Agricultural fine sediment: Sources, pathways and mitigation

By

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Abstract

This project utilises study sites in slowly permeable soils with artificial tile drainage typical of the lowland study region in the North East of England. The three selected sites investigate sediment loss processes at a range of scales, from the field, to catchment.

Sediment monitoring and in a 0.6 km^2 farm ditch catchment indicated that the majority of sediment losses occur during a small number of large runoff events. Source fingerprinting suggested that over 50% of the sediment is derived from arable tile drains, and approximately 10% from the farm track. A small outlet sediment trap was partially effective, with 37% sediment removal on average. It is estimated that a trap of 100-200 m^3 volume may be effective for a catchment of this size.

Detailed plot study of tile drain sediment losses under winter cereals showed conventionally tilled land generated mean sediment losses of 333 $kg.ha^{-1}$ over a single winter. Over 80% of total sediment loss was via tile drain rather than surface flow. The use of a shallow cultivation minimum tillage system significantly reduced mean sediment losses to 183 $kg.ha^{-1}$. Buffer strip treatments were not effective, particularly as they were unable to mitigate the majority of sediment loss occurring through the subsurface pathway.

At the catchment scale the importance of topographically controlled concentrated runoff pathways was recognised as a key sediment loss issue. Channel erosion sediment losses of up to 750 $kg.ha^{-1}$ were estimated for one field in the catchment during a single runoff event. The retention of *c.* 1 t of sediment was recorded during the same event in a field with an engineered flood retention barrier. A novel field method of event sediment loss estimation is presented. An investigation of a wider catchment area demonstrated that tile drainage exceedance during flood events is a major source of field erosion and sediment loss. While surface runoff events are infrequent in this landscape, they are of high magnitude and the relatively small capacity of the drainage network results in concentrated flows of high erosive potential.

The effectiveness of buffer strips is likely to be limited in these landscapes due to the role of concentrated runoff and subsurface sediment pathways. Grassed waterways and in-field soil conservation techniques are likely to be more effective against these two pathways respectively. Engineered structures also appear to be effective, and appear to be most justified where runoff control is also desirable.

Table of Contents

List of Figures

Chapter 2

Chapter 3

Chapter 4

List of Tables

Chapter 5

Chapter 6

Chapter 1: Background

1.1 Introduction and thesis structure

The negative impact of excessive fine sediment on the aquatic environment is an increasingly recognised catchment management issue. As the dominant land use in the UK, sediment loss processes from agricultural land have a large impact at the catchment scale. However, these processes remain poorly understood. Combined with the difficulty in defining thresholds for levels of fine sediment that can be tolerated within healthy catchments, this shortcoming means limited management guidance and action to control sediment loss from agricultural land, particularly in comparison with other industries (such as forestry and mineral extraction for example).

The off-site effects of fine sediment laden runoff from agricultural land have led to a re-evaluation of the need for soil conservation measures in the UK, as well as the need for alternative techniques specifically aimed at reduction of off-site impacts (e.g. water quality, sedimentation, damage to property and infrastructure). Sediment problems in the aquatic environment have long been equated with soil erosion problems, but the validity of this assumption is worthy of further examination. The need for alternative mitigation techniques to reduce off-site impacts is an acknowledgement that agricultural sediment may derive from sources other than field erosion, as well as that sustainable soil loss rates may have unsustainable negative effects on off-site receptors. The following review chapter seeks to examine the sources of agricultural sediment and to what extent the case may be made for soil conservation measures versus other mitigation techniques. The crux of this issue is the relative importance of different sources in their contribution to sediment loads and related off-site effects.

This PhD study investigates agricultural sediment loss processes at three scales; the plot/field, farm and catchment/landscape. The evidence base of the thesis is structured into two sections; field and farm scale (Chapters 2 and 3) and catchment scale processes (Chapters 4-6). It is argued that understanding of the interaction of multi-scale processes is crucial to understanding how risks may be identified and mitigated. The study of sediment loss processes is applied to lowland agricultural areas in North East England, and specifically to the slowly permeable soils formed in glacial till which dominate the lowlands of the region. The existence of artificial sub-surface drainage in these landscapes is a particularly important consideration in evaluating runoff and sediment loss processes and the most effective means of mitigation.

Chapter 2 presents the findings of a farm scale study conducted at Nafferton Farm, Northumberland, an experimental unit and commercial farm operated by Newcastle University. The study focusses on identification of critical runoff and sediment loss processes as well as the potential of drainage ditch sediment mitigation in a tile drained landscape. Chapter 3 presents a more detailed quantification of the role of tile drains conducted under field plot conditions at the University's second experimental farm at Cockle Park, Northumberland, and includes assessment of the efficacy of reduced tillage and buffer strip management to mitigate losses in a tile drained landscape.

The comparative importance of landscape scale topography in controlling risk is evaluated at the sub-catchment and catchment scale. Initially the study utilises the intensively monitored Belford catchment (north Northumberland) to look in detail at sediment and hydrological processes. Validation of these findings is performed using the surrounding wider catchment area (allowing replication of the observations made at the smaller scale). Chapter 4 presents an introduction to the Belford site, including the observed water quality issues in the catchment, and an examination of background data (both primary and secondary) to evaluate local agricultural sediment loss processes. Chapter 5 then presents evidence highlighting the spatially variable but potentially critical sediment loss risks associated with concentrated runoff pathways, and considers their interaction with artificial drainage in determining the level and the nature of risk. Chapter 6 presents the findings of a wider catchment study, and tests the utility of GIS terrain analysis as a remote sensing technology for the identification of local risks resulting from concentrated runoff.

Chapter 7 considers the findings in combination with the wider scientific evidence regarding the effectiveness of different sediment loss management techniques, variously applied as *in field*, *field edge* or *in-channel* sediment mitigation and their suitability as mitigation techniques for specific sediment source-pathways. The conclusions carefully consider how the agricultural, climatic and hydrological typology of the case study area affects the transferability of the findings of the present research to other areas (and vice-versa).

1.2 Soil erosion and conservation

Accelerated soil erosion associated with conversion of natural ecosystems to agricultural production has been recognised as a problem for centuries. Accelerated water erosion presents a sustainability issue to farmers through loss of top soil and nutrients and through blockage of drainage systems. The development of modern soil conservation strategies can be directly linked to the development of the US soil conservation service in response to the agricultural crisis of the 1930s. The basic tenets of soil conservation – minimisation of soil disturbance, maintenance of vegetative cover to protect the soil surface from erosive energy, and the limitation of uncontrolled surface runoff were developed into conservation systems in the US from the 1930's (Harlow, 1994, Helms, 2010). Since that time, systems have been refined, particularly through the development of reduced tillage planting machinery and broad spectrum herbicides which minimize the need for soil disturbance. In livestock systems, management of stocking densities and stock exclusion are aimed at a similar maintenance of vegetative cover (Blanco-Canqui and Lal, 2008). These techniques have been demonstrated to effectively reduce rates of soil loss while maintaining productivity in modern agricultural systems in North America and beyond.

It is important to note that the damaging off-site effects of water erosion have long been recognised as a target of soil conservation programmes, and soil conservation strategies have been increasingly linked with water quality objectives (Walter *et al*., 1979, Blanco-Canqui and Lal, 2008). The logic that soil conservation measures offer the solution to excessive sediment in aquatic environments as well as to soil resource sustainability is questionable however. If the on-site impacts of soil loss are of secondary or minimal importance, the means of promotion of changes in agricultural practice may differ, and an alternative system of cost-benefit analysis is required to place a value on the reduction in off-site impacts (Kuhlman *et al*., 2010). Soil erosion at rates with minimal resource implications for the farmer may still have a significant negative impact on water quality or infrastructure. However, the ability of soil erosion mitigation measures to achieve these more stringent objectives is much less well documented than their ability to prevent high erosion rates.

1.3 Soil erosion in the UK

1.3.1 Lowland erosion

In lowland areas, cases of accelerated water erosion have largely been associated with arable farming. The conversion of large areas of pasture land to cereal production and increases in field sizes from the 1950s onwards (particularly in the south and east) is often cited as the cause of increasing sediment export from agricultural land (Boardman 2002). In a recent summation of UK soil erosion research and policy Evans (2010) describes how incidence of rill erosion on arable land were recognised by the 1960's and systematic monitoring schemes in the 1980s revealed the negative effects of increasing areas of winter cereals. It has been suggested that in other countries increase in winter cereals is believed to *reduce* soil erosion (in comparison with spring crops). Under the UK climate however, winter sown cereals are widely regarded as presenting higher erosion risks than spring (Boardman, 2001). In 1985 a widely quoted assessment concluded that "sustained use of this area [lowland arable] for cereal, sugar beet and vegetable production beyond the first quarter of the next century" was threatened (Morgan, 1985). A similar estimate that 36% of the arable area of England and Wales was at moderate to very high risk of erosion was made by Evans (1990).

Studies of soil erosion in the UK have used a range of direct and indirect methods to assess erosion rates - for example direct measurements from plot experiments, indirect sediment budgeting and alluvial accumulation. This evidence is collated and reviewed by Brazier (2004), who concludes that erosion rates from arable land appear to exceed "acceptable" rates based on estimates of rates of soil formation. However, the general range of these estimates also suggests that the extremely high water erosion rates recognised in the United States and elsewhere are rare. The Soil Survey National Monitoring Scheme from 1983-1986 provides a detailed database of erosion on arable land in England. The findings suggests average rates of erosion are relatively low (3.6-0.4 $m^3.ha^{-1}.yr^{-1}$), although noting that inter-annual variability means that losses in unusual weather conditions can be much higher (Boardman, 2002). Skinner and Chambers (1996) report the results of 1980's ADAS monitoring of high risk arable sites (where erosion had been observed previously). Results show estimated yield losses of over 10% occurred on less than 5% of sites, but clean up costs were associated with 15%.

The evidence suggests that in the UK soil erosion is unlikely to result in significant reduction in fertility (Boardman, 2002). Exceptions are shallow soils (<30 cm) where further decreases in depth are predicted to result in increased droughtiness over the next 50 years (Webb *et al*., 2001). Bakker *et al*. (2007) do not expect crop yields in Northern Europe to be affected by soil erosion this century. The reasons for relatively low erosion rates in the UK are well understood; the low intensity of rainfall and the possibility of year round vegetation cover under a humid maritime climate are the main factors limiting susceptibility to extreme erosion. This is not to say that cases of extreme erosion do not occur. What is indisputable however is that these cases have been insufficiently severe or widespread to lead to the development of national or regional soil conservation strategies. It is also notable that soil erosion research in the UK as in many parts of continental Europe has also received limited investment, and relies heavily on developments in the United States and elsewhere. The EUROSEM project (Morgan *et al*., 1998) was a relatively late attempt to address this shortfall (by providing a soil erosion model calibrated specifically for European conditions).

1.3.2 Upland erosion

While the arguments over sustainability of the soil resource have focussed more on lowland cultivated systems in the UK than on grassland systems, concerns have long been raised over the extent of upland erosion. Upland grassland erosion causing high river sediment loads has been recognised as a problem since Evans' seminal paper (Evans, 1971). Evans (2007) suggests grazing-initiated erosion occurs on upland grassland covering 2.7% of England and Wales and 16.4 % of land area in Scotland, particularly citing the development of scars on slopes by sheep. In peatlands, he argues while erosion is mainly initiated by moorland fires, ditching, air pollution and mass movement, it is maintained and exacerbated in an eroding state by grazing animals. Grieve *et al.* (1995) attribute most erosion to grazing, burning and vehicle tracks in upland Scotland. Holden *et al*. (2007) suggest that gullies and peat landslides are the predominant cause of peatland erosion, and that erosion may be exacerbated by overgrazing and gulleying of artificial drainage ditches. Mchugh *et al*. (2002) conducted a study looking at over 200 upland sites in England and Wales, and identified burning, animal and vehicle tracks, sheep scars, poached areas and drains as the main causes on non-forested sites. A Revisit to the same sites (Mchugh, 2007) found the main change to be re-vegetation or de-vegetation related to sheep grazing intensity, and the authors suggest changes from headage payments to farm area payments under CAP reform may therefore lead to a widespread reduction in upland erosion (although no published supporting evidence is available at the time of writing).

1.4 UK Soil Conservation Policy

The UK, unlike other north European countries such as Belgium and the Netherlands (Fullen, 2003) does not have a dedicated soil conservation programme, reflecting the view that the costs of such a programme were not justified by the threat erosion represents to resource sustainability; Boardman (2002) re-visited Evan's (1971) "case for soil conservation in Britain". He argued that extreme cases of erosion are isolated and rare, and concluded that widespread adoption of soil conservation was not justified in order to maintain agricultural productivity, unless a very long term view was adopted.

The effects of current rates of soil erosion on *water quality* may represent a more urgent sustainability issue however. Boardman (2002) acknowledged that increased concern over the external costs of soil erosion had finally focussed attention on the problem; the need to control the off-site effects, particularly the effect on water quality means there is a pressing need for policy intervention, and to develop a better understanding of the sediment pollution process and its effects on the aquatic environment. However, he stops short of prescribing a soil conservation programme as the solution. Evans (2010) recently provided a summary of soil erosion research in the UK, highlighting the late intervention compared with other countries, and particularly that although long term threats to soil fertility were recognised by the 1980's, little action was taken to address water erosion problems. Evans reports that more recently, increased awareness and concerns over soil erosion has led directly to the production of soil management advice (e.g. Defra , 2004, 2005; Scottish Government, 2005). He makes clear that the two main factors behind this new impetus were indirectly related; CAP reform and the recognition of the off-site impacts – particularly muddy flooding and water pollution.

The need for a better understanding of the sediment pollution problem that Boardman (2002) suggests highlights a key difference between control of *soil erosion* problems and *eroded sediment* problems; in many circumstances where the source of diffuse sediment pollution is poorly defined, the source and the receptor may be separated both spatially and temporally (Boardman *et al*., 2003b) and sediment *quality* may also be more important than absolute quantity. It is much more difficult in these circumstances to undertake the same kind of field risk assessment used to target soil erosion problems, since land areas with significant negative off-site impacts may be distinct from those at high risk of soil erosion using conventional assessment methods.

1.5 Sediment loss to water

The Water Framework Directive (2000/60/EC) for the first time introduces a requirement to meet "Good Ecological Status" for suspended sediment in inland waters. Based on sensitivity of key indicator species such as *salmonid* fish, English Nature set this level at 25 $mg.l^{-1}$ (the same as that of the EC Freshwater Fish Directive [78/659/EC]), and this is now generally adopted as the standard in England and Wales (although achievement of this target is not a *legal* requirement to meet good ecological status). Collins and Anthony (2008a) made an attempt to model the likelihood of exceedance of the 25 $mg.l^{-1}$ target in rivers in England and Wales. They estimated that sediment losses from diffuse agricultural sources need to be reduced by 20% on average in non-compliant catchments, and by as much as 80% in some areas. The validity of an average annual concentration as a threshold target remains questionable however; the association of high sediment losses with short term events means that potentially critical impacts are neglected, for example the effect on gravel bed siltation. Bilotta and Brazier (2008) questioned the use of threshold annual export targets, arguing that short duration events may be responsible for a disproportionate proportion of this export, and that the ecological consequences of elevated suspended solid concentrations for short durations (usually during large runoff events) may be limited. They propose an alternative classification based on the *duration* of exceedance of the 25 $mg.l^{-1}$ threshold target.

While the use of the Freshwater Fish Directive is to continue to be used by UKTAG as a guide to water quality (although not a target) until 2013, an alternative classification scheme was proposed (Cooper *et al*., 2008 - Table 1.1) which utilises upper and lower quartile specific catchment sediment yields as critical thresholds and targets respectively, based on catchment typology as proposed by Walling *et al*. (2008). A second sediment loss prediction study was undertaken by Collins and Anthony (2008b) to assess the likelihood of catchments in England and Wales meeting the quality targets proposed by Cooper *et al*. (2008). This suggested non-compliant catchments are largely limited to uplands in Wales and northwest England, and the chalklands of southern and eastern England. The reasons for the predicted threshold exceedance would appear a function of the relative steepness of topography and high rainfall in the former case, and in the latter the combination of widespread cultivated land uses and low target threshold for the chalkland streams (5-20 $t.km^{-2}.yr^{-1}$ in Table 1.1).

Even the application of this alternative scheme appears problematic, the negative impact of elevated sediment on the natural environment is defined by local conditions – in some circumstances high fine sediment yields will have little impact on target environmental conditions, particularly where target ecological populations are absent. Conversely, there may be locally highly sensitive floral and faunal populations, where low sediment inputs may have an adverse impact and further mitigation is both justified and viable. Whether excess aquatic sediment has a significant negative impact is dependent on the sensitivity of local ecology – the case of *salmonid* fish species presents a good example – numerous catchment sediment problems have now been diagnosed due to the connection between spawning bed siltation and species decline or local extinction, rather than determined on the basis of direct measurement of sediment concentrations or loads. In many cases siltation problems have been recognised only once other limits to *salmonid* migration (e.g. effluent pollution, blocked passage) have been removed. Targets based on suspended sediment concentrations prioritise the effects of suspended material in the water column, and ignore the affect on the benthic environment of streams and receiving lakes and coastal waters.

Catchment type (permeability/rainfall/soils)	Target suspended sediment yield	Critical suspended sediment yield (t km^{-2} yr^{-1}) (upper quartile)
High wet and low peat	50	>150
Low wet other	40	>70
Low dry other	20	>50
High wet and high dry other	10	>20
Low dry and low wet Chalk	2	>5

Table 1.1: Catchment typology and sediment thresholds after Cooper *et al*., 2008.

As well being a *physical* pollutant, sediment derived from agricultural land also constitutes a major *chemical* pollutant; chemicals associated with the 'active' fraction (usually taken as silt and clay sized primary particles less than 63µm) are transported adsorbed on the surfaces of colloidal particles. These contaminants are either associated with the mineral (as in the case of inorganic phosphorus), or with the organic fraction (as with chlorinated compounds which form the basis of many pesticides). Hence control of sediment inputs to freshwaters is a means of significantly reducing total inputs of associated pollutants. It has been estimated that up to 90% of P flux in surface rivers in North America and Europe may be attributable to that associated with suspended sediment (FAO, 1996), while strong associations between pesticide contamination of water bodies and the particulate phase are widely reported (Holland, 2004). The implication therefore, is that sediment from intensive arable land may have particularly detrimental impacts on aquatic ecology, independent of the total contribution to sediment export. The same may be true with regard to the impacts of sediment-associated faecal pathogens and phosphorus derived from grazing areas or fields treated with manures (Ramos *et al*., 2006; Agouiridis *et al*., 2004).

1.6 Causes of sediment pollution in agricultural catchments

Assessment of catchment-scale dynamics clearly involves consideration of sediment sources which are non-agricultural. However, evidence suggests the majority of sediment is derived from agriculture in all but the most urbanised catchments, including all of the priority catchments designated under the Catchment Sensitive Farming Initiative in England for example. Collins *et al*. (2009a) estimate that 76% of sediment discharge to rivers in England and Wales comes from agricultural land, 15% from channel banks, 6% from urban diffuse sources, and 3% from point sources (although it is notable that the focus of this assessment is on lowland catchments and does not include assessment of moorland and forestry activities). While there are likely to be catchments where a sediment pollution problem *is* attributable to non-agricultural activities (for example forestry, construction or extractive industries), these are generally industries for which strong guidelines and regulations exist in comparison with the agricultural sector.

1.6.1 Bank erosion

Stream bank erosion is a natural process which contributes a significant proportion of the sediment load of a river system in undisturbed natural catchments. The relative contribution is affected by the climate, substrate and flow regime, but under natural catchments is in dynamic equilibrium over long time scales. Rates of bank erosion can be accelerated in agricultural catchments, both through channel modifications and land use – particularly the destabilising effects of livestock bank grazing. Catchment sediment contributions from bank erosion appear to be variable, reflecting both the total sediment derived from other sources, and the variability in bank destabilisation between catchments. In catchments dominated by extensive grassland farming with livestock access to watercourses, bank erosion can locally be a very significant or majority source (e.g. Collins *et al*. 2010). A study of bank erosion in the upper Severn upland catchment (Bull 1997) estimated 17% of the annual fine sediment load to be from the banks, but that in individual storm events this could increase as high as 67%. In UK arable-dominated catchments (and those without livestock bank access) however, bank erosion has been found to be a minor source compared with top soil erosion (Russell *et al*., 2001; Walling *et al*., 2006; Collins and Walling 2007). In a summary of UK sediment tracing studies Walling (2005) reports that although contributions have been found to be as high as 48%, the mean value is 16.4% (Q_{25} = 7.5%; Q_{75} = 23%).

1.6.2 Runoff from arable land

As discussed previously, arable agriculture in the UK lowlands has been the focus of concerns over accelerated soil loss rates, primarily from an agricultural productivity perspective. The accumulated evidence demonstrates that high rates of soil/sediment loss occur from arable land through water induced erosion relative to permanently vegetated surfaces. Research into the effects of arable agriculture on catchment sediment budgets has been more limited, but the development of sediment source fingerprinting techniques has suggested that soil loss from arable land often represents a large proportion of catchment fine sediment yield (Owens *et al*., 2002; Russell *et al*., 2001; Walling *et al*., 2006).

Evans (1990) evaluated soil erosion risk of agricultural land in England and Wales and concluded that 36% of the arable land area was at moderate to high risk of erosion. While many studies have focussed on channel (rill and gulley) erosion, Evans (2010) suggests that the effects of sheet wash on erosion rates have been neglected as it was viewed as a relatively minor process. However he states that "the effects on water quality are severe". It has been noted in particular that in chalk catchments in south east England arable land use has had a detrimental effect on stream ecosystems through increased inputs of sediment and associated pollutants (Environment Agency, 2004; Heywood and Walling, 2003). Walling and Amos (1999) among others demonstrated that eroded soil, mainly derived in winter runoff events when arable soils were bare, was responsible for summer problems of siltation and turbidity, as sediment "plugs" were progressively remobilised and deposited.

1.6.3 Runoff from grassland

Lowland grassland systems have been recognised as a significant source of sediment, in some UK lowland catchments representing a greater source than arable farming, and not exclusively those where grassland systems are the dominant land use (Russell *et al*., 2001; Bilotta *et al*., 2008). Poaching of heavily used land areas (e.g. field corners and around feeders) can result in the generation of suspended material, which may reach watercourses particularly where situated in riparian areas, or where connected to an artificial drainage network. Bilotta *et al*. (2010) recently demonstrated that for a small headwater improved grassland catchment suspended sediment yields could reach 1.2 tonnes per hectare per annum, and argue that improved grassland contributions to sediment losses have been wrongly ignored due to an absence of monitoring of soil erosion rates which have concentrated on lowland arable systems.

1.6.4 Land drains

The interaction of climate and soils in the UK has led to a relatively high proportion of land (mainly in the lowlands) being subject to artificial sub-surface drainage improvement. These drains act as an alternative route for sediment and associated pollutants from agricultural land to water courses, and an increasing volume of study has considered the significance of the drainage pathway. Deasy *et al*. (2009a, 2009b) suggest that sub-surface sediment pathways may dominate in under-drained land, and that mitigation options should therefore focus on these rather than surface losses.

Chapman *et al*. (2003) undertook a regional estimate of the risk of tile drain losses in England and Wales, reporting the main factors as the presence of "cracking" soils and proportion of soil subject to artificial drainage. Much of England excluding Cornwall and East Anglia was found to be at moderate to high risk, mainly due to soil type, and particularly the south east where efficient modern drainage systems exist. Chapman *et al*. (2005) suggest that land drains represent a major source of sediment transport particularly when soil moisture deficits are high and cracks allow rapid transport of sediment, with the majority of sediment being derived from the surface. In one study site they report that land drains represented over half of sediment delivery from a field. Grant *et al*. (1996) found that particulate loss through tile drains is particularly associated with major rainfall events, and conclude that annual losses of sediment and phosphorus are significantly underestimated where sampling does not take account of these events. Bilotta *et al*. (2008) tested the effects of grassland drainage on replicate drained and undrained plots. They found losses from drained land were consistently lower than the undrained due to a reduced susceptibility to sediment mobilisation. However, the study demonstrated a significant quantity of sediment was lost from grassland via tile drains despite no visual evidence of the process.

The impact of land drains on catchment sediment budget is complicated by two factors; firstly reduction in surface runoff may result in a decrease in suspended sediment yield – the drain pathway would appear likely to transport less coarse fraction material in suspension. Kladivko *et al*. (2001) reviewing the evidence for effects of drainage on pesticide losses conclude that tile drainage generally decreases surface runoff and sediment loss compared with undrained sites due to lower concentrations in drain flow compared with overland flow. Conversely, the degree of connectivity between a land drain and the receiving water course (a drainage ditch or natural stream or lake) in most cases will be much higher than that of a surface pathway. Much of the sediment mobilised by surface runoff is re-deposited elsewhere in the landscape with no realistic possibility of polluting water bodies. The installation of an artificial subsurface drainage network means that mobilised material is highly likely to reach a water course.

These issues are expressly recognised in the recently developed PSYCHIC model (Davison *et al*., 2008). The connectivity algorithm for land drains is constant at 0.9 (90% of sediment entering the drain is exported to the river network) compared with a variable connectivity for surface runoff depending on proximity to surface waters. A second constant of 0.2 is applied to drain flow to represent the diminution of suspended sediment loads relative to surface pathways, giving a sediment delivery ratio of 0.18. As drainage systems are simulated to carry the majority of runoff, they are estimated as the dominant field source where they exist. Despite the assumption that sediment concentrations per unit runoff are only one fifth that of the surface pathway, much lower estimates of connectivity for the former mean that tile drainage generally appears to increase sediment delivery ratio relative to undrained catchments. PSYCHIC modelling has suggested that across England the higher connectivity engendered by artificial land drainage is often one of the key factors controlling high sediment loss risk (Collins *et al*., 2007, Stromqvist *et al.*, 2008).

1.6.5 Pollution 'hotspots'

Important sediment loss processes involve non-field areas which are not considered in conventional soil erosion risk assessment. Tracks and hard standings provide major runoff sources which interact with field areas to increase runoff energy (run-on) and convey sediment (run-off) to water courses. However, they may also represent a sediment source in their own right. Collins *et al*. (2010) suggest that damage to roadside verges and tracks, particularly by farm machinery and livestock can represent a significant source which should be taken into account in assessing sediment source pressures. Gruszowski *et al*. (2003) estimated for a rural catchment in Herefordshire that 30% of fine sediment was either derived from or transported via roads. Unpaved forestry tracks have also been shown to generate significant quantities of sediment (e.g. Sheridan and Noske, 2008) due to their unsealed surface, usage by heavy machinery and conveyance of large quantities of runoff. Agricultural tracks exhibit similar characteristics. Boardman *et al*. (2009) consider that roads should be considered as receptors for eroded materials from fields and their proximity considered in off-site risk assessment. This reflects both the problems caused by sediment deposition on public highways, and the efficient conveyance of runoff along roads and associated drains , which means that sediment laden runoff reaching a road is also highly likely to reach a watercourse.

Losses from man-made surfaces are particularly significant in terms of the level of contamination of sediment, the impacts of which may far outweigh their contribution to catchment sediment yields. Combined with the role of farm and road networks in conveyance of runoff and associated sediment, this means that mitigation of runoff to and from these areas may be of critical importance to water quality objectives.

1.7 Costs of sediment pollution

The costs of land degradation caused by soil erosion have received much more consideration than off-site effects of erosion-derived sediment. Part of the problem is the difficulty of quantifying many of the costs associated with environmental degradation caused by sediment loss. Sediment transfer to and by water bodies being a natural process, it is also difficult to ascribe environmental dis-benefits, as benchmark sediment levels are difficult to define. However, several attempts have been made to put cost estimates on sediment pollution:

Pretty *et a*l. (2000) estimated from 1992-1997 £34.4 m per year was spent on removing soil from drinking water (£52.3 m if phosphorus removal is included). £13.77 m was associated with direct damage to roads, property and river channels by muddy runoff. The study specified however, that these costs do not include one-off damage costs incurred by water companies, damage to fisheries, or the effects of eutrophication on the environment.

Assessing the cost/benefits of agriculture the Environment Agency (2002) estimated the on-site costs of soil erosion were only £8 million per annum. The direct off-site effect (clean-up operations) was slightly higher at £8.2m. In addition, the agriculture contribution to the costs of flooding was estimated as £114m per annum. This includes on-site impacts on runoff generation such as soil compaction, but also the effects of sedimentation on water courses increasing the risk of flooding (by decreasing bank-full discharge).

In 2007 the Environment Agency (2007) produced a detailed assessment of the external costs and benefits of agriculture, including estimated *non-direct* costs (ecosystem degradation). Although the effects of sediment pollution on ecosystems and water treatment are difficult to separate from costs associated with (dissolved) nutrients and pesticides, the total cost was estimated to be between £445m and £872m per annum. The effects of agricultural pollution on recreation were estimated as £10-23m and on angling as £71m. These figures illustrate that even the costs which are directly measurable through remediate measures and/or loss of revenue appear to greatly exceed the on-site costs. A similar EU assessment of the costs of soil degradation (Görlach *et al*. 2004) suggested that quantifiable off-site costs can exceed on site costs of degradation by a factor of 10, but as neither landowners or insurers bear this cost there are few incentives for mitigation. The study recommended improved targeting of soil protection measures to reduce off site impacts, and the use of the Water Framework Directive to promote good practice where agriculture has a major negative impact on water bodies.

1.8 Assessing risk of sediment loss to water

Numerous schemes exist from around the world for the assessment of agricultural soil erosion risk. Assessment of *sedimen*t loss risk is a similar principle, with an added level of complexity relating to the likelihood of soil mobilised by erosive processes reaching a water course. This is usually referred to as landscape *connectivity*. Interest in connectivity has increased in recent years in correlation with increased concerns over the effects of runoff on surface water quality and quantity (Boardman *et al*, 2009). Methods of assessment vary in complexity and scale of application. The PSYCHIC modelling project attempts to quantify risk based on assessment of Connectivity Index. At a catchment scale the calculation of connectivity is limited to distance from watercourse (Davison, 2008). For field scale assessments an approach has been proposed based on assessment of distance from watercourse, permeability of field exits, presence and condition of artificial drainage, slope characteristics, rainfall and flood risk

(Humphrey *et al.*, 2004a, 2004b), although evidence for the efficacy of this procedure is lacking.

An attempt to adapt existing assessment methods regarding farm-scale soil erosion risk into a risk assessment for off-site effects was made by Boardman *et al*. (2009) adapting Defra's soil erosion risk assessment (Defra, 2005). They suggest that consideration of inter-connectivity of high risk fields is missing from the current Defra methodology but acknowledge this may be difficult without detailed survey, particularly due to lack of information on permeability of field boundaries. They propose a compromise of inclusion a note of whether high risk fields have a road downslope, or a river within 200 m downslope.

1.9 Summary and conclusions

Despite evidence indicating UK erosion rates to be largely sustainable from an agronomic perspective, increased awareness and concern over the off-site effects of water-eroded sediment have called for a new focus on agricultural soil loss issues. However, for several reasons sediment loss processes should be distinguished from soil erosion processes: Sustainable sediment inputs to water bodies are generally much lower than sustainable soil loss thresholds, therefore assumptions that agricultural land use does not present a sediment loss issue may well be inaccurate based on soil erosion risk assessment. Important sediment sources such as track runoff and channel bank erosion are not caused by field erosion, yet may be at least a partial reason for catchment sediment problems. Large parts of a catchment may be effectively disconnected from the surface drainage network, and soil erosion susceptibility is distinct from sediment loss susceptibility in these circumstances. Sediment loss susceptibility requires consideration of greater variety of causal factors, and a different approach to assessment.

Tolerable thresholds for suspended sediment concentrations have some validity, but focus on this issue may neglect a large number of cases where the problem caused by excessive sediment is related to the effects on siltation, which may be caused by relatively few events of large magnitude. Alternative means of quantifying sustainable specific sediment export rates have been suggested, but it is argued here that diagnosis of excessive sediment inputs should be determined locally according to their specific effects on aquatic ecology and or infrastructure. In some circumstances the nature of sediments (particle size or nutrient content) may be more important than absolute quantity, and efforts may therefore be targeted at identifying and mitigating specific sources accordingly.

There remains a significant knowledge gap relating to the role of different sources and pathways controlling the impact of agricultural areas on aquatic receptors. Evidence has indicated the important role of tile drainage in some catchments, although it is unclear whether these findings are applicable at a larger catchment scale. A major shortcoming appears to be the failure to link catchment and plot scale processes, a crucial issue if practical risk assessment procedures are to be successful in mitigating agricultural sediment losses.

1.10 Aims and Objectives

Agricultural sediment loss is an issue which is of increased environmental concern, particularly due to the requirements of the Water Framework Directive. However, agricultural sediment loss processes at a catchment scale are not well understood, particularly in a UK policy context. The evidence presented has highlighted that simply up-scaling soil erosion processes to the catchment scale in order to estimate sediment losses is unsatisfactory. There is a shortage of case study evidence which integrates across scales, and a particular gap appears to exist regarding the role of artificial drainage in sediment loss regimes.

Although a range of mitigation options are available, their efficacy is in large part dependent on the source-pathway of sediment from agricultural land to water bodies. The distinction between sediment and soil erosion processes described in this chapter makes it imperative that mitigation addresses off-site water quality objectives rather than on-site resource protection, and the degree to which existing soil erosion mitigation measures can deliver these different objectives requires further evaluation.

Hence this work takes as its case study the slowly permeable catchments common in the lowlands of north east of England, as well as north west and midland England, Wales and Scotland. The presence of tile drainage has been suggested as a particularly important factor in these landscapes, although case study evidence from this region is lacking, and in general the evidence base is weak. This study will contribute to the understanding of the sediment loss issues associated with lowland agriculture in the UK. It also investigates the potential of different sediment mitigation measures, particularly in the context of this case study catchment typology.

Aims

1. To investigate agricultural fine sediment loss processes at field, farm and catchment scale in a tile drained landscape.

2. To investigate the suitability of sediment loss mitigation strategies in the study landscape.

Objectives

1.1 Determine sediment sources at a farm scale using a chemical fingerprint approach .

1.2 Investigate the temporal pattern of sediment losses at a farm scale using flow and sediment monitoring.

1.3 Quantify tile drainage and surface sediment losses at a plot scale under winter cereals.

1.4 Investigate waterborne sediment dynamics at the small catchment scale using flow and sediment monitoring.

2.1 Evaluate the potential of GIS remote sensing to identify areas at risk of concentrated runoff erosion.

2.2 Evaluate the performance of selected mitigation options.

2.3 Review the potential of source-specific mitigation options for slowly permeable tile drained landscapes.

SECTION 1: FIELD AND FARM SCALE PROCESSES

Chapter 2 Farm scale study – Nafferton Farm

2.1 Introduction

Soil loss from arable land often represents a large proportion of catchment fine sediment yield (Owens *et al*., 2002; Russell *et al*., 2001; Walling *et al*., 2006). Additionally lowland grassland systems have been recognised as a significant source of sediment, and not exclusively those where grassland systems are the dominant land use (Russell *et al*., 2001; Bilotta *et al*., 2008). The farm is a logical unit for mitigation actions for sediment pollution arising from agriculture, since all water-borne sediment problems may be regarded as the responsibility of the land owner prior to exiting the farm boundary, and decisions regarding mitigation may be made appropriately to target identified source pathways. Consequently understanding of the relative importance of various source-pathways is integral to designing successful mitigation schemes. While this relative importance may be expected to vary significantly between individual farms and agglomerations of farms and other land uses into landscapes, an in-depth study of a single farm is useful in identifying in particular the issue of tile drainage losses which other studies have identified as the largest sediment source in lowland landscapes with slowly draining soils (e.g. Deasy *et al*. 2009a, 2009b; Collins *et al*., 2007; Stromqvist *et al.*, 2008).

The following chapter describes a study of sediment loss processes and farm –scale mitigation features conducted at Nafferton Farm as part of the Proactive Experiment investigating soft engineered approaches to mitigating problems of agricultural water quality and quantity. This site provides a case study of sediment loss processes at a farm scale which are investigated in further detail in later chapters. The objectives of this study were to evaluate suspended sediment loss patterns from a typical farm catchment using a detailed monitoring/sampling regime, to quantify suspended sediment sources for the farm catchment using a chemical source fingerprinting approach, and to evaluate the effectiveness of an in-ditch sediment trap in mitigating suspended sediment losses.

2.2 Site Description

Nafferton Farm is operated by Newcastle University School of Agriculture, Food and Rural Development**.** The farm is situated in the Tyne Valley, approximately 15 miles west of Newcastle upon Tyne (NZ 064 656) The total farm area is 294 ha. The farm is a mixed livestock/arable enterprise, with dairy/beef pasture, silage production and arable rotation operated on a commercial basis. The rotation consists of winter and spring cereals with ryegrass leys. In the area around the farm steading, the rotation consists of dairy pasture with occasional cereal crops. Soils are highly uniform, consisting mainly of slowly permeable fine textured soils (Figure 2.2) formed in Devensian glacial till of the Brickfield Series (Cambic

Stagnogley Soils, Avery, 1980). The land is artificially drained by sub-surface tile drains of varying age and efficiency

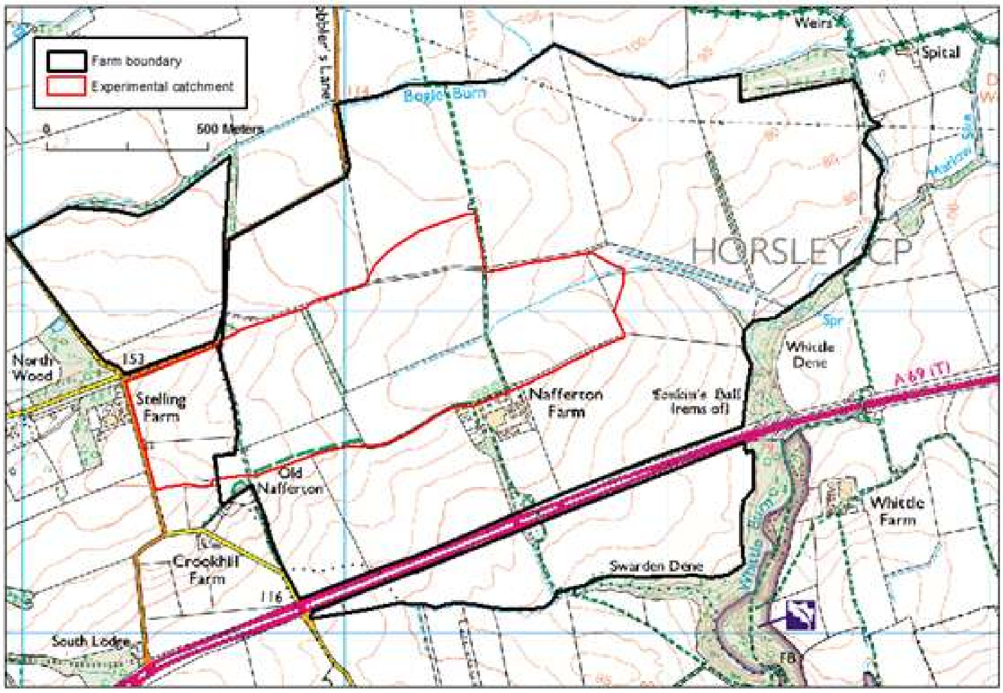

Figure 2.1: Farm and experimental sub-catchment boundary map.

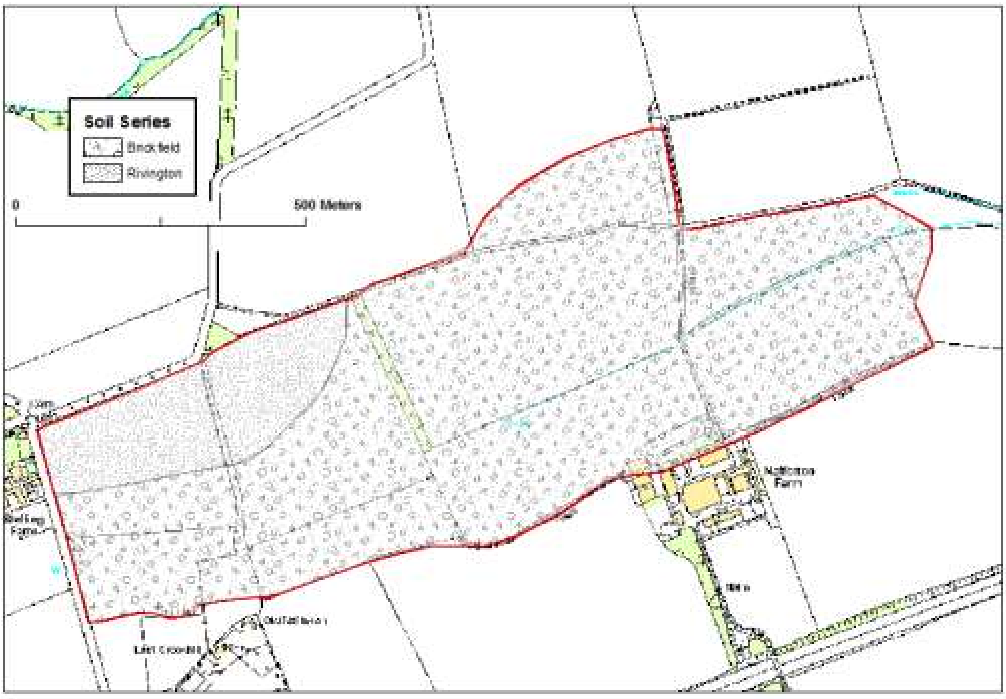

Figure 2.2: Experimental sub-catchment soil map.

Original survey M. Palmer 2008

The Proactive experimental catchment was established in 2006, consisting of an intensively monitored drainage ditch, which contained a linear series of trial water pollution mitigation features. The experimental ditch catchment has an upstream area of approximately 62 ha (0.62 km^2), and occupies the central portion of the farm (Figure 2.1). The catchment consists of permanent dairy pasture and pasture/arable rotation in the area close to the farm steading, and an area of continuous arable cropping in the west of the catchment (Figure 2.3). The catchment also includes the main farm track which is the route taken by the dairy herd to and from the milking parlour. The ditch passes under this track via a cross drain, and receives track drainage during wet periods. The ditch is fed by a number of active tile drains, predominantly from the areas supporting arable cropping. The ditch banks are stock-fenced along the entire length with a 5 m vegetated buffer strip.

As part of the experiment a small scale sediment trap was constructed as the initial mitigation feature (located at the outlet of the catchment shown in Figure 2.3) in November 2008. The trap consisted of two cells; the upper formed by a dam constructed from 1 m panels of recycled farm plastic; the lower by a 1 m dam of tropical hardwood, with a distance between barriers of 10 m (see Photos 14 & 15). Results for mitigation features are reported elsewhere, although testing of sediment trap efficiency is presented in this study.

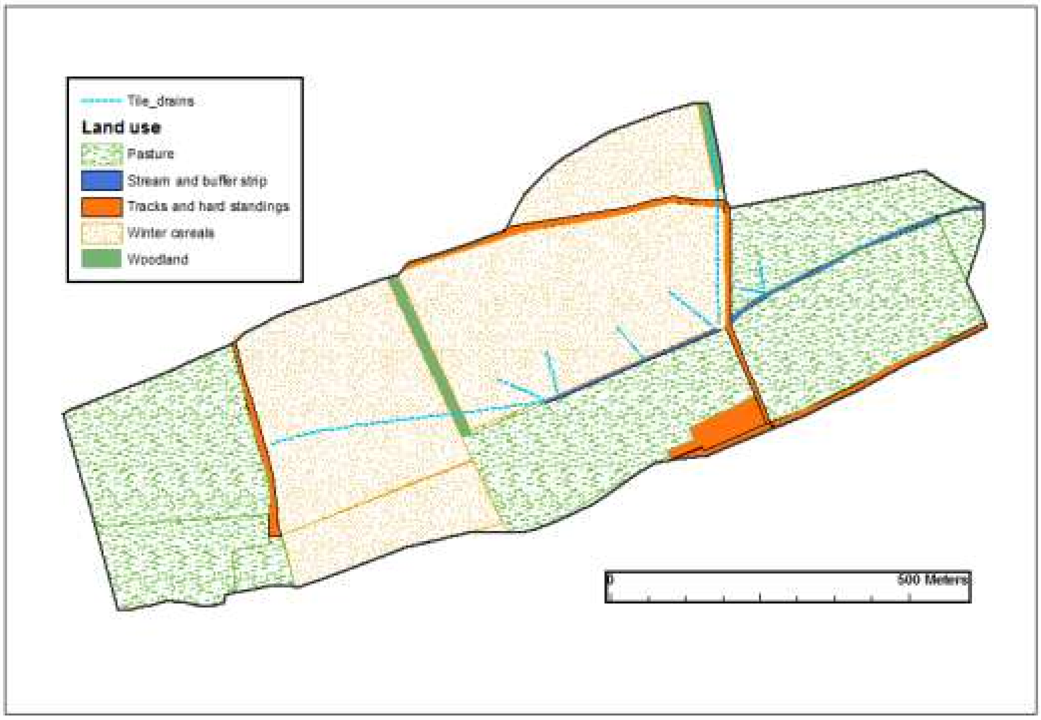

Figure 2.3: experimental sub-catchment land use map 2009.

2.3 Study methods

2.3.1 Water quality and flow monitoring

Flow in the ditch was continuously monitored at the catchment outlet (5 min intervals) between March 2007 and September 2009, at the start and end of the experimental features via a pressure transducer sited on a concrete weir installed as part of the Proactive experiment (not directly undertaken as part of the present study). Flow volume was estimated from stage using a broad-crested weir equation:

$$Q = C.b.H^{\frac{3}{2}}$$

Where Q is discharge, C is the coefficient of discharge 1.534, b is the weir cross sectional width (m) and H is stage (m).

Rainfall was monitored hourly via an automated weather station located on the site.

200 ml ditch water samples were collected between January and July 2009 during runoff events at the catchment outlet, both above and below the sediment trap, to assess flow regimes and trap efficiency using automated samplers (ISCO 6700). Sampling regime was targeted towards the measurement of hydrological 'events' in order to characterise the sediment discharge regime. Samplers were variably programmed to obtain samples at equal time intervals of between 0.5 and 6 hours. Later sampling (May to July) was undertaken hourly following activation of a float trigger to provide samples from large events only. Automated samples were augmented by manual samples where appropriate. A total of 224 samples were collected over the monitoring period. Suspended sediment concentration was measured by filtration of a 100 ml sub-sample through a .45 µm membrane filter paper.

2.3.2 Sediment source fingerprinting

Sediment samples were collected from the ditch between January and December 2009 using a simple sediment trap sampler designed according to the specifications of Phillips *et al*. (2000). Similar samplers were located at the outlet four tile drains within the catchment. One further active drain outlet was located during survey, but the outlet was not accessible for sampling. Three of these tiles drained land under winter wheat during the sampling period. The fourth was under pasture and passed under a field corner water feeder. Because only one active drain was found under grassland, and the potential differences in sediment chemical properties compared to arable drains appeared significant, replication was achieved by collecting samples over different time periods (each sample 6-10 weeks). This replication method was also performed for arable tile drains. While this approach may affect sample independence, as the sampled drains represented a majority of all potential drain inputs this bias is not problematic; the variance between sampling period (crop development stage, presence or absence of livestock) is potentially larger than that between drains over the same time period.

Representative soil samples were collected from the six fields within the catchment from 0-5 cm in October 2009. Bank material was sampled at intervals of approximately 100 m (concentrating on areas which appeared to be subject to active erosion) in October 2009. Samples of track runoff sediment were collected directly at the drain outlet to the ditch during three major runoff events, and supplemented with further samples taken from the track surface close to the drainage point. Sediment retrieved from the sediment trap installed in December 2008 was collected for comparison with material collected by the sediment sampler in December 2009.

Source/end group	Replicates
Arable tiles	6
Grass tile	4
Farm track	6
Channel banks	5
Sediment trap	4
Ditch sampler	6

Table 2.1: Fingerprint sediment sample regime.

Suspended sediment samples were frozen after collection from samplers (ditch and tile drains) and freeze dried. Soil, bank, track surface and sediment trap samples were air dried. All samples were gently screened through a 200 µm mesh sieve to remove stones and coarse sand particles. This was undertaken in order to reduce particle size related errors in source chemical concentrations. Previous studies (e.g. Russell *et al.*, 2001) have used a 63 µm screening, to gain source samples representative of particle size distribution of outlet suspended sediment. However, the use of such a procedure is likely to result either in destruction of coarser primary particles if rigorous grinding is employed, or the rejection of a large proportion of the source material if a gentle screening technique is used. In both cases this potentially introduces a large error, hence the use of a coarser screen in this study.

Samples extracts were prepared by acid digestion using *aqua regia* solution. Samples were analysed for total As, Ca Cd, Cu, K, Mg, Na, Pb, Si, Zn using an inductively coupled plasma optical emission spectrometer (ICP-OES, type Vista MPX, Varion). Extracts were also analysed for total P concentration by acid molybdate reaction. Total carbon and total nitrogen were analysed using *Elemental vario max* CNS analyser. Particle size distribution was performed on hydrogen peroxide treated samples according to the pipette method (Avery and Bascomb, 1974).

2.3.3 Field observations

Detailed field observation was undertaken during the data collection period (2008-2009) to provide evidence to support the quantitative evidence collected (described above). These observations are supported by opportunistic (manual) water sampling where appropriate.

2.3.4 Statistical analyses

Differences in upstream and downstream sediment concentrations were tested using a 2 sample paired t test. Difference in source group sediment chemical concentrations were tested using Kruskal-Wallis test, and Mann-Whitney test to compare individual groups. Differences in source group clay and sand content were tested using a general linear model and Tukey's post-hoc test to compare individual groups. Source fingerprint chemical parameters were selected using Discriminant Function Analysis. This procedure is described in detail in subsection 2.4.3

2.4 Results

2.4.1 Flow and rainfall characterisation

Precipitation and discharge event records over the monitoring period (2007-2009) are summarised in Table 2.2. Peak specific runoff totals vary between approximately 2.5% and 21% of peak hourly rainfall. Daily precipitation totals exceeding 10 mm and hourly intensities exceeding 2 mm are rare (Figures 2.5 & 2.6). No clear seasonal patterns in rainfall are evident, although a tendency towards larger rainfall events is observed in summer months (Jul-Sept) during this monitoring period. Flow duration analysis (Figure 2.4) demonstrates that the ditch flows approximately 94% of the time, suggesting a permanent watercourse. Q_{10} of approximately 6 $l.s^{-1}$ demonstrates high flows to be rare (Q_{50} is approximately 2 $l.s^{-1}$). Very high peak flows relative to average flow conditions demonstrate that the farm catchment produces infrequent, but high magnitude responses to precipitation events, which are characterised by relatively low precipitation intensity (Table 2.2).

Date	Total event P (mm)	Q peak ($l.s^{-1}$)	peak runoff ($mm.hr^{-1}$)	peak hourly rainfall (mm)	mean rainfall intensity ($mm.hr^{-1}$)
13/05/2007	12.6	16	0.09	3.6	0.6
06/07/2007	5.4	19	0.11	2	0.6
20/11/2007	21.2	20	0.12	4	1
01/01/2008	25.6	72	0.42	2.2	0.7
21/01/2008	17.6	76	0.44	3	1.1
12/04/2008	4	16	0.09	1.2	0.9
27/04/2008	17.8	82	0.48	4.4	1.2
30/04/2008	11.4	130	0.75	3.6	0.8
07/07/2008[1]	35.8	95	0.55	-	-
09/07/2008	22.4	91	0.53	4.4	1.6
10/07/2008	12	17	0.10	2	0.6
01/08/2008	14.6	25	0.15	6	1
14/08/2008	24.6	204	1.18	5.8	1.8
18/08/2008	27.2	173	1.00	8.4	1.4
05/09/2009[2]	155.8	-	-	7.4	2.4
01/10/2008	9.4	31	0.18	5.6	2.4
14/10/2008	11.6	23	0.13	3.6	1.5
30/10/2008	11.6	22	0.13	2.2	0.6
09/11/2008	8.8	38	0.22	3.4	1.3
05/12/2008	2.8	22	0.13	1	0.4
13/12/2008	11.8	30	0.17	0.8	0.5
03/02/2009	22.6	129	0.75	3.2	1.1
14/02/2009[3]	0	23	0.13	-	-
01/07/2009	15.4	42	0.24	6.2	1.4
17/07/2009	44.8	129	0.75	3.2	1.7

[1]Rain gauge failure [2]event missing flow record (05/09/2008) [3]Snow melt

Table 2.2: Nafferton catchment Discharge event summary 2007-2009.

'Events' are defined as identifiable increases in discharge from seasonal base/low flow conditions.

Event separation was performed manually using a criterion of return to base flow (assumed where equal to event start base flow). Event rainfall intensity was estimated on the basis of all mean hourly rainfall hours within each event (min. detection = 0.2 mm).

Q peaks are based on instantaneous stage logs collected at 5 minute intervals.

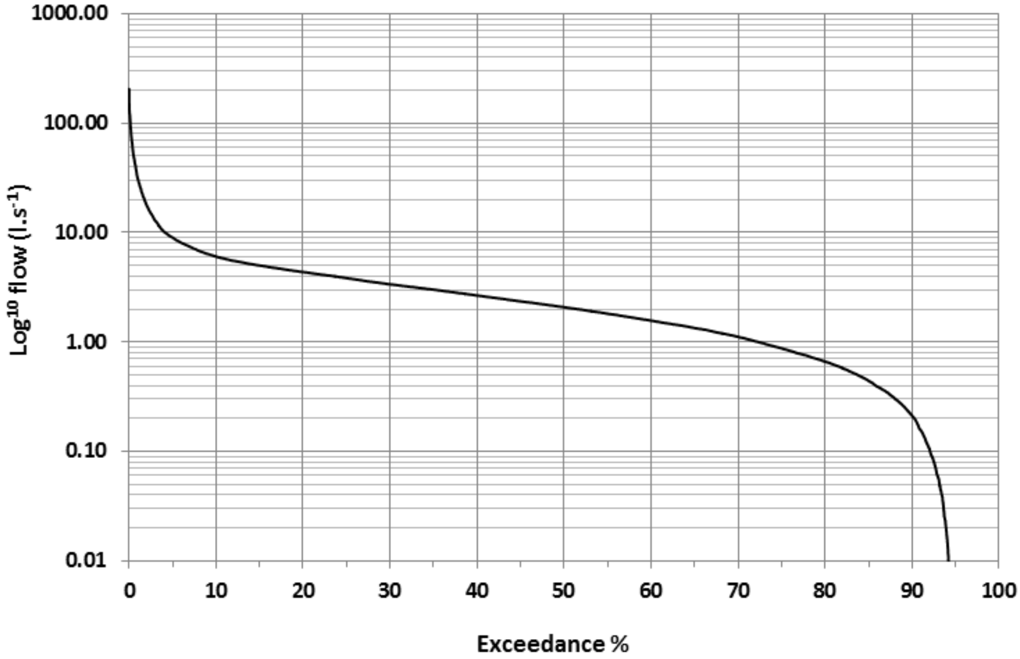

Figure 2.4: Nafferton mean hourly flow duration curve 2007-2009.

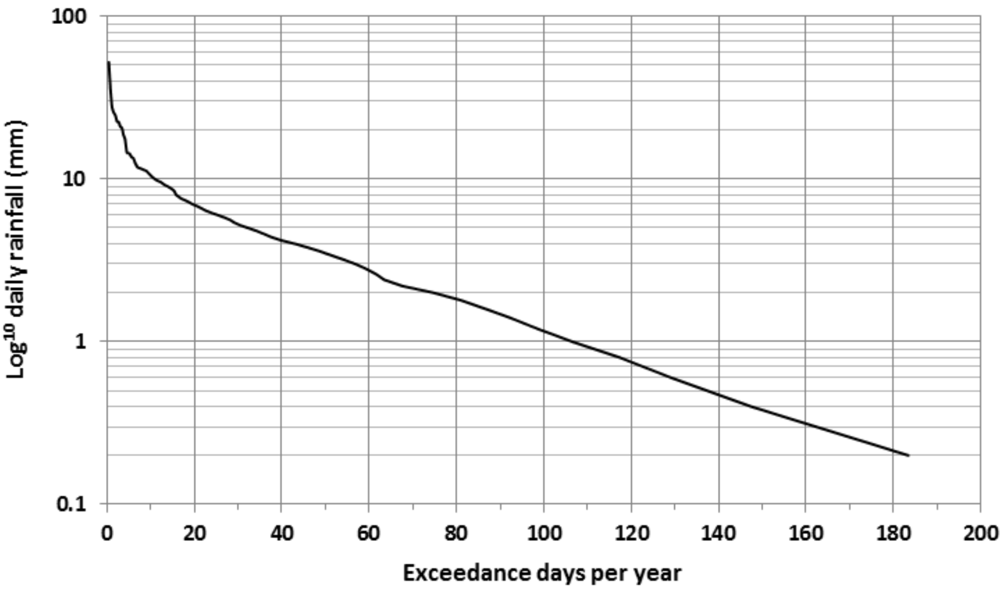

Figure 2.5: Nafferton daily rainfall exceedance frequency 2007-2009.

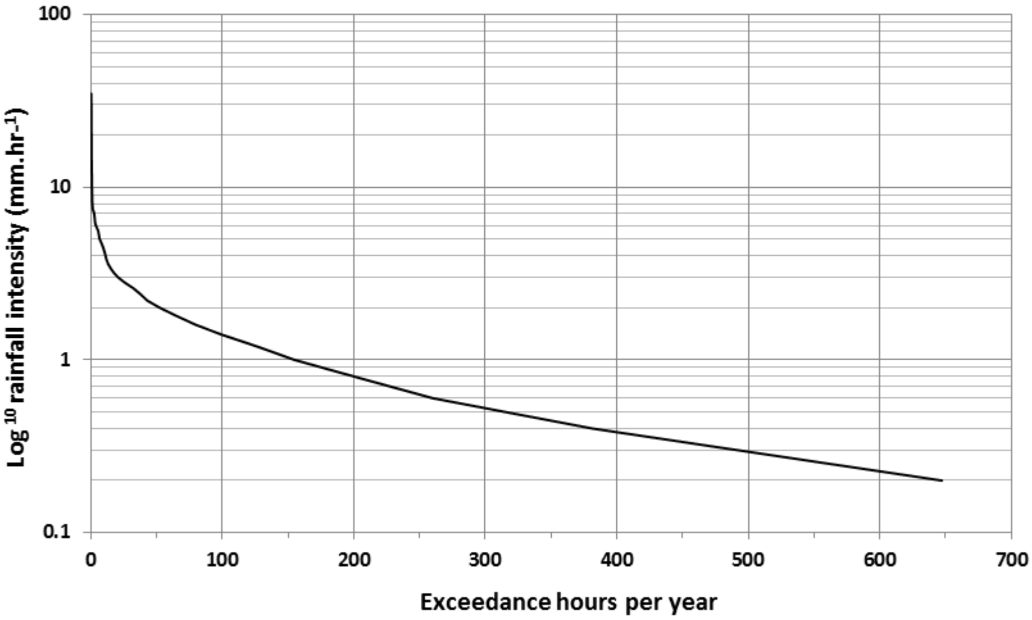

Figure 2.6: Nafferton hourly rainfall exceedance frequency 2007-2009.

2.4.2 Catchment sediment export

Figure 2.7 illustrates that the sampled events (lettered) for the sampling period January-July 2009 are representative of the range of discharge magnitude during this period and that the data presented and derived calculations are a good representation of the full range of hydrological conditions experienced during this six month period. The data presented is selected to illustrate sediment loss process in relation to events of different magnitude and temporal occurrence.

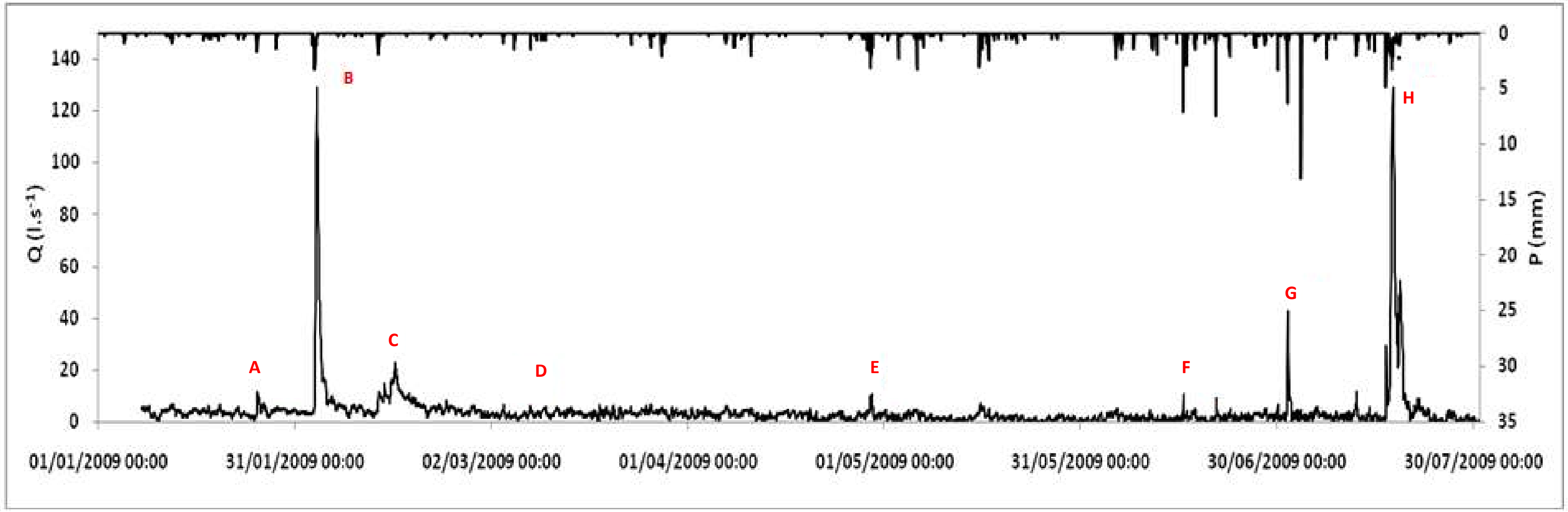

Figure 2.7: Events with sediment sampling record Jan-July 2009.

Event B is a relatively large magnitude event for this catchment. A significant hydrograph lag from peak rainfall intensity is exhibited (Figure 2.8). Rising limb sediment concentration measurements are not available for this event. Falling limb measurements show a rapid decline strongly correlated with discharge.

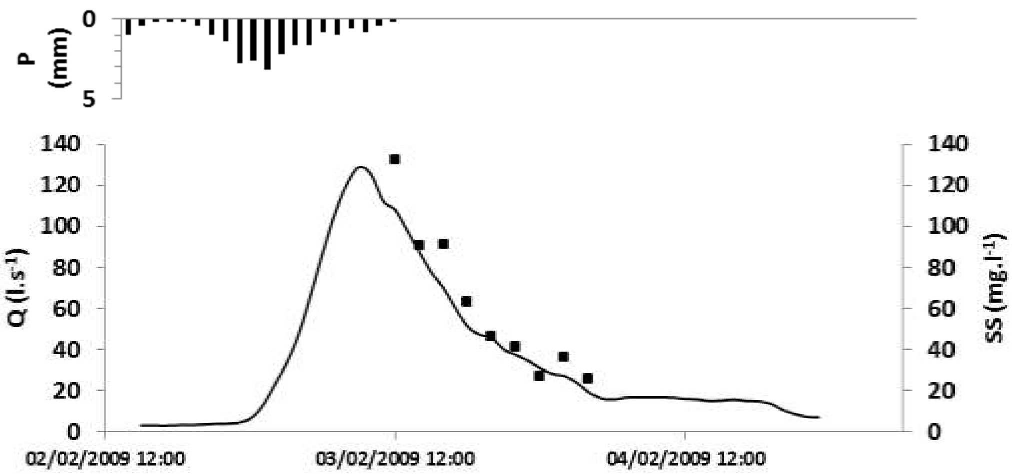

Figure 2.8: Sediment and flow record event B.

Event C is a low magnitude discharge event occurring over a prolonged time period in response to snow melt (Figure 2.9). The concentrations of sediment measured during this event are an order of magnitude higher than other event samples. Although peak concentrations can be seen to coincide with peak discharge in common with other events, uncertainty exists over the accuracy of these figures; neither the magnitude of the rainfall/snow melt event, or the discharge response appear to explain the high sediment concentrations occurring over a prolonged time period.

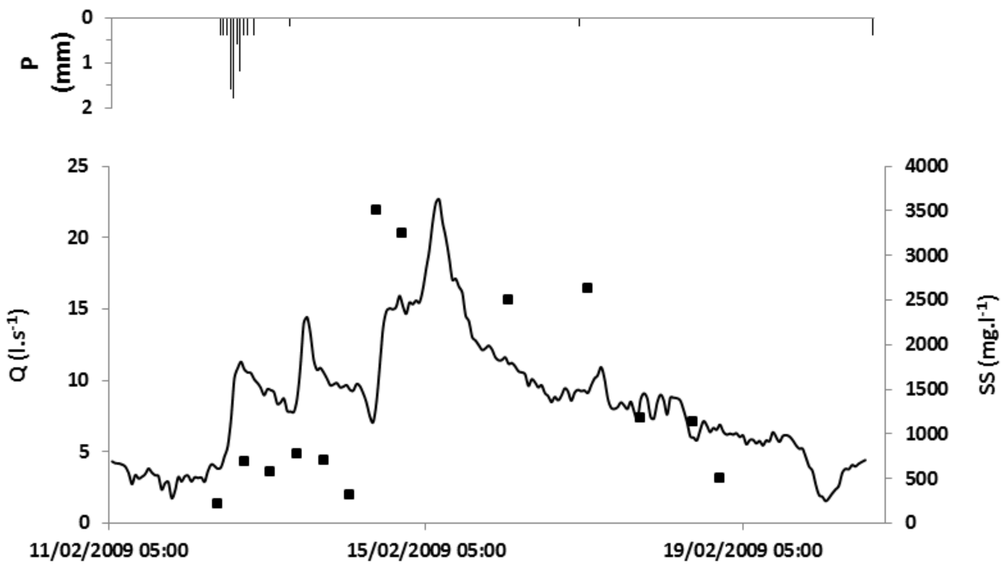

Figure 2.9: Sediment and flow record Event C.

Event E demonstrates a relatively minor discharge response to a large rainfall event occurring during spring. Despite the limited runoff response, sediment concentrations can be seen to peak sharply in coincidence with peak rainfall intensity (Figure 2.10). The second peak in discharge occurs in response to lower rainfall (higher catchment runoff coefficient) and appears to illicit a more limited sediment response.

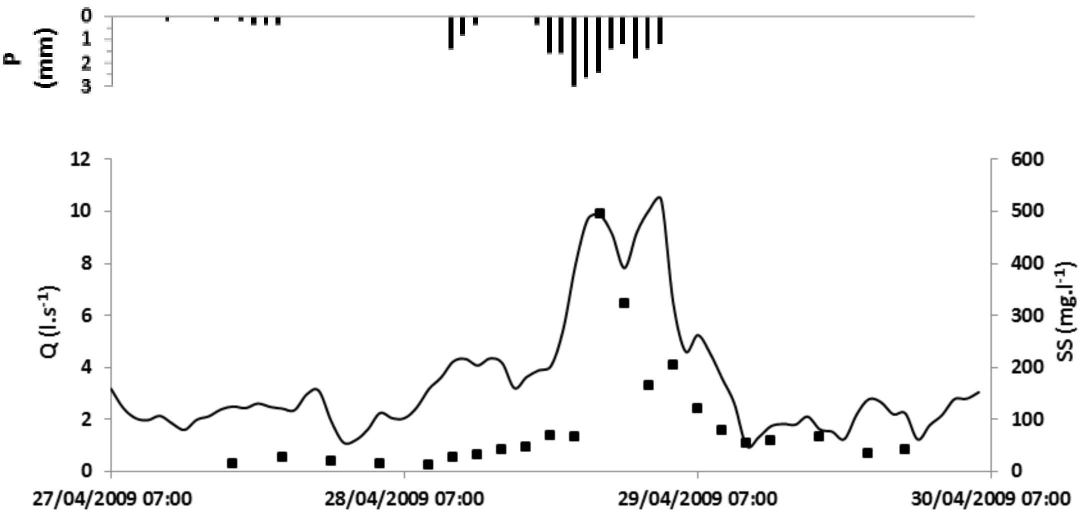

Figure 2.10: Sediment and flow record Event E.

Event F is a relatively intense short duration summer storm with rapid catchment response (Figure 2.11). Sediment concentration peaks and declines rapidly, although outlying values occur at low discharge pre and post peak. Occasional inaccuracies at low flows are expected due to short term input (pulses) of sediment for example.

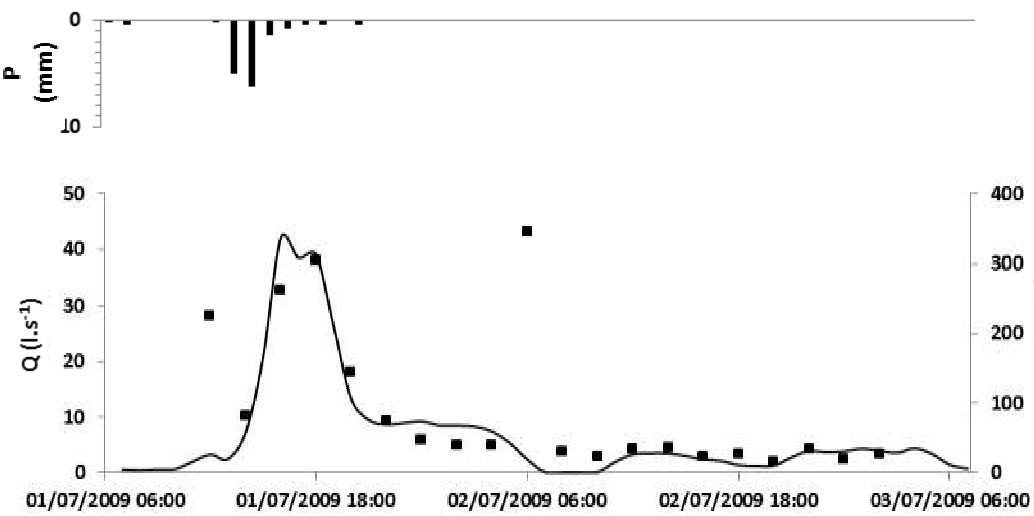

Figure 2.11: Sediment and flow record Event F.

Event H is a relatively high magnitude rainfall event of low rainfall intensity (Figure 2.12). Only spot sample readings are available for this event (further points within the catchment were monitored at the same sample times for exploration of sediment sources and are presented in sub-section 2.4.4). Sediment concentrations at the discharge peak are significantly lower than those during other events presented, including those of much lower magnitude.

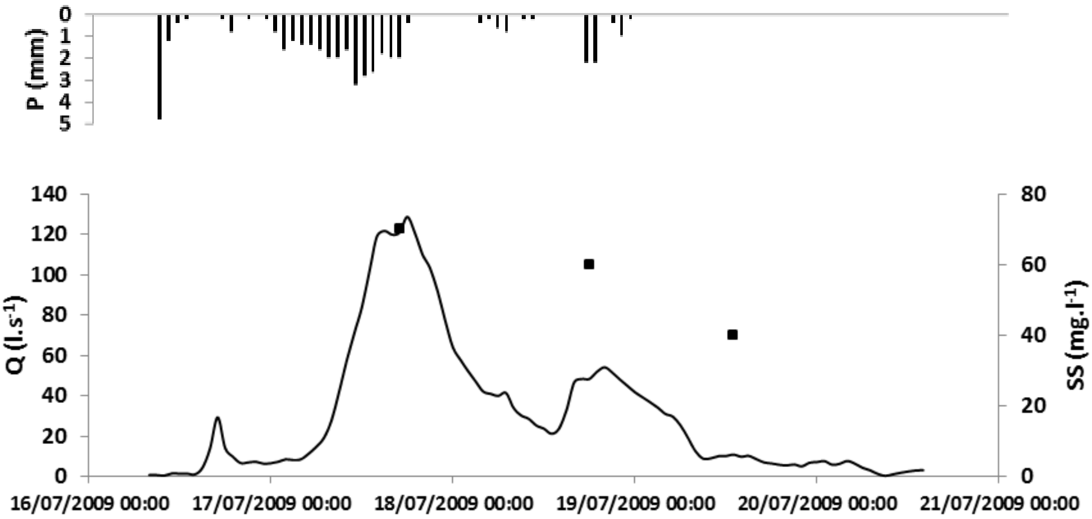

Figure 2.12: Sediment and flow record Event H (manually derived samples only).

Cumulative sediment export (estimated from spot samples and hourly mean discharge) demonstrates that the majority of estimated sediment export was derived from peak discharges (note figures represent the cumulative total for sampled hours only, not for the entire sample period). Due to the uncertainty over event C, exports are calculated both with and without the inclusion of these samples (Figures 2.13 and 2.14). Both indicate the same pattern; over 70% of derived sediment was exported during less than 10% of the sample hours. Figure 2.14 indicates 50% of sediment is delivered in 6 sample hours (2.4% of sampled hours) and 90% in 31 hours (14%).

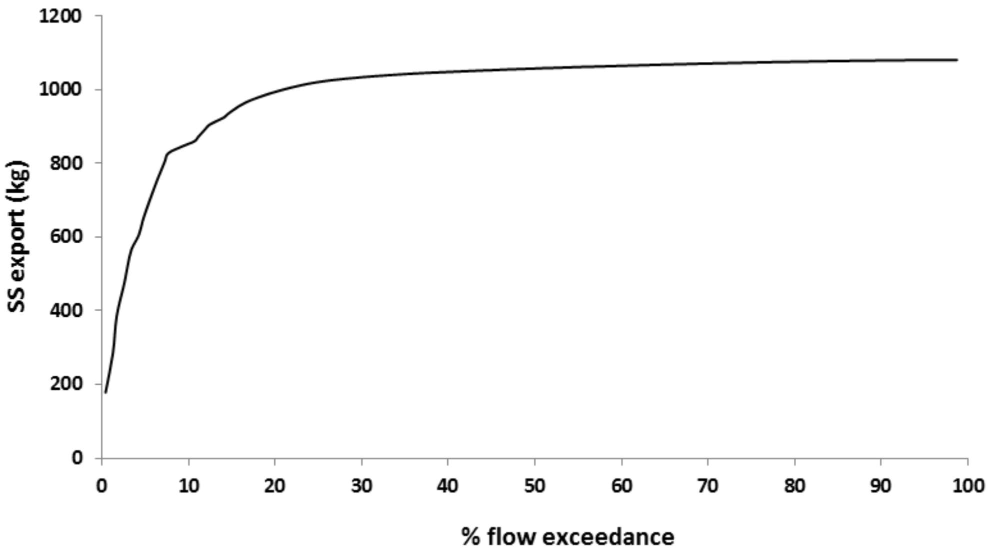

Figure 2.13: Cumulative sediment export Jan-July 2009 for sampled hours (including event C).

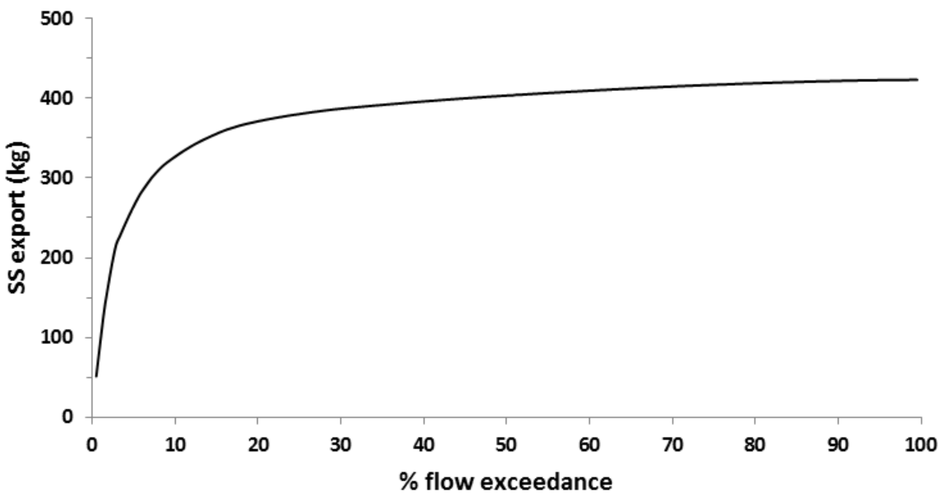

Figure 2.14: Cumulative sediment export Jan-July (excluding event C).

2.4.3 Sediment source fingerprinting

The sediment source fingerprinting procedure is based on that employed in previous studies (e.g. Collins *et al.*, 2007). Following the identification of potential sources and sample analysis as described in section 2.3.2, the following stages are applied as described in detail below.

Stage 1: Parameter screening

An initial screening of fingerprint parameters was conducted to analyse differences between group averages using Kruskall-Wallis Z test and Mann-Whitney U test. The use of non-parametric tests in preference to parametric (Analysis of Variance) was made as the assumption of equal variance between treatments was not upheld for this data; this was particularly the case as *temporally* variable sources (track runoff, tile drains) had a much greater coefficient of variation for most parameters than *spatially* variable sources (soils and channel banks). The use of the Kruskall-Wallis test as a means of eliminating redundant properties in sediment fingerprinting procedures is widely documented in the literature (Collins *et al*., 1997, 2010; Russell *et al*. 2001), and allows initial rejection of parameters which add within-group variability without adding group separation discriminatory power. The additional use of Mann-Whitney U test is suggested here to identify which individual source groups are separable, indicating the discriminatory power of the fingerprint dataset.

Source	Ca	Mg	Na	K	Zn	As	Pb	Cd	Cu	P	C%	N%
Track	44044a	5807	2042a	3810a	125	6.8ab	30a	1.0c	36ab	416a	9.0a	0.30
Grass tile	8983b	3236	737a	3188a	374	11.8a	59a	16.1a	33a	287a	4.0b	0.19
Arable tiles	8170b	1715	1029a	1604b	149	10.0a	67a	2.4b	26b	453a	3.9b	0.11
soil	3041c	1215	382b	1506b	81c	5.9b	29b	0.8c	19c	136b	3.0b	0.10
Banks	3493c	1536	339b	1518b	95c	6.2b	25b	1.0c	19c	89c	1.8c	0.001

Table 2.3: Mean group fingerprint parameter concentrations (mg/kg unless stated).

Means with same letter are not significantly different (Mann-Whitney p>0.05)

The decision was made to assign the grassland tile drain as a separate source to the arable tiles as the fingerprint differed significantly for a number of parameters (Table 2.3). It should be noted that only one active grassland tile drain was identified within the catchment, suggesting that the volume of potential sediment delivery is likely to be limited.

All parameters tested show significant differences (Kruskall Wallis $p<0.05$) between sources other than silicon. Silicon was therefore excluded from further analysis. Three groups of parameters can be identified (Tables 2.3 and 2.4); basic cations (Ca, Mg, Na, K) which show elevated concentrations in tile drain sediments and very high concentrations in track runoff; heavy metals (Zn, As, Pb, Cd) which are at greater concentrations in the track and tile runoff than in soil and bank samples; 'organic' measurements (P, N) which display greater concentrations in surface than sub-surface (channel bank) sources. However, nitrogen data is limited by the insensitivity of the analysis technique at low concentrations, which results in non-detection for some samples, particularly in the bank material. Additionally, although total nitrogen was found to exhibit discriminatory power between soil and channel banks, the variability in other sources and in the outlet sediment limits its potential as a fingerprint tracer; tracers with a high degree of within-source variability are not advisable as the use of a mean value to represent the group is misleading. This problem has been addressed in other studies by assigning weightings to parameters based on their group variability (Collins *et al*., 2010). However, this technique does not seem applicable where within-group variability (residual) differs largely between groups, as here. Total carbon also had similar discriminatory power, but very high within-group variability in the track and drain samples as well as within the end material, which limits its use as a fingerprint property. Due to these limitations, nitrogen and carbon were excluded from further analysis. The problem of soil and bank source separation is addressed later in this section.

	Ca	**Mg**	**Na**	**K**	**Zn**	**As**	**Pb**	**Cd**	**Cu**
Mg	0.933***								
Na	0.354	0.389*							
K	0.636***	0.831***	0.584**						
Zn	-0.035	0.210	0.049	0.401					
As	-0.079	0.099	0.270	0.337	0.733***				
Pb	-0.047	0.101	0.131	0.250	0.442*	0.608**			
Cd	-0.153	0.073	-0.061	0.246	0.969***	0.631***	0.345		
Cu	0.549**	0.745***	0.265	0.859***	0.471*	0.421	0.341	0.299	
P	0.534**	0.575**	0.405*	0.546**	0.205	0.358	0.491**	0.017	0.631***

Table 2.4: Peason's correlation coefficient for fingerprint source parameters.

*p<0.05 **p<0.01 ***p<0.001

Stage 2: Correction factors

Previous studies have suggested that direct use of fingerprint parameter concentrations results in inaccuracies arising from differences in particle size between source and outlet sediment due to selective mobilisation by flowing water. This selectivity may result in significant changes in tracer concentrations compared with the concentrations measured at source, because the ion retention capacity related to specific surface area is many times greater for the clay than the sand and silt fraction. Previous studies have suggested a correction factor based on particle fractionation and analysis of outlet material for *individual* parameters (Russell *et al.,* 2001) or a more general *single* linear correction based on the difference in geometric particle specific surface area (Collins *et al.*, 2010). The present study included assessment of particle size, and confirmed that as in previous fingerprint studies, the outlet clay content was significantly higher than in any of the source materials (Table 2.5), suggesting either selective mobilisation or selective retention of fine particles due to sampling method. Complete fractionation and separate analysis of source materials was not possible given the limitations of the study, and a single linear correction factor is not deemed to be justified; Russell *et al.* (2001) found that particle size correction factors varied significantly between individual chemical parameters involving both positive and negative concentration correction for the same source by factors of as much as 4 or 5. A single unidirectional correction therefore would appear to represent a considerable source of error where the relationships for individual fingerprint tracers are not explicitly tested.

Source	sand	silt	clay
Ditch sampler	12c	43ab	44a
Sediment trap	53a	26b	22b
Soil	50a	29b	21b
Banks	54a	25b	21b
Track	44b	34b	22b
Arable tiles	51ab	39ab	10b
Grass tile	23bc	61a	16b

Means with same letter are not significantly different (Tukey's test $p<0.05$)

Table 2.5: Sediment source mean particle size distribution %

This uncertainty is avoided in this study by using sediment recovered from the outlet sediment trap as a surrogate for samples collected from ditch suspensions. While the particle size of this material is much coarser than the suspended material (Table 2.5), Principal Component Analysis illustrates that the chemical signature is very similar (Figure 2.15) suggesting the effect of particle size to be limited. The trap material , unlike the suspended material was not found to differ significantly in clay content from any of the source materials and from only the grass tile in sand and silt content (Table 2.5). Accordingly, particle size correction is not made in this study.

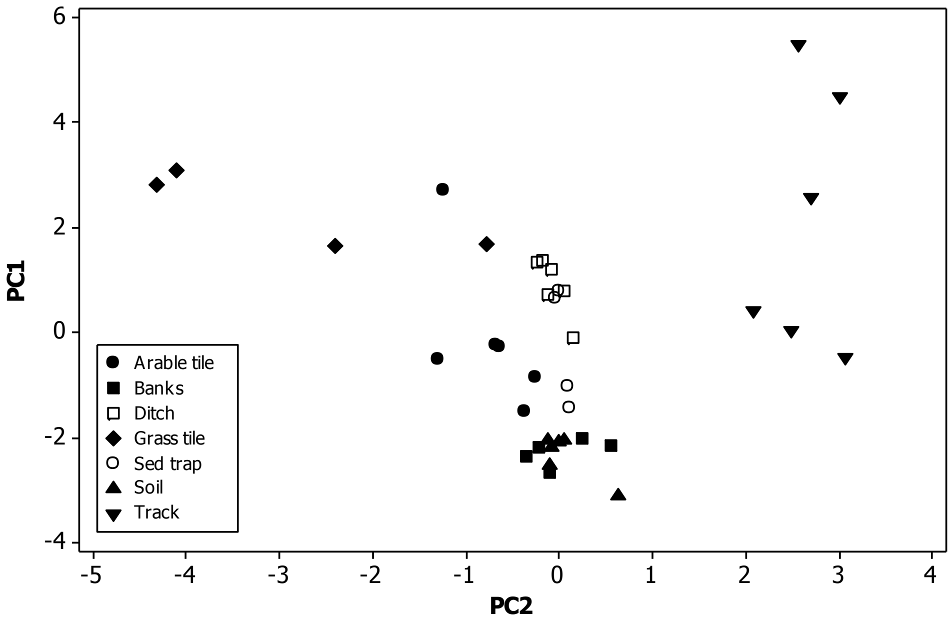

Figure 2.15: Sediment source Principal Component Analysis PC1 vs. PC2 plot.

Stage 3: Fingerprint parameter selection

The third stage involves the selection of the parameter dataset to be utilised in estimation of sediment source contribution. For this purpose Discriminant Function Analysis is employed as adopted elsewhere (Collins *et al*., 1998; Russell et al., 2001; Collins *et al*. 2010). A stepwise procedure is adopted to determine the optimum parameter set, with minimisation of Wilk's Lambda statistic (an indicator of degree of group separation) as the target. Criteria for parameter entry and exit are set at p values of .05 and .1 respectively.

Results confirm that the dataset is able to successfully separate all groups other than soil and channel banks (Figure 2.16; Table 2.6). The use of Ca, Mg, Cd, and P as a fingerprint parameter set minimises Wilk's Lambda and no further improvements can be made by adding further parameters. The output confirms the non-parametric tests for individual properties that there are no significant differences in the dataset between the soil and bank material. Incorrectly classified samples are exclusively within these source sets. The inability to separate these groups is not viewed as problematic to the analysis for two reasons; firstly the principal aim of the application of this technique is to identify the contribution made by track and tile drain sources. Secondly, survey of the catchment confirms there is very limited potential for soil surface erosion to reach the ditch within the catchment – material classified as belonging to one of these two groups is much more likely to consist of bank material.

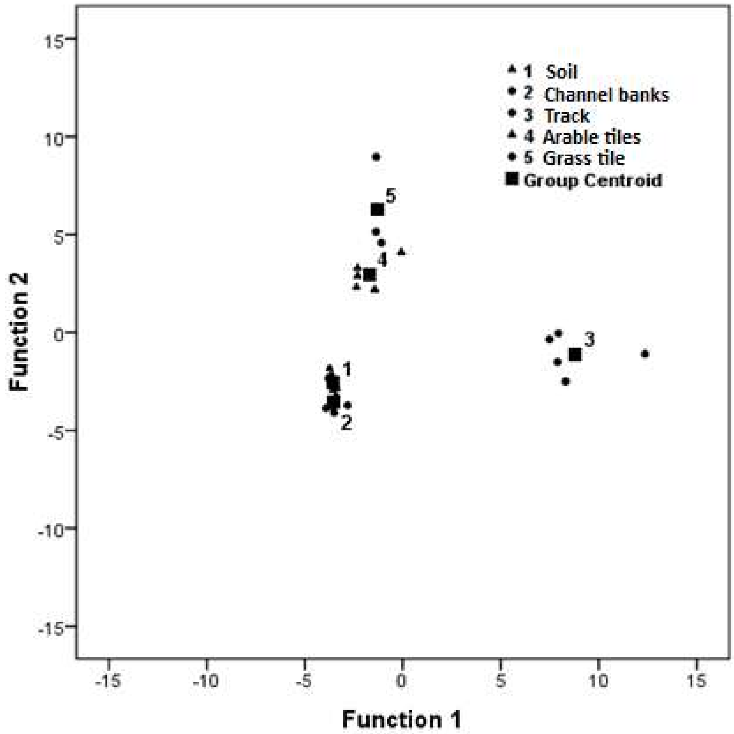

Figure 2.16: Plot of group separation by first two discriminant functions.

Variable	Lambda	Probability
Ca	.033	.000
Mg	.003	.000
Cd	.001	.000
P	.000	.000

Proportion correctly identified = 92.3%

Table 2.6: Fingerprint variables stepwise discriminant function.

Stage 4: Mixing model

The mixing model utilised is based upon those used in previous studies (Collins *et al*., 1997). The target output is the minimization of the sum of route mean square error for the parameter dataset between the outlet measurements and the proportion of contributions from the different sources (lines 9 and 7 in Figure 2.17 respectively). Excel Solver is used to derive the source mix which minimises the error. However, a problem of using this model is that the units vary by several orders of magnitude, meaning that for this dataset the minimisation of error becomes largely a function of the minimisation of the error for Ca and Mg only, which are found in much higher concentrations than Cd and P. To overcome this limitation, the sum of error *coefficients* (the error proportional to the parameter outlet value) is utilised in preference. The optimum output is shown in Figure 2.17.

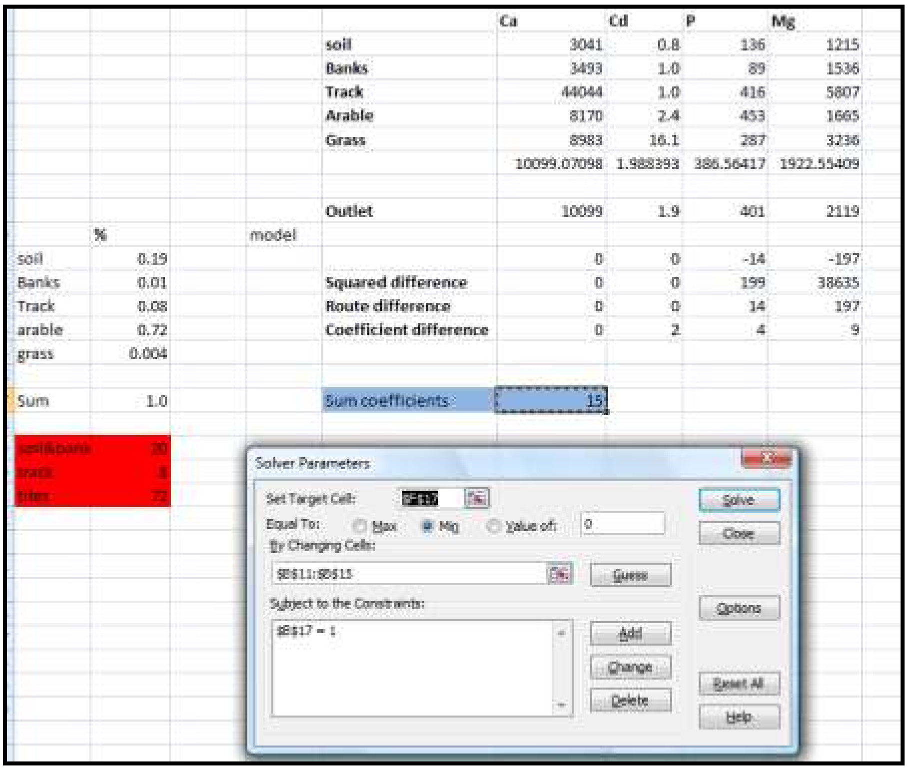

Figure 2.17: Solver solution to minimize proportional parameter error.

Stage 5: Sensitivity Analysis

One potential limitation of the technique is that the solution to the mixing model equation may be diffuse. Uncertainty is introduced by the use of mean sample concentrations for each source, which in reality represent only an estimate of the true mean. Variability of multiple source mean estimates within accepted confidence intervals may result in significant variation in model solutions. The use of a fixed mean estimate for the outlet sediment similarly adds further uncertainty. Although in this case the outlet replicates represent non-independent samples of the same material (the sediment within the sediment trap), variability still exists (which may represent measurement error as well as true variability).

To test the effects of this variability on the model solution, a Monte-Carlo simulation was employed. This involved an initial generation of random datasets based on the replicate numbers for the different source groups (n = 4-6) and mean and standard deviation estimates derived from sample values. A normal (Gaussian) distribution was assumed for random number generation. These ‘bootstrapped’ datasets were used to derive a set of variable mean estimates for each chemical parameter for each source group. 1000 mean estimates for each source were generated. An identical procedure was used to derive 1000 variable estimates of the mean for outlet samples.

The means generated were entered into four worksheets and the model solved 1000 times using a looped macro of Excel Solver using the same structure as in Figure 2.17. In the first worksheet the outlet mean values were held constant at the values used in Figure 2.17. In the second model the variable estimates of the outlet means were employed.

As the fingerprint was not able to reliably separate soil and banks, the relative contributions of these two groups cannot be accurately determined. The first model approach to this problem was to include the groups separately and sum the estimated contributions derived from the mixing model solution. However, this may introduce artificial centrality of groups and lead to inaccurate solutions. To test this effect, in the third and fourth worksheets the effect of combining soil and bank samples to derive a single group mean estimate (rather than including two separate groups) was tested. A similar procedure as above was used to derive 1000 variable estimates of the group mean for a single group (n = 10). Sensitivity was tested with both fixed and variable outlet mean values in worksheets one and two.

Model outputs

The results indicate that best fit solutions are relatively insensitive to the four approaches described above (Figures 2.18 - 2.21). Arable tile drains are predicted to represent the largest proportion, over half of all sediment delivered in the majority of model runs. Losses from the grassland tile drain are predicted to be extremely limited in all models, as the chemical makeup does not match well with the outlet fingerprint. Track runoff is also shown to represent a minority contribution with a high degree of central tendency. This reflects the fact that base metal fingerprint concentrations (Ca and Mg) are very high in this material and any potential solution to the model must necessarily contain a limited contribution, although as the outlet concentrations exceed other sources, some contribution from this source is necessary to minimise error.

These results also indicate that a single solution based on fixed mean estimates of source and outlet sampling may overestimate the contribution from arable tiles and underestimate the soil and bank contribution. This demonstrates that the technique is sensitive to deviations in mean values, which would appear to present limitations to the use of a single best fit model. Only the distribution when the outlet concentration is fixed and soil and banks grouped separately would suggest the single best fit solution to be close to the median source contribution of arable tile drains, soil and bank material (Figures 2.17 and 2.18).

The Monte-Carlo output solutions for all four scenarios were found to be strongly non-normal for all source groups. There are two principal reasons; firstly because the solutions are highly constrained for the track runoff and grass tile, these groups show excessive centrality (leptokurtosis). Secondly, this is due to the occurrence of an unexpectedly high number of values with a large standardised residual. This is particularly the case for low values, where an unexpectedly high proportion of zero values occurs (source group excluded from best fit model). As a result of this problem the use of confidence intervals (based on the standard error) to define the range of group mean contributions is not appropriate. Results are limited to the presentation of mean, median and quartile values (Table 2.7).

	Separate soil & bank groups Fixed outlet				**Single soil and banks group Fixed outlet**			
	soil & banks	**track**	**arable tiles**	**grass tile**	**soil & banks**	**track**	**arable tiles**	**grass tile**
Mean	27.5	9.5	61.7	0.1	28.8	9.4	60.2	0.2
Median	28.2	10.3	60.0	0.0	29.7	9.5	56.5	0.1
Q_{25}	19.1	7.0	53.2	0.0	18.8	6.2	51.3	0.0
Q_{75}	34.4	12.7	71.0	0.2	36.8	12.7	70.7	0.3
	Variable outlet				**Variable outlet**			
Mean	25.2	9.1	64.6	1.1	26.7	9.1	62.9	1.3
Median	24.7	9.2	65.2	0.1	26.7	8.9	62.4	0.2
Q_{25}	17.2	7.0	55.7	0.0	16.5	5.5	52.0	0.0
Q_{75}	33.3	11.3	73.3	1.8	36.4	12.7	74.5	2.2

Table 2.7: Sediment source % mixing model outputs.

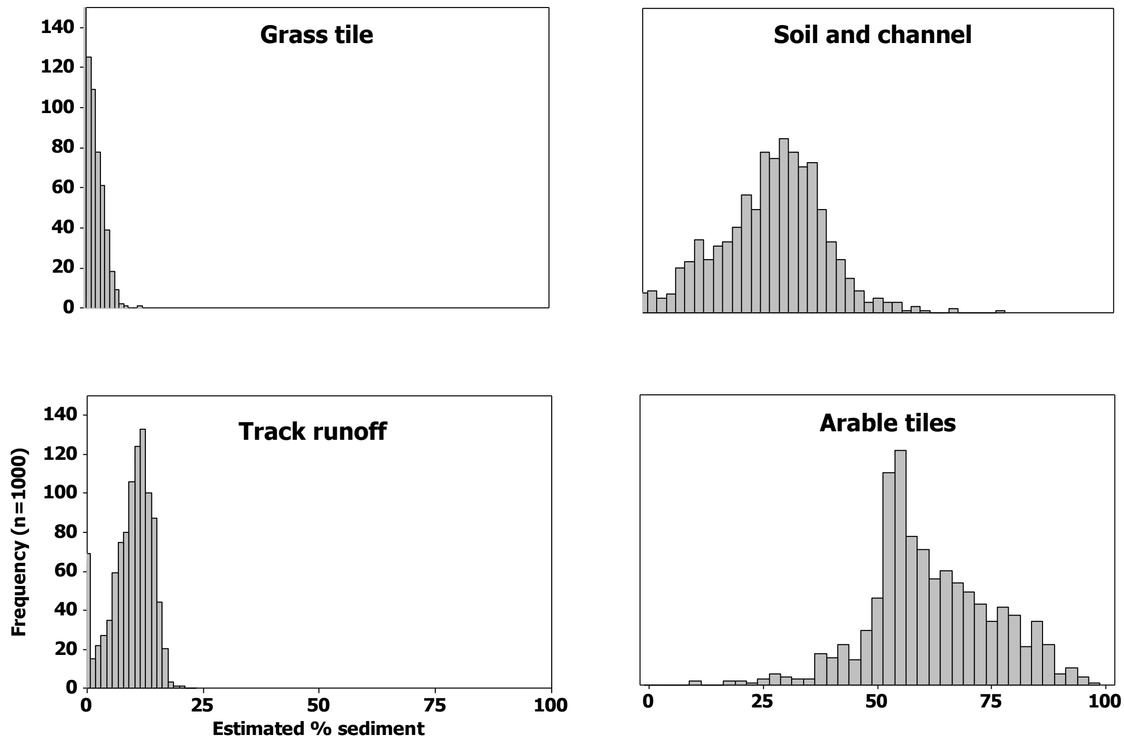

Figure 2.18: Mixing model solutions with fixed outlet values.

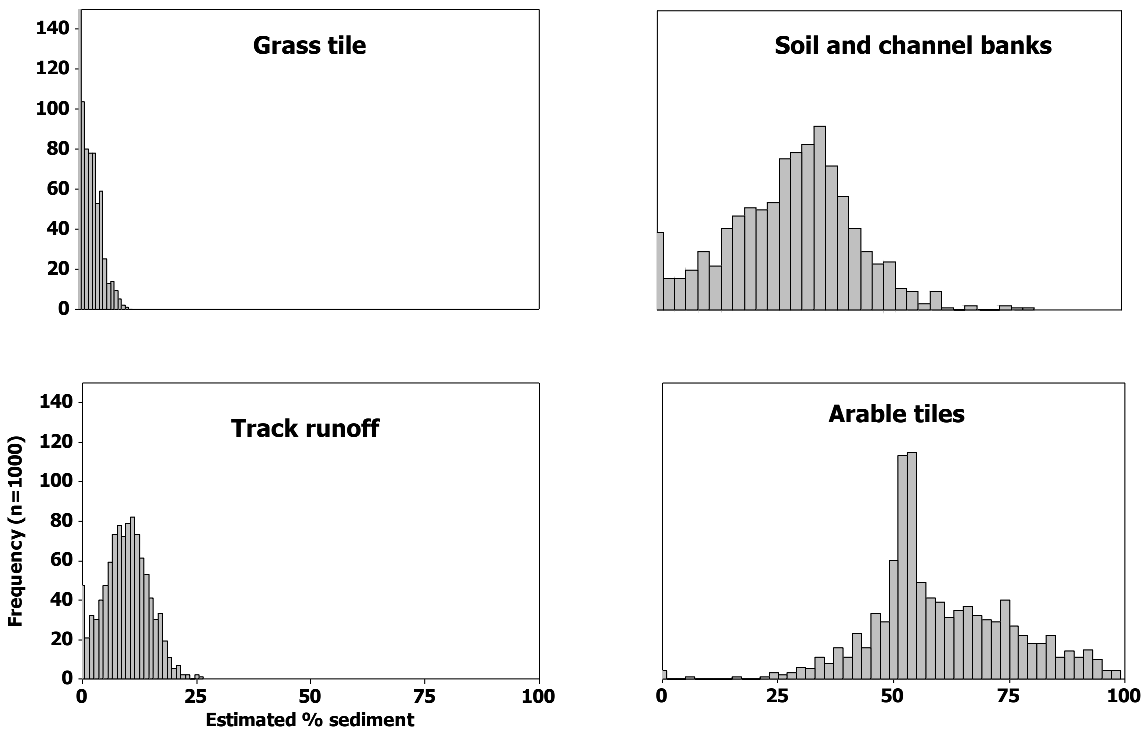

Figure 2.19: Mixing model solutions with variable outlet values.

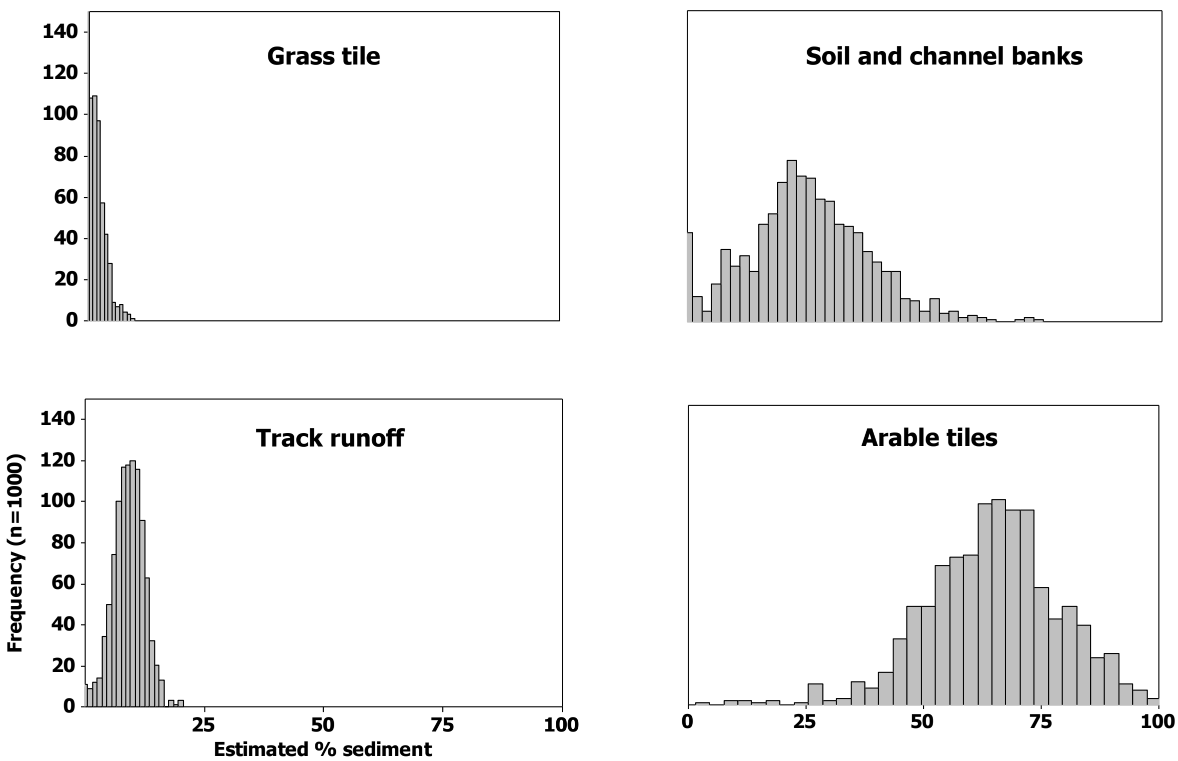

Figure 2.20: Mixing model solution with soil and banks pooled and fixed outlet values.

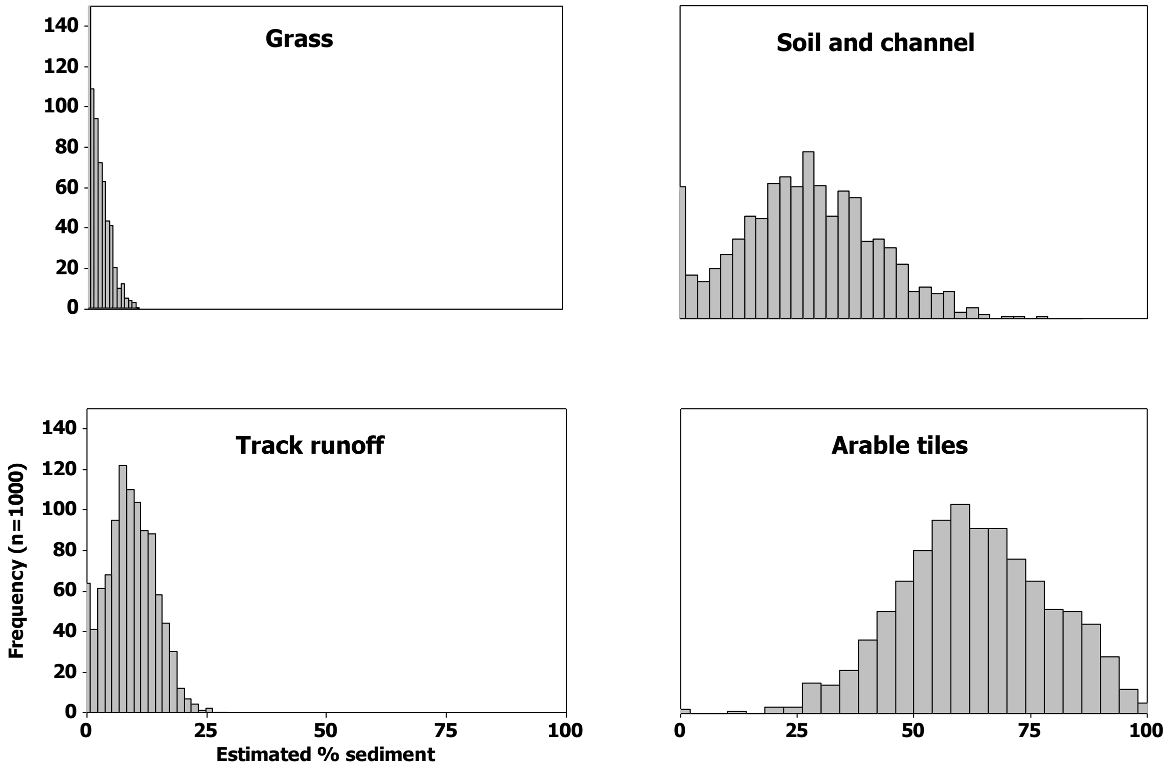

Figure 2.21: Mixing model solution with soil and banks pooled and variable outlet values.

2.4.4 Visual observations of sediment loss processes

Photograph locations are indicated by Figure 2.22.

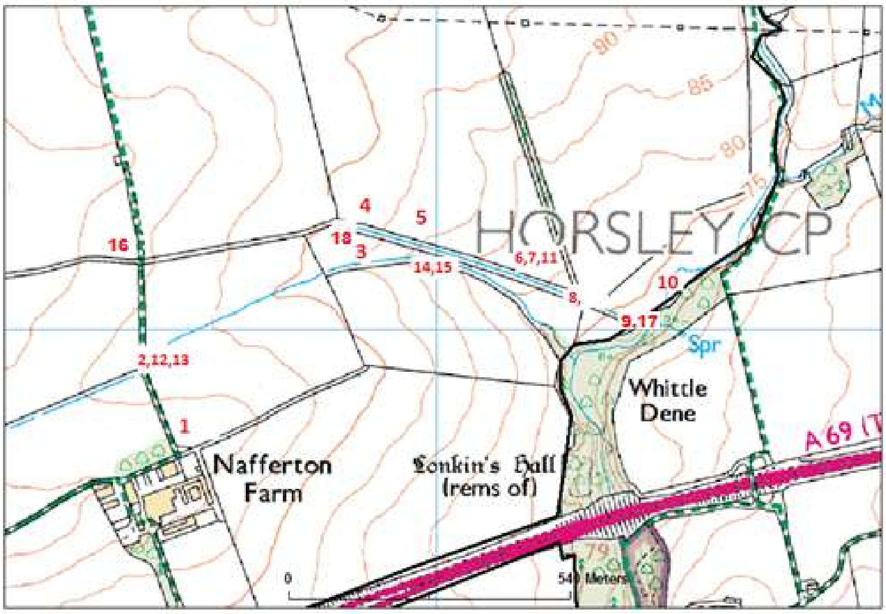

Figure 2.22: Nafferton Farm event photograph locations April 2008, January 2009 and July 2009.

Storm event April 2008

This event occurred in response to a relatively small, low intensity rainfall event (11.4mm over 16 hours, peak intensity 4.8mm.hr^{-1}). However, the event occurred approximately 50 hours following a rainfall event of similar magnitude (Figure 2.23). As a result of precedent soil saturation, large volumes of field runoff occurred during a time of high susceptibility for arable crops, resulting in considerable sediment mobilisation.

Dirty runoff was observed to occur in large volume along the main dairy track previously identified as a major sediment runoff pathway (Photo 2.1). This sediment entered the ditch network through a culvert pipe installed to prevent water ponding on the track (Photo 2.2). Contamination of track runoff water with cattle slurry makes this a critical source of pollution on the farm. Within the experimental catchment few other sources of sediment were identified; all ditch banks are stock fenced with 5 m uncultivated buffers. No surface runoff pathways from arable land were observed within the experimental ditch catchment. Runoff from grassland areas was largely limited to preferential flow along vehicle wheelings (Photo 2.3), which represent a minor sediment source (the compacted fine textured soils of these wheelings are relatively resistant to erosion).

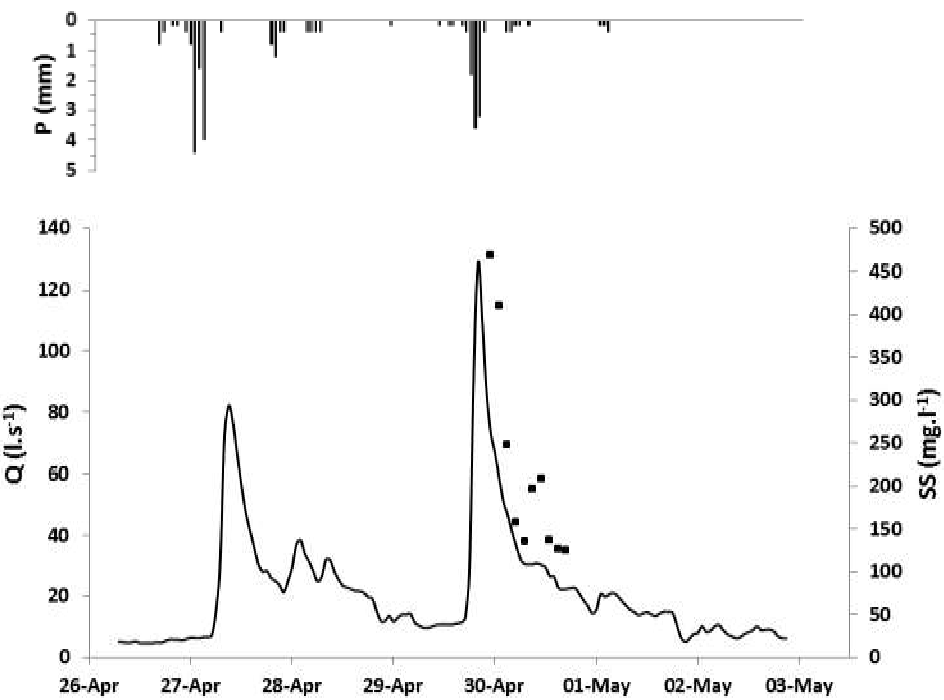

Figure 2.23: Flow and sediment record April 2008 event.

Significant sediment pathways from the farm to the Whittle Burn watercourse *outside* the experimental ditch catchment *were* identified. There was a significant interaction between farm track and field runoff, with tracks acting as a conduit for runoff (and sediment) to and from fields. For example, Photo 2.4 illustrates water from the main track (largely derived from runoff from upstream fields) flowing across a field freshly rolled for a spring crop. The susceptible nature of the recently cultivated soil resulted in significant rill and inter-rill erosion (Photo 2.5). A large sand fan in the lower field was observed (Photo 2.6) together with sediment laden water exiting the field through a farm gate, and ultimately flowing to a surface drainage ditch via the farm track (Photos 2.7, 2.8 and 2.9). This suggests that fine sediment (and the bulk of sediment-phase nutrients) were transferred to the watercourse.

Photo 2.1: Slurry laden runoff along main dairy track

Photo 2.2: Spill point of dairy track runoff to ditch.

Photo 2.3: Runoff along vehicle wheelings on grassland.

Photo 2.4: Run-on from farm track eroding Sixty Acre Field.

Photo 2.5: Rill erosion Sixty Acre Field

Photo 2.6: Sand deposits in lower corner of Sixty Acre Field

Photo 2.7: Field runoff from Sixty Acre Field laden with fine sediment.

Photo 2.8: Runoff from Sixty Acre Field transported via farm track

Photo 2.9: Track runoff and sediment transported to waterway (Whittle Burn) via a surface drainage ditch.

Photo 2.10: Whittle Burn in flood heavily laden with sediment.

Photo 2.11: Colluvial sand deposited in bottom corner of 60 Acre Field – 10-15 cm depth.

Burst pipe January 2009

On the 8th January a water main ruptured at the farm steading, producing large volumes of track runoff which entered the experimental ditch. Opportunistic samples were collected from the different catchment positions at 10 am and 1 pm to determine the maximum effect of the track on suspended sediment concentrations in the absence of runoff from the wider catchment. The results indicate that at low discharge (measured as 3 $l.s^{-1}$ at the catchment outlet) track runoff can have a major effect on suspended sediment concentrations (Table 2.8).

Using upstream and downstream measurements, track runoff discharge can be estimated as approximately 60% and 56% of stream flow at the track spill point at the respective sample times (based on the effect on sediment concentrations in Table 2.8). The increase in concentrations in track runoff at the second sample time can be explained by the effects of farm traffic introducing and mobilising sediment. It is notable that even allowing for the effect of differences between point sample times, and of lag time for sediment transport, concentrations are much lower at the catchment outlet than immediately downstream of the spill point. The short travel distance and small increase in catchment area make it unlikely this can be attributed to flow dilution, suggesting large volumes of sediment are naturally retained within the ditch.

	Sample time	
Sample point	10am	1pm
Ditch u/s track	11	9
track pipe	1420	7880
Ditch d/s track	860	4440
Catchment outlet	69	540

Table 2.8: Suspended sediment concentrations ($mg.l^{-1}$) 09/11/2009.

Photo 2.12: Dirty track runoff emanating from burst water main.

Photo 2.13: Track runoff (top) entering the experimental ditch.

Photo 2.14 and 2.15: sediment trap in operation.

Storm event July 2009

This summer event was of a relatively large magnitude, but occurred at a time of low susceptibility, given full summer crop cover on arable fields. As a result, peak suspended sediment concentrations are relatively low for a large event (see Figure 2.12). It is notable both that the concentrations of sediment in track runoff are an order of magnitude lower than those presented in Table 2.8, and that the contribution of track runoff to stream sediment concentration is much lower (Table 2.9) . During this event the track flow proportion is estimated as approximately 9% and 1% for the 17th and 18th July sample dates respectively. Tile drains would appear to account for a significant proportion of sediment export, but other sources are indicated by the higher concentrations in the stream upstream of the track than the mean for the tile drain outlets. This may represent bank erosion and/or mobilisation of bed sediments.

	Sample date		
	17-Jul	**18-Jul**	**19-Jul**
Arable tile 1	86	30	1
Arable tile 2	26	28	0
Arable tile 3	48	46	5
Grass tile	32	44	4
Ditch u/s track	86	72	34
Track pipe	396	256	n/a
Ditch d/s track	120	74	20
Catchment outlet	70	60	40
Arable field runoff	250	n/a	n/a

Table 2.9: suspended sediment concentrations (mg/l) 17-19/07/2009.

Photo 2.16: Runoff exiting arable field to farm track through gateway

Photo 2.17: Sediment laden flooded ditch (foreground) 200 m above Whittle Burn.

Photo 2.18: Runoff control measures: Clockwise from main image: temporary bund diverting flow from arable field; diversion of flow from track to ditch; flow to ditch.

Outside of the experimental ditch catchment, similar runoff issues were observed as in the event in April 2008 previously described. Field runoff was shown to originate from a wet field corner some considerable distance from the spill point to a surface ditch (Photo 2.16). This runoff travelled along a farm track to the point at which it was previously observed to run on to another field causing the erosion and sediment loss (Photos 2.4-2.7). On this occasion the issue had been recognised by the farm manager, and a temporary bund installed to divert flow from the gateway to a surface ditch running alongside the track (Photo 2.18).

2.4.5 *Sediment trapping*

Observations of sediment trapping indicate partial and variable effectiveness (Figures 2.24-2.28). Although the largest overall concentration reductions occur during runoff events with high loads, the proportion of suspended sediment reduction is higher during intermediate discharges (Figure 2.29). Some caution is required in the interpretation of sediment trap efficiency at low discharges, when the residence time within the trap means samples taken at the same time are not paired. However, overall downstream sediment paired samples show a significantly lower mean concentration of 36.9 mg.l^{-1}, compared to 58.3 mg.l^{-1} upstream of the mitigation feature, a reduction of approximately 37% (Student's t = 4.15, p<0.001; n = 81).

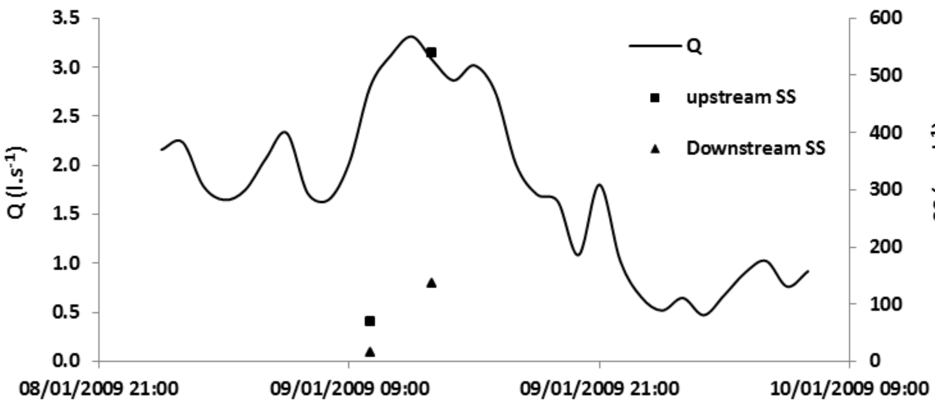

Figure 2.24: Sediment trap manual samples 09/01/2009 (see observations).

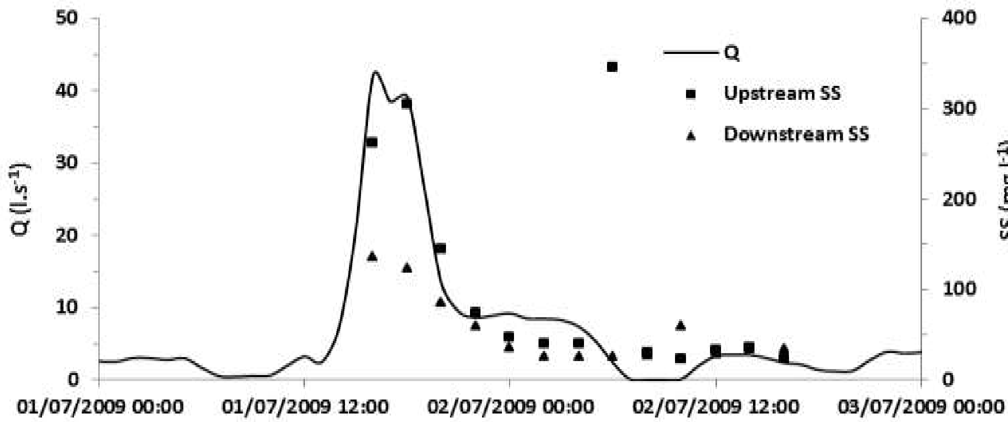

Figure 2.25: Sediment trap record July 1 2009.

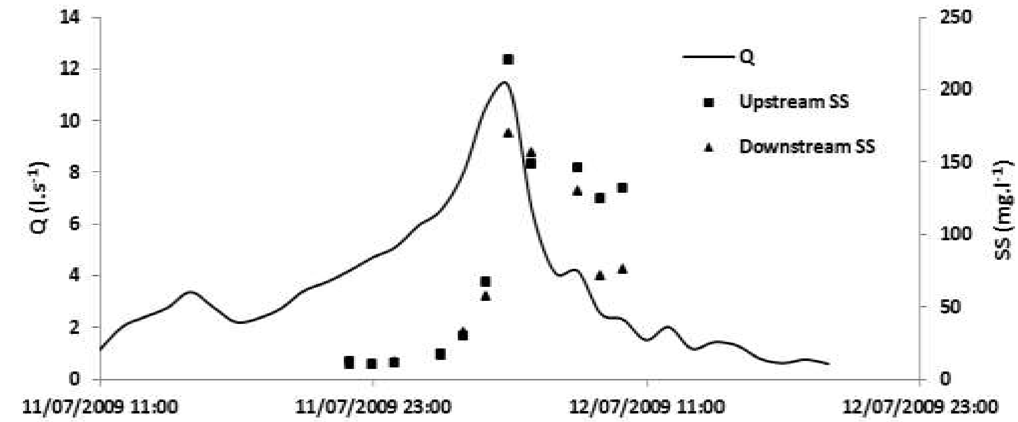

Figure 2.26: Sediment trap record July 11 2009.

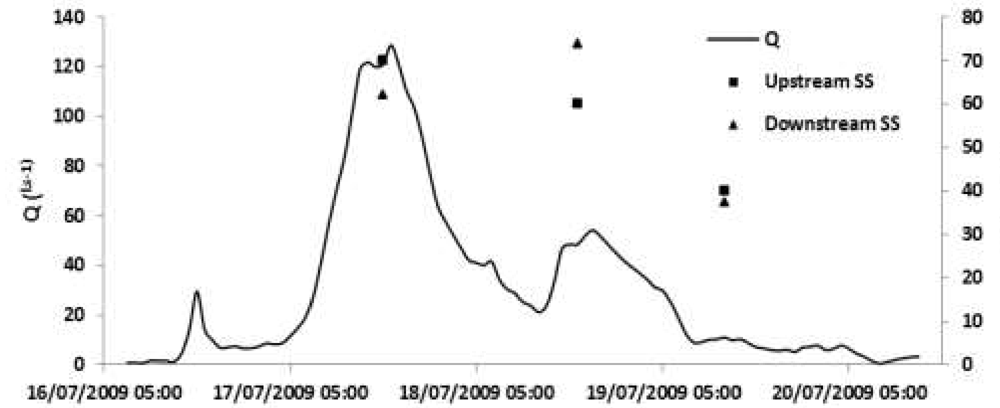

Figure 2.27: Sediment trap manual samples July 17 2009.

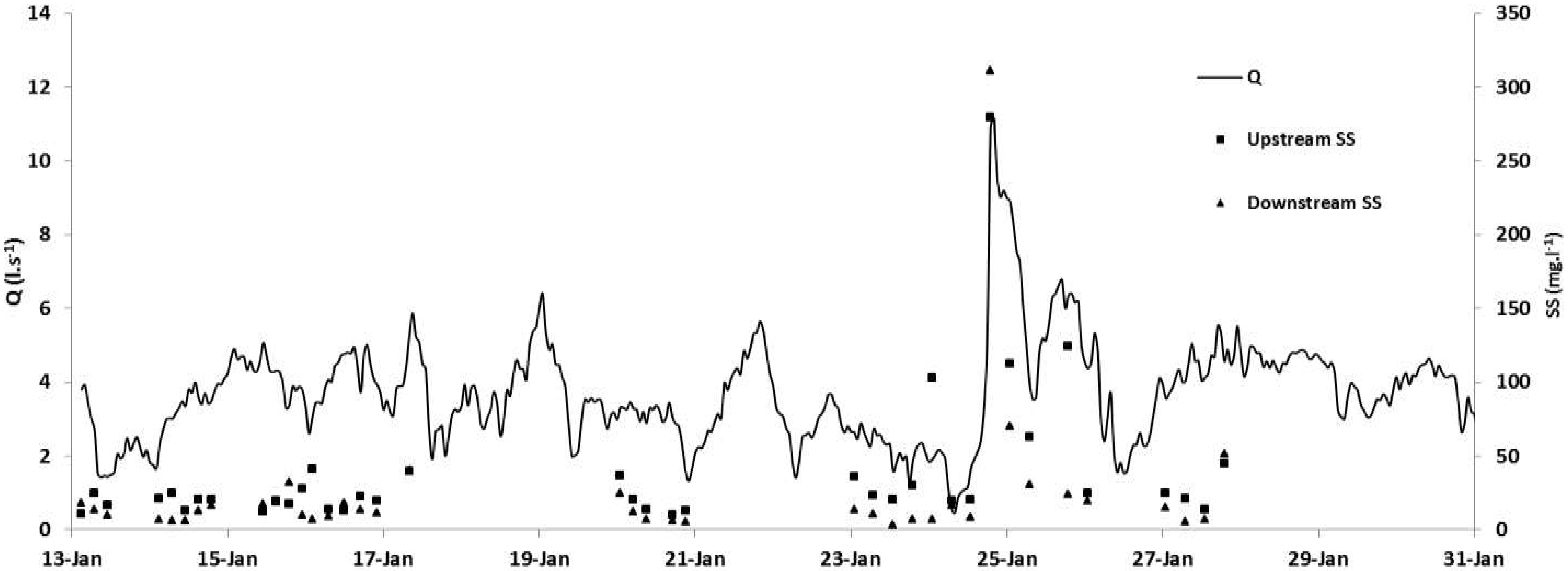

Figure 2.28: Sediment trap extended sampling period January 2009.

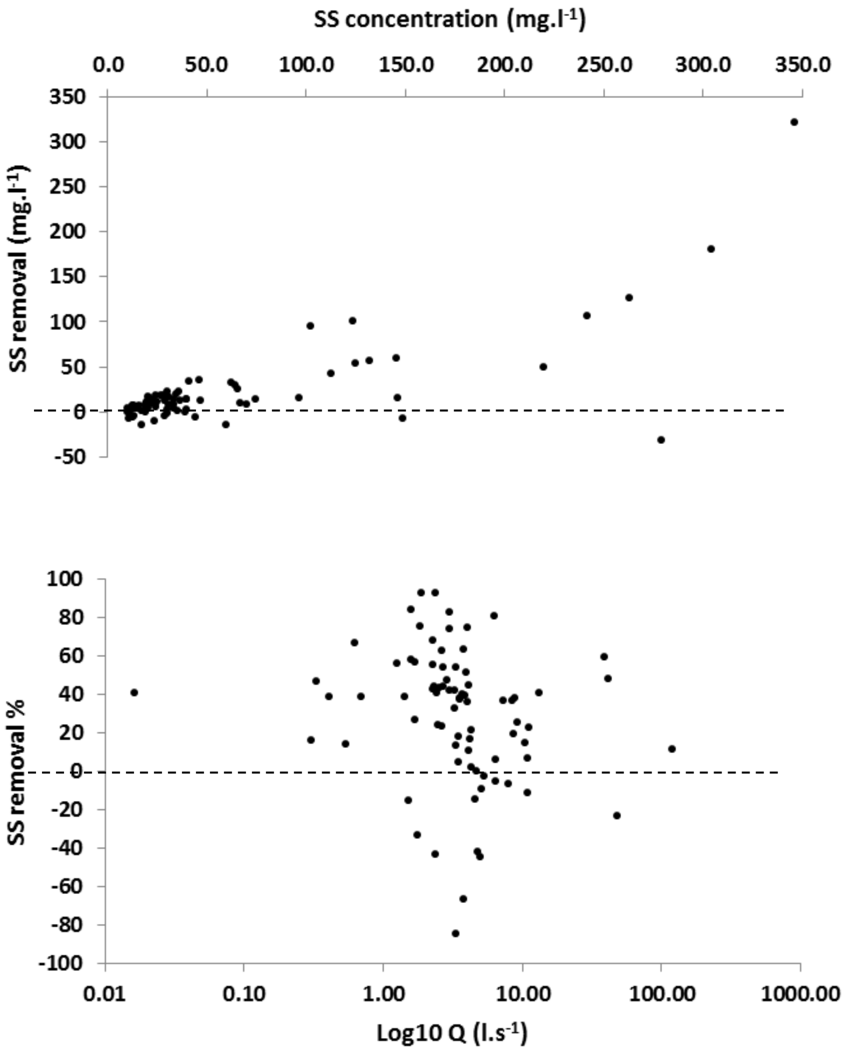

Figure 2.29: Trap efficiency vs. flow and sediment concentration.

2.5 Discussion of findings

2.5.1 Flow and rainfall regime

Flow data (Figure 2.4) indicates a flow regime typical of an artificially drained lowland catchment; the base flow component is low, and runoff response to large rainfall events is relatively high magnitude and short duration. In a tile drained catchment the tile network may be expected to account almost exclusively for low flow (base flow) and medium flows (through-flow); such artificial drainage is designed to divert sub-surface flow more efficiently through the drainage network. This is consistent with a tile dominated flow regime which may be classified as slow (diffuse) intermediate and rapid response (Schilling and Helmers, 2008) where surface runoff is limited to peak discharge when drainage capacity is exceeded. It is notable that in this study however, that both the drainage ditch and the contributing tiles continue to flow even in summer dry weather periods. This appears to indicate relative inefficiency of the existing drainage at this site. In slowly permeable soils such as those at this site, hydraulic conductivity through deeper soil layers can be very slow, and drains installed at depth without secondary sub-soil treatments may continue to transmit water over long periods. This increases the likelihood of effective drainage capacity being exceeded and the generation of overland flow.

Although no clear seasonality of rainfall occurs under the local climate, a response to events of lower magnitude is observable during the winter field capacity period (Table 2.2). Conversely, very large rainfall events were limited to summer months during the study period, meaning that the highest magnitude discharges are observed during summer. Rainfall intensities are relatively low, a well known observation when comparing rainfall erosivity in the UK with observations from North America for example. Daily precipitation totals exceeding 10 mm and hourly intensities exceeding 2 mm are rare (Table 2.2; Figure 2.6). Consequently both soil mobilisation by raindrop impact (rainfall erosivity) and inter-rill erosion generated by infiltration excess (Hortonian) overland flow may be expected to be relatively low. This phenomenon is well reported for UK climates (Beven, 2000), and means that the effects of rill and gully channel erosion caused by flow concentration in parts of the landscape are generally regarded of greater significance than inter-rill hill slope erosion. This may be particularly the case for fine textured soils resistant to lower energy mobilisation such as those at the case study sites in this study (Evans, 2002).

2.5.2 Sediment loss processes

When *absolute* sediment export is considered, the majority of sediment was mobilised during short time periods, almost exclusively associated with discharge peaks in this catchment (Figure 2.14). This may apply to large agricultural catchments as well as small; a study of a 1110 km^2 agricultural catchment in France (Oeurng *et al*., 2010) found over two years, 85% of sediment export occurred during the peak 16% of flow duration and 95% in 20% in 2007 and 2008 respectively. Over longer periods extreme events may further skew this contribution, although sediment losses from larger and more damaging floods become increasingly difficult to mitigate, and are also likely to include catastrophic slope and bank failures as well as erosion losses. If the effects are measured through the potential for channel or siltation or the effects of particulate inputs on lake or coastal eutrophication for example, control of large events becomes the key factor in limiting the impact of agriculture on sediment losses to water. Large amounts of sediment retained within the ditch network may also cause water eutrophication problems through release of dissolved nutrients, although this phenomenon has not previously been observed at this study site (J. Jonczyk, pers. comm.).

Some studies demonstrate that agricultural catchments can be subject to long durations of suspended sediment at high concentrations. Bilotta *et al*., (2010) in a study of a grassland catchment in Devon of comparative size and duration to the present study found that the 25 $mg.l^{-1}$ suspended solids figure used as a water quality threshold for freshwater fish (2000/60/EC) was exceeded over 50% of the time. The current study found that this threshold was exceeded in 54% of measurements. It should be noted that data collection was predicated on event periods, meaning base flow conditions are under-represented. However, measurements at or below median flow ($2.l.s^{-1}$) had a median concentration of 27.4 $mg.l^{-1}$.

Data indicate significant inter-event variability in sediment mobilisation. This may in part be attributed to seasonal differences in land use activities, particularly timings of stock grazing and cultivations in this farm catchment. However, short duration variability would also appear to indicate some effect of antecedent conditions, which may be explained by variation in the supply of bed sediments particularly for smaller magnitude events. Sediment deposition, particularly during flow recessions leads to temporary storage and remobilisation during future events. The precedence of one event with another within a limited time frame may limit the supply of easily mobilised material for the second event. The possibility of such variability means relationships between sediment exports and (seasonal) land use susceptibility, rainfall intensity and runoff coefficient are difficult to recognise given the length of monitoring period.

2.5.3 Sediment sources

Despite the highlighted methodological uncertainties the results of the sediment fingerprinting exercise suggest that the majority of sediment export from the study catchment was derived from arable soil erosion via the tile drainage network. This finding is in broad agreement with previous findings for similar artificially drained catchments (Russell *et al*., 2001; Deasy *et al*., 2009) and with the assumptions of the PSYCHIC prediction tool for sediment loss risk from agriculture (Stroemqvist *et al*., 2008; Collins and Anthony, 2008). Losses from the farm track, identified as a significant pollution 'hotspot' on the farm were shown to be of lesser importance to catchment yields, although of much higher yield per unit area (circa 10% from 0.6 ha) than for the tile drains (circa 60% from 33.4 ha). This evidence is supported by spot samples collected during a large storm event in July (Table 2.7,) when although high concentrations of suspended sediment were observed in track runoff, the effect on ditch concentrations was limited despite the fact that concentrations of sediment derived from upstream sources (tile drains, bank erosion and remobilisation) were relatively low in this event. These spot measurements also appear to illustrate that tile drain sediment losses occur in response to rainfall and are negligible during dry weather. This would support the contention reported elsewhere (e.g. Russell *et al*., 2001) that tile drain sediments are derived primarily from rainfall mobilisation of surface soil.

The relatively low contribution from the farm track appears surprising given the visual evidence supporting losses via this source. However, the small contribution measured during storm runoff (Table 2.7) appears to corroborate these contributions as limited, in absolute terms by the finite supply of sediment, and temporally by runoff mobilisation potential from this small proportion of the catchment. In contrast, runoff through the tile drainage network is derived from a large proportion of the catchment area and is limited only by the drain flow capacity of the pipes, likely to be exceeded only in large runoff events.

It should be noted that as mass balance calculations are not available, the specific sediment export for the catchment cannot be estimated (the absolute contribution from drains is addressed in detail in the next chapter). A caveat must therefore be attached to the finding of majority sediment source through tile drains, as absence of other significant sediment sources may be one explanation for the high contribution. However, the apparently low contribution from track runoff, despite visual observation to the contrary, indicates that this is not an unusually clean catchment and that there are important implications for the use of visual evidence for identification sediment pollution problems; the contribution from drains is likely to be very difficult to assess, doubly so if drain exports occur only at certain times of year (cultivated fields) and only in high rainfall. An over-emphasis on pollution 'hotspots' may therefore result in partially or wholly ineffective mitigation success.

Russell *et al.* (2001) suggest that contributions from drains are highest before and after peak discharge, as greater proportions of sediment are delivered at peak flows in large events via overland flow. The fact that they find annual drain sediment contribution to be much higher than surface erosion suggests that lower magnitude events (plus the rising limb and recession of larger events) represent a greater proportion of total sediment export than that generated during the large event peaks. This appears feasible, but without a longer record it is not possible to take into account very large infrequent erosion events, which may represent a large proportion of sediment export although potentially not occurring within the average year. Deasy *et al*. (2009) using direct measurements of the same study site as Russell *et al*. also found that the drain losses were dominant (24% of total from 26.5% of the catchment), but that surface losses were less important (1%) than the previous study suggested (citing possible effects of land use change) during a 6 month monitoring period. This would appear to illustrate that an efficient drainage system may account for the majority of runoff and hence sediment delivery. However, the lower sediment delivery per unit runoff reported also suggests that infrequent and irregular large surface pathway erosion events make a significant contribution, which is likely to be inaccurately estimated in short- term studies.

The fact that soil surface and bank losses represent a minor contribution in the present study catchment would appear valid due to the absence of surface runoff pathways within the ditch catchment and that of bank damage given the presence of fenced riparian buffer strips. Some bank material mobilisation may be expected in these circumstances, but the relatively low peak flow energy and the cohesive nature of the boulder clay forming the lower banks are likely to mean ditch banks are relatively stable.

Critique of the sediment fingerprinting technique

The current limitations of the sediment fingerprinting technique are robustly critiqued elsewhere (Davis and Fox, 2009). The evaluation of this technique derived from this study acknowledges many of the criticisms in this review. Readers are directed to this review for a fuller evaluation of the state-of-the art.

The suspended sediment fingerprinting approach has evolved from end member mixing model analysis successfully employed as a technique for hydrograph separation for many years (Joerin *et al*., 2002; Davis and Fox, 2009). However, application to sediment source separation involves additional layers of complexity; the number of source-pathways is typically greater, and increases with catchment size. Variability within source groups is also likely to be greater, and direct measurement of eroded source material as delivered to suspension in streams is not possible, unlike for water sources. As the effects of selective particle mobilisation on fingerprint parameters are extremely difficult to quantify, measurements taken from sample points are likely to be a rough estimate, and some qualitative assessment of the relative likelihood of flow connectivity is required to limit the dataset to realistic sources. In larger catchments correct and complete identification of potential sources and minimisation of within-group variability becomes highly complex, for example the requirement for detailed accurate mapping of changes in soil type to avoid over-generalisation when using mean estimates.

The use of a pre-screening test to eliminate parameters which show no difference between groups is necessary to ensure that later steps in the modelling technique do not include parameters where the group differences are unreliable. However, the use of a simple univariate test (Kruskall-Wallis) to include or preclude parameters may be problematic. The risk of Type 1 error increases with the number of parameters included, and where pre-screening is performed without prediction or explanation of group differences the use of very large numbers of parameter sets leads to a high probability of false positives. Conversely correction such as the Bonferroni method may lead to rejection of large numbers of parameters through over stringent entry requirements. It is suggested here that where possible group separation parameters be pre-selected based on prediction, rather than a simple 'data trawling' exercise. If differences between source groups for individual parameters cannot be explained, their inclusion in a fingerprint dataset becomes problematic. For example both the present investigation and a previous study (Russell *et al*., 2001) have indicated that tile drain sediment is easily distinguished from soil and sub-soil (channel bank) material. This difference cannot simply be attributed to differences derived from a mixing of the surface and subsurface soil; tile drain sediments were found to be enriched in both calcium and phosphorus relative to both these two groups in this study. It is possible that changes are as a result of enrichment by adsorption from drainage water passing through sediment, yet this calls into question the conservatism of the tracers used. Conversely, high calcium levels in track sediment in this study can be clearly attributed to the use of limestone as track ballast on the farm, and therefore the high concentrations of calcium in this source group relative to the others identified can be adequately explained.

While the technique is useful, it is as a semi-quantitative technique in conjunction with robust direct measurements and field observation, rather than as a fully objective accurate estimate of source contributions. In this study it points to the high contribution of tile drains, a more limited contribution from a pollution hotspot, and a relatively low contribution from soil erosion and channel banks. These estimates can be at least partly supported by direct measurement, and sound reasoning. The fact that tile drains represent an important if invisible source would appear very important for choice of mitigation strategy, and while mitigation of the pollution hotspot track runoff) may be entirely justified in terms of control of other pollutants (e.g. nutrients and faecal pathogens) the effects on absolute sediment yields may be limited and largely cosmetic.

The need for a 'Whole-farm' assessment

Caution is required in the use of farm catchment water quality data to identify dominant sediment loss processes for this farm in particular, which is likely to be equally important elsewhere. The study catchment area (62 ha) represents less than a third of the total farm catchment area. While it includes the major pollution 'hotspot' associated with the dirty dairy track discharge, it does not include any of the major field points of field runoff, which in interaction with the track appear to represent a major sediment source during large runoff events at sensitive cropping stages. What is clear from visual observation is that the instrumented drainage ditch catchment is not the only route for suspended sediment loss from the farm. The interaction of field run-on and run-off and transport via farm tracks and ultimately drainage ditches outside the study catchment appear to represent a major sediment loss issue, and therefore the low estimates of surface soil erosion contribution for the experimental catchment appear to be a significant underestimate for the farm as a whole.

This illustrates a limitation of low resolution risk assessment techniques applied to agricultural areas; available slope and topographic information would indicate a limited soil erosion risk on these field areas even given a susceptible land use such as late sown winter cereals (Defra, 2005). The soil type would suggest the presence of artificial drainage, which may then be interpreted as the major sediment source pathway due to the much higher delivery ratio (connectivity) between drains and surface waters than the surface pathway based on a generalisation of dominant connectivity at a landscape scale (1 km^2 grid unit). This approach has been recently adopted in UK catchment scale assessment of sediment loss risk (Stroemqvist *et al*., 2008; Collins and Anthony, 2008). However in reality smaller scale landscape features may result in high localised erosion risks, which may be significant or dominant aquatic sediment sources at a farm scale. The erosion risk associated with concentrated runoff is hard to detect remotely, or even from survey in dry weather – this kind of issue will often rely on the knowledge of the land manager (and their willingness to report sediment loss issues). Mitigation design will be greatly affected by the ability to identify such problems – in-ditch sediment traps for example would not be effective in this instance as the route to receiving waters is via surface runoff along farm tracks. However, simple engineering structures, such as a small gateway berm to prevent field run-on are likely to prove low cost and highly effective.

2.5.4 Sediment loss mitigation

As demonstrated below, understanding of the importance of infrequent events of medium to high discharge/rainfall magnitude may have significant implications for choice of mitigation strategy. This study trialled a small sediment trap to mitigate loss of sediment and associated particulate phosphorus. The trapping results show variable effectiveness, but clearly demonstrate that the trap is of limited effectiveness in larger events when residence time is extremely low, when it may even serve as a sediment source through remobilisation of trapped material. The practicality of this mitigation strategy will depend on:

a) Effective catchment area.
b) Cumulative sediment/discharge relationships.
c) The target particle size removal.

Figure 2.30 demonstrates that while a trap of modest size may effectively remove sand (>63 µm) particles for the study catchment area (62 ha), removal of fine silt particles would require a very large construction in order to achieve effective removal in larger events, which account for the majority of sediment export. While sand sized particles may represent a significant proportion of transported sediment, the majority of nutrient losses and siltation degradation of river beds is associated with fine (silt and clay) particles (Kemp *et al.*, 2001). It is probable that a large proportion of the fine particle load will move as part of larger aggregates, meaning sedimentation of a significantly finer particles may be possible than primary particle diameter would indicate; particle size analysis of sediment recovered from the sediment trap showed 48% was in the clay and silt fraction (Table 2.4). However, analysis of material recovered from suspension (ditch samples) indicates that actual export proportions of these fines are considerably higher meaning much of this material was not effectively removed by sedimentation. Fine sediment removal (*circa* 75%) for this case study would appear possible given a trap storage capacity an order of magnitude larger than that installed, using the calculation method in Figure 2.30. Such a scheme would not involve prohibitive costs or land take. However, should catchment area be increased by a further order of magnitude (*circa* 6 $km^{2)}$ the necessary storage capacity would appear to be at the limits of what is practical except where such schemes are justified on other grounds (e.g. flood mitigation). For this reason the use of sediment traps for mitigation appears best limited to small farm catchments less than 1 km^2.

The calculations made in Figure 2.30 are made assuming steady state flow. This is appropriate for the available data given the resolution (1 hour flow intervals); particles with a settling time of less than one hour can be assumed to be unaffected by changes in discharge. However, for smaller particles with a longer settling time, the effect of recession and antecedent hydrograph peaks need to be considered. Figure 2.32 illustrates this effect on required storage capacities to ensure sufficient residence time. For a particle of 2 µm diameter the settling time is greater than the mean interval between hydrograph peaks, meaning high storage capacity is needed, to remove these particles in low as well as high flows (to prevent shortening of residence times by secondary flow peaks). For particles of 10 µm or greater this effect is minimal. Figure 2.31 confirms that the flow exceedance curve for the six month monitoring period is representative of the longer term discharge record for the site.

Available criteria for US sediment basin design for construction site best management practices recommend a dry weather storage volume of 67 cubic yards per acre (126 m^3 per hectare) plus the same volume of storm water storage (California Stormwater Quality Association, 2003). This is equivalent to the volume of water generated by one inch of runoff from 1 acre. It should be noted that these design criteria are to prevent sediment damage by large scale erosion. Although events of this magnitude are unusual for UK conditions, this does indicate that the maximum drainage area for sediment basins is limited by practical scale of construction; designing to these criteria for example would require a storage volume of 7840 m^3 for the study catchment - a circular pond of diameter approximately 70 m diameter if 1 m deep (1.2% of the catchment area). Ineffectiveness against large events may result firstly in a large proportion of the average annual load bypassing the mitigation feature during high flows, and secondly the possibility of remobilisation and export of sediment trapped in previous events. In the absence of detailed climate and flow records the use of an event threshold such as the US example would seem appropriate. For this case study a storage volume approximately equal to 10 mm of specific runoff would appear sufficient to allow trapping of 75% sediment particles greater than 10 μm diameter.

The suitability of sediment traps as mitigation methods is partly dependent on the nature of sediment problems and mitigation targets; where reduction in *total* sediment exports is the target it is clear that trap effectiveness at peak flows (i.e. large capacity construction) is necessary for major sediment reductions (Figures 2.30, 2.31). However, if a reduction in high sediment concentration *duration* is the target, a smaller trap may be effective; for example a trap of 40 m^3 storage capacity would appear sufficient to remove all particles to 10 μm in 75% of duration in this study (Figure 3.30) provided remobilisation of trapped sediment is prevented (by design or regular removal).

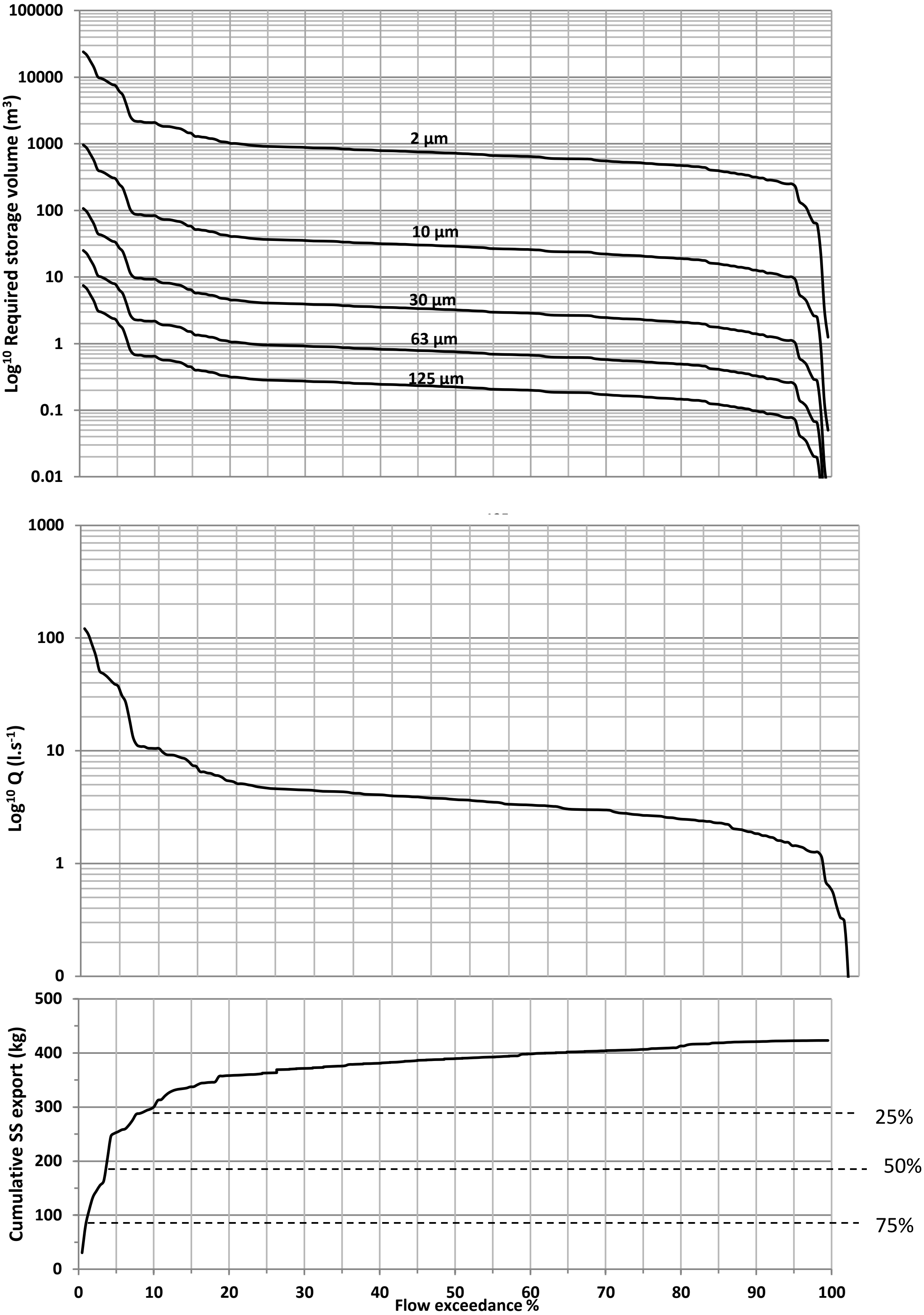

Figure 2.30: Steady state flow storage volume calculations based on required residence time for different sized sediment particles. Settling times based on Stoke's law for 0.5 m at water temperature of 10°C.

$$v_s = \frac{2}{9}\frac{(\rho_p - \rho_f)}{\mu} g\, R^2$$

where: v_s is the particles' settling velocity (m/s) (vertically downwards if $\rho_p > \rho_f$, upwards if $\rho_p < \rho_f$); g is the gravitational acceleration (m/s^2); ρ_p is the mass density of the particles (kg/m^3), and ρ_f is the mass density of the fluid (kg/m^3).

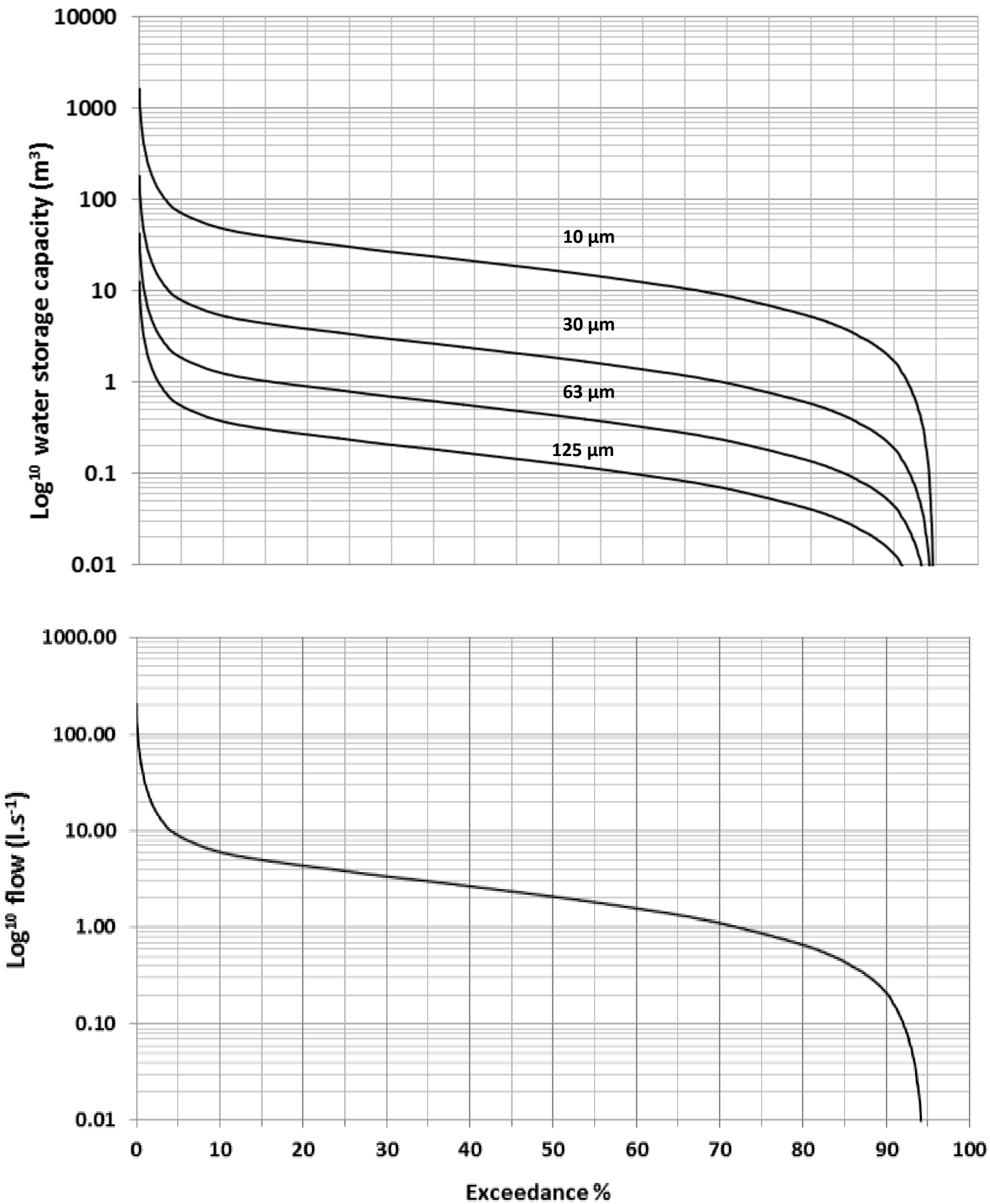

Figure 2.31: Storage volume calculations for steady state flow for full flow record (2007-2009)

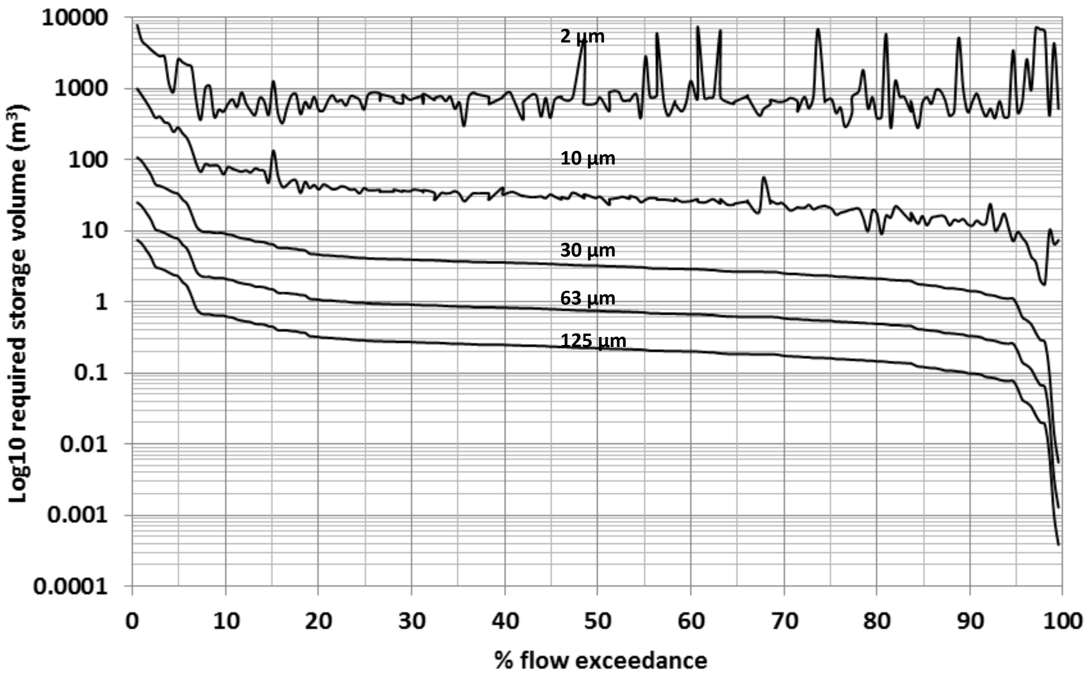

Figure 2.32: Required storage volume based on variable flow

Calculated based on the residence time of water within a trap of constant volume, but variable inflow

3.6 Summary and Conclusions

Sediment losses were dominated by high flow events, with minimal inputs under dry weather conditions This chapter has highlighted that at a farm scale, dominant sediment loss processes may be both generalised (such as losses through tile drains) or localised such as the occurrence of concentrated runoff pathways in resulting from topography and or man-made features. The latter are difficult to identify without detailed local knowledge. The evidence presented in by source fingerprinting suggests tile drains to be a dominant sediment loss pathway from this type of land under susceptible arable uses.

The source-pathway for sediment in agricultural catchments is critical to the identification of appropriate mitigation strategies. While ditch sediment trapping has been demonstrated to be a technique which may successfully mitigate tile drain losses and point discharges, this technique is likely to be less appropriate where significant problems are associated with concentrated surface runoff pathways. The importance of different source-pathways and the relative efficacy of mitigation techniques are examined in later chapters.

Chapter 3: Plot scale sediment loss processes

3.1 Introduction

Sediment losses through artificial sub-surface drains can represent a significant pathway where such drainage systems exist. However, the findings of the initial farm-scale study (Chapter 2) fail to quantify the scale of potential losses or verify the temporal pattern. These are critical issues when considering the importance of drains as a sediment source/pathway. Previous studies (e.g. Kladivko, 2001; Chapman *et al*., 2003) have suggested that the presence of drains may serve to increase sediment losses per unit area relative to undrained land, by increasing the sediment delivery ratio from field surface to watercourse. These findings have also contributed to the formulation of the PSYCHIC model (Davison *et al*. 2008) for estimating catchment sediment and phosphorus export. Applications of this model have identified the presence of agricultural drainage systems as the key risk factor in several catchments (Collins *et al*., 2007; Strőmqvist *et al*., 2008). The implications of such findings are significant for mitigation strategies. For example, widely promoted riparian buffer strips, in-field grassed areas and contour ploughing are measures which may reduce surface runoff, but have very limited impacts on drainage losses. Attempts to reduce sediment losses from under-drainage systems, unlike soil/sediment loss mitigation for surface runoff, are rarely reported in the literature. If the studies described above (together with the evidence in the previous chapter of this study) are accurate, it is necessary to include drain sediment mitigation in agricultural catchment sediment control strategies.

This chapter reports the findings of a study conducted over the winter (Nov-Apr) of 2009-2010. The study utilised a long standing large-scale replicate drainage plot trial restored in 2009 for use in a pesticide study. The main objectives of this study were primarily to quantify the magnitude and temporal pattern of drain sediment losses, with the secondary objective of evaluating the potential of mitigation strategies to reduce suspended sediment exports.

3.2 Site description and methods

3.2.1 Site description

The trial is located at Cockle Park experimental farm near Morpeth, Northumberland (NZ 192 915) and operated by Newcastle University. The drainage trial was originally created in 1978 by the Field Drainage Experimental Unit and consists of nine 0.25 ha (100*25 m) replicate plots. An interceptor drain surrounds the trial at a depth of 1.3 m with permeable gravel back fill to the soil surface. Each plot is isolated by a polythene barrier to a depth of 1 m.

Soils of the site are uniformly fine loamy over clay *Dunkeswick Series*, Typical stagnogley soils (Avery, 1980 – see Table 3.1.) formed in Devensian till derived from carboniferous rocks (principally shales). These soils are the most commonly occurring on the coastal plain of Northumberland where the site is located. The most important hydrological property of the soil is the transition between the relatively permeable top soil, and the poorly structured, slowly permeable sub soil. Previous measurements at the site (Armstrong and Davies, 1984) have indicated an abrupt change in hydraulic conductivity from in excess of 100 mm per day in the top soil to less than 2 mm at 40 cm depth (Figure 3.1). Below 60 cm hydraulic conductivity falls below 1 mm per day (Figure 3.1) and for practical purposes the sub soil is impermeable below this depth. Under natural drainage these soil stand wet to the surface for long periods which restricts machinery access and precludes the growth of spring sown crops in most years and safe machinery access for autumn sown crops in wet years (Jarvis *et al*., 1984). Artificial drainage is required to increase productivity of these soils. Due to the low permeability of the sub soil, secondary sub-soiling has been shown to be necessary to effectively lower perched water tables by facilitating the lateral movement of water to tile drains (Armstrong, 1983).

Depth (cm)	Description
0-25	Very dark greyish brown, slightly stony clay loam. Medium sub-angular blocky structure.
25-45	Greyish brown very slightly stony clay loam with common ochreous and grey mottles. Medium angular blocky structure.
45-70	Brown very slightly stony clay with common dark grey and ochreous mottles and dark grey ped faces. Coarse prismatic structure.
70-100+	As above, but coarse prismatic to massive structure.

Table 3.1: Soil profile description (source: author survey)

Size fraction	**μm**	**%**
Coarse sand	2000-600	1.7
Medium sand	600-212	5.7
Fine sand	212-106	10.7
Very fine sand	106-63	8.5
Coarse silt	63-20	13.3
Fine silt	20-2	28.1
Clay	<2	35.2

Table 3.2: Soil particle size analysis at 60 cm.
(after Armstrong and Davies, 1984)

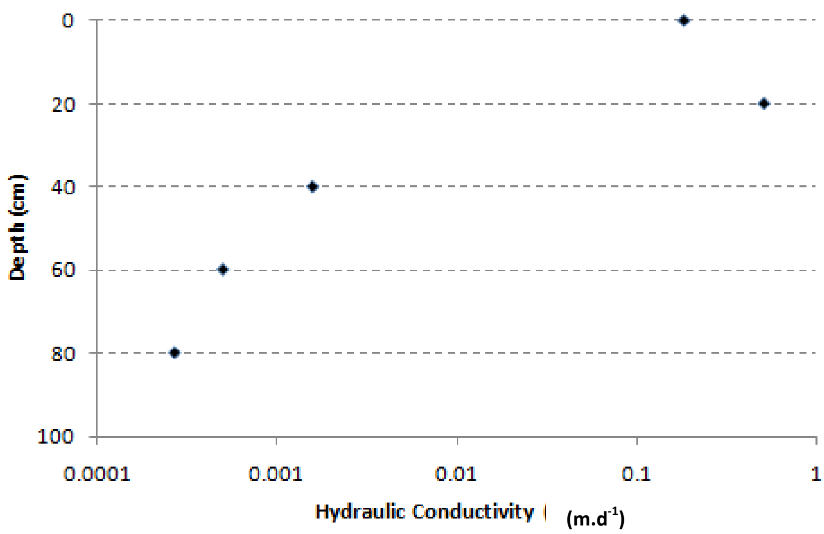

Figure 3.1: Soil hydraulic conductivity depth profile (after Armstrong and Davies, 1984).

3.2.2 Trial design and Methods

The original experimental design consisted of three drainage treatments each with three replicates. The plots were modified in 1996 to provide nine replicate plots of the 'best practice' drainage treatment to act as controlled lysimeter plots for future studies. This drainage treatment consists of wide spaced (40 m) cross-slope collector drains at 0.9 m depth, formed from 60 mm corrugated slotted plastic pipes backfilled with gravel to 0.3 m from the land surface. A secondary treatment of 2 m spaced mole drains are drawn along the slope at a depth of 0.5 m. The collector drains are connected to a sealed drain which leads to a monitoring outfall. Overland flow is monitored via a 0.3 m deep backfilled interception trench at the lower end of each plot, which is connected to a separate outfall via a sealed plastic pipe.

Monitoring equipment was reinstalled in August 2009, consisting of purpose built ¼ 90° V-notch box weirs on both drain and surface water outlets from each plot (Photo 3.1). Flow depth over the V plate was continuously monitored telemetrically via a pressure transducer. The pressure transducer was linked to an automatic pump sampler configured to take a 750 ml sample at flow segmented intervals. Surface water collectors were linked to similar equipment, with bulk samples collected for each storm in a single 5 l sampling vessel. Precipitation data was collected by an automated weather station installed on site.

Photo 3.1: Aerial photo of plots (image 2007)

Photo 3.2: Clockwise from top left: flow monitoring box weirs; water samplers; automated weather station with plots in background.

Cultivation treatments

Plots were sown with winter Oil Seed Rape Variety *Catana* with a Thousand Seed Weight of 5.88g and seed dressing of Chinook® and Thiraflo® on the 12th September 2009. All plots had been sown to spring barley in 2009, having previously been in long term grassland.

The trial tested the effect of three cultivation methods involving progressively lower soil disturbance:

T1 A 'standard practice' cultivation system (mouldboard plough). Following ploughing to approximately 30 cm the plots were power harrowed, then simultaneously rolled and drilled using a *Horsch* combi drill.

T2 An aggressive minimum tillage treatment involving a single pass cultivation system (*Sumo Trio* ®) was applied which encompassed discing, sub-soiling (to 30 cm) and packing. The plots were then rolled (*Cambridge* roller) and drilled separately using a *Vaderstat* drill.

T3 A 'conservative' minimum tillage system involving working to shallow depth (10-15 cm) using a stubble cultivator. The plots were then rolled (*Cambridge* roller) and drilled separately using a *Vaderstat* drill.

Secondary treatments were applied to assess the impact of plot-end riparian buffer strips. These treatments apply to surface runoff only. The buffer strip treatments consisted of the following:

1. 6 m ryegrass.
2. 6 m uncultivated bare fallow.
3. No buffer (plots cultivated to end).

Plot	Cultivation Treatment	Buffer strip (upslope of surface runoff collector)
1	T1 - Plough, power harrow	6m Grass
2	T2 – Min Till , Sumo system	No Buffer
3	T3 – Min Till – Stubble cultivator	6m Bare
4	T2 – Min Till , Sumo system	6m Bare
5	T3 – Min Till – Stubble cultivator	6m Grass
6	T1 - Plough, power harrow	No Buffer
7	T3 – Min Till - Stubble cultivator	No Buffer
8	T1 - Plough, power harrow	6m Bare
9	T2 – Min Till , Sumo system	6m Grass

Table 3.3: Site plan Cockle Park drainage trial.

Cultivation treatments were replicated three times in a blocked design. Buffer strip treatments were randomly assigned to cultivation treatments in each block (each treatment combination represented once in the trial).

Samples were collected from the field at intervals of not more than one week, and analysed for suspended sediment concentration. Suspended sediment concentration was quantified by oven drying (105°C) the filtrate (0.45µm quantitative glass fibre filter paper) from a 150 ml sub-sample. A total of 513 drain water samples were collected and analysed (mean 57 per plot). Bulk surface runoff samples (1 sample per plot per runoff event) were collected during large events. Significant surface flow did not occur outside of these events. Samples were analysed for suspended sediment concentration as above. A total of 32 surface flow storm samples were collected and analysed.

Linear regression modelling was used to predict sediment yield using the continuous flow record and the discharge-sediment concentration relationship. Differences in runoff and surface sediment yield were compared using a blocked General Linear Mode, using Tukey's test to compare individual treatments. Differences in estimated sediment yield between plough and cultivator treatments were compared using a two sample t-test (the reasoning for exclusion of the sumo treatment is detailed in the results section).

3.3 Results

3.3.1 Weather and hydrology

Total rainfall over the monitoring period was close to the long term average for the site (Table 3.4). However, notable deviations from long term monthly averages were observed in November, where a number of large rain storms led to an unusually high rainfall total (Figure 3.2), and April when unusually dry conditions were observed (contributing to a rapid cessation of drain flow at the start of the growing period). Further significant weather events included a prolonged winter cold spell beginning in late December and lasting until late February, during which 78.2 mm fell as snow between Jan 10th and Feb 15th, and a severe weather event over 29th–31st March during which 31.6 mm of rainfall were recorded. The unusually long period of cold weather resulted in a long periods of frozen ground when drain discharge was negligible or zero between late December and February (Figures 3.4 - 3.9). Snow melt and ground conditions (frozen soil and antecedent wetness following thaw) also contributed to drain and runoff response during events over this period, particularly that occurring in Late February (26^{th}-27^{th}).

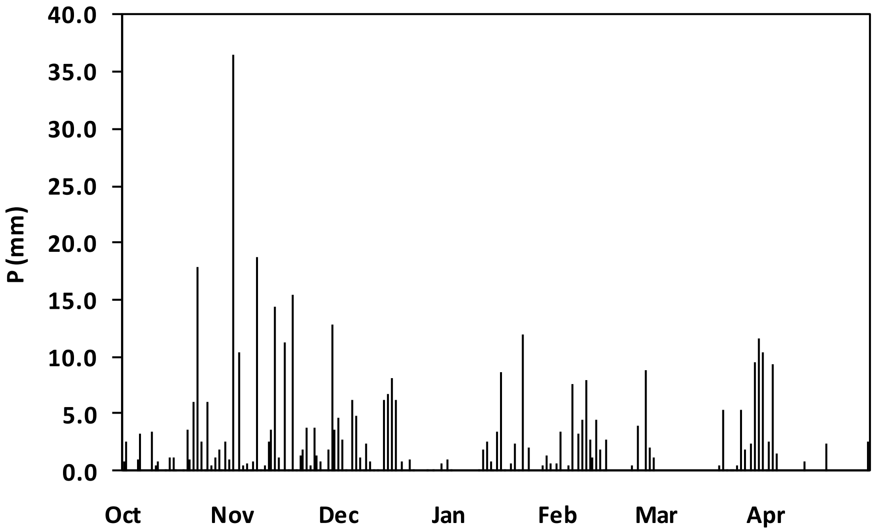

Figure 3.2: Daily precipitation Cockle Park Oct 2009 – Apr 2010.

Month	observed	long term mean (1951-1980)
Oct	59.5	57
Nov	148.6	72
Dec	54	61
Jan	38.6	67
Feb	57.2	53
Mar	48	52
Apr	19.1	43
Total	425.0	405.0

Table 3.4: Cockle Park observed and average monthly precipitation (mm).

Cultivation treatment	Runoff	Block			Mean	SE
		I	II	III		
Mouldboard plough (T1)	Surface	11.6	37.5	33.0	27.4	8.0
	Drain	163.0	121.1	145.6	143.2	12.1
	Total	174.6	158.7	178.6	170.6	20.1
Sumo Trio (T2)	Surface	24.5	4.7	6.2	11.8	6.4
	Drain	247.1	220.6	85.4	184.4	50.1
	Total	271.7	225.2	91.6	196.2	56.5
Stubble cultivator (T3)	Surface	35.0	10.9	12.1	19.3	7.8
	Drain	119.4	122.3	159.4	133.7	12.9
	Total	154.4	133.2	171.5	153.0	20.7

Table 3.5: Mean plot specific runoff (mm) Nov 2009 – Apr 2010.

The surface flow record for the Sumo Trio treatment in Block III (plot 9) indicates error resulting either from failure either of discharge monitoring equipment or the drainage system (Table 3.5; Figures 3.3, 3.6, 3.7). Plot 9 is therefore excluded from further analysis and discussion. Total discharge figures for plough and cultivator treatments indicate good replication (Table 3.5). Total drainage from the Sumo treatments (excluding Plot 9) is significantly in excess of that generated by other treatments ($F = 14.37$; $p = 0.008$) which is attributable to the differences in drain discharge only ($F = 15.44$; $p = 0.007$). Surface water flow accounts for a variable but minor proportion of total discharge (2 – 24%; mean = 12%), and is limited to larger runoff events (Figures 3.4 – 3.9) when the proportion can be much higher; as much as 48% of total discharge for plot 6 (plough treatment II) on Feb 26th (Figure 3.5). Treatments show acceptable replication in total discharges, but large differences can be observed between the proportions of drain and surface flow. A variable increase in surface flow proportion can be observed from the beginning of the monitoring period (Figure 3.3) compared with later events (Figures 3.4-3.9) indicating heterogeneous decline in drain connectivity. This decline results in significant surface runoff generation in later events. However there is no difference in surface flow which can be attributed to cultivation treatment ($F = 0.56$; $p = 0.604$).

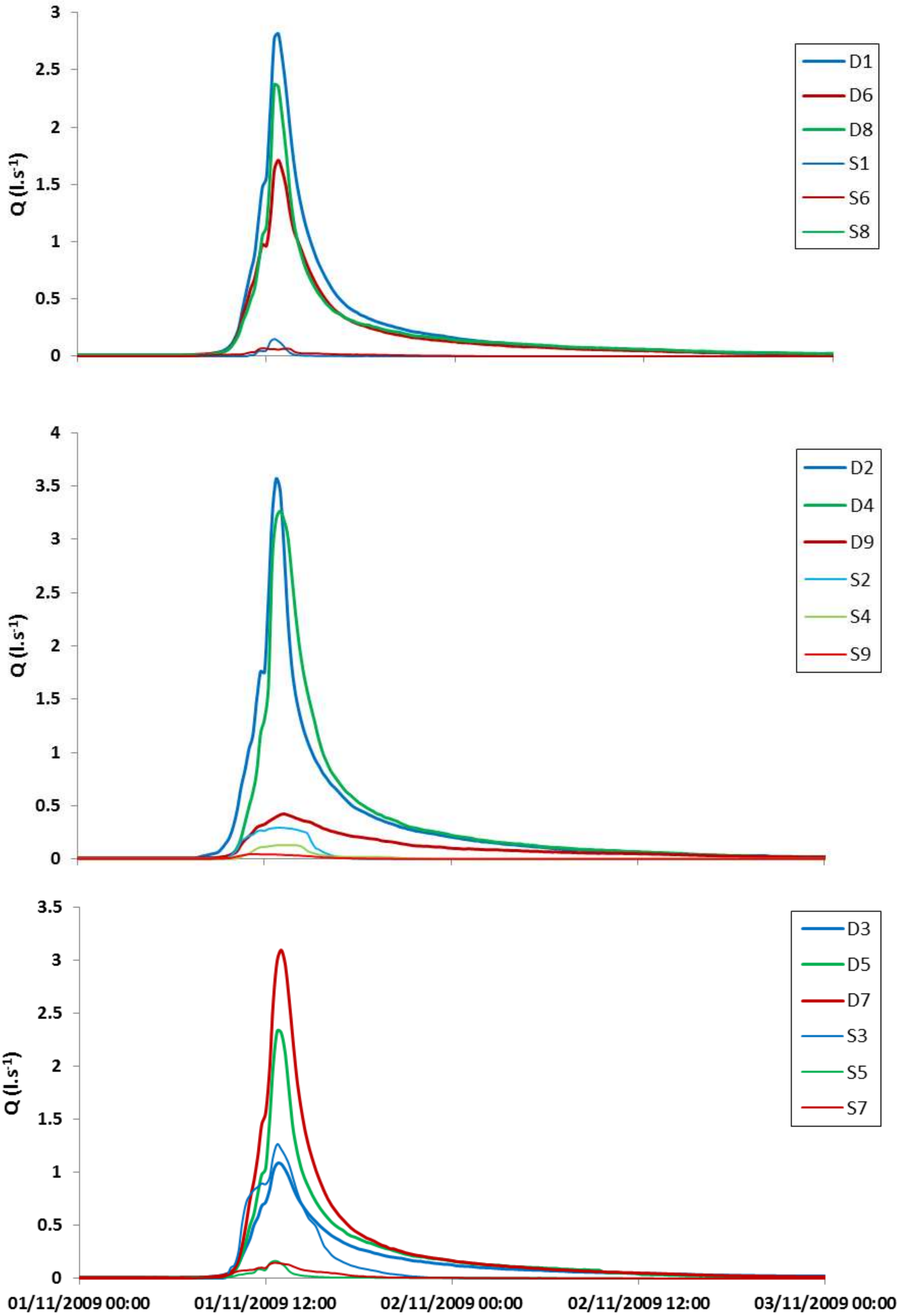

Figure 3.3: Treatment replicate flow record for storm event 01/11/2009 from top: Plough (T1), Sumo (T2), Cultivator (T3).

D(1) = drain flow (plot number); S(1) = surface flow (plot number) Note D9 is suspected drain flow logger error.

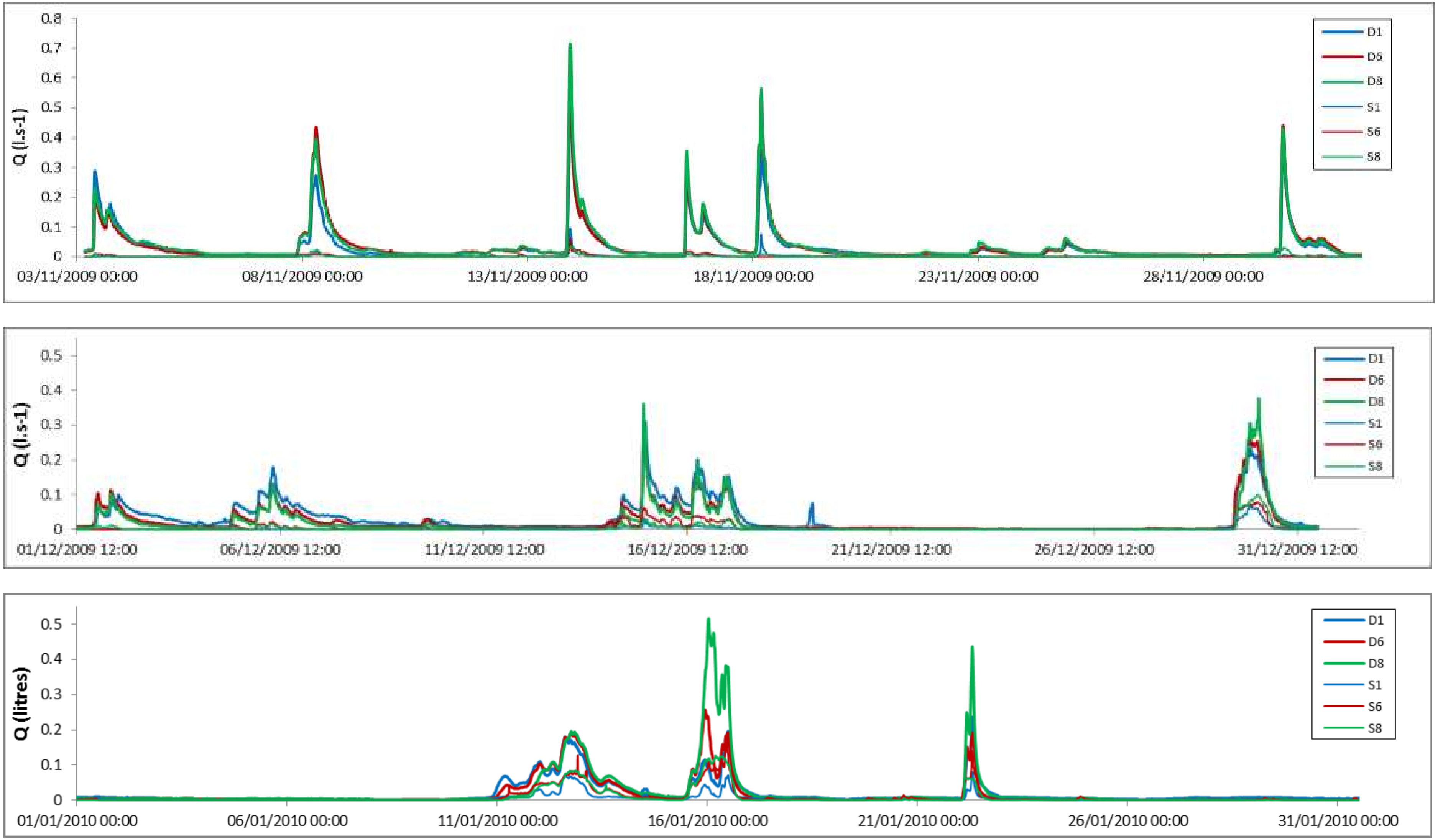

Figure 3.4: Plough treatment (T1) flow record Nov 2009 – Jan 2010.

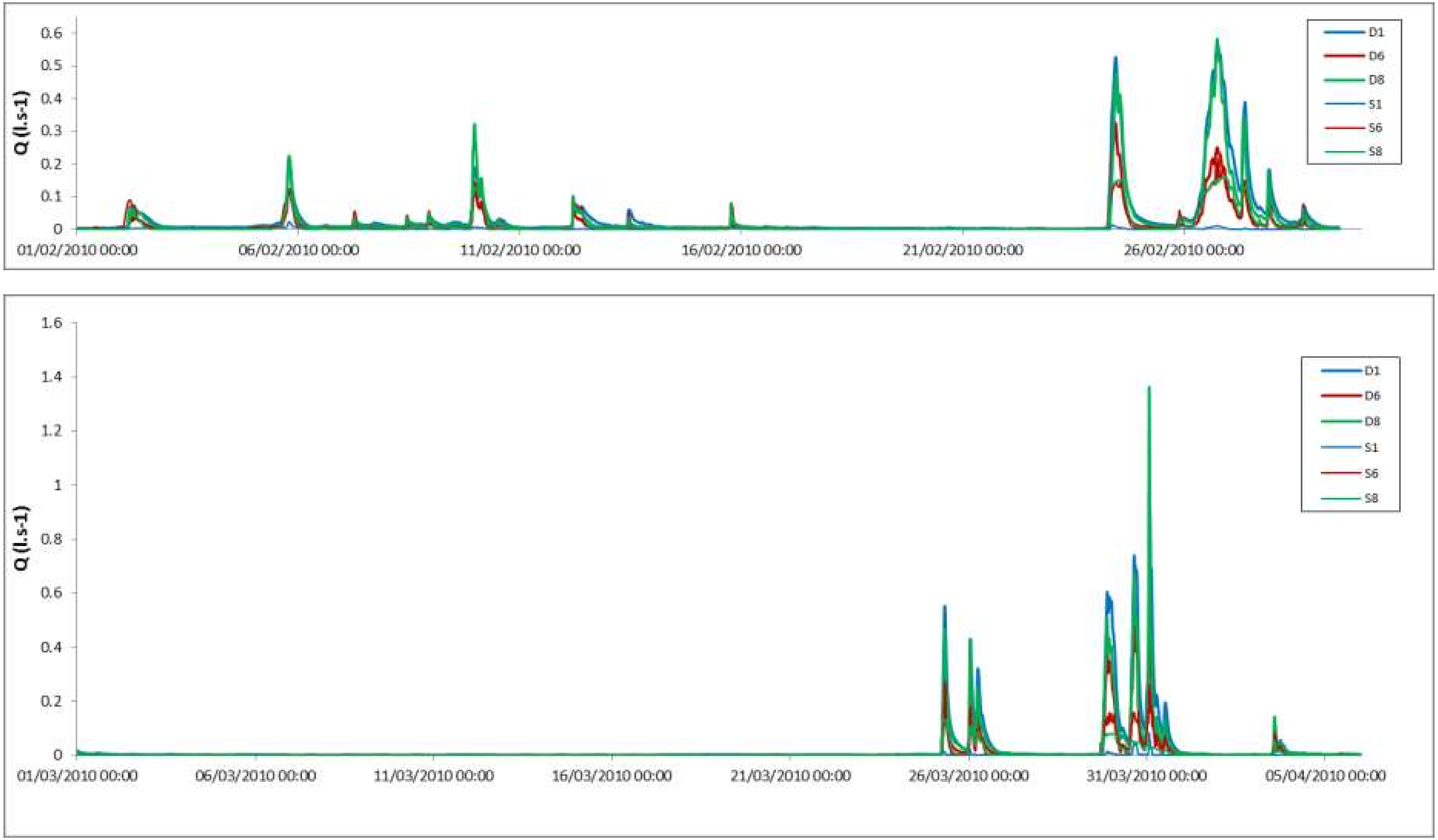

Figure 3.5: Plough treatment flow record Feb – Apr 2010.

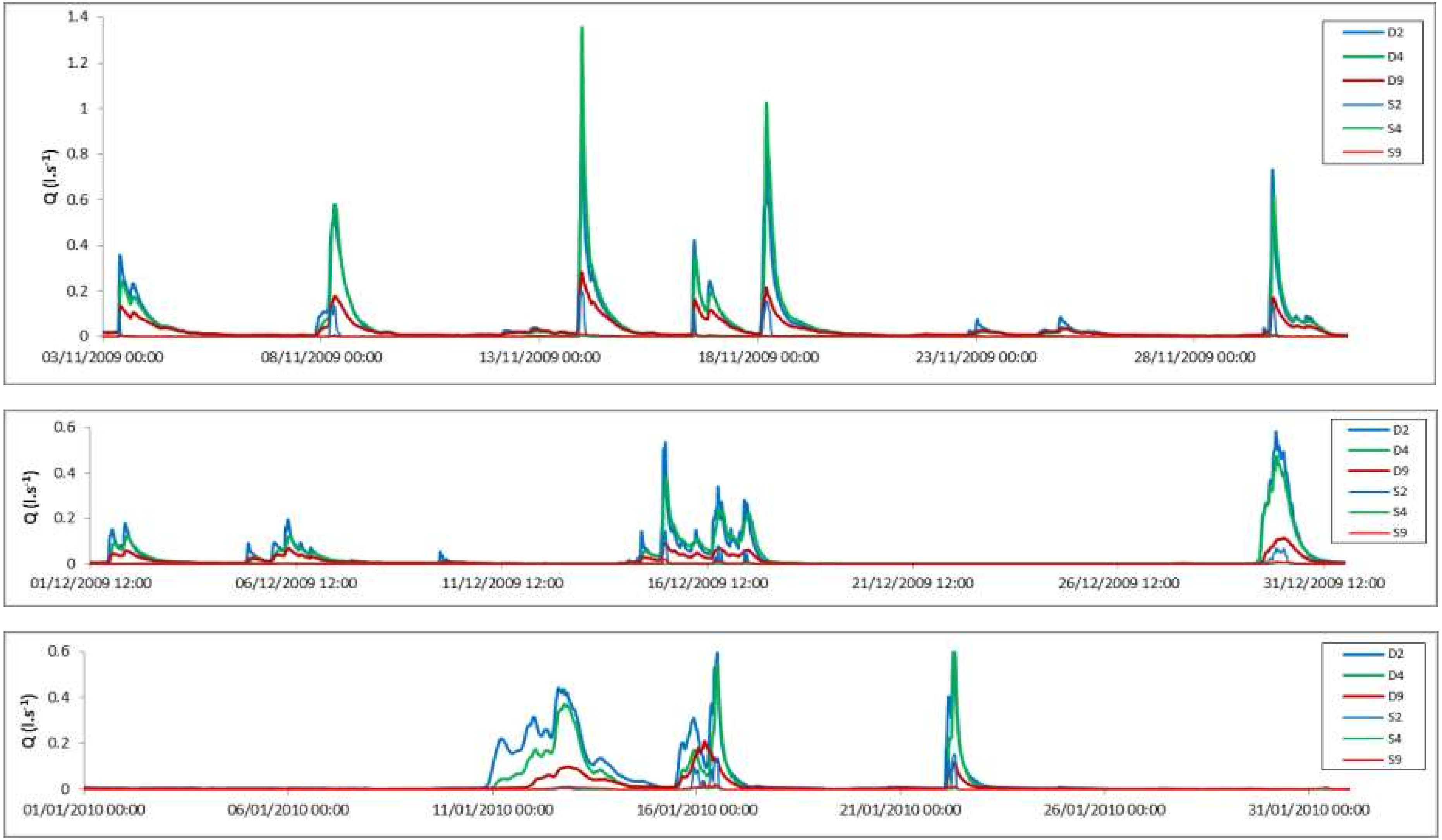

Figure 3.6: Sumo treatment (T2) flow record Nov 2009 – Jan 2010. Note D9 is suspected drain flow logger error.

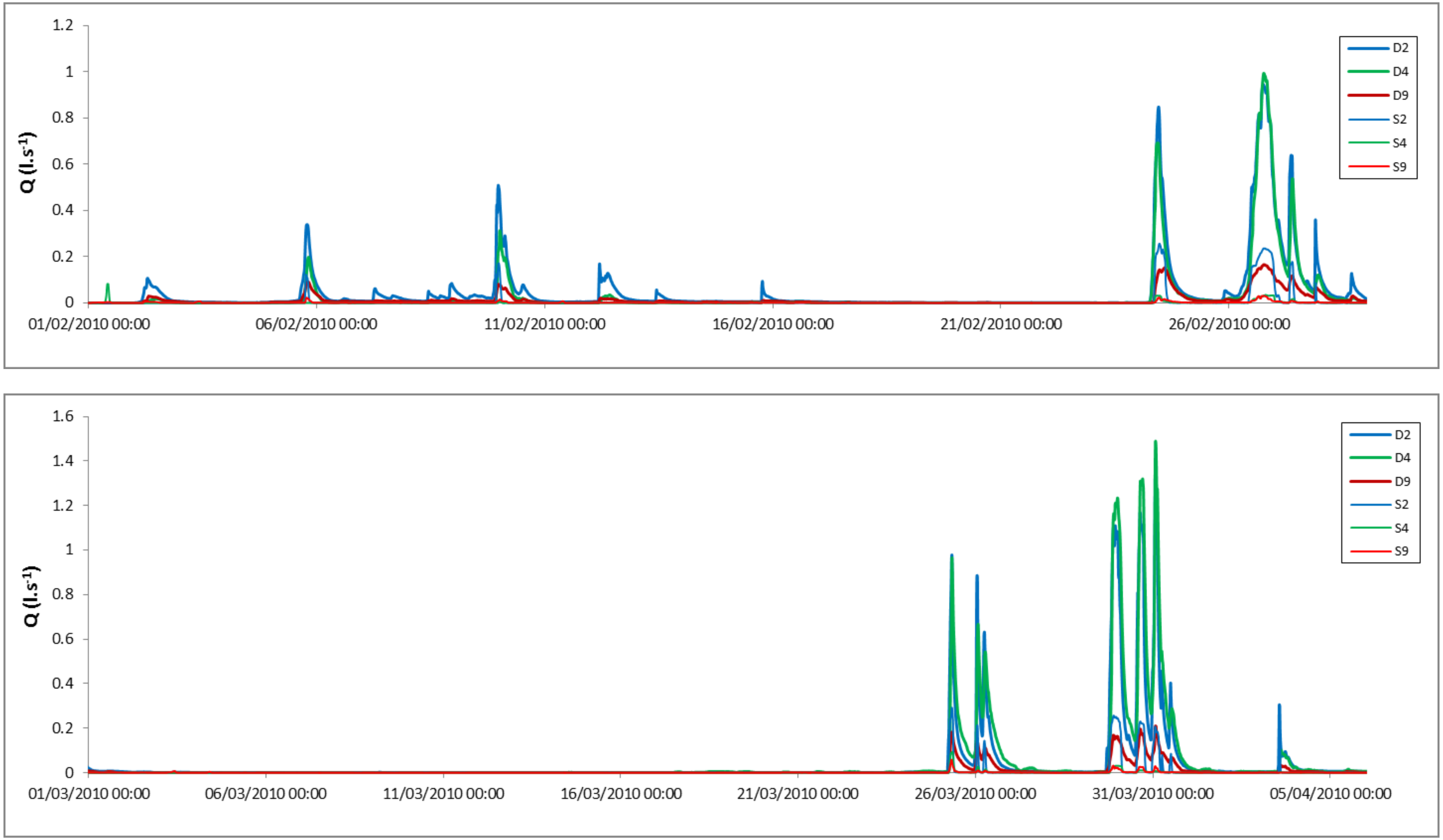

Figure 3.7: Sumo treatment flow record Feb – Apr 2010.

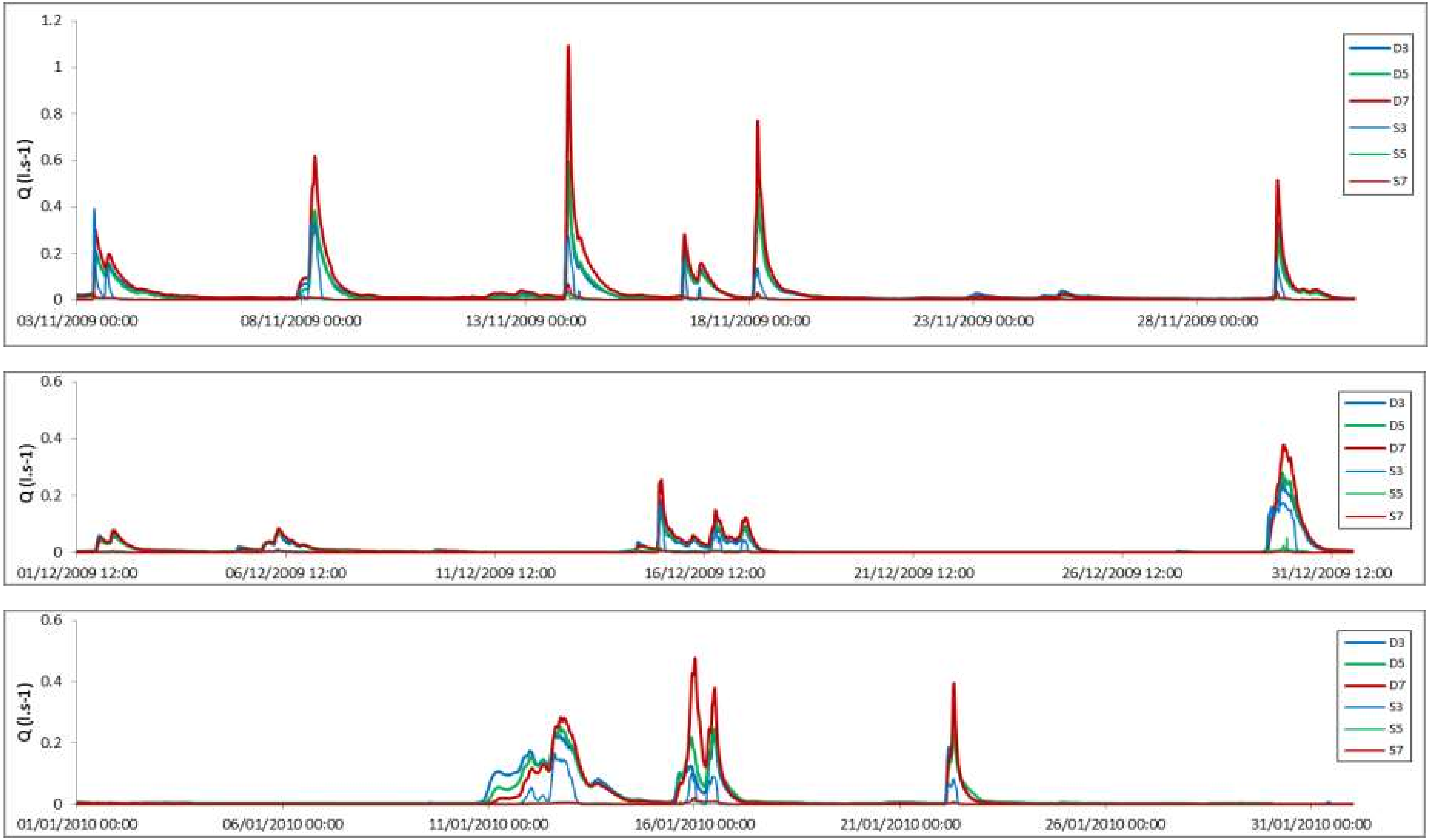

Figure 3.8: Cultivator treatment (T3) flow record Nov 2009 – Jan 2010.

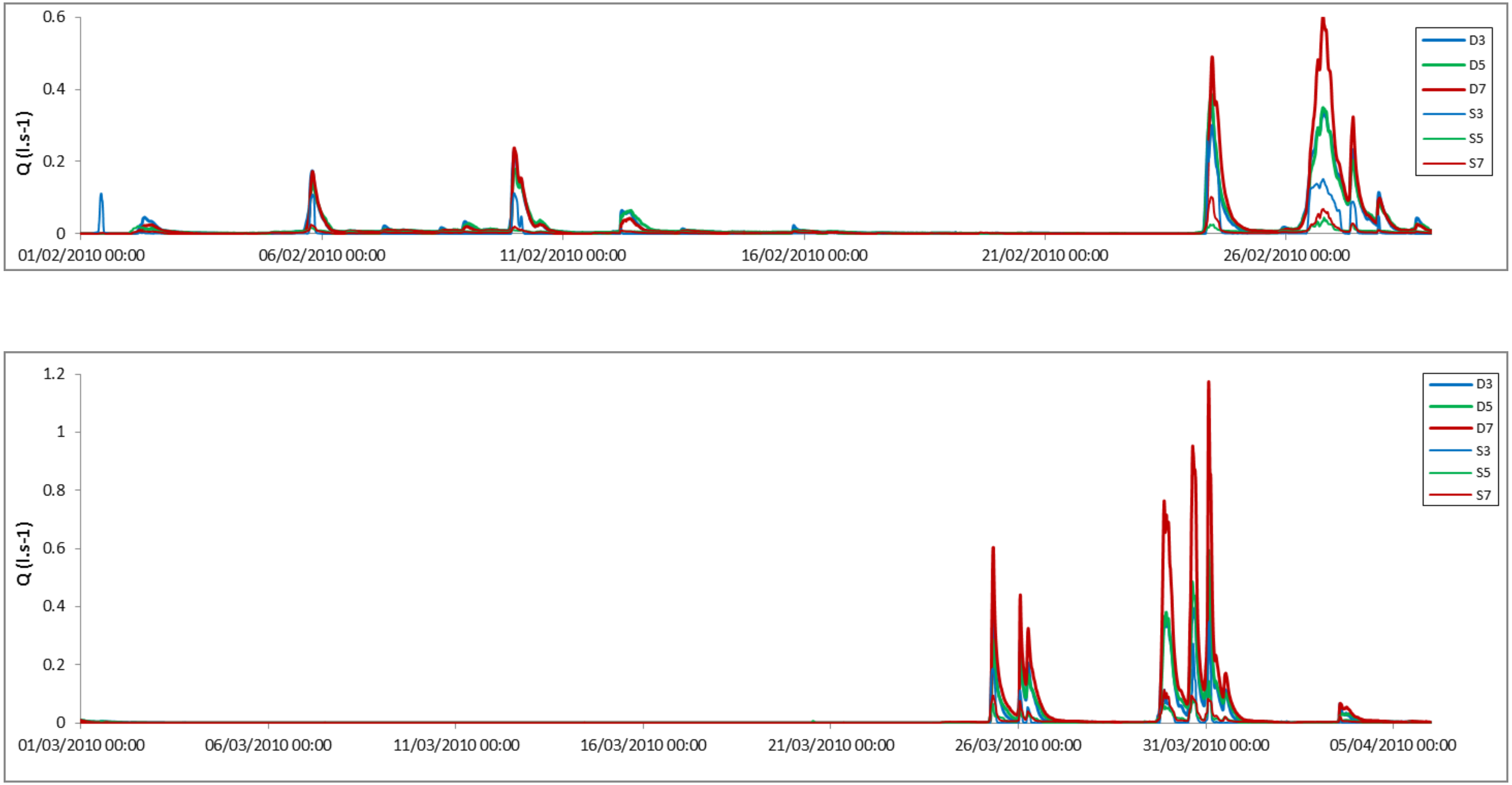

Figure 3.9: Cultivator treatment flow record Feb-Apr 2010.

3.3.2 Calculation of sediment yield

A significant linear relationship between drain discharge and sediment concentration was found for all plots, with the exception of Sumo treatment block III (Figure 3.10; Table 3.6). Errors associated with flow recordings for this plot (9) have been described previously. Both the coefficient and the strength of the relationship varied considerably between plots. Sediment mobilisation per unit runoff appears considerably lower for the zero tillage treatments than for the ploughed plots. However, considerable uncertainty exists within the relationship for some plots.

	Block		
Treatment	**I**	**II**	**III**
Plough	0.746***	0.693***	0.634***
Sumo	0.627***	0.394**	-
Cultivator	0.596***	0.869***	0.809***

Table 3.6: Pearson's correlation coefficients drain Q Vs. sediment concentration.
** $p<0.01$ *** $p<0.001$

A large part of the variability between plots is explained by propensity to surface runoff generation; higher overland flow during storm events appears to increase the sediment concentration in the drain when plots are compared. This effect is clearly demonstrated by comparison of the sumo treatments for block I and II (Figure 3.10); total specific surface runoff for these plots is 24.5 and 4.7 mm respectively.

The effect of surface runoff volume on drain sediment concentrations is clearly illustrated for an event which occurred on February 26th-27th, (the most complete event suspended sediment record). This event occurred in response to snow melt triggered by relatively light rain. The sediment response varied between plots, with no clear treatment effects; peak plot sediment concentrations were not correlated with drain discharge (Pearson's R = -0.219; p = 0.602). However, a strong correlation existed between peak *surface runoff* discharges and *drain* sediment concentrations; the majority of the variation between plots can be explained by the propensity to surface runoff, although this runoff variability is *not* attributable to cultivation treatment (Figure 3.11). Over half of the variability in sediment concentrations recorded during this event can be attributed to variable surface runoff generation (Figure 3.12). Little correlation between sediment concentration and runoff discharge can be seen at low flows (*c.* <0.03 $l.s^{-1}$). The use of *both* drain and surface discharge in the prediction of drain runoff sediment concentrations provides a reliable estimate for this event, indicating some independence between these factors (Figure 3.13).

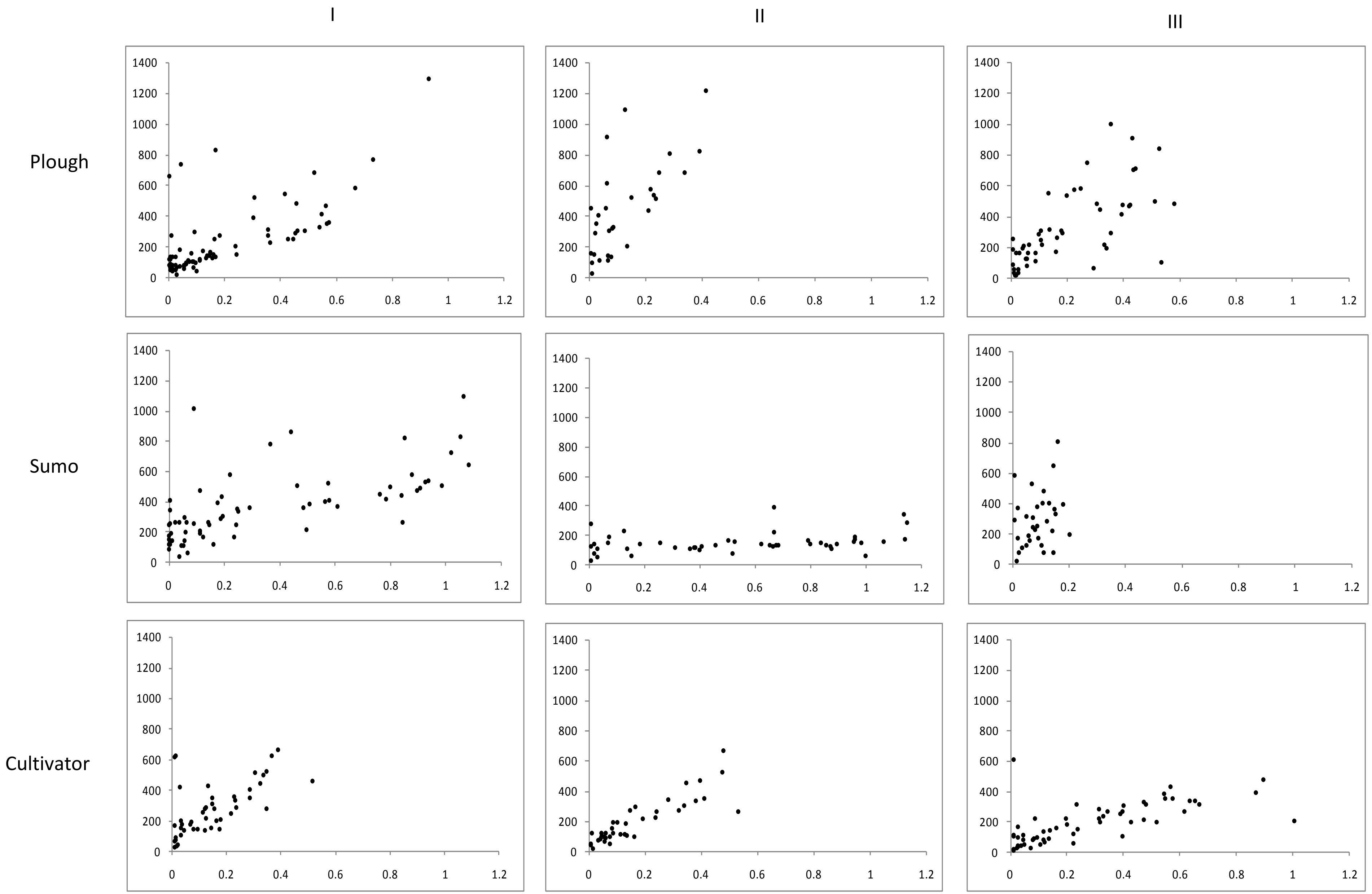

Figure 3.10: Discharge ($l.s^{-1}$) sediment concentration ($mg.l^{-1}$) relationship for individual plots.

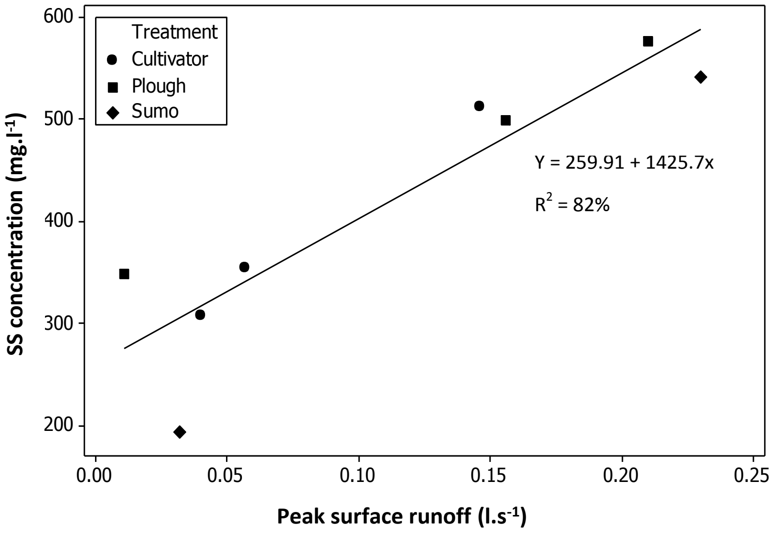

Figure 3.11: Relationship between plot sediment concentration and runoff peaks February 26th 2010.

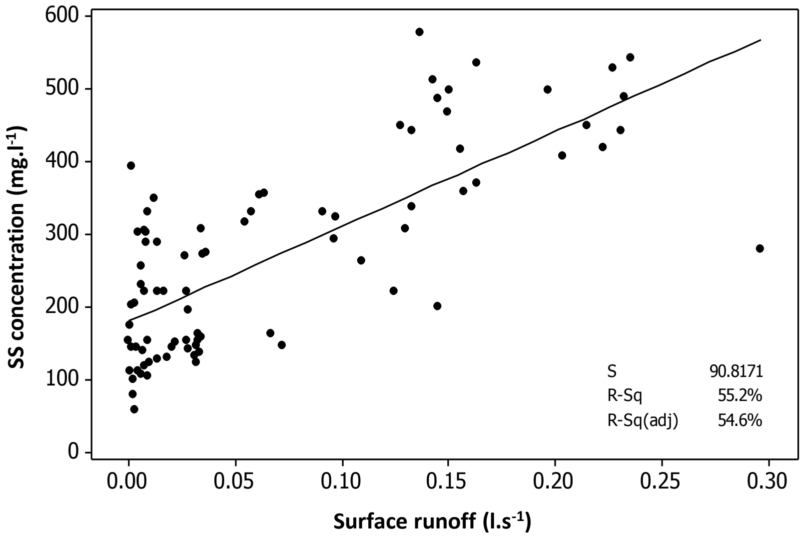

Figure 3.12: Surface runoff discharge effect on drain sediment concentration (all plots) Feb 26-27 2010.

The addition of surface discharge to drain discharge as a predictor in sediment concentrations significantly improves predictive accuracy when applied to all cultivation treatments (Figures 3.14-3.16), for example increasing explained variability from 39 to 58% for the plough treatment. This is attributable to its ability to explain a large proportion of the variability between treatment replicates. The prediction for the sumo cultivation treatment remains poor, although the effect of difference in surface runoff propensity on sediment losses is clearly illustrated (Figure 3.15).

While the addition of surface runoff as a variable explains a significant proportion of the variability observed between treatment replicates, the uncertainty associated with the residual error means that the use of a single rating for each treatment is not appropriate. Better predictive models are derived using rating relationships for individual plots.

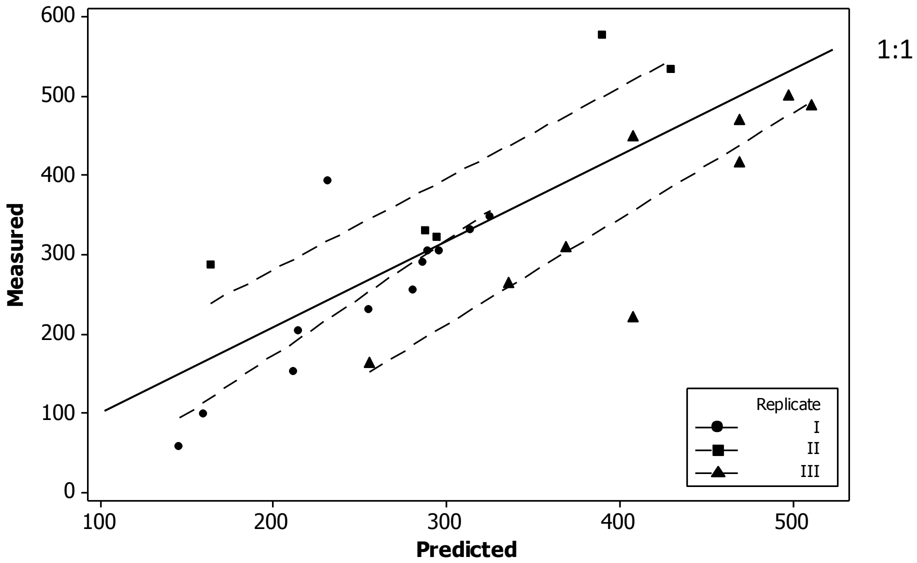

Figure 3.13: Measured Vs. predicted SS (mg.l^{-1}) for plough treatment plots Feb 26 - 27 2010.
Using the regression relationship SS = 138 + 304 drain Q + 1357 surface Q
R^2(adj) = 58.9%, constant T = 3.71 (p = 0.001); Drain Q T = 3.18 (p = 0.04); Surface Q T = 5.16 (p<0.001).

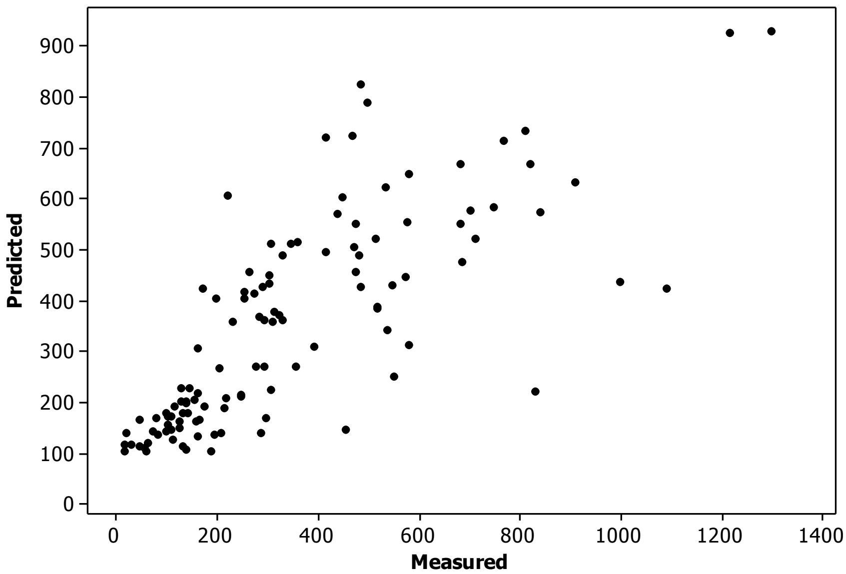

Figure 3.14: Plough treatment sediment concentration prediction (mg.l^{-1}).
Using the regression relationship SS = 91.6 + 695 drain Q + 2265 surface Q
R^2(adj) = 58.2%; Constant T = 3.46 (p=0.001); Drain Q T = 7.79 (p<0.001); Surface Q T = 7.18 (p<0.001).

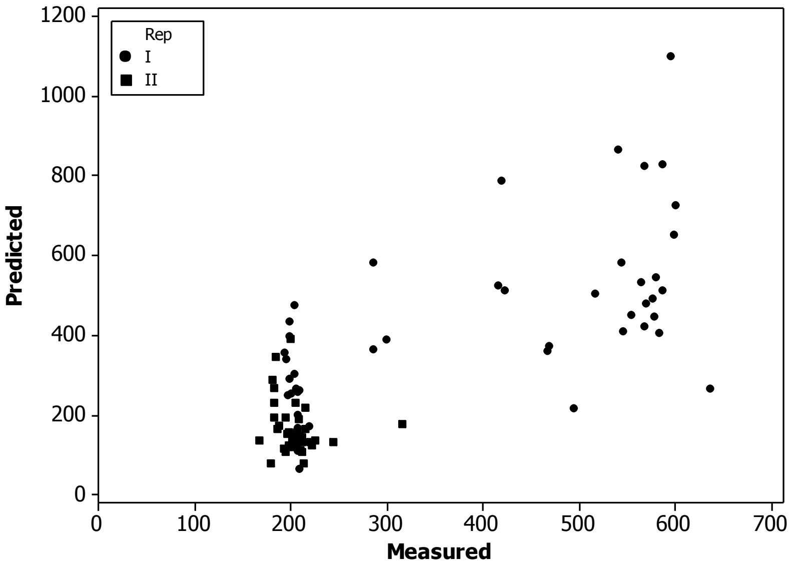

Figure 3.15: Sumo treatment sediment concentration prediction (mg.l^{-1}).
Using the regression relationship SS = 210 - 71.0 drain Q + 1857 surface Q
R^2(adj) = 55.2%; Constant T = 7.51 (p<0.001); Drain Q T = -1.44 (p = 0.154); Surface Q T = 9.54 (p<0.001)

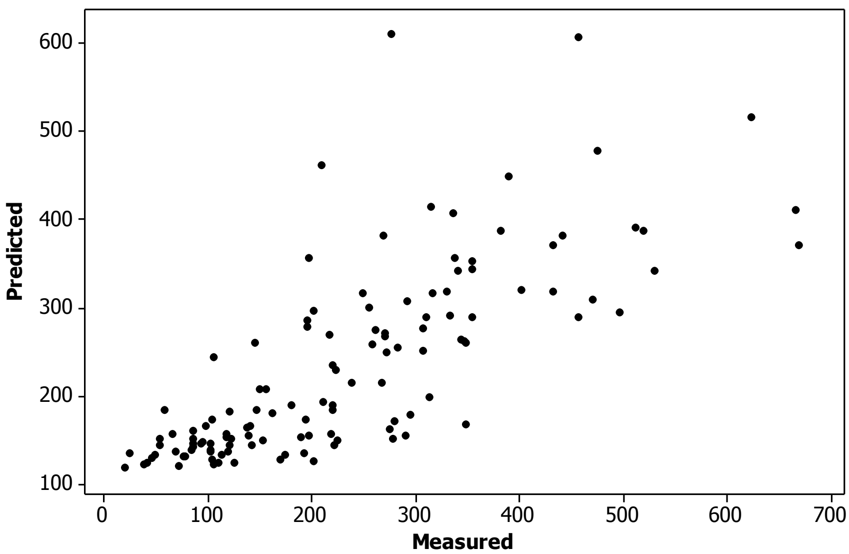

Figure 3.16: Cultivator treatment sediment concentration prediction (mg.l^{-1}).
SS = 114 + 264 Drain Q + 1363 surface Q
R^2(adj) = 57.1%; Constant T = 8.69 (p<0.001); Drain Q T = 5.55 (P<0.001); Surface Q T = 7.36 (p<0.001)

In the majority of cases sediment concentrations were found to be a function of both drain *and* surface discharge, despite considerable co-linearity. The effect of the addition of surface discharge as a variable is most clearly illustrated for plot 1 (Figure 3.17); runoff events with similar drain discharge can be differentiated by the volume of surface runoff, with higher runoff volumes significantly increasing drain sediment concentrations. This plot in particular appears to display different relationships between events generating surface flow and those where minimal flow is observed. As a result different ratings are employed dependent on surface flow conditions for this plot. For other plots a single rating is used. Sediment export estimates for the sumo treatment are not calculated due to the very large uncertainty associated with the poor rating. The decision to include or exclude surface Q as a predictive variable was made individually for each plot dependent on its predictive power. Due to the variability in low flow estimates intercept terms are excluded from all models. Manual samples collected during low flow conditions indicated low sediment concentrations, suggesting overestimation by automated sampling during these flow conditions. Variability in sediment load at low discharge is not seen as problematic however, as the effect on estimation of total sediment exports is minimal.

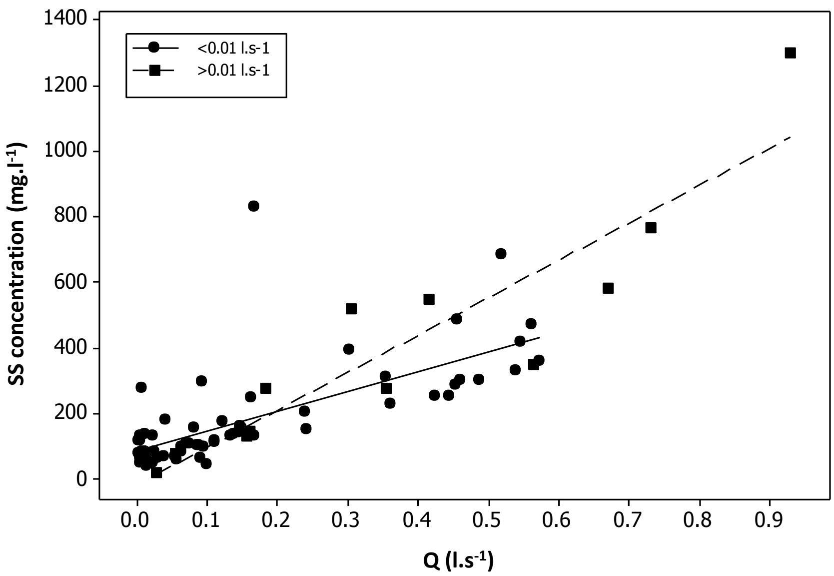

Figure 3.17: The effect of variable surface flow discharge (Q) on the drain discharge/sediment loss relationship plot 1 (plough treatment [T1] block I).

Predictive models for drain flow sediment rating

Plough treatment

Plot 1	If surface Q <0.01 l.s^{-1}: SS (suspended sediment conc.) = 607 Drain Q (R^2 = 70%)
	If surface Q >0.01 l.s^{-1}: SS = 549 Drain Q + 9602 surface Q (R^2 = 93%)
Plot 6	SS = 2512 Drain Q (R^2 = 76%)
Plot 8	SS = 1081 drain Q (R^2 = 52%)

Cultivator treatment

Plot 3	SS = 1416.6 drain Q (R^2 = 60%)
Plot 5	SS= 629 drain Q + 3208 surface Q (R^2 = 95%)
Plot 7	SS = 2281 drain Q + 313 surface Q (R^2 = 86%)

Suspended sediment monitoring included sampling at or near to peak flows, meaning rating can be conducted without significant extrapolation. However, discharges observed during the event of November 1st at the beginning of the monitoring period (Figure 3.3) were significantly in excess of measured values. As a result sediment export rating does not include this event, and figures provided are for the period 2nd Nov – 31st March.

Surface flow sediment

A total of 32 bulked event samples of surface runoff were collected during the monitoring period. Low replication and high variability between estimates resulted in poor estimation of mean values for individual plots. Average sediment concentrations for cultivation treatments decreased in the order Plough > Sumo > Cultivator and buffer strip treatments in the order No buffer > bare fallow buffer > grass buffer, although treatment differences were not significant in either case (Table 3.7). Surface runoff sediment concentrations were therefore based on the grand mean value (444 $mg.l^{-1}$) to avoid false differences based on cultivation treatment mean values. Surface flow sediment export was therefore calculated as

$$\sum \text{surface } Q(l.s^{-1})*444$$

Cultivation treatment	6 m grass	6 m bare fallow	No buffer	Mean	s.e
Plough (T1)	561	571	436	527	112.1
Sumo (T2)	217	581	710	487	123.7
Cultivator (T3)	318	590	270	333	65.0
Mean	343	452	579	444	99.2
s.e	76.8	109.8	104.7		

Table 3.7: Treatment effect on mean surface runoff SS concentrations ($mg.l^{-1}$).

3.3.3 Sediment exports

Total estimated drain sediment exports between 03/11/2009 and 01/04/2010 were significantly higher ($t = 3.82$; $p = 0.03$) for the plough treatment (T1) than the cultivator minimum tillage system (Table 3.8). There was no significant difference in surface runoff sediment yields between the treatments. Surface runoff accounted for 26% and 28% of total sediment export on average for plough and cultivator treatments respectively derived from 22% and 14% of total runoff volume.

Cumulative sediment exports illustrate that a small number of large rainfall/runoff events account for the majority of sediment losses. Events generating significant volumes of *surface* runoff were observed to account for larger export coefficients per unit runoff. These events occurred particularly during late December 2009 and January 2010, and result in both very large short term increases in cumulative drain export, and divergence between drain and total sediment export highlighting the contribution of surface runoff losses during this period (Figure 3.18).

The effects of both surface runoff and cultivation treatment on suspended sediment concentrations (sediment loss per unit runoff) are illustrated below for the February 26th to 27th event (Figure 3.20); much of the variability between treatment replicates is attributable to the plot propensity to surface runoff generation, demonstrated by the difference in concentrations between plough treatment plots 1 and 8. The use of the cultivator reduced tillage treatment sediment mobilisation, but is not attributable to reduced surface runoff (plot 7 Figure 3.20).

	Drain flow		Surface flow		Total	
Block	**Plough**	**Cultivator**	**Plough**	**Cultivator**	**Plough**	**Cultivator**
I	267	216	50	122	317	338
II	370	142	165	45	534	188
III	361	189	143	49	504	239
Mean	333	183	119	72	452	255
s.e.	32.8	21.5	35.1	25.0	67.8	44.2

Table 3.8: Plough and Cultivator treatment total sediment export ($kg.ha^{-1}$).

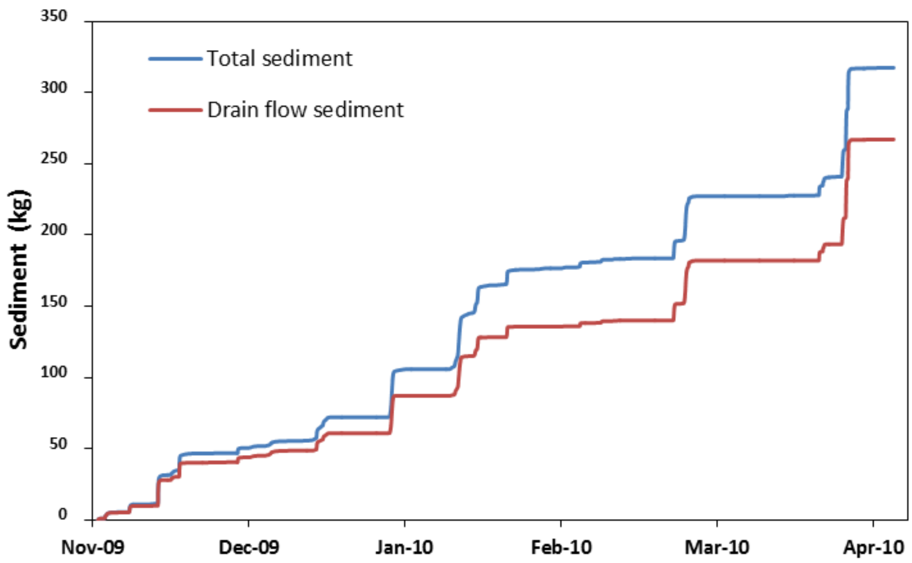

Figure 3.18: Cumulative sediment loss 2nd Nov-31st March Plot 1 (Plough rep I).

Predicted sediment concentrations for plough treatments suggest that while suspended sediment concentrations in excess of 200 mg.l^{-1} occur over only 5% of the winter monitoring period, exports in excess of 25 mg.l^{-1} occur over 25% of the time period (Figure 3.19).

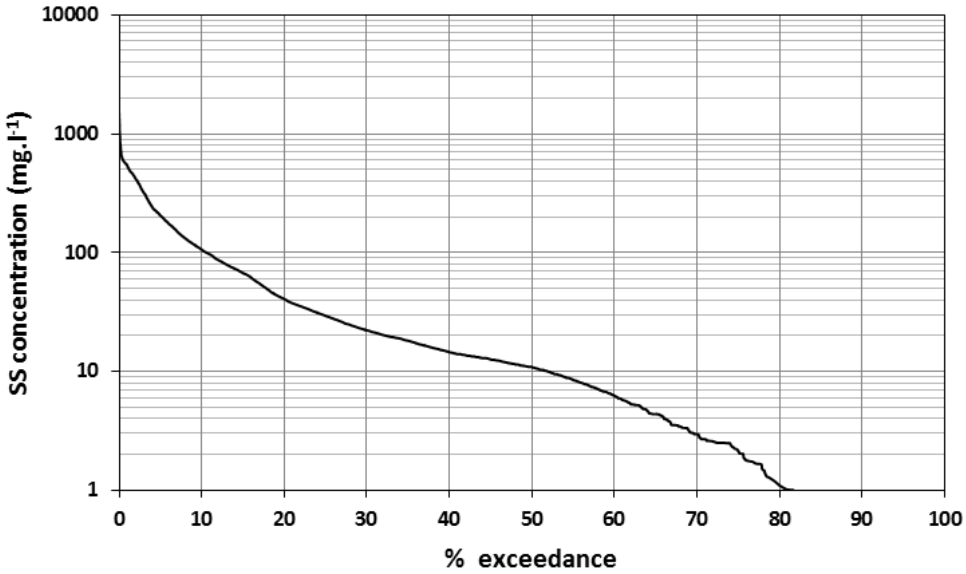

Figure 3.19: Mean suspended sediment duration curve for plough treatment (Nov-Apr).

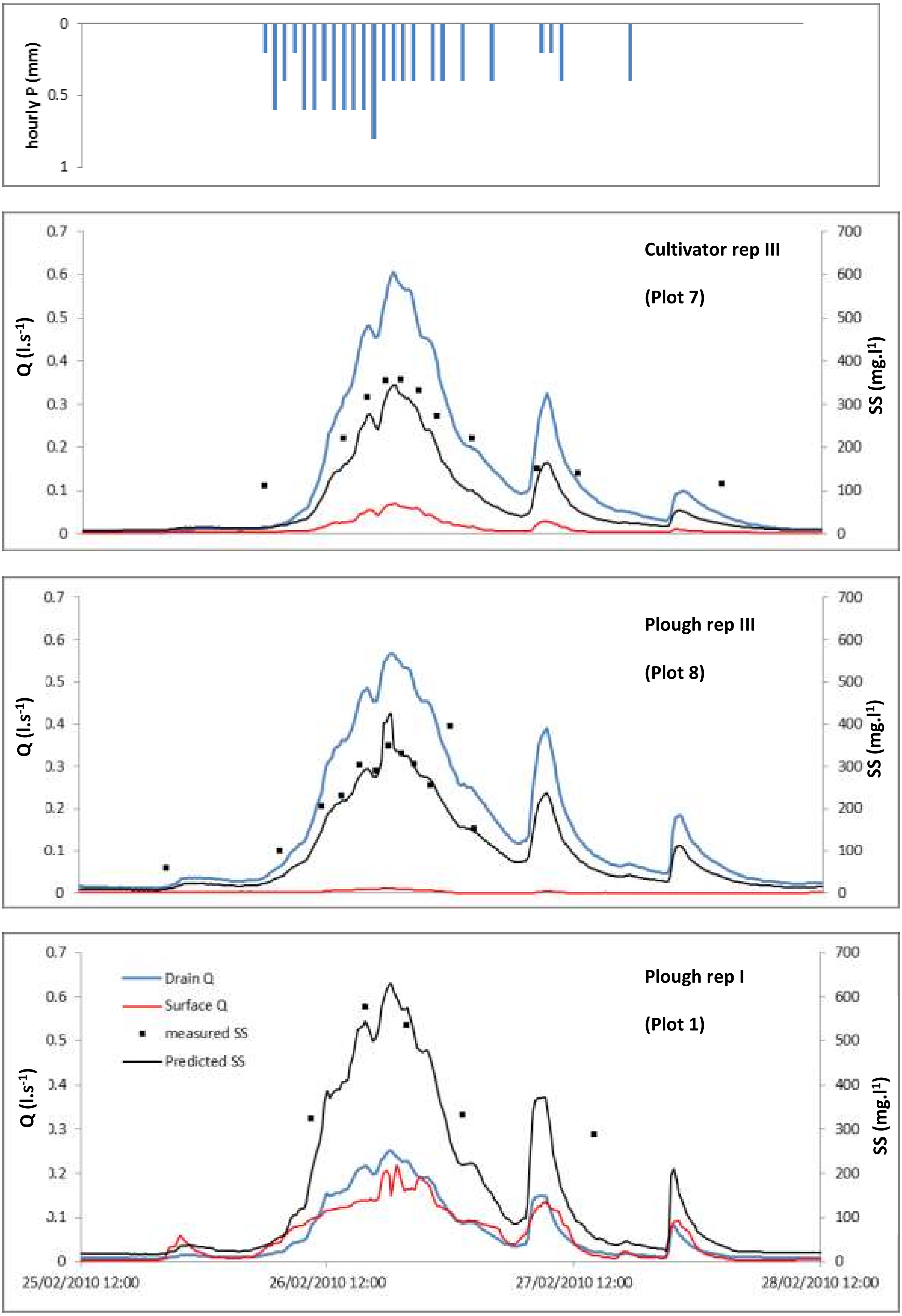

Figure 3.20: Comparative sediment-discharge relationships for February 26th to 27th event.

3.4 Discussion

3.4.1 Sediment losses and processes

It should firstly be noted that there may be factors in the study which influence the representativeness of sediment loss risk relative to median conditions for a catchment in this landscape. Establishment of the crop was late and poor, meaning soil cover over winter was relatively low. Conditions are therefore similar to a late sown cereal crop, and a high land use susceptibility to erosion (Defra, 2005). The drainage system is highly efficient, particularly as these soils are suitable for mole drainage, consequently surface runoff was low. The contribution of tile drains to total flows may therefore be high compared with less efficient systems. While the substitution of surface flow for tile drainage does not imply a reduction in sediment *mobilisation*, over large parts of a catchment remote from surface water networks, sediment *delivery* via drain flow may be much higher. In this study the surface runoff outlet is engineered to ensure runoff efficiency, which may exaggerate specific sediment yields per unit surface runoff compared with catchment scale processes. Assessment of site susceptibility to erosion by conventional methods (e.g. Defra, 2005) would indicate low risk; the slope is gentle, soils fine textured, and the site topography is not conducive to the generation of concentrated runoff pathways. The existence of highly efficient drainage would also be a factor considered to reduce the risk of (surface) erosion.

This experiment confirms the findings of previous studies employing a variety of methods, which have concluded that sub-surface drainage systems represent a significant conduit for agricultural sediment from field areas to surface waters (Russell *et al*., 2001 Schilling and Helmers, 2008; Collins and Anthony, 2008; Deasy *et al*., 2009; Billota *et al*., 2010). The figures do not take account of the largest runoff event occurring at the commencement of the monitoring period (outside the range of the predictive relationship between sediment concentration and discharge). As this event accounted for approximately 10% of drain output and included the highest peaks in both discharge and rainfall intensity, it can be concluded that the estimate provided is conservative with regards to the winter period as a whole. Oygarden *et al*. (2004) report that soil loss through tile drains has been measured at up to 1 $t.ha^{-1}.yr^{-1}$ in Norwegian catchments, further indicating that the observed figures are not extreme.

If the ploughed treatment (T1) is taken as representative of normal cultivation practice, losses of sediment from cultivated tile drained land may be large – in the region of 0.5 tonnes per hectare per annum or 50 tonnes per square kilometre (assuming summer losses when little drainage is expected and crop protection is high to be negligible). The figures indicate a discrepancy between soil erosion and sediment loss risk assessment for agricultural areas; while a rate of 0.5 tonnes per hectare per year would represent erosion within sustainable loss rates (Brazier, 2004) such figures may represent excessive sediment losses for catchments with similar climate and hydrology (Cooper *et al.*, 2008). This indicates erosion assessment techniques are relatively insensitive to the 'chronic' pollution problem resulting from losses through drains over large areas compared with the 'critical' one associated with erodible soils on steep slopes and along concentrated flow paths, which apply to more limited areas (Boardman *et al*., 2009) Although average catchments are likely to include susceptible land uses in mosaic with lower yielding areas, results indicate that a large proportion of cultivated tile drained land within a catchment can be a primary cause of aquatic sediment problems. This supports previous predictions derived from the PSYCHIC assessment of sediment loss risk for catchments in England and Wales (Collins and Anthony, 2008b; Stromqvist *et al*., 2008), which have concluded that the coincidence of arable land uses with slowly permeable soils indicates a sediment pollution risk associated with artificial drainage. Somewhat counter-intuitively therefore, the site factors widely regarded to indicate low *erosion* susceptibility (fine textured slowly permeable soils, low slope) also indicate the likely presence of artificial drainage, and the high degree of connectivity this engenders may mean this land is high risk with regard to sediment loss to water.

The accuracy of current estimates may be verified by comparison with previous observations at the study site. During winter 1980 measurements of suspended sediment losses were used to estimate loads from plots within the central block (plots 4-6). A later study using the same three plots was made over the winter of 1990-1991 (Table 3.9). The treatment with mole drainage represents the same drainage method applied to all plots in the current study. Both estimates are of losses in the hundreds of kilograms, although in both cases these losses were considerably lower than observed in the present study. In both cases estimates were derived from a much smaller number of aggregated samples (to provide overall estimates at intervals of 48 and 24 hours respectively), which may explain lower storm export estimates than derived in the current study; as the majority of flow occurred during runoff events, aggregation of samples or irregular sampling is likely to result in underestimation of peak exports, and hence total seasonal export. Other more recent studies (Russell *et al*., 2001; Bilotta *et al*., 2008; Deasy *et al*., 2009) have also found very high sediment contributions during large runoff peaks.

The earlier studies do not suggest a decline in soil erosion losses from bare soil surfaces as a result of drainage. Such a finding would appear to support the contention that efficient drainage systems are able to transport mobilised sediment with similar efficiency as surface runoff, and indicates that channel erosion by concentrated runoff is of minimal importance on these flat even surfaces.

		Tile and mole drained	Tile drained only	Undrained control
Armstrong, 1980 (ploughed fallow Oct-Apr)	Surface	24.6	35.86	123.4
	Drain	87.6	185.2	-
	Total	112.2	221.06	123.4
Brown, 1990 (winter wheat Dec-Mar)	Surface	91.2	-	321.6
	Drain	194	-	-
	Total	285.2	-	321.6

Table 3.9: Previous measurements of sediment losses ($kg.ha^{-1}$) Cockle Park Drainage Plots.

Total sediment losses appear to be dominated by a small number of large rainfall-discharge events, with minimal losses outside of event flow periods (Figure 3.18). This pattern is similar to seasonal patterns of soil erosion and confirms the view of previous researchers that drain sediments are derived from surface processes rather than mobilisation of sub-surface material (e.g. Beauchemin *et al*., 1998; Luabel *et al*., 1999; Schilling and Helmers, 2008). The degree to which mobilised sediment is able to reach water bodies (the sediment delivery ratio) is a function of the efficiency of the drainage system – in this study the time lag between rainfall and runoff was extremely short (see Figure 3.20 for example). This limits the potential for sedimentation between mobilisation and export, which may be a greater factor in less efficient systems, although conversely, this efficiency reduces the likelihood of surface runoff generation, which has been shown to reduce volume specific mobilisation in this study.

Suspended sediment duration analysis (Figure 3.19) indicates that very high sediment concentrations in drain flow occur infrequently. Mean suspended sediment targets of 25 $mg.l^{-1}$ derived from the Freshwater Fish directive (78/659/EC) have been criticised as inappropriate by Bilotta and Brazier (2008), given that very high sediment concentrations during large events skew averages, and are likely to be less problematic than persistent elevations of intermediate magnitude. The data presented here indicates that suspended sediment from arable tile drains only presents sustained turbidity problems between cultivation and the start of the growing season, and is unlikely to present major problems during ecologically sensitive summer months. However, drainage water is predicted to exceed the 25 $mg.l^{-1}$ threshold for over 25% of the period between November and April, a much higher duration impact than that from surface runoff. The potential impact of sustained inputs of sediment from tile drains on will be dependent on the dilution effect of 'cleaner' water from elsewhere in the catchment, but there is some indication that the duration of exposure to suspended sediment may be increased by the presence of tile drainage, particularly during smaller rainfall events and the rising limbs and recessions of larger events. This may present particular problems in spring and autumn following cultivations.

3.4.2 Treatment effects on sediment losses

The reduction in specific sediment export with the cultivator relative to the plough treatment is not due to drainage differences; neither total runoff yields nor the relative contribution of surface runoff differed significantly. Given that previous management, soil properties and precipitation inputs were identical for all plots, the difference appears directly attributable to cultivation treatment. The apparent reduction in specific sediment yields suggests greater soil resistance to mobilisation. Soil surface conditions did not vary between treatments in the experiment; all plots were rolled to create fine tilth for drilling, meaning that the effect does not appear to be due to differences in seedbed conditions. The difference would therefore appear to be the result of higher aggregate stability in the cultivator plots. Shallow mixing and the absence of the use of a power harrow appear to have limited the generation of easily mobilised fine material. It is unclear whether this effect is most pronounced at the soil surface (where greatest erosive energy is experienced) or within the top soil where rapidly percolating water may mobilise unstable material along macro-pore channels. Daraghmeh *et al*. (2009) found that power harrowing can have detrimental effects on aggregate stability (even under reduced tillage). It is also possible that cultivation above recently drawn mole drains has resulted in soil material entering the drainage network directly during cultivation, although the fact sediment losses did not decrease over time suggests this is less likely.

It is important to note that while reduced tillage resulted in significant reduction in drainage sediment losses of approximately 45%, mean losses of in excess of 183 kg per hectare remain potentially problematic from a water quality perspective. A large proportion of drained arable land within a catchment may mean that this pathway represents a significant contribution to problems of excess sediment in water bodies. Losses may be reduced further by maintaining a degree of soil protection by direct drilling into a growing crop or crop residues, or by the substitution of spring for winter sown crops (with cover cropping) where appropriate.

The results provide weak evidence that the use of vegetated buffer strips may reduce sediment losses along *surface* pathways. It appears likely that a significant part of the mitigation effect is directly attributable to the reduction of losses from the buffer itself (due to non-cultivation), as these narrow grass strips are unlikely to have any major effect on flow velocity or volume in saturated soils of this type. However, it is important to note that in this experiment the majority of flow and sediment travelled through the sub-surface drainage system, and therefore even 100% sediment retention would only mitigate sediment losses by approximately 26 and 28% on average for the plough and cultivator treatments respectively. In situations where the drainage system is less efficient, and erosion hazard higher, the relative surface sediment contribution and the potential mitigation effects of the vegetated buffers may be greater. However, surface runoff from drained slowly permeable soils is likely to be limited to large events when rainfall exceeds drainage capacity under UK conditions. Reductions in flow energy due to increased surface runoff and or infiltration are likely to be limited under these circumstances. Previous reports of buffer strip mitigation effectiveness of the order 60-80% (e.g. Lui *et al*., 2008; Deibel *et al*., 2009; Versatraeten *et al*., 2006) therefore appear particularly inappropriate in a tile drained landscape.

3.4.3 Treatment effects on hydrology

The experiment recorded variable performance for the Sumo cultivator system, but there is evidence that this treatment resulted in greater runoff yield than either the plough or cultivator treatments. As the plots were treated identically prior to cultivation, this would appear to be a direct treatment effect. The difference may be attributable to a reduction in soil moisture storage resulting from greater connectivity between the drainage system and the soil surface. This treatment involved the use of a sub-soiler to 'lift' the top soil and reduce bulk density, which may have served to increase the proportion of interconnected macro-pores. Rapid flow to the drainage system rather than macro-pore storage would explain both the increase in export volumes and the absence of a negative correlation with surface runoff volumes. Under the plough and cultivator treatments discontinuity of macro-pores may result in greater retention of event rainfall, and increased soil moisture storage. Pagliai *et al*. (2004) demonstrated that the use of a shallow depth sub-soiler in preference to a mouldboard plough reduced soil bulk density and increased the proportion of transmission pores at the base of the cultivated layer, which supports the contention that greater pore connectivity in the sub-soiled plots increased by-pass flow and reduced retention in soil surface layers. It is assumed that under the climatic conditions of northern England these soils remain at field capacity over the majority of the winter period (Jarvis, 1984). Previous measurements at the site have indicated that efficient mole drainage systems can result in long delays in return to field capacity (Brown, 1993). The difference between field capacity and maximum soil moisture storage (saturation) is also large and variable for the top soil; poor structure and connectivity may result in limited drainage even where the soil moisture content exceeds field capacity. The difference between field capacity and saturation for these top 30 cm of these soils is sufficient to account for a large proportion of the difference between treatments as well as the discrepancy between observed rainfall and drainage (Tables 3.4 & 3.5) under field capacity conditions. Previous studies have also demonstrated that tillage systems can affect tile drainage flow volume on poorly drained soils; Randall and Iragavarapu (1995) reported zero tillage increased tile drainage volume compared to conventional, while Tan *et al*. (2002) reported lower winter soil moisture storage in reduced tillage plots compared with mouldboard ploughing.

3.5 Summary and conclusions

The evidence confirms that under susceptible land uses tile drains may represent a chronic sediment loss source largely independent of site erosion susceptibility, requiring consideration in water quality improvement programmes. The findings support those of Chapter 2, which suggested that drain losses may represent a large proportion or majority of annual sediment export from an agricultural landscape. Losses of sediment through this pathway are likely to be greatest during rainfall events occurring at susceptible times under UK conditions – particularly in autumn and winter under winter sown cropping, or spring for spring cultivations. The water quality impacts are dependent on the importance of sediment as both a direct pollutant and vector for other pollutants at high flows and outside of summer base flow periods. Impacts are more likely through the effects of channel siltation of rivers lakes and reservoirs and nutrient retention in lakes and coastal waters, than as a result of the short term effects on water chemistry and turbidity. The high connectivity of tile drains with surface waters means that the this sediment issue may apply to large areas of an agricultural catchment, unlike localised soil, bank or track erosion problems, which may represent much greater yields per unit land area, but apply to a much more limited proportion of the catchment. Consequently the understanding of the routes of sediment loss has important implications for the assessment of sediment loss risk as well as the efficacy of different mitigation strategies. This study indicates that reduced tillage methods may represent an effective mitigation strategy, but that the effectiveness of buffer strips may be relatively limited in a tile drained landscape.

SECTION 2: CATCHMENT SCALE PROCESSES

Chapter 4: Background to Belford Burn study site

4.1 Introduction

Chapters 4-6 investigate suspended sediment loss processes at the larger sub catchment to catchment scale. These studies take place in similar lowland Northumberland landscape to the sites utilised in Chapters 2 and 3, characterised by slowly permeable soils. The study area was selected for the study due to the existence of a detailed flow monitoring network; evidence from measurements also suggested that agricultural pollution is having a detrimental impact on freshwater ecology in the area. Consequently the chapter introduces the Belford Burn study area, including background to previous work conducted in the catchment. The objectives of this chapter are to characterise flow and suspended sediment regime in the study catchment and to examine the evidence for the effect of agriculture on water quality. The chapter is divided into two sections. Section 4.2 describes the wider context of Budle Bay analysing secondary evidence to assess the major water quality issues and the impact of agricultural diffuse pollution on aquatic receptors. This secondary evidence does not include direct evidence regarding suspended sediment regimes (which is not a component of standard EA water quality monitoring). In section 4.3 primary evidence regarding flow and sediment regimes in the Belford sub-catchment of Belford Burn is presented.

4.2 The Budle Bay catchment

Budle Bay is classified as a sheltered meso-tidal embayment (CW8), and is located on the north east coast of England close to the Scottish border (NU 150,356; Latitude 55:37N). The bay has an area of approximately 315 ha and a total catchment area of around 9400 ha (94 square kilometres). Budle Bay is located within the Berwickshire and North Northumberland Coast Special Area of Conservation (England/Scotland) under the Habitats Directive (92/43/EEC). It forms part of the Lindisfarne Special Protection Area protected under the Birds Directive (79/409/EEC), and is also designated a Natura 2000 site and Ramsar wetland. The extensive intertidal mudflats and sandflats of the ecosystem support large beds of narrow leaved eelgrass (*Zostera angustifolia*), dwarf eelgrass (*Z. noltei*) and a diverse infauna. This infauna in turn supports internationally important populations of water fowl.

The catchment is formed of lowland topography, draining mainly west to east from a dip slope formed from the Carboniferous Fell Sandstone outcrop. Solid geology is mainly composed of mid-Carboniferous sandstones, mudstones and limestones, and the major Whin Sill dolerite intrusion outcropping east to west and forming the elevated steeply sloping southern margins of the bay. Solid geology is mainly covered by Devensian glacial till, with smaller areas of glacial sands and gravels, particularly in low lying areas in the south of the catchment. The dominant soils are fine loamy and exhibit slow permeability being formed from till deposits, with smaller areas of freely draining soils occurring on higher ground to the east of the catchment where drift cover is absent, and over glacial sands and gravels.

The catchment drainage network is comprised of a number of small streams known locally as ‘burns’. The Waren Burn flows into Budle Bay from the south west, and Ross Low from the north west, the product of the confluence of Belford, Middleton and Elwick Burns (Figure 4.1). These burns along with those of Holy Island Sands to the north are known collectively as the ‘Lindisfarne Coastal Streams’.

The North Northumberland Coast has a relatively low population density hence the Budle Bay catchment is predominantly agricultural, with a mixture of intensive pasture and arable cropping (mainly cereals) particularly in the flat coastal area adjoining the bay. The upper slopes of the catchment to the west are mainly used for lower intensity grazing and conifer plantation. The town of Belford (population around 1000) is the only sizeable settlement within the catchment. The town is served by a water authority sewage treatment works, which discharges treated effluent to Belford Burn approximately 1 km downstream of the town centre and 2 km upstream of Budle Bay.

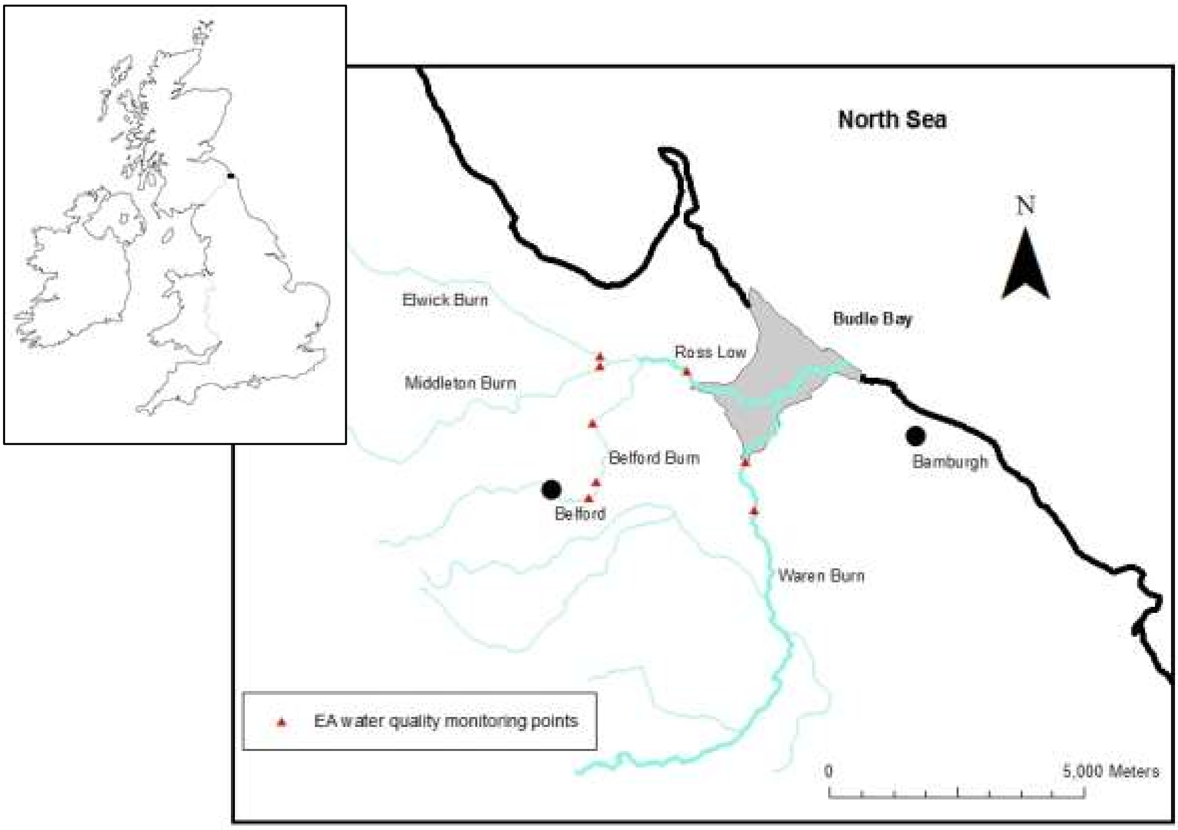

Figure 4.1: Map of the Budle Bay catchment.

4.2.1 Ecological status and water quality

Over recent decades, increasing summer blooms of macro-phytic algae (mainly *Enteromorpha/Ulva intestinalis*) have occurred in Budle Bay. Increased macro-algal growth replacing a sea grass dominated community is consistent with eutrophication in transitional waters. The macro-algae have been observed to form thick mats of sufficient density to inhibit the growth of the eelgrass beds. A study of algal cover conducted by Natural England between 1995 and 2000 reported these blooms to be of significant extent (Table 4.1), and that the area of dense mat in particular appeared to be increasing. The distribution of the mats occurs exclusively at the landward side of the bay particularly around the inlets of the freshwater tributaries (Figure 4.2).

The percentage of the total bay area covered by opportunistic blooms of macro-algae in summer months, and the negative impact of these blooms on the status of the *Zostera* beds and associated benthic inflora is sufficiently high to limit the ecological status of Budle Bay to "moderate" according to the WFD status indicators for England and Wales (Environment Agency, 2010). The assessment of macro-algal status in the UK WFD classification is based on the extent (% area) and density (g.m^2) of algal cover. Moderate ecological status regarding opportunistic macro-algae applies where mat densities of over 500g.m^2 cover between 15 and 25% of the designated area. The failure to meet good ecological status with regard to this biological quality element derives from the consensus that at this density the algae are likely to result in some modification of the underlying sediment and its benthos (UKTAG, 2008). This may in turn negatively impact the food source for wading birds.

	Cover (ha)			
Date	**Dense**	**Sparse**	**Total**	**Percentage Cover***
Jun-95	0.4	3	3.4	1.1
Sep-95	0	8	8	2.5
Aug-96	9	37	46	14.6
Aug-97	34	36	70	22.2
Sep-97	37	36	73	23.2
Sep-98	36	36	72	22.9
Jun-99	31	42	73	23.2
Sep-99	43	21	64	20.3
Jul-00	48	28	76	24.1
Sep-00	58	25	83	26.4

Table 4.1: Summer macro-algal cover Budle Bay. (Source: Environment Agency records)
* % Total Area of Budle Bay (315 ha)

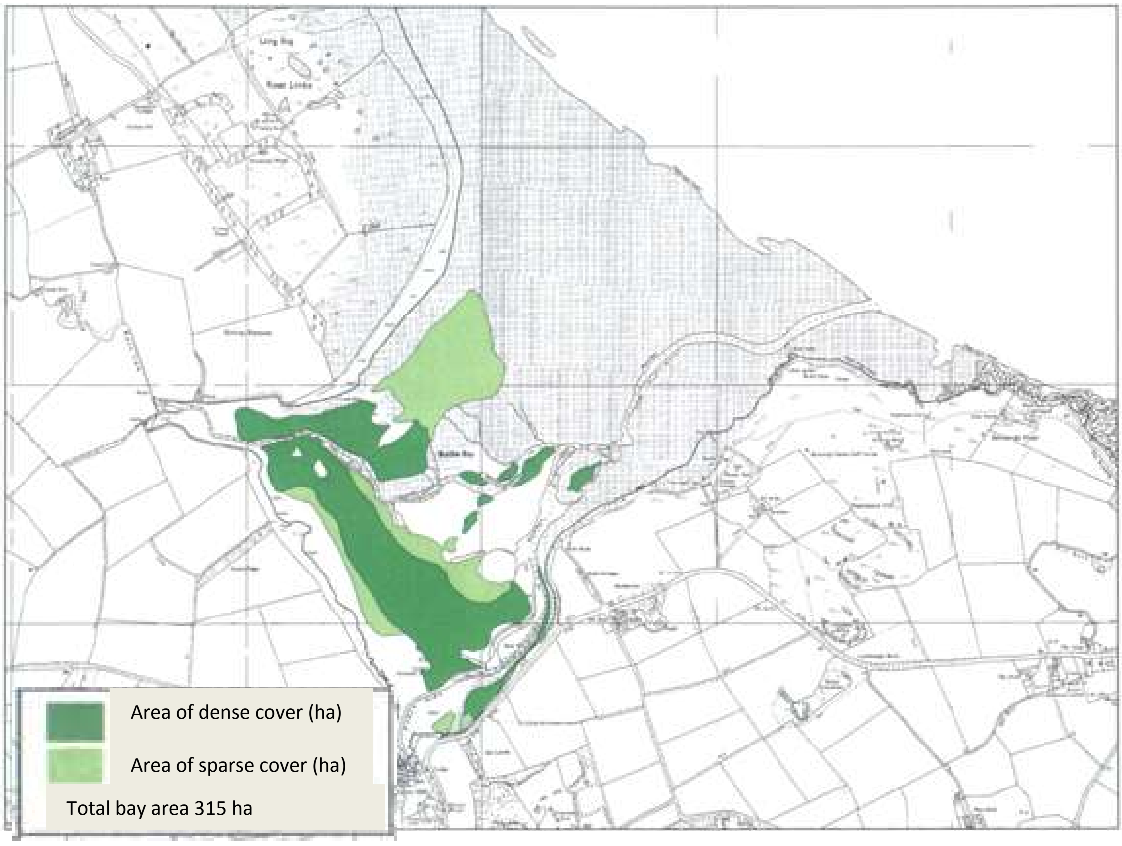

Figure 4.2: Distribution of algal mats in Budle Bay in July 2000 (Source: Environment Agency records).

4.2.2 Methods

Results are presented from a long term water quality and quantity monitoring record from Budle Bay catchment collected by the Environment Agency at statutory monitoring points on a monthly basis from 1998-2008. Spot water samples were collected at approximately monthly intervals at designated sample points on all freshwater tributaries to Budle Bay (Figure 4.1). Flow monitoring data was collected at the mouth of Waren Burn only (Figure 4.1) between 2000 and 2008.

Differences between in Dissolved Reactive Phosphorus (DRP) and nitrate concentrations between different time periods were analysed using 2 sample t tests. Differences in mean upstream and downstream concentrations from Belford STW were analysed using paired 2 sample t tests.

4.2.3 Results

Freshwater nutrient inputs

Mean dissolved nitrate concentrations were 4.07 and 4.12 $mg.l^{-1}$ nitrate as N from Waren Burn and Ross Low (the two freshwater tributaries of Budle Bay), respectively between 2000 and 2008. Mean Dissolved Reactive Phosphorus (DRP) concentrations were 0.04 and 0.17 $mg.l^{-1}$ respectively over the same period. Results indicate that the much higher phosphate concentrations in Ross Low can be attributed to inputs from its Belford Burn tributary (mean 0.27 $mg.l^{-1}$). Belford Sewage treatment works accounts for the majority of the DRP in Belford Burn (Table 4.3).

The Belford treatment works was upgraded to include tertiary phosphate stripping in 2004 due to the reduction in phosphate discharge consent introduced under the Wastewater Treatment Directive in response concerns over the effects of DRP discharges on Budle Bay. The results indicate that this upgrade resulted in a significant reduction in phosphate levels in Belford Burn ($t = 5.01$, $P = <0.001$) and subsequently in Ross Low ($t = 4.72$, $P = <0.001$) (Tables 4.2 and 4.3), although concentrations entering the bay from Ross Low remain significantly higher than those from Waren Burn. Under the same use of the Wastewater Treatment Directive a small consented discharge for sewage effluent at Waren Mill on Waren Burn was also removed in 2004 (via installation of a soakaway drain). Average DRP concentrations fell significantly as a result ($t = 3.66$, $P = 0.001$). There were no significant changes in dissolved nitrate concentrations in either tributary over the same period. Measurements upstream and downstream of Belford STW indicate that diffuse sources represent a much larger proportion of nitrate than phosphate (Table 4.3).

Nutrient export and stream discharge

Dissolved phosphate concentrations are significantly increased by inputs from Belford sewage treatment works (Figure 4.3). However, there are a number of sample occasions when high concentrations are observed upstream of the sewage works and the effect of the effluent on downstream concentrations is markedly less; grouping the relationships demonstrates that while for the main group (*point source pollution*) the downstream concentrations are 9.7 times higher than the upstream, this increase is minimal (1.07 times) for the *diffuse pollution group*. Comparison of this relationship with nitrate concentrations for the same samples (Figure 4.4) illustrates that the same pattern does not apply; there is a significant correlation ($r = 0.462$, $p<0.001$) between samples collected upstream and downstream of the STW on the same date, (a downstream increase of approximately 25% on average).

The relationship between flow conditions and nutrient levels over the 2000-2008 sample period indicates that high DRP concentrations occur during periods of high discharge (Figure 4.5). When these concentrations are converted to nutrient export rates (Figure 4.6) there is evidence to suggest that these relatively infrequent events may account for a significant proportion of nutrient loss from the catchment.

	DRP ($mg.l^{-1}$)		Nitrate as N ($mg.l^{-1}$)	
	2000-2003	**2004-2008**	**2000-2003**	**2004-2008**
Waren Burn	0.06	0.02	3.98	4.09
SE	0.009	0.002	0.12	0.13
Ross Low	0.29	0.09	4.14	4.09
SE	0.04	0.017	0.10	0.13

Table 4.2: Nutrient concentrations Budle Bay tributaries before and after treatment works upgrade.

	DRP ($mg.l^{-1}$)		Nitrate as N ($mg.l^{-1}$)	
Sample point	**2000-2003**	**2004-2006**	**2000-2003**	**2004-2006**
Upstream STW	0.08	0.05	4.49	4.20
SE	0.04	0.01	0.24	0.17
Downstream STW	0.48	0.17	5.17	5.6
SE	0.05	0.03	0.17	0.22

Table 4.3: Nutrient concentrations Belford Burn before and after treatment works upgrade.

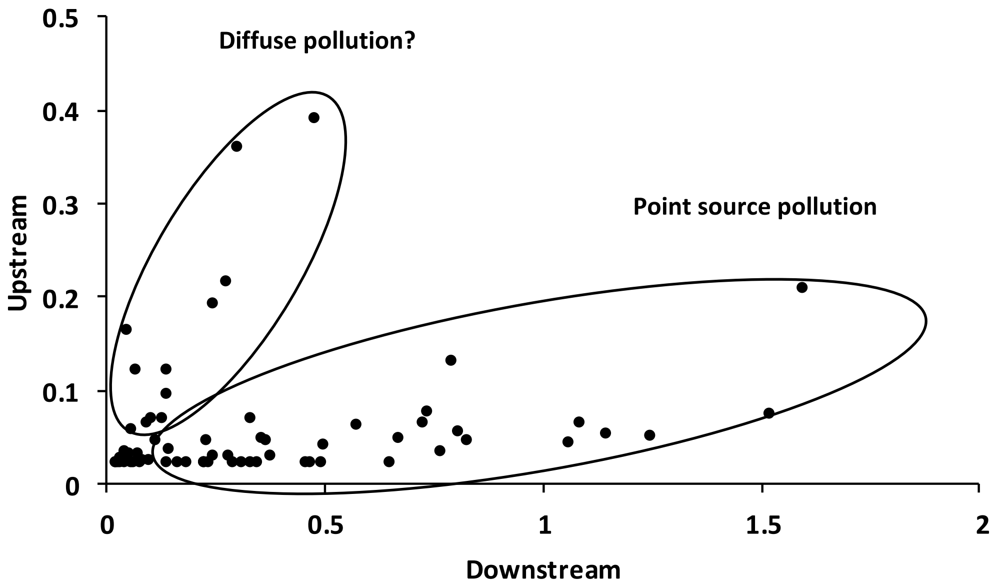

Figure 4.3: Comparison of paired dissolved reaction phosphorus concentrations ($mg.l^{-1}$) upstream and downstream of Belford STW.

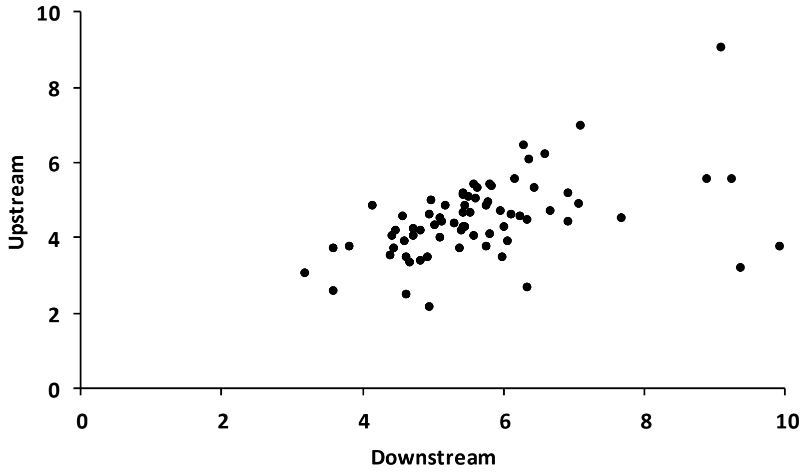

Figure 4.4: Comparison of paired nitrate (as N) concentrations (mg.l^{-1}) upstream and downstream of Belford STW.

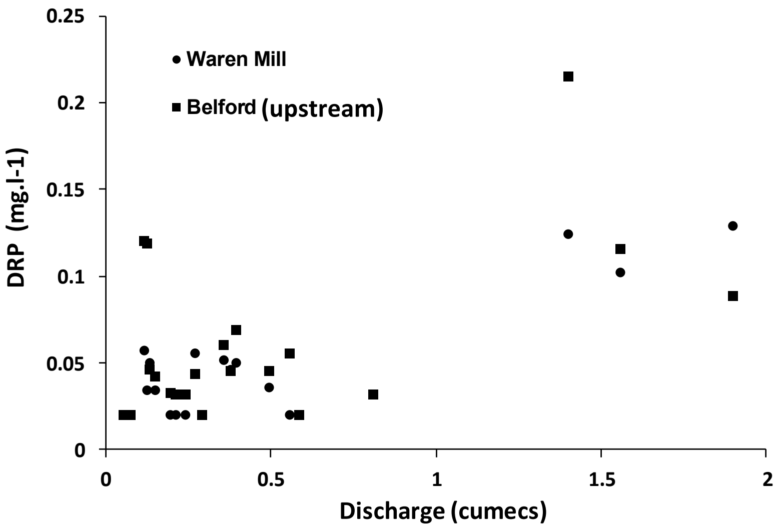

Figure 4.5: relationship between mean hourly discharge (Waren Mill) and dissolved reactive phosphate concentrations (Discharges <0.05 m^{3}.s^{-1} removed).

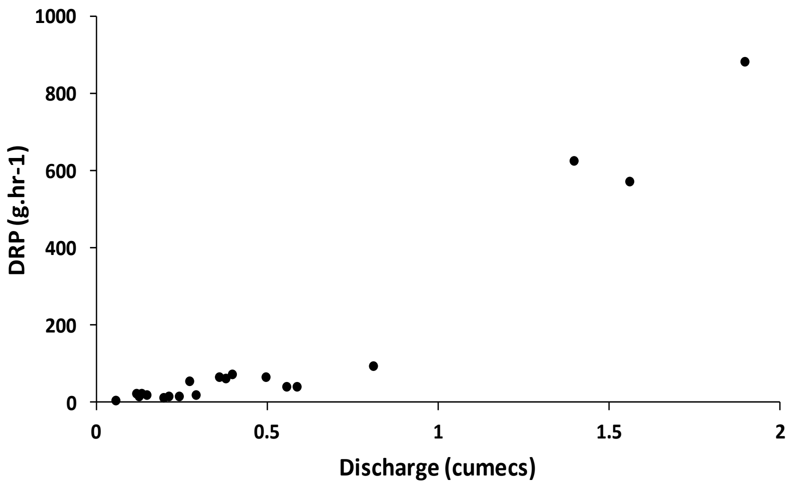

Figure 4.6: relationship between mean hourly discharge and dissolved reactive phosphate exports (Waren Mill).

4.2.4 Discussion

Analysis of flow patterns indicates that the majority of samples are collected during low flow conditions. A sampling regime randomised for flow conditions (by sampling at defined intervals) is unlikely to provide adequate data high flow events, even for a relatively long dataset such as the one presented here. Such a strategy is arguably appropriate for river water quality monitoring, where the infrequent elevated input of dissolved nutrients during flow peaks is unlikely to have a major impact on ecology. However, where the flows impact a receptor such as transitional waters and sheltered coastal bays and inlets with longer residence times and retention of nutrients in particulate as well as dissolved form, an understanding of the effects of these short-term events on nutrient dynamics becomes important.

Analysis of DRP concentrations upstream and downstream of Belford STW indicates that while on most sample dates elevated downstream concentrations can be attributed to effluent from the sewage treatment works, on a number of occasions significant upstream pollutant sources are observed (Figure 4.3). In the absence of flow records for Belford Burn it is not possible to confirm whether these dates coincide with higher flow levels as observed in the Waren Burn catchment, although comparison with samples collected on the same date from Waren Burn suggests this to be the case (Figure 4.5).

The data collected by the EA regarding the water quality of freshwater tributaries indicates that prior to 2004 Ross Low failed to meet good ecological status for nutrients on the basis of the concentrations of dissolved phosphorus. This failure can be attributed to the input from the sewage treatment works at Belford. Following the installation of improved phosphate treatment at the works the water quality of this input improved significantly, although remaining significantly higher than the comparable input from the Waren Burn. However, the distribution of algal mats in Budle Bay is not visually greater around the mouth of Ross Low than Waren Burn (see Figure 4.2). In 2009 a study was initiated to investigate whether the installation of phosphate stripping at Budle Bay STW had had any effect on the eutrophication issue in Budle Bay. No decrease in macro-algal assemblages has yet been observed (Mill, 2009). Neither was any strong correlation between the distribution of macro-algae and phosphate concentrations in the sediments of Budle Bay, although concentrations are reported between 6.8 and 26.8 mg.l^{-1} (Mill, 2009), which indicates that phosphate is unlikely to be a limiting nutrient. The spatial distribution of the algal mats would appear to be a function of other environmental variables such as exposure to tidal energy and salinity which determine algal habitat suitability. However the presence of the algal blooms would still appear to indicate eutrophication, and the most likely source would appear to be the freshwater tributaries.

Given that the sediments described by Mill (2009) appear to exhibit high concentrations of nutrients – many times higher than that of the input water (Table 4.2), it is possible that the source of the sediments is the cause of the eutrophication, although phosphate inputs from organic inputs within the bay may also be an important factor. Although there is no evidence to suggest there are unusually high agricultural sediment inputs to Budle Bay, the nature of this tidal embayment may mean that near-shore deposition causes greater ecological impacts than in more common less sheltered coastal ecosystems. The distribution of mud flats suggests that large quantities of riverine clay and silt are deposited close to the mouth of the tributaries. The much higher exports of nutrients during runoff events appear to indicate a greater role of agricultural runoff than average concentrations would indicate. The lack of suspended sediment measurements from the suite of standard EA tests (which are performed on filtered samples) means that this may only be inferred however.

Management of nutrients in Budle Bay

Previous to the recent publication of River Basin Management Plans under the WFD, a Habitat Management Plan was drawn up by Natural England following the designation of the Natura 2000 Berwickshire and North Northumberland Special Area of Conservation. This acknowledged the freshwater eutrophication problem in Budle Bay. The habitat management plan reported that diffuse agricultural pollution was the main concern, but acknowledged that there was no available mechanism for regulating these pollution inputs to the bay. A Catchment Management Plan was called for.

The River Basin Management Plan for Northumbria River Basin District (Environment Agency, 2009) details the proposed measures to improve ecological status in the region in accordance with requirement of the WFD. This RBD covers 9000 square kilometres from the Scottish border to North Yorkshire and includes 170km of coastline. Budle Bay is one of nine Special Areas of Conservation (SACs) and one of six Special Protection Areas (SPAs) within the RBD. Annex D of the River Basin Management Plan (Environment Agency, 2009) reports that the Budle Bay protected area is not achieving its environmental objectives and attributes this to a water quality problem caused by agricultural runoff.

The Habitats and Directive (92/43/EEC) requires the achievement of “favourable status” of Natura 2000 sites, under a new timeframe of 2015 set out for water dependent SACs in the

WFD. The Habitats Directive may result in the development of different monitoring and standards (e.g. phosphorus in coastal waters) for Natura 2000 sites than those recommended in the WFD. While both the Berwickshire and Northumberland SAC and the Lindesfarne SPA have been assessed as achieving favourable status, measures have been proposed to reduce water pollution from agricultural runoff affecting Budle Bay, through the creation of a “Pollution Action Plan” to be operational “not later than 2012”.

The *Lindisfarne Streams* are included as one of 50 priority catchments in England as part of the *Tweed, Aln, Coquet and Coastal Streams* catchment under the Catchment Sensitive Farming Initiative. The CSFI aims to promote good practice primarily through voluntary farm visits targeted at businesses with high identified risks, previously recorded pollution incidences and low engagement with other stakeholders and stewardship schemes. Funding for actions to reduce pollution risks which do not fall within the current regulatory framework are anticipated to be made through stewardship schemes as part of the Single Farm Payment. Stewardship schemes include farmer payment for activities which may reduce nutrient losses, such as the installation of buffer strips along water courses under Entry Level Stewardship. Increasingly the availability of Natural England stewardship funding is expected to be targeted towards a limited number of sensitive areas such as CSFI priority catchments to achieve best value from a limited fund. Higher Level Stewardship options for resource protection include compulsory completion of manure and nutrient management plans, and payments are received for selected activities including arable reversion to grassland and seasonal stock removal in high risk areas.

To-date CSFI activity has been limited in the Budle Bay catchment, due to the need to monitor and assess the impact of nutrient reductions achieved under the Wastewater Treatment Directive. However, the emphasis placed by the regulatory bodies on agricultural pollution suggests the extension of this initiative to Budle Bay catchment may be highly important to realising the objective of having a pollution management plan in place by 2012.

The Belford Proactive Catchment Flood Solutions project is specifically referred to in the Northumbria RBMP as an “action to deliver water quality objectives” in addition to the Environmental Stewardship and Catchment Sensitive Farming programmes (Environment Agency 2009). This project is described in detail in the following section.

4.3 The Belford Catchment

The Belford Burn above the town of Belford has a catchment area of approximately 5.9 km^2. Catchment elevations range between 207 and 51 m O.D, with over 95% of the catchment at (lowland) elevations below 200 m. The Burn flows approximately east to west, rising from a small linear mire at the foot of a Fell sandstone dip slope, although the issue point of the channel appears to have progressed upslope due to ditching of a valley wetland.

Geology consists mainly of Carboniferous limestones and mudstones with two notable exceptions: the upper slopes around the western watershed are formed of hard impermeable sandstone of the Fell Sandstone formation. In the north-east of the catchment, dolerite outcrops of the Whin Sill occur. In both instances these resistant rocks form steep promontories in contrast to the relatively gentle gradients of the rest of the catchment. Drift cover is extensive, and consists largely of Devensian till which covers the majority of the catchment excluding the elevated Fell Sandstone and Dolerite outcrops. Small areas of deep peat occur in valley depressions in the upper catchment. The annual average rainfall in the catchment has been estimated as 695 mm (Wilkinson *et al*., 2010).

The extensive boulder clay deposits means that deep slowly permeable soils (Stagnogleys) dominate the catchment. These have previously been mapped as belonging to the *Dunkeswick Association* (Jarvis *et al*., 1984). Shallower, better drained soils are found where the Fell Sandstone and Dolerite outcrop (Jarvis *et al*., 1984). Reconnaissance survey conducted as part of this study has confirmed the accuracy of this mapping, although the soils in the upper catchment in particular are dominated by the fine loamy *Brickfield Series* member of the *Dunkeswick Association*, and the well-drained soils over the Dolerite are not previously mapped over a small outcrop in the north west of the catchment. Small areas of peat which occur in valley mires are also not mapped. These discrepancies however are unlikely to have any significant implications for catchment hydrology (which is likely to be controlled by the response of the boulder clay soils which cover over 90% of the catchment area) .

The catchment is agricultural and can clearly be divided into an upper and lower catchment on the basis of land use (Figure 4.7). The upper catchment is predominantly permanent beef pasture, with steeper slopes occupied by rough grazing and coniferous plantation, and a mixture of deciduous and coniferous woodland along the main stream corridor. The lower catchment is dominated by arable rotation – mainly cereals with both grazed and meadow grassland and occasional break crops. The gradient of the channel decreases in the lower section of the catchment despite the fact that the average hill slope gradient is steeper (Table 4.4). This is explained by the fact that although most of the land in the lower catchment is more gently sloping, the occurrence of dolerite outcrops results in localised ridges with extremely steep slopes (maximum 43.3 degrees). The channel gradient although not extreme is relatively steep for a lowland watercourse (Table 4.4)

	Catchment zone		
	Upper	**Lower**	**Total**
Catchment area (km^2)	3.44	2.46	5.90
Mean overland flow length (m)	624 (628)	812 (545)	702 (602)
Mean slope (degrees)	4.49 (3.93)	5.11 (4.97)	4.79 (4.08)
Drainage density (km/km^2)	1.59	1.92	1.73
Mean main channel gradient (degrees)	3.01	2.17	2.64

Table 4.4: Belford catchment characteristics (numbers in parentheses S.D.).

The coverage of most of the catchment with slowly permeable soils means that artificial drainage is extensive. This mainly consists of tile drainage, including areas of permanent pasture as well as arable rotation. In a number of locations piped drainage appears to have incorporated former surface ditches, some of which may have originated from natural channels. Two large piped tributaries join the small burn in its upper reaches, and two further piped tributaries occur in the lower catchment. Only one surface tributary occurs, joining the burn within Belford town (Figure 4.7).

4.3.1 The Belford Proactive Flooding project

The village of Belford is regularly affected by flood events, with 31 properties and several businesses at risk of flooding (return period 200 years) and a number at risk from a flood with return period of only two years. The East Coast Mainline and the A1 pass nearby the town and have also been affected. Recent flood events were recorded in 1997, 2005 and 2007. A number of factors have been suggested for the flooding susceptibility, but chief among them appear to be the constriction of the Belford Burn channel in the town by walls and bridges, and the vulnerable location of properties. The gradient of the catchment is not excessively steep (Table 4.4) and rainfall is moderate. Relatively intensive land use has also been suggested as causing soil compaction and greater runoff (Wilkinson *et al*., 2010).

The small number of properties at risk meant that Belford did not meet the criteria for flood defence funding. Funding was made available from the Environment Agency's Local Flood Levy. In addition to more traditional flood alleviation measures undertaken within the town (small flood walls and channel dredging and widening), part of the funding was earmarked for the construction of *Runoff Attenuation Features* (R.A.F.'s) on farmland upstream of the town, undertaken by Newcastle University in collaboration with the Environment Agency. A secondary objective of the programme was to assess the potential of soft-engineered R.A.F.'s to mitigate sediment and nutrient losses from agricultural runoff, as a well as provide improved wildlife habitat on farmland, thus achieving multiple benefits.

Catchment monitoring commenced in autumn 2007 with the installation of three stream gauges (R1-3, Figure 4.7). A further gauge was installed at the end of 2008 (R4, Figure 4.7). Installation of Runoff Attenuation Features began in August 2008 with the construction of three flow diversion storage ponds adjacent to the main stream channel designed to store part of the bank full discharge. A small reservoir was also constructed for in-channel storage. In 2009 construction of a small number of runoff *interception* features was initiated, consisting of earth bunds constructed in field corners and across natural lines of overland flow concentration, with the objective of providing temporary storage of runoff at flood peaks to reduce flood magnitude. It was hypothesised that where these interception features are located on land at risk of soil erosion and sediment loss they may have significant potential as water quality mitigation options. Six of these features had been constructed by the end of 2010.

The limited progress in feature construction prior to 2009 meant that it was not appropriate to examine the cumulative effect of runoff mitigation features on catchment sediment losses. Presented in this study is a more simple consideration of sediment loss processes and evaluation of the relative potential of engineering structures to mitigate sediment losses.

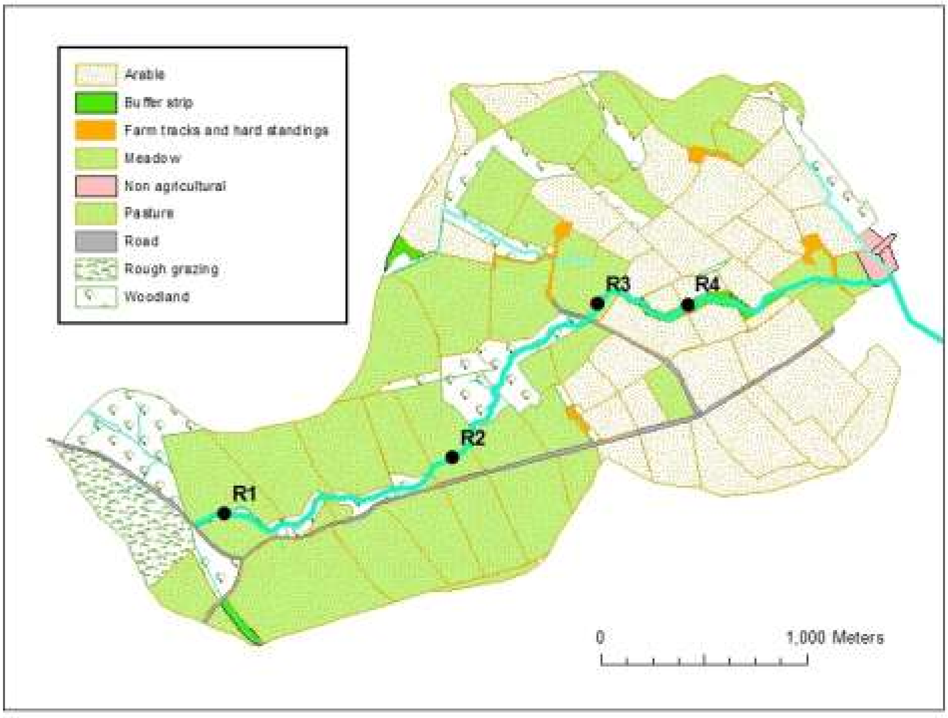

Figure 4.7: Belford catchment land use (2008-2009) and flow gauging points.

4.3.2 Methods

Flow monitoring

River stage is continuously recorded at 5 minute intervals using logging pressure transducers at each of the four gauging stations (Figure 4.7). A limited number of manual discharge measurements have been conducted using a propeller gauge between 2008 and 2010 at each of the flow points to produce rating for these stage recorders. However, given the absence of gauging measurements for high flows, a second estimate has been provided by a lumped hysteric hydraulic model (Ewen and Birkenshaw, 2007) which is used to extrapolate discharges beyond bank full levels (to estimate discharge during floods). The model is calibrated using channel gradient and cross sectional measurements. Comparison of simulated discharge and manual stage recordings (Figure 4.8) suggests acceptably accurate simulation is achieved. The simulated stage-to-discharge conversion is therefore used in further analysis.

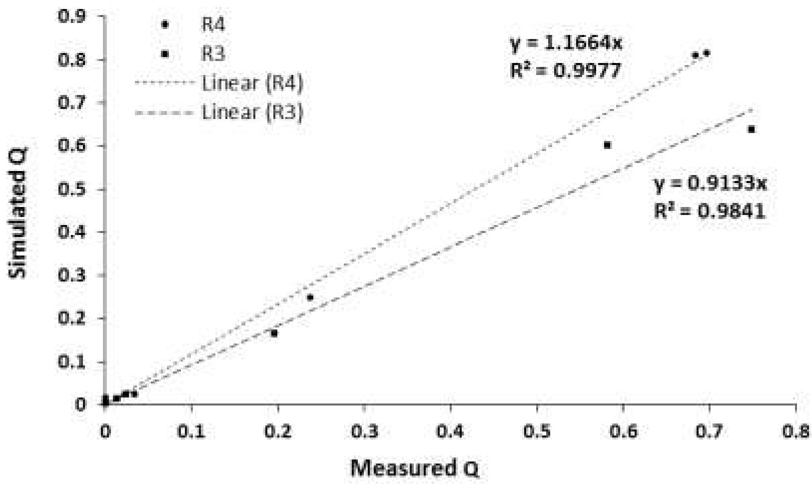

Figure 4.8: Relationship between Belford Burn measured and simulated discharge.

Sediment monitoring

Sediment measurements were collected between January and December 2010 at the catchment outlet, which for the purposes of this study is taken at R4 (Figure 4.7). This decision is justified due to the construction of a small in-stream reservoir downstream of this point which may have significant impacts on the sediment regime. Sediments were collected using a combination of 200 ml manual dip samples collected on a 2-3 week basis and targeted event sampling. The latter were collected using ISCO 7500 automated samplers. The samplers were set to a float switch triggered as stage increased. Twenty four 200 ml samples were collected hourly from the point of programme initiation. Suspended sediment concentration was determined by filtration through a .45 µm membrane filter. A total of 163 samples were collected

4.3.3 Results

Hydrological characterisation

Flow analysis indicates that Belford Burn has a relatively low base flow index and that flow peak response is large relative to both average and low flows (Table 4.5, Figure 4.10). The regime appears to display considerable inter-annual variability in yield, affecting both the annual runoff proportion and the Base Flow Index (Table 4.5). This may be explained by the much more limited runoff response to precipitation during the growing season (April-September) than in winter months (Figure 4.9); in 2010 the precipitation in both Jan–March and October-December was proportionally much greater than in both 2008 and 2009, which resulted in higher annual runoff. Notably during 2010, runoff events are limited to the winter period while in 2009 and 2010 the largest events are in response to extremes daily rainfall occurring in summer months (Figure 4.10).

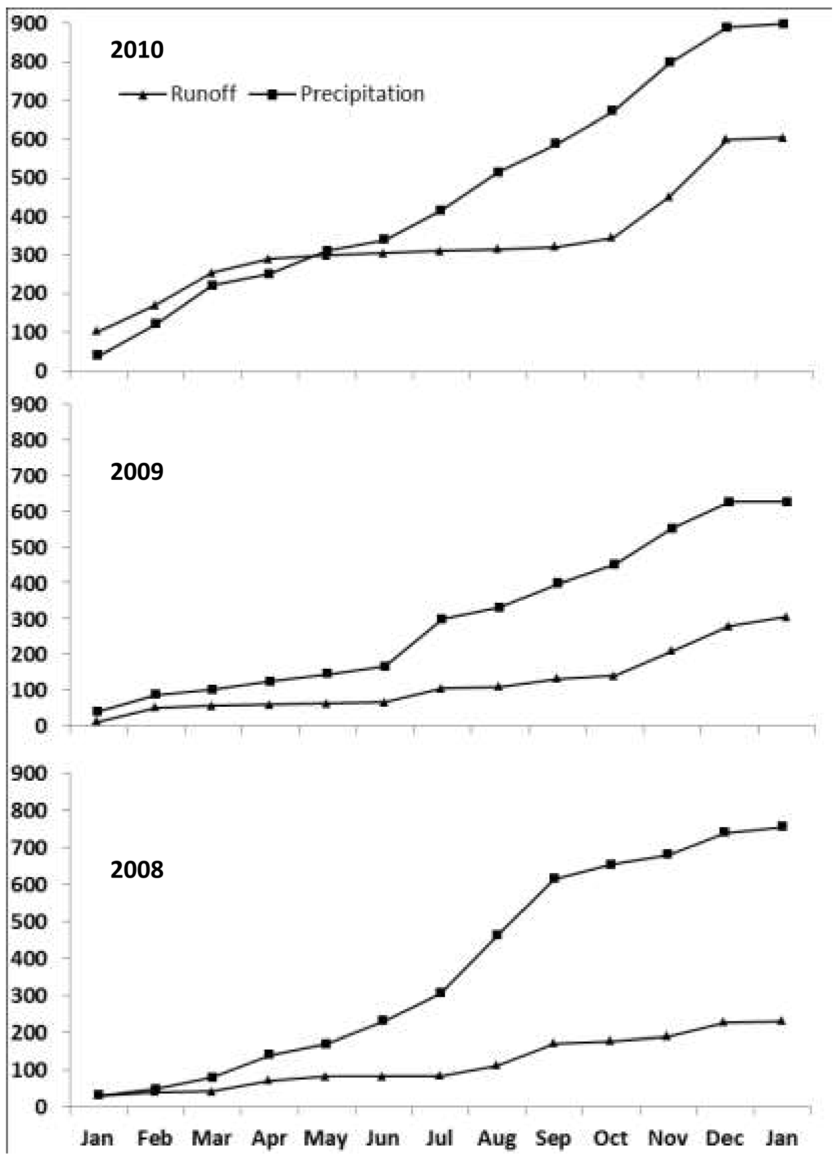

Figure 4.9: Belford Burn cumulative runoff and precipitation (mm) 2008-2010.

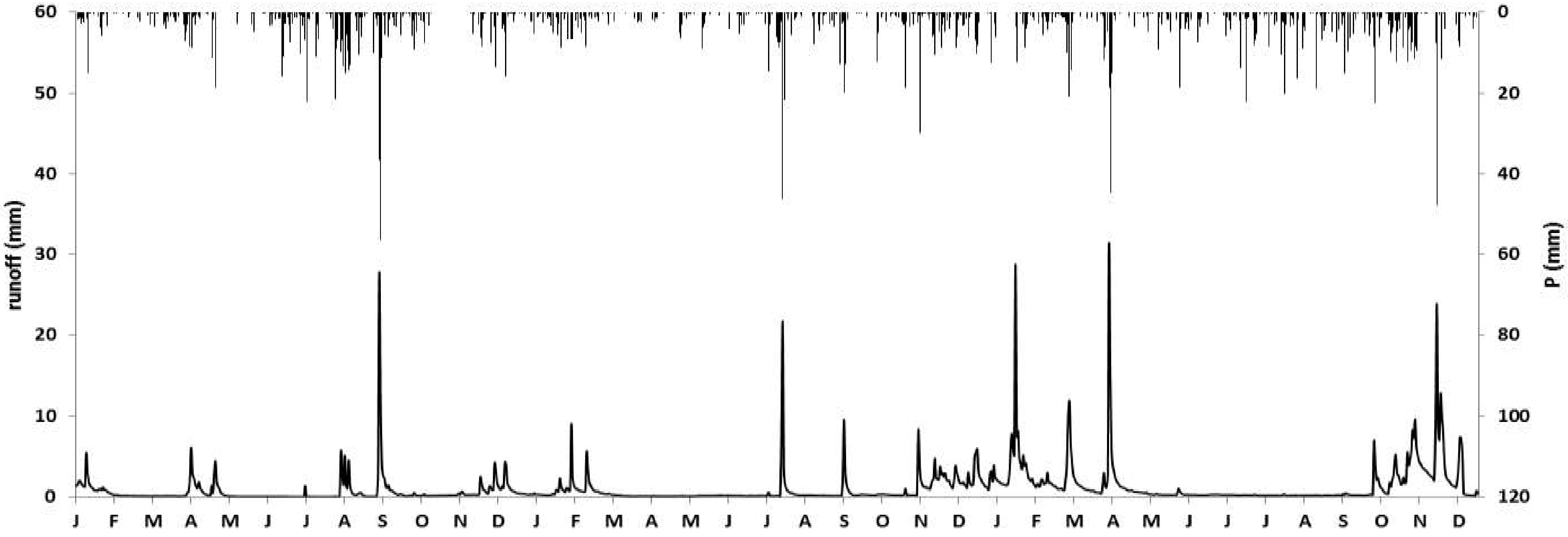

Figure 4.10: Belford Burn flow and precipitation record 2008-2010 (Gauge R3).

	2008	2009	2010	Full record
BASE Flow Index[1]	0.22	0.40	0.38	0.27
Mean daily flow (cumecs)	0.020	0.027	0.052	0.033
Max hourly Q (cumecs)	1.99	2.52	2.80	1.99
Max hourly specific runoff (mm)	2.61	3.30	3.66	3.66
Q_{10} (cumecs)	0.043	0.062	0.136	0.078
Q_{50} (cumecs)	0.0046	0.0061	0.0148	0.0071
Q_{95} (cumecs)	0.0001	0.0018	0.0044	0.0004
Annual P (mm)	756	626	898	760
Annual runoff (mm)	230	305	604	380
% runoff	30.5	48.7	67.3	48.8

Table 4.5: Belford Burn flow parameters 2008-2010 (Gauge R3, Figure 4.7). [1] Institute of Hydrology method (Gustard et al, 1992).

Sediment measurements

Suspended sediment concentrations display a strong positive skew (Figure 4.11); sediment concentration under normal flow conditions is low. High sediment concentrations are limited to a small number of high flow events, which notably are confined to the winter during the 2010 monitoring period (although this is not the case in previous years as described previously). Within these events, peak sediment concentrations and exports appear variable; little relationship between discharge and sediment concentrations can be observed (Figure 4.12). However, it is notable that short duration, high magnitude sediment loads coincide with flow peaks, although the peak concentration varies considerably between events. Extremely high peak sediment concentrations are observed during the largest annual event in March 2010 (Figure 4.13). Peaks of a similar order of magnitude are observed during two smaller events in the succeeding winter (October and November 2011; Figure 4.14).

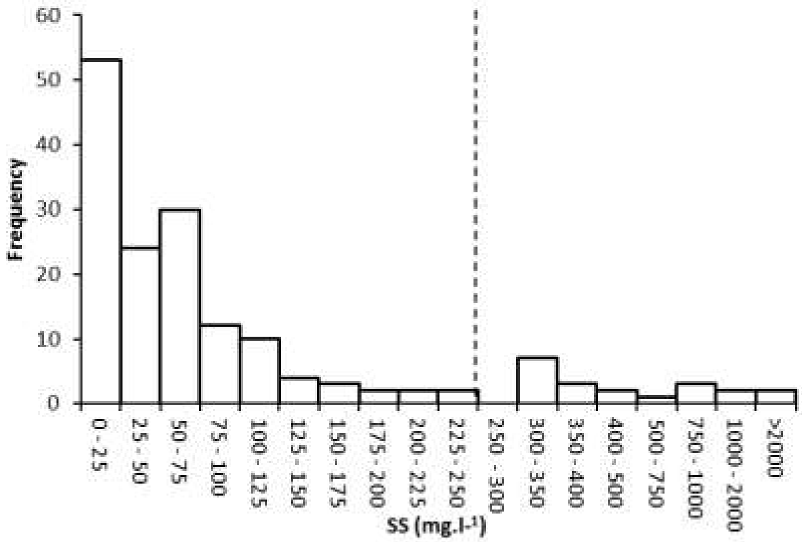

Figure 4.11: Distribution of suspended sediment concentration measurements Belford Burn 2010.

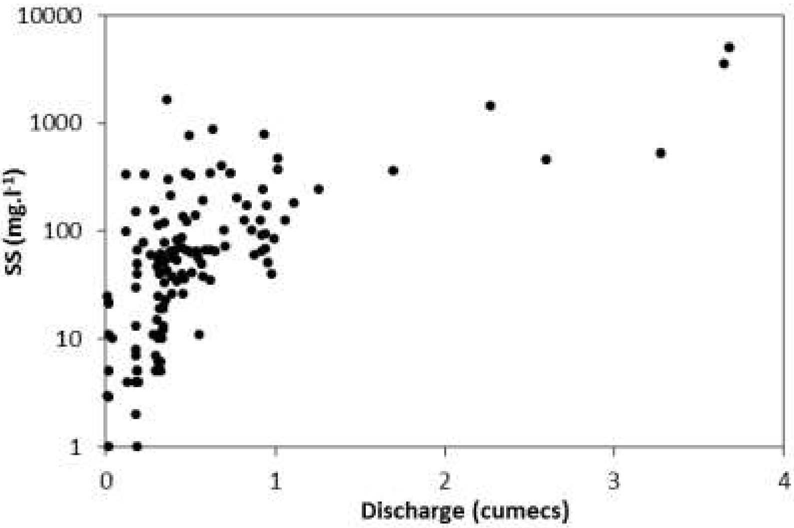

Figure 4.12: Discharge – sediment concentration relationship Belford Burn 2010.

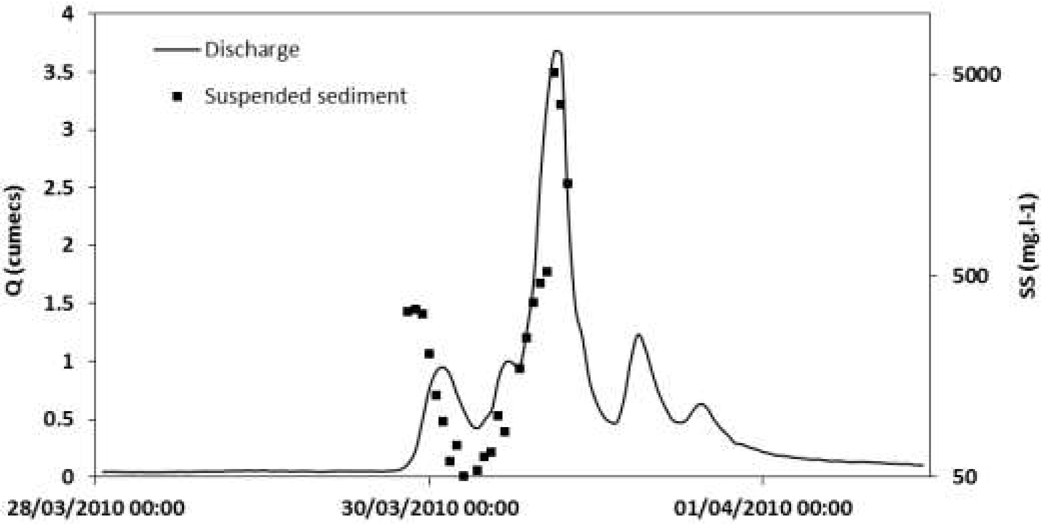

Figure 4.13: Belford Burn Discharge and suspended sediment for event 29th-31st March 2010.

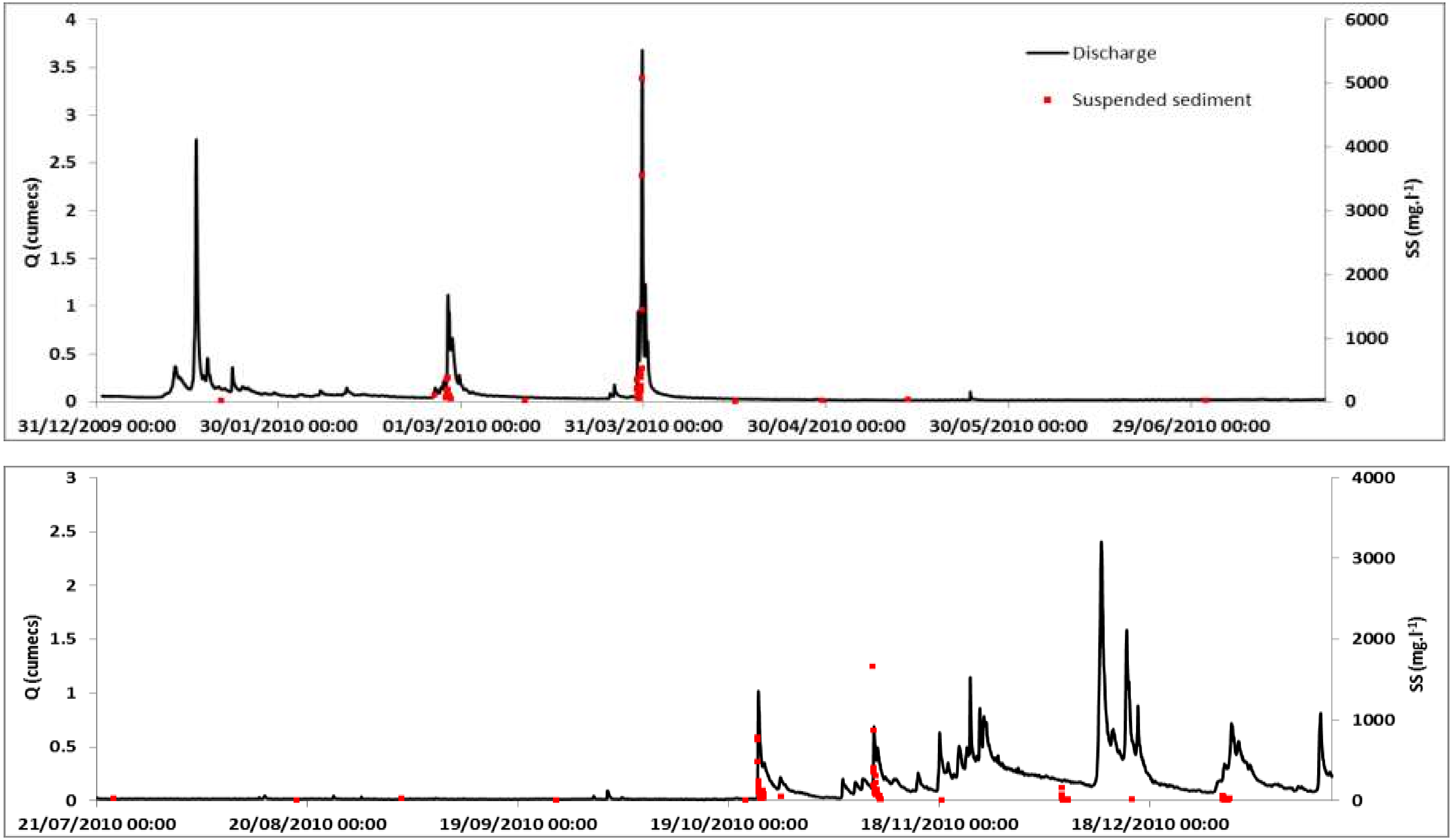

Figure 4.14: Belford Burn discharge and suspended sediment concentrations 2010.

4.3.4 Discussion

Catchment hydrology

Measurements since 2008 reveal that the flow regime of Belford Burn is characterised by low base flow, but with relatively high magnitude events (a 'flashy' regime). One notable characteristic of the catchment is a flow sink located in the mid reach of the Burn (approximately 300 m upstream of R3 gauge - see Figure 4.7). At low flows the entire discharge is diverted to a subterranean route through *karstic* limestone which outcrops in a narrow band at right-angles to the stream channel. This is thought to resurface at a spring to the south of Belford. The result is that the base flow observed at the catchment outlet is almost entirely from the lower catchment. The geology of the catchment is such however, that even allowing for the effects of the sink described above, base flow is likely to be minimal; the dominant soils are relatively deep, but have limited infiltration below approximately 60 cm (Jarvis *et al*., 1984). The active hydrological zone is therefore largely limited to within 1 m of the ground surface, and in the absence of a true groundwater component, the base flow is mainly accounted for by a limited amount of 'slow flow' through sub-soil layers.

During the growing period (March to October) hydrological response to rainfall is limited by the relatively high available soil storage capacity, which results in few summer events and prolonged low flow periods (Figure 4.10). During winter, the available storage capacity is much more limited, as the soils remain at field capacity for long periods. The uniformity of slowly permeable sub-soils in the catchment means that simultaneous runoff generation is likely once soil saturation conditions are reached. Of particular note is that runoff response to moderate rainfall is much more frequent in winter (Figure 4.10). The absence of significant lag times for any part of this small catchment enhances the magnitude of peak flow and leads to flood hydrographs of short duration. Generally it appears that the catchment has a high capacity to absorb precipitation, but has a rapid, high amplitude response once storage capacity is exceeded.

Although some Northumberland rivers have been noted for their susceptibility to extreme floods, this phenomenon applies to those with a significant upland catchment (such as the Tyne and Tees). Lowland rivers such as the Wansbeck and Blyth have been reported to have a relatively benign flood regime (Archer, 1992). This difference is attributable to the much lower extreme rainfall totals for lowland compared to upland Northumberland, together with the effects of unusually rapid snow melt periods in the Pennine uplands which affect the Tyne and Tees but not the lowland catchments. In comparison to the Blyth and Wansbeck, the small catchment area and resultant short lag times of Belford Burn above Belford town mean that relatively high specific hourly discharges are recorded. Therefore while rainfall runoff response is relatively infrequent and event rainfall totals relatively low, the magnitude of specific discharge response to moderately large rainfall events (return period circa 0.5 years) is often high. While Belford Burn has no notable susceptibility to *major* flooding, frequent small local floods would appear a function of the catchment characteristics. Short term flood storage would therefore appear a feasible mitigation strategy to reduce the frequency and magnitude of these regular flood events (Wilkinson *et al*., 2010), although of more limited potential to decrease the magnitude of larger events which are mainly the function of event precipitation totals rather than high runoff percentages.

The *natural* susceptibility of Belford Burn to regular flood events described above is in contrast with similar examples presented for small catchments in the South Downs over permeable geology, where increased frequency of flood events has been clearly linked with land use factors - the decrease in soil infiltration rates resulting in conversion from grassland to arable cropping (Boardman *et al*., 2003). The impermeable geology of the Belford catchment means that *saturation* rather than *infiltration* excess largely controls runoff processes, and while these problems may be exacerbated by compaction caused by agricultural land use, there is no strong evidence that this has played any significant impact in this catchment. Hence while mitigation strategies designed to increase infiltration were regarded as preferable to runoff interception storage in the South Downs (Boardman *et al*., 2003), they are of much more limited potential in Belford.

Effect of artificial drainage

The presence of artificial drainage means that the sub-surface flow response to events is more rapid, and consequently base flow is minimal. In winter, the presence of artificial drainage ensures the soil is largely maintained below saturation, and some storage is therefore available (saturation minus field capacity). However, it is likely that hill slope lateral flow will mean that this storage is rapidly exceeded in receiving sites, resulting in surface runoff or ponding. The drains may therefore serve to decrease low flows (reduced base flow component) and may have some capacity to decrease the magnitude of minor flow peaks through the maintenance of available temporary saturation storage. However, the drains do not appear to have any major impact on the frequency or magnitude of flood events. This has also previously been reported by Robinson (1990) in a major review of the issue.

Sediment

The Belford Burn may be characterised as a lowland catchment with impermeable geology (Walling *et al*., 2007; Cooper *et al*. 2008). Such catchments have been suggested to exhibit relatively high suspended sediment yields due to their susceptibility to runoff and high proportions of susceptible land use. The figures collected by manual sampling suggest low average suspended sediment concentration – below the suggested threshold of 25 $mg.l^{-1}$ used in the Freshwater Fish Directive (78/659/EC) and subsequently adopted as an indicator for the Water Framework Directive (2000/60/EC) in England and Wales. The impacts of agricultural sediment inputs do not appear to have a significant impact on water turbidity given the short duration of sediment runoff periods. However, there is evidence that large infrequent events occurring at susceptible times of year may result in very significant losses of sediment. The event of 30^{th} March 2010 suggests a short period during this flood when very large sediment inputs occur. While there is no replication of events of this magnitude, it does indicate that critical individual events may account for a large proportion of annual sediment export.

The critical nature of events such as 30^{th} March (Figure 4.14) may be explained by overland flow mobilisation of soil, particularly from fields which had recently been cultivated for spring planting. Input from bank erosion cannot be disregarded as a potential source, although no significant bank failures were observed in the catchment following this event. Observations undertaken in the catchment were made prior to the flood peak on the morning of March 30^{th}, and confirmed significant overland flow mobilisation of sediment was taking place (see Chapter 5). Given that the discharges recorded in the river at the flow peak were several times higher than those at the time these observations were made, much higher sediment delivery from agricultural fields at the flow peak would appear probable.

The presence of extensive artificial drainage means that the frequency of small surface runoff events is reduced, and under these conditions tile drains are likely to represent the major pathway for sediment from field areas to receiving waters. Large surface runoff events are relatively frequent but are likely to account for the majority of soil erosion and related sediment losses. Therefore, in evaluating the major sediment source-pathways at the landscape scale, the importance of short periods of concentrated overland flow is highlighted.

As described in section 4.2, there is some (indirect) evidence that the impact of agricultural diffuse pollution on water and sediment quality of Budle Bay may be important if the effects on total inputs of nutrients are considered (rather than dissolved active fractions). Agricultural exports occur with irregularity prompted by runoff events, unlike consented discharges. The crucial issue is the degree of retention within the receptor. Given the low energy coastal environment of Budle Bay, large amounts of sediment are retained, particularly in the near-shore area around the entry points of the freshwater tributaries. It appears possible therefore, that occasional large inputs of sediments and associated nutrients could result in significant damage to the sea grass communities. However, while the runoff attenuation project in Belford Burn has been cited as a management technique to limit agricultural impact on Budle Bay in the River Basin Management Plan (Environment Agency 2009) the potential of this approach remains largely untested and the connection between agricultural runoff and ecological status unproven.

4.4 Summary and conclusions

Belford Burn is susceptible to regular surface runoff events due to its high proportion of impermeable geology. When measured using standard techniques such as the Environment Agency aquatic sampling regime in the Budle Bay catchment, the impacts of agriculture on both suspended sediment and eutrophication appear relatively limited. The direct impacts of agricultural sediment on stream water quality are indeed unlikely to be significant in this type of catchment as the duration of exposure to high concentrations of suspended sediment is extremely short. However, there is a relatively high potential for large sediment loss events when major runoff coincides with susceptible land use periods, particularly winter and spring cultivation periods. This regime means there is the potential for significant negative impacts on substrate quality, bed sediment nutrient release and on downstream conservative receptors such as Budle Bay, where the impacts of sediment phase nutrients may be particularly important. Under these circumstances it may be crucial that mitigation options to control sediment are targeted towards effectiveness at high flows if total sediment exports are to be significantly reduced. The control of erosive catchment runoff through interception and retention may therefore be effective in mitigating water *quality* impacts as well as controlling the negative impacts of water *quantity*. This issue is investigated in more detail in Chapter 5.

Chapter 5: topographic controls of sediment delivery

5.1 Introduction

Evaluation of agricultural sediment loss processes at a catchment scale requires additional consideration of the effect of topography. Previous research has suggested that concentrated runoff along topographically controlled pathways represents a significant erosion process in the UK (Chambers and Garwood 2000; Evans 2002). Studies conducted in other countries have directly observed that a very large proportion of catchment fine sediment losses result from this source-pathway (Poesen *et al*., 1998; Gordon *et al*., 2008) due both to its propensity for high energy mobilisation and a high connectivity with surface water networks.

Chapters 3 and 4 illustrated that while dominant processes at the field scale (such as losses through land drains) may be up-scaled to catchment scale, important run-on/runoff interactions cannot be determined at such scales. At a catchment scale, spatial variability in sediment loss risk may be attributable to landscape-scale features controlling runoff pathways during sediment loss events. The aim of this chapter was to evaluate the relative importance of landscape scale processes, by quantifying suspended sediment losses during runoff events in susceptible areas, and predicting the frequency and timing of such events.

Within the Belford study catchment, remote (GIS) topographic analysis of detailed Lidar terrain mapping had previously been used as part of the Belford Burn flooding project to identify major overland flow routes which may be selected as sites for construction of flow attenuation features (Wilkinson *et al*., 2010). As the interception and storage of runoff along these routes (see Figure 5.2) may also have the potential to mitigate sediment losses, an investigation was conducted to evaluate the role of these pathways in mobilising and transporting soil to the drainage network. This chapter demonstrates that these land *facets* are of significant importance to fine sediment losses, and that greater attention should be given to management of these critical areas in order to mitigate catchment sediment losses.

The results are divided into two main sections; section 5.3.1 describes the quantification of concentrated runoff-derived sediment losses following a minor flood event in 2010. Section 5.3.3 details a simple simulation used to estimate the frequency and magnitude of drainage excess overland flow events in these tile drained soils, and therefore evaluate the relative contributions of the drain and surface pathways to suspended sediment exports. Section 5.3.2 provides visual evidence to verify the findings of the earlier sections.

5.2 Methods

5.2.1 Field monitoring and sampling

The catchment was monitored from October 2009 to May 2011. During this period land use patterns were recorded, and sites at risk of erosion were identified. Based on the standard Defra soil erosion risk assessment field method (Defra 2005) few fields were at high risk of erosion (Figure 5.1). This is attributable to the fine texture of most top soils, and to the interaction between topography and land use; steeply sloping land (>7 degrees) where landscape susceptibility is high was generally under low susceptibility land uses (permanent pasture or woodland). Three fields identified as high erosion risk were underlain by medium textured top soils on steep slopes under winter cereals in 2010-11. Particularly high) was associated with the late sowing (mid-October) of winter cereals on two fields (Figure 5.1). Late sowing means the degree of crop cover established prior to winter dormancy is less than 30%, and the land is therefore at continued high risk over winter rather than during the short establishment period associated with early sown winter or spring cereals (Defra, 2005).

Qualitative observations and photographic evidence were collected at regular intervals throughout the period October 2009 – May 2010. During the study period major runoff events were recorded on the 30th March and 11th December 2010.

Quantification of erosion from a single event

Following the event of December 11 2010, significant channel erosion was observed in three arable fields within the catchment referred to hereafter as *Lady's Well*, *West Hall* and *F3* (Figure 5.2). Two of these fields were those previously identified as high risk using the standard Defra method (Figure 5.1). Quantitative assessment of soil erosion, deposition and sediment loss was made using the methods described below. Field measurements were collected between the 13th and 26th January 2011.

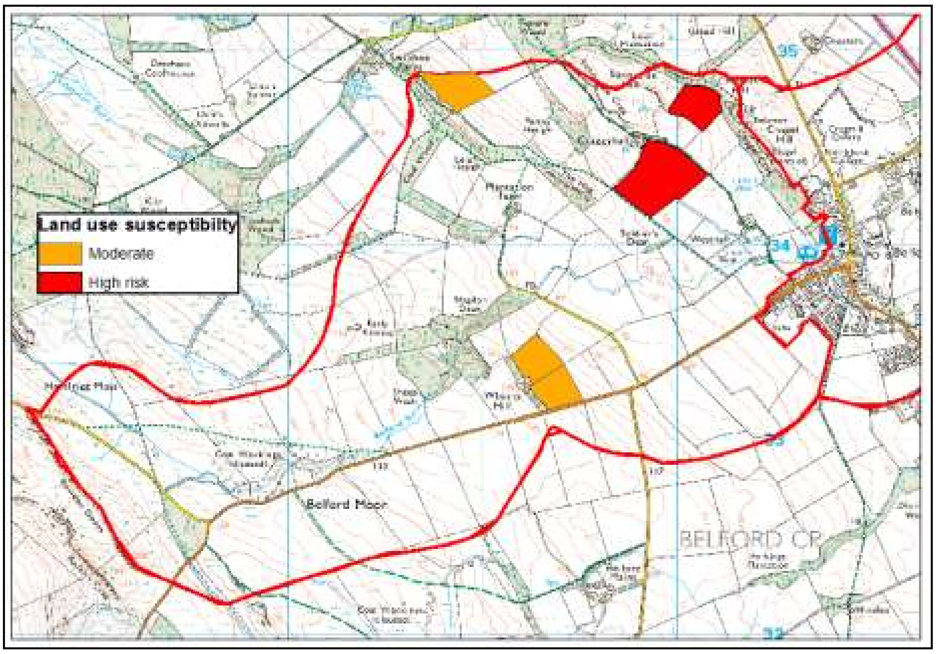

Figure 5.1: Fields at moderate or higher risk of soil erosion 2010-11 rotation.

(After Defra, 2005 field assessment method)

Field descriptions

Lady's Well

The field is *v-shaped* with average hill slope gradients of approximately 6 degrees. Maximum hill slope length is 120 m. The central valley *thalweg* (the line following the lowest part of the valley) has a gradient of approximately 4 degrees. The valley originates to the north in a small drained headwater wetland at the foot of a steep dolerite outcrop (under permanent pasture). Both the upstream catchment and the study field are tile drained; a six inch (15.2 cm internal diameter) leader drain is located along the course of the valley thalweg, and outlets at a surface ditch below the adjoining downslope field (Figure 5.2). The upstream catchment area of the field is approximately 4.1 ha, the area at the outlet from the field 9.9 ha and at the tile outlet to the surface ditch 15.6 ha. The maximum flow path length along the valley thalweg within Lady's Well Field is 230 m. The field was in arable (winter wheat) cultivation throughout the monitoring period.

A trial mitigation bund designed to retain flood runoff (with the secondary objective of reducing sediment loss) was constructed at the downslope end of the thalweg by the farmer on the 21st September 2009. In December 2010 a large rill formed along the valley floor (Plate 5.1). In addition to the rilling, sizeable *blow-out* holes were observed along the thalweg (Plate 5.1) apparently in response to pipe surcharge. Sediment deposits of predominantly fine material (fine sand, silt and clay) material was observed in the area covered by the temporary pond located behind the retention bund. A limited amount of small-scale hill slope rilling along downslope vehicle wheelings was observed (Plate 5.1), but these rills were too small to quantify.

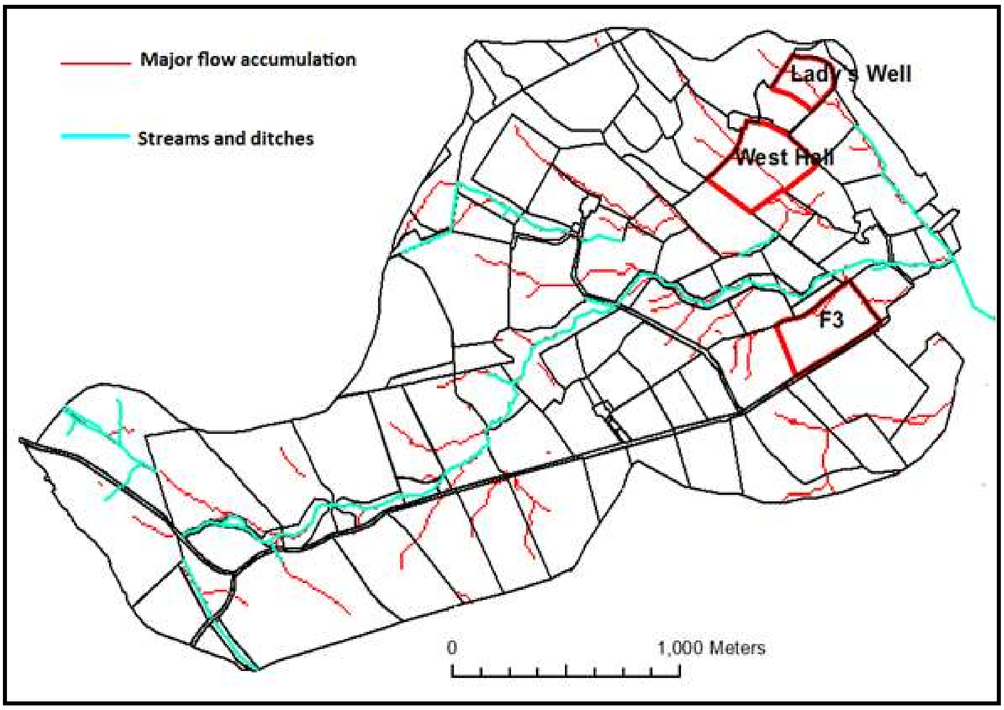

Figure 5.2: Belford Burn catchment eroded fields 2011.

Flow accumulation estimates are computed from LIDAR topographic data using ArcGIS software (*D8* method).

West Hall field

The field is v-shaped with moderately sloping hill slopes of average gradient of approximately 5 degrees. Maximum hill slope length is approximately 180 m. The field contains a relatively large-scale valley depression, which originates over 500 m upstream, and crosses five field boundaries. The average gradient of the thalweg across the field is approximately 2.5 degrees. The maximum flow path length along the valley thalweg within the field is 280 m and is underlain by a leader tile drain. This originates in the field directly upstream, where a nine inch (22.5 cm internal diameter) open-inlet clay tile receives the drainage from a surface stream from a steeply incised valley with coniferous woodland (Penny Heugh Wood – see Plate 5.2). The 9 inch tile drain continues some distance downstream (as observed at open inspection points), where it discharges directly to Belford Burn. The catchment area upstream of the field is 56.8 ha and at the field outlet is 65.7 ha. The field was in winter cereal production during the monitoring period (barley 2009-10, wheat 2010-11).

Surface runoff was observed in March 2010, and inferred from field evidence in December 2010 when major channel erosion occurred, creating a broad (approximately 3 m width) shallow channel. The presence of coarse gravel (>6 cm^2) sized stones deposited within the channel on the second of these occasions indicates considerable flow energy (Plate 5.2). A large sand fan formed at the lower margin of the field where the topography levels, although further patches of sandy deposition were observed for approximately 50 m in the downstream field. Vehicle wheelings on adjoining hill slopes did not display significant rill erosion.

Field F3

This field is gently sloping (approximately 3.5 degrees) and relatively flat in plan compared with West Hall and Lady's Well fields. The upslope contributing area is also gently sloping. Despite the absence of major convergent topography which may indicate susceptibility to concentrated flow erosion, a minor valley depression does occur in the north eastern corner of the field extending into the adjoining upslope field. The upslope catchment area is 5.7 ha (although potentially larger when runoff from a larger contributing area exceeds the capacity of road drains which intercept storm runoff from steep upslope fields). Catchment area at the flow outlet point at the base of the field is 8.4 hectares. The approximate maximum flow path length within the field is 390 m. The presence of sub-surface drainage is not confirmed, but the presence of a tile outlet to a surface channel approximately 70 m downslope in the adjoining field suggests tile drainage to be present. The field was in grazed pasture in 2009-2010 and winter wheat in 2010-11.

Significant rill erosion in along the eastern margin of this field was noted in January 2011 (Plate 5.3). The rilling appeared to originate from run-on from upslope fields, and erosion was exacerbated by the channelling of this run-on along downslope vehicle wheelings. However, rilling was not confined to the wheel tracks, indicating considerable flow volume. A sand fan deposit occurred at the lower margin of the field.

Field measurements of erosion

Two methods of estimating soil and sediment loss were deployed as described below:

Method 1: Channel mass measurements

Channel soil loss was determined by measuring rill cross sectional areas at representative points along the entire rill length at 10 – 20 m intervals. Total volumetric soil loss was calculated as the sum of the estimated channel volume for each interval-defined section. In Lady's Well field the clearly defined erosion channel (see Plate 5.1) meant that an accurate measure of channel volume could be achieved. For West Hall field similar channel volume measurements were collected, although the broader, less-defined channel (see Plate 5.2) meant that this data may be less reliable. A second estimate of erosion is therefore provided using method 2 below and the results compared. The complex pattern of rilling in Field F3 (see Plate 5.3) meant that channel volume estimates were not feasible; erosion estimates are based on the figures obtained using method 2 only.

Method 2: Depositional mass measurements

Depositional volume was estimated by measuring depth of deposited material using intensive grid survey (1 measurement per 5 m^2). In two of the fields (West Hall and F3) these deposits consisted of sizeable fresh sand fan deposits at the lower field margins. In Lady's Well field sandy deposits were found to occur in irregular patches along the rill length, which made volume estimation impossible. However, the presence of a field bund mitigation feature resulted in fine sediment (mainly clay and silt) deposition in a definable area behind the mitigation pond (plate 6.1). Depth of this material was measured as described above to provide an estimate of the fine material retained within the field.

Plate 5.1: Lady's Well 2011

Clockwise from top left: gulley initiation at tile blow-out hole; hill slope erosion; channel gulley looking north; sandy deposits upstream of bund; tile blow out hole at base of field; silty pond deposits looking north from bund.

Plate 5.2: West Hall 2011: Clockwise from top left: Recessional flow following major runoff event (looking south); channel erosion (looking south east); sand fan deposit at base of field (looking north – Penny Heugh wood in background); gravel bar in erosion channel (looking north).

Soil sampling

Three representative soil samples were collected from each of the three fields by dividing the rill into equal upper, middle and lower sections and bulking 10 sub-samples of undisturbed soil (0-10 cm) derived from within 5 m of the rill channel from each section. Three samples of depositional material were also collected. Due the lack of field bottom sand fan deposits in In Lady’s Well field, deposits were sampled from within the channel along its route, subdivided into upper, middle and lower sections as with the soil sampling strategy. From West Hall and F3 fields, a mixed representative sample (10 bulked sub-samples) of the sand fan deposit material was collected. A mixed representative sample of the material trapped behind the mitigation feature in Lady’s Well field was also collected. All samples were stored in sealed containers at 2 degrees prior to analysis.

Representative stone free bulk density was estimated from the mean of three cores collected from undisturbed areas adjacent to the upper, middle and lower sections of the erosion channel in each of the three fields. Mass of soil loss from erosion channels (method 1) for Lady's Well and West Hall fields was corrected by subtracting stone volume. Due to the shallow deposition depth within the Lady's Well mitigation pond, bulk density sampling was not possible. Less precise estimates of deposition mass are made using the stone free bulk density of undisturbed soil samples within the field.

Plate 5.3: F3 2011: Top: downslope rilling; bottom: sand fan at base of field.

5.2.2 Laboratory analyses

Soil loss estimates were converted to sediment loss by subtracting the proportion of in-field deposits from the total erosion. Estimate of the exported soil mass was made by comparing the particle size distribution of undisturbed soil and deposited material. However, because a significant proportion of fine primary particles may exist as part of water stable soil aggregates, *absolute* particle size may lead to an inaccurate estimate of the true particle size distribution derived by erosion processes. To counter this problem, three measurements were made of particle size distribution following different disruptive energy inputs; a gentle shaking procedure hereafter referred to as *actual* particle size distribution, a vigorous shaking procedure hereafter referred to as *effective* particle size distribution and a vigorous shaking following the addition of a chemical dispersant hereafter referred to as *absolute* particle size distribution. Preliminary evaluation of the estimate provided by each method was used to select the most appropriate for use in sediment loss calculations.

The method was based on the standard ADAS laboratory method for estimation of dispersible clay content (MAFF, 1982). The standard method applies the test to ground air-dry soils, which are rapidly wet by immersion in water. This provides an easily replicated test of relative soil stability. The drying-rapid wetting pre-treatment may be justified on the grounds that the maximum water erosion susceptibility will occur when precipitation contacts dry surface soil causing slaking and dispersion. However, in the current context this is likely to misrepresent channel erosive energy; moist or wet soils are not subject to the same slaking energies as air-dry soils, and the air-drying procedure is therefore not a good approximation of winter conditions. Kjaergaard *et al*. (2004) illustrated that rapid wetting of air-dry soil generally results in higher release of dispersible clay than the same soils at lower matric potentials, while conversely showing that for soils with moderate to high clay contents, air-drying resulted in increase in cohesive strength, and *reduction* in clay dispersion. Measurement at field moisture content was therefore regarded as a more appropriate representation of processes in this study. As samples were collected within 4 weeks of the erosion event during a particularly cold mid-winter period, soil moisture is assumed to be at field capacity, and particle disruption susceptibility is therefore representative of field conditions under which the erosion occurred.

***Actual* particle size distribution**

Approximately 10 g (oven dry weight) of field moist sample were weighed and added to 200 ml of deionised water in a 1000 ml glass measuring cylinder, then made up to 1000 ml. Samples were left to equilibrate for 30 minutes before being manually inverted end-over-end for 1 minute at a rate of 1 inversion per second. Samples were taken at designated time intervals according to the standard pipette method for particle size distribution (Avery and Bascombe, 1982) to determine the content of 63-40, 40-20, 20-10, 10-2 and <2 micron particles. Following collection of pipette samples, the inversion procedure was repeated and the re-suspended contents gently screened through a 63 micron mesh sieve. Sieves were oven dried and the contents transferred to a nest of sieves of 2000, 600, 202 and 106 micron mesh and screened. Weights of the sand-sized fractions were recorded.

Effective **particle size distribution**

Approximately 10 g (oven dry weight) of field moist sample were added to 200 ml of deionised water and shaken on a reciprocal shaker for 1 hour. The contents were then transferred to a 1000 ml cylinder and the procedure described above repeated.

Absolute **particle size distribution**

Approximately 10 g (oven dry weight) of field moist sample were added to 200 ml of deionised water and 10 ml of 1 M sodium hexa-metaphosphate (Calgon® flake) and shaken on a reciprocal shaker for 1 hour. The contents were then transferred to a 1000 ml cylinder and the procedure described above was repeated.

5.3.3 Statistical analyses

Differences in the fine fraction (<106 µm) content of soils and in-field deposits were compared between fields using one way Analysis of Variance, and differences between individual fields using Tukey's post-hoc test.

5.3 Results

5.3.1 Estimation of event erosion

	Actual			***Effective***			***Absolute***		
	Sand	**Silt**	**Clay**	**Sand**	**Silt**	**Clay**	**Sand**	**Silt**	**Clay**
Lady's Well deposits	86	9	5	81	12	7	86	8	6
Lady's Well soil	87	11	2	65	28	8	64	24	11
Mitigation pond	39	45	16	21	55	24	31	50	20
West Hall deposits	94	4	2	93	4	3	91	5	3
West Hall soil	76	17	6	67	21	12	68	17	15
F3 deposits	97	1	1	94	5	1	92	5	3
F3 soil	90	8	1	71	22	7	60	29	11

Table 5.1: Particle size distributions for eroded fields using increasing dispersive energy.

The outputs of the three dispersion techniques indicate that generation of fine particles (silt and clay) is notably lower for the *actual particle size* low energy method. This finding applies to soil samples only, as the deposited material consists primarily of sand grains and is therefore unaffected by increasing energy (Table 5.1). Increasing dispersive energy with the *effective particle size* test releases material from sand-sized aggregates to aggregates and primary particles of silt and clay size. The *absolute particle* size test demonstrates a relatively limited further release of sand and silt sized particles to primary clay particles with the addition of a chemical dispersant.

Detailed analysis of particle size contents (Figure 5.3) indicates that in-field deposits are enriched in particles larger than approximately 100 µm in diameter (medium and coarse sand) relative to the soil from which they are derived, indicating primary particles and stable aggregates of fine sand, silt and clay size have been selectively transported from the fields.

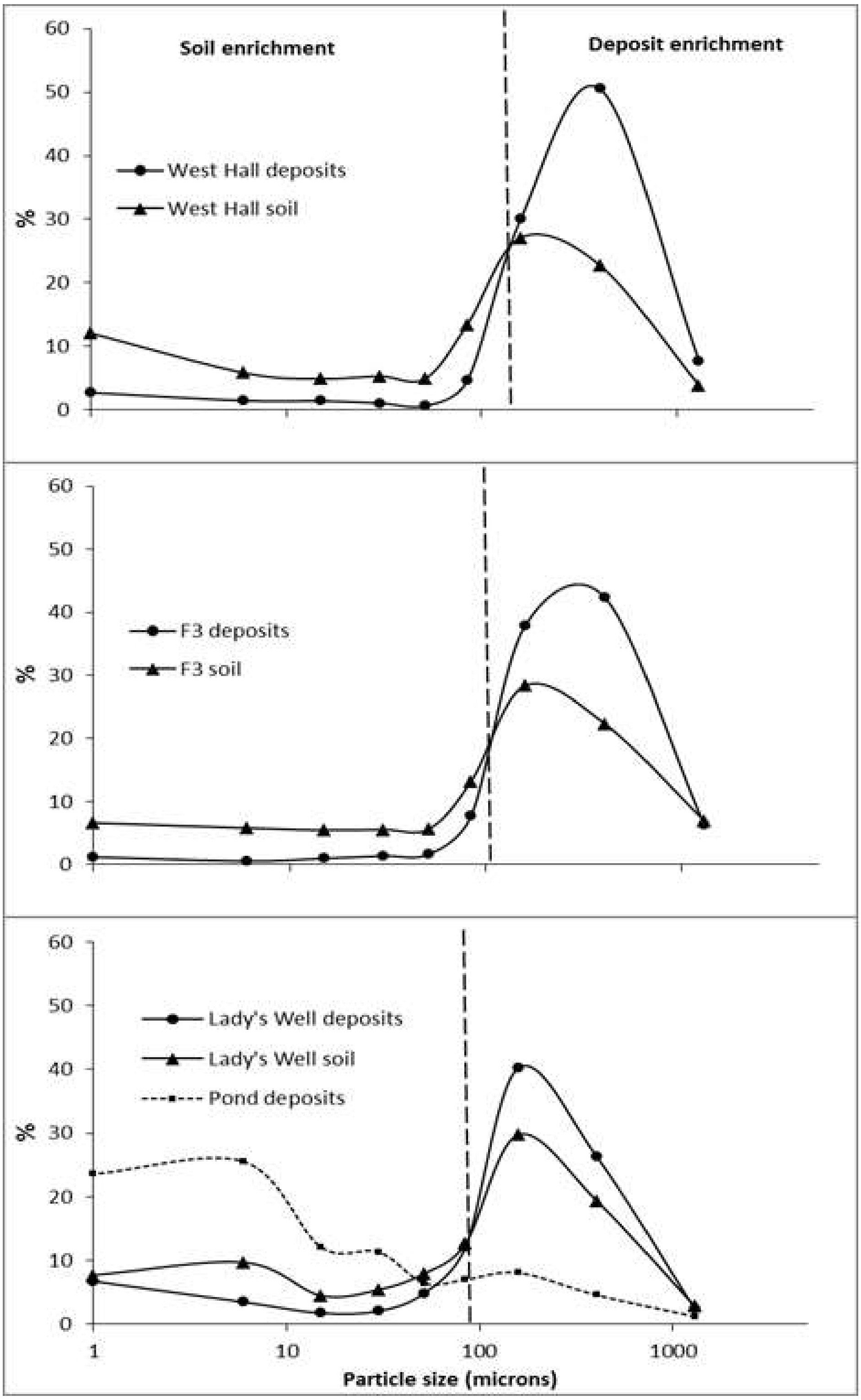

Figure 5.3: Paired comparison of effective particle size of soil and in-field deposits for 3 study fields.

Sediment loss and soil erosion calculations

Calculations of soil loss are made using the *effective particle size* test, which appears more representative of the fractionation effects of moving water than the *actual particle size*. The proportion of effective particles less than 106 microns (fine sand, silt and clay) is selected as the measure of fine material loss, an approximation of the findings presented in Figure 5.3. Subtraction of the proportion of effective fine particles of in-field deposits from the content in the parent soil provides an estimate of the proportion of the total mobilised soil material which is exported from the field (Table 5.2). Material deposited in the Lady's Well field demonstrates a significantly higher fine content than either of the other two fields despite the fact that there was no significant difference in the fine content of the soils in the three fields ($F = 0.37$; $p = 0.819$).

	<106 µm%		
	Soil	**Deposits**	**% loss**
Lady's Well	48.0a	31.1a	17
West Hall	46.3a	11.7b	35
F3	42.3a	13.5b	29
Mean	45.6	18.8	
s.e.	3.74	3.08	

Table 5.2: Mean fine particle loss (effective).
Column means with same letter are not significantly different (Tukey's $p>0.05$).

The % loss of effective fine particles (Table 5.2) is used to calculate soil loss from each field. The presence of a well-defined erosion channel in the Lady's Well field made it possible to gain a quantitative estimate of soil loss volume. This method was necessary in this instance, as the occurrence of continuous deposition as well as erosion made quantification of depositional volume impractical. In West Hall and F3 fields, the presence of well-defined lower field sand fan deposits made measurement of deposit volume possible. The implications of the method of estimation are discussed later in the chapter.

The estimated sediment loss from West Hall field was an order of magnitude greater than that for the other two fields (Table 5.3). Comparison of the two methods of estimation for West Hall field suggests an acceptable level of agreement, with a lower estimate provided by the deposition method (method 2). Estimates for Lady's Well field indicate that the total eroded soil mass represented by the fine material retained in the mitigation pond is in excess of the soil eroded from the gulley channel.

Field	field area (ha)	In-field deposition (t)	Fine sediment loss (t)		Soil erosion (t)[1]	
			Total	per ha	Total	per ha
West Hall	10.31	10.30	4.97 (7.73)[2]	0.48 (0.75)[2]	15.27 (22.33)[2]	1.48 (2.17)[2]
F3	9.01	1.68	0.63	0.07	2.31	0.27
Lady's Well.	4.15	-	0.17	0.04	0.92	0.22
Pond deposits (LW)	4.15	0.99	-	-	1.43[3]	0.35[3]

Table 5.3: Erosion and sediment loss estimates.
[1] Defined as in-field deposition + sediment loss.
[2] Figures based on estimate of channel volume.
[3] Estimate based on the difference in the content of particles >106 µm between soil and pond deposits.

5.3.2 Further observations of sediment loss processes

A previous large runoff event occurring 30th to 31st March 2010 was observed and recorded to provide visual evidence of flow and erosion/sediment loss processes (Figure 5.4, Plate 5.4). The results support the findings reported above that concentrated flow is a prevalent sediment delivery process during large runoff events. Observation of the West Hall catchment confirmed that run-on to the field originated from the field above (Photo 1, Plate 5.4) at the open tile inlet of the woodland stream. The secondary flow path to the north-east suggested by flow accumulation routing (Figure 5.2) is also confirmed in this event (Photo 2, Plate 5.4). This evidence also confirms that the runoff is largely unimpeded by field boundaries and is connected with the permanent surface water network (photos 2-4, Plate 5.4).

In the Lady's Well catchment runoff was found to be generated only from the lower section of the study field at the time of observation, but the mobilisation of significant quantities of sediment is visually confirmed (Photos 5 & 5a, Plate 5.4). The creation of a settlement pond behind the mitigation dam is also confirmed, although it is clear that a considerable sediment load remains in suspension through the overflow pipe, and that this sediment is carried in suspension to the surface network (Photos 5a & 6, Plate 5.4) despite the fact that it has travelled a distance measured at 195 m across the downstream pasture before reaching the ditch. The final photographic evidence (Photo 7, Plate 5.4) demonstrates that relatively small pathways contained within single fields may also be accurately identified by remote-sensed flow routing (Figure 5.2).

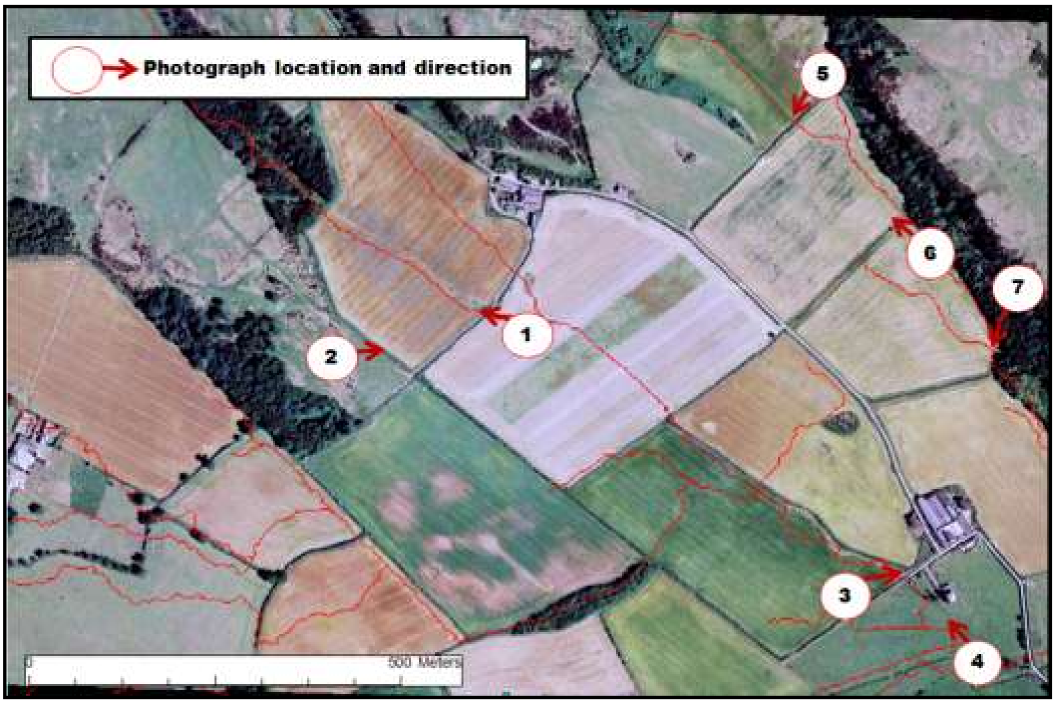

Figure 5.4: Photograph locations for event March 30th 2010.

Further observations of note were made during the March 2010 event. Firstly it was observed that although the major routes of concentrated overland flow and sediment delivery could be accurately predicted, other minor routes were either not identified or were spatially inaccurate. As in the bottom left photograph of plate 5.5, run-off in landscape areas of low natural convergence can be easily diverted by man-made flow paths such as field edge open furrows. In other instances, these runoff pathways were either absent (false positives) or not identified (false negatives – for example top right and left photos Plate 5.5). The event in question would appear to have resulted in sufficiently high sediment concentrations to have impacted the sensitive downstream receptor at Budle Bay (bottom right, Plate 5.5).

Plate 5.4: Runoff observations March 30th 2010.

Plate 5.5: Miscellaneous catchment pictures March 30th 2010.
Clockwise from bottom left: rill erosion along open furrow caused by field run-on; muddy runoff from freshly rolled field; muddy runoff from same field spilling to road (and storm drain); sediment plume meeting tidal water, Budle Bay.

5.3.3 Modelling drainage exceedance overland flow

The occurrence of erosive concentrated overland flow in vulnerable landscape positions has been confirmed in the previous section. The connectivity of these pathways during events means that they are both major sources and pathways for sediment to surface waters, and may therefore represent a significant sediment loss issue. These surface pathways occur despite the presence of an intensive sub-surface drainage system. The drainage system is likely to represent the most frequent route for runoff and sediment (during minor rainfall events) although it may not be the most important pathway with regard to total annual sediment yields for two reasons; firstly the majority of the flow in major events (which account for a large proportion of sediment losses) may occur via the overland flow route. Secondly significant overland flow unlike drain flow has the capacity to mobilise as well as transport sediment, and as demonstrated in the previous section, channel erosion may represent a much larger risk than hill slope erosion, particularly for fine textured soils. However, these overland flow events are infrequent and their magnitude as sediment loss events depends on their coincidence with susceptible land use periods, particularly periods of low crop cover in a cereal-based system. The return period of these events and its interaction with land use susceptibility control the degree of risk. Therefore an investigation of surface runoff generation frequency and magnitude is critical to evaluating this risk.

The interaction between the drainage system and overland flow generation is illustrated using the West Hall field catchment as a case study, and then compared with that of the Lady's Well field catchment. Two outlet points are selected for comparison in West Hall catchment; the open inlet for the woodland stream above the upstream field (catchment area 33.4 ha), and the point at which two flow paths converge in near the upstream boundary of West Hall field (catchment area 56.8 ha, Figures 5.4, 5.5). For Lady's Well Field the downstream outlet (catchment area 9.9 ha) is preferred due to the large influence the catchment area of the field appears to have on the generation of drain exceedance (meaning consideration of upstream 'run-on' only will significantly underestimate erosive potential).

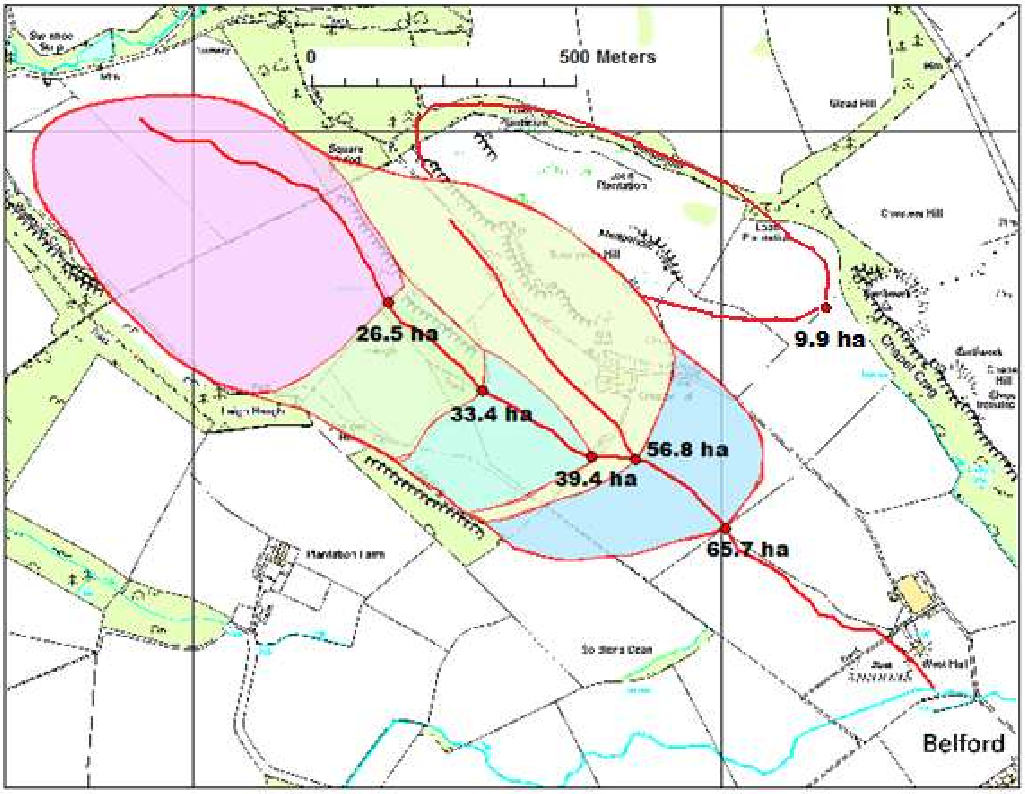

Figure 5.5: Catchment area at selected flow points in West Hall catchment and Lady's Well catchments.

The approach taken to simulating runoff processes assumes the leader drain is the outlet for all sub-surface drainage, that the flow 'bottleneck' occurs at this point, and not due to infiltration excess (Hortonian) overland flow on hill slopes. The point along the leader at which pipe flow capacity is exceeded by measured catchment runoff may therefore be taken as the point of overland flow initiation. The volume of overland flow may be simply estimated given catchment outlet runoff data and estimation of the pipe flow capacity. The assumption that all runoff occurs either through the drainage system or as overland flow seems valid in this case; the location of the drain in a valley floor with low longitudinal gradient (measured as 2%) means that soil storage in the valley bottom between pipe capacity and surface saturation represents a very small proportion of catchment specific runoff, and may be disregarded without causing significant calculation errors. Lateral flow through the soil profile along this course is expected to be low given the gradient and the poor structure of these soils below approximately 40 cm. In any case, pore permeability will result in return flow in the event of drain surcharge which would effectively impede upstream lateral flow (and force it to the soil surface).

Direct measurement of the catchment outfall were not available, which meant that specific discharge had to be extrapolated from flow recordings made upstream on the main stream channel (note the catchment area for the gauging points does not include the case study sub-catchment – see Figure 4.7). Data for two gauging stations is available; the upstream of the two (R3) has a three year record, while the downstream record (R4) is two years. Comparison of the two stage records indicate that event response is closely correlated (Figure 5.6), and that therefore the upstream gauge may be effectively used as a surrogate for the downstream. Greater uncertainty as to the catchment area of the lower gauge potentially variable during events due to low-flow drainage diversions returning to their natural course) means the use of the upstream gauge is considered more reliable.

The validity of extrapolation of the data to the study sub-catchment may be justified with reference to land use, geology and soil type; the entire catchment is in agricultural use, and the proportion of impermeable surfaces is extremely low, meaning the land use (mixed livestock, woodland and winter cereals) of the West Hall catchment is entirely representative of the catchment as a whole. Over 90% of the Belford catchment is covered by fine textured slowly permeable till, and the resultant slowly permeable soils with sub-surface drainage dominate both the whole catchment and the West Hall and Lady's Well catchments alike. The exceptions are areas of shallow coarse loamy soils which occur over the Fell sandstone outcrop in the west of the main catchment, and over the Whin Sill dolerite outcrop on the steep slopes around Penny Heugh wood within the West Hall sub-catchment. These soils are have lower storage capacity, and may elicit a more rapid runoff response than the deeper fine textured soils on the gentler slopes. By comparison with the whole catchment, the proportion of these soils in the study sub –catchment is higher, which may mean that specific runoff is greater and response quicker. However, this minor inconsistency is not a major limitation, and at least indicates that the use of extrapolated figures is unlikely to represent an overestimation of runoff response. The figures are likely to be less transferable to the Lady's Well catchment, as the smaller area means heterogeneity of soils and topography may be greater than the generalised catchment (although the soil type is confirmed to be the same – *Dunkeswick Series*).

Calculations

The internal pipe diameter (22.5 cm), gradient (2%) and roughness coefficient for clay ware tiles were used to estimate the maximum flow capacity of the pipe using published data for drainage system design (Castle *et al.,* 1984). The calculated flow capacity was 42 $l.s^{-1}$, equivalent to 151.2 $m^3.hr^{-1}$ or 15.12 $mm.hr^{-1}.ha^{-1}$. The critical catchment area for pipe capacity exceedance at a given discharge may therefore be calculated as

$$15.12/\text{observed runoff } (mm.hr^{-1}) \qquad [1]$$

Hence the critical catchment area for pipe surcharge and overland flow generation for 1mm.hr^{-1} is 15.12 ha, for 2 mm 7.56 ha and 5 mm 3.02 ha.

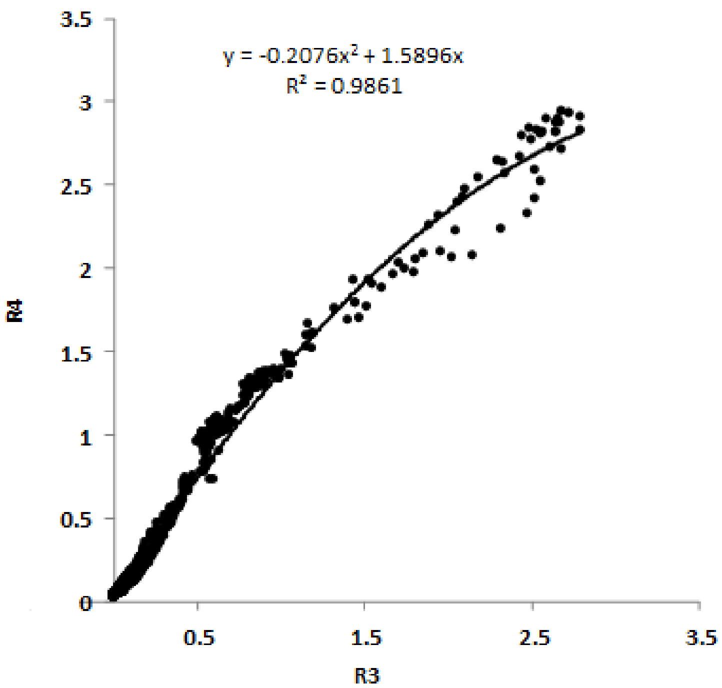

Figure 5.6: Catchment specific runoff (mm) comparison for Belford Burn lower flow gauges. Data displayed from large a single largest discharge event in record (July, 2009)

The runoff excess in $m^3.hr^{-1}$ is assumed to be uniform across the catchment and can be calculated as:

([specific runoff*10]*total catchment area) - ([specific runoff*10]*critical catchment area) [2]

The second bracketed term in equation 2 is constant at 151.2 $m^3. ha^{-1}. hr^{-1}$

Simplified this means that the hourly surface runoff discharge (m^3) at a selected outlet is given by

([specific runoff*10]*total catchment area) – 151.2. [3]

Negative values are corrected to zero (no runoff) in the model. Very low discharges taken from the stream gauge record reflect local groundwater table (base flow) and may not be accurate for the case study sub-catchment. However, the groundwater contribution to storm discharge appears very small in this catchment, so this does not introduce any notable error to the calculation.

The drain flow capacity for Lady's Well field was estimated as 16 $l.s^{-1}$ and runoff simulations repeated using the calculations described above.

Simulation outputs

The simulated runoff data indicates an increased frequency of runoff events with increasing catchment area in West Hall catchment, although for both catchment sizes the largest number of events are of relatively low magnitude (<50 $l.s^{-1}$ peak discharge – Figure 5.7). Increasing drainage area increases the magnitude (and duration) of large infrequent runoff events (Table 6.4). Figures 5.8 and 5.9 indicate that during the observed major sediment loss events in March and December 2010, the majority of catchment runoff occurs as overland flow. The estimated proportion of annual runoff occurring as overland flow increases with catchment area, although in both cases the drain flow pathway is much more important on an annual basis (Table 5.4).

Comparison of runoff from the West Hall and Lady's Well catchments indicates that although the propensity to surface runoff in the latter is similar to that for the 33.4 hectare West Hall catchment, the magnitude of peak runoff is much lower (Table 5.4).

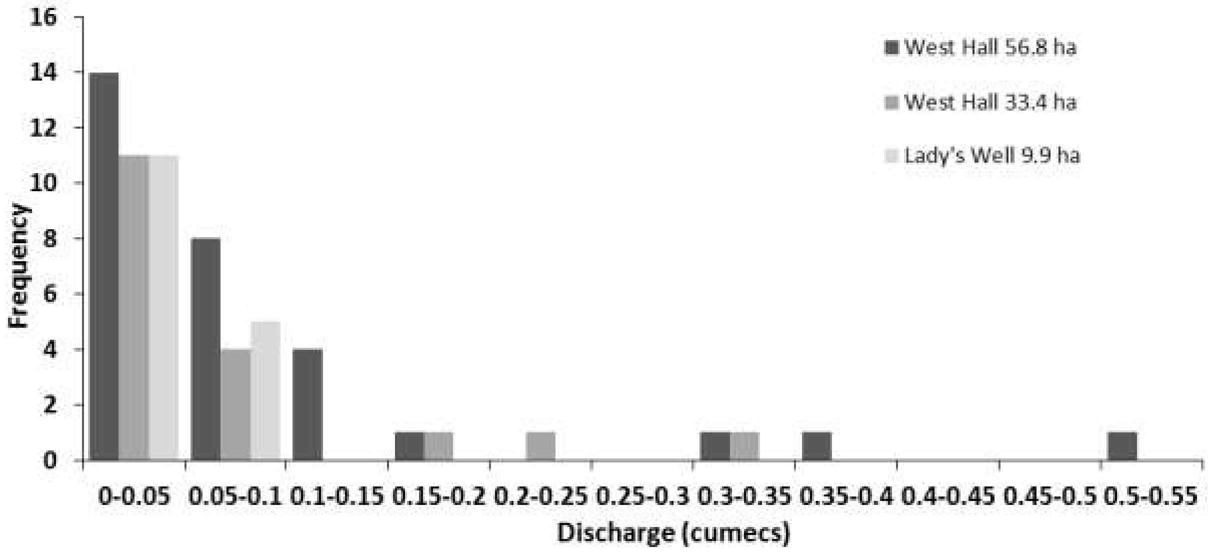

Figure 5.7: Event peak discharge frequency for case study catchments 2008-2010.

	West Hall 33.4 ha[1]				West Hall 56.8 ha[1]				Lady's Well 9.9 ha[2]			
	2008	**2009**	**2010**	**Full record**	**2008**	**2009**	**2010**	**Full record**	**2008**	**2009**	**2010**	**Full record**
Peak Q (cumecs)	0.23	0.28	0.31	0.31	0.41	0.51	0.55	0.55	0.063	0.08	0.088	0.088
Annual P (mm)	756	626	898	760	756	626	898	760	756	626	898	760
Annual spec. runoff (mm)	230	305	604	380	230	305	604	380	230	305	604	380
Annual surface runoff (m^3)	7814	7320	23855	12996	20295	18886	70168	36450	1838	1739	5452	3010
Predicted surface runoff%	10.1	7.2	11.2	9.4	15.4	10.9	19.3	16.4	8.1	5.8	8.6	7.8
Predicted drain flow%	89.9	94	88.8	90.6	84.6	89.1	80.7	85.6	91.9	94.2	91.4	92.2
Total events[3]	5	4	9	18	9	10	10	30	3	4	6	13
Winter events	1	2	9	12	2	8	10	21	1	2	6	9

Table 5.4: Simulated runoff statistics 2008-2010.

[1]Flow capacity for 22.5 cm internal diameter clay tile at 2% gradient estimated as 42 $l.s^{-1}$

[2]Flow capacity for 15.2 cm internal diameter clay tile at 2 % gradient estimated as 16l.s-[1]

Figures from Castle *et al.,* 1984.

[3]A separate event is defined as a surface runoff period separated by at least 24 hours

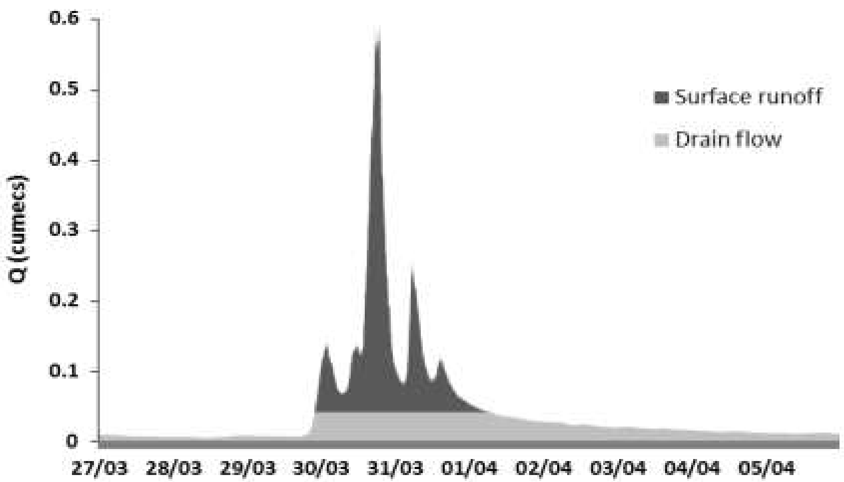

Figure 5.8: Simulated run-on for West Hall Field (56.8 ha) March 2010 event.

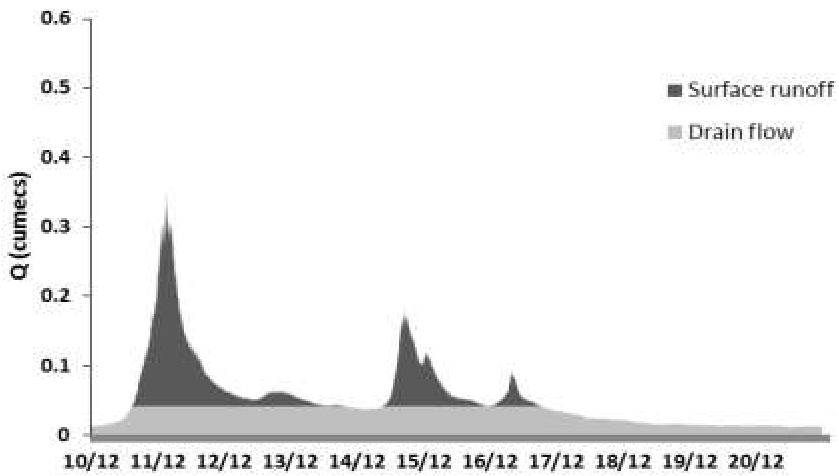

Figure 5.9: Simulated run-on for West Hall Field Dec 2010 event.

5.4 Discussion

5.4.1 Erosion measurement methods

The poor definition of the erosion channel in West Hall field led to concerns over the accuracy of channel volume loss estimates (method 1). In particular it was expected that natural ground curvature over a broad (circa 3 m) channel could result in overestimation of channel cross sectional area and thus overestimation of soil loss. However, comparison with the outputs of method 2 suggests that this was not the case; while the channel loss estimate is approximately 58% higher than the deposition estimate (Table 5.3), the presence of a significant volume of sandy deposits of in the downstream field confirm that method 2 represents an underestimation of total channel erosion. Allowing for this underestimation, the figure provided by method 1 would appear acceptably accurate. The use of deposition as a means of estimating soil loss is likely to lead to underestimates of soil erosion losses, particularly where the particle size distribution of deposited material differs from the source (Evans, 2002). This problem is partly addressed through laboratory measurements in this study, although not the underestimation due to downslope deposition. The possibility also exists that at peak flows a proportion of coarser particles may have remained in suspension and been exported from the catchment. This suggests that the estimate of sediment loss from field F3 is also conservative.

Figures for Lady's Well field illustrate a discrepancy between the estimate derived from channel volume and from the volume of fine sediment trapped by the mitigation feature (Table 5.3). Even assuming that the feature was 100% effective in retaining fine sediment, the volume of trapped material suggests erosion losses considerably in excess of the channel loss volume. Given that the trap is known to be only partially effective (see Plate 5.4, Photos 5-6) this discrepancy appears significant. There are two likely explanations: firstly it is evident that sub-surface erosion caused by return flow from surcharged drainage pipes contributed to the total eroded material. Two large holes were formed above the lateral drain (plate 5.1). The upstream of these was measured at an internal diameter or 55 cm and a (visible) depth of 65 cm, representing an estimated soil loss of 185 kg. Secondly, it is evident that there was a contribution was made by hill slope erosion (see Plate 5.1) in addition to that caused by the concentrated flow channel. The use of channel volume methods therefore represents an underestimate of total field sediment loss, which becomes increasingly problematic as the proportion of eroded material derived from hill slopes increases.

5.4.2 Sediment loss rates

While the event sediment loss in all three fields was significant, sediment loss from West Hall field appears is of a higher order of magnitude, and may also be sufficient to represent an unsustainable soil loss. The reasons for the difference in soil loss would appear to be explained by the difference in upstream catchment area; 56.8 ha compared to 4.1 ha for Lady's Well and 5.2 ha for F3. The thalweg flow path of West Hall field was less steep than that of the two other fields, and although the catchment hill slope gradients for West Hall were significantly greater than those of F3, this is not the case for Lady's Well, meaning that differences in gradient-controlled runoff erosivity is also an unlikely explanation. This indicates that the flow *volume* rather than the flow *velocity* largely controls the erosive energy (and that unlike for hill slope erosion, these factors appear largely independent). Simulation data (Figure 5.7, Table 5.4) illustrates that surface runoff discharge increases rapidly with catchment area; a similar exceedance of drain flow capacity (specific runoff) can lead to much larger discharges and erosion/transportation potential as catchment area increases, as observed in West Hall compared to Lady's Well catchment.

The values calculated for West Hall may appear a severe soil loss given they occur during a single event lasting less than 48 hours. However, the likely return frequency of both runoff and erosion events of this magnitude must also be considered. The runoff event would not appear to have a particularly long return period, and as larger events have been observed in 2008, 2009 and 2010 (see Figure 4.10), a return period of less than one year appears probable. However, for erosion of this magnitude to occur, the event must coincide with periods of high land use susceptibility. All three study fields were sown to winter cereals relatively late in the year (late October) in 2010. Late sowing of winter cereals is regarded as a high erosion risk land use (Defra, 2005). A lack of winter crop cover prior to the end of the growing season results in a long winter period of high susceptibility (See Plates 5.1-5.3). Therefore under the observed cropping, significant event sediment loss is likely if not probable in these fields in an average or wet winter.

Early winter sowings reduce the period of highest erosion susceptibility, and make the coincidence with large runoff events much less likely (Defra, 2005). As the land in question is in arable and pasture rotation, some (pasture) years are also likely to have a low susceptibility, although short term reseeded grass may also represent a high risk. Spring cereal cropping may also decrease erosion event probability provided the soil is protected over winter. Although long term assessment of the contribution of these events to average annual catchment sediment yields is beyond the scope of this study, they may be regarded as a likely to occur once during a winter under late sown cereals (or autumn grass reseeds), and significantly less frequently under early sown winter cereals, or spring sown cereals with residue retention or catch crops. Overall therefore these processes represent a very significant, yet avoidable risk which should be taken into account in land use planning where hazards are identified. In the absence of such planning they are likely to represent a significant contribution to annual sediment catchment exports. Evans (2002) suggests serious rill erosion occurs every 2-4 years on susceptible arable fields in the UK, which would appear a conservative estimate for the West Hall catchment.

Under circumstances where hill slope erosion is significant, the presence of a concentrated runoff channel is often crucial to whether derived sediment reaches an off-site receptor. The visual evidence provided in the March 2010 storm event (Plate 5.5) illustrates that concentrated runoff over permanently vegetated surface retains sufficient transport capacity to connect source areas with remote receptors; under saturation excess flow conditions the effects of vegetation cover is not sufficient to cause sedimentation due to flow impedance. Extensive studies of the effects of grassed waterways in preventing sediment losses from the South Downs area (Boardman *et al*., 2003) have led to suggestions that grassed waterways can be effective in preventing sediment losses by promoting infiltration and sedimentation. However, this appears to apply to circumstances where flow is initiated by infiltration excess caused by surface sealing of cultivated soils. In these circumstances slowing of flow over vegetated surfaces of higher infiltration causes sedimentation. The potential of this mitigation strategy under saturated soil conditions appears much more limited.

Evidence indicates that erosion by topographically concentrated runoff is the most common cause of erosion on UK arable land, particularly but not exclusively on fine textured soils which are relatively resistant to the lower erosive forces resulting from raindrop impact. Chambers and Garwood (2000) report ADAS monitoring of water erosion incidence on arable land in England and Wales between 1990 and 1994. The most common factor in the initiation (30% of cases) of erosion was linked to valley floor features which concentrated runoff, exceeding other factors including poor crop cover, wheelings and tramlines. In c. 95% of cases rainfall events causing erosion were greater than or equal to 10 $mm.day^{-1}$ and c. 80% were > 15 $mm.day^{-1}$. Standard methods of erosion risk assessment are likely to ignore this significant factor – although Defra advice (Defra, 2005) gives guidance on the assessment of concentrated runoff, this advice is much less detailed than that applied to inter-rill assessment. Evans (2002) details the results of long term field monitoring of water erosion in England and Wales during the 1980's. He reports that both the proportion of erosion resulting from channel erosion (as opposed to hill slope erosion) *and* the proportion of soil leaving the catchment increase with clay content. Evans also states that although the findings of higher erodibility for sandy and silty soils now forms the basis of Defra advice to farmers, these studies do not demonstrate a strong negative correlation between soil clay content and erosion rates**.**

The high connectivity of concentrated flow channels with surface waters during runoff events would also appear to represent the dominant means of surface export for land not directly adjoining surface channels (i.e. land other than riparian areas). This type of erosion is unlikely to be controlled by some commonly applied mitigation measures such as riparian buffer strips or contour ploughing. The estimates from this study suggest mean figures of 29% and 35% of total eroded material to be exported from the unmitigated fields (F3 and West Hall respectively – Table 5.2). Where concentrated overland flow channels meet watercourses without major impedance this sediment may be expected to reach permanent water courses. Prasuhn (2011) reports the results of a 10 year study in the Swiss Midlands, where median erosion rates are low (<1 t per ha per annum), which suggests that 52% of eroded material was deposited within the field of origin, and 20% reaches water bodies. Given that only fine sediment export (<106 µm) was measured in the present study the current estimates bear comparison with these figures, although lacking the benefits of long term study required to achieve satisfactory mean estimates.

5.4.3 Effect of tile drainage

An important role of artificial sub-surface drainage systems in this channel erosion has been identified in this study. Replacing open ditches with under-drainage offers the advantage of simplified mechanised field operations and is therefore attractive to farmers. This presents a problem in fields with natural flow concentration, particularly where the drain replaces a permanent natural stream, as appears to be the case in West Hall catchment. The main leader drain of a tile system must be laid at the lowest topographic point to be effective in draining the whole field, meaning it must follow the valley floor in convergent topography. Where valleys continue for some distance, the catchment area can become large, leading to increasing pipe capacity requirement. The required capacity to ensure adequate drainage in moderate to heavy runoff events increases rapidly with drainage area to the point where the required pipe diameter becomes impractical. The question in these circumstances is whether the advantages of sub-surface drainage (ease of machinery access) outweigh the disadvantages (soil erosion and crop damage).

The characteristics of the Belford catchment - low base flow due to the effects of drainage on the perched groundwater table and flashy hydrographs when soils are saturated appear typical of slowly permeable lowland topography. The result, as in this case, of inappropriately small pipe diameter is a high propensity to overland channel flow generation. The probability of this type of erosion occurring is dependent on the catchment area, pipe size and gradient, with the potential for channel generation dependent upon the return period of specific runoff events relative to the pipe capacity. The output of the flow modelling exercise indicates that drainage exports in excess of the capacity of the tile drainage system are infrequent; under most circumstances under the local climate the tile network is able to both minimise soil waterlogging and prevent overland flow (which may lead to erosion). It is important to note that the intended purpose of these drainage systems is to minimise the duration of soil saturation, and the drainage system may be regarded as effective provided this criteria is met.

Castle *et al.* (1984) report drainage systems for arable agriculture should be designed with capacity such that the five-day rainfall should not exceed the five day drainage capacity more than 1 year in 5 – estimated as 8 mm per day (40 mm over five days) for lowland North Northumberland. Even at the larger West Hall catchment area (56.8 ha), the estimated 24 hour drainage capacity is 6.42 mm. This may be acceptably efficient for the cultivation of lower value cereal feed crops being grown on this land (a 1 in 2 year return period should be sufficient under these circumstances; Castle *et al*., 1984). However, as illustrated (Table 5.3, Plate 5.4, Figures 5.8 & 5.9) designing according to these requirements is not sufficient to prevent significant, potentially erosive concentrated runoff; the specific drainage capacity for the West Hall catchment is estimated as 0.27 $mm.hr^{-1}$ – a runoff magnitude observed to be exceeded multiple times per year.

In the short term the occurrence of drainage excess is only likely to present a problem for the landowner if it causes flooding which damages crops or infrastructure, or if erosion problems become sufficiently severe to affect productivity. The loss of yield, seed and fertiliser from an eroded channel plus burial of crop in depositional areas are likely to be bearable losses, particularly given higher profit margins for cereal cropping compared to alternative land use such as pasture. In some circumstances (such as potentially in West Hall field) the loss of soil due to channel erosion may be sufficient to compromise sustainable fertility, although even in these circumstances there are unlikely to be any short term consequences (Boardman, 2002; Chambers and Garwood 2000). However, the loss of sediment and associated nutrients and contaminants may have tangible off-site impacts which make a stronger case for mitigation. In these circumstances incentives needed to facilitate management changes differ considerably, however.

5.5 Summary and conclusions

This chapter has demonstrated that some parts of an agricultural landscape may be susceptible to sediment losses due to topographically determined erosion and high connectivity arising from the concentration of runoff. Unlike risks associated with tile drained land under susceptible uses, or cultivation of erodible soils on steeply sloping land, these risks cannot be effectively evaluated either on a field by field basis, or using generalised remote data. It is necessary to determine the direction, scale and impedance of concentrated runoff, often over considerable distances and across multiple boundaries. The results presented in this chapter illustrate that sediment yields associated with these landscape processes may be appreciable, and therefore need to be included in risk assessment and subsequent mitigation strategies. Initial use of GIS terrain analysis has indicated potential in conjunction with field validation. This potential is investigated in further detail in Chapter 6.

Chapter 6: Remote asessment of concentrated overland flow risk

6.1 Introduction

Concentrated runoff along ephemeral channels can represent a considerable erosion risk under susceptible land uses (Chapter 5). While the frequency of such erosion events is a function of the complex interaction of (rotational) land use susceptibility and runoff event return period, the location of areas at risk of channel erosion may be predicted by topography. In much the same way that soil type and gradient largely control landscape susceptibility to hill slope erosion, catchment area and other topographic characteristics control the susceptibility to channel /concentrated runoff erosion (Desmet *et al*., 1999). Moreover, the generation of concentrated surface runoff increases the connectivity of hill slopes to surface waters, meaning that even in the absence of significant channel erosion risks (e.g. with grassed waterways), the existence of these flow paths increases the probability of sediment derived from adjoining slopes reaching water bodies (Gordon *et al*., 2008).

Location of these pathways requires either historical knowledge of previous events, detailed ground survey or considerable skill by remote surveyors. The tendency for large, infrequent events may lead to neglect of this issue in sediment loss risk assessment; concentrated runoff risks are much less commonly reported than those for steeply sloping fields with susceptible soil types, and published guidance (Defra, 2005) is much more rudimentary. As these may represent critical areas for sediment loss, their identification is important in considering appropriate management and mitigation with regard to agricultural sediment pollution.

As these pathways are topographically controlled, given the availability of topographic mapping of suitable resolution, it should be possible to locate runoff generating areas, and consider land use induced risks appropriately. A simple means of identifying high risk areas would aid planning of mitigation schemes, and GIS terrain analysis techniques may offer a potential solution to this problem. The objective of this study was therefore to identify whether remotely derived topographic parameters could be used to identify areas at risk of erosive concentrated overland flow.

6.2 Background: *GIS methods for flow routing.*

A large number of GIS-based algorithms exist for simulating the concentration of flow within a three dimensional landscape. The basic information requirement is a Digital Terrain Model (DTM) of sufficient resolution to accurately determine flow paths. The main difference between algorithms is the manner in which they deal with concentration and dispersion of flow. Early simple algorithms treated moving water in the manner of a ball rolling downslope;

the flow following the line of maximum slope, usually represented by a limited number of downslope compass points – 8 cardinal points in the classic D8 algorithm (O'Callaghan and Mark, 1984). This approach is limited by the fact that diffuse flow may follow a number of flow directions (not simply one of eight cardinal directions). Alternative algorithms allowing dispersive flow were therefore developed, such as the FD8 model (Quinn *et al*., 1991), which forms the basis of the widely utilised topographic wetness index calculation used to identify critical areas with high moisture storage which are likely to generate runoff early in storm events (Beven and Kirkby, 1993; Quinn *et al*., 1995).

Further developments have introduced adaptations to allow flow in infinite slope directions, but constrained to a limited number of primary pathways to prevent erroneous dispersion. Tarboton (1997) compared a number of routing algorithms, and produced an alternative known as D-infinity, arguing it to be superior due to the prevention of erroneous dispersion in other multi-directional algorithms, without the limitation of grid bias represented by D8. The DEMON algorithm (Costa-Cabral and Burges, 1994) attempted to solve the problem by representing flow as two rather than one dimensional, meaning that flow is possible along pathways of variable width (number of grid cells), preventing false concentration, but without dispersion errors which limit the use of multi-directional algorithms at the catchment rather than hill slope scale.

Wilson *et al*. (2007) compared the effects of the use of different algorithms on allocation of contributing area. They found the main differences to occur between uni-directional algorithms (D8 and Rho8) and multi directional (Dinfinity FD8, DEMON) due to more scattered distribution of low value cells with the former methods, whereas these were restricted to ridge crests and upper slopes with the latter. Desmet and Govers (1996a) specifically investigated the use of different flow routing algorithms for use in prediction of ephemeral gulley formation, and concluded that routing algorithms which allocate flow to only one or two downslope cells are more fit for purpose than multiple flow directions, as they were better able to accurately determine main drainage lines. However, these algorithms appeared to predict gulley initiation higher on hill slopes than was observed through ground-truthing. Although the slope position of gulley initiation may be influenced by soil erodibility (shear resistance to channel flow) this phenomenon appears most likely attributable to the tendency to 'force' channel flow where dispersive flow occurs at the hill slope scale using simpler flow algorithms.

Topographic indices

Attempts have been made to utilise the outputs of flow accumulation maps in the identification of runoff and erosion. The Topographic Wetness Index (Beven and Kirkby, 1993) is one such as example, and is a dimensionless representation of catchment wetness expressed as a simple product of slope and catchment area:

$$TWI = \ln(\frac{a}{\tan\beta})$$

Where a is the specific catchment area per width of grid cell and *β* is the local slope angle in radians.

A related means of estimating the transport capacity of flowing water is the Stream Power Index (SI), which has been employed as a simple estimate of channel erosion risk:

$$SI = \ln(\mathrm{a}.\tan\beta)$$

It can be seen that the stream power index operates as the inverse function of the wetness index; stream power being increased rather than decreased with slope. The wetness index is an approximation of the movement of sub-surface water. When used as an indicator of the likelihood generation of surface runoff, the assumption is that runoff is controlled by saturation excess rather than infiltration excess – in the latter case increasing slope may be assumed to increase rather than reduce the risk of surface runoff generation. This is in accordance with the theory of the TOPMODEL hill slope simulation model (Beven and Kirkby, 1979).

A limitation in the use of these indices is that the relative impact of a unit change in slope or catchment area is not calibrated. An alternative widely adopted approach to evaluating sediment/erosion risk in a GIS framework involves substitution of contributing area for slope length in estimations based on RUSLE principles (Desmet and Govers, 1996b). The approach has been widely employed, although the original empirical basis for the coefficients was based on short hill slopes (<100 m). This has been adopted for watershed applications in the USPED model (Mitasova *et al*., 1996) which attempts to predict deposition as well as erosion based on changes in transport capacity (derived from slope curvature). Different derivatives of the LS factor have been employed. Moore and Wilson. (1992) calculate these factors as:

$$L = 1.4\left(\frac{A_s}{22.13}\right)^{0.4}$$

$$S = \left(\frac{\sin\beta}{0.0896}\right)^{1.3}$$

Where A_s is the specific catchment area and β the local slope angle in radians.

This computation is known as the *unit stream power* approach.

Desmet and Govers (1996b) derived the L equation in finite difference form:

$$L_{i,j} = \frac{(\chi_{i,j,\text{in}} + D^2)^{m+1} - \chi_{i,j,\text{in}}^{m+1}}{D^{m+2} x_{i,j}^m (22.1)^m}$$

where $\chi_{i,j.\text{in}}$ is the contributing area for overland flow into the cell, *D* is the size of the cell (length of the sides of the cell) and *x* is a factor which accounts for variations in flow width that depend on the direction of flow relative to cell orientation.

The latter form is intended to take account of flow divergence and convergence, essentially the change in erosive potential resulting from changes in flow width, as well as in contributing area.

It is apparent that despite the availability of GIS terrain analysis techniques for several decades, and numerous applications to catchment erosion and sediment studies, to date there remain many questions about the validity of the outputs of RUSLE simulations at a catchment scale. Very large numbers of catchment models have employed adaptation of RUSLE to a 3D landscape, but given uncertainties over the transfer of empirical calculations validated at the plot scale to the catchment scale, the outputs can only truly be utilised as 'index' scores rather than an accurate representation of sediment losses. The prediction of deposition as well as erosion using empirical coefficients (e.g. Mitasova *et al*., 1996) appears a further deviation from the intended application of the RUSLE model. Perhaps most crucially, in catchments where field sediment losses may be dominated by channel rather than hill slope erosion, the assumption that RUSLE is able to simulate these processes in any meaningful sense appears flawed, as the empirical basis is limited to hill slope erosion processes. Reliance on this method is unlikely to represent any real improvements on existing field based assessment techniques (e.g. Defra, 2005) which are based largely on similar hill slope plot data. Boomer *et al*. (2008) demonstrated that while RUSLE-based sediment models produce similar results to RUSLE erosion models, these processes do not successfully explain observed differences in *sedimen*t yield between catchments.

Faulkener *et al.* (2010) applied GIS risk mapping to an area of the South Downs where long term erosion hazards associated with muddy floods are well documented. The calculated index was a simple logarithmic transformation of the product of local slope angle, catchment area and an expert-based land use susceptibility factor. Although the reliance on expert knowledge for the latter is acknowledged, they were able to produce patterns broadly recognisable by land owners (although it is clear that the findings also suggest the use of the technique at the large scale did not provide new information to this study group).

The recently developed SCIMAP risk mapping framework makes use of both stream power and wetness indices to derive fuzzy risk indices for landscape pollutants, particularly sediment. The computation makes risk a function of erosion hazard (stream power index*land use index) and landscape connectivity (the minimum wetness index along a flow path). The derived index has been applied to the River Eden catchment in the North Pennines, and concluded that while predicted sediment risk for arable land uses appeared exaggerated, landscape susceptibility could be successfully identified using a purely remote sensed technique (Reany *et al*., 2011). While the method represents a further simplification of RUSLE-based techniques (minimising data requirements) the development of a quantitative index of connectivity represents an important development towards sediment loss assessment, rather than simply modified soil erosion assessment techniques which generally are much less well validated for sediment delivery to water courses.

Minimum information methods relying mainly on topographic data offer the advantage of rapidity and data availability, and in their simplest form are largely free from the need for expert knowledge. However, as the examples of the application of SCIMAP (Reany *et al*., 2011) and the expert system employed in the South Downs (Forester *et al*., 2010) demonstrate, there is a temptation to incorporate somewhat subjective expert interpretation in order to compensate for inaccurate or uninformative identification of sediment loss risk in its absence; in both these studies 'expert' land use susceptibility judgements were made and later recalibrated on the basis of ground-truthing, with little scientific basis. This suggests that while the remote sensing methods may be capable of accurately predicting the geographic derivation of runoff, the interaction with land use which results in erosion and sediment transport is more complex. The combination of topographic and land use susceptibility to produce a single risk index used in the above examples may appear attractive, but limits the ability of these tools to provide useful information for land use decision making; the separation of landscape and land use susceptibility assessment (e.g. Defra, 2005) facilitates identification of unsuitable land uses on susceptible land areas. Combining these elements means that the consequences of land use change is not a component of output risk maps . Where local expert knowledge becomes embedded in the system, the requirement for such expert knowledge for predictive accuracy would appear contrary to the purposes of such a method.

The relative utility of different flow routing algorithms has been evaluated in a number of studies, with diverse conclusions. It is apparent that 'success' depends in large part on the intended application; where the accurate description of hill slope processes is required, for example for calculation of topographic wetness index, multiple flow direction algorithms are more capable of representing subsurface movements, which in turn appear to largely control runoff occurrence at these scales. Conversely these methods misrepresent the flow of a confined channel, particularly where the channel topography is not represented (by low resolution DTM's). However, the generation of ephemeral channels across fields is intermediate between these two scales, often having large catchment areas but poorly defined flow routes. Desmet and Govers' (1996a) study is the most relevant to the current study, although appears to be targeted to smaller scale channel erosion features. The assessment of choice of flow routing method appears worthy of further study.

6.3 Description of methodological approach

To investigate the utility of GIS remote risk assessment techniques to identify channel erosive features, a multi-stage investigation was conducted. In the first stage (Section 6.4) the Belford Burn catchment was used as a case study area to investigate the potential of the technique to predict observed erosion patterns; previous study (see Chapter 5) had indicated the existence of major routes of concentrated overland flow which had resulted in a number of instances in channel erosion and significant sediment loss through the stream network during rainfall events. Preliminary data processing using GIS terrain analysis had further suggested that many of these flow routes were predictable as topographic positions with large upslope contributing area (see Figure 5.2). The Belford study site was therefore used to evaluate the following:

- The utility of different flow routing algorithms for the identification of concentrated overland flow routes.
- The effect of topographic data resolution.
- The utility of different topographic indices.

Section 6.4 includes preliminary discussion and conclusions with regard to the three objectives described above, which justify the approach taken in later sections of this chapter. However, the small size (5.9 km^2) of the Belford catchment means that predicted concentrated flow routes are limited in number, as are the number of observed erosive features. This presents a study limitation in that the important issue of the generation of false positive and negatives (i.e. the susceptibility of the method to misidentify risks) cannot be adequately interrogated.

Section 6.5 therefore applies the selected method to a wider catchment by selecting a 15*15 km (225 km^2) study area which includes the Belford Burn catchment. This comprises a quantitative evaluation of the topographic properties which characterise areas of channel erosion. Methods for each section are presented separately. As the methodology applied at the larger scale is a product of the evaluation conducted in the Belford Burn catchment, the results, discussion and conclusions of the latter are presented *a priori* the wider catchment study.

6.4 Belford catchment study

6.4.1 Methods

Input data

High resolution topographic imagery (20 mm elevation accuracy, 2 m resolution) was derived from *Lidar* survey data of the Belford catchment provided by the Environment Agency. This was compared with the national Ordnance Survey *Landform* data (10 m resolution) derived from digitisation of map contours from the 1:25000 Ordnance Survey Explorer Series. This data was downloaded through the Digimap service.

Data processing

Digital Elevation Model raster datasets were derived using the Arc/Info® "Topo to raster" interpolation method to derive smoothed topography. Grid data was imported to SAGA open access GIS software (Conrad, 2006). SAGA GIS facilitates rapid processing of large raster datasets, and being developed as an open access terrain analysis tool allows the use of a wide range of different algorithms for flow routing and different computations of topographic indices (e.g. LS, SPI, TWI) using previously written code by programme participants. These capabilities make it ideally suited for an evaluation and selection procedure for topographic parameters. Sinks/pits were filled using the Planchon and Darboux (2001) method automated facility of SAGA.

Choice of flow accumulation algorithm

Within SAGA GIS intial comparison of flow routing algorithms was undertaken to determine the effect of choice of algorithm in the computation of specific catchment area (and indices derived from the calculated catchment area). The following flow routing algorithms available as SAGA modules were compared:

- D8 (O'Callaghan and Mark, 1984).
- Multiple Flow Directions (Quinn *et al.*, 1997)
- Dinfininty (Tarboton, 1997)
- DEMON (Costa-Cabral and Burges, 1994)
- Multiple Triangular Flow Directions (Siegbert and McGlynn, 2005)

All raster outputs were computed using a 5 m resolution pit-filled DEM raster dataset derived from Lidar data for the Belford Burn catchment. These were calculated using SAGA GIS software. Qualitative comparison of the outputs was made using graphical and flow accumulation map outputs for the Belford Burn catchment.

Effect of data resolution

In order to evaluate the effects of the resolution of available topographic data on flow accumulation route accuracy, qualitative comparison was made of the outputs of data derived from detailed Lidar data and lower resolution OS data. Raster interpolation was conducted using ArcGIS 'topo to raster' at 10 m 5 m and 2 m grid cell resolution for both Lidar and OS datasets. Visual qualitative comparison was made of flow accumulation map outputs for the Belford Burn study catchment.

Topographic inidices

Visual qualitative comparison was made of the topographic index map outputs for the Belford Burn catchment. The following topographic indices were calculated for the 5 m resolution Lidar dataset:

- Topographic Wetness Index (Quinn *et al.*, 1995)
- Stream Power Index (Moore *et al.*, 1993)
- LS Index (Moore and Wilson, 1992)

Validation against study catchment processes

The assessment made is validated against field observations made during the period July 2008 – March 2011 (including but not limited to the data presented in Chapter 5). Use was also made of free historical aerial imagery for the catchment available using Google Earth software in order to investigate the potential recurrence of channel erosion in areas of high predicted risk.

6.4.2 Results

Comparison of flow routing algorithms

Comparison of the D8 method with multi-directional algorithms illustrates that the latter methods result in less very low contributing area cell values; the unidirectional D8 method results in more cells with single cell contributing area, or a multiple of only a small number of cells. The distinctive pattern produced by the scatter diagram below (Figure 6.1) can be explained by the limited number of unique values which occur in hill slope areas using the D8 method – grid cells generally are either part of a flow line or have no upslope contributing cells. In comparison the multiple flow direction methods allow flow to 'diffuse' across the hill slope.

Comparison of the different multiple flow direction algorithms illustrates a high degree of similarity between outputs (Figure 6.1). The Multiple Flow Direction (MFD) algorithm output can be seen to diverge from other methods in 'hill slope' areas; the MFD method has a relative tendency towards equal distribution of flow to grid cells due to dispersion. The other methods lead to more concentrated flow paths and hence high or low cell (fewer intermediate) values. The DEMON algorithm in particular tends to concentrate flows at the hill slope scale relative to other algorithms, resulting in less cells of intermediate catchment size. However, despite discrepancies between methods there is a clear convergence in predictions at contributing areas of approximately 10000 m^2 or greater (1 ha; log = 4.0).

Effect of choice of flow routing algorithm

The use of the D8 algorithm has the advantage of simplicity, leading to clearer outputs with flow paths of a single cell width. However, a limitation of this method is that flow divergence is not simulated. This appears of particular significance in catchment areas where the ratio of maximum slope to other downslope angles is low, notably areas of reduced curvature where natural dispersion of flow may be expected. Neither is it possible to accurately determine points of highly convergent (headwater) flow. This limits the potential to predict areas of channel initiation (erosive) or channel dispersion (depositional). This issue is illustrated in Figure 6.2; whereas the Dinfinity and MFD methods are capable of detecting points of potential flow dispersion (circled), the D8 algorithm allows for flow accumulation but not dispersion. The same appears to apply to the DEMON algorithm. The significance of these areas in the case study catchment is discussed later in the chapter.

The Multiple Flow Direction method appears the most representative of the commonly used algorithms for the simulation of *sub-surface* water movement. The effect of this algorithm is to increase dispersion of concentrated flow at points where distinct topographic concentration into flow channels is not detected. At a hill slope scale this means that *all* cells at the foot of a slope have a significant contributing area – a reasonable approximation of sub-surface hill slope hydrology. At this scale the false concentration of flow into channels derived from other algorithms may represent an inaccurate indication of the spatial distribution of runoff. Even where channel erosion *does* occur on steep hill slopes with small catchment areas, the exact location is unlikely to be predictable without extremely detailed topographic survey data.

The ability of the MFD algorithm to route flow in multiple directions may be useful in determining areas where significant dispersion of concentrated overland flow occurs, where valley topography diverges for example. However, this property is likely to present issues with low resolution topographic data, where natural channels are not detected (resulting in erroneous flow dispersion). For example, historic or prehistoric erosion processes (or even depressions formed by the installation of drainage) can result in small channel depressions in agricultural landscapes which are not indicated by map contours, meaning they are not a feature of contour-derived Digital Elevation Models. In these circumstances false dispersion may be predicted using this algorithm, and while other methods will be unable to detect the exact route, the channel erosion potential within an individual field should still be identifiable.

The Multiple Triangular Flow Direction and Dinfinity algorithms both permit a degree of divergence of channel flow, although in both cases divergence is constrained compared to the MFD method. The outputs of these methods appear particularly well correlated (Figure 6.1).

Selection of flow routing algorithm

The choice of flow accumulation algorithm does not appear to be a crucial factor in locating major overland flow paths; at catchment areas in excess of 1 hectare outputs of all methods appear relatively well correlated. However, the ability to locate potential points of channel initiation and dispersion with multiple flow methods appears to offer advantages over the traditional D8 method. For the current purposes the Dinfinity method (Tarboton, 1997) is selected for further study based on its widespread usage and its ability to detect dispersion points (without the risk of over-prediction of dispersion presented by the MFD technique).

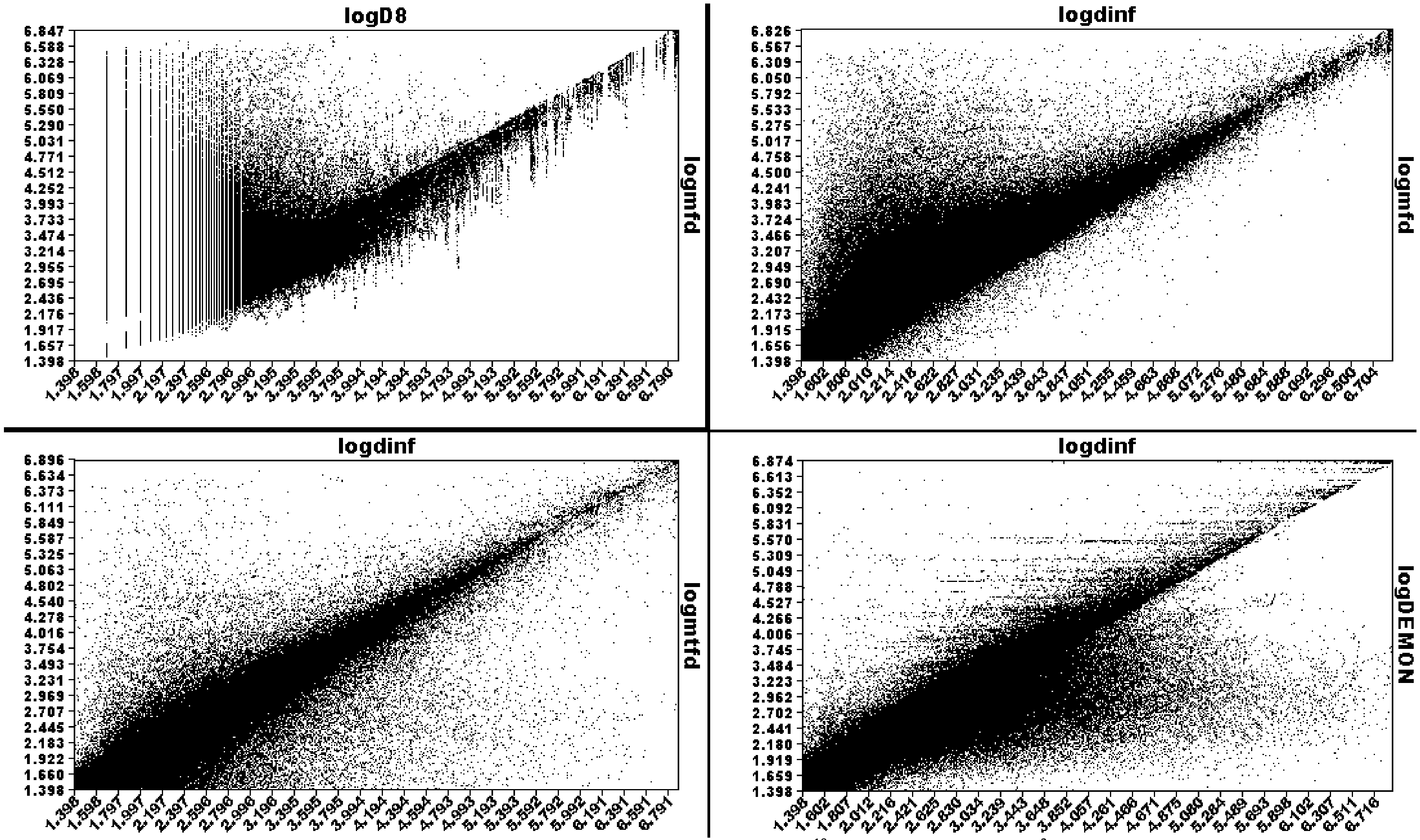

Figure 6.1: Selected comparisons of flow accumulation algorithms. * All scale show $\log^{10}$ cell contributing area (m^2)

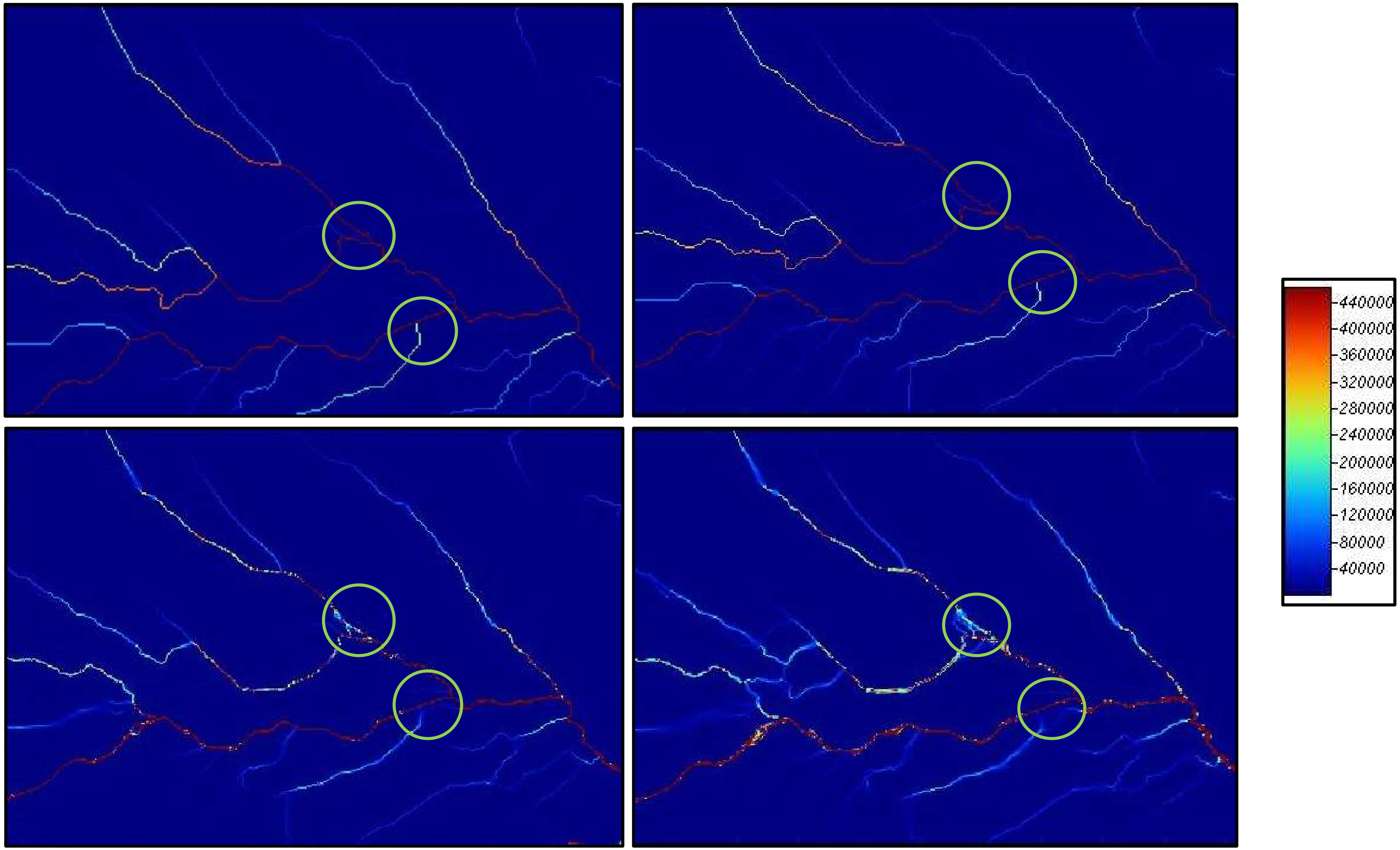

Figure 6.2: Comparison of flow accumulation maps for Belford catchment: Clockwise from top left: D8; DEMON; MFD; Dinfinity.

Comparison of DEM input data

Comparison of maps produced at the same grid cell resolution, but varying input resolution (Figure 6.3) demonstrates that the low resolution Ordnance Survey Landform and high resolution Lidar data produce very similar results. There are some areas of discrepancy; topographic errors in the Ordnance Survey data result in greater divergence of major flow lines (allocation of flow in more than one direction). In this example there is also some incorrect routing using the OS data; notably the river channel is erroneously diverted to the north at the circled point (Figure 6.3).

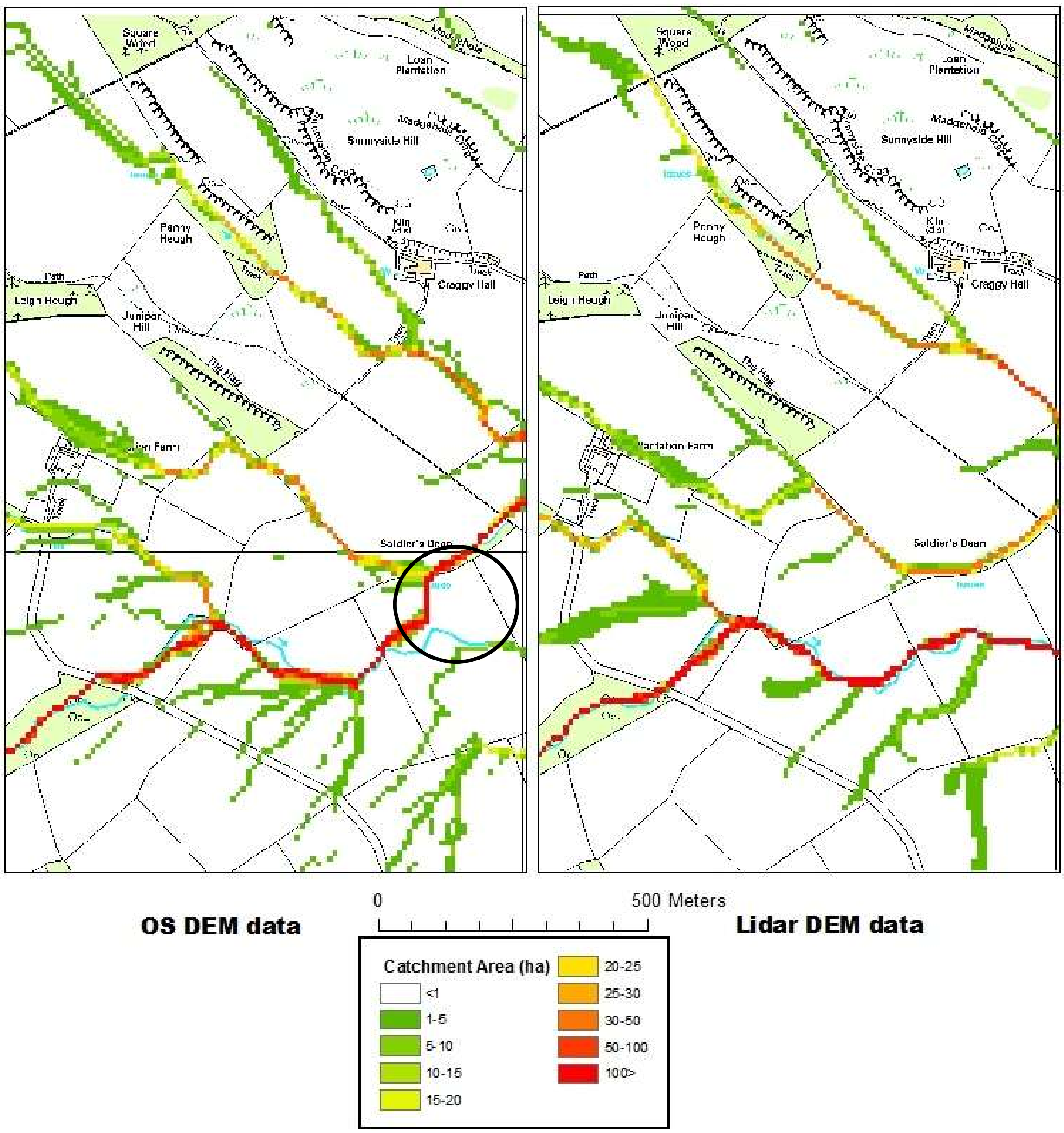

Figure 6.3: Comparative D infinity flow accumulation map (10 m grid resolution).

Increasing grid resolution for Lidar survey data from 10 m to 5 m had minimal effect on outputs (Figure 6.4). However, further increase in resolution to a 2 m grid resulted in the production of dual flow lines rather than flow line dispersion which occurs at lower resolutions (Figure 6.5). It also results in continuity of flow paths where dispersion is predicted at lower resolution in some cases. The circled point shown on Figure 6.5 is significant; a large sand fan was observed and quantified at this point occurring at a point of flow dispersion along a major erosion channel (see Chapter 5).

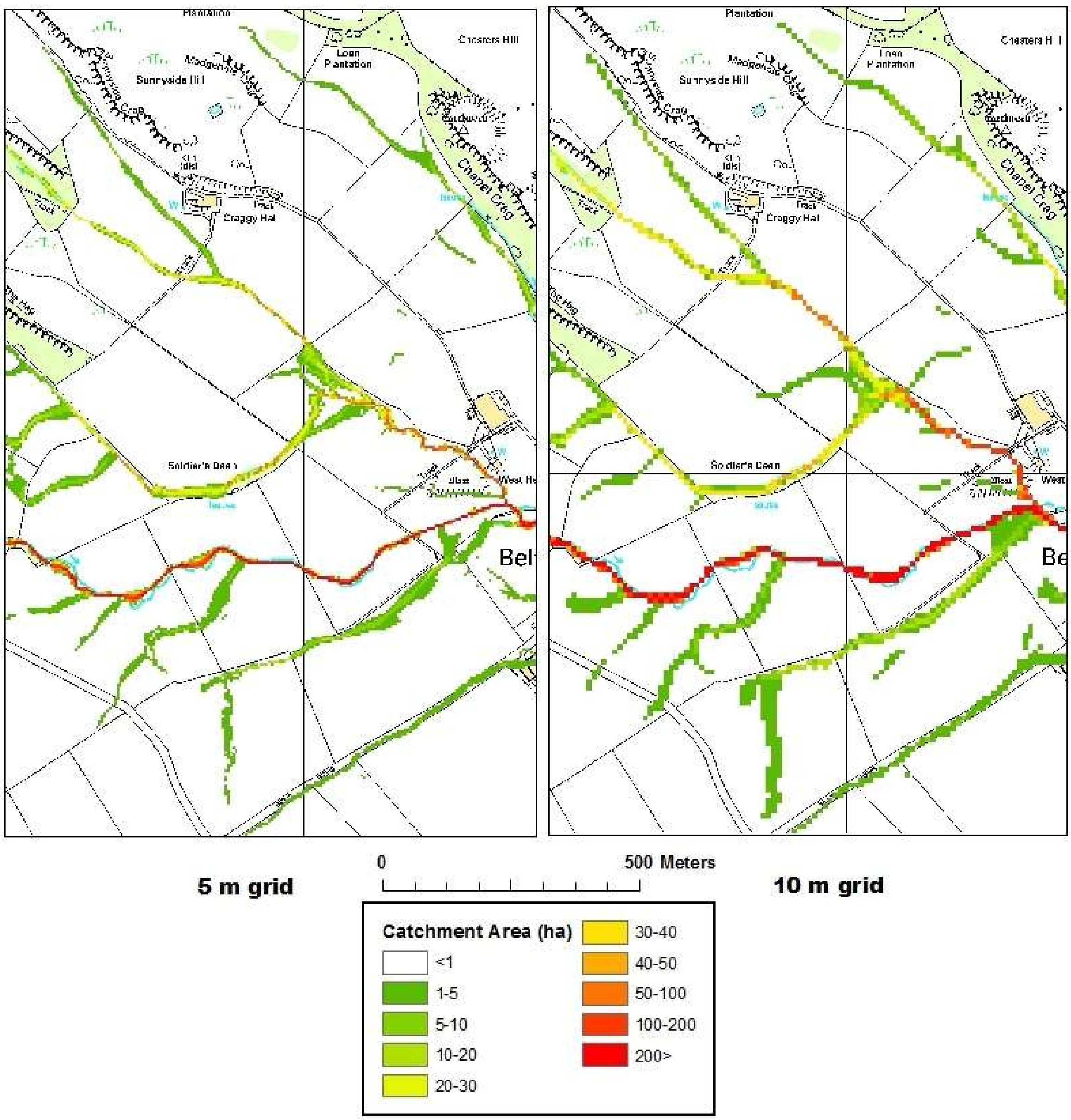

Figure 6.4: Comparative Lidar Dinfinity flow accumulation map: 5 Vs 10 m grid cell size.

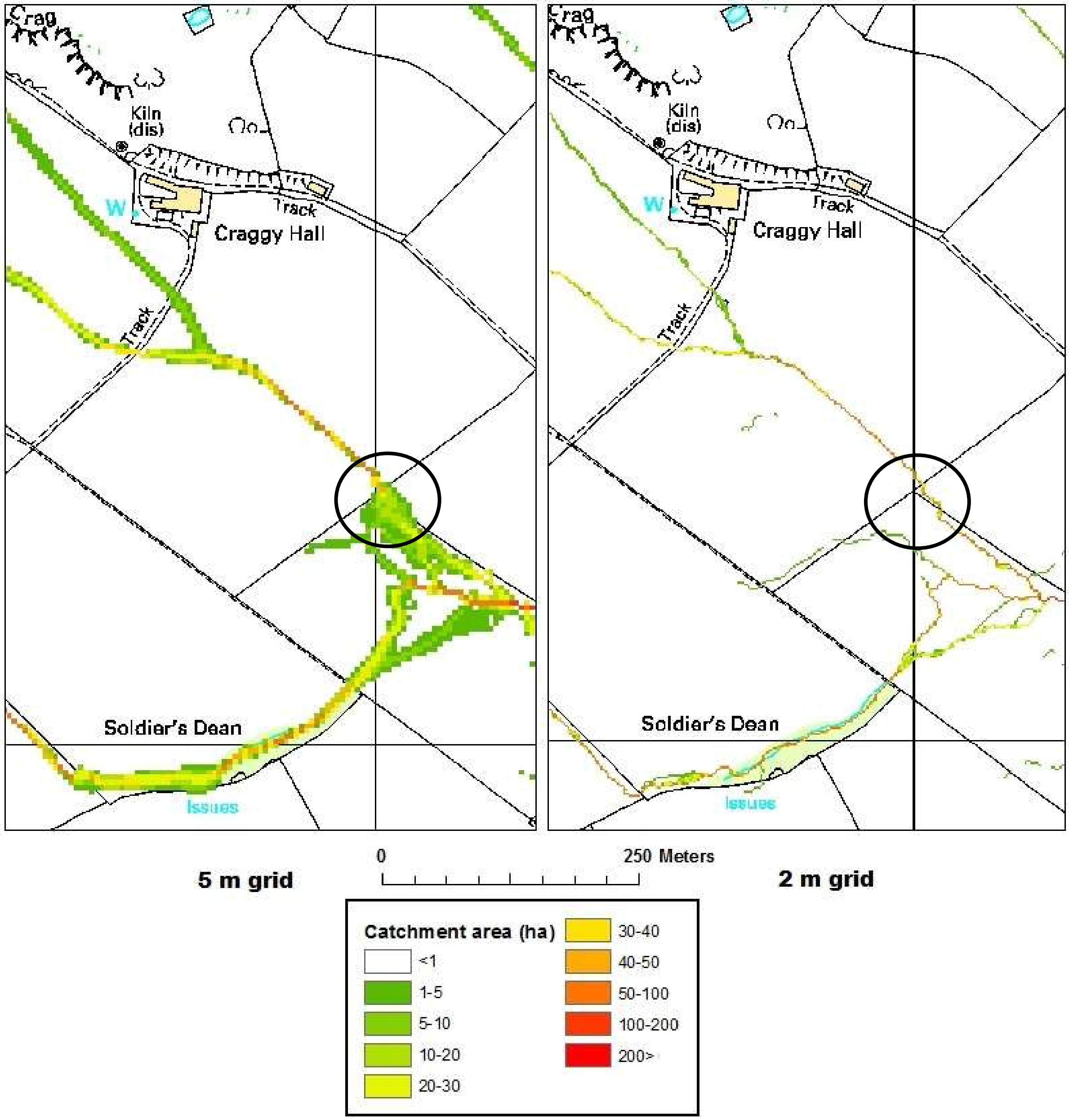

Figure 6.5: Comparative Lidar Dinfinity flow accumulation map: 5 Vs 2 m cell size.

Effect of DEM data resolution

It appears that high resolution data reduces error in flow routing. In particular it appears able to detect small linear depressions across fields which are either poorly defined or not detected by interpolated data from OS map contours. It should be noted that the 1:10000 OS DEM data has already been interpolated by digitisation of maps of smaller scale (1:25000), so the resolution is not well suited to identification of micro-topography. However, high resolution remote-sensed data such as Lidar scanning does also suffer limitations where vegetation cover prevents ground sensing, for example in woodland areas and permeable hedge boundaries. In both these cases flow routing errors can occur. However, this evaluation concludes that where available these datasets are generally preferable to OS data.

Higher resolution data is only available for limited areas, and an important consideration is whether nationally available OS data can be effective at a field scale – capable of predicting erosive flow lines at least within the correct land parcel. The conclusion of this preliminary study is that the lower resolution national dataset retains much of the utility of higher resolution data, and may be similarly employed where it is the only data available. In general it is able to detect large scale features (with catchment areas in excess of one hectare). If identification of smaller scale features is required, accuracy is likely to be compromised without more detailed topographic input data. However the complexity of land management effects on channel formation at these scales means the use of a remote-sensed method of risk assessment is in any case questionable.

As well as the resolution of the input data, the resolution of the GIS *outputs* is an important consideration. Examples using both datasets shown above demonstrate that errors can result where grid cell values are calculated at resolutions approaching the resolution of the input data; in the OS dataset the stream channel was incorrectly routed. This is an example of the problem created where topographic features (such as stream channels) are at a smaller scale than the input data – in this case the channel topography has been interpreted as flat, meaning that flow is free to divert along a locally steeper gradient than that of the channel profile. In the Lidar data example, dispersive flow at low resolution may be due to selection of arbitrary routing directions rather than due to detection of minor topographic channels. This may be problematic, particularly during large runoff events where flow dispersion or ponding are more likely outcomes of a fall in local gradient than the formation of smaller 'dendritic' flow channels. It is therefore recommended that during data processing, cell resolution is limited to the input data resolution or coarser where possible to avoid these errors.

6.4.3 The utility of topographic indices

The previous section illustrates that given sufficiently accurate topographic mapping, it is possible to derive a flow accumulation map which gives indication of points of flow divergence as well as the prescence of major flow lines. This property is important in the evaluation of continuity of flow pathways with regard to suspended material. However, the use of catchment area as a determinant of sediment loss risk does not take account of the effect of gradient on flow energy, important in determining the transport capacity, and crucial in assessing areas of potential sediment mobilisation and channel initiation. Estimating connectivity and channel erosion risk remotely with minimum information are difficult concepts, although several methods for quantification have been developed.

Stream Power Index and Topographic Wetness Index

The widely used wetness and stream power indices are used to identify critical source areas for runoff generation and high energy (erosive) runoff respectively. Both indices are a simple function of catchment area and local slope, but while TWI uses slope as a divisor, for SPI it is a multiplier. In theory therefore, these indices should be negatively correlated; as slope decreases runoff energy decreases and the propensity to sub-surface saturation increases. However, Figure 6.6 illustrates that while for small specific catchment areas the wetness index does indeed represent an inverse function of the stream power index, when catchment areas exceed 10000 units (1 ha per metre width of contour for a 1 m resolution flow accumulation grid) the effect of changes in slope are relatively minor, and the two indices are well correlated regardless of differences in slope; in effect both indices become a function of catchment area.

Figure 6.7 demonstrates that for large concentrated flow lines (those where repeated channel erosion risk is likely to occur) the outputs of both indices are remarkably similar. Discrepancies may be detected where low local slope angle along a large flow line results in greater increase in TWI than SPI (circled). However in both cases the index value increases in a downslope direction due to the increase in catchment area until a dispersion point is detected. The circled area was observed and quantified as an area of significant channel erosion during a large event in winter 2010/2011 (see Chapter 5).

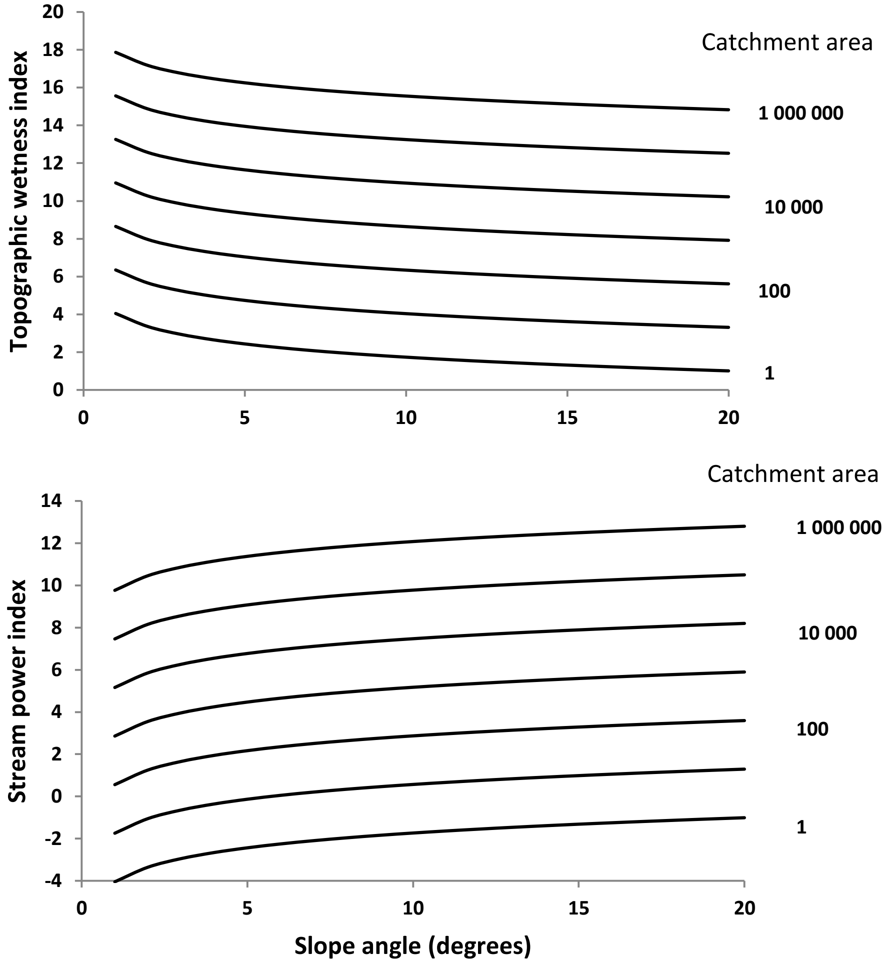

Figure 6.6: Effect of specific catchment area (m^2) and local slope angle on topographic wetness index and Stream Power Index.

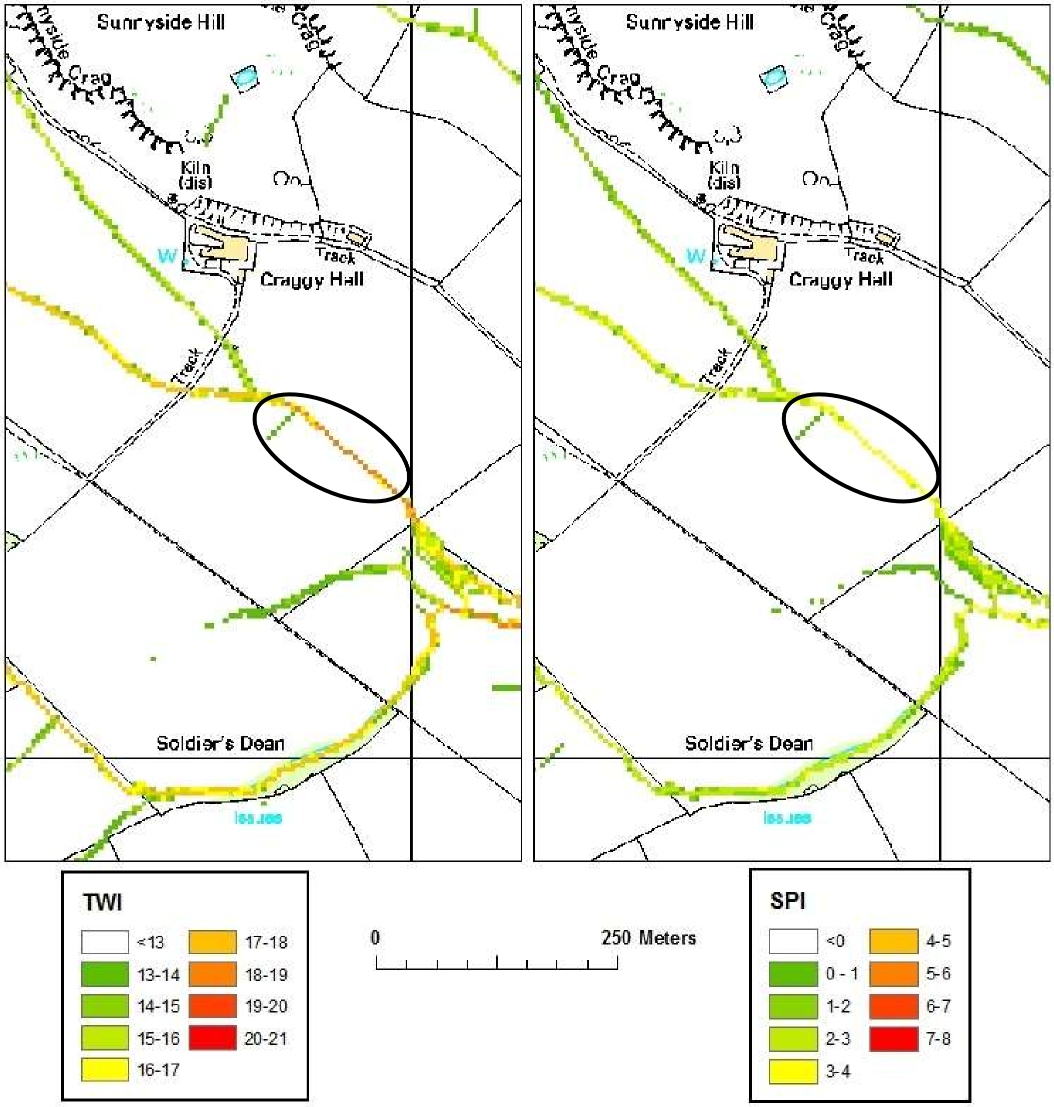

Figure 6.7: Comparison of Topographic Wetness (TWI) and Stream Power (SPI) indices for Belford catchment.

Computation of LS factors

Comparison of the unit stream power method of Moore and Wilson with the unit hillslope method of Desmet and Govers demonstrates broad agreement (Figure 6.8). The use of both methods allows differentiation between areas with large predicted catchment area but minimal gradient and those with similar catchment areas but greater gradient-derived erosive potential. The Moore and Wilson method predicts somewhat higher values in areas of large catchment area but low slope angle, and predicts greater continuity of erosive flow lines. Generally the effect of slope gradient on both hillslope and channel erosion risk is exaggerated using the Desmet and Govers method. The circled areas represent areas where channel erosion has previously been observed (see section 6.4.5 below and Chapter 5).

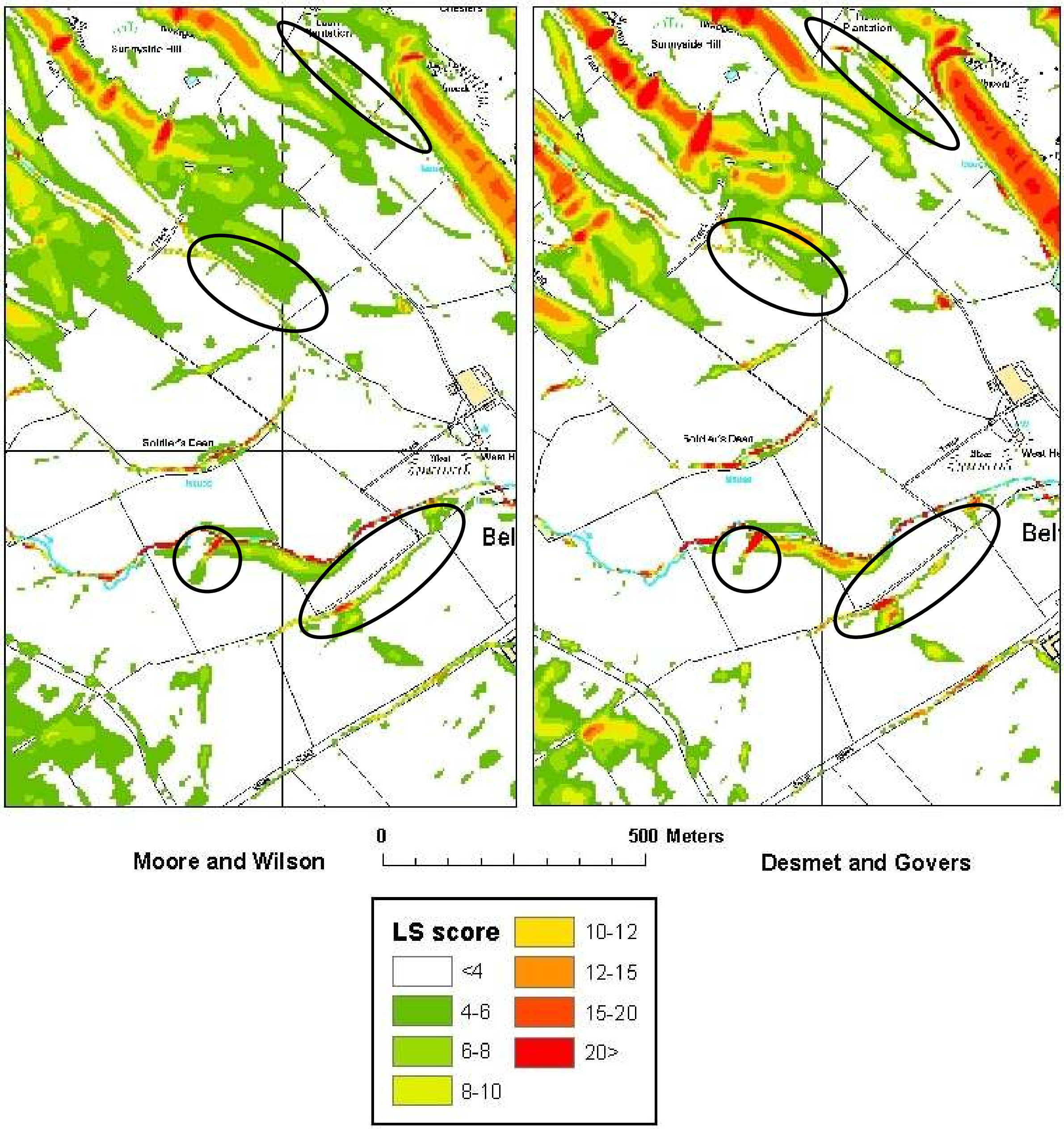

Figure 6.8: Comparison of GIS computation methods of RUSLE LS factor.

Choice of topographic indices

Comparison of topographic wetness and stream power indices, two similar calculations based on contributing area and local slope angle, illustrates that for points with large contributing areas the outputs are strongly positively correlated while the opposite is true for smaller contributing catchments (threshold circa 10000 m specific catchment area). The implication is that these indices become largely a function of catchment area for features large enough to be mapped accurately. Therefore it is questionable whether the use of both terms is useful. The exception may be areas with low local slope (less than 1 degree) where relatively low erosive potential may be predicted using SPI, but a high volume of runoff may cause the formation of large high energy runoff channels (and may be correctly interpreted using the TWI). However, given that no knowledge is assumed regarding soil type and susceptibility to runoff it does not appear valid to apply the TWI to indicate the threshold at which channel formation is initiated. This is particularly the case in a landscape with tile drains of unknown efficiency. The SPI retains greater independence from these unknown factors; assuming runoff initiation is uniform across a landscape, the factors controlling runoff energy are gradient and flow accumulation. For this reason the use of the Stream Power Index is preferred for further study.

Previous studies have demonstrated that soil loss is considerably more sensitive to changes in slope angle than slope length (Mcool *et al*., 1987). The difference in erosive potential between a 1 and 20 degree slope appears likely to be greater than that suggested by Figure 6.6 for large contributing areas. This issue is addressed by the RUSLE LS factor, which applies separate weightings to slope and length. However, the majority of GIS developmental work has focussed on the geo-spatial processing aspect of the problem, rather than the calibration of the original model for catchment areas much larger than the intended scale of the computation. It is therefore uncertain how well this algorithm can be applied in catchments dominated by channel erosion.

It would appear that the main algorithms used in GIS risk assessment are largely a function of the catchment size for flow pathways at scales beyond the hill slope. As there is currently very limited experimental basis for the use of correction factors for slope or the effect of predicted soil wetness, it appears the most logical low input solution is to use flow accumulation as a surrogate for channel erosion potential. The impact of local channel slope on erosive potential may be assessed qualitatively relatively simply. In any case there is considerable interaction between slope steepness and propensity to flow convergence, and between slope angle and land use for convergent flow; high risk land uses with convergent flow on steep sites are highly likely to lead to serious channel erosion affecting farm activities and creating permanent erosive landscape features which prohibit continued use.

Desmet *et al*. (1999) tested the relative importance of catchment area and slope in the prediction of ephemeral gulleys, noting that an exponent value is required for specific contributing area to calibrate the relative importance of the predictors. They found that while a catchment area exponent of 1 (i.e. no adjustment) was most suitable for prediction of gulley *pathways* in the Belgian study area, the *initiation* of gulleys was better predicted where this exponent was 0.2, indicating a much higher importance of slope in determining where gulleys begin. However, the use of such exponents clearly requires considerable ground-truthing and calibration for specific site characteristics based on local climate, geology and soil type.

Ludwig *et al*. (1995) investigated the interacting effects of contributing area and slope on channel erosion in northern France by dividing predicted flow lines into segments of roughly constant gradient. They found that where contributing area exceeded 2 ha and slope 5% most cultivated land was affected by gulley erosion. However, they also found that even when gradients were less than 2%, over 50% of flow segments with over 2 ha contributing area were affected by erosion, indicating that areas with catchments large enough to generate significant surface flow channels could result in significant erosion independent of slope. This demonstrates that erosive channels are not confined to steep slopes. Given the tendency towards lower slope angles as contributing area increases in many catchments, it may therefore be argued that once catchment size reaches a size of several hectares, significant landscape erosion risks may exist irrespective of the gradient in the direction of flow.

A major limitation in assessing relative risk of sediment loss is the ability to predict flow dispersion as well as convergence. This limitation particularly affects areas where ephemeral channels level and widen leading to deposition. GIS processing, including multiple flow direction algorithms appears insensitive to these changes, as often the steepest path is relatively great compared to other directions (particularly the case where flow directions are corrected by automated pit filling).

6.4.4 Visual validation

Chapter 5 illustrated using Belford Burn case study catchment, that where major overland flow pathways coincide with susceptible land uses, significant channel erosion and sediment loss can occur during large runoff events. A number of channel erosive features identified through a combination of field observations and historic aerial imagery are indicated in Figure 6.9. Features A, B and E correspond to the three eroded fields quantified in Chapter 5 (West Hall, Lady's Well and F3 respectively). The location of major runoff pathways appears to be predictable using remote survey data (Figure 6.9) which includes circumstances where the risk is not easily recognisable from topographic attributes observed in the field such as features E and F. In this catchment, channel initiation appears to have been exacerbated by a tile drainage network of insufficient capacity to carry the volume of runoff occurring in events of moderate to large magnitude (see Chapter 5).

The use of historical Google Earth aerial imagery demonstrates that channel erosion observed along two of three pathways shown in Chapter 6 (features A, B, C and D, Figure 6.9) has also occurred in previous years (Photos 6.1 & 6.2). These images were taken over the winter of 2002/2003 and appear to be in response to a major flood event. Review of historical documents suggests a major flood occurred in Belford on 21-22 October 2002 (Berwick Advertiser article 24 October, 2002). It is therefore likely that the erosive features observed were initially caused by runoff during this event. A further channel erosion feature was identified from the same date of aerial imagery although in a location where no similar feature was observed in 2010/11 (location F, Figure 6.9; Photo 6.3). The steeply sloping field margin has since been taken out of production, although some sediment delivery to the watercourse appears to occur despite the presence of a 12 m vegetated buffer strip (photo 6.4).

Channel erosion is not observed on the 2002 imagery in areas B, C or E whereas areas A, D and F were not observed to erode in 2010/11. This difference is entirely attributable to the effects of crop rotation; all of the channel erosion features identified from field observations in 2011 (n = 3) or historical imagery in 2002 (n = 3) occur on land under winter cereals. When these points did not display channel erosion in the respective observation periods they were exclusively under grass rotational stages.

The use of a simple flow accumulation grid clearly identifies the catchment areas where major flow accumulation occurs. In some instances (such as the flow line indicated at points A and B Figure 6.9) the topographic attributes of fields present very clear channel pathways, and as such it is arguable that the occurrence of channel flow lines can be easily identified in the field or from map contours. Even in these circumstances however, the total catchment size may be surprisingly large and not immediately apparent where headwaters are gently sloping. In other instances, such as points E and F (Figure 6.9), channel erosion is not visually obvious from topographic attributes. In these situations it appears that the combination of moderately steep local slope and upstream areas of high saturation propensity provide the causal factors for high energy runoff. This saturation propensity distinguishes the channel erosion risk in such landscape areas from hill slopes of steep slope but small catchment area, where the risk of saturation excess runoff is relatively low. Under conditions where infiltration excess runoff generation is less common, the ability to identify critical saturation excess source areas is a major improvement on risk identified by local slope angle. This would appear to justify the use of contributing area in calculating landscape risk, rather than the use of gradient only as in currently used methods (e.g. Defra, 2005).

It appears it may be possible to qualitatively identify landscape points where deposition may occur due to changes in slope and plan curvature (flow dispersion). For example depositional features observed in the field downslope of features B and E illustrated in the previous chapter occur at dispersive flow points. Comparison of the outputs of a flow accumulation map with a RUSLE LS factor map (Figure 6.9) indicates that while the LS has its limitations, the partial calibration of the respective influence of slope and catchment area means it may be possible to distinguish flow paths which are erosive from those which are merely transportational or depositional (for example the circled areas of LS map Figure 6.9 show examples of validated erosive paths). However, from the perspective of sediment loss assessment this property may be less useful than for soil erosion assessment, as a transportational flow path with high connectivity to the permanent drainage network may still transport significant quantities of sediment eroded upslope. As described in Chapter 5, it is important to appreciate that fine sediment (fine slit and clay) is unlikely to be deposited within the landscape where continuity of flow path exists, regardless of dissipation in transportation energy.

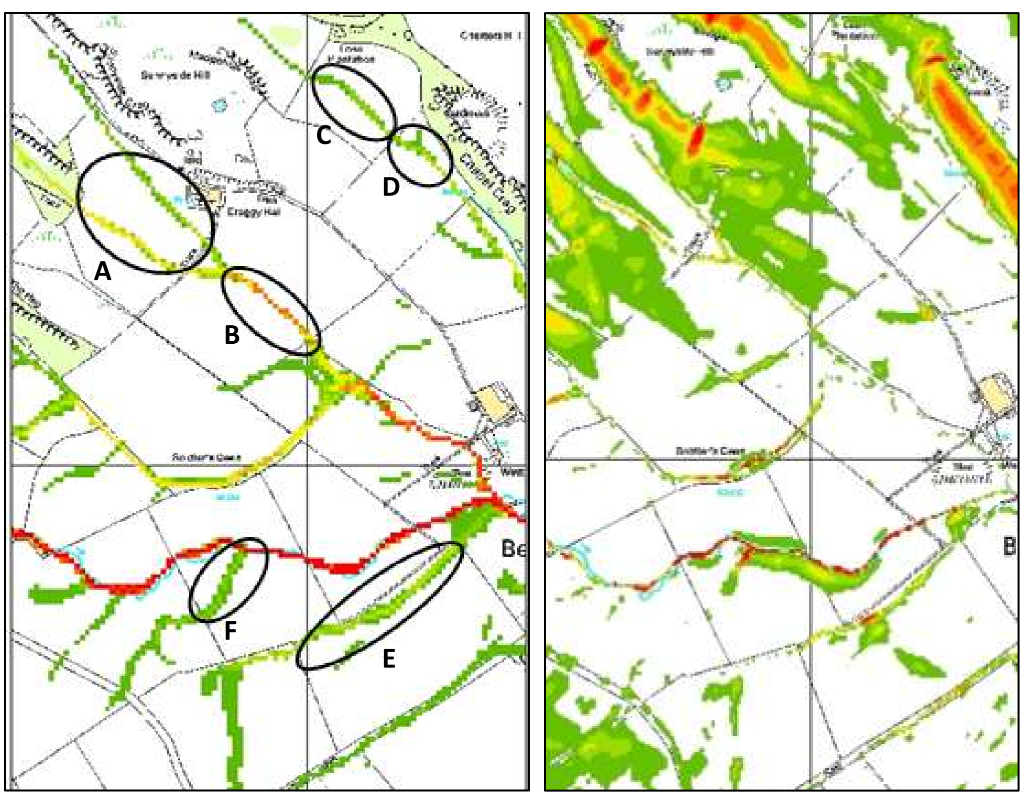

Figure 6.9: Flow accumulation and RUSLE LS maps of Belford catchment erosive features (circled).

Photo 6.1: 2002 erosion area A.

Photo 6.2: 2002 erosion area D.

Photo 6.3: 2002 erosion area F.

Photo 6.4 : Sediment deposition within a grassed buffer strip erosion area F 2010.

6.5 Summary and conclusions

The GIS terrain analysis method has potential as a risk assessment tool for sediment loss from agricultural land; major flow routes during runoff events are identifiable even using low resolution national Ordnance Survey topographic data. The critical threshold area for channel erosion initiation appears dependent upon other factors however. These may include both local and average overland channel profile gradient, topographic curvature (the propensity to flow concentration) and convergence of flow (catchment area increase per channel length). The potential of the use of additional topographic parameters in combination with catchment area appears worthy of further investigation.

Given the limited number of erosive features recorded in this catchment a larger dataset is required to interrogate whether common topographic attributes may be identified. A particular issue to be addressed is the susceptibility of the method to false positives and negatives. The small number of positives within the Belford catchment means a meaningful assessment is not possible at this scale. These issues are addressed in the next section.

6.5 Catchment scale study

6.5.1 Methods

Input data

The study area comprised a 15*15 km (225 km^2) area including the Belford Burn catchment, part of the remaining coastal streams catchment encompassing the Low, Elwick and Middleton Burns to the east and the catchment of the Horton Burn tributary of the River Till to the west (Figure 6.10). Both the coastal streams and River Till have been found to be affected by water quality issues related to agriculture in recent years, and are Catchment Sensitive Farming areas. The river Till in particular has experienced problems associated with excess sediment loads which affect the River Tweed fishery in this cross border catchment (Natural England, 2008). Soils of this area are dominated by slowly permeable soils formed in glacial till of the Dunkeswick and Brickfield Associations (Jarvis, 1984) in common with the nested Belford study area.

Figure 6.10: GIS risk assessment study area.

The digital elevation model for this area was derived from the national OS archive of digitised contour elevation data at 10 m grid cell resolution (higher resolution data was not available). The raster DEM was processed from point data using the ArcInfo “topo to raster” function to derive a smoothed image. Sinks/pits were filled using the Planchon and Darboux (2001) method automated facility of SAGA.

Erosion features were derived from free historical imagery from Google Earth. These images were dated as winter 2002/2003 and appear to preserve the effects of a large runoff event occurring on the 22nd October 2002.

Data Processing

The following topographic parameters were derived using SAGA GIS from the DEM for the study area:

- Contributing catchment area – Dinfinity method (Tarboton, 1997)
- Stream Power Index (SPI)
- LS index (Moore and Wilson, 1992)
- Plan curvature
- Topographic Convergence Index

Plan curvature is the second derivative of the DEM (slope of the slope) at 90° to the maximum slope aspect (the assumed direction of flow).The convergence index calculates the degree of aspect convergence for a grid cell relative to the maximum possible convergence (8 surrounding cell aspect directions converge) or divergence.

Statistical methods

Erosion features were digitised and statistical values for topographic parameters (mean, min, max) were derived for each individual feature. In addition parameter values at the point of feature *initiation* (upslope limit) were derived, as well as the catchment area at the start and end point of each feature. A total of 41 features were identified.

A comparative dataset of equal size (n = 40) of linear cross-field flow accumulation features was derived by randomly selecting the closest flow accumulation path to each erosion feature. As erosion features were only observed to form on arable land, selection of ‘paired’ features were similarly limited to arable fields. Initial analysis of the erosion features indicated that a threshold catchment upslope area of approximately 1 ha existed below which channel erosion initiation was rare. Accordingly, the paired dataset was limited to flow lines in excess of this threshold.

Parameter statistics (as described above) for digitised features were obtained (minus initiation/end point data).

Erosion feature parameters were compared with values for all grid cells in the study area using a 1 sample t test (where appropriate) to determine whether the features represented outlying values. Relationships between parameters were tested using Pearson's Correlation Coefficients and Variable Cluster Analysis. The predictive power of the parameters was tested by comparing feature values with the uneroded control dataset using a two sample t test, and using discriminant function analysis to predict group membership (eroded/uneroded). The minimisation of Wilk's Lambda was taken as the target of the latter.

Ground truthing of erosive features

Sites of channel erosion features observed on 2002/2003 imagery were visited in March 2011 to evaluate whether the linear erosive features had been replicated following the event occurring in December 2010. The major runoff event in December 2010 which caused major channel erosion in the Belford catchment (Chapter 6) was assumed to have also occurred in the larger catchment area given its proximity. At the time of survey no significant crop growth had occurred in the intervening period which may have removed the visual evidence of channel erosion.

6.5.2 Results

Topographic attributes of erosive features

	mean	**range**	**s.e.**	**CV%**
Length (m)	226.3	711.0	21.14	60
Start catchment area (ha)	26.7	375.9	9.89	237
Max catchment area (ha)	46.1	453.8	15.04	209
Start slope (°)	2.42	7.6	0.28	74
Max slope (°)	5.22	9.86	0.36	44
Start SPI	4.7	13.1	0.40	54
Start LS	7.3	48.2	1.65	145
Start plan curvature	-55.8	204.4	7.83	90
Start convergence index	-22.2	68.0	2.63	76
Mean slope (°)	2.6	8.6	0.23	57
Mean SPI	4.5	6.6	0.20	28
Mean LS	5.7	17.4	0.59	67
Mean plan curvature	-31.5	73.9	3.09	63
Mean convergence	-15.9	33.3	1.29	52

Table 6.1: Parameter statistics for channel erosion features (n=41).

Parameter	Min	Max	Mean	S.D.
Catchment area (ha)	0.01	3346	3.84	60.6
Slope (°)	0	32.6	2.88	3.19
SPI	-87.6	12.8	0.67	3.60
LS	0	308.4	1.27	3.04
plan curvature	-179.8	180	-2.57	42.7
Convergence index	-98.5	99.9	-0.38	15.0

Table 6.2: Cell parameter statistics for study area (225 km^2; cell n=2250000).

Erosive features were found to be located almost exclusively in extremely convergent landscape positions (strong negative Convergence index values), and were particularly found to be *initiated* at strongly concave, convergent locations (Tables 6.1, 6.2). Catchment area of erosion features was highly variable (Table 6.1). However, 88% of observed erosive features were initiated at catchment areas in excess of 1 hectare, 71% over 2 hectares and 56% over 5 hectares. Maximum catchment values are an order of magnitude larger than average cell values for the study area (Table 6.2), although heavily skewed by high outlying values. Twenty six out of the forty one features (63%) reached a maximum catchment area over 10 ha and seventeen (44%) over 20 ha. Stream power and LS index values for erosive features were significantly higher than for the study area as a whole (start SPI $t = 10.1$; $p<0.001$; mean SPI $t = 19.17$, $p<0.001$; start LS $t = 3.41$, $p = 0.001$; mean LS $t = 6.76$, $p<0.001$). Stream Power Index displayed low variability between eroded features relative to other parameters (Table 6.1) Average maximum feature slope appeared relatively low (Table 6.4), although significantly higher than mean catchment values ($t = 6.59$, $p<0.001$). Initiation point and mean feature slope were not significantly higher than the mean for the catchment however (Tables 6.1, 6.2). While mean gradients of erosive features were found to be relatively low, they were largely limited to average slopes in excess of 1 degree (forty of forty one instances). In twenty seven of forty one cases (66%) average slope exceeded 2 degrees. Of the eight erosive features where *initiation* slopes were less than 1 degree, the mean catchment area was 41.6 ha compared with 21.9 ha for the main group (26.7 ha overall mean).

Analysis indicates that erosion indices (stream power and LS) are more closely related to catchment area than slope (Figure 6.11; Table 6.3). Very strong correlation was found to exist between curvature index and plan curvature values, and between start and end catchment area (Table 6.3). End catchment area and plan curvature were therefore viewed as superfluous parameters and excluded from further analysis.

One difficulty in this remote analysis of historical imagery is the inability to differentiate between erosive magnitudes (the volume of eroded material per unit length) because field measurements to quantify channel volume are not possible. The only observable quantitative parameter which may be related to the volume of eroded material is channel length. However, channel length was not found to be correlated (Pearson's $p>0.05$) with any of the other measured topographic parameters

Channel erosion prediction

Parameter	Group	mean	range	s.e.	CV%	t	P
Max. slope (°)	non eroded	4.50	17.00	0.45	64	2.57	0.213
	eroded	5.22	9.86	0.36	44		
Mean slope	non eroded	1.83	4.85	0.17	60	1.26	0.012
	eroded	2.57	8.60	0.23	57		
Min. c.a. (ha)	non eroded	0.71	15.0	0.37	334	0.67	0.503
	eroded	1.19	22.5	0.61	328		
Max. c.a. (ha)	non eroded	15.0	138.2	3.66	154	2.01	0.051
	eroded	46.1	453.9	15.05	209		
Max. SPI	non eroded	5.97	4.48	0.16	17	2.71	0.008
	eroded	6.74	6.12	0.23	22		
Mean SPI	non eroded	3.37	6.14	0.20	38	4.08	0.000
	eroded	4.54	6.61	0.20	28		
Max LS	non eroded	13.2	38.7	1.29	62	2.80	0.007
	eroded	20.4	53.4	2.23	70		
Mean LS	non eroded	3.25	9.26	0.33	64	3.55	0.001
	eroded	5.66	17.37	0.59	67		
Min. plan curvature	non eroded	-116.2	146.8	5.33	29	0.02	0.987
	eroded	-116.1	163.3	6.88	38		
Mean plan curvature	non eroded	-26.2	53.0	2.11	51	-1.41	0.162
	eroded	-31.5	73.9	3.09	63		
Min. convergence index	non eroded	-39.2	45.1	2.06	33	-0.35	0.726
	eroded	-40.4	58.0	2.54	40		
Mean convergence index	non eroded	-14.0	25.6	1.03	46	-1.10	0.274
	eroded	-15.9	33.3	1.29	52		

Table 6.4: Comparison of eroded and non-eroded channel characteristics.

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	Start C.A.	End C.A	Start slope	Start SPI	Start LS	Start curv.	Start conv.	Mean slope	Mean SPI	Mean LS	Mean curv.
End c.a	0.914***										
Start slope	-0.198	-0.011									
Start SPI	0.437**	0.349*	0.381*								
Start LS	0.413**	0.486**	0.415**	0.510***							
Start curvature	-0.048	-0.029	0.232	-0.224	-0.039						
Start convergence	-1.07	-0.048	0.139	-0.302	-0.165	0.817***					
Mean slope	-1.67	-0.221	0.579***	0.097	0.092	0.072	0.025				
Mean SPI	0.440**	0.450**	-0.030	0.497**	0.490**	-0.219	-0.241	-0.001			
Mean LS	0.316*	0.335*	0.161	0.395*	0.525***	-0.116	-0.141	0.257	0.825***		
Mean curvature	-0.106	-0.101	0.069	-0.178	-0.229	0.379*	0.552***	0.133	-0.345*	-0.211	
Mean convergence	-0.086	0.616	0.081	-0.148	-0.156	0.384*	0.576***	0.120	-0.366	-0.266	0.920***

Table 6.3: Pearson's correlation coefficient for linear erosion feature attributes *p<0.05; **P<0.01; ***P<0.001

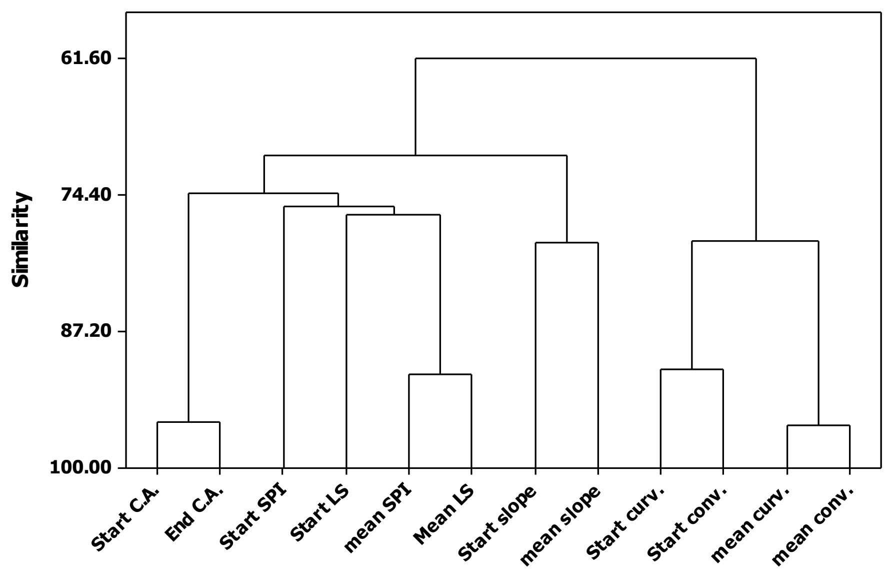

Figure 6.11: Variable Cluster Analysis plot for erosive feature parameters.

A significant proportion of the features in the eroded group are distinguishable from the control group in having steeper gradient; seventeen of the control group have average gradients below 1.5 degrees but only five of the eroded features. Eroded channels had significantly higher minimum and maximum SPI and LS values than the non eroded channels (Table 6.4). The within-group variability of these parameters is relatively low.

The best single predictor of group membership is mean SPI. Maximum SPI has very similar explanatory power, and maximum slope only slightly lower (Table 6.5). However, all single parameter models offered low discriminatory power (note 50% correct prediction is random in a two group dataset). The most efficient discriminant model includes all the parameters in Table 6.5, although the proportion of the variability explained by the addition of multiple factors was not found to be significant (Lambda F>0.05). The best dual predictor model includes mean slope and SPI, and retains almost all or the predictive power of the inclusion of all parameters (Table 6.5).

Parameters in model	% correct	Wilk's Lambda Chi-sq sig.	Discrimination %
Max CA	58.0	-	16
Mean slope	59.3	-	18.6
Max slope	61.7	-	23.4
Max SPI	64.2	-	28.4
Mean SPI	65.4	-	30.8
Mean slope/mean SPI	71.6	<0.001	43.2
All	74.1	0.001	48.2

Table 6.5: Best fit discriminant models.

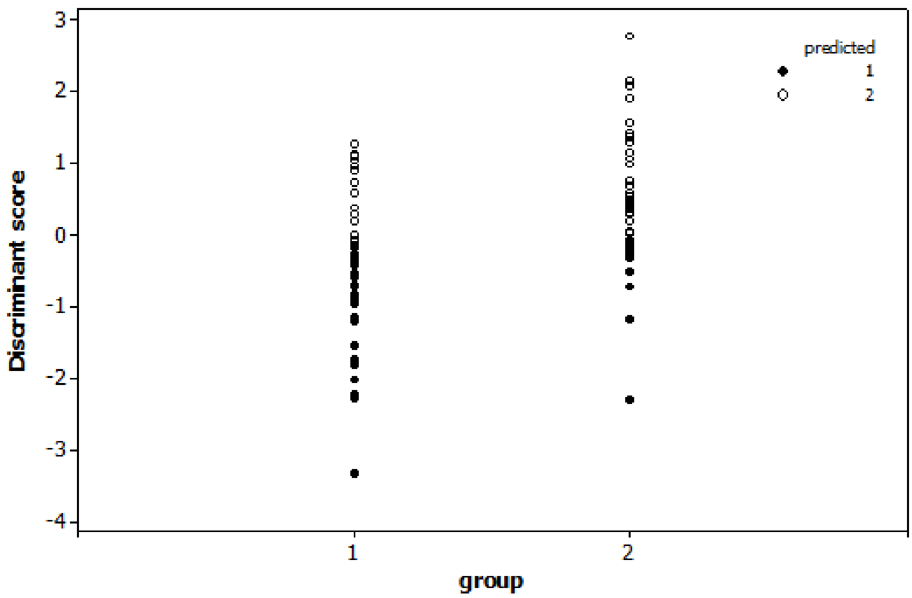

Figure 6.12: Group discriminant scores using mean slope and mean SPI as predictors.
1 = eroded, 2 = uneroded

Of the ten features in the eroded group (Group 2, Figure 6.12) wrongly classified as unerosive (27.8%) four occur on low gradients with very large catchment area (>20 ha). The very low gradients of these features results in low SPI as well as mean slope. The remaining six had a significantly lower SPI value than the correctly identified features ($t = 3.70$; $p = 0.008$) which was attributable to significantly smaller catchment size ($t = 2.44$, $p = 0.02$). There was no significant difference in average slope ($t = 1.71$, $p = 0.104$) between eroded and uneroded features.

Ground truthing of erosive features

Ground survey during 2011 demonstrated that channel erosion had recurred on a number of the features observed from historical imagery. A majority of the locations were under grass rotational stages at the time of survey and had therefore not eroded, with the exception of the feature in Plate 6.2 where the grass had recently been reseeded. Other sites had been recently cultivated and evidence was therefore obscured. Two sites (Plates 6.1 and 6.4) had been heavily gulleyed by events occurring since 2002, and these had become permanent landscape features.

A recurrent theme in this landscape is that the major flow pathways predicted by GIS terrain analysis were found to be underlain by tile drains (Plates 6.1-6.5). In a number of instances (see for example plates 6.1, 6.4, 6.5) these drainage networks incorporated permanent streams (former surface channels).

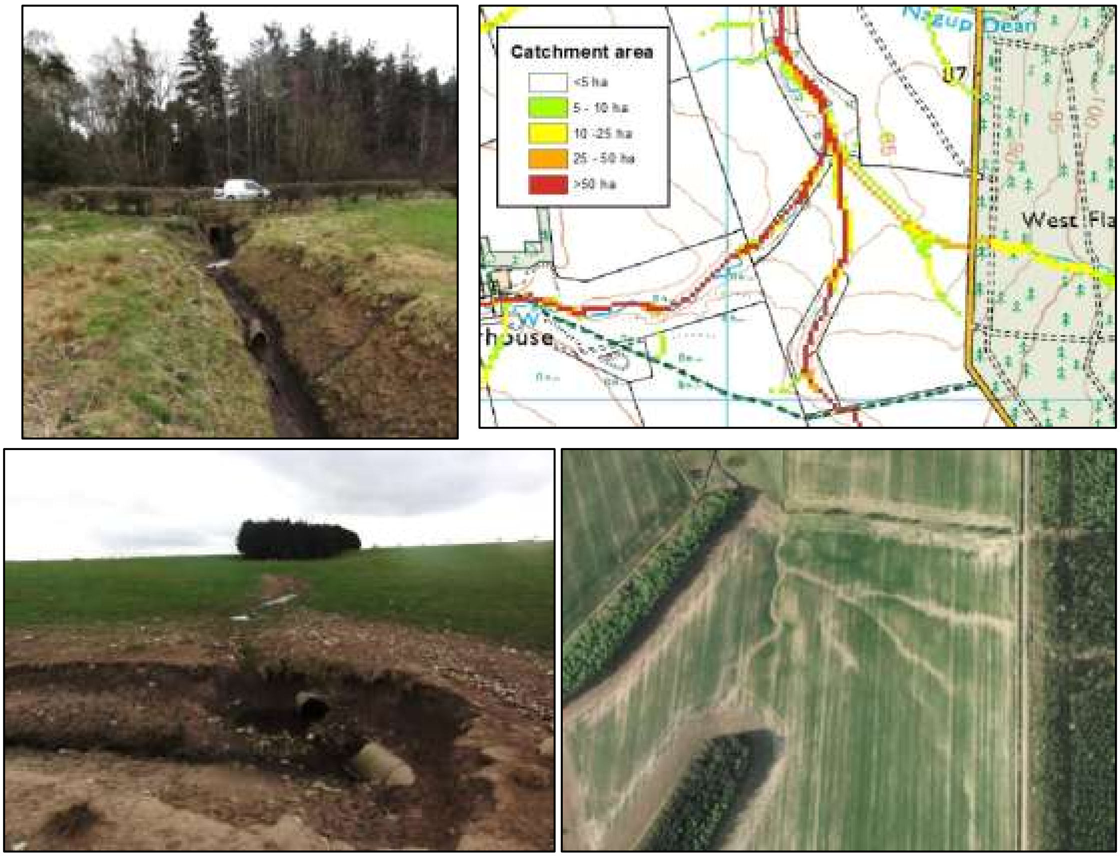

Plate 6.1: Clockwise from top right: flow accumulation map; erosion 2002; gulleying 2002 (view south – note exposed large diameter pipe); gulley origin at road cross drain from woodland (view west).

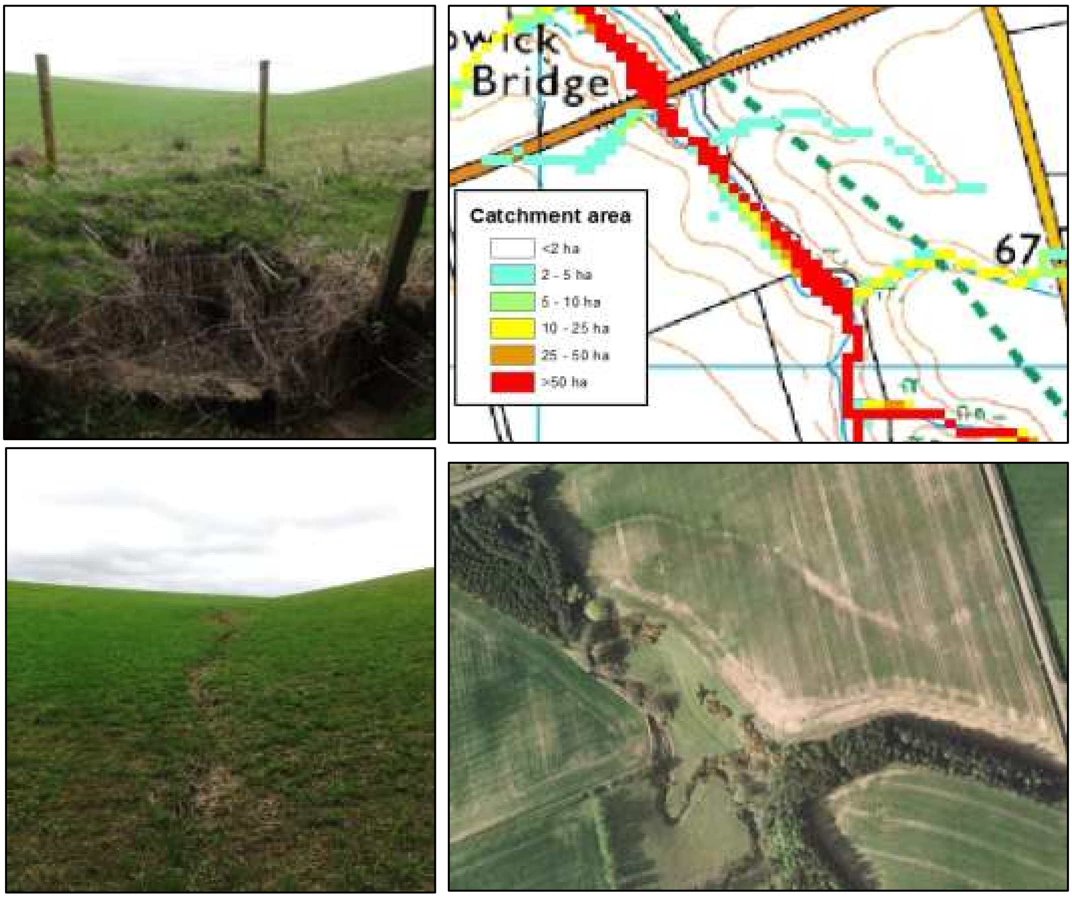

Plate 6.2: Clockwise from top right: Flow accumulation map; erosion 2002; erosion 2011; blocked tile outlet.

Plate 6.3: Clockwise from top right: contributing area; rill erosion 2002; tile outlet; rill erosion 2011.

Plate 6.4: Clockwise from top left: Contributing area; erosion 2002; stream exiting woodland to east of field; permanent gulley feature 2011 (piped stream outlet from northern flow path).

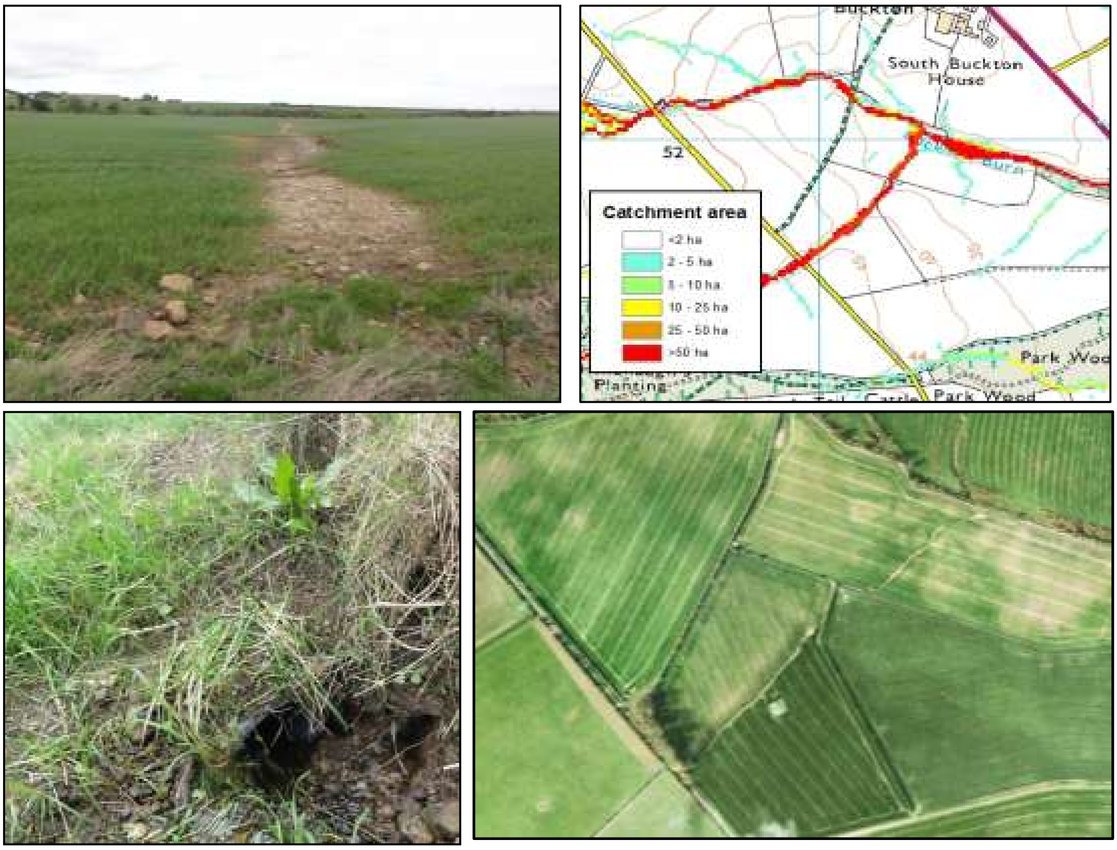

Plate 6.5: Clockwise from top right: Flow accumulation map; erosion 2002; stream pipe inlet; erosion 2011.

6.5.3 *Discussion*

Significance of channel erosion

The results clearly demonstrate that the large scale erosion features mapped using remote imagery are found exclusively in areas of high flow convergence; in dry valley 'thalwegs' and headwater depressions. In this landscape at least, large scale erosion features are not the result of susceptible land use on steep gradients. It may be the case that hill slope erosion is less visible (although perhaps more widespread) and/or that land management in this area excludes high susceptibility uses on steep slopes. However, given that there are a large number of arable fields in the study area located on slopes exceeding 7 degrees, the latter at least is unlikely. Evans (2002) reported that the proportion of erosion caused by channelled flow increases with soil clay content, as low erosivity soils resist the more limited shear energy generated by sheet flow, which may explain the pattern in this landscape characterised by slowly permeable soils formed in fine textured till. Long term study (Chambers and Garwood, 2000) has also indicated that channel erosion is the most prevalent form in England and Wales. This being the case, it is therefore arguable that channel erosion represents the greater risk on arable land in this study area. Moreover, because the degree of connectivity of natural drainage lines (temporary streams) with the permanent stream network is likely to be much higher than for the average hill slope, the sediment loss risk from channel erosion is likely to be substantially in excess of that presented by hill slope (sheet and minor rill) erosion processes.

The relatively gentle gradients over which linear erosive features are located demonstrates that steep slopes are not a primary control of the occurrence of these features; maximum gradients below 5 degrees account for thirty seven of forty one erosion features, and thirty nine of forty one mean channel gradients. These would not be would not be regarded as high risk sites (Defra, 2005) in the absence of concentrated flow. The absence of features on steeper slopes may be a function of the scarcity of areas with both high gradient and large catchment area under susceptible land uses. Steeper areas tend to either consist of non-convergent (planar) or divergent topography, and where convergent areas do occur, the landscape erosion risk may serve to preclude high susceptibility land uses due to the high probability of soil damage. It also appears the case that lowland areas favour arable land use as well as flow concentration (being lower in the catchment) and that the higher risk is therefore predominantly in these areas.

A considerable literature now exists regarding what is generally termed *ephemeral gulley erosion*. Ephemeral gulleys are regarded as those which repeatedly occur in the same landscape position as a result of concentrated runoff and are subsequently removed by cultivation. A growing consensus has indicated the importance of this previously neglected erosion process, particularly the large contribution made to sediment export (Foster, 1986; Desmet *et al*., 1999; Valcarel *et al*., 2003). Many authors now suggest that this form of erosion is the dominant source of water-borne sediment in agricultural catchments (Gordon *et al*., 2008). Poesen *et al*. (1998) estimated that 44% of total catchment sediment loads in central Belgium were the result of agricultural ephemeral gulley erosion, with bank and hill slope erosion contributions an order of magnitude lower. Vandaele *et al*., 1997 attributed this gulley erosion to infiltration excess overland flow to which capping under cultivation made these Belgian loess soils highly susceptible. Poesen *et al*. also report that estimates of ephemeral gulley sediment contributions measured in US catchments vary from 19 to 81%.

The fine textured slowly permeable soils of the current study are of relatively high shear resistance, but are highly susceptible to saturation excess, apparently despite the presence of artificial under-drainage. The landscape in this study may therefore be considered less susceptible to gulley sediment losses than the Belgian example, particularly given the comparatively large proportion of land in permanent or semi-permanent pasture. However, the apparent lower risk contributes to a current lack of consideration of the issue in land management decisions and knowledge transfer to land managers, which mean that mitigation measures are currently much less prevalent than those in high risk landscapes such as those described in Belgium, or the South Downs in the UK (Boardman 2002).

Prediction of channel erosion

The absence of predictive power attributable to catchment area may appear to suggest that erosion susceptibility is unrelated to this factor. However this is unlikely to be the case. In reality it appears that the adoption of a 1 hectare catchment area threshold was sufficient to 'pre-select' high risk areas. The fact that SPI values were significantly higher for eroded features than for the study area as a whole despite the fact that gradient was not supports this assertion. Catchment area was therefore an important factor in locating channel erosion features. Although the mean initiation catchment size of observed features suggests that high risk areas consist of catchments larger than a single field (26 ha), this statistic is strongly influenced by a small number of features of very large catchment size; eighteen of forty one features (44%) were initiated at catchment sizes less than 5 hectares. This indicates that features are not exclusively associated with large multi-field catchments. However, it was rare for features to initiate in catchments of less than 1 ha, irrespective of gradient, and while the use of historical imagery precludes assessment of erosive magnitude, it would appear likely that the features with larger catchment size result in a greater soil loss per unit length.

A significant proportion of the features in the eroded group are distinguishable from the control group in having higher stream power index and/or gradient. Seven of the control group have average gradients below 1 degree while only one of the eroded features. Seventeen features in the control group have gradients less than 1.5 degrees but only four of the ten features in the eroded group wrongly classified as unerosive (27.8%) occur on gradients less than 1.5 degrees. All four have unusually large initiation catchment area (<20 ha). This indicates that erosive energy of these features is a function of runoff volume (and upslope energy) rather than due to local gradient. The remaining six wrongly classified may be described as *minor* erosive features (although this may not be quantitatively verified given the reliance on historical imagery).

The discriminant output suggests that approximately 50% of eroded pathways may be distinguished from the uneroded group by steeper gradient and/or very large catchment area. It is of note that occurrences of the latter factor are only found in the eroded group, which may indicate that the high volume of runoff makes erosive channels inevitable in these cases. Previous studies have also found a combination of slope and catchment ('stream power') to be a good predictor of the location of ephemeral gulleys (Moore et al., 1988; Desmet *et al*., 1996; Kier *et al*., 2007). Other studies have demonstrated that gulley magnitude (length width, depth, volume) can also be predicted using these factors (De Santisteban *et al*., 2005; Hancock and Evans 2006). Desmet *et al*., (1999) attempted to calibrate catchment area and slope to derive accurate locations of ephemeral gulley formation in central Belgium. Their results demonstrated that while catchment area alone defined the *route* of the gulley, slope was much more important in defining the point of gulley *initiation*. However, their findings did not suggest that predictions of gulley initiation points were significantly improved by applying coefficients to contributing area and slope factors compared with a simple (uncalibrated) Stream Power Index. The implication of their findings is that there may be a critical threshold of catchment area and slope at which channel erosion occurs. However, as in the current study, as observed erosion features are the result of a runoff event of defined magnitude, this threshold is not transferable temporally or spatially.

Moore *et al*. (1988) demonstrated that the use of a stream power (catchment area*slope) in conjunction with the topographic wetness index (ln(A*s)) resulted in good prediction of the location of ephemeral gulleys in a small cultivated catchment, and that the combination provided a better prediction than either factor individually. Kier *et al*. also found that a combination of sediment transport capacity (stream power) and topographic wetness index provided the best predictor gulley location in a Mediterranean climate. As indicated by Figure 6.6, these factors are strongly related (inverse functions of effect of slope), and the fact that the combination of factors improves regression fit compared with either in isolation appears to indicate non-linearity; erosion risk may be underestimated for a large catchment area with low gradient using SPI only, or on an area of moderately steep slopes using wetness index. The former example has been observed in the present study. Although the index values produced in different studies are not transferable, this suggests there may be critical thresholds of gradient and catchment area where the interaction between factors is superseded by a high value of one factor.

Both eroded and uneroded feature groups occurred in strongly convergent topographic locations. As the control group was selected purely on the basis of contributing catchment area, this indicates that major flow lines (greater than the 1 ha catchment area threshold) are only predicted in convergent areas. While this may appear a truism, contributing areas on this scale may theoretically occur on planar slopes (as often occurs in classic gulley erosion). The correlation between catchment area and flow convergence in this study means that although both plan curvature and convergence index appear strong indicators of the location of high risk sites, the predictive power provided by catchment area alone was not improved by the use of these factors. Previous studies have suggested that the use of curvature in addition to catchment area and slope can aid location of gulley risk sites (Thorne *et al*., 1990; Moore *et al*., 1991; Parker, 2010). The resultant "Compound Topographic index" is an attempt to differentiate flow along valley thalwegs from that occurring on planar slopes. However, the present study demonstrates that in this landscape at least, large contributing areas (<1 ha) do not occur in none convergent landscape positions, and this distinction is therefore unnecessary.

In general it appears from the analysis that there are only minor differences between the eroded and control groups in the present study; although slopes were generally slightly steeper, resulting in higher SPI values, reliance on an SPI threshold value is inappropriate as indicated by a number of false positives in the control group (Figure 6.12), which suggests that the erosion risk is affected by other factors including land management. Sowing date for example, is known to affect susceptibility to erosion (Defra, 2005) primarily through soil protection from erosive energy (including channel shear).

Ground-truthing of erosive features illustrated that they commonly follow the path of artificial piped drainage in this landscape. This is not surprising given the necessity for under-drainage for arable land use in these slowly permeable soils, and the fact that an effective drainage design requires the main collector to follow the lowest point of the topography. In a number of cases these pipes involve the integration of a natural stream into a tile drainage network as was observed in the Belford study catchment. The size of the catchment of these streams means discharges exceeding pipe capacity are probable in large runoff events. In some circumstances (see Plate 6.5) the size of the catchment would appear so large that the capacity is likely to be exceeded so regularly that large-scale channel erosion becomes inevitable rather than probable. In at least two localities attempts to pipe streams (exiting plantation woodland) have resulted in permanent gulleying (Plates 6.1 and 6.4). The observation of major channel erosion associated with tile drainage exceedance does not appear to have previously been documented in the UK, although Oyegarden (2004) reports that in studies conducted in Norway severe gulley erosion was observed down to the depth of tile drains in similar circumstances.

6.6 Summary and conclusions

Landscape factors appear key to controlling the occurrence of large (visible) channel erosion features. These features were found to be limited almost exclusively to landscape positions with a high degree of convergence/curvature such as natural dry valleys and headwater depressions, and not due to high risk land use on steeply sloping land. While erosion channels are gentle (less than 5 degrees), their occurrence is limited to average slopes in excess of 1 degree excepting where catchments are very large. While catchment area does not appear to need to be particularly large to generate erosive features, in most cases catchment area is in excess of 2 hectares in this study. This suggests that such erosion channel features may be relatively few in number and absent from many fields. Accordingly, the identification of at risk areas on a farm or catchment basis would appear practical.

While contributing area and slope can successfully indicate potential risk areas, the results indicate that knowledge of these factors alone is insufficient to determine whether channel erosion occurs. This is also a function of management (the degree of susceptibility risk and length of exposure) and weather (the coincidence of runoff events with susceptible periods) which interact to control both the geographic and the temporal occurrence of channel erosion. Minor topographic features such as field boundaries and roads may also affect risk locally. It is also likely that topographic data resolution affects the accuracy of estimations in some cases. Therefore a complete reliance on remote sensed risk assessment is not appropriate.

Despite limitations, the results demonstrate that the important sediment loss risks presented by concentrated runoff pathways may be successfully identified using GIS terrain analysis techniques. Management decisions and mitigation techniques may therefore be targeted accordingly where ground survey confirms the existence and continuity of these pathways. Sole reliance on remote sensing is not recommended, but may be effectively used to focus mitigation activities on at-risk areas, particularly when covering large catchments.

Chapter 7: Mitigation options

7.1 Sediment loss processes in the study landscape

This study was targeted at lowland landscapes with slowly permeable soils typical of the North East of England, and also common in the lowlands of northern and midland England, Wales and Scotland. These landscapes are characterised by a high proportion of tile drained land, particularly (although not exclusively) land used for arable cropping. The results presented in Chapter 2 and 3 support previous evidence that where present under susceptible uses, tile drains are a significant sediment loss pathway to surface waters. Failure to address tile sediment losses in these landscapes may therefore lead to failure to achieve water quality objectives with regard to sediment where catchment sediment problems are identified. However the nature of this pathway has significant implications for the suitability of different mitigation options as described below.

While tile drainage pathways appear to dominate sediment losses for the majority of the agricultural catchment area under susceptible arable uses, critical source areas exist where topographically-controlled flow accumulation occurs under these uses, classically in dry valleys where piped drainage of has replaced surface drainage, and the more limited capacity leads to drainage excess overland flow. The interaction between surface and drainage pathways is somewhat complex, and the relative importance is dependent both on hydrological conditions and the efficiency and capacity of the drainage network installed. Evidence presented in Chapters 5 and 6 illustrates that during major runoff events the surface runoff pathway dominates the quick flow hydrological response, and particularly sediment losses due to the propensity for erosive concentrated runoff. In contrast Chapters 3 and 5 demonstrate that piped drainage systems, even those draining large sub-catchments are generally sufficient to carry all rapid runoff and limit surface runoff to a small number of annual large events (which may not coincide with susceptible land use periods). In the intervening period the tile drainage represents the major pathway, apparently increasing sediment loss per unit area and the duration of exposure to high sediment concentrations from received arable runoff water. This is a classic chronic/critical problem, and the appropriate mitigation response is dependent upon whether either or both are considered to represent a sufficient impact upon off-site receptors to demand action.

Runoff response to rainfall events is often may be muted during the growing season due to the storage capacity of these deep fine textured soils, but appears of high magnitude once soils are at field capacity due to the absence of significant lags associated with the groundwater pathway. The susceptibility of such catchments to runoff events will vary with average annual water balance (P-ET), and more frequent occurrence of major runoff events is therefore likely in North West England, Wales and Scotland than in North East England, and may therefore mean that there is a greater susceptibility to surface runoff derived sediment loss to water in these areas, as well as greater tile runoff and sediment generation.

The study landscape included a relatively high proportion of lowland livestock agriculture, where water erosion of field surfaces is likely to be a less prevalent source of aquatic sediment, particularly in comparison with losses due to channel bank damage. However, the high nutrient loads associated with sediment derived from arable soils coupled with other associated pathogens such as pesticide residues mean that control of this source appears of critical importance to achieving water quality objectives. While this fact provides justification for the concentration on arable land processes, the relative importance of accelerated bank erosion in catchment fine sediment losses appears an important for further research in these landscapes.

Implications for mitigation

The importance tile drainage sediment losses under low-moderate runoff conditions, and of concentrated overland flow derived sediment during major event peaks are major considerations affecting the efficacy of mitigation options for the study landscape. Adopting a classic *source-pathway-receptor* conceptual model of pollution control, the ability to influence the pathway with regard to tile drainage is severely restricted given that the efficiency artificial sub-surface drainage is critical to removing the wetness limitation presented to arable agriculture in this land type. Therefore even if it is accepted that the reduction in runoff connectivity associated with reduced drain flow and increased surface runoff will lead to reduction in sediment export ratio, pathway mitigation is extremely unlikely to be a mitigation option which is acceptable to arable farmers.

Concentrated runoff occurring at event peaks places particular demand on mitigation features in that they must by definition be designed to function at maximum efficiency during high flows, as failure to do so may render them partially or totally redundant. Whether *in-field*, *field margin* or *in-channel* mitigations are most appropriate will depend upon both the relative potential to mitigate sediment at peak flows, and desirability and cost implication from a farming system perspective. An additional consideration is whether runoff quantity (flood control) is an consideration along with runoff quality: Some mitigation options may reduce the susceptibility of land surfaces to sediment mobilisation without affecting the volume or velocity of runoff.

A total or partial replacement of piped drainage with open ditches may be a specific option which allows on-farm mitigation (in channel features). This would also appear likely to reduce sediment derived by overland flow, by removing the dry valley ephemeral gulley phenomenon described in Chapters 5 and 6 and elsewhere. It is important to note that the reason for existence of piped drainage replacing natural streams or ditches has been to aid simplicity of machinery access and movement, often critical to the feasibility of arable agriculture in these areas. Whether the restoration of surface water courses is acceptable would require careful consideration in discussion with land owners. However Chapter 6 in illustrated that in some circumstances the susceptibility of these concentrated pathways to gulley erosion has resulted in *impromptu* reversion to surface channels, and it may be that the attempt to culvert large catchments does not represent efficient management in these circumstances.

7.2 Mitigation options review

Previous chapters have illustrated that the issue of agricultural sediment pollution is complex; treatment of aquatic sediment problems simply as the off-site effects of soil erosion ignores two key points. Firstly agricultural sediment can be derived from important sources other than field soil erosion. These include accelerated channel bank erosion and losses from man-made surfaces such as farm tracks and steadings. The impacts of these sources are independent of in-field soil erosion processes. Secondly landscape connectivity - the spatio-temporal variation in the sediment delivery ratio, means that where high erosion risks exist, the risk of soil reaching an aquatic receptor may be low (hence sediment loss risk does not equate to soil erosion risk). Conversely as threshold sustainable sediment inputs may be an order of magnitude lower than agriculturally sustainable soil loss rates, a field with a particularly high sediment export ratio may be problematic despite relatively low erosion risk. Reduction in the connectivity of the sediment pathway without management changes in field may be preferable in these circumstances.

These two points mean that the range of potential mitigation options for agricultural sediment is more diverse than for soil erosion. These options may be broadly defined as *in-field*, *field margin* or *in-channel* options. The most appropriate mitigation option or combination of options will vary according both sediment source and pathway. The evidence for the effectiveness of these options are reviewed and discussed in this chapter. This review is directed particularly to the implications of the findings of the current study, but is wider in scope, considering a wider range of sediment source and mitigation responses. Direct UK evidence is supplemented with that from programmes in other countries where necessary.

7.3 In-field options

7.3.1 Conservation agriculture

Conservation agriculture is a generic term applied to practices designed to minimise soil disturbance and exposure in cultivated systems. Its three core principals are minimal soil disturbance, permanent vegetation cover and crop rotation. It is an important component of many conservation programmes around the world. Although it should be acknowledged that no single solution exists suitable for universal application, the basic principles may be applied to upland arable systems in diverse agro-climatic zones. The use of the term Conservation Agriculture is used here as a general umbrella for practices including reduced tillage and residue retention either used in combination in accordance with the formal definition, or as individual practices aimed at reducing soil disturbance.

The use of conservation agriculture principles (minimum tillage, residue retention) is much less prevalent in the UK and Europe than other parts of the world. Higher production costs, greater management difficulties and lower yields appear to be controlling disincentives, although the absence of knowledge transfer programmes which have been crucial in widespread adoption in other countries also appears to be a major limiting factor (Holland, 2004). A large literature exists to support the benefits of these practices in controlling soil erosion (Fawcett *et al*., 1994; Uri *et al*., 1998), although much of this evidence comes from plot studies, which are difficult to translate into *sediment* loss due to uncertainties over sediment delivery ratios at the landscape scale. Some studies (mainly from the US) have shown the effects of conservation practices on sediment loss to water directly; Owens *et al*. (2002) found annual sediment losses of 532, 828 and 1152 $kg.ha^{-1}$ for no till, chisel plough and disk cultivation respectively in a two year paired field-scale watershed study in Ohio. Zhou *et al*. (2009) suggested that zero tillage agriculture could reduce sediment loss by over 90% in susceptible catchments on the basis of Water Erosion Prediction Project (WEPP) simulations.

Studies of the effect of conservation principles on rates of soils erosion are limited in the UK (Holland, 2004), and those which directly consider the effect on sediment losses are more limited still. Some evidence is available from plot studies; Quinton and Catt (2004) considered ten years of runoff and erosion data from the Woburn Plots on sandy soils, and found that a combination of minimum tillage and cross contour ploughing reduced average event soil loss by approximately 75%, although neither treatment in isolation had a significant effect. The major effect is attributable to reductions in runoff afforded by both treatments (and greatest in combination). A comparison of concentrations of sediment and P in runoff from small plots on Greensand and Chalk soils showed them to be consistently lower when the soil was minimally tilled rather than ploughed (Withers *et al*., 2007). The study also found that early sowing reduced losses by an order of magnitude and that tramline compaction was a key factor on the chalk soil, and surface sealing for the greensand soil. Withers *et al*., (2006) found significant differences in runoff quantity, and sediment concentrations where tramlines ran up and down slope compared to areas without tramlines for conventional tillage, but no significant difference with minimum tillage. The authors suggest that this is due to the lower depth and greater consolidation of tramlines in untilled systems. The trial was conducted in sandy soils very susceptible to capping (so even wheelings have little resistance to erosion). Stevens *et al*., (2009) tested the effect of reduced tillage in Leicestershire. They found that the use of shallow depth disk cultivation resulted in no significant reduction in sediment losses compared to a conventional system (mouldboard ploughed). This evidence appears to support the contention of proponents of conservation agriculture principles, that residue retention as well as reduced soil disturbance is essential to prevent soil mobilisation by water. Blanco-Canqui *et al*. (2009) also found sediment loss under zero tillage to be as high as conventional tillage in plot studies where soils are exposed by residue removal.

Recently the Defra-funded MOPS (Mitigation Options for Phosphorus and Sediment) project has sought to address some of the knowledge gaps by testing the efficiency of a range of mitigation options in England and Wales. Deasy *et al*., (2009a,b) report that of the measures tested to date, tramline management was the most consistently effective way of reducing sediment losses, but that minimum tillage was the only mitigation measure to have net cost savings for the farmer (others are roughly neutral). However, this cost saving considers only the savings in time and energy, not the marginal costs of change of system or the potential for yield decrease. MOPS found a 75-99% reduction in suspended sediment delivery was afforded by running a tine through the wheelings although this was trialled only in sandy and silty soils. Contour cultivation resulted in sediment reductions of 40-43% and minimum tillage of 45-79%. In all cases there was a similar reduction in runoff (Deasy 2009b) which indicates that a large proportion of the benefit was in maintaining infiltration rates to prevent Hortonian runoff generation. Stevens *et al*. (2009) report results from similar experiments at one of the sites over slowly permeable clay soils and found *no* reductions in runoff *or* erosion as a result of reduced tillage, which suggests that the benefits may be more limited under saturated soil conditions.

Evidence appears to suggest that the economic justification for adoption of conservation agriculture is dependent on high costs associated with lost productivity and off-site environmental damage – which means that cost effectiveness is dependent on very high existing rates of erosion; where erosions rates are lower, the benefits are more limited. For example Leys *et al*. (2007) used plot simulations in Belgium to compare CT and ZT on different soils in different states. They found that while ZT reduced runoff and soil losses in most cases, the large reductions occurred primarily where conventional tillage soil losses were very high. This has been used to justify a targeted approach focussing resources on the critical areas of a landscape where the greatest risks occur. However, in circumstances where sediment but not erosion problems are identified (i.e. where soil loss rates are sustainable), both the absolute sediment mitigation effectiveness and the cost effectiveness are much less well proven.

While the impacts of reduced tillage techniques appear well grounded in evidence particularly from North America, quantitative evidence regarding the success of major investment in these programs in improving aquatic habitats is scarcer. One study (Richards and Baker, 1998, cited in Uri *et al*., 1998) was found detailing efforts to reduce the eutrophication in Lake Erie that began in the early 1970s. A monitoring station was set up at Bowling Green, Ohio on the Maumee River which feeds into Lake Erie. The major crops are corn, soybeans and wheat. Between 1975 and 1995, implementation of conservation tillage increased from less than 5% to more than 50% of planted acreage. The adoption of conservation tillage led a reduction in suspended sediments of 19%. This finding indicates much more limited effectiveness in water quality mitigation at catchment scales than soil erosion studies would indicate.

7.3.2 (In-field) runoff control features

Grass strips

Of the available control structures, terracing is not considered here as a viable sediment control option for UK conditions; previous experience from soil conservation programmes has suggested very high construction and maintenance costs, which are only justified by significant benefits in terms of soil conservation (Bradley et al., 2007; Zhou *et al*., 2009). Under UK agricultural systems the need to cultivate erosion-prone soil on steep slopes is relatively low, meaning that the substantial subsidies required to promote terracing programmes are unlikely to be a desirable option in comparison with other measures.

Specific studies of the effectiveness of *in field* grass strips (rather than *field edge* grassed buffer strips) have been limited. Laboratory simulations have suggested high performance as sediment filters (Ligdi and Morgan 1995). Dabney *et al*. (1995) found contour grass strips to range from 15% to 79% in their effectiveness. However, Stevens *et al*. (2009) found no significant reduction in runoff or sediment loss with the use of grass strips on clay soils on gentle slopes (2-6 degrees). In general the effects of these strips may be regarded as similar to field margin grassed buffers in slowing flow, promoting infiltration and dispersing rill pathways. The evidence for these features is discussed in detail in a later section.

Grassed waterways.

The importance of concentrated (channel) erosion for sediment loss processes has been demonstrated in this study as well as others (Foster, 1986; Poesen *et al*., 1998; Desmet *et al*., 1999; Valcarel *et al*., 2003). Evans (1990) has suggested that this may be the most prevalent form of erosion in fine-textured soils in England and Wales. The mitigation response to this form of erosion may differ significantly from hill slope erosion processes; as demonstrated in Chapter 5, the erosion risk may apply only to a small area of a field, which means mitigation may be appropriately targeted. Grassed waterways or 'swales' are permanent uncultivated strips which follow recurrent flow pathways, particularly in valley bottoms. Evidence has indicated that they may be highly effective in reducing sediment losses. The effect of the permanent vegetation may be twofold; firstly the increased flow resistance offered by standing vegetation may serve to slow runoff as well as prevent surface sealing which restricts infiltration rates. In this way grassed waterways have been shown to reduce runoff volumes. Secondly, the increase in soil shear resistance afforded by a dense root system reduces soil mobilisation by flowing water relative to cultivated land.

Dermisis (2010) demonstrated that increasing grassed waterway length reduced the generation of runoff by approximately 20% and sediment by 60% on average in a small Iowa catchment. Fiener and Auerswald (2003) observed reductions in runoff volume of 90% and 10% for catchments with GWW compared to paired catchments without, and sediment delivery reductions of 97% and 77% respectively in a 7 year study in Germany. They attribute these reductions to infiltration and reduction in flow energy (due to reduced volume and velocity) rather than shear protection.

Reductions in flow volumes apply only to circumstances where runoff generation is attributable to infiltration excess (where soils are unsaturated). Such conditions occur where rainfall intensity is high, gradient is steep or soils are particularly susceptible to capping (or some combination of these three). However, this type of runoff is applicable only in soils with free drainage; where sub-soil permeability is restricted these convergent landscape positions will rapidly saturate once runoff is initiated on adjoining hill slopes, and reductions in runoff volume do not occur. Evrard *et al*. (2008) observed no infiltration into grassed waterways, and for their study in Belgium suggested that grass strips have a *lower* infiltration rate than most cultivated soils. In these circumstances the effectiveness of grassed waterways is due primarily to increased shear resistance and possibly to flow velocity reductions, but importantly this means that the entire flow path length requires mitigation to prevent channel erosion (unlike the former circumstance where surface runoff and resultant erosion may be eliminated by a more limited grassed waterway length). Under conditions of saturation-generated concentrated runoff, full length grassed waterways would appear likely to be effective provided the major sediment source is the flow channel rather than the hill slope.

Zhou *et al*. (2009b), using WEPP simulations for arable catchments in central Iowa, suggested that grassed waterways were capable of reducing sediment yield from 5.09 to 2.67 $t.ha^{-1}$ for a chisel plough system, but had very little effects where reduced tillage (strip tillage or no till were applied). This can be explained by the effectiveness of these reduced tillage systems in reducing sediment yields, (1.2 and 0.74 $t.ha^{-1}$ respectively). It is notable that the climatic and edaphic risk factors for soil erosion appear much higher under these study conditions than for the UK, and the remaining sediment loss risk for chisel plough systems (2.67 $t.ha^{-1}$) would appear to illustrate that while the grassed waterway was able to prevent channel erosion, hill slope (rill and sheet erosion) derived sediment was only partially mitigated. Zhou *et al*. (2009a) consider the cost-benefits of conservation structures and also suggested that grassed waterways were most justified in the absence of in-field conservation (zero tillage). In assessing the cost-benefits however, they conclude that the economic justification applies mainly in areas of high water erosion potential; the lost productivity due to soil loss and severe effects on the environment justify extra costs and potentially reduced yields. However in “flat” areas where soil losses are lower they view both zero tillage and structural measures (terraces and grassed waterways) as poor value.

7.3.3 Land use change

Targeted land use change from high to lower risk activities on erosion-prone land is an alternative to management system changes such as tillage practice. In a UK context the land use change usually entails change from high risk cultivation (vegetables or maize in particular) to either cereals or grasses. In general this may be seen as the ‘option of last resort’, but when targeted at specific areas of a farm or catchment (i.e. individual fields) does not necessarily mean major changes in farming system. It may also be argued that change of unsustainable practices in these areas represents a rational economic decision. However, when land use sustainability applies to off-site rather than on site resource protection, incentives may be required. Boardman *et al*. (2009) assess available options for sediment loss mitigation in regard to prevention of muddy floods in the South Downs, and point out that while reversion to grass is effective, it is rarely an option farmers consider, particularly as livestock farming is a relatively minor component in this area.

In the United States targeted grassland reversion is a common component of soil conservation schemes, and has often been applied as a blanket measure where soil erosion problems are particularly severe. Kuhnle *et al*. (2008) found a 20% conversion of the Goodwin Creek watershed in Mississippi from cropland to permanent cover between 1982 and 2005 under the Conservation Reserve Program resulted in more than a 60% reduction in annual sediment yields. The cost of compensation to farmers on such large scales to achieve *water quality* objectives is a more difficult argument to justify, although it may be appropriate where targeted at critical catchment areas. Khanna *et al*., (2003) investigated the potential of applying a new method of identification of critical catchment areas aimed at reducing sediment in rivers in Illinois. Only land within 900 feet of water course was considered. Potential overland flow channels were generated by assessing topography. The study adopted the Agricultural Non Point Source model (AGNPS) to simulate sediment yields and an economic model to estimate the costs of achieving different sediment reduction targets. They estimated that a 20% reduction in sediment could be achieved by retiring only 8.4% of cropland in the target area – less than 3% of the total cropland in the catchments. They suggested that targeting highly sloping erodible cropland in a narrow zone adjacent to streams was most cost effective, and that existing target schemes should include less flat floodplain land which was not a cost-effective use of financial incentives.

A change in arable land use from winter to spring sown cereals may have the potential to reduce annual sediment losses by reducing the period of soil vulnerability to erosion from a prolonged winter period to a shorter window following spring cultivation prior to crop establishment. Such a benefit from spring cropping is dependent on winter cover being maintained (either under grass in a mixed rotation or through residue retention or cover cropping following a previous arable crop). Studies of the effects of such a change are not available from the UK, but the practice has been adopted in Norway to mitigate high sediment and phosphorus losses in a number of catchments. Lundekvam and Skøien (1998) citing evidence accumulated from plot studies suggested spring tillage reduces the annual soil loss by 90% compared with autumn ploughing in Norway because most of the runoff and soil loss occurred between October and March. Bechmann and Stalnacke (2005) studied three catchments in southern Norway monitored from 1983-2001 and showed that conservation measures produced significant reductions in stream SS and total P in the most polluted catchment dominated by arable production. From the early 1990s, between 25 and 36% of the arable area was not cultivated in autumn in this catchment through agricultural subsidies targeted at high soil erosion risk areas. As highest SS losses occurred in winter and early spring, this was effective in reducing annual losses significantly during this period (no significant reductions occurred at other times of year). Reduction in sediment losses resulting from a switch to spring cropping in the UK is also possible, although it should be noted that the soil-climatic interaction in the North east of England where the current study was undertaken means the potential for regular spring sowings is generally limited (Jarvis *et al*., 1984).

7.3.4 Livestock management

An investigation of sediment sources across England and Wales using the PSYCHIC prediction tool (Collins and Anthony, 2008a) suggested that stream bank erosion represents a relatively minor source of sediment overall. However, the outputs are constrained by the inability of the model to predict accelerated bank erosion due to disturbance. While the estimated contribution made by bank sources varies between catchments, this difference is mainly attributable geomorphological controls on bank stability and the relative inputs from other sources. Data regarding the occurrence of accelerated bank damage by stock is mainly unavailable and difficult to generalise. Other research has suggested that in livestock dominated catchments, bank erosion can be both the primary sediment source and that this source can reach problematic levels with regards to aquatic ecosystem health (Walling *et al*., 2003). Collins *et al*. (2010) studied the effects of bank fencing in rivers in the south west of England where bank erosion had been demonstrated to be the main contributor to gravel sedimentation in *salmonid* spawning grounds. Revisiting the same sites ten years after the initiation of fencing suggested the contribution of bank (sub-surface sources) to gravel siltation was dramatically reduced relative to surface sources in several catchments. It is acknowledged by the authors however that a reduction in absolute sediment inputs can presently only be inferred. There is currently a shortage of quantitative evidence regarding the effects of bank protection on sediment regimes in the UK.

Some further evidence is available for US catchments. Zaimes *et al*. (2008) tested rates of bank erosion under different riparian land use in Iowa and found that stream bank fencing (22-47 $t.km^{-1}.yr^{-1}$) or forest buffer strips (5-18 $t.km^{-1}.yr^{-1}$) dramatically reduced bank erosion rates compared with continuous grazed banks (197-264 $t.km^{-1}.yr^{-1}$) or rotational grazing (124-153 $t.km^{-1}.yr^{-1}$). However, these extremely high figures suggest severe erosion issues which are difficult to explain - bank erosion rates in row cropped fields was found to be higher (304 $t.km^{-1}.yr^{-1}$) than any of the grazed systems. Owens *et al*. 1996 tested the effect of stream fencing in a small (26 ha) Ohio catchment with permanent cattle grazing. Average annual sediment concentrations were reduced by over 50% and annual soil losses by 40%.

Schilling *et al*. (2011) recently tested the effects of reversion to native prairie on sediment loads in paired catchments in central Iowa. They found that despite the fact that a 46% reduction in soil loss was predicted following land use change in one of the catchments, there was no significant change in sediment loads. There was a 9.2% increase in row crops in one catchment and a 14.9% decrease in the other due to native prairie reconstruction. Estimates indicated that bank erosion is the dominant sediment source in the mitigated catchment, and rill and sheet erosion in the unmitigated. The implication is that the benefits in reduced field erosion following grassland reversion were entirely offset by an increase in bank erosion.

These findings indicate that bank erosion can be severe enough to cause excessive sediment and that the issue may be successfully addressed by limiting stock access to stream banks. The problem is that plot scale testing which forming much of supporting the evidence for other mitigation techniques is not available or suitable for the effects of fencing, which can only reasonable be measured at the small catchment scale or larger. The lack of suitable risk assessment techniques also mean that bank erosion problems may also be underestimated.

Reduced stocking

High stocking densities can lead to loss of vegetative cover and increased risk of sediment loss. This has particularly been noted as an issue in the UK uplands where topographic and climatic controls mean that relatively high sediment losses can result from vegetation damage. It has therefore been suggested that reduced stocking densities may result in significant reduction in sediment loses. Evans (2005) reports evidence gathered from long term studies in the Peak District indicates that erosion caused by overstocking slows rapidly once livestock densities are reduced. Mchugh *et al*. (2002) observed that erosion caused by sheep was the most prevalent in the uplands of England and Wales and that a short term study of changes in erosion rates suggested improvements derived from a reduced stacking density were likely to result in re-vegetation of exposed sites and reductions in erosion. Posthumus and Morris (2010) suggest that current changed economic circumstances for livestock enterprises are likely to lead to extensification, and a reduction in erosion (and runoff) pressures.

7.4 Field margin options

7.4.1 Buffer strips

The use of vegetated buffer strips as a mitigation option for sediment loss from cultivated land has received a large amount of research attention. This may be attributable to the desirability of achieving resource protection objectives through what may be regarded as both the lowest technology and lowest cost of mitigation options. A number of studies have demonstrated that buffer strips may be highly effective; Lui *et al*. (2008) suggest from evidence of over 80 studies that sediment removal efficiency varies from 45 to 100%, with increasing removal up to 10 m width. Yuan *et al*. (2009) also review sediment trapping efficiency of riparian buffer strips. They find that while buffer strips of 3 m width are efficient at low slope angles, they may be less effective where slope exceeds 5 degrees. They conclude that studies suggest buffer strips of 5 m width remove 80% of sediment and that a 6 m buffer strip is sufficient, regardless of slope or buffer vegetation type.

The effectiveness of sediment removal appears dependent on particle size, although the evidence is somewhat conflicting. Gharabaghi *et al*., (2006) reported more than 95% of the aggregates larger than 40 µm in diameter were trapped within the first five metres of a filter strip. However, Owens *et al*. (2007) found collected material was primarily sand sized, thus effective in reducing sediment loads, but of lower importance for reduction of associated phosphorus. Abuzreig (2001) found using computer simulations that trapping efficiency was 47% for clay compared with 92% for silt particles.

Buffer strip effectiveness is also controlled by flow conditions. Both Lui *et al*. (2008) and Yuan *et al*. (2009) in their detailed review of the evidence report that concentrated flow significantly compromises the effectiveness of riparian buffer strips because much of the sediment transported by these higher energy pathways is not effectively filtered. While the evidence suggests that vegetated riparian buffer strips are highly effective in reducing sediment eroded from field surface from reaching water courses, some care is required in interpreting the potential impacts of buffering all water courses at a catchment scale. A large number of studies assessing buffer strip effectiveness are performed at the plot scale, on uniform *planar* slopes. While slope uniformity is desirable in plot replication, it masks the reality that large proportions of runoff from fields and crossing buffer strips will be convergent, along plough furrows, vehicle wheelings and natural thalwegs. It means that the majority of field runoff is concentrated at critical spill points. The effectiveness of buffers in slowing this runoff and allowing sedimentation is likely to be much lower than in non-convergent flow.

Dilhalla *et al*. (1989) conducted a buffer strip plot study simulating the effects of cross slopes generating concentrated flow along plot edges, compared with planar surfaces. They found that plots were 61-70% less effective in sediment reduction where concentrated flow occurred, and that over 60% of all the flow was concentrated. Helmers *et al*. (2005) modelled (MIKE SHE) the effect of converging overland flow in two dimensions. The study found that convergence through the buffer strip reduced trap efficiency from 80% to 64%, and when convergence occurred in the field (before reaching the buffer) this was reduced to 57%. Verstraeten *et al*. (2006) used a RUSLE- based modeling approach to simulate the effectiveness of buffer strips at a catchment scale. They found that under conditions of an even slope (no flow convergence) buffer strip effectiveness was similar (62%) to that reported in plot tests. However at a catchment scale this fell to 21%. They point out that buffer strips along riparian margins where runoff to water courses does not occur have no effect, and therefore estimates of available sediment reductions based on catchment buffer strip length are misleading.

The evidence presented in Chapter 3 of this study highlights the fact that under saturated conditions buffers may function much less effectively than in circumstances where runoff is the result of infiltration excess; where soils are saturated the function of vegetated buffers in promoting runoff infiltration and sedimentation does not apply. In these circumstances the sedimentation potential of buffers trips is limited to the reduction in flow energy caused by physical resistance of vegetation. The evidence presented in this study has illustrated that most sediment loss occurs in a small number of high magnitude events in this landscape. The effect of grass strips in reducing energy of large volumes of runoff appears to be limited. The beneficial effect of buffer strips in these circumstances may be limited to the reduction of sediment loss from the buffer area, rather than reduction of sediment from the wider field area. Chapter 3 also illustrates that where artificial sub-surface drainage systems exist, these may be the primary pathway for suspended sediment, against which buffer strips are entirely ineffective.

While the evidence suggests that buffer strips are highly effective in mitigating sediment losses in some circumstances, and at least partially effective in all circumstances, some caution is required in transferring the figures from the literature to the catchment scale. The issue of flow concentration described above indicates that the effectiveness of buffers is negatively correlated with the degree of runoff occurring along concentrated pathways. As during large events these pathways may account for the majority of runoff, the effectiveness of the buffers may be regarded as much lower than plot derived figures would suggest. In fact as Qui (2009) argues, the channelling of flow at spill points means that the majority of a riparian buffer length has no sediment mitigation function and is therefore an inefficient land use allocation in this regard. Plot studies (and model mitigation algorithms based on these studies) ignore the fact that sediment delivery ratios at the base of a sloping plot are unlikely to be the same as those reaching a water course; even under the most intensive arable systems some degree of natural vegetation, or at the very least a diversionary field edge furrow is likely to occur between the cultivated field and a water course. In England for example, Cross Compliance regulations prohibit cultivation and other field activities with 1 m of a stream bank (and in many cases even this is not practically possible). Estimated sediment loss reductions available through promotion on vegetated buffer strips should therefore take into account the existing attenuation in target catchments.

7.4.2 Field edge dams, fences and hedges

The use of field edge structures such as small dams has been used specifically to address problems of ephemeral gully erosion (Boardman, 2003, National Research Council, 1986). However evidence is lacking as to the effectiveness of these structures. Fiener *et al*. (2005) studied the construction of small retention ponds at the lowest margins of fields as part of a catchment sediment mitigation trial in Germany. They estimated sediment retention at 50-80%. Initial high sedimentation and maintenance were reduced by dramatic fall in soil loss due to additional installation of soil conservation measures. They conclude that the combination of retention ponds and conservation measures is effective at reducing sediment losses and runoff. However, use of dams has qualitatively been judged unsuccessful in the South Downs (Boardman, 2003; Boardman and Evans 2003) on the grounds that storage capacity was insufficient for large events and that ultimately the dams failed, exacerbating the problem. Similarly, although the removal of hedgerows and field boundaries during agricultural intensification is an often cited cause of accelerated erosion in the UK in particular (Boardman 2002), evidence regarding the effectiveness of traditional or modified field boundary structures in mitigating soil losses is lacking.

The data collected as part of this study (see Chapter 5) indicates that large quantities of fine sediment (circa 1 t from a single large event) may be prevented from field export by field edge retention structures. The efficacy of this mitigation approach may be justified in circumstances where soil losses are not extreme; the redistribution of large quantities of fine material within a field may result in management issues due to loss of soil function in eroded areas, and through burial in depositional areas. In such circumstances *source* mitigation (conservation practices or land use change) would appear more appropriate. The use of field edge structures has the advantage that no major changes to land management are required. These interventions may be preferable to in-field options particularly in circumstances such as the Belford example where mitigation of runoff quantity as well as quality are targets; while options such as buffer strips and grassed waterways are likely to have limited impacts on runoff volumes in saturated soils, carefully designed retention structures have the potential to reduce peak flows. The fact illustrated in chapters 5 and 6, that run-on from adjoining land is often responsible for channel erosion on susceptible fields, means that structures constructed on *unsusceptible* fields may be effective in mitigating sediment losses (and runoff volume) despite these areas not being a significant sediment source. This may be important to the efficacy of this approach, as retention structures in permanent pastures or farm woodlands may present much more limited management problems than those on arable land, and may therefore be more acceptable to land managers.

7.5 In-channel options: sedimentation basins and traps

Where sediment loss problems are associated with losses from non-field areas (e.g. hard standings and tracks) or where tile drainage is a significant vector, mitigation options located within surface drainage networks may have greater efficacy than those discussed above. Engineered structures designed to intercept and store water are common in dryland farming systems. Evidence from the United States suggests that small farm ponds and reservoirs are significant sinks for sediment at a catchment and basin scale (Renwick *et al*., 2005, Chin *et al*., 2008). Verstraeten and Prosser (2008) used the WATEM/SEDEM model to simulate land use impacts of installation of dams on a Murrumbidgee catchment (New South Wales, Australia). They estimated that farm dam construction had reduced sediment delivery to rivers by 47%. The role of farm dams as a sediment mitigation structure may be less cost effective in circumstances where farm water storage is less crucial to farm operations such as the UK (although forecast increases in droughts may make this an increasingly viable option). In such circumstances due to the high costs of construction and maintenance, such structures are likely only to be justified where off-site effects of sediment loss are extreme, and particularly where this is combined with flood risk. Evrard *et al*. (2008) measured the effect of control dams in a 300 ha catchment in loess belt of Belgium. The study estimated a 93% reduction in suspended sediment concentrations at the outlet compared with upstream of the three dams. They claim their results demonstrate that this approach can militate against muddy floods even for large events in the absence of other conservation measures. They also estimated that the construction costs, plus maintenance payments to farmers of €21 per hectare were less than the costs of damage caused during a single large flood event.

Smaller engineered structures may be appropriate to intercept field-scale runoff. Baskerud (2001) found that a constructed wetland representing 0.06% of the catchment area and 0.5 m deep could remove 50-60% of suspended sediment, and had potential in combination with other conservation measures. Wang *et al*. (2009) report that the use of over 200 gulley plugs (check dams along eroding channels) contributed to a 52% reduction in runoff and 86% reduction in sediment loss in a 22.5 km^2 Texan catchment.

It has been shown in a number of studies that during large storm events surface runoff is inevitable and can overwhelm structures employed to limit surface flow and encourage infiltration. In these circumstances easily eroded material captured by mitigation features in smaller runoff events may serve as sediment sources. Short term sediment concentrations may thus be increased and net reduction is minimal (e.g. Owen *et al*., 2007). As storm events are predicted to increase in both frequency and magnitude in the UK as a result of climate change, the need to control emissions during high runoff periods will be increasingly important (Mainstone *et al*., 2008). Boardman *et al*. (2009) discuss the use of engineering options to control muddy flooding in the South Downs, but state that these on their own have not worked, and are unlikely to without other control measures because of their susceptibility to failure during flood events.

7.6 Catchment scale mitigation

Although in some circumstances catchment scale interventions aimed at agricultural sediment reduction have focussed on the use of a single mitigation technique, realistic intervention on a catchment scale involves the use of a suite of mitigation options. A number of studies have been performed (although no evidence from the UK is available). A study of the Upper Thames River catchment in Ontario, Canada (Yates *et al*., 2006) demonstrated that a combination of BMP interventions including non-cultivation of fragile land, vegetation of riparian areas, "clean water diversion" (steading water separation) and grassed waterways resulted in ecological water quality improvement. Only in-field erosion control measures appeared to have a significant impact on water quality when applied in isolation. In a 5 year study of the effects of BMP's on water quality of tributaries of Consesus Lake, New York State, Makarewicz *et al*. (2009) found that in 3 of 6 catchments suspended sediment reductions were observed. The greatest reductions (72%) were achieved through the conversion of 60% of cropland to alfalfa (72%), 71% reduction through a combination of sediment basins and a 28% reduction in cropland, and a 65% reduction using a whole farm approach (multiple BMP's including stream fencing, grass buffers, cover crops and contour strips) .

Zhou *et al*. (2009) used field validated WEPP simulations to test the cost effectiveness of different secondary sediment control measures for reducing sediment yields on chisel ploughed, disk ploughed and no till in a maize-soybean rotation in an Iowa watershed. Without supplemental conservation measures, predicted sediment yield was 22.5, 17.7, and 3.3 $t.ha^{-1}$ yr^{-1} respectively. When factoring in the value of soil, no-tillage was the most cost-efficient practice with the highest net benefit of \$94.5 $ha^{-1}.yr^{-1}$. Secondary treatments (terraces, filter strip, grassed waterways) were more cost effective with mouldboard and chisel plough systems. However, the very high erosion rates should be noted.

A recent review (Tomer and Locke, 2011) evaluated the effectiveness of USDA conservation strategies in improving water quality across 14 large agricultural catchments. They concluded that while conservation strategies did improve water quality, problems persisted in water quality at a large catchment scale, citing among the explanations that strategies did not take account of connectivity in allocating resources, and that in the catchments where problems persisted, bank erosion was more problematic than soil erosion (hence the strategies did not address the root of the problem). They also suggest that mitigation of sediment loads while allowing runoff to continue increases the potential of flow to erode channel banks.

Lemke *et al*. (2011) tested the effects of multiple BMPs on a large (4000 km^2) tile drained catchment in Illinois. Land use was >90% row crop agriculture. Outreach successfully increased BMP implementation rates for grassed waterways, stream buffers, and strip-tillage within the treatment watershed, designed to reduce surface runoff and soil erosion. However, no significant changes in total suspended sediment, or hydrology were observed after implementation of these BMPs over 7 years of monitoring. The explanation for this failure is that the tile drainage means that BMPs such as grassed waterways and buffer strips are bi-passed. Although the focus of their discussion is dissolved nutrients, the fact that suspended and total phosphorus are not mitigated suggests either that tile drains are the major pathway or that as they point out, the vast majority of export occurs in storm events for which BMPs may be much less effective.

7.6.1 Sediment mitigation in the UK: The South Downs case study

The South Downs is an area which over the late Twentieth Century experienced an increase in intensive cereal production, mainly on naturally thin loamy soils in a landscape of relatively high relief and increasingly large field sizes. As such, in a national inventory, this land could be expected to be at relatively high risk of productivity losses caused by soil erosion by water. However, during the 1980s, field monitoring of erosion events in an area of 36 km^2 showed that average rates of erosion were low (0.5-5.0 $m^3.ha.yr^{-1}$), although occasional storms were observed to result in losses of over 200 $m^3.ha.yr^{-1}$ on individual fields (Boardman 2003). Boardman noted that little runoff occurs from grassland, with most runoff from arable land occurring as infiltration excess, particularly due to crusting, and suggested that conversion to grass under set aside (new schemes) could be effective in reducing erosion. Boardman (2003) concluded farming in the South Downs to be unsustainable without changes in land use. However, the threat to the already thin soils had been previously reported to be a low priority for farmers (Robinson 1999).

The off-site threat of muddy flooding to housing estates through runoff from dry valley systems was well reported in the South Downs; 138 incidents of muddy floods were recorded between 1976 and 2000 (Boardman *et al*. 2003). Greatest occurrence of flooding occurred in autumn and early winter when higher rainfall coincides with bare soils with fine tilth. Figures are quoted for estimated flooding costs of £445 000 in South Downs in 1987, of which £112

000 was uninsured loss borne by householders. The main response by the council was construction of emergency earth dams. These proved ineffective due to breaches. A second attempt included construction of permanent features. However, research has illustrated that the storage capacity of these dams is still way short of required given the large contributing areas. (Boardman *et al*., 2003; Evans and Boardman 2003). The recommendations of the researchers were the use of grassed waterways and selected grass reversion to encourage infiltration and sedimentation and prevent gulley erosion.

The evidence from the South Downs illustrates that even in extreme cases, farmer change in practice due to on-site effects of soil erosion were limited. It was the off-site effects which eventually led to change in practice. While threats of legal action over negligence were raised in some cases, it was the acceptance of the runoff problem by farmers caused by winter cereals which led to voluntary changes through the introduction of grass reversion under set aside and grasses margins under an Environmentally Sensitive Areas scheme (Boardman *et al*., 2003, Evans and Boardman, 2003). The authors report no further muddy flood problems over the next seven years following the

The extent to which the findings of the South Downs case study are transferable to the UK as a whole is worthy of consideration. Certainly the issue of conversion of grassland to winter cereals has been a national phenomenon, as arguably is the removal of field boundaries, increasing the erosive potential of runoff waters. The cultivation of shallow soils on upland topography however, is a more specific issue, and it appears likely that part of the problem of high runoff coefficients can be attributed to these conditions. The occurrence of ephemeral streams during heavy rainfall is also likely to be exacerbated by free drainage as a result of the chalk geology of the area, which causes surface runoff over land surfaces along dry valley floors, meaning the risk of uncontrolled flooding is likely to be much higher. These streams occurring on cultivated surfaces are likely to result in rill and gulley erosion.

Importantly the current study has illustrated that a similar issue of uncontrolled surface runoff is also an issue in a tile-drained landscape; the replacement of surface watercourses with sub-surface pipes of limited capacity has resulted in artificial dry valleys similar to those which occur as a natural feature in a chalk landscape. Unlike the Downs example however, the occurrence of surface runoff appears largely the result of saturation rather than infiltration excess, meaning that surface sealing of exposed soils is less of a factor, and the mitigation impacts of preventing of this phenomena are much more limited.

The problems resulting from conventional arable cultivation in the South Downs were slowly remedied, despite early diagnosis of the problem. It is notable that positive changes when finally introduced were brought about voluntarily, through farmer compensation under agricultural subsidies. This would appear a relatively painless solution, particularly compared with the alternative option of engineering, which in any case was not effective without land use change. However, whether targeted land use changes such as these will be as effective in mitigation sediment pollution problems as the more extreme muddy flood problems in the South Downs is debatable.

7.6.2 Sediment mitigation case studies from Europe

While uptake of sediment and erosion control measures in many parts of Europe has been similarly slow, some countries have given the issue more attention than UK. In Flanders, the off-site effects of soil erosion, particularly siltation of flood control structures has led to national scale intervention, with local municipalities given a multi-million euro budget to create an erosion control scheme, and 75% funding for construction under the "subsidy of small-scale erosion control measures to be taken by local authorities' decree. The scheme focuses on engineering approaches such as flood retention ponds. Funding for in-field conservation measures (as in UK) is made through other schemes (Verstraeten *et al*., 2003). Verstraeten *et al*. (2002) investigated the effectiveness of various measures (using a modelling approach) to control sediment losses to water and emphasised the greater effectiveness of in-field measures compared to off-field.

Incentives (subsidies and direct payments) began in the late 80s and 90s in Norway to reduce soil and P losses, including reduced tillage, buffer zones, grasses waterways (Bechmann and Stalnacke, 2005). Because no tillage in autumn had been shown to reduce soil erosion in Norway, payments for no autumn tillage were introduced in 1991 at a rate of about 125 € ha^{-1} irrespective of erosion risk. Later subsidies were made available for under-sown catch crops. In some parts of Norway after 2001 erosion risk mapping facilitated a sliding scale of payments of 50 to 163 €.ha^{-1} for reversion from autumn ploughing depending on erosion risk. Subsidies are made for grassed waterways to protect water bodies. Two hundred and fifty sediment ponds were built by 2001 at a cost of 1.2 m Euros (Lundekvam et al, 2003). Oyegarden *et al*., 2004 report that the policy has already meant targets for phosphorus have been reached and sediment targets are close to being achieved. Lundekvam (1997) found that 40% of erosion was due to concentrated flow erosion in a 2.7 ha catchment, and was virtually eliminated by converting to zero tillage under autumn sowing.

7.7 Policy measures

A recent national scale modelling estimate (Collins *et al*., 2009) projected a 9% reduction in total sediment losses from the sector by 2015 as a result of farming change and predicted uptake of mitigation methods primarily through environmental stewardship schemes. However, a lot of conjecture is used in predicting the effectiveness of these measures. The Single Farm Payment Scheme now includes some requirements for land owners to recognise soil erosion risks, and to prohibit high risk of detrimental effects on and off site ecology, or off site infrastructure. There are also limited schemes (e.g. SOWAP) which provide advice to farmers on a voluntary basis on soil protection. However, the focus appears to be on identifying high risks that can be easily avoided (e.g. non-cultivation of steep land) and BMP's such as riparian buffer strips which can reduce sediment loss without affecting land management. As previously described in earlier chapters, reliance on soil conservation measures alone to mitigate sediment losses is a limited approach which is unlikely to be a cost effective means of achieving water quality objectives.

Kuhlman *et al*. (2010) estimated the cost of introducing appropriate soil conservation measures in Europe. They base these estimates on the cost of mitigating soil erosion risks of varying degrees of severity scenarios for different levels of soil erosion. Costs of serious (>10t $ha^{-1}yr^{-1}$), serious-moderate (2-10 $t.ha^{-1}.yr^{-1}$) moderate (0.5-2 $t.ha^{-1}.yr^{-}$) and low (<0.5 $t.ha^{-1}.yr^{-}$) were estimated at €296, €140, €120, and €116 per hectare respectively. Costs are assumed equal to the rate of single payment subsidy in this analysis (the subsidies being assumed to accurately determine the offset value of lost production). Khulman *et al*. calculated €1292 million per year was spent in EU on soil conservation through CAP to 2005. They conclude that the on-site benefits are exceeded by the costs, meaning these incentives are required to promote off-site benefits. The costs of siltation plus pollution are estimated as €1.8bn per year (equivalent to an average of €80.38 per hectare); illustrating that such an incentive may be economically justified. Evidence suggests that costs of soil conservation in England may be lower than for Europe as a whole; Evans (1990) estimated that only 6% of agricultural land in England and Wales was at high risk of soil erosion, with a further 18% at moderate risk. Using the mitigation cost figures quoted by Kulman *et al*. (2010) Costs of conservation measures for this land would total €197m for the highest risk and €209m for moderate risk. However, as argued previously, concentration of resources on land at high erosion risk does not appear an effective means of mitigating sediment losses at a catchment scale.

7.8 Conclusions

Conservation measures appear effective in reducing the mobilisation of sediment. Reduced tillage appears to be effective in high risk areas, particularly under winter sown crops - a land use which has been demonstrated to represent a prolonged period of susceptibility to sediment losses. This measure may also reduce losses to sub-surface drainage systems, but requires residue retention in addition to tillage reduction to protect the soil surface from raindrop impact. Arable reversion may be an option on very high risk land, where rates of soils loss threaten sustainability. However, the emphasis on conservation use of these areas (low or no stocking) in subsidy schemes means it may not be the most attractive option for farmers.

This review has indicated that of the available mitigation options, field edge and riparian buffer strips have received the greatest amount of recent research attention. Buffer strips appear effective, low cost and multi-benefit, which explains their attractiveness. However, they may be significantly less effective than results suggest in improving water quality, and are unlikely on their own to mitigate in high risk areas. They appear particularly ineffective where sub-surface field drains are present, and of much lower effectiveness against concentrated runoff. In these circumstances their function appears largely limited to reduction of sediment mobilisation from zone of greatest connectivity – the buffer strip itself.

The use of control structures such as field edge barriers or grassed waterways generally entails higher costs and greater management restrictions. In order for the costs to be justified judicious targeting of these features to areas where concentrated flow occurs appears necessary. The use of remote terrain analysis in conjunction with field survey may have potential in this regard. Sediment basins and traps are higher cost solutions, which may be justified particularly where runoff in extreme events threatens property and infrastructure. These may be combined with wetland creation schemes and buffer strips to achieve conservation benefits.

In livestock systems, prevention of access to stream banks where sediment pollution is a concern would appear a simple solution, combined with infrastructure improvement (water feeders, bridges, armoured crossings). This issue has not been the focus of the present research, but it is acknowledged that there is a current knowledge gap regarding the relative importance of stock bank damage and field losses from cultivated surfaces in mixed agricultural landscapes.

Chapter 8: Conclusions and recommendations

This study has identified the distinctiveness of agricultural sediment loss processes in a slowly permeable tile-drained landscape. These processes can be broadly divided into field and landscape processes. At the field scale, sub-surface (tile) drains appear to be the major sediment loss pathway, particularly due to their high permanent connectivity with surface waters. This connectivity means that sediment losses in response to events of low magnitude will be large relative to those from soils without artificial drainage. It also means that a greater proportion of a catchment under susceptible land uses becomes relevant for sediment loss mitigation, as the negative relationship between sediment load and flow distance to receptors (due to energy reduction and interception) does not apply as to surface runoff pathways.

Landscape processes are defined as those which are controlled by topography and man-made features. The importance of these landscape idiosyncrasies has been neglected in past attempts at sediment loss risk assessment, particularly due to the need for high resolution survey information which is rarely available. However, as this study has demonstrated, landscape features which result in concentrated runoff may be a critical sediment source-pathway (as well as representing a threat to sustainable land use in some circumstances), and 'generalisation' of catchment units may therefore result in failure to address the main sediment loss issues. This is particularly the case in the study landscape, where the replacement of a relatively high density surface drainage network with a sub-surface piped system of limited capacity significantly increases the probability of erosive high energy concentrated overland flow (while indirectly increasing risk by facilitating a change to higher susceptibility arable land uses). Man-made surface features such as farm tracks may result in similar recurrent overland pathways resulting in significant runoff and sediment generation during susceptible periods, particularly winter and spring cultivations. Landscape processes appear to be locally much more critical to sediment losses than tile drains, although applying to a smaller proportion of the land area.

Identifying risk

The existence of tile drainage has previously been identified as the key factor in catchment scale studies of sediment loss risk in England and Wales. This is largely attributable to the increased connectivity promoted relative to the average connectivity of the surface pathway. This study supports this conclusion, demonstrating that the export through this pathway under susceptible land use (arable cultivation) may be significant and may represent the largest source proportionally from catchments with a high proportion of tile drained land. Therefore the inference made in the studies above, that tile drainage can be assumed to be a major sediment loss risk factor in catchments with soil types requiring drainage appears valid. However the efficiency of the tile network, and in particular its liability to capacity exceedance in major runoff events have also shown to be important considerations which affect the potential to quantifiably estimate tile drainage losses.

The study has illustrated the importance of concentrated runoff pathways in mobilising and transporting sediment to water bodies. Although this risk appears to apply to only limited parts of the study landscape, the potential of these pathways to mobilise relatively large quantities of sediment, coupled with their high connectivity with water bodies during runoff events mean that the associated risk is significant. The fact that the number of these areas was found to be relatively small in the study landscape, and these areas may be relatively easily identified using simple GIS techniques means that targeted mitigation is feasible. The use of topographic remote sensing as demonstrated in this study is a useful identification tool, but should be used in combination with local knowledge to identify issues.

Mitigation strategies for the study landscape

The fact that runoff hydrology is dominated by saturation rather than infiltration excess in this landscape has significant implications for agricultural sediment loss mitigation. The mitigating effect of a reduction in runoff energy observed in permeable geology in other studies does not apply in slowly permeable soils of this type under UK conditions. Therefore mitigation options designed to encourage runoff infiltration are likely to be much less effective than reported elsewhere.

The study has provided evidence that for large parts of the landscape (those not subject to concentrated runoff and sediment mobilisation), the artificial tile drainage network represents the major pathway for mobilised sediment from agricultural fields to aquatic receptors. Field edge mitigation treatments are wholly ineffective against this pathway, meaning that reliance on buffer strips to mitigate field sediment losses is likely to be unsuccessful. As a reduction in drain system efficiency is unlikely to be desirable, mitigation strategies should either target in-field susceptibility and/or seek to trap sediment within receiving ditches. In-field options to reduce drain sediment losses include conservation measures such as residue retention, cover crops and reduced tillage. This study has demonstrated that a reduction in tillage disturbance may be partially effective. It has also demonstrated that ditch sediment traps of moderate size may be an effective treatment at the farm scale. The best suited of these two options will depend upon the dis-benefits to the farmer of a change in cultivation system relative to the costs of construction and land-take associated with sediment trap installation. It is important to note that while less visible the sub-surface pathway should not be neglected in mitigation strategies: The evidence presented illustrates that losses can be substantial.

As areas at high concentrated runoff risk can be mapped using relatively low resolution GIS topographic data analysis, and apply only to a relatively limited proportion of a catchment, identification and appropriate mitigation of these areas is practical and likely to have a significant positive impact on losses of sediment and associated pollutants to water. The fact that these critical source pathways appear limited in number and land area means appropriate mitigation may be more efficiently achieved than tile drain losses, which may require more widespread adoption of in-channel sediment trapping or in-field conservation practices. Suitable mitigation options include the use of in-field grassed waterways, or construction of interception dams on field margins, the latter being particularly desirable where runoff control is also a target, either to mitigate flooding or to reduce the damaging effects of run-on to adjoining downstream fields. A more radical solution may involve reinstallation of surface drainage in some circumstances in order to prevent the very large volumes of sediment which may be mobilised by concentrated overland flow. However the impact of such a change on land management needs to be fully explored.

Recommendations for further study

The potential of conservation agriculture techniques to mitigate tile drainage losses has been partially investigated during this study. However, the effect of residue management as well as tillage reductions, and of zero tillage techniques appear worthy of further investigation. The large areas subject to tile drainage in predominantly arable catchments is likely to present a significant impediment to attempts to reduce agricultural fine sediment inputs unless a suitable option can be identified.

This study has not addressed the important issue of accelerated bank erosion, particularly that caused by livestock. Bank sediment inputs are difficult to measure, although sediment source fingerprinting techniques offer potential to provide improved understanding. Some more recent evidence has suggested that bank inputs may have been previously underestimated in national assessments. While the solution to bank damage (stock fencing) is relatively simple a better understanding of its importance relative to field losses is required if issues of excessive sediment loss from mixed land use agricultural catchments are to be successfully addressed.

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