings.

6.4 Dilution Effects

Risk at a location may not necessarily be transferred through the hydrological network to receiving areas and it is possible that a degree of risk may be diluted as a result of inputs of water from a large upslope contributing area. The effects of this possible dilution are incorporated into SCIMAP's risk predictions by dividing the accumulated fine sediment risk by the rainfall weighted upslope contributing areas (Figure 6.4). The scale bar runs from red (high risk) through to green (low risk) in classes of one third of the standard deviation of the mean of the risk value. In areas where the standard deviation value is higher than the mean (red areas), the risk of pollution is increasing faster than it can be diluted and thus the area is identified as having a higher risk input associated with it. Conversely, in locations where the standard deviation is less than the mean, the risk of pollution is less than the dilution potential and thus the area is classified as having lower risk inputs to the channel.

6.5 Final Risk Outputs

Areas which are highlighted by SCIMAP as being at greatest risk of diffuse pollution, once the effects of land cover and dilution are taken into consideration, are mainly located in the central parts of the catchment (Figure 6.4). The low-lying agricultural areas that surround the main channel of the River Eden and River Petteril between Penrith and Carlisle are considered to be of particular high risk. Other notable “hotspots” can be observed to the north-east of Kirkby Stephen and on the far north-western fringes of the catchment. As discussed previously, arable and intense pastoral farming can be found in these areas and this is the main driver of high risk at these locations.

Small hot spots of risk, as a result of a degree of horticultural activity across the area, are identified by SCIMAP to the north-east of Kirkby Stephen. Similar hotspots can be seen at Appleby-in-Westmorland. In the north-west of the catchment, following the channel of the River Caldew, patches of high risk can again be seen. Land use in this area is predominantly improved grassland which is unlikely to contribute to heightened risk, unless it is very intensely grazed. However, the area does have a smaller upslope contributing area due to it lying on the fringes of the catchment and hence there is little chance for the dilution of pollutants.

Locations with low risks are found on the south-western edges of the catchment on the Lakeland fells around Swindale Beck and in the east where the main river channel flows down through the Pennine Hills. Although these areas have high levels of erosive rainfall and steep slopes, they also have low risk land uses which substantially minimises the risk of fine sediment erosion.

Areas with the lowest risk are identifiable as being in the upper reaches of the north-west of the catchment where the River Irthing flows down off the Pennine Hills towards Carlisle. Further downstream, along the main channel of the Irthing, risk can be seen to increase substantially around the point where the River Irthing meets its confluence with King Water. This increase in risk is likely to be a result of the change in land use between the upper parts of the River Irthing catchment where most areas are covered by bracken and land around King Water which is predominantly arable based land covers.

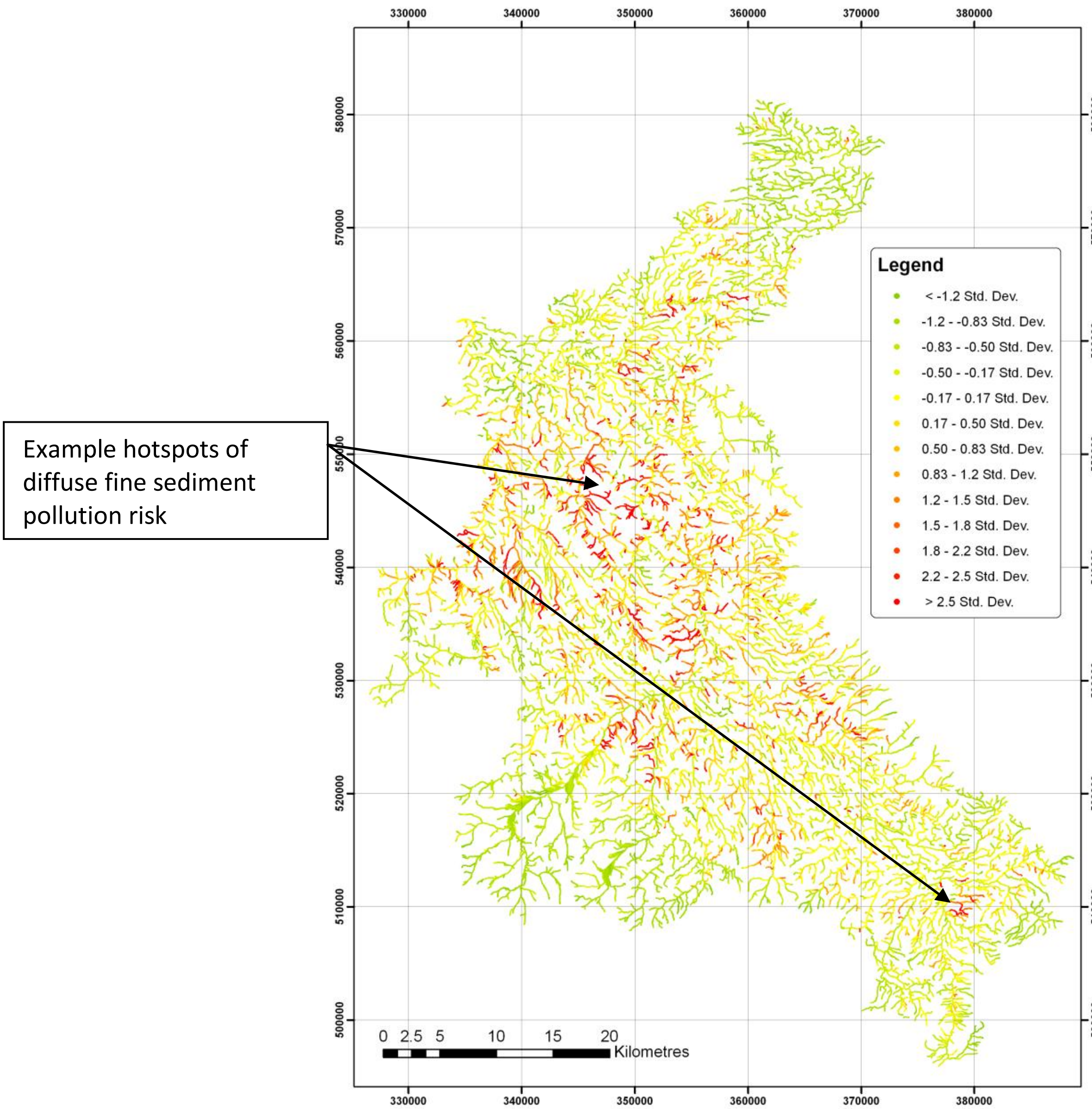


Figure 6.4: Final SCIMAP fine sediment erosion risk output for the River Eden Catchment.

6.6 SCIMAP Performance Assessment

The results obtained from applying the SCIMAP framework to the River Eden catchment are compared with spatially distributed ecological and water chemistry data gathered from the River Eden catchment. This allows us to determine the performance of SCIMAP in making spatial predictions and whether correlation can be identified between areas deemed to be most at risk of fine sediment pollution by SCIMAP and records of salmon and trout fry abundance (Reaney et al., 2011). SCIMAP has also been assessed against spatial patterns of water quality based on the Environment Agency general quality assessment (GQA) data (Milledge et al., 2012 in press) and gravel siltation mapping (Dugdale, 2007, *unpublished PhD thesis*).

6.6.1 Electrofishing Data

The Eden Rivers Trust has been carrying out electrofishing surveys throughout the River Eden catchment since 2002, as a method of assessing the status and distribution of juvenile salmonids (salmon and trout fry – aged less than one year). Catchment wide studies have now taken place at over 200 sites (Eden Rivers Trust, 2009).

Focusing on salmonid fry abundance is a valuable tool for comparison with the SCIMAP framework predictions for two reasons. Firstly, salmonid fry abundance is strongly influenced by exposure to local conditions, and hence contain a spatial signal that reflects the spatial variability in these local conditions. Secondly, processes that are associated with the surface transport of eroded material, including fine sediment, are thought to impact upon the recruitment of salmonid fry. For example low fry counts have often been attributed to fine sediment infiltrating into spawning gravels (Soulsby et al., 2001). The records of abundance of fry obtained from the electro-fishing data will reflect potential limits upon fry numbers and hence can draw attention to areas where it is possible fine sediment may be creating a risk. This spatial pattern can then be compared to the SCIMAP predictions.

Two disadvantages of using this approach have been identified. Firstly, the abundance of salmonid fry will not just be affected by fine sediment pollution and there may be other potential limits upon numbers. Therefore data may contain a signal that is driven by other factors. Furthermore, surveys have only been carried out on semi-quantitative basis to allow a larger number of sites to be sampled, data is therefore based on a single pass rather than multiple surveys (Reaney et al., 2011).

Once salmonid densities for each area have been collected, each surveyed section is assigned to a class: absent, poor, fair, good or excellent – based on the number of fry found there. It should be noted at this point that this classification was specifically designed by the Eden Rivers Trust for the River Eden catchment; hence calibration of results would be required before these classifications could be compared to other national electrofishing datasets (Eden Rivers Trust, 2009).

Past research by Reaney et al. (2011) has successfully applied the SCIMAP framework to salmonid fry data in the River Eden catchment. Findings show that arable land covers are relatively unimportant as drivers of fry abundance and instead it is areas of intensive pasture which become risky locations when they are readily connected to the river system by overland flow.

6.6.1.1 Upper Eden catchment

The Upper Eden is a large and varied landscape and forms the south eastern corner of the catchment. The headwaters of this part of the catchment lie on high fells where extensive sheep grazing is the main land use. Further downstream the catchment has long lowland stretches where farming becomes more intensive, gradients are shallower, population densities are higher and major infrastructure, such as the M6 motorway, can be found (Eden Rivers Trust, 2009).

Results from the SCIMAP modelling framework show the upper parts of the River Eden catchment to generally be at low risk of being critical source areas of diffuse fine sediment pollution. Data obtained as part of the ERT electrofishing surveys supports the risk weighting given to this area by SCIMAP. In all but a handful of the surveys carried out in this part of the catchment a number of fry, both salmon and trout, were found and at the majority of all other sites fry of one species were found.

Within this area of the catchment SCIMAP identifies Birkett Beck, to the west of the village of Hartley, to be the only area at very high risk of being a critical source area of pollution. However, electrofishing results show the area to have high densities of both salmon and trout fry implying that fine sediment pollution is not a major issue here. It is likely that SCIMAP assigns a high risk weighting to this area as a result of there being a disused quarry and several small areas of housing and development close by which could potentially be a source of large amounts of sediment. SCIMAP assumes this sediment risk must be transferred downstream and will have little time to dilute before reaching Hartley. In reality however the stream rarely has flow and so the required energy for transport is not available (Eden Rivers Trust, 2009).

A handful of electrofishing surveys found no salmon or trout fry in waterways deemed by SCIMAP to be at low risk of fine sediment pollution. However, this is not necessarily a reflection on the performance of the model and is instead likely to be a result of one of the many other factors that influence salmonid densities and determine whether habitats are suitable for them. The area may have a low risk of fine sediment pollution, as correctly identified by SCIMAP, but if another habitat controlling factor is making conditions unsuitable for salmonids this could explain why none were found as part of the electrofishing surveys.

6.6.1.2 Lower Eden catchment

SCIMAP predictions suggest areas most at risk of being critical source areas of diffuse fine sediment to be in the central lowland areas of the catchment between Penrith and Carlisle. The main channel of the River Eden and that of its tributary, the River Petteril, are both found in this area and it is well documented that water quality and ecological health is an issue along the Petteril. In recent years much research has focused on attempting to find possible explanations for this (Environment Agency, 2006; Eden Rivers Trust, 2009) and findings suggest that fine sediment pollution is indeed a serious issue in this part of the catchment.

Electrofishing results from the River Petteril appear to suggest that SCIMAP is correct to categorise this area of being particularly high risk. Much of its catchment is under intensive dairy production, and the river has been heavily modified and constrained over a long period of time. Previous work by the ERT has identified the river as particularly at risk from various pressures, most notably water quality and habitat issues resulting from intensive agriculture and the various major transport links running parallel to the river (Eden Rivers Trust, 2009).

Surveying results from 2008 show the River Petteril to have some of the lowest numbers of salmonid fry across the entire catchment and no salmon fry were found by the Trust at any site on the river in 2009 (Figure 6.5). There are no known barriers to migration on the River Petteril and so the absence of salmon fry is assumed to be related to poor quality water, suggesting that the modelling results from SCIMAP are accurate in suggesting the area to be at high risk.

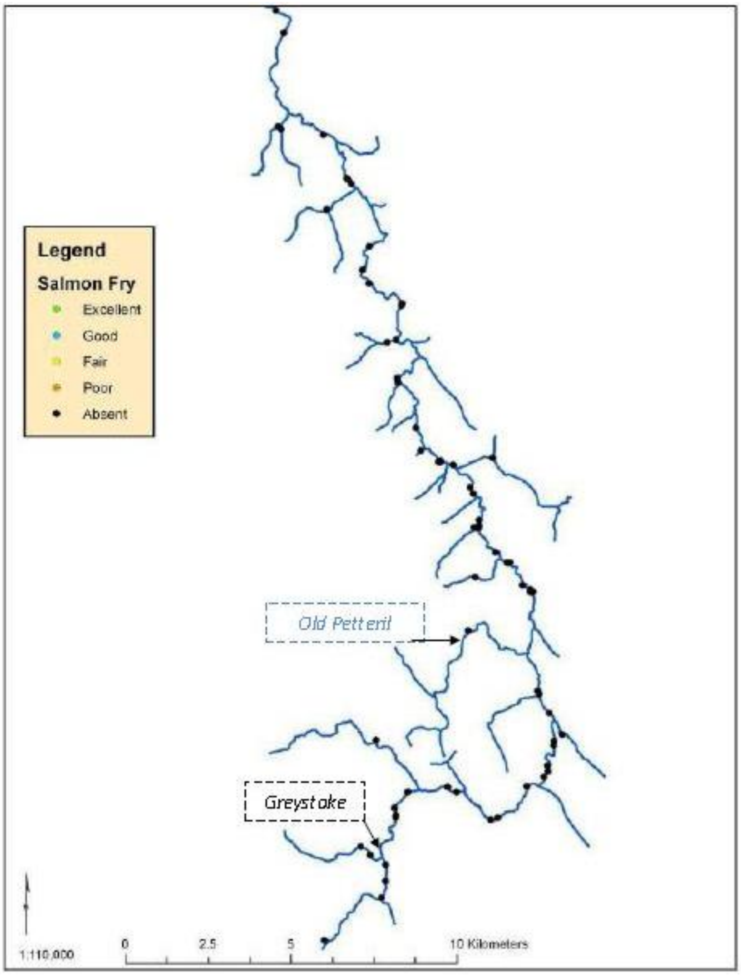


Figure 6.5: Electrofishing survey results, River Petteril, 2006. Source: Eden Rivers Trust, 2009.

A relationship can also be seen between the intermediate, delivery index layer of the SCIMAP mapping process and the number of salmonids recorded across the catchment. Areas with the highest network index values are found in the north-west of the catchment along the channel of the River Petteril where, as discussed previously, low numbers of salmonid fry are found. Along the course of the Petteril the probability of continuous flow to the river channel network is high, risk from upstream is therefore unlikely to become captured and material will, in all likelihood, be delivered to the receiving waterway. At the same time however, areas where connectivity risk is very low also have very low salmonid numbers as inputs of nutrients and sediment required to create suitable habitats are likely to be restricted.

Throughout the River Eden catchment the mean delivery index is consistently lower for sites where trout and trout fry are present compared to sites where they are absent. Analysis of this by Dugdale (2007, *unpublished PhD thesis*) show the differences to be statistically significant ($p<0.05$) for all surveyed years. Trends between the mean delivery index and the presence or non-presence of salmon fry are slightly harder to determine, however it has been shown that a significant relationship between the two does exist but that it is non-linear (Dugdale 2007, *unpublished PhD thesis*).

6.6.2 Gravel Siltation Mapping

The accuracy of the predictions made using the SCIMAP framework is highlighted further when delivery index values for the River Eden Catchment are compared with gravel siltation mapping carried out by the Eden Rivers Trust in 2004, 2005 and 2006 (Dugdale, 2007, *unpublished PhD thesis*). The presence of siltation was assessed visually by inspecting the substrate and also by the surveyor digging into the substrate to test the degree of cohesiveness. In areas where siltation levels are high and bed material has been infiltrated by finer material, coarse material will be difficult to dislodge.

Findings support the theory that reaches with a high delivery index have a greater probability of fine sediment deposition and the siltation of gravels. Sites which had high levels of siltation and points within the catchment predicted by SCIMAP as being at high risk of fine sediment pollution appear to correlate well. Furthermore, sites with observed gravel siltation consistently reported lower numbers of salmonid fry than those without siltation (Dugdale, 2007, *unpublished PhD thesis*).

Although the methods used in this instance to map gravel siltation are subjective, results suggest that levels of gravel siltation can be related to landscape parameters such as the type of upstream agricultural activity and the degree of hydrological connectivity in an area. These results support the assumption that SCIMAP is an appropriate tool for recognising areas most at risk of being critical source areas of diffuse fine sediment pollution.

6.6.3 Water Quality Samples

Milledge et al. (*in review)* used water quality data that are available through the Environment Agency for England and Wales (EA) General Quality Assessment (GQA) monitoring network to show that SCIMAP is able to reproduce observed water quality patterns across several UK catchments. The GQA scheme is designed to collect one sample per month and data are available for a 15 year period, 1990-2005, with a mean of 155 observations per site. Several dozen monitoring sites are located across the River Eden catchment (Milledge et al., *in review*).

Results from Milledge et al (*in review)* show that SCIMAP performs well for upland catchments that have higher and more variable annual rainfall. By using the SCIMAP approach, which incorporates a simple static metric for hydrological connectivity, both the frequency and duration of connection can be accurately captured (Lane et al., 2009). The accuracy of this capture is not lost when the model is applied over large areas (e.g. ~2300 km^2 for the River Eden catchment) at fine resolutions (<20m), making it a worthy tool for capturing both catchment scale and more localised risk (Milledge et al., *in review*).

Water samples of suspended sediment concentrations, carried out by the Environment Agency in 2007, further support the predictions from the SCIMAP framework. Along the main channel of the River Petteril suspended sediment levels were found to regularly exceed 20mg L^{-1} compared to large proportions of the rest of the catchment where samples rarely found levels of sediment to be above 10mg L^{-1} (Environment Agency, 2006). This adds support to the results produced during the SCIMAP modelling that this area of the catchment is most likely to be at greatest risk of fine sediment pollution. Further analysis of available water quality data, from across the entire catchment is required before further comparisons can take place.

6.7 Conclusions

Results obtained from the SCIMAP framework have highlighted the main valley of the River Eden Catchment as being most at risk of being a critical source area of fine sediment pollution. Initially, when assessing risk from a topographic perspective, the most at risk areas were in the headwaters of the catchment where a high risk of hydrological connectivity combines with an equally high risk of soil erosion. Once land cover weightings had been applied to the model, the focus of concern shifted, and lower areas of the catchment, where intense arable farming means land is often left bare for a large proportion of the year, becomes the focus of attention. Here, land is easily connected to the main river channel and hence there is a high chance of material that is entrained being transported all the way to a receiving waterway. In Chapter 7, the effects that variations in land use have on fine sediment erosion risk over the course of a year are investigated in detail.

Chapter 7: Image Processing and Analysis

As discussed previously fine sediment erosion risk is, to a degree, controlled by vegetation cover and the protection this offers to soils. To establish the degree of protection that soils have from rainfall over the course of the year, percentage crop cover for three different crops was assessed. Aerial photographs of three fields, collected using the methods outlined in Chapter Five, were analysed using MATLAB programming software. This information was then used to inform the TRIP fine sediment erosion risk framework, as discussed in Chapter Eight.

7.1 Analysis of Test Fields

Aerial photographs were collected across three test fields at Sewborwens Farm from March 2011 to July 2011 (Figure 7.1) using a Microdrone remote controlled helicopter. Fields with different crop covers were selected to enable direct comparison between the fine sediment erosion risks associated with various crops over the course of a year.

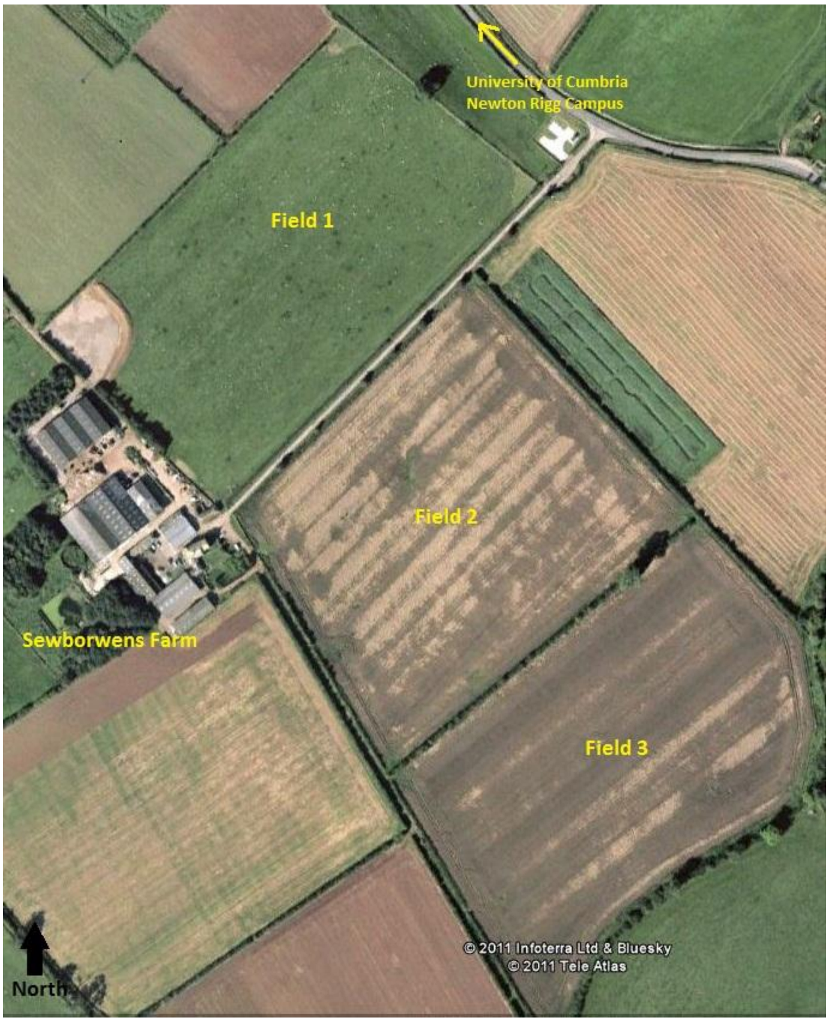


Figure 7.1: Location of study sites at Sewborwens Farm, Newton Rigg, Cumbria.

7.1.1 Field 1

Photographs collected from field 1 track the growth of a crop of winter wheat. This was planted during October 2010 and will be ready for harvest in August 2011. The first sets of images for analysis were collected in March 2011. Each image was taken using a 12 megapixel camera attached to the Microdrones MD4-200 remote controlled helicopter from a height of approximately 50 metres.

To ensure at least one of the images collected was suitable for use; more than 25 photographs were taken over the field. Once the images had been uploaded on to a computer, initial analysis showed that several would be unsuitable for processing. Reasons for this included: 1) the image being blurred; 2) other items, for example power lines, obscuring the image; and 3) the image being taken at too great a height, meaning areas of bare and vegetated ground could not be easily distinguished.

Figure 7.2: One of the initial images collected of field one in March 2011. When studied alongside Figure 7.2, the red box shows areas of particularly high growth between March and April 2011.

From all the images collected, the one with the highest degree of clarity was selected for processing using MATLAB. Use a coding sequence that was created as part of this research, the image was re-sampled based upon coordinates that were input into the code sequence to show areas of vegetation and bare ground. Markers, showing the noted areas of vegetation and bare ground, were then displayed on the image before a nearest neighbour classification was run and a colour label assigned to each pixel of the original image. By using a label matrix, objects in the original image could now be separated by colour or, in other words, each pixel could now be assigned to either a “vegetation” or “non-vegetation” class. A full description of the MATLAB coding used in this analysis can be found in Appendix 1.

Once the initial image had been processed in MATLAB, the overall number of pixels per image was determined to enable the percentage of crop-covered ground and bare ground to be calculated. Results show 52% of the ground captured in the image to be covered by vegetation meaning 48% is currently bare soil. As identifiable in Figure 7.2, some of this bare ground is between individual seedlings of winter wheat which have yet to sufficiently grow and spread out; however large proportions of the bare soil are in the tractor lines around the edges of the field.

A second set of aerial images were collected in early April. As shown in Figure 7.3 the crop had grown considerably during the previous month and fewer bare patches of soil are identifiable within the image. Several parts of the field which in early March had large amounts of bare ground, such as the area identified on Figures 7.2 and 7.3 by the red box, now had a much thicker vegetation covering. Through processing the image using the MATLAB software, it was found that vegetation cover had increased from 52% to 69% during the four-week period between the first and second set of images being collected. By early April therefore only 31% of soil at this site was exposed to erosion.

Figure 7.3: Aerial photograph of Field 1 taken in April 2011. As can be seen grown cover has increased considerably since the photograph in Figure 7.2 was taken in early March.

Further images were collected in the middle of May; however, the quality of the images was lower than those gathered previously due to poor weather. Many of the images were very blurry and initial calculations proved inconclusive as the MATLAB program was unable to determine between areas of vegetated and non-vegetated land. To try and ensure results could be obtained from the image, an increased number of markers were manually applied to the image before the nearest neighbour classification was run and a colour label assigned to each pixel of the original image. Results showed a percentage crop cover of almost 77%, an increase of only 8% since the last images were collected the previous month. This suggests a much slower period of growth throughout April than that which was observed in March. These findings correspond well with previous research into the growth patterns of wheat which found evidence of the crop going through a period of fast growth when tillering is occurring before growth slows down as stem elongation begins (Fowler, 2002). It is also possible that the period of rapid growth early on in the season may have been a result of the exceptionally warm and sunny weather across the UK during late March and early April 2011. Records show that temperatures were 3.7 °C above average during April and the spring of 2011 was the equal-warmest since 1910 (Meteorological Office, 2011). April was a particularly sunny month with 48% more sunshine than normal. Provisional records suggest April 2011 to have been the sunniest since 1929 (Meteorological Office, 2011).

Growth continued throughout the late spring and early summer, and by the middle of June crop cover had increased to approximately 90%. At the point when the final selection of images were collected in the middle of July 2011 over 94% of the soil within field one had a covering of wheat.

Figure 7.4 shows the pattern of percentage crop cover increase throughout the time period studied in this research. As discussed previously, and as identifiable in Figure 7.4, the wheat experienced a period of rapid growth in the early spring months before continuing to grow at a steadier rate over the course of the rest of the study period. There is little erosion risk associated with winter wheat crops at this time of the year, and so Figure 7.4 is of little interest from a fine sediment pollution perspective. The trends and percentages of average crop cover do however correspond well to previous research (e.g. Fowler, 2002), thus showing the aerial photography and image analysis methods developed in this research to be a useful research tool.

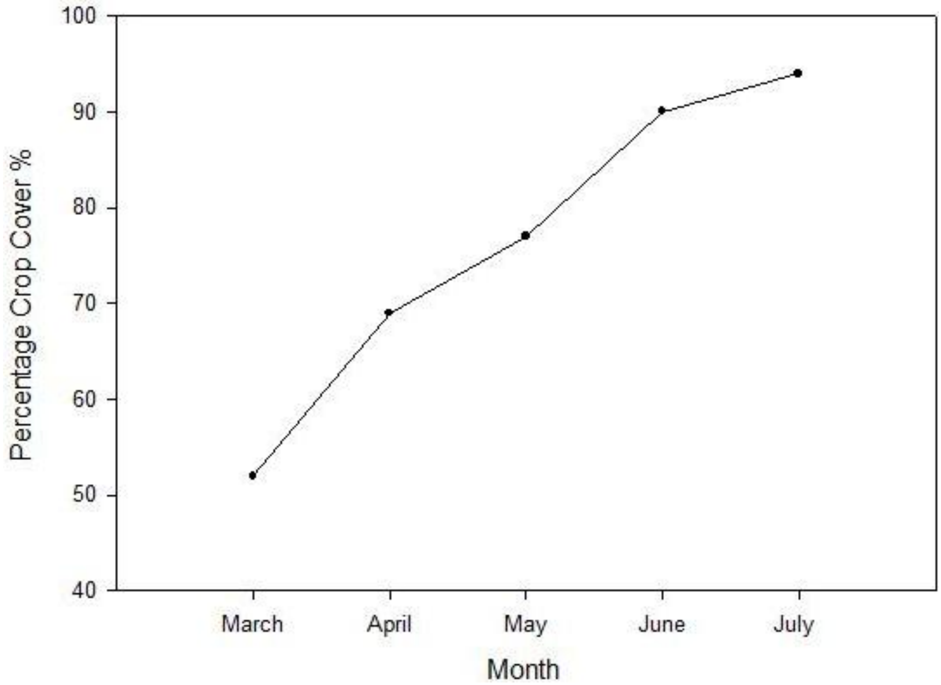


Figure 7.4: Pattern of winter wheat growth at Sewborwens Farm as assessed using aerial photography analysis.

7.1.2 Field 2

Photographs collected from the second test field at Sewborwens Farm track the growth of a crop of oilseed rape. This was planted in August 2010 and will be ready for harvest by late July 2011. When the first set of images were collected from this field in early March very few areas of vegetation could be identified and over 90% of the soil was exposed to erosion (Figure 7.5). The small areas of vegetation that are identifiable are weeds and grasses that began to colonise the soils over the winter months since the last crop was harvested in September 2010.

Figure 7.5: Exposed soils in Field 2, March 2011.

Little growth occurred during March, and by early April images show the field to have changed little over the course of a month. Some larger areas of weeds were beginning to develop, particularly around the most central parts of the field, but growth rates appear to be generally quite slow.

The next set of images were collected in early May; as was the case with the images collated for Field 1, the quality of picture was poor and large amounts of the picture were blurred (Figure 7.6). In order to process the image, the bottom half of the picture was therefore discarded to enable an accurate measurement of vegetation cover to be taken. Analysis of the image collected in May shows a large amount of growth had occurred in the previous month and crop cover was now just less than 60%. Rapid growth during the early stages of development of any crop is important as it enables the plants to compete with and smother out weeds in the soil below.

Further analysis of vegetation cover in early June shows the oilseed rape crop to be covering 85% of the soil. Considering at the start of April practically no vegetation cover was identifiable, this represents a period of extremely fast growth.

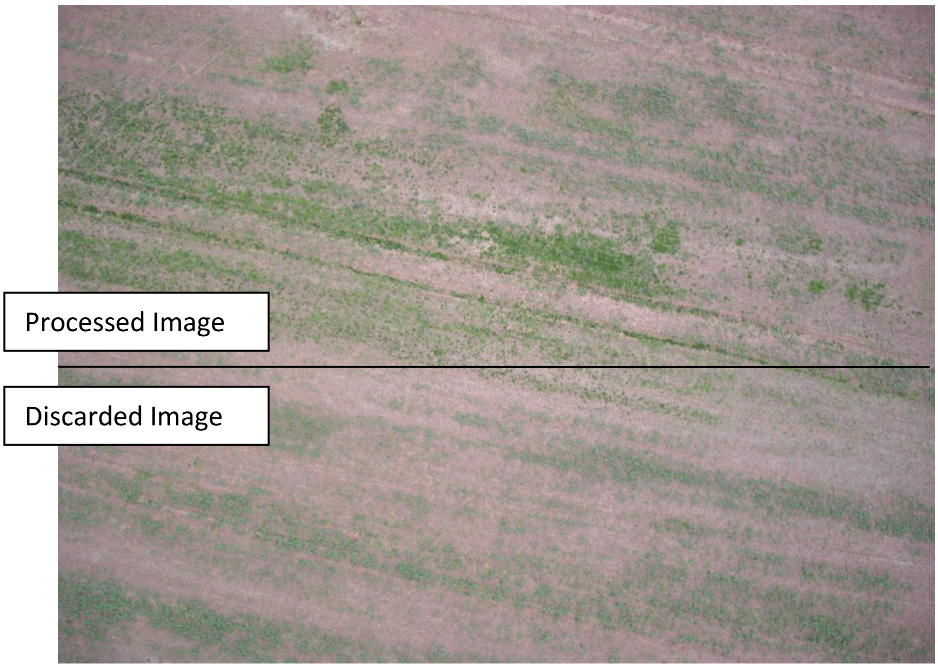


Figure 7.6: Vegetation developing in Field 2 June 2011.

The rapid growth observed during April continued throughout May and early June and when images were collected at the end of June large areas of the field were now covered with the flowering oilseed rape (Figure 7.7). Percentage crop cover at this point was found to be approximately 93%.

Figure 7.7: Flowing oilseed rape crop in Field 2, late June 2011.

Analysis of aerial images collected from Field 2 have therefore shown the growth patterns of oilseed rape differ considerably to those of other autumn-sown crops. Whereas wheat grows at a relatively steady pace throughout the winter months, oilseed rape appears to grow at a much slower rate initially before suddenly experiencing a period of rapid growth in May and June (Figure 7.8). Split dressings of fertilisers are usually applied to oilseed rape in March, April and May and this may go some way to explaining the sudden growth the crop experiences at this time.

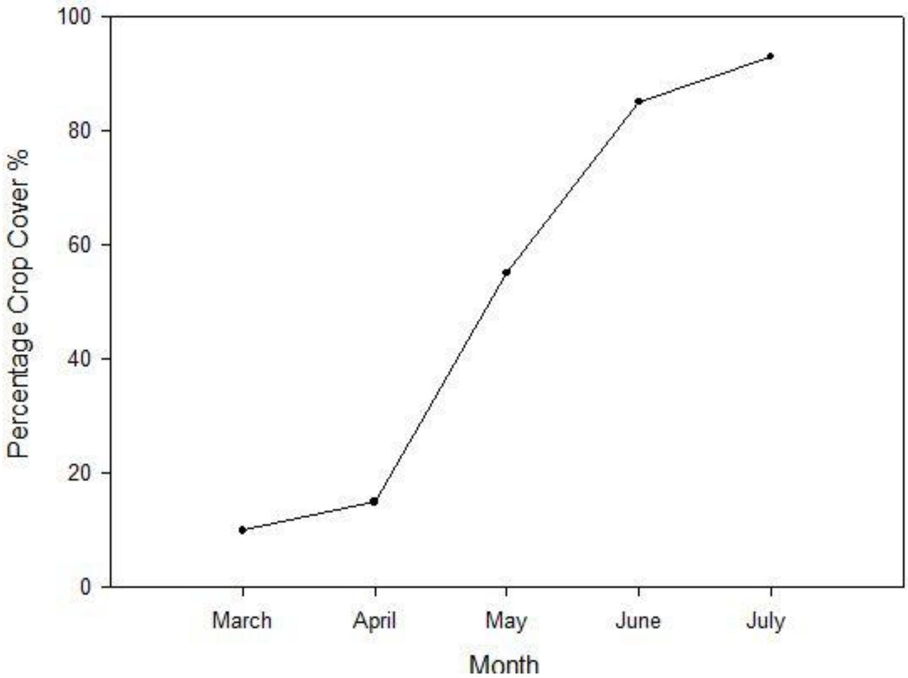


Figure 7.8: Growth rates of the oilseed rape crop in Field 2.

The results shown here for an oilseed rape are somewhat unusual compared to findings from other research. There is good evidence that oilseed rape usually covers the ground quickly after initial drilling in September time, thus a greater crop cover over the winter months would be expected compared to what was observed in this research. Having been planted in August, oilseed rape crops are usually able to take advantage of the warm late summer weather and autumn rainfall, percentage ground cover by November is therefore approximately 25%.

7.1.3 Field 3

Photographs collected from Field 3 track the growth of a crop of spring-sown Barley. Unlike the other two study sites the crop here was not planted during autumn 2010. Instead, the barley was planted at the end of the winter in March 2011 and will be harvested late on in 2011, probably in September. Crops of spring barley are relatively frost-sensitive and so in northern catchments such as the Eden are rarely sown earlier than the middle of March.

When the first two sets of aerial images were collected in March and April 2011, soils within this field appeared completely bare and no vegetation was identifiable when processing the images within MATLAB. Typically, the first shoots of barley will emerge after 10-30 days depending upon temperature. It is unlikely therefore that many would have reached emergence by the time the second site of images were collected at this location in early April. Over the period of weeks where the barley had yet to emerge, soils were extremely exposed and had no protection against erosive rainfall; the fine-sediment erosion risk for this field at this point in time was therefore extremely high (Figure 7.9).

Figure 7.9: Bare soil in Field 3 where the crop of spring barley have yet to develop, April 2011.

Once seedlings have emerged leaves generally develop every three to five days, depending upon the variety of wheat being grown and the climate of the area. The rate of leaf area establishment usually depends on temperature, but can be increased by high nitrogen fertilisation. This process of canopy formation is captured in the aerial images taken of the site and by May total vegetation cover was found to have increased to 35%.

Figure 7.10: Spring Barley after emergence as leaves begin to develop

Canopy expansion in spring-sown barley tends to be more rapid than in autumn-sown wheat and so by June a large increase in levels of vegetation cover was expected. This was identifiable in the aerial photographs collected and analysis showed vegetation cover in June was 49%. By the time the final selection of images was gathered in July 2011, crop cover in this field had reached 67%.

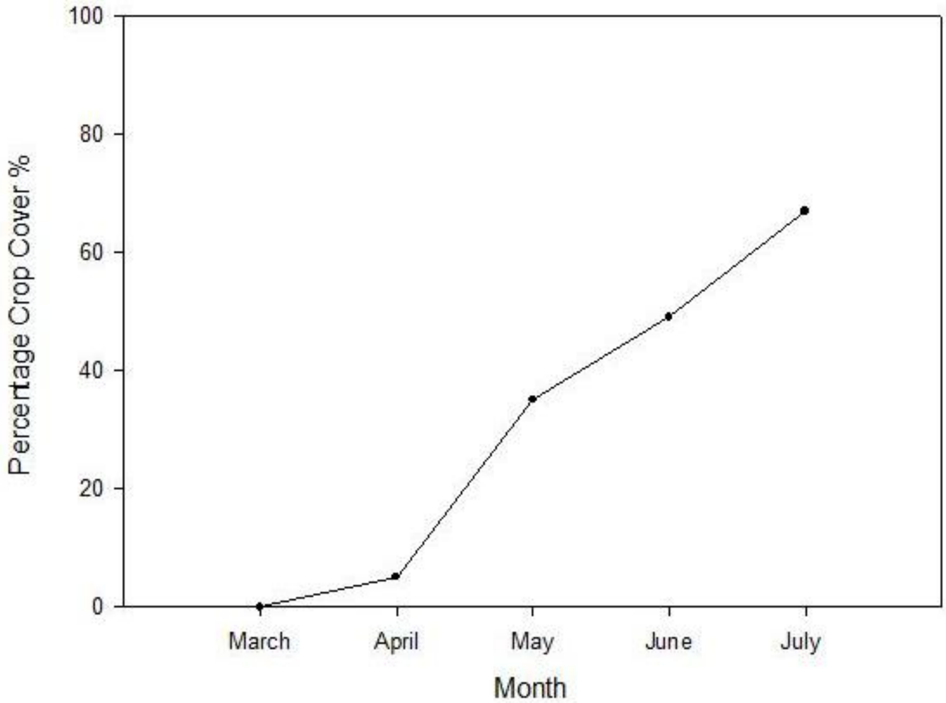


Figure 7.11: Growth rates of the spring barley crop in Field 3.

7.2 Validation of Images

In order to begin to validate the results obtained from the TRIP framework, field surveys, where crop cover was manually measured, were carried out on the same day that images were collected using the remote control helicopter using the methodology outlined in Chapter Five. A full record of the field survey results gathered from each field can be found in Appendix 2.

7.2.1 Field One

Results from the field surveys obtained from Field 1 at Sewborwens Farm support the percentage crop cover estimations produced from the aerial images using the MATLAB software. The field survey carried out at the same time as images were collected at the start of June found crop cover to be 90% towards the centre of the field. Analysis of the aerial photograph of the same section of the field showed cover to be 95%.

A field survey was also conducted along the edge of the field where evidence of tractor markings and erosion around the gateway into the field could be seen. Results from the field survey suggested that crop cover at this point was slightly lower with only 80% of the soil being protected by vegetation. Analysis of the aerial photographs of this area also allows these slightly barer patches of land to be captured, and results from the picture analysis suggest crop cover to be in the region of 83%.

In both of the instances discussed above the software used to assess areas of vegetation is slightly over-estimating crop cover compared to what is observed on the ground. This is likely to be a result of the image being taken at slightly too higher height making it very hard for the software to identify the smallest gaps between plants where soil will be exposed.

7.2.2 Field Two

Comparisons between results obtained from the MATLAB image processing and field surveys in field two show strong correlations. Image processing found crop cover to be in the region of 76% in early June and field surveys results support this (69%). At the edge of the field, particularly around the entrance gateway and along the hedgerow at the west edge of the field, crop cover is considerably lower and this can be identified in both the results from the image processing and the field surveys. Of the 300 points sampled along a transect at the edge of Field 2 only 145 had some degree of vegetation cover over the soils meaning crop cover was only 48%. This is significantly lower than cover in the central parts of the field and thus highlights the effect that tractor lines and similar disturbances can have on increasing soil exposure to erosion.

7.2.3 Field Three

Unlike Fields 1 and 2, which were near full growth when field surveys were carried out in June 2011, the spring barley crop in Field 3 had only been planted three months previous to the field surveys and hence had had less time to develop. It was immediately noticeable how much smaller the plants were in this field and results from the surveys suggest crop cover to be in the region of approximately 54% in early June.

When results from the field surveys were compared with the predictions of vegetation cover obtained from the aerial image analysis it became apparent that there was quite a large discrepancy between the two methods. Analysis of the aerial photographs of Field 3 suggested crop cover to be in the region of just 40%, over 10% less than what was found during the field surveys. It is likely that this is a result of the software being unable to detect very small areas of vegetation on the photographs, thus meaning the program will under-estimate total cover. It is therefore worth noting that in fields where crops have only recently been planted, or where large amounts of soil are exposed, it may be necessary to collect aerial photographs at lower heights to enable accurate identification of small patches of vegetation.

7.3 Variations Between Crops

As discussed above, assessing the degree of vegetation cover within a field is a difficult process when the crop is in its early stages. When using the MATLAB programming method the software struggles to locate small patches of vegetation, and when using the field survey approach it is difficult for the observer to accurately say whether a point has a very small amount of vegetation cover or not. One may therefore expect the two methods to converge as crop cover increases and findings from each of the 3 fields support this theory.

Over the course of the five month period when aerial photographs were collected for the 3 test fields vegetation cover varied considerably between each site. As shown in Figure 7.12, Field 1 which was being used to grow a crop of autumn-sown wheat, initially had a more comprehensive crop cover than the surrounding oilseed rape and spring-sown barley fields. This is due to the longer period of time the wheat crop had had to develop since being sown. However, by June the oilseed rape being grown in the second test field had almost as much soil covered by vegetation as the first field. Oilseed rape is known to experience a period of very rapid growth between May and June whereas autumn-sown wheat grows at a much more constant rate over the course of the year. All of the test fields experienced a slowing in growth rates between June and July as each of the crops began nearing full growth. At this point canopy cover is extremely thick, protecting soil from erosive rainfall.

Throughout the period studied in this research the third field, being used to grow spring-sown barley, had the lowest vegetation cover. Initially, Field 1 had over 50% more vegetation cover than field three. However the spring-sown barley grew at a faster rate than the wheat crop over the early summer months, and by the time the final set of images were collected from each field in July the difference was a little over 25%. If aerial photographs continued to be collected each month it would be expected that Field 3 would have a greater degree of vegetation cover in the late summer and early autumn of 2011 as the crop would be harvested later than both Field 1 and Field 2.

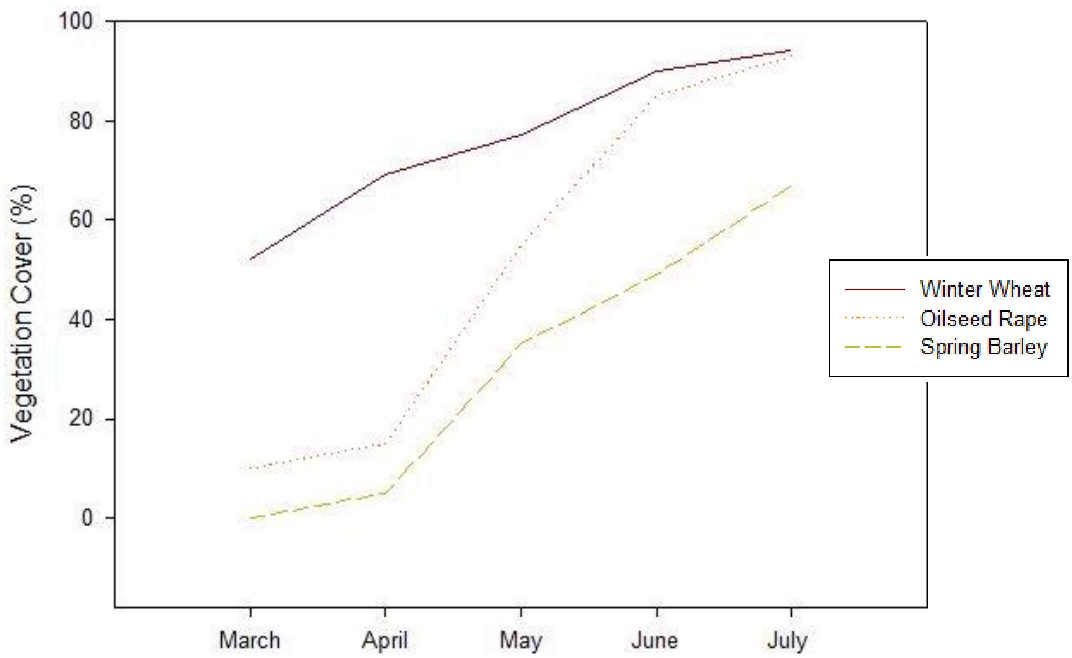


Figure 7.12: Variations in vegetation cover between 3 test crops at Sewborwens Farm March 2011 – July 2011.

7.4 Conclusions

This chapter has discussed the results of using a piece of coding within MATLAB software and aerial photographs to assess percentage vegetation cover within three fields at a test site in the River Eden catchment. Results from this analysis have highlighted the varying growth rates experienced by different crops and how total levels of cover can vary considerably between one crop and the next. Estimations of crop cover obtained from the image analysis were compared with field survey observations in order to validate the photograph analysis methods developed in this research as a suitable method for determining vegetation cover. Comparisons between the two methods found the aerial image approach to be a suitable way of assessing cover, particularly when crops are well developed. However, it has also be found that particular care needs to be taken to ensure photographs are not taken at too great a height when crops are in their infant stages, otherwise the programming software is unable to accurately identify areas where sparse vegetation cover is present.

The results from this analysis will now be used to simulate the effects that changing patterns of vegetation cover and rainfall have on fine-sediment erosion risk over the course of a year. The results of this simulation can be found in Chapter 8.

Chapter 8: Scenario-Based Minimisation of Risk

A significant limitation of the SCIMAP framework is that it cannot account for changes in fine-sediment erosion risk over time. To investigate the effect that various crop cycles have on diffuse fine sediment risk over time, several cycles were analysed using the newly created TRIP framework. Six cycles, which are typically found across the River Eden catchment, were used to establish the diffuse fine sediment pollution risks associated with current land management. Initially, this chapter presents the results of processing the six current crop cycles within the TRIP framework and assess which of these cycles would minimise fine sediment erosion risk to the greatest extent. Leading on from this, latter parts of this chapter present the effects that projected climate change and changes in land management may have on fine sediment erosion risk in the future.

8.1 Simulating Current Risks

Current fine sediment erosion risks for six crop cycles that are typically found across the River Eden Catchment were inputted into the TRIP framework. Initially, a single risk value for each of the cycles was determined before this was broken down to enable the analysis of variations in risk across annual and monthly timescales.

8.1.1 Long-Term Risks

Using the method discussed in Section 4.2.3, six crop cycles, each based upon a standard five-year cycle, which accurately represent land management and land use within the River Eden catchment, were selected for processing within TRIP. This enabled a single overall fine sediment erosion risk value for each of the cycles to be determined. The overall risk values for each of the cycles are shown below in Figure 8.1. Cycles with the higher values are deemed to be more of a diffuse fine-sediment erosion risk and those with lower values are classified within the TRIP framework as causing less risk.

As shown in Figure 8.1, having a five-year cycle which involves just grassland minimises the risk of diffuse fine sediment erosion to the greatest extent. The highest risks are associated with 5-year cycles of maize and spring-sown barley where land is intensively worked throughout the cycle and soils are left bare over the course of the winter months. Cycles 2 and 3, which both involve several years of intense cropping followed by a period where land is left as grassland, have similar moderate to low risks. Cycle 1, which involves a five-year rotation of three different crops is classified by TRIP as being a lower risk than both the maize and spring barley cycle; however, the risk is still considerably higher than with cycle 2 or 3.

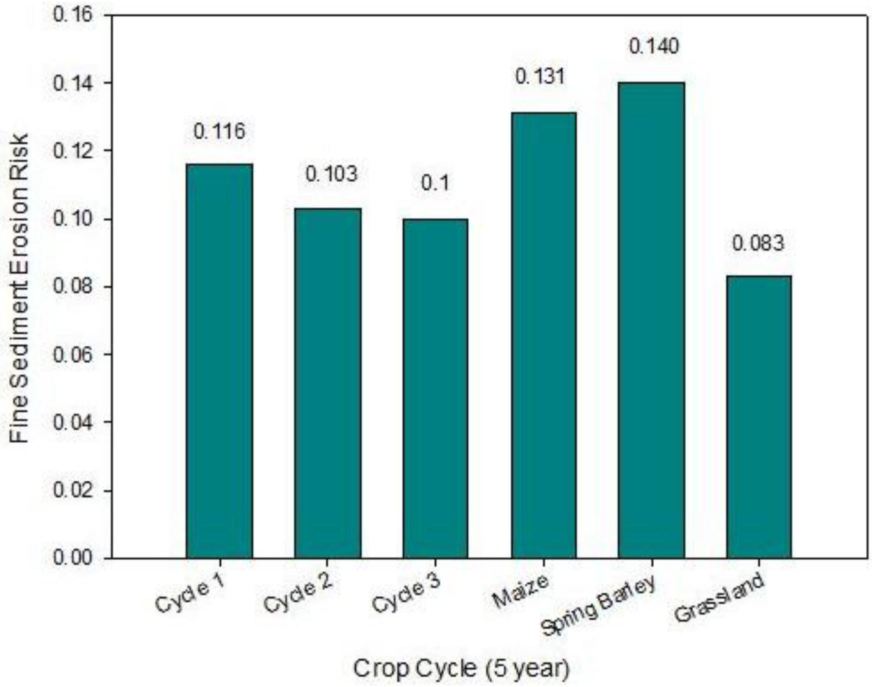


Figure 8.1: Annual average fine sediment erosion risk for various crop cycles within the River Eden catchment.

As expected, TRIP assigns the lowest fine sediment erosion risk values to the cycle where land is left as set-aside grassland for the whole of the five-year cycle. Set-aside ensures soils have a significant covering of vegetation throughout the course of the year and disruption by ploughing or other management practices is minimal. However, in reality, having land in a 5-year grassland rotation may not be economically viable for farmers and thus may not be a realistic management option to minimise fine sediment erosion risk. The exception to this, although unlikely, would be if farmers had land available that they did not need for a long period of time. Some 5-year grassland rotations are found across the River Eden catchment; however these areas are used as grazing ground for dairy herds rather than being set-aside grassland.

As already mentioned, Cycles 2 and 3 have similar risk predictions associated with them. Cycle 3, with the slightly lower risk, incorporates three years' worth of set-aside and two years' worth of winter crop. Cycle 2, with the slightly higher risk, has the opposite: two years worth of set-aside grassland and three of winter crops. The minimal difference in risk between these two cycles suggests the overall benefits of introducing an extra year of grassland into a five-year cycle to be minimal. From a management perspective, suggesting a three-year set-aside period within a cycle is therefore unnecessary. This is an extremely useful finding as from an economical perspective: a three-year set-aside period will not be possible for many farmers.

The highest risks of diffuse fine-sediment pollution are associated with crops of maize. These are usually planted in April and May of a given year and harvested in September of the same year thus meaning during the winter months soils are bare and exposed prior to spring planting. This exposure, which occurs during the wettest periods of the year, explains why TRIP assigns such high risk to these crops. In coming decades, when winter storms are predicted to become more intense, the risks attributed to maize are likely to increase further. There is the added risk of erosion from early summer thunderstorms when the tilled soil has little crop cover (Boardman et al., 1996)

The crop cycle which minimises risk as much as possible whilst still being economically viable for farmers is crop cycle 2 (winter wheat > winter wheat > winter barley > set-aside grass > set-aside grass). The three years of autumn-sown crops offer a degree of protection to the soil over the winter months and, as the new crop is planted approximately a month after the previous one has been harvested, there is only a short window of time where land is bare. Winter wheat is planned in August, hence by the time the wetter autumn and winter weather arrives, seedlings have had a chance to establish themselves and some ground cover is evident. This is also true to a lesser extent with autumn-sown crops of barley. These are usually planted in September thus meaning that by late autumn some ground cover is evident. The main risk period for winter cereal erosion, as captured in TRIP, is between October and December when low crop cover is combined with high average rainfall.

8.1.2 Monthly Risk Variability

The fine sediment erosion risk of a particular crop varies over the course of a year as a result of differences in vegetation cover percentages from one month to the next (Figure 8.2). When the risk for each crop is assessed over the course of a calendar year, it can be seen that maize has by far the highest risk value for the first eight months. Maize is usually not planted until May and hence land is bare, possibly with the exception of some residual stubble, over the wet winter months, and continues bare for a while after drilling, of course. Furthermore, wide un-vegetated rows exist between the maize plants even when the crop is nearing maturity. However, during the autumn months fine sediment risk from maize is considerably lower than with other crops as its late harvest time means soils are well protected during autumn storm events. Harvesting of maize usually occurs in October in the Eden Valley and issues surrounding erosion on post-harvest maize fields are well documented (Boardman et al., 2009; Defra, 2010). Post October, just at the time when average monthly rainfall is increasing, once maize has been harvested, fields are lacking sufficient crop cover to inhibit runoff and prevent erosion (Boardman et al., 2009). Another factor resulting in increased erosion risks associated with maize is that harvesting later on in the autumn often means using heavy machinery on wet ground. This leads to further compaction of the soil and damage to soil structure, erosion rates therefore increase further.

Maize is often grown repeatedly from year to year in the same field, and fields are frequently left over winter prior to cultivation and pre-drilling the following spring. This further exacerbates the issue of soils being exposed over the wet winter months.

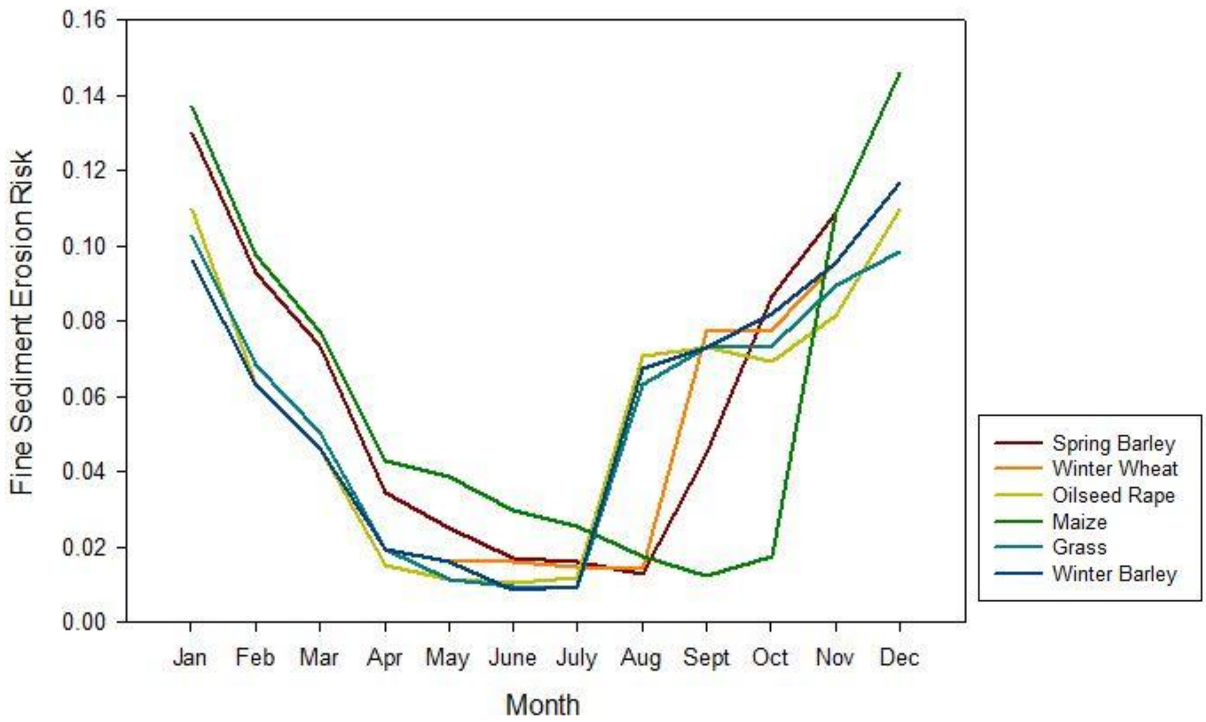


Figure 8.2: Monthly fine sediment erosion risks for various crops within the River Eden catchment as calculated using the TRIP framework.

During the summer months grassland has the lowest risk of contributing to a diffuse fine sediment erosion problem; this is a result of ground cover often exceeding 95% in the summer months, offering a great degree of protection to soils. TRIP also assigns very low risks to autumn-sown wheat and barley crops, both of which are near to full growth and harvest at this time of year. Percentage ground cover for both of these crops rarely reaches more than 90% even once fully developed hence the slightly higher risk compared to grassland.

Managing fine sediment erosion risk during November and December is extremely important when attempting to minimise overall risk at a location. Research shows oilseed rape to the optimum crop for minimising risk at this time of year. Having been planted in August, oilseed rape crops are able to take advantage of the warm late summer weather and autumn rainfall; percentage ground cover by November is therefore approximately 25%. This is in contrast to other crops, for example autumn-sown wheat, which are planted later on in the autumn and have ground cover of only 10-15% by late December period. Boardman (1992) noted that for soils to have adequate protection against erosion, a minimum of 30% crop cover is required (Figure 8.3). In the River Eden catchment this requirement is likely to be met in some mild, wet autumns when new seedlings are planted early enough.

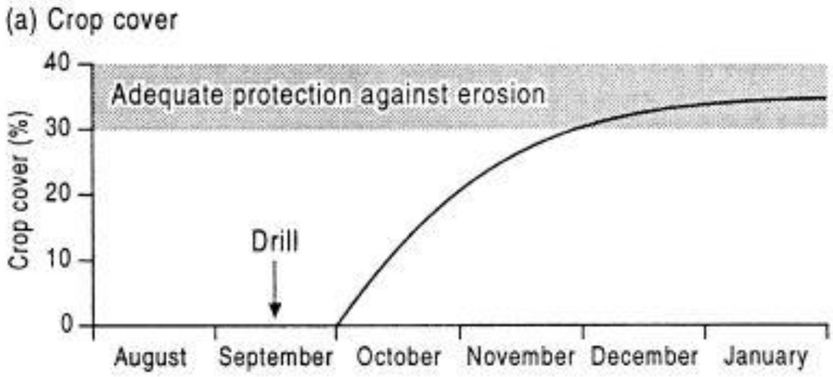


Figure 8.3: For soils to be adequately protected over the winter months crop cover needs to be above 30%. From: Boardman (1992).

8.2 Simulating Future Risks

Projected changes in land management have the potential to drastically alter diffuse fine sediment erosion risks across the River Eden catchment. The potential effects that two of these changes could have on fine sediment erosion risk are discussed below. The land management changes considered are: 1) a decline in winter cereal crops and 2) an increase in the amount of land being cultivated as opposed to being managed as grass pasture.

A decline in winter cereal crops such as winter wheat and winter barley has been studied in this research for two main reasons. Firstly, as highlighted in research by Boardman et al. (1990), it has been predicted that the number of winter cereal crops will decline, as a result of predicted wetter winter weather, and these will be replaced by more erodible spring-sown crops such as maize. Secondly, in the River Eden catchment a decline in the number of winter crops is already being observed (Cleasby, per comm) and this trend is likely to continue in future decades. Research suggesting a decline in the amount of land used to cultivate winter cereal crops is therefore highly applicable to this study catchment.

The second projected change considered in this research was a potential decline in the amount of land left as set-aside grassland. The River Eden catchment is already experiencing such declines (Cleasby, per comm.) and this is an issue that is likely to become more prevalent in coming years (UKCIP, 2009). Research has shown that more farmers are starting to cultivate land previously deemed to be unsuitable for arable crop growing (Inman, 2006). This is due to a combination of factors including: increased financial hardship for farmers meaning they need to maximise profits from crops as much as possible; technological advances meaning machinery can now be used on land previously too difficult to access; and an increased demand for food production (Inman, 2006).

8.2.1 Decline in autumn-sown cereal crops

The effect that winter crop cover has on levels of fine-sediment erosion risk was investigated by altering a current crop cycle used in the River Eden catchment to a five-year cycle where spring barley is grown every year. Spring barley is usually planted in March or April and is then harvested in August or September of the same year. Once the crop has been brought in, a stubble cover is usually left over the autumn and winter months before a new crop of barley is planted again the following spring. Although this stubble offers some degree of protection to soils between October-December when erosion risks are highest, total ground cover is less than if a winter cereal crop had been planted. It has also been noted that, despite various subsidies being available to farmers for leaving stubble on the ground over the winter month, many farms still do not do this (Cleasby, per comm.).

Results from the TRIP model show that when an autumn-sown crop, such as winter wheat, is replaced with a spring-sown crop, the fine sediment erosion risk for the location in question increases considerably (Figure 8.4). This is particularly the case in the first quarter of the year when the spring crops have yet to be sown, hence the only protection soils have from erosion is provided by the stubble that has been allowed to develop over the winter months. It is not until August, when both crops have an average ground percentage cover of approximately 90%, that the risks associated with the two become more even. For a short time during the autumn, the risk associated with winter sown crops becomes higher than the risk from the spring ones. This is due to wheat being harvested earlier on in the autumn season. Hence, for a short period of time winter wheat fields are bare whereas fields with spring-sown crops such as maize are still covered. By October, however, the fine sediment risk from the winter wheat crop has begun to fall as the newly sown crop begins to show signs of early growth offering some protection to the previously exposed soils.

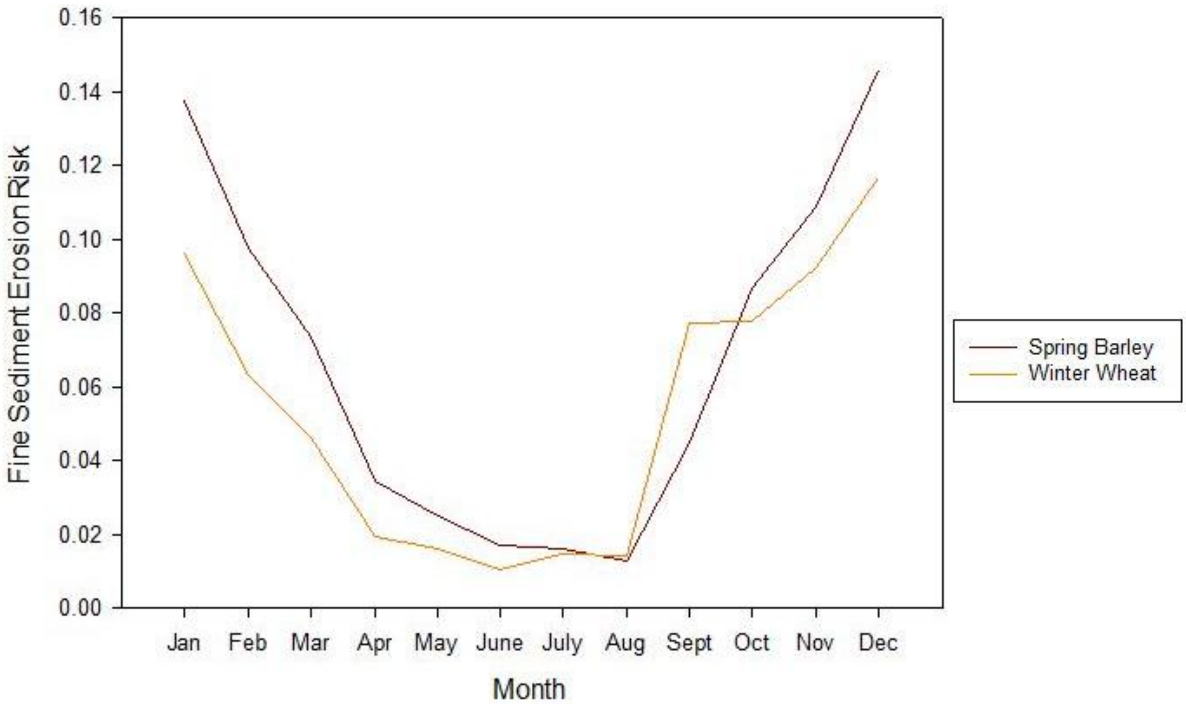


Figure 8.4: Variations in fine sediment erosion risk over the course of a year between two different crops as calculated using the TRIP framework.

From the TRIP model outputs it can therefore be seen that planting winter wheat is an extremely effective way of minimising fine sediment erosion risk. In the winter months (December, January, February) the erosion risk associated with arable land being used to grow this crop is considerably lower than the risk on land which has been harvested in the autumn and has not had a new crop sown. This can be seen in Figure 8.4 where a crop of spring-sown barley has much higher risk over the winter months than any of the autumn-sown crops. In the wetter parts of the catchment, where intense winter rainstorms have high erosive powers, such variation in risk is likely to have big implications on diffuse fine-sediment erosion rates. Intense winter storms may also lead to crusting of soils further effecting rates for erosion.

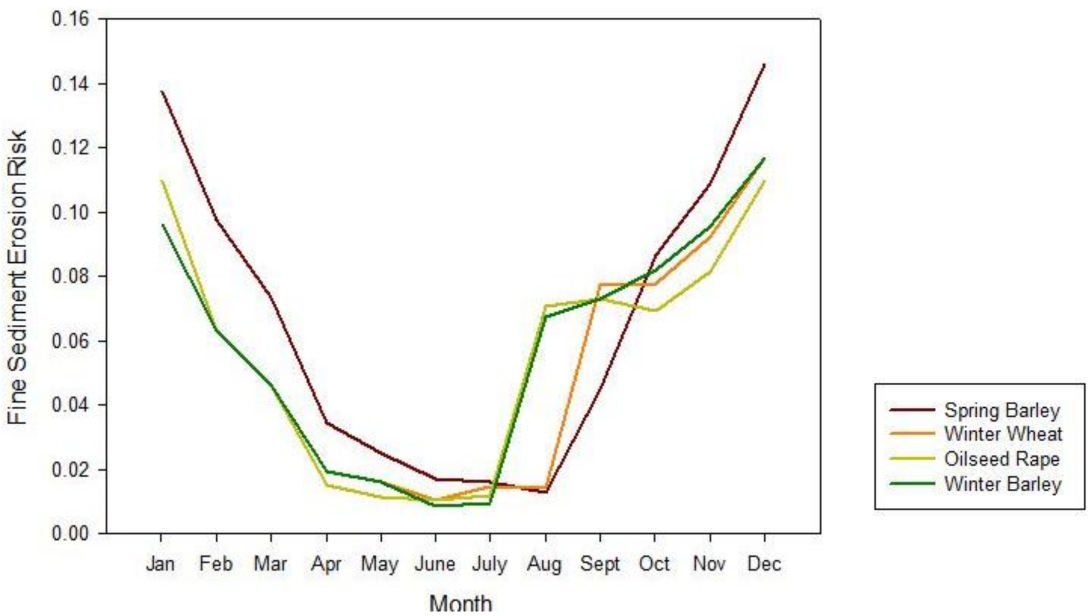


Figure 8.5: Monthly variation in risk between a winter cover crop and spring-sown alternatives as calculated using the TRIP framework.

The effects of reducing the amount of winter crop cover provided to land within a five-year crop cycle vary when considered spatially across the River Eden catchment. At Appleby, a relatively wet slightly higher area which receives more than 90mm of rain on average in both December and January, the effects of replacing an autumn-sown crop with a spring one are more strongly felt. As identifiable in Figure 8.6, there is a sharper increase in risk when changing a crop from autumn-sown wheat to spring-sown barley at Appleby than when the same change occurs at Broadfield House, an area further downstream in the catchment which receives rather less rain over the course of the winter months.

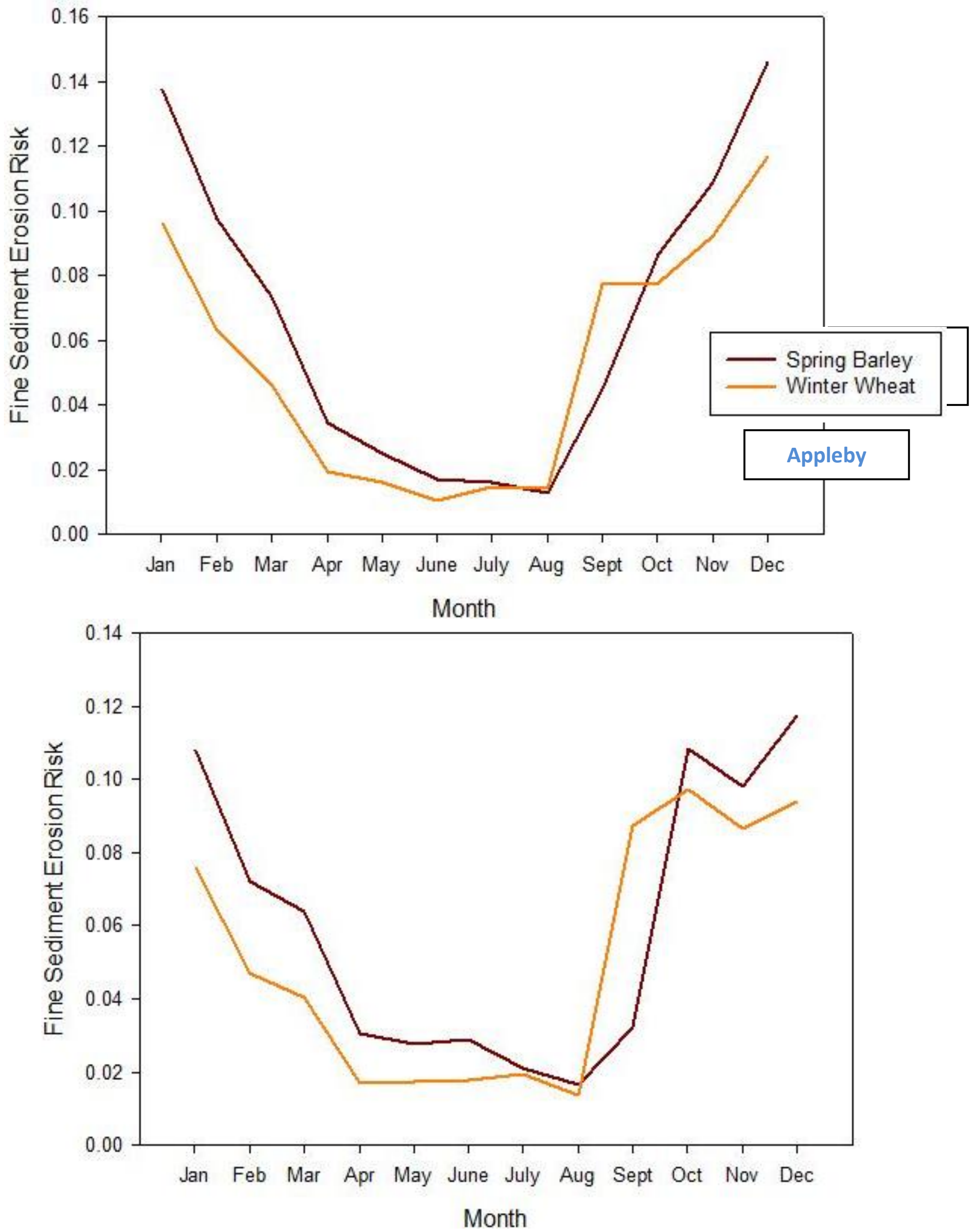


Figure 8.6: The effect of changing from a winter cover crop to spring-sown alternative at two locations within the River Eden catchment as calculated using the TRIP framework.

8.2.2 Decline in Grassland

The second factor to be investigated with regards to changing agricultural land management was the effect that a decrease in the amount of land left as set-aside grassland would have on fine sediment risk. A standard crop cycle, which is currently widely used across the River Eden catchment, was modified in order to simulate the decline of set-a-side grassland (Table 8.1).

Input Cycle	Year 1	Year 2	Year 3	Year 4	Year 5
Original Cycle	Winter Wheat	Winter Barley	Winter Barley	Grass	Grass
Edited Cycle	Oilseed Rape	Winter Wheat	Winter Barley	Winter Barley	Oilseed Rape

Table 8.1: Original and edited crop cycles inputted into TRIP to simulate the projected decline in set-a-side grassland.

Results from the TRIP framework predict the total risk for the original cycle, which includes a two-year period of set-aside grassland, to be 0.09. However, in the edited cycle, where the grassland is removed, the risk increases to 0.11. Removing the two years of set-aside from the end of the cycle significantly increases overall fine sediment erosion risk in years four and five. In the original cycle, soils would have had a degree of grass cover throughout the course of the year and during the summer months vegetation cover would be over 85%. In the modified cycle however, the replacement of the grass with a crop of winter barley followed by a cycle of oilseed rape means soils are much more exposed during the autumn and winter months when new seedlings have yet to have the opportunity to develop. When grass is not cut in the autumn for hay winter ground cover is approximately 40%. This is in comparison to a typical ground cover of 15% for cereals such as wheat and barley.

The results obtained from the TRIP model thus highlight the importance of incorporating years of set-aside grassland into five-year crop cycles to ensure that soils are given ample time to recover from previous crop seasons. Similarly, particularly in areas where fine sediment erosion is known to be a problem, a winter covering of grass can significantly reduce erosion risks at the time of year when highly erosive storms are most likely.

8.3 The Effects of Projected Future Climate Changes on Risk

UKCP09 climate change predictions suggest that current monthly levels of rainfall are likely to alter considerably across the River Eden catchment in forthcoming decades. Current predictions, based on a medium emissions scenario, have found that by the 2080s winter rainfall at Kirkby Stephen is likely to have increased by 16% and summer rainfall will have decreased by 22%. Analysis of the long-term monthly rainfall record for Kirkby Stephen shows that an increase in winter rainfall of 16% would mean an extra 63mm of rain falling on average during the winter months. In the summer season, if total rainfall were to decrease by 22% as predicted, 47mm less rainfall than is currently received could be expected on average through June, July and August.

By editing the current monthly rainfall averages obtained from the British Atmospheric Data Centre these future predicted changes in precipitation levels were modelled into the TRIP framework. Land use cycles were held constant whilst rainfall levels were varied to determine the effects that changing precipitation levels would have on the overall fine sediment risk for the area.

Increases in winter rainfall, inputted into the TRIP model as storm events, are likely to increase fine sediment erosion risk in the River Eden catchment as shown in Figure 8.6. Although the increases in risk are relatively small, this would still lead to an increase in the amount of fine sediment being made available for transport. Increases in winter rainfall totals and the number of winter storm events have been predicted to continue beyond the 2080s; hence it is likely that fine sediment erosion risk in the winter months will continue to increase beyond the time period of this study.

When the projected decline in summer rainfall is modelled in TRIP, a reduction in fine sediment risk can be identified. Changes in levels of rainfall over the summer season would mean that by the 2080s only 2% of the total number of annual storms occurring within the catchment would occur in June and only 4% would occur in July. Such a reduction in both rainfall totals and rainfall intensities, combined with the high levels of crop cover during these months, leads to a reduction in the erosion risk. It should be noted, however, that although TRIP suggests a reduction in summer storm events will reduce fine sediment erosion risk, a feedback system between rainfall and vegetation cover could reverse this effect. Prolific reductions in summer rainfall could hinder crop growth, thus reducing vegetation cover and therefore increasing fine sediment erosion risk. Currently such a feedback mechanism cannot be accounted for within the TRIP model; however, this would be possible with future development of the current TRIP framework through linking the approach to a crop growth model, such as EPIC (AGRIS, 2012).

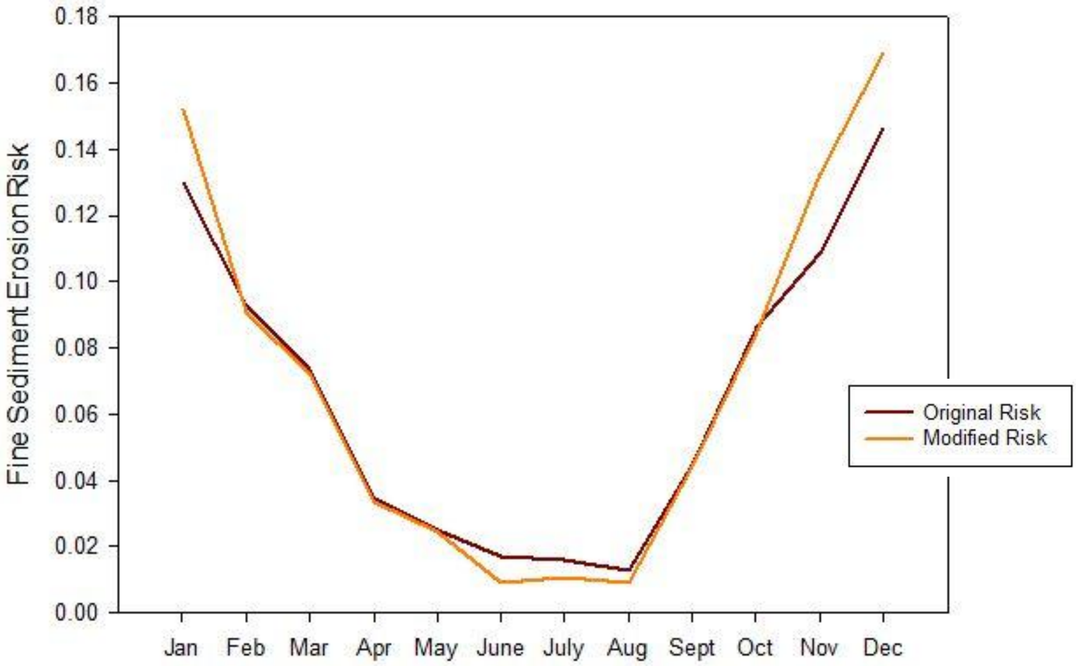


Figure 8.7: Projected changes in fine sediment erosion risk when projected climate changes are simulated in TRIP.

8.4 Projected Climate Change and Land Management Interactions

Changing patterns of precipitation alone may do little to alter fine sediment erosion risk at locations within the River Eden catchment. However, if changes in the erosive power of rainfall are combined with alterations in annual crop cover, then significant changes to erosion risk are possible. As discussed in earlier parts of this chapter, a decline in the number of autumn-sown crops grown within the River Eden catchment has been widely projected as farmers move towards crops that are cheaper to produce, such as maize, and can earn the farmer a higher price per tonne, such as oilseed rape . Potentially, this could mean that large areas of previously cultivated land are now left bare over the winter months, if the field is used to grow maize or a similar spring-sown crop, just at the time when increases in the number of storm events will be resulting in more erosive rainfall.

Figure 8.8 shows the original fine sediment erosion risks for autumn-sown barley and spring-sown barley as calculated by TRIP. Also shown are fine sediment erosion risks for these two crops when projected changes in precipitation levels are input to the TRIP framework. It can be seen that, when a change in crop cover (from autumn-sown barley to spring barley) is combined with the projected variations in precipitation patterns, fine sediment erosion risk increases substantially. This is particularly the case over the winter months when a lack of crop cover and an increase in heavy winter storms combine to create high erodibility and erosivity risks. If areas that have previously been left as grassland are bought into arable production at the same time as changes in winter precipitation levels, this will further increase the fine sediment erosion risk.

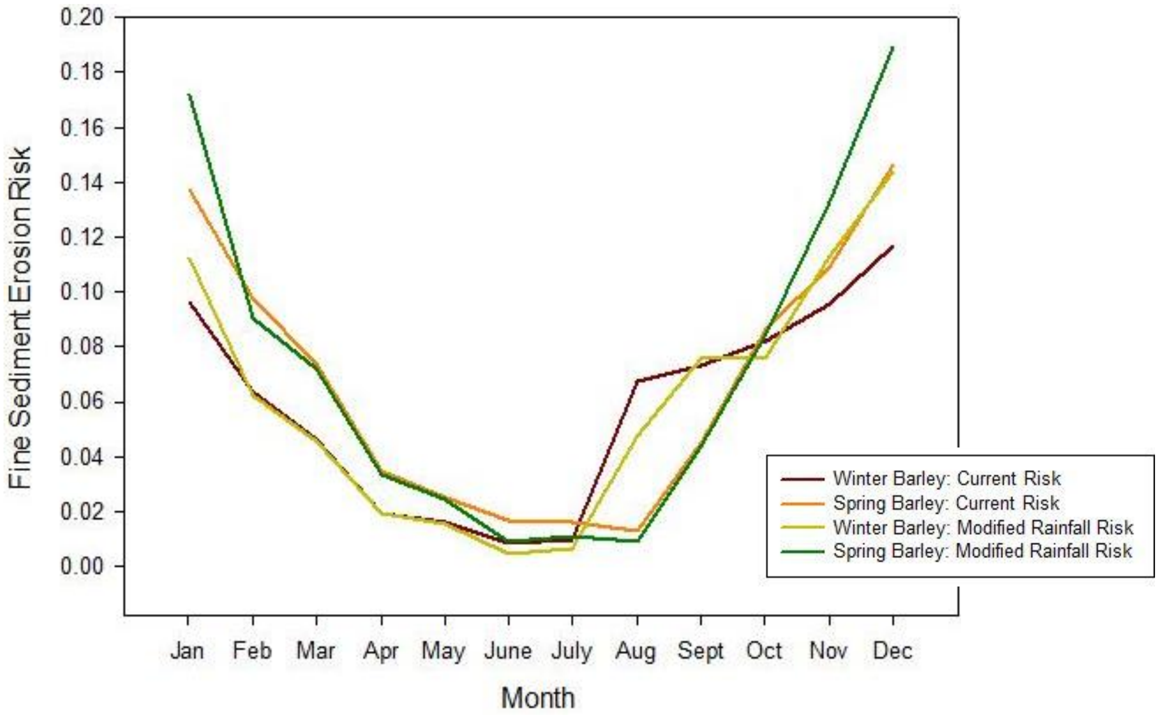


Figure 8.8: Current and future fine sediment erosion risks for winter and spring barley. The green lines represent projected changes in risk as a result of changing patterns of annual precipitation.

8.5 Future Critical Source Areas of Pollution

TRIP is only able to capture changing erodibility risks at a defined location within the River Eden catchment. Hence, it is necessary to input these values back into the SCIMAP framework to determine the likely effects that projected changes in climate and land management will have on erosivity across the catchment. By doing this, future likely locations of critical source areas of diffuse fine sediment pollution can be identified.

Initial results from the SCIMAP framework (as discussed in Chapter 6) were calculated using a network index-based approach to determine connection patterns between areas of saturated land and receiving waterways (Lane et al., 2009). In order to simulate the effect that projected climate change may have on pathways of future connectivity risk, the probability of connection was re-evaluated in this research using the network index value distribution curve (Figure 8.9) based on research conducted by Reaney et al. (2009). This approach uses projected precipitation data, collected as part of UKCP09 research, to establish future variability in the probability of hydrological connection based on a medium emissions scenario for the 2080s.

The projected variations in levels of connectivity are inputted into the SCIMAP modelling framework as part of the SAGA processing stage. When selecting a method for "mapping of netwet to connectivity" the CRUM3 2080s option is selected as opposed to the usual CRUM3 Baseline method.

Results from this analysis show the probability of hydrological connection, especially in areas with the lowest relative Network Index values, decreases under the 2080s emissions scenario when compared to current, baseline trends (Figure 8.9). A decline in the likelihood of connection can be observed for all Network Index values, but it appears that areas with the lowest Network Index values are much more likely to become disconnected under future climate change scenarios. This is likely a result of the projected reduction in levels of summer rainfall across the catchment.

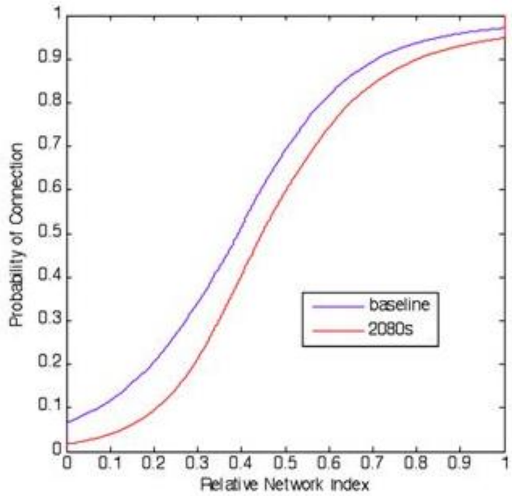


Figure 8.9: The Network Index and Probability of Connection. Source: Reaney et al. (2009)

When variations in the likelihood of hydrological connection are inputted into the SCIMAP framework, fine sediment erosion risk across the catchment is affected (Figure 8.10). Initial comparisons between the original SCIMAP output and the new modified version suggest little change in the location of critical source areas of diffuse fine sediment pollution. However, when the individual sub-layers produced as part of the modelling process are analysed further variations between the original and modified maps can be identified.

When the two network index maps are compared, differences in the number and location of areas that are most likely to connect to a receiving waterway can be identified. In the initial modelling process, the results of which were discussed in Chapter Six, areas in the wide, gently sloping landscapes of the western side of the catchment have the highest rates of surface connectivity. When baseline trends are modified and the probability of connection changed, these areas still have the highest likelihood of connection but the extent of areas with high connection risk has decreased (Figure 8.10).

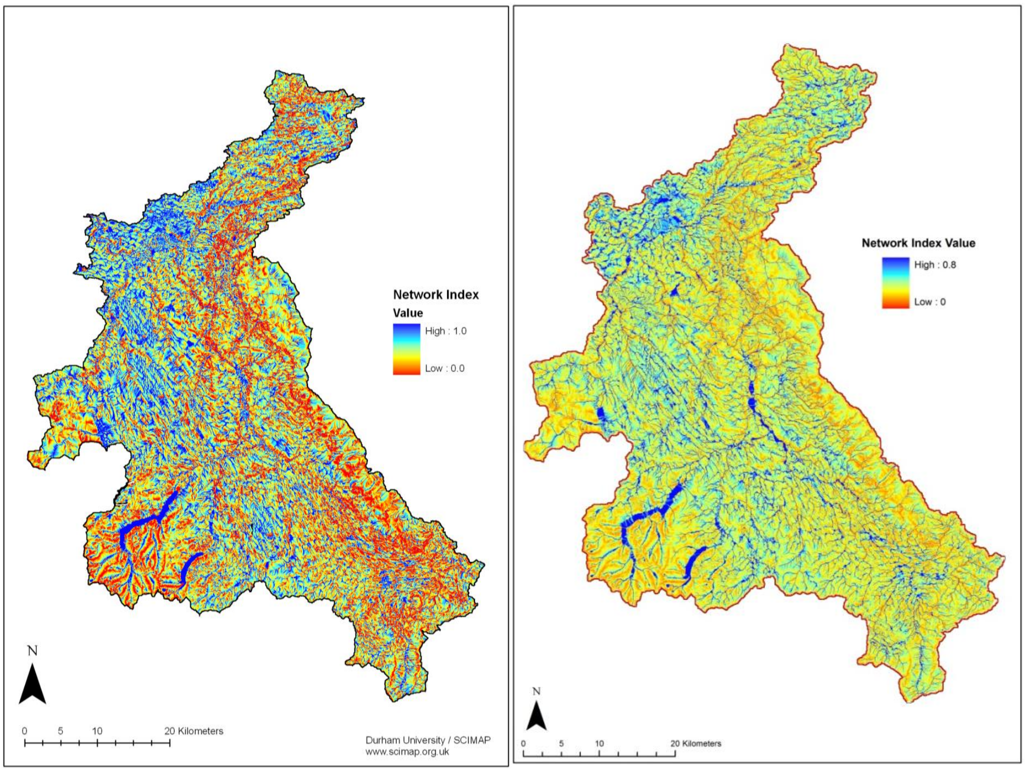


Figure 8.10: Probability of connectivity maps for the River Eden catchment. Left hand image – original map showing locations currently most likely to connect to waterways. Right hand image – modified network index values showing areas most likely to connect under projected climate change.

The final SCIMAP fine sediment erosion risk output map shows little variability in the location of critical source areas of diffuse pollution between the original baseline conditions and the modified projected conditions. It is likely, however, that the observed reduction in the number of hydrologically connected areas may mean that areas previously at high risk of being source areas of fine sediment erosion are no longer at such high risk. Although these areas will still be capable of producing fine sediment material, they will not be connected to a waterway hence the risk will remain captured at its source.

In the north-west of the catchment, where areas are likely to become much less connected as a result of projected climate change, subsequent fine sediment erosion risks are also likely to decline. Conversely, in the east of the catchment, where rates of connection are likely to increase as upland areas experience increasing amounts of winter rainfall, associated fine sediment pollution risks are likely to increase.

To ensure successful management of the effects that projected future climate change may have on fine sediment erosion risk across the River Eden catchment, further research into the potential changes to precipitation patterns is required. Currently research is based upon UKCP09 2080s medium emissions scenarios. It is highly possible therefore that the degree of projected climate variability may be under or overestimated and the climate conditions in the decades leading up to the 2080s will, to a degree, determine the catchments response to climate change.

It is important to consider the projected changing pattern of annual rainfall with amount of winter rainfall increases and summer rainfall decreasing. This may mean that there is greater winter connectivity and less in the summer. This scenario could lead to a non-linear increase in diffuse pollution export due to the greater rainfall at the times of the year when the soil surface is most vulnerable.

It is also important to pay close attention to the fact that projected variations in the frequency and total amounts of precipitation alone will not control future patterns of hydrological connectivity and associated diffuse fine sediment erosion risk. The projected changes to patterns of precipitation therefore need to be combined with associated variations in soil moisture, vegetation cover and river flows to accurately establish the effects that climate change will have on the hydrology of the River Eden catchment.

8.6 Socio-Economic Considerations

Much of this chapter has sought to determine the most effective way of managing arable farmland so as to minimise fine sediment erosion risk in the River Eden catchment. However, what is and what is not appropriate and effective management cannot be assessed on an environmental basis alone. Instead social and economic factors need to be considered alongside environmental ones to find the optimum management measure.

It has now become widely recognised that farming systems need to work within the boundaries of natural resources whilst still providing the highest standards of production. An important challenge for policy makers across the UK is to find suitable programmes of land management which maintain and enhance food production whilst also improving the overall health of environmental systems and the rural economy (Defra, 2002). Recent government initiatives and programmes of investment are working hard to increase the amount of food grown within the UK. In order to achieve these targets, farmers will be under obligation to increase overall efficiency and productivity and these requirements need to be carefully balanced alongside the issue of reducing fine sediment erosion risk. Many farmers will have quotas to meet as part of the conditions of their Single Farm Payment and hence the productivity of the farm needs to be taken into account when implementing management designed to minimise fine sediment erosion risk.

As discussed in Chapter 8.1, results obtained from the TRIP model show spring-sown barley and maize to be the two crops with the highest associated risk of fine sediment pollution. When considering the optimum crop cycle for minimising fine sediment erosion risk, these two crops should therefore be discounted and farmers should be actively encouraged not to plant them. However, the costs associated with growing barley and maize are considerably lower than costs for wheat or oilseed rape (Table 8.2). From a financial perspective, these are therefore very appealing to farmers. Furthermore, the hardy nature of barley makes it an easier option for many farmers as it can be grown on poorer soils than many other crops and need little attention once planted. Hence, in areas that cannot support winter wheat, it is often used as an alternative. Clearly, rural land owners will want to achieve maximum yield from all areas of their farm and so will often plant a spring barley crop on land which otherwise would probably remain fallow. If farmers do plant spring-sown crops, then the environment impacts need to be taken into account and appropriate mitigation measures implemented.

Crop	Seed	Fertiliser	Sprays	Other	Total
Winter Wheat	£79	£212	£123	£0	£424
Winter Barley	£76	£200	£86	£11	£373
Spring Barley	£66	£138	£56	£8	£268
Maize	£181	£45	£53	£145	£467
Oilseed Rape	£45	£180	£98	£0	£323
Grassland - Grazing	£16	£84	£15	£0	£115

Table 8.2: Commonly grown UK crops and their associated costs (£/ha). Source: SAC (2010).

Like spring barley, maize is associated with high rates of fine sediment erosion risk. Despite this, however, it is still a popular crop within the UK and is grown widely on farms across the country. Although the initial costs of planting maize are high, mainly due to contractors normally being required for planting and harvesting, the crop then requires little spraying and, once successfully established, yields can be very high (SAC, 2010). Growing maize is somewhat of a pollution trade-off: more fine sediment is eroded but fewer pesticides are added to the crop thus reducing the risk to nearby waterways posed by nitrate and phosphate.

In recent years, the costs of fertilisers and sprays have increased considerably; many farmers are now attempting to turn away from traditional autumn-sown wheat that require large amounts of spraying and are instead concentrating on finding crops that require little attentions once planted. These minimal costs and high yields make maize an appealing option to farmers. Furthermore, maize is commonly used as livestock feed, farmers are unwilling to cease growth as this would force them to buy in a greater proportion of their cattle feed.

The final price that farmers receive per tonne varies considerably between crops. For a standard yield of wheat, a farmer can expect to receive £125 per tonne and for barley the average rate is currently £105 per tonne (SAC, 2010). Oilseed rape, however, can often earn a farmer in excess of £270 per tonne and so is much more financially profitable. Although oilseed rape requires a relatively large amount of fertiliser, its overall profit margin is much greater than the margins available for wheat and barley thus if land is suitable for oilseed rape it can be an extremely profitable use of space for farmers. Although winter cereals can produce yields almost 20% higher when compared to spring cereals, an overall change in the markets means farmers can earn more from a smaller yield of oilseed rape or maize than a large yield of wheat.

Historically, a lack of awareness by farmers and policy makers, along with subsidies and financial incentives, has led to farmers growing arable crops on erosion prone land (Boardman et al., 1994). Attempts at growing cereals in such unsuitable areas have resulted in increases in the erodibility of soils (Burt, 2001) and hence the strength resistance of soil to erosion by overland flow has decreased (Brunsden, 1993). Over the course of the past decade it has been recognised that a fundamental change in the way agricultural subsidies are managed is required if the scheme is going to remain financially sustainable, whilst also minimising environmental risks (Defra, 2010). As a result, a range of policy measures and management tools are currently available for encouraging changes in the way farmers manage their land (Pretty et al., 2010). These include the use of regulatory and legal measures along with subsidies and financial incentives and it has been suggested that successful management depends upon the use of a mixture of these options. In the UK such schemes have already been initiated and payments are now made to many farmers to encourage the conservation of landscapes, their wildlife and history (Natural England, 2011).

A recent Defra-commissioned study found that almost 40% of UK farms would fail to make a profit if subsidies were abolished and only 5% would be in a stable enough financial position to be able to contemplate expansion and reinvestment (Defra, 2010). The removal of subsidies could lead to changes in the type of products UK farmers produce as they move to less risky crops, cheaper production methods and different scales of production.

Any future land management projects therefore need to minimise diffuse fine sediment erosion risk whilst at the same time ensuring recommendations are economically viable for farmers. Fine sediment erosion risk along with economic and social considerations will vary considerably across a catchment, particularly one as large as the River Eden; hence management needs to be carefully tailored to meet the needs of each individual area.

8.7 Conclusions

This chapter has highlighted the important role that winter crop cover plays in minimising fine sediment erosion risk. Furthermore, it has been seen that a decline in rates of incorporation of grassland in crop rotations is likely to lead to increased fine sediment erosion risks across the River Eden catchment. An increase in the amount of maize being grown within the catchment is likely to exacerbate this problem further.

Through analysing several crop cycles within a newly created temporal risk framework, it has been shown that a five-year cycle which incorporates three years of an autumn-sown crop such as barley or wheat before two years of set-aside grassland is the optimum cycle for minimising fine sediment erosion risk within the catchment. Crops of maize are associated with the highest fine sediment erosion risks. This is a direct result of land being left bare without vegetation cover over the wet winter months.

It is not only land management changes that have the potential to significantly alter fine sediment pollution risk dynamics within the study catchment; the effects of changing patterns of precipitation are likely to have significant effects on hydrological connectivity across the River Eden catchment. By recoding parts of the SCIMAP framework, it has been possible to determine how the location of critical source areas of diffuse fine sediment pollution are likely to change in future decades. Results from this analysis have shown areas with the lowest network index values are less likely to be connected to receiving waterways under future emissions scenarios.

This chapter concludes by highlighting the importance of considering social and economic driving factors when assessing available management strategies for a location. Without such assessment, it is highly possible that the management tools which are implemented may be unsuitable for the location in question and could disrupt the social and economic balance of an area.

Chapter 9: Research Synthesis and Conclusions

9.1 Research Overview

This research has investigated the potential impacts of projected climate change upon hydrological connectivity and associated diffuse fine sediment pollution risk in an upland UK river catchment. Previous research has shown it is highly likely that patterns of connectivity will alter in response to changes in the intensity, duration and frequency of rainfall events (Reaney et al., 2011). However, little research has focused on the effects that such changes may have on altering the risk of diffuse fine-sediment pollution by changing patterns of water movement through catchments (Pringle, 2003; Reaney et al., 2011). Therefore, this project was split into three research objectives. The three objectives were designed to: 1) assess current hydrological connectivity dynamics within the River Eden catchment; 2) simulate future diffuse pollution dynamics under projected climate change scenarios; and 3) study the relationship between diffuse fine sediment pollution risk and rural land management.

This chapter briefly re-iterates how each research aim was investigated and outlines the key results from each. In the latter part of this chapter recommendations for future research are made.

9.2 Research Aims: Key Findings

1) ***Conceptualise and assess the interactions between climate change, hydrological connectivity and diffuse pollution risk within the River Eden Catchment using the SCIMAP framework.***

Current hydrological connectivity dynamics and diffuse fine sediment risk within the River Eden catchment were modelled using the SCIMAP framework. The basis of the SCIMAP analysis is the joint consideration of the probability of a unit of land producing a particular environmental risk and then of that risk reaching the drainage network (Lane et al., 2006). Hydrologically well-connected and risky land uses should be the prime focus of management activities, and hence the result of SCIMAP is a method for determining where finite management resources should be concentrated in order to achieve optimum environmental protection (Lane et al., 2006).

Initially the intermediate layers of the SCIMAP modelling process are analysed independently of each other to see which areas had 1) the highest amounts of energy available for erosion, 2) the highest erosivity risk and 3) the highest probability of becoming hydrologically connected to a receiving waterway. From the SCIMAP framework it was found areas with the highest amounts of energy available for erosion are found in the upland areas, particularly in the western fringes of the catchment in the upland parts of the Lake District hills. The heightened risk at these locations can be attributed to the high levels of annual precipitation. Increasing rainfall levels are usually matched by increasing rainfall intensity and raindrop size; this reduces soil capacity for infiltration and thus increases the likelihood of infiltration-excess runoff later on during storm events. Furthermore, the steep slopes in these areas serve to act as a way of channelling water off the hillslope very quickly, creating overland flow travelling at erosive velocities.

Areas with the lowest levels of erosive energy were found to be in the valley bottoms, around the main channel of the River Eden. This is most likely a result of these areas receiving less than 40% of the rain that falls annually in upland areas and the low slope gradients.

For a point to export risk a continuous connected flow to the river channel network is required. This was assessed in the SCIMAP framework through the Network Index based prediction of hydrological connectivity, calculated from the prediction of the spatial pattern of soil moisture and hence the susceptibly of each point in the landscape to generate saturation-excess overland flow and allow overland flow to cross that part of the landscape. Results show that areas in the wide, gently sloping landscapes of the western side of the catchment have the strongest surface connectivity. The concave nature of the slopes in this region makes the lower parts of the slopes prone to saturation and hence during storms events they quickly become active and saturation-excess runoff is a common occurrence. It is likely therefore, that any fine sediment eroded at these locations will quickly become connected to a receiving waterway thus transferring the risk from its source area further downstream.

Areas of high connection risk, when combined with polluting areas, become critical source areas for diffuse pollution (Lane et al., 2006). To establish the location of these potentially polluting areas, each point within the catchment was assigned a risk value signalling the likelihood of the generation of fine sediment risk as parametised by expert judgement based on landscape parameters, in this case land cover (Lane et al., 2006). Each land cover classification was scaled between 0 (lowest risk) to 1 (highest risk) based upon its potential for producing soil erosion. Results show that once these land cover weightings are integrated into the SCIMAP modelling process, the focus for concern shifts away from the high, upland areas of the catchment to the central lowland parts of the River Eden valley. Land in these lowland areas is predominantly improved grassland or used to grow arable crops meaning it is likely that it will be regularly disturbed through ploughing and other forms of intensive management. Furthermore, in areas where land is used to grow arable crops soils are likely to be left bare for large amounts of time further heightening the risk of fine sediment erosion.

The results from applying the SCIMAP model to the River Eden catchment show areas at greatest risk of being critical source areas of diffusive fine sediment pollution are found in the central lowland areas of the catchment. Risk here is high as a result of the interactions between high levels of hydrological connectivity and risky land uses. In this instance it is therefore these areas where management resources would be best targeted in order to minimise fine sediment erosion risk.

2) *To assess what a changing pattern of hydrological connectivity means for changing fine sediment diffuse pollution risk.*

It was a hypothesis of this research that patterns of hydrological connectivity will alter in response to changes in the intensity, duration and pattern of rainfall (Reaney et al., 2007; Bracken & Croke, 2007). Recent research has predicted changes in the size of hydrologically well-connected areas and the number and frequency of connection-disconnection cycles (Reaney et al., 2009). In order to simulate predicted changes in patterns of hydrological connection UKCP09 projections of precipitation and temperature were combined with CRUM3, a hydrological model which can predict changes in the hydrological connectivity (Lane et al., 2009). By developing a relationship between these two models' predictions of hydrological connectivity, it was possible to import different relationships between the Network Index and the predicted connectivity under different climates. An assessment was then made on the potential impact of changing climate on fine sediment diffuse pollution risk export.

Results from this analysis show the probability of hydrological connection, especially in areas with the lowest relative Network Index values, decreases under the 2080s emissions scenario when compared to current, baseline trends. When the two network index maps are compared, differences in the number and location of areas that are most likely to connect to a receiving waterway can be identified. In the initial modelling process, the results of which were discussed in Chapter Six, areas in the wide, gently sloping landscapes of the western side of the catchment have the highest rates of surface connectivity. When baseline trends are modified and the probability of connection changed, these areas still have the highest likelihood of connection but the extent of areas with highest connection risk has decreased.

3) *To assess how changing patterns of land management may affect fine sediment diffuse pollution risk.*

Connection pathways and diffusive pollution risk are closely linked to management practices at both the catchment and the field scale (Walling & Quine, 1991). Fine sediment erosion risks associated with various crops were investigated using a newly developed risk filter (Temporal Risk Integration Process, TRIP) which calculates a time integrated risk by analysing the temporal patterns of vegetation cover and precipitation on annual and inter-annual time scales.

The TRIP model works on the basis that fine sediment erosion risk within a catchment is a product of the erosive force of the rainfall, the protection to the soil surface offered by vegetation cover and the natural erodibility of the soil. TRIP is able to capture both temporal variability in fine sediment erosion risk as a result of changing crop cover and farming practices whilst at the same time having the capacity to capture the effects that future rural land management and projected climate change may have on risk levels. The objective was to optimise land management so as to minimise the risk of fine sediment reaching receiving waters. The suitability of these practices was discussed, jointly considering the economic and social implications alongside environmental factors. From this, necessary changes to land management practices, to mitigate against the effects of projected climate change upon diffuse pollution risk, were identified.

The standard SCIMAP framework does not account for temporal variations in risk; hence, to enable an assessment of monthly variation in risk, the temporally integrated framework was developed. In order to successfully run the model, accurate measurements of percentage vegetation ground cover during each month of the year and average monthly rainfall totals were required. Monthly precipitation totals were obtained from the long-term daily rainfall record at Appleby, Cumbria, UK, which dates from 1891. Vegetation cover, in three test fields in the River Eden catchment, was surveyed on a monthly basis using a Microdrones MD4-200 remote controlled helicopter with a 12 megapixel camera attached. Each image was taken at a height of approximately 50 metres and images were collected on days when skies were mostly bright and clear to ensure the quality of the image was the best possible. These pictures were then analysed, using a newly written piece of programming code, to determine what percentage of the ground was covered with vegetation and what was not.

The results from the aerial image analysis found that total crop ground cover during the summer months can be as high as 90%; however, earlier on in the growing season percentage cover is often lower than 20%, leaving soils very exposed to erosion. Between the months of May and July autumn-sown crops go through a period of rapid growth when percentage ground cover increases from approximately 45% to over 80%.

Monthly vegetation cover was predicted for six standard crop cycles, found throughout the River Eden catchment. Annual fine sediment erosion risks were then obtained for each in order to determine which of the cropping cycles is likely to be causing the largest fine-sediment erosion risk and which is likely to result in the lowest risk. It was found that autumn-sown crops offered the best protection to soils as some growth is possible before the start of the wet winter months when erosivity levels are highest. The highest risks of diffuse fine-sediment pollution are associated with crops of spring-sown barley and maize. These are usually planted in March and April of a given year and harvested in September of the same year thus meaning during the winter months soils are left bare and exposed. Although these crops are therefore the least effective at minimising fine sediment risk, they are widely used by farmers across the UK due to their low fertiliser requirements and hardy nature.

Results from the TRIP framework highlight the importance of managing fine-sediment erosion risk during November and December, which are often two of the wettest months of the year with intense winter storms. Outputs from the TRIP model show oilseed rape to be the optimum crop for minimising risk at this time of year. Having been planted in August oilseed rape crops are able to take advantage of the warm late summer weather and autumn rainfall, percentage ground cover by November is therefore 25-30%. This is in contrast to other crops, for example autumn-sown wheat, which are planted later on in the autumn and have ground cover of 15-20% by late December.

Once risk values had been obtained for each of the original crop cycles, alterations to the cycles were made to incorporate possible future land use changes. Firstly, the predicted decline in the use of winter cereal crops and their replacement with spring varieties was simulated. Results from the TRIP model show that when a typical winter crop, such as wheat, is replaced by spring-sown barley, the fine-sediment erosion risk for the location in question increases considerably. This is particularly the case in the first quarter of the year when the spring barley has yet to be sown, hence soils are completely bare. From the TRIP model outputs it can therefore be seen that the use of a autumn-sown crop is an extremely effective way of minimising fine sediment erosion risk. In the winter months (December, January, February) the erosion risk associated with arable land being used to grow any winter crop is considerably lower than the risk on land where a non-winter cover crop is being grown.

Secondly, the effect that a decrease in the amount of land left as set-aside grassland would have on fine-sediment risk was analysed. A typical five-year crop cycle was modified by replacing the two years of grassland at the end of the cycle with two further years of autumn-sown crops. Results show that removing the two years of grassland from the end of the cycle significantly increases fine sediment erosion risk at a location. The results obtained from the TRIP model thus highlight the importance of incorporating years of set-a-side into five-year crop cycles to ensure soils are offered a degree of protection from the risk of fine-sediment erosion for at least part of the crop cycle. This is particularly important in areas where fine-sediment erosion is known to be a problem; a winter covering of grass can significantly reduce erosion risks at the time of year when highly erosive storms are most likely. Similarly, by incorporating periods of grasslands into cycles soils will be given time to recover from previous cropping seasons.

Projected climate change has the potential to dramatically alter the connectivity and fine-sediment dynamics of river catchments across the UK. A reduction in rainfall totals and rainfall intensity during the summer months, combined with the high levels of crop cover is likely to lead to a reduction in the erosion risk. It should be noted, however, that although TRIP suggests a reduction in summer storm events will reduce fine-sediment erosion risk, a feedback system between rainfall and vegetation cover could reverse this effect. Large reductions in summer rainfall could hinder crop growth, thus reducing vegetation cover and therefore increasing fine sediment erosion risk. Currently, such a feedback mechanism cannot be accounted for within the TRIP model.

The full implications of future changes in land management and the effects that this may have on diffuse fine-sediment pollution risk are still not completely understood. In future decades, it is likely that projected climate change will result in further changes to the way agricultural land around the UK is managed and changing patterns of hydrological connectivity are likely as a result of this. At the same time, however, changes to the way agricultural land is used and managed will also be driven by economic factors including market prices and the availability of government subsidies. It may therefore be that economic factors continue to be the main driving force behind agricultural land management and associated variability in hydrological connectivity as opposed to projected global climate change.

9.3 Recommendations for Future Research

In order to further develop the results obtained from this research, several recommendations for future studies are made. These include suggestions as to how the methods used in this research could be improved and how best to apply the research carried out here to other UK river catchments.

9.3.1 Validation of SCIMAP Outputs

Some discussion is given in this research on the attempts to compare the results from the SCIMAP framework against spatially distributed electrofishing data (Chapter 6.8). Time constraints on this study meant independent analysis of electrofishing data could not be carried out and all attempts at validation rely on previous work by the Eden Rivers Trust and Reaney et al. (2011). One recommendation for future research would therefore be for a more robust attempt at comparing the outputs from the SCIMAP modelling process. This could be achieved by an in-depth comparison between electrofishing survey results from the River Eden and SCIMAP models, by using Environment Agency suspended sediment survey results or gravel mapping data from the Eden Rivers Trust. It has also been suggested that field surveys could be used to assess possible source areas of fine sediment; comparisons could then be made between these locations and locations of high risk as highlighted within the SCIMAP framework.

9.3.2 Development of Results

Results obtained within this research could be further developed by collating aerial photographs from an increased number of locations located across the River Eden catchment. Although the fields surveyed in this research at Sewborwens Farm were deemed to be an acceptable representation of agricultural land use in the catchment, further collection of images would allow patterns of vegetation cover in areas of different land use to be captured. Hence, a more accurate picture of the fine sediment risk associated with these land covers could be developed.

9.3.3 Application of Approach to Other Catchments

Whilst it is important to remember that no two river catchments are the same, the methods used during this research can be applied to other upland catchment across the UK. One approach would be to apply the SCIMAP framework and TRIP model to the other catchments selected as part of the DEFRA Demonstration Test Catchment project. These are the River Wensum catchment in Norfolk and the River Avon catchment in Hampshire. Alternatively, the methods used could be applied in similar catchments across northern parts of the UK. Detailed monitoring is already currently being carried out on several catchments including the River North Esk and the River Dee, both near Aberdeen (Northern Rivers Institute, 2011), and large amounts of the data required for both SCIMAP and TRIP have already been collated.

9.3.4 Distribution of SCIMAP Framework

Currently, the SCIMAP framework and the data produced from it are available for public use but require some specialist GIS knowledge to use the system. To make the results more widely available, it would be viable to suggest that the X and Y coordinate locations of points within the catchment, along with their associated risk values, be exported into KML format and used within the Google Earth system. The Google Maps set up is a user-friendly, public GIS system which includes both satellite and terrain data along with street maps. Integrating the SCIMAP framework into Google Earth would enable the framework to be used by a larger number of stakeholders who could then discuss and investigate the results being produced by SCIMAP further. An example of this can be seen as part of the work currently being carried out on the River Eden catchment as part of the Demonstration Test Catchment project (EdenDTC, 2012)

9.4 Concluding Remarks

This study aimed to address the effects that changes in spatial and temporal connectivity dynamics would have on pollution risk in the context of future climate change. Connectivity dynamics were modelled using the established SCIMAP risk framework and projected climate change, based on predictions from the UKCP09 study, were incorporated into the model based on previous work using the Network Index approach (Reaney et al., 2011). In the case of the River Eden catchment, this research has found that projected changes in seasonal precipitation and associated changes in land use and land management have the potential to alter the location of critical source areas of diffuse fine-sediment pollution. Furthermore, the TRIP framework which was developed as part of this research to address a well known weakness in SCIMAP, has highlighted temporal variations in risk in the catchment. The framework has also highlighted the important role winter cereal crops play in minimising fine sediment erosion risk and has captured the effect that the incorporation of periods of set-aside grassland into typical five year crop cycles can have on minimising risk. By carrying out simulations using the TRIP framework the most effective ways of minimising fine sediment erosion risk within the catchment have been identified. Results show that a five year crop cycle incorporating three years of autumn-sown crops to be the most effective and yet still economically viable, option for minimising risk. Overall, the combined SCIMAP-TRIP analysis framework has been shown to have the capability to tackle the difficult issues of making predictions for the management of diffuse pollution under changing climate and land management.

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Appendix 1: MATLAB Image Processing Code

```
image1=imread('IMGP0274.jpg');
figure,imagesc(image1); title('Orignal');
h=fspecial('average', 70);

for n=1:3
   S(:,:,n)=filter2(h,image1(:,:,n),'same');
end

%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%
%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%

nColors = 3;
sample_regions = false([size(image1,1) size(image1,2) nColors]);

A=[673 687 676 656 654 659 673];
B=[809 783 743 737 763 794 808];

C=[3747 3800 3800 3760 3747];
D=[1593 1668 1660 1598 1593];

E=[2176 2186 2180 2176];
F=[1704 1695 1689 1704];

   sample_regions(:,:,1) = roipoly(image1,A,B);
   sample_regions(:,:,2) = roipoly(image1,C,D);
   sample_regions(:,:,3) = roipoly(image1,E,F);

%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%
%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%

cform = makecform('srgb2lab');
lab_image1 = applycform(image1,cform);

%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%
%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%


a = lab_image1(:,:,2);
b = lab_image1(:,:,3);
color_markers = repmat(0, [nColors, 2]);

for count = 1:nColors
  color_markers(count,1) = mean2(a(sample_regions(:,:,count)));
  color_markers(count,2) = mean2(b(sample_regions(:,:,count)));
end

%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%
%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%

disp(sprintf('[%0.3f,%0.3f]',color_markers(2,1),color_markers(2,2)));
```

```
%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%
%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%

color_labels = 0:nColors-1;

a = double(a);
b = double(b);
distance = repmat(0,[size(a), nColors]);

for count = 1:nColors
  distance(:,:,count) = ( (a - color_markers(count,1)).^2 + ...
                  (b - color_markers(count,2)).^2 ).^0.5;
end

[value, label] = min(distance,[],3);
label = color_labels(label);
clear value distance;
%Step 4: Display Results of Nearest Neighbor Classification
       %The label matrix contains a color label for each pixel in the image1 image. Use the label
matrix to separate objects in the
         %original image1 image by color.
       rgb_label = repmat(label,[1 1 3]);
segmented_images = repmat(uint8(0),[size(image1), nColors]);

for count = 1:nColors
  color = image1;
  color(rgb_label ~= color_labels(count)) = 0;
  segmented_images(:,:,:,count) = color;
end

figure,imagesc(segmented_images(:,:,:,1)), title('Crop');
figure,imagesc(segmented_images(:,:,:,2)), title('Not crop dark');
figure,imagesc(segmented_images(:,:,:,3)), title('Not crop light');
%figure,imagesc(segmented_images(:,:,:,4)), title('Water');
%figure,imagesc(segmented_images(:,:,:,5)), title('Road');
%figure,imagesc(segmented_images(:,:,:,6)), title('Road markings');

A=(segmented_images(:,:,:,1));
mask=A(:,:,:)>0;

figure,imagesc(mask); title('Mask');
number=pixcount(mask);
number=0;
for n=1:size(mask,1)
   for m=1:size(mask,2)
      if mask(n,m)==1
         number=number+1;

      end
   end
end
C=(number);
```

Appendix 2: Field Survey Results From Sewborwens Farm

C = Vegetation Present

0m	10	20	30	40	60	70	80	90	**1m**	110	120	130	140	150	160	170	180	190	**2m**	210	220	230
	C	C		C	C	C	C	C	C	C	C	C	C	C	C		C	C	C	C	C	C
240	250	260	270	280	290	**3m**	310	320	330	340	350	360	370	380	390	**4m**	410	420	430	440	450	460
C	C	C	C	C	C		C	C	C	C	C			C	C		C	C	C	C	C	
470	480	490	**5m**	510	520	530	540	550	560	570	580	590	**6m**	610	620	630	640	650	660	670	680	690
C	C	C	C	C	C	C	C	C	C	C	C		C	C	C	C	C				C	
7m	710	720	730	740	750	760	770	780	790	**8m**	810	820	830	840	850	860	870	880	890	**9m**	910	920
C	C	C	C	C	C	C	C	C	C	C	C		C	C	C	C	C	C		C	C	
930	940	950	960	970	980	990	**10m**															
C	C	C	C	C	C	C	C															

0m	10	20	30	40	60	70	80	90	**1m**	110	120	130	140	150	160	170	180	190	**2m**	210	220	230
	C	C	C	C	C	C	C		C	C	C	C	C	C	C	C	C	C	C	C	C	C
240	250	260	270	280	290	**3m**	310	320	330	340	350	360	370	380	390	**4m**	410	420	430	440	450	460
C	C		C	C	C	C	C		C	C	C	C	C	C		C	C	C	C	C	C	C
470	480	490	**5m**	510	520	530	540	550	560	570	580	590	**6m**	610	620	630	640	650	660	670	680	690
C	C	C	C	C	C	C			C	C	C	C	C	C	C	C	C	C	C	C	C	C
7m	710	720	730	740	750	760	770	780	790	**8m**	810	820	830	840	850	860	870	880	890	**9m**	910	920
C	C	C	C	C	C	C	C	C		C	C	C	C	C	C	C	C	C	C	C	C	
930	940	950	960	970	980	990	**10m**															
C	C		C	C	C	C	C															

0m	10	20	30	40	60	70	80	90	**1m**	110	120	130	140	150	160	170	180	190	**2m**	210	220	230
	C	C	C	C	C	C	C	C	C	C		C	C	C	C	C	C	C	C	C	C	C
240	250	260	270	280	290	**3m**	310	320	330	340	350	360	370	380	390	**4m**	410	420	430	440	450	460
C	C	C	C	C	C	C	C	C	C	C		C	C	C	C	C	C	C	C	C	C	C
470	480	490	**5m**	510	520	530	540	550	560	570	580	590	**6m**	610	620	630	640	650	660	670	680	690
C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C
7m	710	720	730	740	750	760	770	780	790	**8m**	810	820	830	840	850	860	870	880	890	**9m**	910	920
C	C	C	C	C	C	C	C	C	C		C	C	C		C	C	C	C		C	C	C
930	940	950	960	970	980	990	**10m**															
C	C	C	C	C	C	C	C															

FIELD ONE **Field Centre** **300 measurements > 29 without vegetation > crop cover of 90%.**

0m	10	20	30	40	60	70	80	90	**1m**	110	120	130	140	150	160	170	180	190	**2m**	210	220	230
	C	C		C	C	C	C						C	C	C		C	C	C	C	C	C
240	250	260	270	280	290	**3m**	310	320	330	340	350	360	370	380	390	**4m**	410	420	430	440	450	460
C	C	C	C	C	C							C	C	C	C		C	C	C	C	C	
470	480	490	**5m**	510	520	530	540	550	560	570	580	590	**6m**	610	620	630	640	650	660	670	680	690
C					C	C	C	C	C	C	C		C	C	C	C	C				C	
7m	710	720	730	740	750	760	770	780	790	**8m**	810	820	830	840	850	860	870	880	890	**9m**	910	920
C	C	C	C	C	C	C	C	C	C	C	C		C	C	C	C	C	C		C	C	
930	340	950	960	970	980	990	**10m**															
C	C				C	C	C															

0m	10	20	30	40	60	70	80	90	**1m**	110	120	130	140	150	160	170	180	190	**2m**	210	220	230
	C	C	C	C	C	C	C		C	C	C	C	C	C						C	C	C
240	250	260	270	280	290	**3m**	310	320	330	340	350	360	370	380	390	**4m**	410	420	430	440	450	460
C	C		C	C	C	C	C		C			C	C	C		C	C	C	C	C	C	C
470	480	490	**5m**	510	520	530	540	550	560	570	580	590	**6m**	610	620	630	640	650	660	670	680	690
C		C	C	C	C	C			C	C	C	C	C	C	C		C		C	C	C	C
7m	710	720	730	740	750	760	770	780	790	**8m**	810	820	830	840	850	860	870	880	890	**9m**	910	920
C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C						
930	940	950	960	970	980	990	**10m**															
C	C	C	C	C	C	C	C															

0m	10	20	30	40	60	70	80	90	**1m**	110	120	130	140	150	160	170	180	190	**2m**	210	220	230
	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C
240	250	260	270	280	290	**3m**	310	320	330	340	350	360	370	380	390	**4m**	410	420	430	440	450	460
C	C	C	C	C	C	C	C	C	C	C		C	C	C	C	C	C	C		C	C	C
470	480	490	**5m**	510	520	530	540	550	560	570	580	590	**6m**	610	620	630	640	650	660	670	680	690
C	C	C	C	C	C				C	C	C	C	C	C	C	C	C	C	C	C	C	C
7m	710	720	730	740	750	760	770	780	790	**8m**	810	820	830	840	850	860	870	880	890	**9m**	910	920
C	C	C	C	C	C	C	C	C	C		C	C	C	C	C	C	C	C		C	C	C
930	940	950	960	970	980	990	**10m**															
C	C	C	C	C	C	C	C															

<u>FIELD ONE</u> **Field Edge** **300 measurements > 61 without vegetation > crop cover of 80%.**

Appendix 2

0m	10	20	30	40	60	70	80	90	**1m**	110	120	130	140	150	160	170	180	190	**2m**	210	220	230
C	C	C	C	C	C	C	C	C				C		C	C		C			C	C	C
240	250	260	270	280	290	**3m**	310	320	330	340	350	360	370	380	390	**4m**	410	420	430	440	450	460
	C	C					C			C		C	C	C	C	C		C	C	C		C
470	480	490	**5m**	510	520	530	540	550	560	570	580	590	**6m**	610	620	630	640	650	660	670	680	690
C	C	C	C	C	C	C	C	C	C	C	C		C	C	C	C	C	C	C	C	C	C
7m	710	720	730	740	750	760	770	780	790	**8m**	810	820	830	840	850	860	870	880	890	**9m**	910	920
C				C	C		C	C	C	C	C		C	C	C	C	C	C	C	C	C	C
930	340	950	960	970	980	990	**10m**															
C	C	C	C	C	C	C	C															
0m	10	20	30	40	60	70	80	90	**1m**	110	120	130	140	150	160	170	180	190	**2m**	210	220	230
	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C
240	250	260	270	280	290	**3m**	310	320	330	340	350	360	370	380	390	**4m**	410	420	430	440	450	460
	C	C	C		C	C	C					C	C			C	C	C	C			
470	480	490	**5m**	510	520	530	540	550	560	570	580	590	**6m**	610	620	630	640	650	660	670	680	690
C	C	C	C	C	C	C			C	C	C	C				C	C	C	C			
7m	710	720	730	740	750	760	770	780	790	**8m**	810	820	830	840	850	860	870	880	890	**9m**	910	920
						C													C	C	C	C
930	940	950	960	970	980	990	**10m**															
		C	C	C	C	C																
0m	10	20	30	40	60	70	80	90	**1m**	110	120	130	140	150	160	170	180	190	**2m**	210	220	230
				C	C	C	C	C	C				C	C	C			C	C	C	C	C
240	250	260	270	280	290	**3m**	310	320	330	340	350	360	370	380	390	**4m**	410	420	430	440	450	460
C		C	C	C		C	C	C	C	C	C				C	C	C			C	C	
470	480	490	**5m**	510	520	530	540	550	560	570	580	590	**6m**	610	620	630	640	650	660	670	680	690
		C	C	C	C	C	C	C						C	C	C	C	C	C	C	C	C
7m	710	720	730	740	750	760	770	780	790	**8m**	810	820	830	840	850	860	870	880	890	**9m**	910	920
C	C		C	C		C	C	C	C	C	C	C	C		C	C		C	C	C		
930	940	950	960	970	980	990	**10m**															
C	C	C	C	C	C	C	C															

FIELD TWO **Field Centre** **300 measurements > 91 without vegetation > crop cover of 69%.**

0m	10	20	30	40	60	70	80	90	1m	110	120	130	140	150	160	170	180	190	2m	210	220	230
												C		C	C		C			C	C	C
240	250	260	270	280	290	3m	310	320	330	340	350	360	370	380	390	4m	410	420	430	440	450	460
	C	C					C			C		C	C					C	C	C		C
470	480	490	5m	510	520	530	540	550	560	570	580	590	6m	610	620	630	640	650	660	670	680	690
C	C			C	C			C	C	C	C		C	C	C	C	C	C	C	C	C	C
7m	710	720	730	740	750	760	770	780	790	8m	810	820	830	840	850	860	870	880	890	9m	910	920
C	C	C	C		C		C	C	C	C	C		C	C					C	C	C	C
930	340	950	960	970	980	990	10m															
C	C			C	C	C	C															

0m	10	20	30	40	60	70	80	90	1m	110	120	130	140	150	160	170	180	190	2m	210	220	230
	C	C	C	C			C	C			C	C	C	C		C	C	C		C	C	
240	250	260	270	280	290	3m	310	320	330	340	350	360	370	380	390	4m	410	420	430	440	450	460
	C	C	C		C	C	C					C	C			C	C	C				
470	480	490	5m	510	520	530	540	550	560	570	580	590	6m	610	620	630	640	650	660	670	680	690
C	C			C		C			C		C					C	C	C	C			
7m	710	720	730	740	750	760	770	780	790	8m	810	820	830	840	850	860	870	880	890	9m	910	920
						C													C	C		C
930	940	950	960	970	980	990	10m															
		C	C	C	C	C																

0m	10	20	30	40	60	70	80	90	1m	110	120	130	140	150	160	170	180	190	2m	210	220	230
				C		C	C	C	C				C	C	C			C	C	C	C	C
240	250	260	270	280	290	3m	310	320	330	340	350	360	370	380	390	4m	410	420	430	440	450	460
C		C	C	C		C	C			C	C				C			C		C	C	
470	480	490	5m	510	520	530	540	550	560	570	580	590	6m	610	620	630	640	650	660	670	680	690
		C		C	C			C						C	C		C	C	C	C		
7m	710	720	730	740	750	760	770	780	790	8m	810	820	830	840	850	860	870	880	890	9m	910	920
C	C		C	C		C			C	C	C	C	C		C	C				C		
930	940	950	960	970	980	990	10m															
	C	C			C	C	C															

FIELD TWO **Field Edge** **300 measurements > 155 without vegetation > crop cover of 48%.**

0m	10	20	30	40	60	70	80	90	**1m**	110	120	130	140	150	160	170	180	190	**2m**	210	220	230
	C	C	C	C			C	C	C			C			C				C	C	C	C
240	250	260	270	280	290	**3m**	310	320	330	340	350	360	370	380	390	**4m**	410	420	430	440	450	460
C	C	C	C			C		C	C		C	C		C	C		C	C	C			
470	480	490	**5m**	510	520	530	540	550	560	570	580	590	**6m**	610	620	630	640	650	660	670	680	690
C		C	C		C	C		C	C	C	C		C		C	C	C			C		
7m	710	720	730	740	750	760	770	780	790	**8m**	810	820	830	840	850	860	870	880	890	**9m**	910	920
C	C	C				C					C	C	C		C		C	C	C			
930	340	950	960	970	980	990	**10m**															
C	C	C				C																

0m	10	20	30	40	60	70	80	90	**1m**	110	120	130	140	150	160	170	180	190	**2m**	210	220	230
			C	C			C	C				C	C	C	C	C			C			C
240	250	260	270	280	290	**3m**	310	320	330	340	350	360	370	380	390	**4m**	410	420	430	440	450	460
C	C	C	C	C	C	C		C	C			C		C		C	C		C	C	C	C
470	480	490	**5m**	510	520	530	540	550	560	570	580	590	**6m**	610	620	630	640	650	660	670	680	690
C		C	C		C	C	C	C		C	C		C				C				C	C
7m	710	720	730	740	750	760	770	780	790	**8m**	810	820	830	840	850	860	870	880	890	**9m**	910	920
C	C	C				C					C	C	C		C			C	C			
930	940	950	960	970	980	990	**10m**															
C	C	C				C																

0m	10	20	30	40	60	70	80	90	**1m**	110	120	130	140	150	160	170	180	190	**2m**	210	220	230
	C		C		C	C	C	C	C					C	C			C				C
240	250	260	270	280	290	**3m**	310	320	330	340	350	360	370	380	390	**4m**	410	420	430	440	450	460
C			C					C				C					C		C			C
470	480	490	**5m**	510	520	530	540	550	560	570	580	590	**6m**	610	620	630	640	650	660	670	680	690
		C								C	C				C	C				C	C	C
7m	710	720	730	740	750	760	770	780	790	**8m**	810	820	830	840	850	860	870	880	890	**9m**	910	920
C			C			C	C	C	C			C	C	C		C		C	C	C		
930	940	950	960	970	980	990	**10m**															
C	C	C	C	C	C	C	C															

<u>FIELD THREE</u> **Field Centre** **300 measurements > 138 without vegetation > crop cover of 54%.**

0m	10	20	30	40	60	70	80	90	**1m**	110	120	130	140	150	160	170	180	190	**2m**	210	220	230
	C		C	C			C	C	C			C			C				C			C
240	250	260	270	280	290	**3m**	310	320	330	340	350	360	370	380	390	**4m**	410	420	430	440	450	460
	C		C			C		C	C		C	C		C			C		C			
470	480	490	**5m**	510	520	530	540	550	560	570	580	590	**6m**	610	620	630	640	650	660	670	680	690
C		C	C		C	C			C				C		C	C				C		
7m	710	720	730	740	750	760	770	780	790	**8m**	810	820	830	840	850	860	870	880	890	**9m**	910	920
C						C					C	C	C		C		C		C			
930	940	950	960	970	980	990	**10m**															
C	C					C																

0m	10	20	30	40	60	70	80	90	**1m**	110	120	130	140	150	160	170	180	190	**2m**	210	220	230
			C				C	C				C	C	C	C	C			C			C
240	250	260	270	280	290	**3m**	310	320	330	340	350	360	370	380	390	**4m**	410	420	430	440	450	460
C	C			C	C	C		C				C		C		C	C			C	C	C
470	480	490	**5m**	510	520	530	540	550	560	570	580	590	**6m**	610	620	630	640	650	660	670	680	690
C			C		C		C			C	C		C				C				C	C
7m	710	720	730	740	750	760	770	780	790	**8m**	810	820	830	840	850	860	870	880	890	**9m**	910	920
C	C					C					C	C	C		C			C	C			
930	940	950	960	970	980	990	**10m**															
C		C				C																

0m	10	20	30	40	60	70	80	90	**1m**	110	120	130	140	150	160	170	180	190	**2m**	210	220	230
	C		C		C	C	C	C	C					C	C			C				C
240	250	260	270	280	290	**3m**	310	320	330	340	350	360	370	380	390	**4m**	410	420	430	440	450	460
C			C									C					C		C			C
470	480	490	**5m**	510	520	530	540	550	560	570	580	590	**6m**	610	620	630	640	650	660	670	680	690
		C								C	C					C				C	C	C
7m	710	720	730	740	750	760	770	780	790	**8m**	810	820	830	840	850	860	870	880	890	**9m**	910	920
C			C				C	C	C			C	C			C		C		C		
930	940	950	960	970	980	990	**10m**															
C	C				C	C	C															

FIELD THREE — **Field Edge** — **300 measurements > 175 without vegetation > crop cover of 42%.**