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# Towards Guidance for the Design and Placement of Vegetated Filter Strips

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# Abstract

A combined field, laboratory and modelling approach to the study of vegetated filter strips (VFSs) was carried out in order to provide guidance on optimum design and placement for trapping sediment from overland flow. Monitoring of fifteen established filter strips in the Parrett Catchment, England, informed on the complexity of intercepting flow pathways to optimise filter strip performance. Results suggest that a 6 m VFS will trap an average of 1.74 t $year^{-1}$ of material from a field of 1 ha, but this is highly variable depending on design, placement and management factors. In most cases the majority of coarse sediment is trapped at the upslope edge of the VFS and is typically >85% sand. A revised Morgan-Morgan-Finney model was tested against a range of field and laboratory datasets and an efficiency coefficient of 0.7 was achieved. When testing the model against the field results from the Parrett Catchment, an active filter strip area was used. This took into account only the area of the filter strip effective in trapping sediment due to the convergence and bypassing of flow pathways. In the field, filter strip performance will be improved by reducing concentrated flow reaching the strip and ensuring that flow does not bypass the strip through burrows and gateways, using in field erosion control, maintaining level ground between the field and filter strip edge and managing the strip to maximise the density of vegetative material, particularly the number of vegetative stems. Potential applications for the research include a field based Decision Support System, design of filter strip biophysical architecture and catchment planning.

# Acknowledgments

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# Chapter 1 Introduction

## *1.1 Overview of problem*

### 1.1.1 Diffuse pollution

Diffuse pollution enters the environment via a multitude of pathways, as opposed to point source pollution which refers to a discrete point of discharge (Defra, 2002). It is defined by D'Arcy et al. (2000) as:

> *"Pollution arising from land-use activities (both urban and rural) that are dispersed across a catchment, or subcatchment, and do not arise as a process effluent, municipal sewage effluent, or an effluent discharge from farm buildings".*

Whilst agriculture is not the only source of diffuse pollution, it is a key generator (Defra, 2002), constituting 76% of the land area of England and Wales (Defra, 2001). Impacts on surface water quality result from activities such as the tillage and ploughing of land, the spreading of slurries and farm yard manures, use of pesticides, veterinary medicines and fertilisers (Defra, 2002). Table 1.1 summarises the main agricultural pollutants, example sources and impacts. This includes pollutants such as farm chemicals, silt, organic wastes, faecal pathogens, nitrogen and phosphorus which can present adverse impacts to aquatic life, health risks and an economic cost borne by the loss of water resource value. Estimates of pollutant loads in surface waters (e.g. Johnes et al., 1994) suggest that diffuse pollution is increasing throughout the UK.

| *Pollutant* | *Example Sources* | *Impacts* |
|---|---|---|
| Farm Chemicals Including<br>• Oils & hydrocarbons<br>• Pesticides<br>• Veterinary Medicines | Agricultural machinery maintenance, accidents, spill storage disposal.<br>Yard runoff – beef and dairy systems<br>Use of pesticides of arable and fodder crops<br>Use of pesticides of horticultural crops<br>Use of sheep dips and other mecidines on livestock to stop spread of disease and maintain high levels of animal husbandry | Toxicity to aquatic species.<br>Contamination of stream sediments.<br>Groundwater contamination. Nuisance (surface waters). Taste (potable supplies) Toxicity. Contamination of potable supplies |
| Silt | Runoff from arable land. Upland erosion by sheep farming<br>Lowland erosion by outdoor pig systems, dairy and beef herds | Destruction of gravel riffles. Loss of breeding and rearing sites.<br>Sedimentation of natural pools and ponds. Costs to abstractors (e.g. fish farms, potable water supplies) |
| Organic Wastes | Agricultural wastes – slurry, silage liquor, and surplus crops – dairy and beef herds<br><br>High production of waste products from intensive livestock units<br>Waste for land application | Lowering of oxygen levels.<br>Toxicity to aquatic life<br>Loss of habitats<br>Nutrient enrichment. |
| Faecal Pathogens | Application of livestock organic wastes to farmland<br>Use of dirty water for irrigation<br>Non separation of dirty and clean water | Health risk to humans. Non compliance with recreational bathing water and shellfish standards |
| Nitrogen | Agricultural fertilisers – use for horticulture, improved grassland, fodder and arable crops, Atmospheric deposition – emission of ammonia from livestock | Eutrophication. Contamination of potable supplies (rivers and groundwater). Accelerated plant growth. Changes in community structure. Acidification. |
| Phosphorus | Agricultural fertilisers – addition to fodder, improved, horticulture and arable crops<br>Soil Erosion – loss of phosphorus combined with soil particles from livestock and arable farming activities | Eutrophication<br>• Changes in community structure<br>• Accelerated plant growth<br>• Increased filtration of potable supplies |

**Table 1.1 Diffuse pollution sources and impacts from UK agriculture (Source: Defra, 2002.)**

There are several drivers in the UK for improving water quality. The Government has commitments to legislation at national, European and international levels as well as their own targets for England and Wales (Table 1.2). As well as meeting these commitments an improved understanding of water quality is required in order to inform policy development and to maintain best practice for land owners.

| Scope | Legislative driver or target | Government requirement |
|---|---|---|
| National/EU | EU Water Framework Directive | Meeting of ecological and chemical water quality targets by 2015. |
| | Nitrate Vulnerable Zone Action Programme | Protection of surface and groundwaters from diffuse pollution by nitrogen compounds of agricultural origin. |
| | Bathing Waters Directive | Compliance with microbiological standards for identified bathing waters by 2015. |
| | Habitats Directive | Maintenance or restoration of priority habitats and species. |
| | Drinking Water Directive | Meeting of minimum drinking water quality standards |
| International | UK Biodiversity Action Plan | Restoring status of priority habitats and species |
| | Agenda 21 | Sustainable development |
| | OSPAR Convention | Reducing anthropogenic inputs from land-based sources to the North Sea |
| England and Wales | River Quality Objectives | Compliance with objectives |
| | Sites of Special Scientific Interest | 95% in favourable conditions by 2010 |
| | English bathing waters | Compliance with mandatory coliform bacteria standards |

**Table 1.2 Legislative drivers for the control of diffuse pollution.**

### 1.1.2 Sediment as a diffuse pollutant

"Soil erosion is the major cause of diffuse pollution and sediment is the most visible pollutant" (Campbell et al., 2004). Gilliam (1994) also identified sediments as the diffuse pollutants that cause the greatest environmental damage. "Currently there is no national silt budget, but by far the highest proportion of suspended solid loading in rivers is from diffuse sources, rather than sewage or industrial effluents (Ferrier and Ellis, 2001). The relative contributions from rural or urban diffuse sources vary across the country, depending upon the characteristics of the catchment. In rural catchments the load is heavily dominated by soils lost by erosion from agricultural land." (Defra, 2002).

The main catchment sources of soil particles to surface waters are a) erosion of banks and stream beds and b) erosion of soils in the agricultural landscape and their subsequent transport to water via surface and subsurface pathways (Defra, 2002). Impacts include the siltation of riverbeds and channels, sedimentation of reservoirs, modification of the channel substrate and an increase in the concentration of suspended solids. The effects of this, as illustrated in Table 1.3, represent severe degradation of aquatic environments and a reduction in the economic value of water resources.

<table>
<tr><th>Impact</th><th>Effect</th><th>Consequence</th></tr>
<tr><td rowspan="4">Siltation of riverbed</td><td>Fine sediment deposited on spawning gravels</td><td>Decline in success of salmonid fisheries in England and Wales.</td></tr>
<tr><td>Reduced transfer of oxygen from water body to fish eggs and fry</td><td>Intra-gravel life stages of fish species are impacted</td></tr>
<tr><td>Blockage of voids</td><td>Difficult for alevins to emerge</td></tr>
<tr><td>Mortality in invertebrate communities</td><td>Fish food abundance reduced</td></tr>
<tr><td>Siltation of channels</td><td>Decreased root penetration of macrophytes</td><td>Increased vulnerability to displacement during high flow</td></tr>
<tr><td>Sedimentation in reservoirs</td><td>Reduced storage capacity</td><td>Reduced economic usability</td></tr>
<tr><td>Modification of substrate</td><td>Unsuitability for certain species</td><td>Decreased species diversity</td></tr>
<tr><td rowspan="10">Suspended solids</td><td rowspan="2">Altered water turbidity</td><td>Decrease in efficiency of hunting</td></tr>
<tr><td>Hindered fish migration</td></tr>
<tr><td>Reduced sunlight penetration</td><td>Decreased photosynthesis and hence oxygen availability for fish and macroinvertebrate communities</td></tr>
<tr><td>Clogged fish gills and shellfish filter feeding mechanisms</td><td rowspan="2">Increased mortality</td></tr>
<tr><td>Toxicity to aquatic life</td></tr>
<tr><td>Transport of adsorbed pollutants</td><td>Pollution from pesticides, faecal pathogens, toxic metals and nutrients such as nitrogen and phosphorus</td></tr>
<tr><td>Reduced survival rate of egg to fry</td><td>Degraded benthic habitats and communities</td></tr>
<tr><td>Blanketed benthic habitats</td><td>Changed ecological dynamics of the water course</td></tr>
<tr><td>Breakdown of organic material by micro-organisms resulting in rapid de-oxygenation of waters</td><td>Fish kills</td></tr>
</table>

**Table 1.3 Impacts, effects and consequences of increased sediment transport to water courses.**

### 1.1.3 Buffer features for controlling diffuse pollution

When soil erosion is not controlled at source, soil loss and delivery may be reduced by intercepting the pathways of sediment transport. Strategically placed buffer features provide the potential to reduce the movement of sediment and associated pollutants. When combined with appropriate land management practices such as crop residue management, nutrient management and winter cover crops, buffers should allow farmers and land owners to achieve a measure of economic and environmental sustainability in their operations. Buffer features can also enhance wildlife habitat and protect biodiversity, thus offering a potentially effective tool for achieving Water Framework Directive targets (Defra 2002) as well as a wider range of landscape quality objectives. Soil conservation and sediment control techniques have been empirically tested and modelled, and are well documented in available literature. However, given the lack of available tools for buffer feature implementation, it is likely that the literature does not provide sufficient basis for the design and placement of effective buffer features. The following study starts with a thorough examination of the level of current understanding of buffer performance and analyses the extent to which this may be transferred into design and placement guidance. From this, five research hypotheses were formed which provide the framework for the rest of the study.

## *1.2 Rationale*

The key driver for investigating the performance of buffer features for mitigating diffuse pollution was provided by a UK Department for the Environment, Food and Rural Affairs (Defra) funded project "PE0205 - The strategic placement and design of buffers for trapping sediment and phosphorus". PE0205 was commissioned by Defra who have a range of environmental commitments, such as those outlined in Table 1.2, as well as a requirement to inform policy development. The target audience for the work therefore included policy makers and those advising farmers and land managers on best practice for mitigating diffuse pollution.

PE0205 aimed to determine the potential effectiveness of features within current or future agri-environment schemes for the trapping of soil and phosphorus (P) and to identify where in the landscape buffer features could be most effectively placed, with consideration for the cost effectiveness of proposed options, as well as the maintenance of landscape quality and biodiversity benefits. The key output from the project was a mechanism by which strategic placement could be determined by local Defra staff, advisors/consultants and farmers;

A subset of PE0205 focused on sediment and on a set of buffer features termed vegetative filter strips. That work is reported in this thesis. Following a comprehensive literature review, the following research aims and hypotheses were formulated for the thesis. These are further discussed in Section 2.18.

### 1.2.1 Thesis research aims

The work described in this thesis targets some of the gaps in field data on established vegetative filter strips (VFSs), and in guidance on VFS design and placement. The overall approach is to test whether a process-based soil loss model can be applied to the design and placement of vegetated filter strips in a UK agricultural landscape. The following five hypotheses are tested in order to achieve this aim:

1. Stem diameter has a significant influence on the trapping efficiency of VFS.

2. A simple soil erosion prediction model can be used to predict the sediment trapping efficiency of laboratory simulated VFSs.

3. The model can be used to simulate the influence of different plant stem diameters on the sediment trapping efficiency of VFSs.

4. The model can be used to predict the sediment trapping efficiency of established VFSs in the field.

5. The model can be used to derive the optimum design and location for VFSs in the field.

# Chapter 2 Literature review

## *2.1 Introduction*

The purpose of the following review is to set out criteria, by which it is possible to gauge the scope of the information base, for advocating and guiding best practice for the use of buffer features to trap and retain sediment. From this base, limitations in understanding and in design guidance are identified and the research objectives for the thesis are defined. The criteria, and hence the questions asked of the literature, are:

1. What is the optimum buffer design?

2. What is the optimum buffer placement?

3. How should a buffer be managed? How long will a buffer perform its role and what magnitude of storm will it withstand?

4. What level of water quality improvements can be expected?

5. In what form does practical guidance exist for establishing buffer features?

The review initially defines buffer features and provides information on UK policy for implementing buffers in agricultural environments from 1986 to 2007. The remainder of the chapter concerned with establishing buffer features in UK agri-environments and focuses on vegetated filter strips (VFSs). The literature in this section has been evaluated in order to determine how much guidance it can provide on buffer design, location and maintenance. Information in this half of the review is discussed in relation to the questions posed above.

## *2.2 Buffer features for mitigating surface water pollution*

### 2.2.1 Defining buffer features

"A buffer acts to protect from impact or to 'cushion the blow'" (Dabney et al. 2006). The impact or "blow" that this study is concerned with is the diffuse pollution of surface waters. In this context, the purpose of a buffer feature is to intercept field runoff and to trap sediment and sediment-associated pollutants before they can enter a watercourse (Dosskey M.G. et al., 2002). Figure 2.1 illustrates features in the landscape which might offer a buffering role by intercepting flow to watercourses.

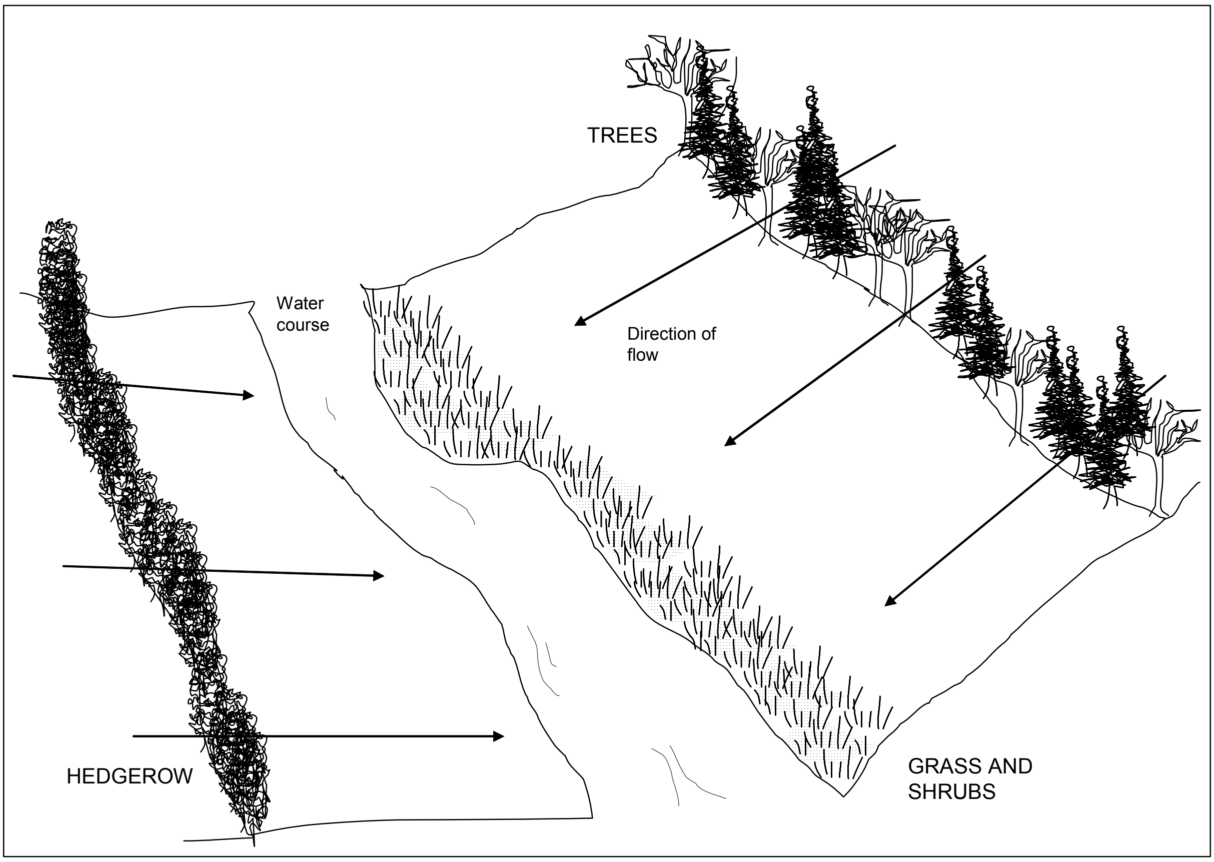



**Figure 2.1 Some landscape features with the potential to buffer sediment by intercepting field runoff.**

### 2.2.2 Buffer types and their role in trapping sediment and associated pollutant

Water quality may be managed at source (i.e. the field or hill slope) by reducing nutrient inputs or preventing the dislodgement of soil particles and at transition by intercepting the transport of soil particles over the landscape or, once the pollutant has reached the water, through in-stream amelioration techniques. Buffering can be used at the sediment transport stage to perform one or more of the following functions, which in turn impact on the delivery of sediment and associated pollutants to watercourses:

- Reduce the velocity of runoff by decreasing surface runoff (this may also involve reducing the slope length, angle or both).
- Delay the flow time and reduce the peak flow of runoff through the system.
- Trap the pollutants (including sediments) carried in runoff before they reach the river.

In the context of surface water quality protection, the term 'buffer feature' usually describes a vegetated strip between a river or stream and an adjacent upslope land-use activity (Hickey and Doran, 2004). However, the term may include infrastructure features, water retention features and management practices that have the potential to perform one of the functions listed above. It includes features located adjacent to a water course as well as further up in the catchment; within-field and field edge positions; features intended for water pollution mitigation as well as those which are incidental, having been established for an alternative purpose such as creating habitats or maintaining biodiversity. Further information on each of the features below is included in Appendix 1. Images are presented in Figure 2.2.

1. Grass features - riparian buffer zones, grass strips (field margins, beetle banks, conservation headlands, set-aside) grass hedges or vegetative barriers.
2. Hedgerows and related features – hedgerows, ditches, banks.
3. Trees and woodland – riparian multi-species buffer zones, in-field trees, woodland barriers.
4. Water retention features - wetlands, ponds, floodplains, grass waterways.
5. Infrastructure - fences, stone walls.
6. Management practice - strip cropping, contour cropping, terracing (or contour bunds), contour cultivations, soil (or vegetated) berms or a combination of practices.

Grass filter strip

(http://www.howardswcd.org/index.html)

Wetland

(http://www.howardswcd.org/index.html)

Grass waterway

(http://www.howardswcd.org/index.html)

Strip cropping

(http://www.omafra.gov.on.ca/english/environment/soil/bmp.htm)

Hedgerow

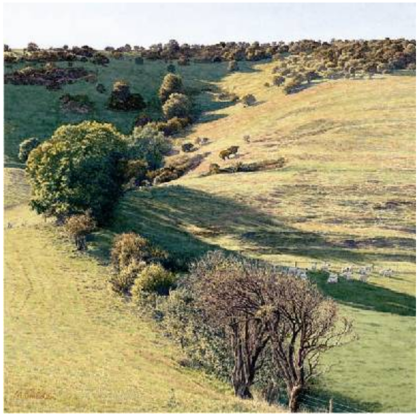
(http://www.biodiversitysussex.org/hedgerows.htm)

Contour cropping

(http://sitemaker.umich.edu/section8group1/possible_solutions)

Riparian multi-species buffer zone

(http://www.howardswcd.org/index.html)



Woodland barriers

(www.ecologyandsociety.org)

**Figure 2.2 Images of VFS types**

## *2.3 UK levers for implementing buffer features*

### 2.3.1 Agri-environment Schemes

Agri-environment schemes were established in the UK in 1986 to provide the government with a mechanism for funding farmers and landowners for establishing or improving environmentally beneficial practices on farmland. Table 2.1 provides a summary of Agri-environment schemes and their aims. The current protocol for the implementation of buffer features is described by Defra's Environmental Stewardship scheme, launched in 2005. (Defra operates in England and Wales. Similar schemes exist in Scotland and Northern Ireland and are implemented by the Scottish Executive and the Northern Ireland Department of Agriculture and Rural Development respectively.)

| Scheme | Implementation | Aims |
|---|---|---|
| Environmentally Sensitive Areas (ESA) | 1986 | **To protect threatened landscape and ecology and create improvements in public areas** |
| Nitrate Sensitive Areas (NSA) | 1989 | **To improve water quality on waters under threat of nitrate pollution by reducing fertilizer applications** |
| Countryside Stewardship | 1991 | **To include conservation as part of farming practice and land management for the protection of targeted landscapes, wildlife habitats and historic features. To improve opportunities for public access.** |
| Habitat | 1994 | **To create, protect and enhance wildlife habitats through environmentally beneficial land management.** |
| Countryside Access | 1994 | **To increase the benefits from land entered in the Arable Areas Payment Scheme by providing incentives for farmers to improve public access.** |
| Organic Aid | 1994 | **To assist farmers who wish to convert to organic production** |
| Moorland | 1995 | **To protect and improve moorland by encouraging extensive grazing practices** |
| **Environmental Stewardship**<br>• **Entry Level Stewardship**<br>• **Organic Entry Level Stewardship**<br>• **Higher Level Stewardship** | **2005** | **To continue or introduce beneficial environmental land management** |

**Table 2.1 Summary of Agri-environment schemes in England and Wales and their aims.**

The primary objectives of the Environmental Stewardship scheme are to

- Conserve wildlife (biodiversity)
- Maintain and enhance landscape quality and character
- Protect the historic environment
- Promote public access and understanding of the countryside
- Protect natural resources

Within these objectives it has the secondary objectives of

- Genetic conservation
- Flood management

Entry Level Stewardship (ELS) is open to all farmers and landowners and promotes simple and effective land management. Higher Level Stewardship (HLS) is another element of the Environmental Stewardship scheme which encourages targeted environmental management. A complete list of the options available can be accessed in the scheme handbooks (Defra, 2005a; Defra 2005b). Based on the preceding discussion those options likely to perform a buffering function include 2, 4 and 6 m buffer strips, hedgerow management, ditch management, stone wall protection and maintenance, protection of in-field trees and maintenance of woodland fences and edges.

### 2.3.2 Catchment Sensitive Farming

The Catchment Sensitive Farming (CSF) programme aims to develop measures to tackle diffuse water pollution from agriculture to meet Water Framework Directive requirements. The England Catchment Sensitive Farming Delivery Initiative, which commenced in 2006, aims to promote voluntary action by farmers in 40 priority catchments to tackle the problem of diffuse water pollution by agriculture. CSF includes encouraging best practice in the use of fertilisers, manures and pesticides; promoting good soil structure to maximise infiltration of rainfall (including in-field grass strips) and to minimise runoff and erosion; protecting watercourses from faecal contamination (e.g. with fencing and livestock crossings) and from sedimentation and pesticides (e.g. with buffer strips); reducing stocking density or grazing intensity and reverting arable land to grass. (More information on CSF can be found at http://www.defra.gov.uk/farm/environment/water/csf/delivery-initiative.htm.) It should be noted that no guidance on the size or other properties of in-field grass strips and riparian buffer zones is available, nor suggestions on best placement.

## *2.4 Non-UK levers for implementing buffer features*

In the USA, buffers are encouraged through financial incentives available through the United States Department of Agriculture conservation programs (e.g. the Conservation Reserve Program). In addition to state and local government some private organisations offer further financial incentives for installing conservation buffers. In certain cases blanket installation along all perennial streams has been undertaken (Dillaha et al., 1989). In New Zealand a policy of ‘retirement’ of riparian zones (i.e. removal from active use such as ploughing or trafficking) has been implemented along all perennial streams to protect riparian and aquatic habitats. The Landcare Program promotes training and extension through local committees in Australia. Whilst the nature of the schemes varies they all comprise financial compensations to farmers voluntarily applying approved land management practices.

## *2.5 Summary*

Table 2.2 summarises the buffering functions of the features discussed. Figure 2.3 illustrates the most likely location of each buffer in a catchment, whilst Figure 2.4 provides a general summary of the hydrological and sediment control processes associated with buffers. Based on the literature reviewed the buffer type which appears to offer the greatest number of benefits consists of, either solely or in combination with other types, a linear vegetated strip. This may be a beetle bank, field margin or grass barrier, part of a large, mixed species riparian area, floodplain or wetland or the ground cover to a forested area. The vegetation, which is usually grass, is widely reported to provide the potential to slow and disperse runoff, to increase infiltration and to encourage filtration, deposition and retention of sediment and associated nutrients. Vegetated strips are the most common buffering features in UK agriculture and are strongly promoted within Defra’s Environmental

Stewardship Scheme. The remainder of the review will therefore focus on vegetated strips and will use the term vegetated filter strips (VFS) to include linear vegetated features which intercept surface runoff and entrained material. The following sections examine the design and placement of VFSs.

| **Class** | Grass features | | | Hedgerows | | Trees | | Water retention features | | | | | Infrastructure | | Management practice | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Buffer type** / **Buffer function** | Riparian buffer zones | Vegetated filter strips | Grass hedges | Hedgerows | Ditches | Woodland barriers | In-field trees | Wetlands | Retention ponds | Detention basins | Floodplains | Grassed waterways | Fences | Stone walls | Strip cropping | Contour cropping | Terracing | Contour cultivation | Soil berms/contour bunds |
| Spread out and reduce velocity and turbulence of runoff | * | ** | ** | * | * | * | | ** | ** | ** | ** | * | * | * | * | * | * | * | * |
| Encourage filtration, deposition and retention of sediment and nutrients | ** | ** | ** | * | * | * | | ** | ** | ** | ** | * | * | * | * | * | * | * | * |
| Increase infiltration capacity | ** | * | ** | * | | * | * | | | | | | | | * | * | * | * | * |
| Promote adsorption and absorption of nutrients | ** | * | * | | | | | * | | | * | * | | | | | | | |

**Table 2.2 Summary of functions for different buffer types. ★ means the feature is likely to perform the function moderately well and ★★ means it is likely to perform the function very well.**

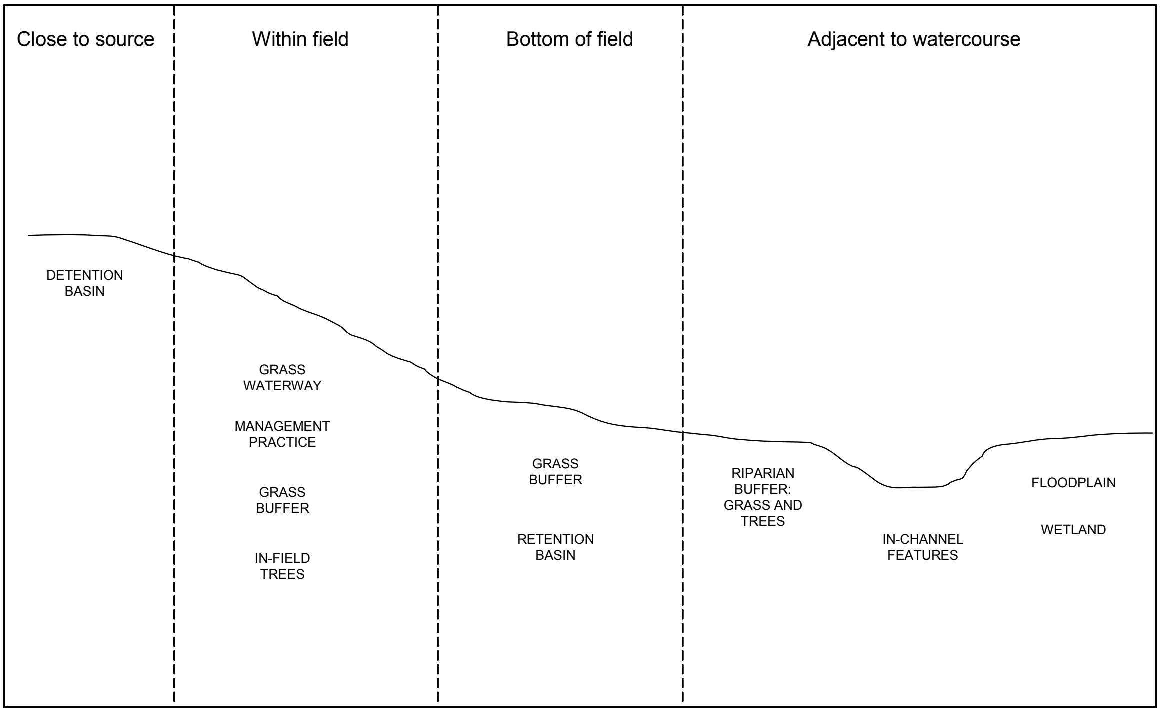



**Figure 2.3 Typical catchment position for different buffer types.**

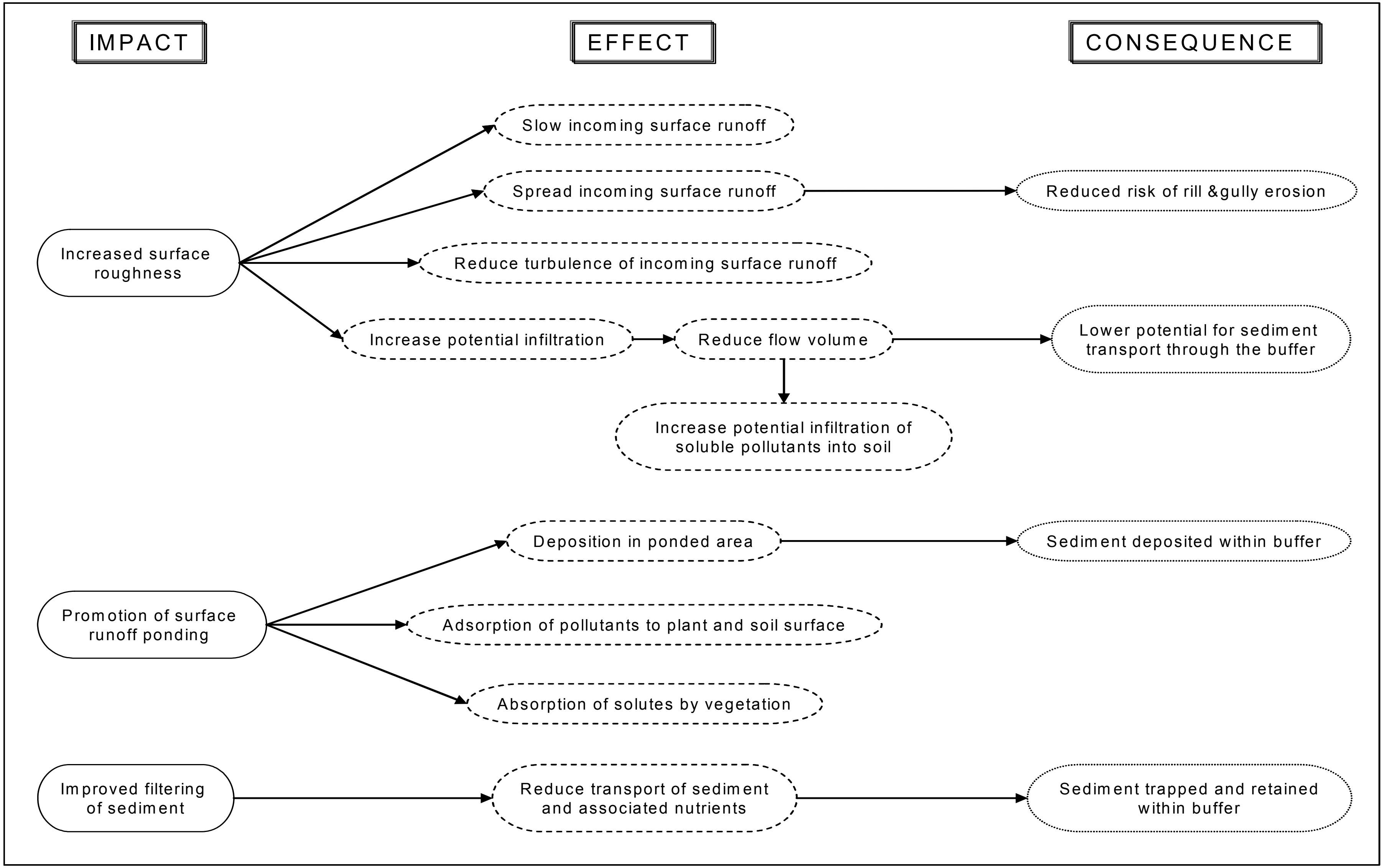



**Figure 2.4 Buffer processes presented as causes, effects and consequences.**

## 2.6 Implementing Vegetative Filter Strips: Design

This section of the review aims to answer, from the literature, the questions posed at the beginning of the chapter concerning VFS design, placement, management and performance. Factors deemed important have been selected from the literature and are discussed in relation to their importance and whether guidelines for optimum values have been established.

### 2.6.1 Height

Prosser and Karssies (2001) proposed that the height of the vegetation is an important factor in determining VFS performance. Whilst heights of 100 to 150 mm have been recommended (Dillaha T.A. et al., 1986) longer grass should not pose a problem as long as the stem density near to ground level is still high and does not suffer from lack of light (Prosser and Karssies, 2001). However, experiments by Hook (2003) do not support the use of height as a predictor of VFS capacity and Pearce et al. (1997) reported that, as long as submergence does not occur, vegetation height is not a significant variable in VFS performance.

Pearce et al. (1998a, b) applied simulated rainfall, run-on and sediment to 10 and 2 m long plots that were clipped to the ground surface, clipped to 100 mm or left unclipped. Stubble height generally did not affect runoff or sediment retention except over distances less than 2 m (Pearce et al., 1998a, b). In deeper flows, representative of concentrated runoff or overbank flooding, short stubble enhanced sediment deposition the most but taller vegetation was more effective in retaining the deposited sediment (Abt et al., 1994; Clary et al., 1996).

The limited research available on sediment retention in rangeland riparian areas shows that where herbaceous vegetation is dense, commonly recommended stubble height criteria in the 50 to 150 mm range (Clary, 1995; Clary and Leininger, 2000) will probably maintain vegetation capable of trapping sediment in either shallow overland runoff (Pearce et al., 1998a, b; Hook, 2003) or overbank flow (Abt et al., 1994; Clary et al., 1996).

*Summary: vegetation height may be a factor, 100 to 150 mm being most commonly recommended, but it may be outweighed by density of ground cover.*

### 2.6.2 Density

A number of studies have found an increase in VFS effectiveness with an increase in vegetation density. Results from Hook (2003) indicated that the dense vegetation of moist and wet riparian sites generally retained sediment effectively, whereas lower sediment retention was associated with sparse vegetation. The difference in sediment retention was attributed to gross differences between flow patterns.

Pearce et al., (1998a) found that plant density, cover, growth form and bare soil area influenced sediment retention. In both simulated and real filter strip vegetation, sediment retention has been found to increase with stem density (Tollner et al., 1976; Munoz-Carpena et al., 1999; Ghadiri et al., 2001). Abu-Zreig et al., (2004) observed a steady increase in sediment trapping efficiency with an increase in percentage cover ($R^2$ = 0.96).

Upland range studies show that erosion and sediment transport are sensitive to surface cover, especially when very sparse (Hook P.B., 2003). In a study of grassed filter strip performance in south-central Montana cropland, presence of vegetation was more important than the species planted (Fasching R.A. & Bauder J.W., 2001).

*Summary: there is fairly strong evidence that buffer efficiency will increase with vegetation density. What is not clear is whether the relationship between sediment retention and cover is linear or a negative exponential, or whether there is a threshold cover beyond which further increases in density have little effect. Species may only be important if certain species can provide enough cover and others can not. Hook (2003) suggests that practical field indicators of a site's potential for sediment retention should emphasize major differences in vegetation that can be evaluated visually rather than accurate, quantitative measurements. However, getting to this point is likely to require detailed studies with appropriate measurements.*

### 2.6.3 Vegetation properties and species type

The majority of VFS literature focuses on grasses, very few on tree species and no studies were found on the filtering effect of other vegetation types. Deletic (2000) attributes the importance of grass in conservation techniques to it being easy to establish, hardy, quick to grow, very well spread and able to grow in most climates. According to Wilson (1967) a suitable filter grass species should comprise a deep root system, a high stalk density, insensitivity to submergence and droughts and an ability to grow through sediment coverage. For maximum effectiveness, a filter strip should have the best grass stand possible and be maintained in a good condition (William and Nicks, 1988).

Taddesse and Morgan (1996) provide one of the few comparisons of the buffering effect of two grass species. They found that a dense, uniform grass strip can effectively reduce erosion, on an erodible sandy loam soil, on slopes up to 13°, whereas a VFS formed from a more open grass with a less dense rooting system is effective only up to 9°. They recommend a perennial species with an erect, dense and uniform rather than tussocky structure; a wide range of stem and leaf angles with respect to ground surface, providing a barrier to flow; and a system of roots to bind the soil and reinforce its strength. The species of grass favoured was *Festuca ovina* which Hayes et al. (1978) also recommends for forming a dense, rigid and permanent barrier.

Whilst grass is promoted as an effective filter medium, Deletic (2000) notes that there are problems relating to runoff over grassed surfaces that remain unresolved such as: how soil properties are affected by grass establishment, how evapotranspiration, interception and surface retention are affected by grass cover and how the roughness of a grass surface may be characterised. Furthermore, whilst there is general agreement across the literature with the properties proposed by Wilson (1967), very little specific guidance exists on optimum vegetation properties such as stem number, diameter, rigidity, angle and branching pattern. This may reflect the difficulty in isolating the influence of vegetative factors on VFS performance. Abu-Zreig et al. (2004) found that experimental evaluation of the influence of vegetation type on filter performance was problematical because of the difficulty in constructing identical filters varying only in vegetation type.

*Summary: The literature generally favours grass but very little information exists on ideal species or on properties that could be related to typical species. Species selection is based on experience and anecdotal evidence rather than engineering properties associated with the function of the vegetation.*

### 2.6.4 Stem characteristics

The diameter of a stem determines the cross-sectional area of plant material that comes into direct contact with incoming water, i.e. water flowing into a VFS. It is not unreasonable to assume therefore, that this parameter may play an important role in VFS flow and sediment deposition processes. In fact it seems unlikely that the density of a VFS can be considered without taking into account both the number and the diameter of the stems within. Furthermore, these are factors that may be measured in the field relatively easily, yet are not discussed in the literature on VFS performance.

*Summary: Not discussed in the literature despite potential influence on the interception of incoming flow to a VFS.*

### 2.6.5 Age

Vuurmans and Gelok (1993) identified grass age as an important factor because it determines its stiffness. Young grass was more flexible and bends more easily than older grass. Old grass was slightly more effective in reducing sediment movement and was more effective in retarding the water flow. As well as greater stem stiffness, its greater effectiveness has been attributed to higher grass density and lower frequency of mowing activities (Van Dijk et al., 1996). Schmitt et al. (1999) observed that 25 year old grass strips were more effective than 2 year old grass strips. The older strips delayed the onset of outflow longer and had higher infiltration than the 2 year old strips, suggesting improvements in runoff and sediment reduction over time. No studies were found that compared species types at different ages.

*Summary: older grass found to be more effective than younger grass but this is not quantified.*

### 2.6.6 Length of VFS in direction of flow

Length is used in the current study to describe the distance from the upslope to the downslope edge of a linear feature (as illustrated in Figure 2.5). This dimension is often cited in VFS literature as width but this becomes confusing when used in discussions of the length of the flow through a linear feature and when defining the dimensions of fields and buffers as elements for modelling.

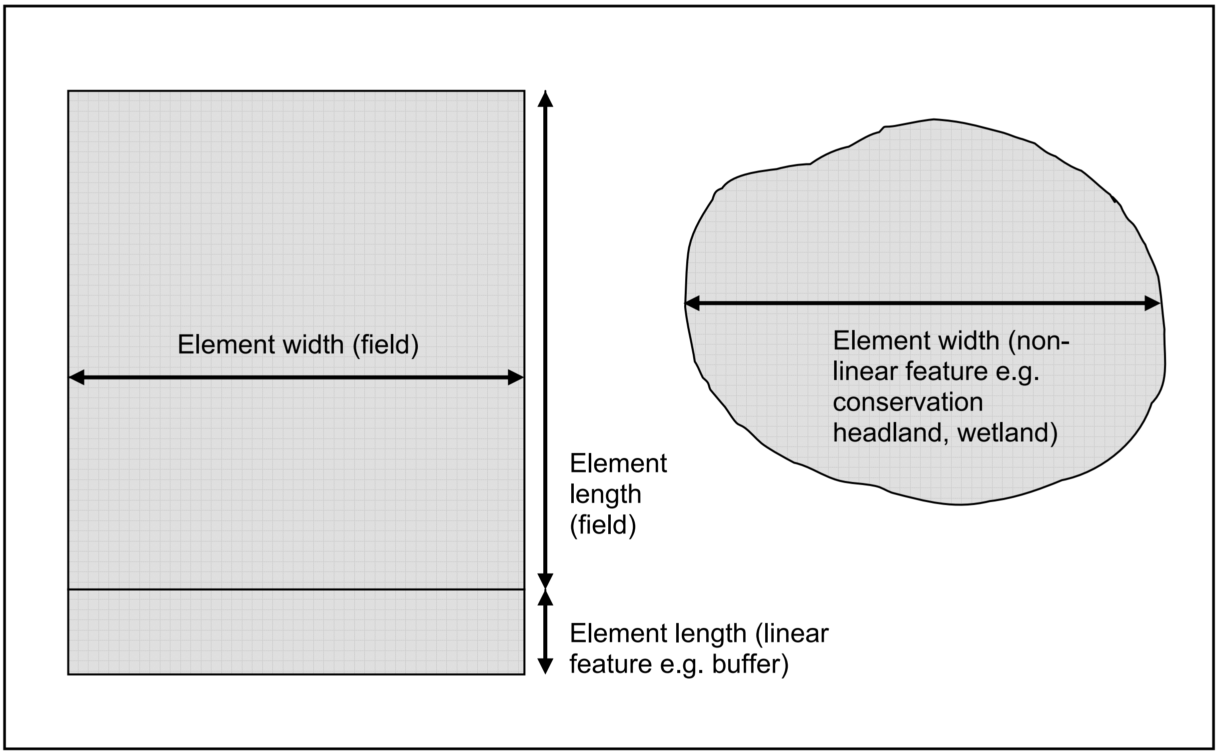



**Figure 2.5 Terms used to describe feature dimensions.**

Trapping efficiency is generally improved when the length of the VFS is increased (Magette et al., 1987; Chaubey et al., 1994; Barfield et al., 1998), with longer VFSs increasing the percentage of mass reduction of both nutrients and sediment (Lee et al., 2000). When tested against other factors filter length has been stated as having the greatest influence on the sediment trapping efficiency of VFSs (Abu-Zreig, 2001); Abu-Zreig et al., 2003, 2004; (Hook P.B., 2003). For a given soil type, filter length also has an effect on the amount of infiltration and consequently on the volume of water outflow and peak flow rate (Abu-Zreig, 2001).

Many examples have been cited in the literature. Lalonde (1998) found that trapping efficiency varied between 68 to 98% as filter lengths increased from 2 to 10 m (Abu-Zreig, 2001). Abu Zreig (2001) observed that the average trapping efficiency for a 15 m filter was three times higher than a filter 1 m long (95 and 30% respectively).

Dillaha et al. (1989) tested six filter strips with lengths of 0, 4.6 and 9.1 m. Sediment trapping efficiency (STE) varied from 53 to 86% on the 4.6 m strips and from 70 to 98% on the 9.1 m filter strips. Coyne et al. (1995) reported a sediment removal efficiency of 99% in two 9 m long filter strips vegetated with tall fescue and Kentucky bluegrass. Similar results were also obtained by Magette et al., (1987) who tested filter strips vegetated with Kentucky-31 fescue established on a silty loam soil. Removal efficiencies of 66% and 82% were obtained for 4.6 and 9.2 m strips respectively.

According to Hook (2003), averaged across other factors (vegetation type, slope and stubble cover), decreasing VFS length from 6 to 1 m reduced average sediment retention from 99 to 83 % which corresponded to 13 times more sediment in runoff. In contrast, moving from wetlands to uplands, or increasing slope from 2 to 20 %, reduced average sediment retention from 96 to 91 % which corresponded to just 2.5 times more sediment.

A number of studies, however, do not show such an obvious relationship. Neibling and Alberts (1979) found very little difference between the amount of sediment retained in strips of length 0.6 and 4.9 m and most of the sediment was deposited at the upslope side of the strip despite its length. Line (1991) observed no significant difference in trapping efficiency between 3 m and 6.1 m long VFSs containing a mixture of ryegrass and fescue. Daniels and Gilliam (1993) obtained lower trapping efficiencies of 45 to 58% for a 3 m long VFS compared with 57 to 62% for a 6 m long VFS. Their study used natural rainfall which is likely to be less intense than simulated rainfall and therefore perhaps not as sensitive to differences in VFS length.

Effective filter length will depend on the size of particles to be trapped, with finer particles requiring longer filter strips to reduce flow rate and hence reduce transport capacity (Jin and Römkens, 2001); Wilson, 1967). Neibling and Alberts (1979) observed that grass strips as short as 0.61 m reduced sediment discharge rates by 90% but the sediment discharge of particles less than 0.002 m through the same length strip was reduced by just 37%. Gharabaghi et al. (2002) suggest that most of the larger sediment particles may be removed in a 5 m grass buffer, but finer particles may require 10 m. Abu Zreig (2001) advocates a poorly vegetated 3 m long filter as sufficient for silt-sized particles, whilst for clay-sized particles the length of the filter should be more than 15 m. This in turn has implications for the pollutant retention efficiency as there is often an inverse relationship between pollutant concentration and particle size, for example Owens and Walling (2002) found this for phophorus.

Factors other than particle size are likely to influence the relationship between VFS length and trapping efficiency. The optimum length of a VFS has been associated with the number of successive rainfall events, soil type, landscape, geology, whether the flow meeting the VFS is uniform or concentrated and the type and load of pollutant. Parsons et al. (1990) noted that increasing the length of a herbaceous filter from 4.3 to 8.6 m reduced runoff and sediments passing through the VFS but a longer VFS was less effective during very intense rainfall, probably due to increased runoff speed. Manning's roughness coefficient ($n$) has a moderate influence on filter performance compared with length and slope. The increase in trapping efficiency with $n$ is much higher at shorter filter lengths compared with longer filter lengths (Abu-Zreig, 2004).

Dabney et al. (2006) suggest that it is wrong to assume that short VFSs do not improve water quality; rather, they suggest that the presence of a continuous VFS edge is critical because the first upslope section of a VFS has a much larger impact than any subsequent section. It is generally agreed that the majority of sediment retention takes place in the first 1 to 3 m of a filter strip (Tollner et al., 1976; Daniels and Gilliam, 1996; Robinson et al., 1997; Pearce et al., 1998b and (Jin C.X. & Romkens M.J.M., 2001). Dillaha et al. (1989) demonstrated an acceptable effect (50 to 80%) with 3 to 5 m wide buffers.

Based on study results, Hook (2003) suggests 6 m as a starting point for designing rangeland buffers for sediment retention, consistent with cropland filter strip guidelines. Under the conditions tested, even the least favourable sites studied by Hook (2003) (10 to 20% slopes) achieved 95 % sediment retention at a 6 m length, suggesting that this length may be effective across diverse rangeland riparian sites. (This length is also recommended by Defra's Environmental Stewardship scheme in the UK.)

Whilst some authors suggest 10 m as the minimum length for satisfactory pollution control (e.g. (Castelle et al., 1994), others (e.g. Neibling and Alberts, 1979; Robinson et al., 1996) observed no appreciable difference between the sediment trapping efficiency of 10 m and 15 m long filters and Schmitt et al. (1999) found that incremental improvements in sediment retention are relatively small beyond 7.5 m lengths.

Even where recommendations exist there appear to be exceptions suggested. VFSs of a given length are less effective for the retention of relatively mobile contaminants e.g. fine sediment (such as clay-sized particles), bacteria and dissolved nutrients than for coarse sediment (Schmitt et al., 1999). Although increasing the filter length beyond 15 m is ineffective in enhancing sediment removal it is expected to further enhance phosphorus removal (Abu Zreig et al., 2003). On level or gently sloping areas with dense vegetation, such as moist floodplain sites, lengths even shorter than the widely recommended 6 m may be adequate (Hook P.B., 2003).

Figure 2.6 is useful in illustrating the complexity of deriving appropriate VFS lengths based on previous studies. Based on the graph a 10 m VFS could provide anything in the range of 0 to 100% pollutant removal. The graph does not, however, distinguish between the natural variability in results and that introduced by differences in experimental methods, scale and site characteristics. Figure 2.7 summarises maximum, minimum and mean VFS lengths based on available literature. It also illustrates the range in suggested optimum VFS lengths but again there is no confirmation of the interaction between experimental parameters. Parkyn (2004) observed that the length required to optimise nutrient removal has been debated with little systematic study of the issue and that one problem in assessing minimum lengths is that many studies have had to use existing VFS lengths, rather than deriving them experimentally. An approach, therefore, may be to model optimum VFS lengths and to validate the simulations using field experiments. A further factor in setting optimum VFS lengths, noted by Hickey and Doran (2004), is the length of VFS that is politically acceptable or the area that landowners can reasonably be expected to lose from production.

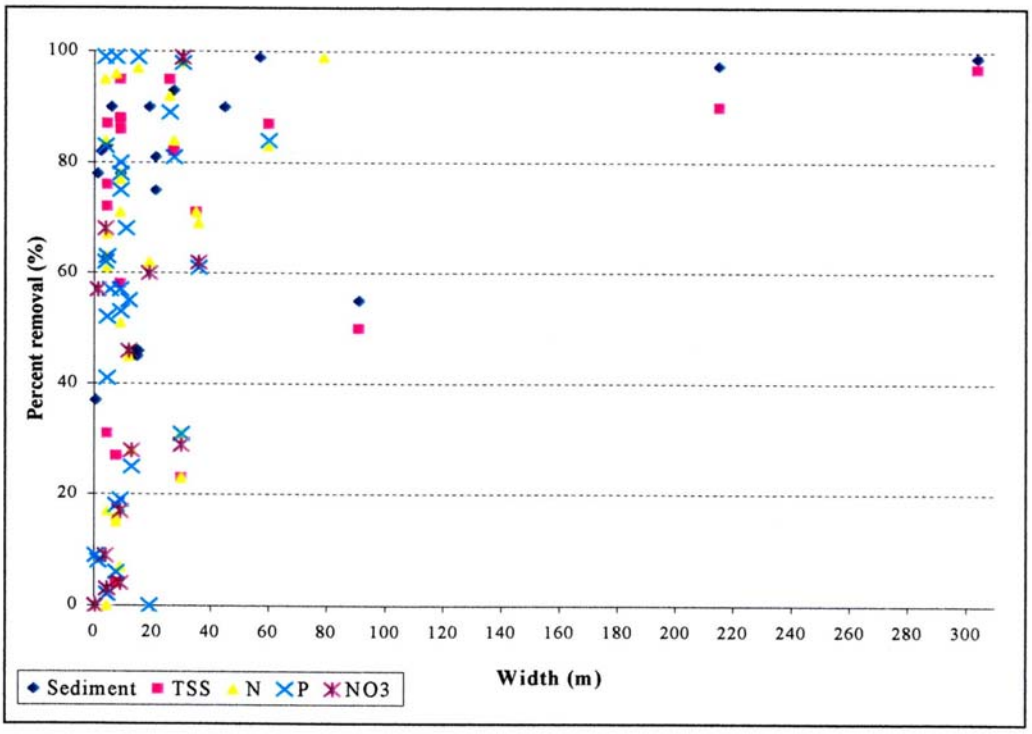



**Figure 2.6 VFS pollutant removal efficiency with length (referred to as width here) (Ducros, 1999).**

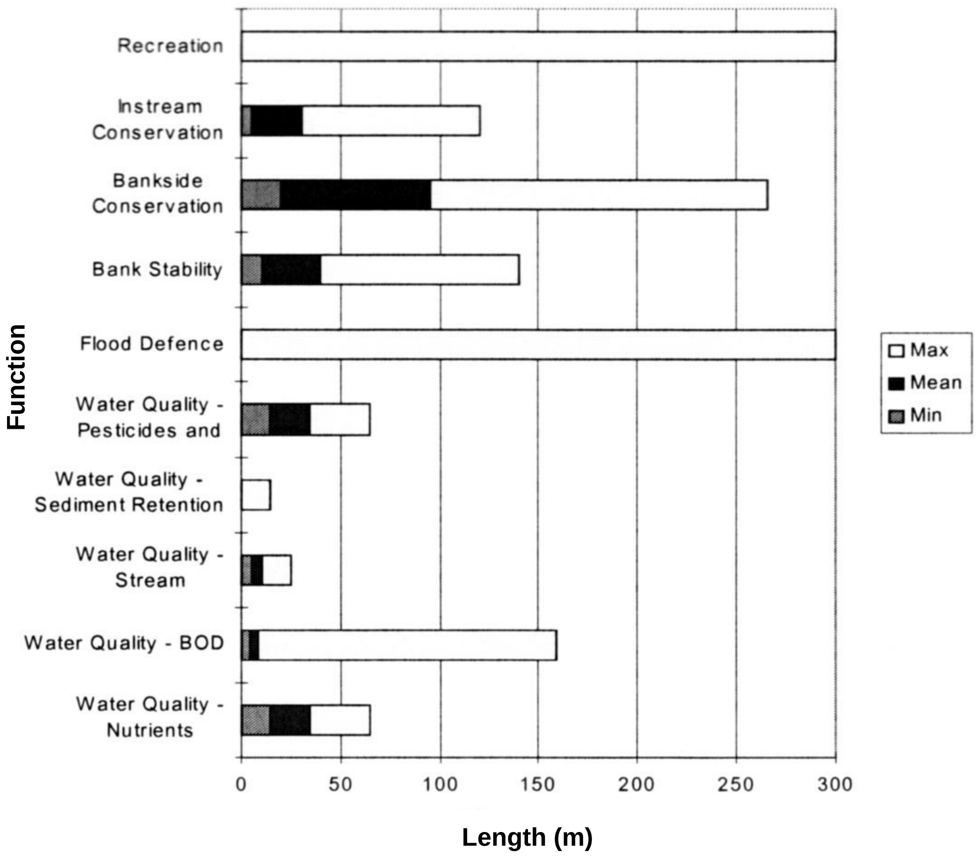



**Figure 2.7 Guidelines on VFS length (Haycock and Muscutt, 1995).**

*Summary: An optimum length is not obvious from the literature although it is suggested that for sediment, increases in length beyond 10 m are unlikely to increase efficiency. This length may be more for clay particles to which pollutants such as phosphorus may be associated.*

### 2.6.7 Other design factors

VFSs may also vary in terms of their shape and the shape of the contributing area. This is likely to affect whether the area delivers runoff to the VFS uniformly or in concentrated channels. The ratio of VFS area to upslope source area, and also the size of the event will determine how much water and sediment are delivered to the VFS. There seems to be no published knowledge of how these factors affect deposition rates in VFSs and what size of event the VFS can cope with. There may also be differences in hydrologic contact time with surface flow, characteristics of the upslope or leading edge and infiltration capacity. VFSs may be exposed to different rates of flow, sediment loads and types of pollutant. These factors do not receive enough coverage within the published literature to be able to quantify or determine their effect on VFS performance.

## *2.7 Implementing Vegetative Filter Strips: Placement*

Related to optimum VFS design is the selection of an effective landscape location. It has already been suggested from Figure 2.3 that different VFS types will have an ideal location in the landscape and this may not always be adjacent to a watercourse. Within the following section the literature is used to determine whether there is an ideal place to locate VFSs and to define what constitutes a 'right' and 'wrong' place.

### 2.7.1 Is placement important?

Soil erosion and sediment delivery processes vary in space with different kinds of processes operating at various locations (Verstraeten et al., 2002). Therefore, the implementation of a conservation measure will be most effective when applied to those locations where it is most suited (Verstraeten et al., 2002).

It has been noted that the position of VFSs in the landscape significantly affects their ability to disperse runoff and remove sediment and associated pollutants from runoff (Qui, 2003). When VFSs were placed along certain segments of a perennial stream in a catchment in southern Iowa, Tim et al. (1995) found that sediment yields were disproportionately reduced. It is thus advised that, in order to obtain the maximum benefits of vegetated VFSs, researchers and resource managers must understand the impact of the physical characteristics of VFSs (e.g. length, placement or location along major streams) on catchment quality.

Blackwell et al., (1999) maintain that riparian filter stripss are often bypassed where natural hydrological flows are intercepted by ditches or drains (which are common features in the UK landscape). Similar conclusions have been drawn by Haycock and Muscutt (1995) who reported that while 85% of some sub-catchments of the River Avon in Hampshire, UK, were served by effective riparian buffer zones, 60% of the polluting material in the river was delivered by roads and drains that effectively bypassed them. Observing the same pattern in Ontario, Canada, Hickey and Doran (2004) suggested that buffer zones may be most effective in preventing the deterioration of water quality in areas where the natural drainage patterns are intact. This suggests that a detailed survey of soil types, ditches and water channels (permanent and ephemeral) should be made prior to VFS implementation in order to ensure a full understanding of field to field to water connectivity.

### 2.7.2 Catchment scale

"There is a requirement to understand more about catchment level hydrology so that buffers are not established in the wrong place" (Wigington, pers. comm., 2004). Verstraeten et al. (2002) suggested that many soil conservation and sediment control techniques are known and widely studied and yet their impact at a catchment scale generally remains unclear. In a more recent study, Verstraeten et al. (2006) used a model approach to investigate the impact of riparian VFSs on river sediment delivery at different spatial scales. Simulating a field plot with a straight slope produced similar sediment reduction results to those obtained by experimental set ups elsewhere (i.e. > 70%). However, at the catchment scale sediment reduction was much less (i.e. ± 20%). This was attributed to a) overland flow convergence reducing VFS trapping efficiency and b) a proportion of the sediment bypassing the VFS via ditches, sewers and road surfaces. Information such as this is crucial in catchment planning for diffuse pollution mitigation.

### 2.7.3 Type of buffer / field position

Model simulations using EUROSEM (Rickson, 1994) suggest that grass strips are effective at reducing soil losses from upslope elements but are not effective at reducing runoff volumes. EUROSEM works on the basis that flow is potentially erosive if its transport capacity has not been reached. Therefore, if a grass strip has removed all or much of the sediment, but not reduced the flow, then the flow's potential erosivity (i.e. transport deficit) is increased. Regarding VFS placement, Rickson (1994) suggests that if the model output is realistic this implies that the grass strip should always be placed at the bottom of the slope, to reduce erosion downslope of the VFS, rather than as a method for breaking slope length. However, Dabney et al. (2006) suggests that various benefits may be gained from different field positions.

At the field scale Dabney et al. (2006) categorise a number of VFSs according to the position at which they might best perform their intended function and describe the benefits of each location. (This is in good agreement with Figure 2.3.) They list:

a) In-field VFSs: grassed waterway, contour buffer strip, alley cropping, vegetative barrier.
b) Edge-of-field VFSs: field border, filter strip, riparian forest buffer, vegetative barrier.
c) After-field VFSs: constructed wetlands, vegetated ditches, channel bank vegetation.

In-field VFSs are recommended close to the source because they offer the best opportunities to encounter sheet flows and therefore can most effectively reduce runoff and control erosion and pollutant transport. These VFSs are complementary to edge-of-field and after-field features. On flat lands, VFSs that impede drainage within agricultural fields are impractical so edge-of-field features are advocated. After runoff has left the field and passed through edge-of-field VFSs, water quality can still be improved by after-field VFSs (Dabney et al, 2006).

*Summary: VFS types have been classified according to location at which they are likely to best perform. There is evidence to suggest that VFS position may influence flow and sediment deposition dynamics.*

## 2.7.4 Critical areas

Identifying critical areas provides a means of directly linking VFSs with problem sites, thereby targeting specific locations rather than promoting widespread implementation of VFSs throughout a catchment. Few attempts have been made to establish criteria for selecting critical areas for implementing VFSs (Maas et al., 1985). Maas et al. describe two distinct critical-area perspectives: the land resource perspective and the water resource perspective. Critical areas, from the land resource perspective, are those lands on which soil loss exceeds the soil loss tolerance (i.e. defined by Maas et al. (1985) as the rate at which soil can be replaced by natural processes). However, areas of severe soil loss may not always be the most critical areas for the treatment of agricultural diffuse pollution. Therefore the water resource perspective recognises areas where the greatest improvements to an impaired water resource can be obtained for the least investment in best management practice (i.e. this may be based on the value of the water resource or the proximity of the sediment source to a watercourse). Although not mentioned by Maas et al. (1985) it should be noted that the land resource perspective may also be based on management. It is also possible that the land resource perspective holds less relevance for the UK, where soil loss is likely to be considered permanent, given that the rate of natural soil replenishment means that soil is effectively a non-renewable resource.

### 2.7.4.1 Determining critical areas

Determining critical areas requires consideration of factors such as the type of water quality impairment, the dimensions and dynamics of the impaired water resource, the hydrology of the catchment, the magnitude of pollution source areas, and the investment in Best Management Practices (BMPs). Implied in this approach is the concept of treatability of the resource (Maas et al., 1985). Out of a number of projects investigated by Maas et al. (1985) erosion rate, distance to nearest watercourse, on-site evaluation and present conservation status were the most commonly used factors for critical area selection. Other factors included type of water resource impairment, manure sources, fertilizer rates and timing, pathogen source magnitude, distance to nearest watercourse, planning timeframe and designated high priority sub-basin.

## 2.7.5 Source-river connectivity

Collins and Walling (2004) recognise the importance of understanding catchment suspended-sediment sources for designing and implementing targeted sediment control measures. Their work explores approaches to documenting catchment suspended-sediment sources and the following paragraph summarises the problems they outline in relation to VFS placement.

Sediment mobilisation and transfer at the catchment scale is spatially complex. Few methods for establishing catchment suspended sediment sources take into account source-river connectivity and the uncertainties associated with sediment routing. The potential for different parts of a catchment to contribute to the downstream suspended sediment flux is controlled by factors categorised into strength (e.g. low strength may be exhibited by bare rolled soil), morphological (i.e. terrain), locational (e.g. proximity of sediment source to river) and filter (e.g. density of roads, paths, tracks or field drains and presence of VFS) resistance. These factors of resistance interact in controlling sediment mobilization and delivery to river channels and the interplay is catchment specific. For example, a sediment source may be characterized by low strength and morphological resistance to erosion but if location and filter resistance are high, the sediment released may not contribute significantly to the suspended-sediment load measured downstream. Such concepts, as discussed by Collins and Walling (2004), suggest that sediment sourcing may be an important basis for the effective placement of linear practices such as VFSs.

### 2.7.6 Precision conservation

Dosskey et al. (2005) propose the use of precision spatial information technologies and procedures to implement VFSs. This is based on the premise that some locations along a VFS will be more likely to receive runoff than others and that VFS efficiency can be improved by targeting those sections. The approach allows for site specific management recognising that field runoff typically converges in some areas and diverges in others due to uneven topography and differences in soil conditions and farming practices. The approach therefore designs variable length VFSs in order to obtain a uniform level of pollution control along the VFS. This is achieved by calculating the runoff load for field areas contributing runoff and computing the length of VFS required to achieve a desired level of control on the runoff load. The approach has limitations in the cost of data, software and equipment, which may be offset by the gains in more efficiently placed VFSs. Farmers can alter the direction of flow over a field through their direction of ploughing so runoff discharge points may be ends of furrows at the sides of fields and not at the base of fields (i.e. along the direction of the field slope). This would be difficult to build into a GIS. Variable length buffers also pose an inconvenience to farmers compared with straight field margins.

*Summary: Various strategies have been proposed for placing VFSs. However, attempts to validate have been few and have been area specific. It is likely that strategic targeting is more practical and cost effective than widespread implementation but "best" locations are likely to be site and problem specific, precluding the likelihood of a "one size fits all" strategy.*

## 2.7.7 Soil properties

### 2.7.7.1 Texture

Smaller sediment particles will have a larger surface area and hence more propensity to attach to pollutants. It is well known that, for sediment-associated elements such as certain metals and nutrients, there is a relationship between particle size and the concentration of the chemical elements. For heavy and trace metals and radionuclides etc there is a fairly large amount of literature (for example, Van der Perk, 2006) on this. Work by Owens and Walling (2002) however, is one of few studies to examine this relationship for nutrients. Examining a bulk suspended sediment sample from the River Aire, England, they found a very strong relationship between phosphorus content and particle size, as measured by specific surface area. This demonstrates a strong affinity of phosphorus for particulate material and supports statements by others (for example, Pionke and Kuniski, 1992) on the likely relationship between phosphorus and particle size. Studies such as this suggest that a VFS capable of trapping clay particles will be more effective in mitigating diffuse pollution arising not only from the siltation of surface waters, but from sediment-associated pollutants.

### 2.7.7.2 Drainage

Soil drainage properties influence VFS performance, with free draining soils minimising the generation of surface runoff both on the hillside and within a VFS (Parkyn, 2004). Therefore, for maximum removal efficiency the VFS soil should have a high permeability to enable infiltration of a large amount of runoff (Lee et al., 2000). Infiltration not only decreases runoff, and thus reduces sediment transport capacity, but is also important because finer particles and soluble nutrients enter the soil profile along with infiltrating water (Lee et al., 2000). This suggests that VFSs on clay soils, which typically have a low infiltration rate, will be less effective than those on free draining sandy soils. However, being more resistant to erosion, clay soils are less likely to be a source of sediment.

### 2.7.7.3 Depth

Runoff is generated quickest, and seepage forces (that encourage erosion) are greatest on the shallowest soils in a catchment. Dabney et al. (2006) suggests that these are, therefore, the zones that need the greatest protection and are also amongst the least productive areas from an agricultural point of view. Against this, some erodible soils are easier to work with and do give high crop yields, especially sandy loams. Shallow soils ensure that ground water plumes have hydrological contact with (and do not pass beneath) the riparian root zone, increasing potential for buffering nutrient fluxes by denitrification and for nutrient uptake (Tabacchi et al., 1998).

### 2.7.7.4 Erodibility

Soil erodibility is defined by McIntosh and Laffan (2005) as “the inherent susceptibility of soil particles or aggregates to become detached or transported by erosive agents such as rainfall, runoff, throughflow, wind or frost”. The two most significant soil characteristics influencing erodibility are infiltration capacity and structural stability, which are in turn affected by soil properties such as organic matter content, soil texture, the nature and amount of swelling clays, soil depth, tendency to crust and the presence of impervious layers (Brady and Weil, 2002). Few VFS studies make reference to differences in soil erodibility and its potential impact on VFS performance. However, the soil erodibility of the contributing field will influence the amount of sediment that is delivered to a VFS. A VFS will most likely perform better where it is not inundated by sediment delivery. Of course it is areas where sediment delivery is a problem that will benefit from VFS placement, but where sediment delivery is high enough to inundate the VFS, in-field measures to control sediment detachment should be combined with VFSs. The literature does not indicate such threshold values for VFS inundation by sediment, and it is likely to vary with design and site characteristics.

*Summary: VFSs will perform best on high porosity soils. Shallow soils are likely to require the most protection. Erodibility of the contributing field will influence sediment flux to the VFS, and where possible should be controlled by in-field measures.*

### 2.7.8 Slope angle

Jin and Römkens (2001) proposed that as slope increases, the trapping efficiency of a VFS will decrease as deposition moves downslope through the VFS. De Ploey et al. (1976) found, in their experiments that when slope exceeds 10°, grass covered slopes could result in more erosion than bare soil slopes because of concentrations of fast flowing run-off between individual clumps of vegetation. Similarly, Ligdi and Morgan (1995) observed that contour barriers (simulated with metal rods) provided good protection on slopes up to 6° but resulted in double the erosion of bare slopes at 10°. Furthermore, this confirms recommendations by the FAO (1976) and the experiments of Wenner (1981) with respect to the suggested maximum slope for which VFSs are an effective soil conservation measure (see Table 2.3 for values).

Tadesse and Morgan (1996) deduced that on the steeper slopes the main effect of the barrier is to filter sediment, whereas on the gentler slopes the main effect is to pond the runoff and thereby increase infiltration and reduce runoff volume. On steep slopes the ponded area is small and rills, developed in the non-cohesive material in the deposition zone, are able to extend their channels through the barrier and on to the slope below. Abu-Zreig (2001) observed that with a vegetative cover, slope angle showed a minor effect on VFS trapping efficiency, with the effect decreasing with increasing VFS length.

At the farm scale Dillaha et al. (1989) observed that in hilly areas water concentrated in natural drainage ways within the fields before reaching the VFSs. Consequently, flow across the VFSs was primarily concentrated, resulting from larger storm events, and the filters became inundated and ineffective, with little sediment accumulation observed. VFSs were more effective in flatter areas and significant sediment accumulation was observed on uniform slopes. However, at the upslope edge of older VFSs (one to three years) deposited sediment had created a ridge higher than the adjacent field, along which runoff would flow until a point where it crossed the VFS as concentrated flow. “Recognising the importance of flow redirection caused by edge-of-field berms, [US] national practice standard for VFSs specifies that the gradient along the edge of the filter strip must be less than 0.5% and it calls for the field upslope of the filter strip to have a slope steepness of between 1 and 10%" (Dabney et al., 2006).

The slope angle at which buffers may perform effectively can be increased where dense vegetation exists and this has been attributed to the greater reduction in flow velocity achieved with a more uniform vegetation structure. Jin and Römkens (2001) simulated a VFS using bunches of polypropylene bristles inserted into a flume. They found that whilst a VFS with a density of 2,500 bunches/m$^2$ was effective in trapping sediment on slopes up to 4%, a VFS with a density of 10,000 bunches/m$^2$ was effective on slopes up to 6%. Taddesse and Morgan (1996) suggested that barriers with a dense root system and an interwoven mat of rigid leaves were effective on slopes of up to 23%, whereas more open tussocky species were effective only up to 16%. Boubakari and Morgan (1999) confirmed this, proposing that perennial grasses with dense leaf, stem and root systems will control erosion on steep slopes largely by ponding water upslope of the barrier, encouraging infiltration and promoting deposition of sediment within the ponded area. Conversely, grass species with low stem densities and low rigidity will concentrate flow within the strip and provide passages whereby rill channels can extend through the barrier on to the slope below (Taddesse L.D. & Morgan R.P.C., 1996).

| Recommended maximum slope (%) | Slope angle (°) | Reference |
|---|---|---|
| **10** | 5.7 | Hayes & Dillaha (1992) |
| **10** | 5.7 | Herson-Jones et al. (1995) |
| **11** | 6.3 | Ligdi & Morgan (1995) |
| **15** | 8.5 | Neiswand et al. (1990) |
| **15** | 8.5 | FAO (1976) |
| **16** | 9.1 | Taddesse & Morgan (1996) |
| **18** | 10 | DePloey et al. (1976) |
| *Mean = 13.6* | ***7.7*** | |

**Table 2.3: Recommended maximum slope angles taken from the literature**

*Summary: The literature reviewed presents concurring recommendations on slope angle (Table 2.3). Based on those studies the mean maximum slope angle for effective sediment trapping by VFSs can be calculated as 7º and, with a dense uniform grass cover, this might be increased to 12°. Differences in experimental approach have not been factored into this calculation.*

## 2.7.9 Economic cost

Another factor in determining the best placement of VFSs is the economic cost. Nakao and Sihngen (2000) suggest that the cost of reducing soil erosion varies with site characteristics across a catchment. They suggest that very few studies have assessed how cost effective riparian buffer zones (or other filter strips) are for reducing environmental impacts. Their research revealed that the costs of riparian buffers are likely to vary with factors such as field shape, tillage methods and field size, plus soil type and the effectiveness of the buffer.

The Defra report, PE0205, associated with the work reported in this thesis produced an analysis of the cost benefits (financial and environmental) of installing grass filter strips (Wood et al., 2007). However, this is not discussed further here as it does not explicitly provide placement guidance based on economic cost.

*Summary: Cost effectiveness of VFSs is likely to vary with site characteristics such as field shape, tillage methods, field size, soil type and VFS effectiveness.*

## 2.7.10 Buffer management

"Longer-term performance of filter strips can decline if not managed properly" (Dosskey, 2001).

### 2.7.10.1 Time

VFS performance is expected to vary over time and to depend on management criteria (Borin et al., 2005). A better understanding of long-term accumulation of pollutants has important implications for management of VFSs to maintain filtering capability (Dosskey M.G., 2002). For example, according to Vellidis et al. (1994) the efficiency of new or restored VFSs initially increases with time. This effectiveness later decreases as sediments accumulate and encourage concentrated flow across the VFS (Dillaha T.A. & Inamdar S.P., 1997). McKergrow et al. (2004) observed a release of previously trapped sediment from a wooded VFS over three years when vegetation density was low due to a lack of understory. Conversely, Parkyn (2004) suggested that dissolved nutrient uptake by plants may be greatest during early growth phases and decline as vegetation matures. During experiments by Dillaha et al. (1989) the effectiveness of VFSs for sediment removal decreased with time as sediment accumulated in the VFS. They proposed, however, that this may not be a problem in "real world" VFSs because vegetation should be able to grow through most sediment accumulations. The success of VFSs in surviving burial by sediment will be a function of random variables associated with rainfall, runoff, vegetative growth rate, depth of sediment accumulation (Dillaha T.A. et al., 1989) and other factors including the plant species and the age of the VFS at the time of submergence.

Rodriguez (1997) claimed that as roots and other plant residues lodge against a grass hedge, hydraulic resistance progressively increases, causing deeper backwaters, longer settling lengths and increased sediment trapping so long as the grass hedge is strong enough to remain erect. Over 6 years it was observed that grass hedges (in Venezuela) caused substantial sediment deposition as evidenced by the appearance of berms in front of the hedges.

Whether sediment removal is effective over the long term appears to be a matter of debate. Cooper et al. (1987) found that a riparian zone was a sediment sink over the 20 year period they studied. Lowrance et al., (1986) reached a similar conclusion examining sediment deposition over a 100 year period. Both of these studies were conducted in catchments that were characterised by >50% forest cover. Whether narrow VFSs are able to retain sediments over the long term is not clear as most studies have been too short in duration to detect remobilisation of sediments during infrequent intense storms (Hickey and Doran, 2004).

#### 2.7.10.2 Runoff characteristics

Borin et al. (2005) suggests that the amount of water leaving the field is the driving force determining pollutant losses to surface water bodies. They observed that total pollutant losses from fields with VFSs were mainly due to very few events. Furthermore, runoff depth from their experiments was positively correlated with total suspended-sediment concentration suggesting that low flow volumes are more easily filtered.

Field and plot studies generally confirm that VFSs are less efficient for pollutant retention when concentrated flow occurs (Dosskey et al., 2002). Dillaha et al. (1989) suggests that 'unless vegetated filter strips can be installed so that concentrated flow is minimized, it is unlikely that they will be very effective for agricultural non-point source pollution control' (which includes sediment retention). Concentrated runoff may develop on fields with long slopes or where the terrain creates natural drainage channels perpendicular to the VFS. Compared with sheet flow the higher velocity and depth of concentrated flow is likely to create breach or failure points along a VFS. When concentrated flow forces the grass to bend, the sudden drop in resistance encountered by the flow creates an increase in velocity and a decrease in sedimentation (Vuurmans and Gelok, 1993). Where water reaches the VFS at a few points as concentrated runoff, sedimentation will be less than expected on the basis of plot measurements (Dillaha et al., 1989).

Dosskey (2002) quantified the effect of concentrated flow by comparing VFS performance with uniform and non-uniform flow. He used mathematical relationships to estimate that VFSs at four farms could potentially remove 99%, 67%, 59% and 41% of sediment from field runoff if the runoff is uniformly distributed over the entire VFS area. However, because of non-uniform distribution it was estimated that actual sediment removal would be only 43%, 15%, 23% and 34% respectively.

As mentioned earlier, concentrated flow may also cross the VFS where sediment accumulation creates berms that divert subsequent field runoff along the front of VFSs to low points. A similar effect was observed by Dillaha et al. (1989) on farms where mouldboard ploughing was practiced. When soil was ploughed away from the VFS, a shallow ditch was created parallel to the VFS. If this ditch was not removed, runoff would concentrate and flow along the front edge of the VFS until it reached a low point and crossed as channel flow. Maintenance of sheet flow conditions may require periodic removal of accumulated sediment or other modification of surface topography (Dosskey, 2001).

Dosskey et al. (2002) suggest that flow should be managed to improve the distribution of field runoff across a VFS. This includes removing sediment accumulations and tillage berms to maintain the VFS at a lower height than the upslope field. Correll (2005) suggests that, in order to create sheet surface flows, the grass component of any VFS should be intensively maintained by mowing and berms of trapped sediment re-contoured and replanted. Within the field it has been suggested that crop row direction and VFSs are located along the contour, rather than at a constant distance from the stream (Dosskey et al. 2002), and that runoff is dispersed from any drainage ways or ephemeral channels at the foot-slope so that VFSs can function without being overloaded (Daniels and Gilliam, 1996).

The United States Department of Agriculture (USDA) standards require that concentrated flow be dispersed by in-field measures before entering a VFS. However, the precision conservation approach proposed by Dosskey et al. (2002) (and described in the earlier section on VFS placement) promotes that the VFS itself be used to address concentrated flow by varying length according to the risk of concentrated flow.

#### 2.7.10.3 Other management features

Management practices that might improve VFS performance include:

- In-field practices that reduce total sediment load to VFSs, such as implementing appropriate tillage, land shaping and in-field features can improve the trapping efficiency of VFSs so long as other runoff characteristics, such as size distribution of sediment particles, remain relatively unaltered (Dosskey M.G. et al., 2002).

- The formation of berms at the edge of VFSs, by tillage operations parallel to contour VFSs and perpendicular to waterways, act as linear elements that interact with topography and soil properties and may alter runoff patterns (Dabney et al., 2006). To avoid these berms, contour filter strips must be periodically renovated (USDA-NRCS, 1999).

- In contrast, VFSs can be designed to control runoff by using these berms as miniature gradient terraces to redirect runoff to a stabilised concentrated flow outlet (USDA-NRCS, 1999).

- To promote plant growth and the trapping of suspended solids, herbaceous VFSs should be mown, and hence vegetation height managed, and the residues removed two to three times per year (Dillaha T.A. et al., 1989).

Management practices that might impede VFS performance include:

- Traffic on the field lanes during wet seasons can expose bare soil which then serves as a secondary field edge or sediment source (Daniels and Gilliam, 1996).

- Uphill-downhill farming is not recommended because it increases erosion rates and counteracts the improvement in the trapping efficiency that VFSs create by encouraging runoff distribution (Dosskey et al., 2002).

*Summary: Hydraulic resistance of forested VFSs increases over time but sediment in narrow VFSs may be at risk from mobilisation by intense storms. VFSs may be less efficient when flow is concentrated. VFS performance can be enhanced or impeded depending on management practice.*

## *2.8 Buffer performance*

### 2.8.1 Reduction in sediment and runoff

Performance is most commonly reported in the literature as the difference in soil loss from a buffered field compared with that from a non-buffered field of the same dimensions and characteristics; commonly referred to as VFS trapping efficiency. Runoff reductions may also be derived in the same way but this is less commonly reported. Table 2.4 summarises the results of a selection of VFS studies representing a range of experimental locations and conditions. It is difficult to directly compare the studies because of the range in characteristics of both the contributing area and the VFS itself. Further to the characteristics listed in the table, the experiments are likely to differ in rainfall characteristics, flow patterns, length of monitoring period, soil moisture and climate, as well as methods of sediment collection. At just one site Le Bissonnais (2004) observed a range in VFS efficiency from 7 to 100% and attributed such a high variability in inter-event efficiency to differences in rainfall characteristics, soil moisture and surface conditions.

### 2.8.2 Sediment particle size

A small number of the studies discuss VFS performance in relation to the size of trapped sediment. Le Bissonnais et al. (2004) observed that sediments deposited in the grass strip were enriched with sand and coarse silt, whilst the material moving through the VFS contained twice as much clay and fine silt as the soil surface horizon. McKergrow et al (2004) recorded an 80% reduction in sediment with 2 to 4 mm diameter aggregates of clay and silt but only a 25 to 65% reduction in suspended-sediment less than 2 mm. The majority of sediment > 2 mm was deposited at the upper edge of the VFS. Daniels and Gilliam (1996) also observed a greater removal of sand in the VFS and a progressive increase in the proportion of silt and clay with flow distance through the filter.

Despite the experimental and site variables, in order to investigate some of the design and placement factors discussed earlier, Figure 2.8 and Figure 2.9 illustrate sediment trapping efficiency plotted against VFS length and against the slope of the contributing area respectively. The studies represent a small range of VFS lengths but there does appear to be an increase in sediment trapping efficiency up to 10 m. The relationship with slope is less obvious with the $R^2$ value showing that less than 1% of the variation in sediment trapping efficiency is due to slope.

| | *Contributing area* | | | | *VFS* | | | |
|---|---|---|---|---|---|---|---|---|
| **Reference & country of study** | **Land use** | **Length (m)** | **Slope (°)** | **Soil type** | **Species** | **Length (m)** | **Reduction in sediment* (%)** | **Reduction in runoff* (%)** |
| *Daniels & Gilliam (1996) USA* | Row crops | | 4 to 15% | Sandy loams & silt loams | Grass (*Festuca arundinacea*) | 6 | 60 to 90 | 50 to 80 |
| *Dosskey (2002) USA* | Corn, grain, sorghum, soybeans | | 1 to 9% | | Trees & grass | 5 to 61 | 67 (potential), 29 (actual) | |
| *Barfield & Albrecht (1982) USA* | Previously mining activities | | 17% | | Grass (Kentucky fescue) | 21 | 95 to 100 at the start, 75 at end of 10 month period | |
| *Borin (2005) Italy* | Maize, winter wheat, soybean | 35 | 1.8% | Loam | Trees (*Plantanus hybrida*), shrubs (*Viburnum polulus*), grass (*Festuca arundinacea*) | 6 | 78 | 80 |
| *Abu-Zreig (2001) USA* | | | 2.3 to 5% | | Grass | 2, 5, 10 & 15 | 65 to 92 | |
| *McKergrow et al. (2004) Australia* | Banana crops | | 7 to 13% | Clays | Grass (*Brachiaria decumbens*) | 15 & 20 | > 80 (25-65 for suspended sediment) | |
| *Van Dijk (1996) The Netherlands* | | | 2.3 to 8.5% | | Grass | 1, 4.5 & 10 | 50-60 (1 m), 60-90 (4.5 m), 90-99 (10 m) | |
| *Le Bissonnais et al. (2004) France* | Winter wheat | 54 | 3 to 5.8% | Silt loams | Ray grass | 3 & 6 | 76 in 1$^{st}$ season and 98 in 2$^{nd}$ | 77 |

* Average values or ranges

**Table 2.4 Summary of contributing area and VFS characteristics for a selection of VFS studies.**

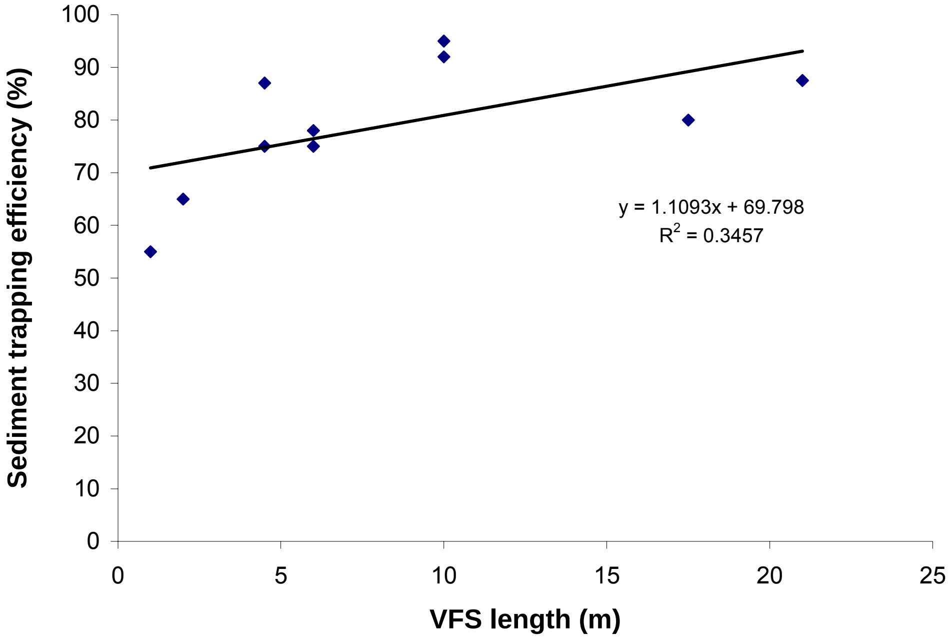



**Figure 2.8 Sediment trapping efficiency plotted against VFS length.**

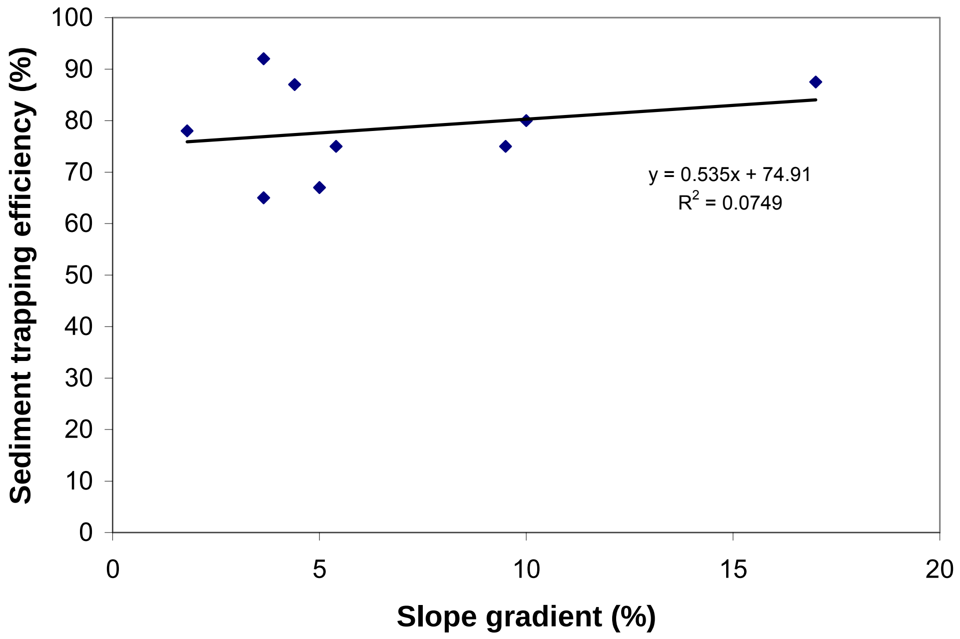



**Figure 2.9 Sediment trapping efficiency plotted against slope of the contributing area.**

## *2.9 Available guidance*

Design criteria for VFSs in the UK are vague (Deletic, 2000). Readily accessible guidance is restricted to the list of options available within the Environmental Stewardship Schemes detailed earlier. Similarly in the USA, the United States Environmental Protection Agency provides a document entitled “Guidance Specifying Management Measures for Sources of Non-point Pollution in Coastal Waters” (USEPA, 1993) but despite its title it again lacks advice on actual design and placement properties.

## *2.10 A modelling approach to VFS design and placement*

Models provide a useful tool for simulating VFS performance, particularly given the lack of long term monitoring data available. A modelling approach allows for the creation of different combinations of design and placement parameters, thus enabling scenario analysis. A number of models have demonstrated success in the evaluation and design of VFSs.

Soil erosion models such as EUROSEM, CREAMS and WEPP, for example, have been adapted for use in predicting VFS performance, whilst models such as REMM, TRAVA and VFSMOD have been developed specifically for applying to VFS work. However, many of the models are data intensive and require an experienced modeller to use them. The majority have been tested only in the US and therefore may not be applicable to conditions elsewhere. Very few models take into account concentrated flow, assuming instead that flow uniformly reaches the upslope edge of a VFS. Furthermore, representation of the vegetative component of a filter strip is, in most cases, limited to the use of coefficients in place of measurable properties, whilst representation of sediment texture is limited to a single size class.

It is clear from the literature that, for models to gain acceptance by land managers, planners and policy makers, they must be based on comprehensive process understanding whilst remaining easy to use, quick to develop and not parameter intensive. Modelling approaches to VFS evaluation and design are reviewed in Chapter 5.

## *2.11 Summary, conclusions and gaps in literature*

In this section, each of the questions posed at the beginning of the chapter are addressed.

### a) What is the optimum VFS design?

Table 2.5 summarises the information on VFS design and placement found within published literature and where possible guidelines are included. Studies have compared various design variables such as differences in VFS height, length and density. Trends have been observed and in most cases these are broadly consistent across experimental studies. However, results are generally site specific and it is difficult to derive general guidance. It may be that it is not possible to provide general rules and that VFS design should always be considered specific to the site characteristics and potential sediment loading. In particular there is a dearth of literature describing the relationships between vegetative parameters and the ability of VFSs to filter and retain sediment. For example, whilst the importance of stem density is discussed, it is not quantified with respect to the ideal number and diameter of the stems. This level of design information is crucial in order to develop empirically based design rules to promote best practice whilst avoiding designs and locations which may, at worst, exacerbate conditions.

The current review has focused on sediment and sediment associated pollutants but when taking a holistic diffuse pollution mitigation approach there may be conflict in optimal VFS characteristics for different pollutants. For example, saturated conditions will encourage denitrification but will not be beneficial for the removal of phosphorus. VFS design should ideally be targeted to the specific pollution problem being addressed. However, this must be balanced against the problem of "pollutant swapping" whereby the reduction by VFSs of one pollutant is offset by increases in another.

#### b) What is the optimum VFS placement?

Buffer design has been widely studied through controlled plot scale experiments, but clearly a full appreciation of VFS performance can only emerge through consideration of the catchment characteristics in which the VFS operates. Since little evidence exists to illustrate the implications for incorrect VFS placement it is difficult to determine from the literature whether there is a "right" or "wrong" place to locate VFS. It is possible however, that poorly placed VFSs may enhance erosion beyond that which might occur with no intervention (e.g. Rickson 1994). The inter-relational effects of VFS placement must be considered, for example, the possibility that stripping of sediment from flow may actually increase its transport capacity or erosivity.

It is not clear, at the catchment scale, which locations are most effective and whether placement should be based on sediment yield, on connectivity to a watercourse or on the value of the watercourse to be protected. At the field scale the most commonly studied factors are slope angle and VFS length. At the field and laboratory scale previous work has been undertaken on planar slopes (Verstraeten et al., 2006) and little information is available on whether the mechanisms of sediment trapping and the effectiveness of the barriers alter if the barrier is placed on convex or concave breaks of slope.

Effective VFSs need to be located to divert or intercept hydrological pathways through which sediment and associated pollutants are transported. VFSs must be understood and modelled in respect of the landscape and catchment characteristics on which they operate in order to allow advice to be transferred to policy makers, farmers and land owners.  On relatively small farms, typical of UK agriculture, where farmers might be reluctant to take large areas out of production, the strategic placement of VFSs (and buffer features in general) may be critical. Indeed it is likely that the strategic placement of VFSs at a few key locations may be the most cost-effective solution.

#### c) How should a VFS be managed? How long will a VFS perform its role and what magnitude of storm will it withstand?

Very few long term studies exist so it is difficult to draw conclusions on the effectiveness of VFSs over time. It is generally agreed that appropriate management will maximise the value of VFSs. The key elements of VFS management are minimising concentrated flow and preventing the submergence by deposited sediment of filter vegetation. No thresholds were found within the literature for storm magnitudes causing VFS failure. In fact, there is very little on observed VFS failure.

### d) What level of water quality improvements can be expected?

Despite the results recorded for sediment retention and runoff reduction in VFS experiments (Table 2.4) there is a lack of reported results for water quality improvements following VFS establishment. It is likely that studies have not been performed over the length of time required to obtain these results. It is also likely that the impact of VFSs would be difficult to isolate from other changes in land use and catchment management. Paired catchment studies may be the most effective way to monitor changes in water quality.

### e) In what form does practical guidance exist for establishing VFSs?

Despite the wealth of studies on VFSs very little practical advice exists for appropriate VFS implementation. One factor contributing to this may be the difficulty in amalgamating information which refers to such a range of terms (including filter strip, buffer zone, grass contour strip, grass hedge, riparian buffer, vegetative barrier) and which is spread over a wide range of scientific disciplines. This may well impede the transfer of information to those that are involved with the practical establishment of VFSs. The sections at the beginning of this review provide some clarification of the various terms used in published literature.

### Other gaps in the literature

Most of the studies reviewed have been conducted in laboratory or controlled field experimental conditions and few quantitative data exist on VFS performance under natural field conditions (McKergow et al., 2004). Rickson (2006) demonstrated that, due to differences in erosion processes and erosion rates at different scales simple "scaling up" is not possible in erosion studies. "There are still very few studies that measure the effectiveness of either VFS or riparian forest buffers under natural rainfall conditions at a scale appropriate to represent management units realistically" (Lowrance and Sheridan, 2005). Whilst valuable for observing processes, the storm, vegetation and topographic conditions of simulated, scaled studies may not accurately represent those of natural systems.

Related to this there is a lack of detailed monitoring following the establishment of VFSs (Briggs et al., 1994) or any established methods for such monitoring. Not only does this have implications for source-sink relationships during extreme events but also for seasonal differences in VFS performance and whether sediment and associated pollutants stay within the VFS or move through it. If VFSs are at risk of becoming pollutant sources over time then mowing and harvesting may be required to remove pollutants. In this case non-riparian VFSs may be favourable in order to keep the destabilisation and erosion associated with harvesting further away from watercourses. However, the majority of the literature focuses strongly on riparian zones.

Whilst minimising concentrated flow through VFSs is recommended, there is little quantitative information on the impacts of breach or failure points due to concentrated flow caused by flow diversion, gully formation, poaching, farm gates etc. This information would be valuable in determining whether entire field widths should be buffered or whether the same level of sediment retention could be achieved, more cost-effectively, by focusing on areas of the field where flow concentrates.

| *Section* | *Factor* | *Level of guidance* | *Guidelines provided by available literature* |
|---|---|---|---|
| Design | Height | General agreement on optimum range but this may only reflect range of heights tested | 100 to 150 mm |
| | Density | No quantitative guidance but agreement that higher is better | High density |
| | Species / vegetative properties | No guidance on species and very little on properties | 1. Uniform, erect grass with a range of stem & leaf angles<br>2. Dense rooting system |
| | Age of vegetation | No quantitative guidance | Older grass |
| | Length | General agreement that performance improves with lengths up to 10 m but contradictory results on optimum length | 6 to 10 m for sediment and particulate pollutants, longer for dissolved pollutants |
| | Other | | 1. Contributing flow lengths < 45 m<br>2. Water table close to the ground surface<br>3. Permeable soils |
| Placement | Critical areas | Different approaches to defining critical areas including VSA approach | Consider: erosion rate, distance to nearest watercourse, onsite evaluation, present conservation status. |
| | Soil type | Not enough studies to derive guidance | 1. Permeable, un-compacted soils<br>NB Shallow soils most likely to need protection. On the most erodible soils combine VFS with in-field measures. |
| | Slope angle | General agreement the performance declines with increasing slope gradient but differences in recommendations | Average maximum recommendation of 7.5° |
| Cost | Economic cost | Very few studies of relative costs for different designs & locations | Dependent on soil type, land opportunity costs, tillage practices, field size, VFS size and relative size of field to buffer |
| Management | Time | Effects uncertain depending on ability of vegetation to grow through sedimentation.<br>Lack of long term >4 year performance monitoring | Regular mowing to remove sediment build-up from VFS |
| | Runoff characteristics | Studies confer that VFS are most efficient when flow is slow, shallow and diffuse | 1. Minimise concentrated flow<br>2. Entry runoff velocity < 0.45 m/sec |
| | Other | | 1. Minimise total sediment load to VFS<br>2. Avoid trafficking on VFS<br>3. Farm along contours |

**Table 2.5 Summary of design and placement guidance derived from published literature.**

## *2.12 Justification for research aims*

Five research hypotheses were identified at the beginning of the chapter. Table 2.6 relates these hypotheses back to specific sections of the literature review and explains why each one is a research priority.

| *Relevant section of literature review* | *Research hypothesis* | *Why a research priority?* |
|---|---|---|
| 2.12.3 | 1. Stem diameter has a significant influence on the trapping efficiency of VFSs. | General lack of vegetation parameterisation. Parameterisation is likely to maximise understanding and enable more accurate modelling. Stem characteristics are not quantified in the literature. Stem characteristics offer a field measurable property and one which will differ between species. |
| 2.12, 2.13 | 2. A simple soil erosion prediction model can be used to predict the sediment trapping efficiency of simple laboratory simulated VFSs. | Given the lack of long term monitoring and experimental evidence, the ability to model VFS and validate with simple laboratory conditions will enable scenario analysis for optimum design and placement combinations without the variability of field experiments and monitoring. |
| 2.12.3, 2.17 | 3. The model can be used to simulate the influence of different plant parameters on the sediment trapping efficiency of VFSs. | Better representation of vegetative parameters is likely to a) enhance understanding of VFS design and b) enable key parameters to be identified and used in species selection. |
| 2.17 | 4. The model can be used to predict the sediment trapping efficiency of established VFSs in the field. | There is a lack of simple models for VFS performance, in particular with the capacity to deal with specific vegetative parameters and different sediment particle sizes. Modelling is important given the lack of long term modelling. |
| 2.15 | 5. The model can be used to derive the optimum design and location for VFSs in the field. | There is a lack of tools for guiding VFS design and placement in the field, particularly in UK conditions. This is important given that VFSs are promoted under Agri-environment schemes. |

**Table 2.6 Literature review sections, research hypotheses and justification**

## 2.13 Thesis structure

Chapter 3 describes the research methods employed during the study. Chapter 4 presents the results of laboratory experiments conducted to test the vegetation parameters of small scale VFS. Chapter 5 presents the results of the monitoring of established VFSs in the field, using the Parrett catchment as a case study. Chapter 6 introduces the concept of soil erosion modelling, reviews its application to the study of VFS design and performance, and describes the selection process for the model used in the current study. Chapters 7 and 8 provide a sensitivity analysis and testing of the selected model. Chapter 9 describes the validation of the model with the field data. Chapter 10 discusses potential application of the model in VFS design and placement. Chapter 11 presents the main conclusions to the study and relates the work in the thesis to the hypotheses described above.

# Chapter 3 Research design and methodologies

## *3.1 Introduction*

Chapter 3 provides details of the research design and methodologies that were adopted to address the aims and hypotheses identified in Chapter 1. The overall research design is based on a modelling approach. A set of laboratory experiments and a field monitoring campaign were also carried out to provide data for model testing. Methods for these two components are described with the respective results in Chapters 4 (laboratory) and 5 (field).

Overall research designFigure 3.1 illustrates the methods employed in relation to the hypotheses proposed in Chapter 1 and how they are related. As well as informing on gaps in knowledge, the literature review provided information on VFS design, guided the design of the laboratory studies (Chapter 4) and assisted with the selection of field sites and the calibration of model parameters (Chapter 6). The field component (Chapter 5) produced information on the design, placement, condition and efficiency of established VFSs which was used in the selection and development of the model parameters. The model enabled analysis of a greater range of parameters and conditions than could be observed in the field (Chapters 7 to 10). Finally, the information and data generated from all components was used in the development of guidance on VFS design and placement (Chapter 10).

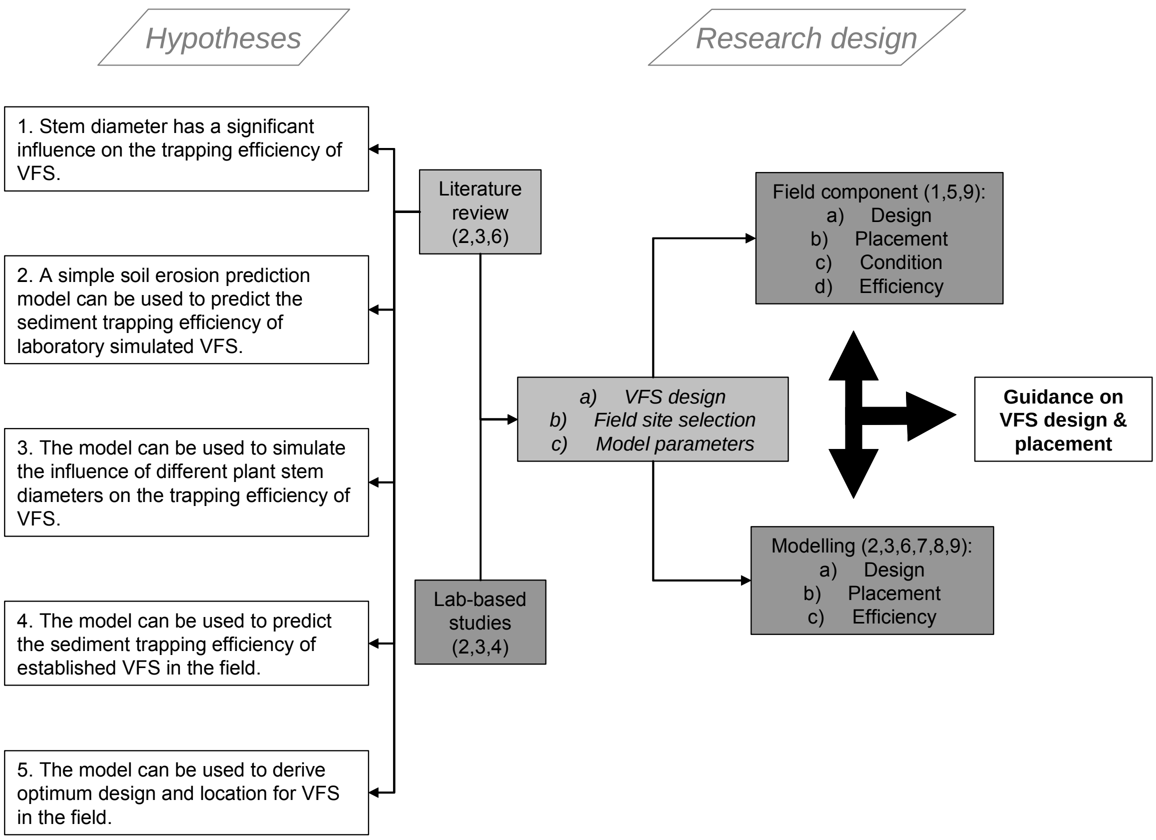



**Figure 3.1 Research design. Numbers in brackets indicate the chapter each step is reported in.**

## *3.2 Modelling research design*

For this component of the study it was first necessary to select a suitable model and then to evaluate its performance and ability to meet the objectives of the study. Once this was satisfied, the model was applied to designing tools to support decisions on VFS design and placement. The model selection and evaluation procedures are described in this section. Actual model selection and application of the evaluation procedures are described in Chapters 6 to 9.

### 3.2.1 Model selection

"An understanding of soil erosion is closely related to the development of physically based equations forecasting sediment transport" (Kirkby et al., 2003). Hence models have been developed to describe the processes occurring during soil erosion by rainfall or flow. According to Foster (1987), variability of climate requires that at least 10 years of data be collected under the "best of conditions" in order to obtain an accurate measure of the average rate of annual erosion. When time and economic resources for achieving this level of accuracy are limited, models provide an alternative to real world observation and the direct measurement of soil loss from an area (Ogunmokun, 1990). Numerous models have been developed for this purpose, all of which involve the use of one, or a series of, mathematical expressions to describe the relationship between erosion factors and the rate of soil loss. However, mathematical models of particle trapping in grass are very rare, primarily due to a lack of detailed physical studies of the processes that occur in grass during wet weather (Deletic, 2001).

Models reported in the literature were assessed for their suitability for the study based on the following requirements. The list was compiled with the following in mind: ability to address gaps identified in the literature, end users (i.e. appealing to land managers and policy makers), research hypotheses and study resources. The list is presented below but justification for each criterion is discussed further in Chapter 6.

Primary requirements:

a) Physically based.
b) Simple to use i.e. does not require a trained user.
c) Describes soil erosion, runoff and deposition.
d) Accounts for vegetative characteristics.

Secondary requirements:

e) Based on or modified with UK/European data.
f) Describes particle size fractions.
g) Annual/seasonal model.
h) Available or readily attainable data.
i) Enables routing over multiple elements comprising the field, farm and small catchment scale.
j) Can be combined with land cover and management options.
k) Reasonable simulation time requirements.
l) Ideally has been applied to VFS design.
m) Ideally has been applied to VFS placement.
n) Low cost to obtain.

### 3.2.2 Model evaluation

A good model evaluation uses more than one measurement of accuracy to pass judgement on the quality of the model (Laflen J.M. et al., 2004). Pilgrim (1975) suggests that assessment of a model should include the following:

- A rational examination of its structure;
- An evaluation of its parameter values;
- Testing of its accuracy; and
- An estimation of its range of applicability.

Hence model evaluation normally includes model calibration (to estimate parameter values for those with high uncertainty or those which cannot be measured), model verification (to test both the accuracy and the rationality of the model) and sensitivity analysis of the model's parameters (to estimate the range of applicability and to pinpoint important parameters and/or processes) (Ogunmokun, 1990).

The model selected for this study was evaluated by:

a) performing a sensitivity analysis on a number of model parameters to establish which have the greatest control over the model output and to determine whether the model behaves rationally (Chapter 7),
b) verifying the model, using available data from previous studies and available literature, in order to investigate its predictive power (Chapter 8),
c) evaluating the model against field and laboratory data collected within the study (Chapter 9), and,
d) performing simulation runs to test the range of its applicability (Chapter 10).

### 3.2.3 Sensitivity analysis

The objective of sensitivity analysis is to quantify the effects of parameter variations on calculated results. Such analysis can be performed either comprehensively or partially by considering only selected parameters. The methods for sensitivity analysis vary in complexity and each method comes with its own specific advantages and disadvantages. The most common procedure for assessing the effects of parameter variations on a model is to vary selected input parameters, rerun the model, and record the corresponding changes in the results or responses. The model parameters responsible for the largest relative changes in a response are then considered to be the most important for the respective response. "Thus, the scientific goal of sensitivity and uncertainty analysis is not to confirm preconceived notions, such as about the relative importance of specific inputs, but to discover and quantify the most important features of the models under investigation" (Cacuci, 2003).

Sensitivity analysis may be used during several stages of model development. During model calibration, sensitivity analysis allows those parameters and variables, to which the model is sensitive, to be identified and classified. Sensitivity analysis is used in the validation of models to quantify the effect that an error in the value of an input parameter has on the model output. An effort can then be made to measure those parameters more accurately (Quinton, 1994).

### 3.2.4 Testing

Model calibration consists of changing values of model input parameters in an attempt to match field conditions within some acceptable criteria. A full calibration of the selected model was not carried out partly due to the limited data available. There was little data from the laboratory and field components to form both a calibration and a validation dataset. Therefore the results of previous studies and available literature were used in place of a calibration dataset to verify the model performance. This involved running the model with the conditions described for the studies and comparing the measured (field or laboratory) and the simulated (model) results. In most cases, however, the datasets were still limited to a single rainfall event and a single location in the catchment which, according to DeRoo (1996), would preclude a clear assessment of the validity of a model. As such it was difficult to judge whether a poor agreement between the measured and simulated datasets was due to model performance, experimental error or to natural variability. Testing was limited to the range of conditions described by the experiment. It was also observed that various sets of parameters could be adjusted, each providing equally good agreement between the measured and the observed datasets (Beven, 1996). Furthermore, fitting the model output to a single experimental dataset risked making the model specific to a particular set of conditions which is not the purpose of a universally applicable model.

For reasons such as this DeRoo et al. (1996) went so far as to suggest that, from a theoretical viewpoint, calibration of a physically based model should be avoided altogether. He further claimed that if a physically based model is calibrated its physically based nature is compromised or is lost entirely. Beven (1989) agrees that physically based models are “not well suited to calibration of model parameters because many different parameter sets will give equally acceptable fits to observed catchment behaviour”. DeRoo (1996) proposed that calibration prevents the identification of errors in process description and catchment representation and James & Burges (1982) commented that calibration is handicapped by short data series. Yet, in almost every physically based modelling study, calibration has been applied, which DeRoo et al. (1996) concludes is because calibration is probably necessary to obtain a reasonable fit between measured and simulated data, especially if the model and the data set relate to different spatial and temporal scales. Within the current study, obtaining this fit was achieved by verifying the model with a number of datasets (both field and laboratory). This was found to be a satisfactory approach to overcoming some of the problems with small datasets and to gaining information on the behaviour of the model under different scenarios.

### 3.2.5 Evaluation

The model was evaluated by comparing its results with the measured results from both the laboratory experiments and the field sampling and monitoring. The field results consisted of deposition of sediment per mat within the VFS. These values were converted to kg of deposition per $m^2$ and compared with output from the model as sediment deposition. Model output was compared with field data for an annual period, for each sampling period and for the nature of the material deposited.

### 3.2.6 Application

The model was applied to investigating the influence of stem characteristics on VFS performance. This was achieved through performing simulated model runs for a range of conditions and investigating the model output in terms of soil loss.

## *3.3 Developing guidance on design and placement*

The literature review (Chapter 2) revealed a lack of currently available tools for guiding VFS design and placement, for UK conditions. Options were therefore investigated for using the results of this study to support decisions on appropriate VFS practice. One of the options developed was a field-based decision support system. This involved pre-running the model for a number of scenarios, based on parameters measured in the field, and using the results to determine appropriate options for soil and slope combinations. The observations recorded on the evaluation forms were used to provide additional maintenance advice. Also investigated was the option of using the model as a simple "soil loss calculator" run in a spreadsheet as well as incorporating the model into a GIS for catchment planning.

## *3.4 Statistical analysis*

Statistical analysis of all data was achieved using Statistica © version 7.0 (StatSoft Inc., Bedford, UK). The specific analysis used is detailed in the relevant chapters.

# Chapter 4 Laboratory component

## 4.1 Introduction

In this chapter detailed methods and results are presented for the laboratory component. The use of laboratory simulation experiments is potentially beneficial in informing on the fundamental mechanisms operating within a VFS system which, in the field, may be too complex to isolate. In this regard, a suite of laboratory experiments was designed and executed to examine different vegetative parameters in order to assess their significance to overall VFS performance. However, the experimental design met a number of limitations, which restricted the statistical analyses that could be performed with such a small dataset and precluded the use of the results in model evaluation and verification, as originally intended. This chapter summarises the experiments performed, provides detailed recommendations for experimental design, including materials, scale and number of replicates, and presents suggestions for further work on the testing of VFSs at the laboratory scale.

The overall aim of the component was to determine how VFS trapping efficiency is related to plant parameters and to provide data for the testing and evaluation of the modelling component. The thesis hypotheses related to this chapter are:

1. Stem diameter has a significant influence on the trapping efficiency of VFSs.
2. A simple soil erosion prediction model can be used to predict the sediment trapping efficiency of laboratory simulated VFSs.
3. The model can be used to simulate the influence of different plant stem diameters on the sediment trapping efficiency of VFSs.

In order to address these hypotheses the following objectives were derived for the component:

1. To determine the vegetative properties of the VFSs.
2. To identify which of these are important in controlling the filtering efficiency of the VFSs.
3. To establish how trapping efficiency relates to the vegetative properties.
4. To investigate the nature of the trapped material.
5. To investigate when, where and in what form sediment gets deposited.
6. To generate data for use in evaluation of a VFS model.

Three sets of experiments were carried out to test a) the materials and experimental set up; b) the vegetative parameters of grass species and c) the vegetative parameters of shrub species.

## *4.2 Laboratory research design*

Morgan (2005) suggests that whilst field measurements may provide realistic data on soil loss, conditions vary in both time and space, so it is often difficult to determine the main causes of erosion or to understand the processes at work. He recommends that experiments designed to lead to an explanation are best undertaken in the laboratory, where the effects of many factors can be controlled. The use of an experimental flume has become an established procedure for investigating the processes of water erosion in laboratories (e.g. Babaji, 1987; Ligdi and Morgan, 1995; Deletic 1999; Ghadiri et al. 2001; Rose et al. 2003; Jin and Römkens, 2002.) Such a procedure is advantageous to this study because the influence of plant canopy architecture can be isolated from other factors. Laboratory experiments have additional advantages over field plot experiments in that replication is more controllable, the weather and time of year do not need to be considered and water supply is less restricted. Previous studies were consulted for examples of materials used to simulate VFSs in laboratory experiments.

### 4.2.1 A brief history of simulated VFS experiments, from nails to live vegetation

#### 4.2.1.1 Nails

Using metal rods to simulate the effects of contour grass strips, Ligdi and Morgan (1995) observed significant differences in the effectiveness of sediment trapping for different plant densities, slopes and discharges. Rose et al. (2003) used beds of nails of various densities to simulate the hydraulic resistance offered by grass strips and found good agreement with the theories he tested on hydrologic performance. Simulated VFSs were proven to be useful in producing replicable experimental conditions.

Ghadiri et al. (2001) investigated the influence of both natural grass and of porous barrier strips, constructed from beds of nails, on runoff hydrology and sediment transport. This enabled some comparison of the performance of nails with the live grass that it was representing. On gentler slopes the grass strips showed a much higher degree of efficiency than the nail strips of any density in blocking the passage of both water and sediment. However, this high efficiency could not be maintained on slopes steeper than 6%, where the grass collapsed under the flow of water and sediment. The grass stems provided a low surface coverage close to the ground but greater coverage at about 30 to 40 mm above the ground where leaves expanded laterally from the stems. This gave rise to a situation where, as the height of the flow increased, the strip became more efficient in blocking the flow and its sediment load. Additionally, the leaf blades appeared to show some degree of hydrophobicity which added to the strip's efficiency in preventing water and sediment from penetrating the strip. Some protection of the soil appeared to be provided by the grass strip although this may have been partly due to differences in the preparation of the grass and nail strips. The study concluded overall that strips, whether solid (nails) or flexible (grass), behave like impeding barriers to flow but that experimental grass strips behave more like natural VFSs.

#### 4.2.1.2 Wooden dowel

Babaji (1987) used wooden dowel to demonstrate the retardation potential of plant stem properties. This proved a successful method for conducting consistent experiments with a range of stem densities and sizes. Babaji (1987) attributed a high degree of variability to the narrow range of his experiments but was still able to conclude that:

a) Increasing plant stem number leads to a reduction in flow velocity and an increase in flow depth, and
b) At the highest level of density investigated the overall retardation increases with increases in stem size. Thus the influence of stem size is only manifested under high stem population.

#### 4.2.1.3 Polypropylene bristles

Jin and Römkens (2001) simulated VFSs using polypropylene bristles inserted in bunches into a flume. The experiments were effective in investigating the influence of different stem densities, flume slopes, flow rates, sediment concentrations and sediment materials in determining trapping efficiency. The study provides useful relative results between treatments. However, like the nails and dowel, polypropylene bristle is an artificial material and therefore does not represent natural variability in biophysical parameters such as stem height, resilience to bending, diameter and spacing.

#### 4.2.1.4 Artificial turf

Artificial turf was used by Deletic (1999) to simulate sediment transport in grass filter strips. Experiments were conducted for different grass densities, flow rates, sediment inflows and sediment types. It was possible to establish a relationship between particle fall number ("the terminal velocity attained by a particle when it settles under gravity in a fluid" (Lovell and Rose, 1988) and the percentage of particles trapped within the grass strip. However, the study recommended that the relationship should be verified using natural grass in order to study, in particular, the downward infiltration of water and particles as well as the influence of grass blades bending.

#### 4.2.1.5 Live grass

Taddesse and Morgan (1996) and Boubakari and Morgan (1999) used natural grass to examine differences in trapping efficiency between two grass species on a range of slope angles. Both studies concluded that the effectiveness of grass strips is dependent on the grass species used and the second study suggested that this is more important than slope angle in controlling soil loss. There were differences in performance between the species tested which were attributed to variation in stem rigidity and in leaf, stem and root system density.

Examining the effect of vegetation height on sediment filtration, Pearce et al. (1997) concluded that using natural grass in laboratory experiments is an effective method for investigating VFS performance. The study concurred, however, with Magette et al. (1987) that the effectiveness of VFSs is highly variable due to the formation of concentrated flow, which may cause the VFS to fail.

### 4.2.1.6 Summary

The studies discussed provide confirmation that simulated laboratory scale experiments can inform on aspects of VFS performance. It should be noted however, that no examples were found of laboratory studies that have been validated using established field VFSs and no studies were found that examine the implications for directly scaling up laboratory experiments to the field, farm or catchment scale. However, laboratory scale experiments do provide an environment relatively isolated from the variability of field conditions; one where specific relationships between variables can be examined and replicable conditions created. Different materials can be effective in simulating VFS performance, depending on the objectives of the study. Whilst natural grass provides more information on the biophysical parameters influencing VFS behaviour, the variability of growing patterns makes tightly controlled replication almost impossible. Replication may be easier to obtain with artificial materials.

In order to get the most comprehensive results, and to diminish the disadvantage of variability, whilst balancing cost, time and practicality, a series of experiments was carried out as follows.

a) Experiments using **Astroturf** strips were first performed in order to develop a protocol for further experimentation. This included testing whether the rainfall intensity and duration, slope angle and soil type used were appropriate for generating enough runoff and sediment for analysis. Height and density were tested because they are easy to manipulate and to observe, providing a good indication of the usability of Astroturf. The Astroturf was also tested by comparing its performance with **live turf** so that it could be assessed for its suitability as a proxy for using live vegetation.
b) In a second set of experiments VFSs were created by growing **grass seeds**. These experiments were designed to test different aspects of buffer management likely to have an effect on stem diameter and density.
c) A final set of experiments was carried out using a combination of **nails** and **wooden dowel**. In these experiments the variability introduced by growing natural vegetation was removed so that variables and relationships between them could be tested in isolation.

## 4.2.2 Experimental layout

The experiments were all conducted using the same equipment and layout (Figure 4.1), which is discussed in the following sections. The generic method consisted of applying simulated rainfall over a plot of soil, installed at a 5° angle, and measuring runoff and soil loss from the downslope end of the plot, for conditions with and without VFSs. This approach is described here whilst aspects specific to each set of experiments are described with the results in Chapter 4.

### 4.2.2.1 Soil properties

The properties of the soil are summarised in Table 4.1. A sandy loam soil was selected because it is easily erodible and therefore likely to respond to the experiment. It is the soil texture used in the simulated erosion research experiments of Taddesse and Morgan (1996), Rimal (1995) and Boubakari and Morgan (1999).

| | *Soil property* | *Units* | *Value* |
|---|---|---|---|
| pH value & salinity | pH (1:2.5 soil/water extract) | | 6.5 |
| | Electrical conductivity (1:2.5 soil/water extract) | uS/cm | 90 |
| | Electrical conductivity ($CaSO_4$) | uS/cm | 2010 |
| Organic matter & nutrient status | Organic matter | % | 3.2 |
| | Total nitrogen | % | 0.18 |
| | Extractable phosphorus | mg/l | 27 |
| | Extractable potassium | mg/l | 72 |
| | Extractable magnesium | mg/l | 43 |
| Particle size analysis & stones | Clay (<0.002 mm) | % | 7 |
| | Silt (0.063 – 0.002 mm) | % | 18 |
| | Sand (2.00 – 0.063 mm) | % | 75 |
| | Texture class | UK class | SL |
| | Stones (2 – 20 mm) | % by dry weight | 4 |
| | Stones (20 - 50 mm) | % by dry weight | 3 |
| | Stones (>50 mm) | % by dry weight | 0 |

**Table 4.1 Soil properties used in the experiments**

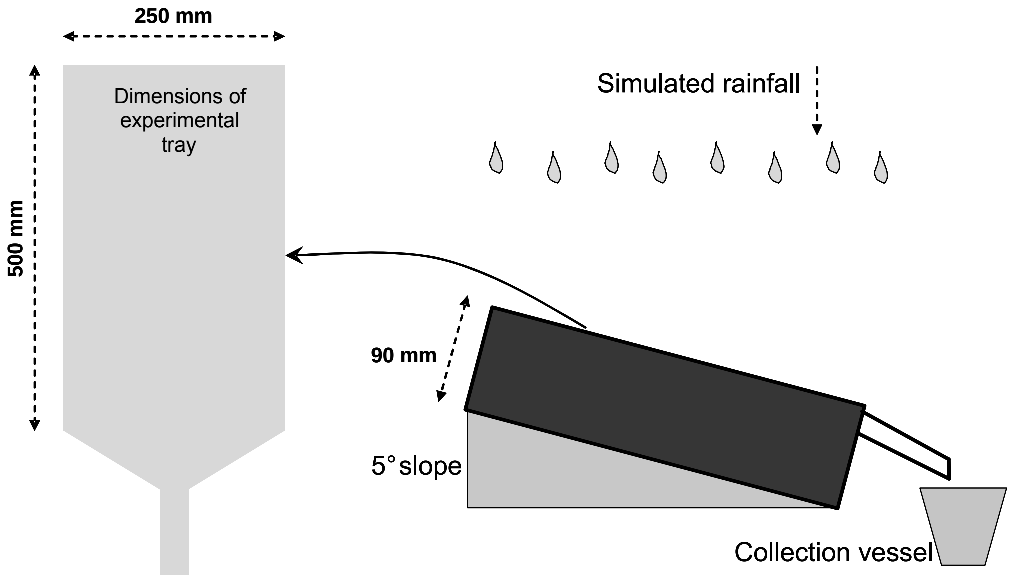



**Figure 4.1 Experimental set-up for the laboratory experiments showing a plan view of the tray on the left and a cross-sectional view on the right.**

## 4.2.2.2 Rainfall simulation

Simulated rainfall provides the opportunity to take many measurements quickly without having to wait for natural rain. The erratic and unpredictable variability of natural rain is eliminated and the operator is able to maintain control of the rainfall characteristics such as intensity, energy and length of storm. However, there are disadvantages. For example, it is very difficult to produce simulated rainfall that is similar in character to natural rainfall, especially at very low intensities and therefore most rainfall simulations for erosion studies use relatively high intensities (Rickson, pers. comm., 2003). A further problem is the relatively small area to which rain can be applied by most rainfall simulators. However, depending on the erosion process that is being studied, this may not be a limitation, for example, detachment requires small plot sizes whilst transport requires a plot length sufficient to generate runoff (Rickson, 2001).

For the experiments presented in this thesis, rainfall was supplied to the soil surface from a 9 m high gravity fed rainfall simulator at Cranfield University, Silsoe. Gravity fed rainfall simulators allow raindrops to fall under gravity, ideally to reach their terminal velocities, so mimicking natural rainfall. The height required for drops to reach their terminal velocity is dependent upon their size, with drops of 5 mm having a terminal velocity of approximately 9 m per second (Hudson, 1995). Epema and Riezebos (1983) present the fall velocity of drops from different heights.

For the simulator used in this study, raindrops are formed by 860 hypodermic needles fixed into a tray (0.86 x 0.46 m) (Figure 4.2 and Figure 4.3). Hypodermic needles are effective small raindrop formers (Miller, 2004). The external diameter of the hypodermic needles is 1.0 mm and the internal diameter is 0.7 mm. The unnaturally uniform drop size distribution produced by the regularly sized needles is overcome by the installation of a metal mesh screen 2.2 m beneath the needles. The screen, of 7 mm aperture, breaks up the drops into random sizes and produces a random spatial distribution of rainfall.

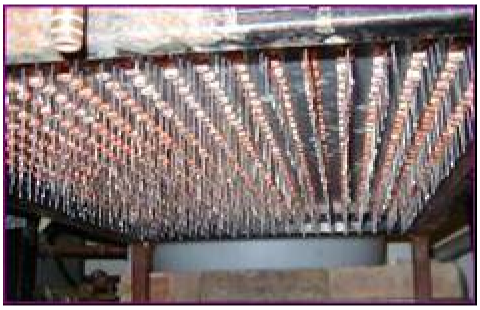


**Figure 4.2 Hypodermic needles fixed into a tray.**

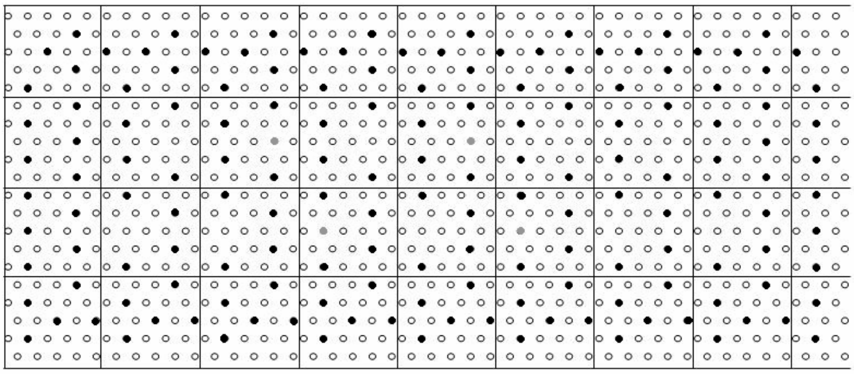


**Figure 4.3 Hypodermic needle pattern for 60 mm h$^{-1}$ rainfall intensity. Positions in black were blocked with rubber bungs and those in white left open.**

In order to allow the rainfall intensity to be adjusted and to maintain constant rainfall intensity throughout an experiment, a weir device is fitted to one side of the needle bed. The weir allows for a constant head of water above the needles, so regulating the rate of drop formation. Once the desired head is reached, the excess water drains away. The rainfall simulator was calibrated in order to achieve the desired rainfall rate for each experiment. To ensure that the water chemistry is consistent throughout the series of experiments deionised water is fed through the simulator. This also prevents the build up of calcium, lime scale and salt.

### 4.2.2.3 Rainfall characteristics

Table 4.2 summarises the characteristics of the rainfall used in the experiments. A rainfall intensity of 60 mm $h^{-1}$ is commonly used in experiments using simulated rainfall e.g. Gabriels and Moldenhauer (1978) and Proffitt et al. (1991). Although this is fairly high compared with natural rainfall, a high intensity is essential to achieve even and reproducible rainfall. Also, the simulator used within this study functions well at this intensity.

Rainfall of this intensity does not occur frequently for long periods in Europe. In Bedfordshire, UK, the return period for 30 minutes of rainfall at an intensity of 63.7 mm $h^{-1}$ is 100 years. The return period for a five minute rainfall event, at an intensity of 60 mm $h^{-1}$, in the same area, is 1.7 years. However, some climate change scenarios suggest that longer duration, more intense rainfall events will in future become increasingly common in Europe (Hardy, 2003).

| *Rainfall characteristic* | *Value* |
|---|---|
| Rainfall intensity (mm $h^{-1}$) | 60 ± 3 |
| Christiansen's coefficient of uniformity (%) | 89.05 |
| Median drop size, $D_{50}$ (mm) (by volume) | 1.03 |
| Drop velocity, based on $D_{50}$ (m $s^{-1}$) | 3.59 |
| Total kinetic energy for 30 minute duration rainfall event (J $m^{-2}$) | 221 |

**Table 4.2 Rainfall characteristics.**

### 4.2.2.4 Trays

Soil was packed into red oxide coated steel trays (0.5 x 0.25 x 0.09 m) designed for use in small scale erosion experiments (Figure 4.4). Erosion processes can be studied at a range of scales depending on the objectives and desired outcomes of the research. For example, at the field scale, rainsplash, overland flow, gullying and mass movement may all be dominant processes, whereas, at smaller scales only rainsplash, overland flow and limited deposition are active processes (Rickson, 2001). The main aim of the experiments undertaken within this study was the comparison of variables in VFS design. It was important therefore that sediment was detached and transported through simulated VFSs. The highly variable nature of VFS behaviour reported in the literature suggests that replication is important. Therefore, the scale selected for experimentation represented a balance between a) suitability for the study of sediment filtering and b) maintaining a soil preparation time which, if lengthy, would compromise the number of replications possible within the time available.

The bases of the trays are made from two layers of fine polyester voile fabric to allow drainage and are fitted with an outlet tube at one end, at the height of the soil surface, to allow for the collection of overland flow. Trays were packed to a density of 1.3 g $cm^{-3}$ and always prepared by the same person to reduce error. The packing density was achieved by placing a 5 kg weight (base area of 240 by 100 mm) over the soil surface after the addition of first 2000 ml and then a further 1000 ml of soil. A smooth surface was maintained by use of a spirit level.


**Figure 4.4 Experimental tray packed with soil prior to simulated rainfall.**

Soil in the trays was wetted by capillary rise for 24 hours and then allowed to drain for a further 24 hours in order to reach field capacity. The trays were tilted to an angle of 5° which was a suitable angle for generating runoff. It is also typical of those slopes observed at the field sites described in the following section and in Chapter 5. Runoff and sediment were collected, for analysis, from the outlet tube at the downslope end of the tray during the experiments.

## *4.3 Experiments*

### 4.3.1 A) Testing materials and experimental set up

#### 4.3.1.1 Introduction

Initial experiments were designed to develop a protocol for further experimentation and to test the potential of Astroturf against live grass turf for simulating VFS behaviour. This was tested by creating the treatments detailed in Table 4.3. The objectives were to:

i. Quantify overland flow discharge and sediment loss from a single soil using laboratory scale rainfall simulation.
ii. Quantify overland flow discharge and sediment loss from soil with VFSs of different vegetative properties.
iii. Make recommendations on the use of vegetative and non-vegetative material for simulating VFSs.

#### 4.3.1.2 Methods

Astroturf was cut to 100 by 250 mm strips. Following packing of the trays with soil the Astroturf strips were placed on the soil surface at the downslope edge (Figure 4.5). Following wetting up, rainfall was applied to the trays at 60 mm $hr^{-1}$ for 45 minutes. A second rainfall event was repeated 24 hours later. All water and sediment draining from the trays, as overland flow and throughflow, was collected over 15 minute intervals. Soil storage was calculated as the difference between the rainfall input, and the output of overland flow and throughflow. Sediment concentration was determined from a representative dried sample of the discharge. As discussed earlier the storm intensity and duration were designed to generate enough runoff for sediment analysis. The values for the height and density treatment were selected to allow as much difference as possible between the treatments. Discharge was collected for six treatments, over three time periods, for two storm events.

| *Number* | *Material* | *Height (mm)* | *Density (stems per $m^2$)* |
|---|---|---|---|
| 1 | Bare soil control | n/a | n/a |
| 2 | Live grass turf | 40 | 40000 |
| 3 | Astroturf | 40 | 2667 |
| 4 | Astroturf | 40 | 853 |
| 5 | Astroturf | 10 | 2667 |
| 6 | Astroturf | 10 | 853 |

**Table 4.3 Treatments used in first set of experiments.**

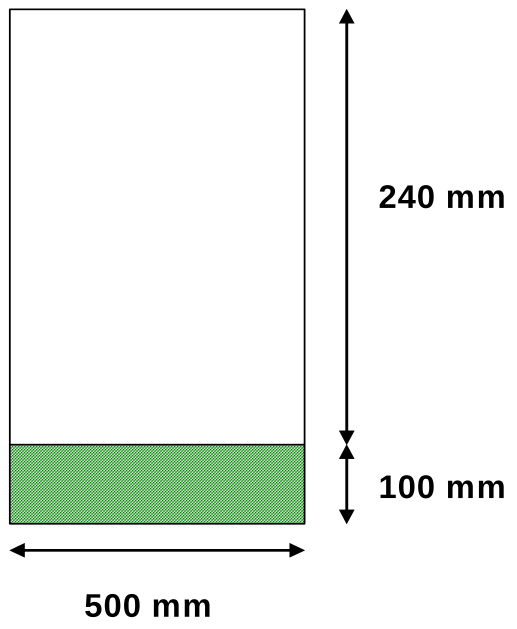



**Figure 4.5 Experimental tray to show dimensions of the bare soil surface (top rectangle) and Astroturf strip (bottom rectangle).**

#### 4.3.1.3 Results and discussion

All results are presented in Table 4.4 and Figure 4.6. It can be seen that, in general, overland flow and sediment concentration were greater with the bare soil (1847 ml and 0.063 g/l) than with any other treatment and that overland flow and sediment concentration were generally greater during Event 2 (1873 ml and 0.09 g/l) than Event 1. A student's t-test was used to test for any statistically significant differences, at a 95% confidence limit ($p<0.05$), between events, time intervals and treatments. In general, the grass treatment produced less overland flow and sediment loss and more throughflow than any other treatment.

*Overland flow*: Grass turf reduced the flow compared with the bare soil control. Astroturf reduced flow but less so than the grass turf. The difference in volume between the control and the buffered plots reduced throughout the time of the experiment. The height and density of Astroturf did not have a significant effect on the volume of overland flow. Between Events 1 and 2, only the short dense treatment had a significantly different overland flow rate and the grass turf treatment had a significantly different sediment concentration. For the short dense treatment overland flow increased, and for the grass treatment sediment loss increased, in Event 2 compared with Event 1. Overland flow from the grass treatment was significantly less than from the control and the short dense treatment (Event 1 only). No significant differences were observed between treatments for Event 2.

The reduction in overland flow is likely to be due to the friction imparted on the flow by the VFSs. This will reduce the velocity of the flow and encourage infiltration within and in front of the VFS. The Astroturf had less of an influence on infiltration due to the less permeable nature of its base. Overland flow did not exceed the top of the VFSs so it is not surprising that height did not influence overland flow. The difference in density between experiments was small which may explain why this parameter appeared to have very little influence on the results. There was no statistical difference in overland flow with vegetation height and density or with time interval and event.

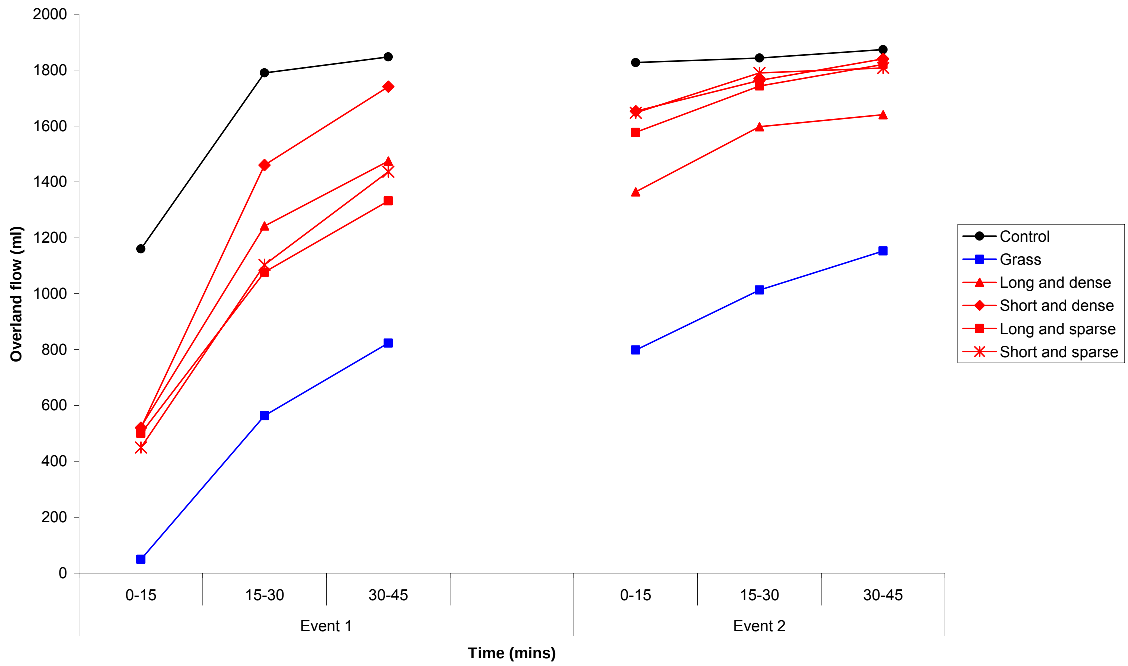
*Throughflow*: The throughflow showed the opposite trend to overland flow, decreasing through time as overland flow increased. The grass turf produced the highest through-flow over the duration of the experiments. These patterns are both likely to be caused by the increased wetness of the soil over time and due to the root network of grass turf respectively. There was a significant difference in through-flow with time interval and with event. Throughflow from the grass turf treatment was significantly greater than from the control, short dense and short sparse treatments (Event 1 only). Throughflow was significantly greater for the long dense treatment than for the short dense, long sparse and short sparse treatments (Event 2 only).

*Sediment concentration*: An initial flush of loosely bound sediment from the bare control plot was not evident for any of the buffered plots. Sediment concentration from the buffered plots remained fairly constant throughout the experiments. The grass VFS produced the lowest sediment concentration but the effect of height and density of the VFSs was not obvious. The short and less dense VFS breached at the final measurement leading to a sudden high concentration of sediment in overland flow. There was no significant difference in sediment concentration with time interval or event. Sediment concentration from the grass treatment was significantly less than from the long sparse and short sparse treatments (Event 1 only).

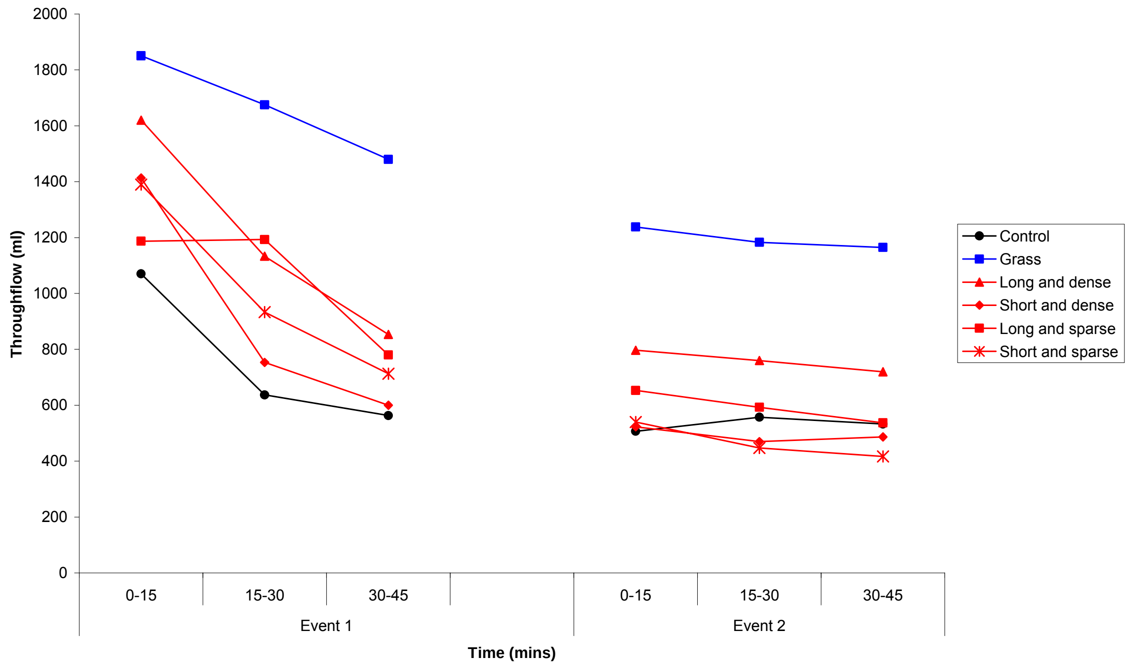
All of the VFSs were effective in trapping sediment. This is apparent from the low sediment concentrations compared with the control plot (Table 4.4) and can be seen in the photographs in Figure 4.7. It was observed that ponds formed at the upslope edge of the VFS which encouraged deposition of sediment here. Throughout the rainfall events sediment moved further across the VFS. There was no statistical difference in sediment concentration with vegetation height or density.

| | | *Event 1* | | | | | | *Event 2* | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Overland flow (ml) | | Throughflow (ml) | | Sediment Concentration (g/l) | | Overland flow (ml) | | Throughflow (ml) | | Sediment concentration (g/l) | |
| Treatment | Time interval (mins.) | Mean | Standard error | Mean | Standard error | Mean | Standard error | Mean | Standard error | Mean | Standard error | Mean | Standard error |
| Bare soil (control) | *0 to 15* | 1160 | 310 | 1070 | 270 | 0.135 | 0.080 | 1827 | 184 | 507 | 77 | 0.085 | 0.04 |
| | *15 to 30* | 1790 | 275 | 637 | 193 | 0.082 | 0.020 | 1843 | 150 | 557 | 92 | 0.083 | 0.04 |
| | *30 to 45* | 1847 | 239 | 563 | 121 | 0.063 | 0.010 | 1873 | 188 | 533 | 48 | 0.090 | 0.05 |
| Grass turf | *0 to 15* | 50 | 50 | 1850 | 189 | 0.035 | - | 798 | 272 | 1238 | 213 | 0.023 | 0.01 |
| | *15 to 30* | 563 | 113 | 1675 | 206 | 0.031 | 0.01 | 1013 | 275 | 1183 | 244 | 0.029 | 0.01 |
| | *30 to 45* | 823 | 210 | 1480 | 216 | 0.018 | 0.01 | 1153 | 264 | 1165 | 224 | 0.026 | 0.01 |
| Astroturf: long & dense | *0 to 15* | 521 | 307 | 1620 | 250 | 0.031 | 0.00 | 1364 | 152 | 797 | 44 | 0.030 | 0.01 |
| | *15 to 30* | 1243 | 327 | 1133 | 300 | 0.037 | 0.01 | 1597 | 52 | 760 | 50 | 0.036 | 0.01 |
| | *30 to 45* | 1474 | 192 | 853 | 179 | 0.037 | 0.02 | 1640 | 72 | 720 | 72 | 0.036 | 0.01 |
| Astroturf: short & dense | *0 to 15* | 520 | 171 | 1413 | 312 | 0.049 | 0.01 | 1653 | 70 | 523 | 84 | 0.053 | 0.02 |
| | *15 to 30* | 1460 | 147 | 753 | 170 | 0.055 | 0.02 | 1763 | 91 | 470 | 56 | 0.051 | 0.02 |
| | *30 to 45* | 1740 | 90 | 600 | 121 | 0.065 | 0.02 | 1840 | 64 | 487 | 77 | 0.050 | 0.02 |
| Astroturf: long & sparse | *0 to 15* | 500 | 500 | 1187 | 338 | 0 | - | 1577 | 116 | 653 | 64 | 0.041 | 0.01 |
| | *15 to 30* | 1077 | 500 | 1193 | 387 | 0.050 | 0.01 | 1743 | 86 | 593 | 28 | 0.044 | 0.01 |
| | *30 to 45* | 1332 | 374 | 780 | 240 | 0.074 | 0.02 | 1820 | 72 | 537 | 17 | 0.038 | 0.01 |
| Astroturf: short & sparse | *0 to 15* | 450 | 325 | 1390 | 350 | 0.049 | - | 1647 | 127 | 540 | 110 | 0.025 | 0.01 |
| | *15 to 30* | 1103 | 459 | 933 | 309 | 0.058 | 0.01 | 1790 | 26 | 447 | 66 | 0.032 | 0.01 |
| | *30 to 45* | 1437 | 314 | 713 | 210 | 0.050 | 0.01 | 1807 | 57 | 417 | 58 | 0.097 | 0.07 |

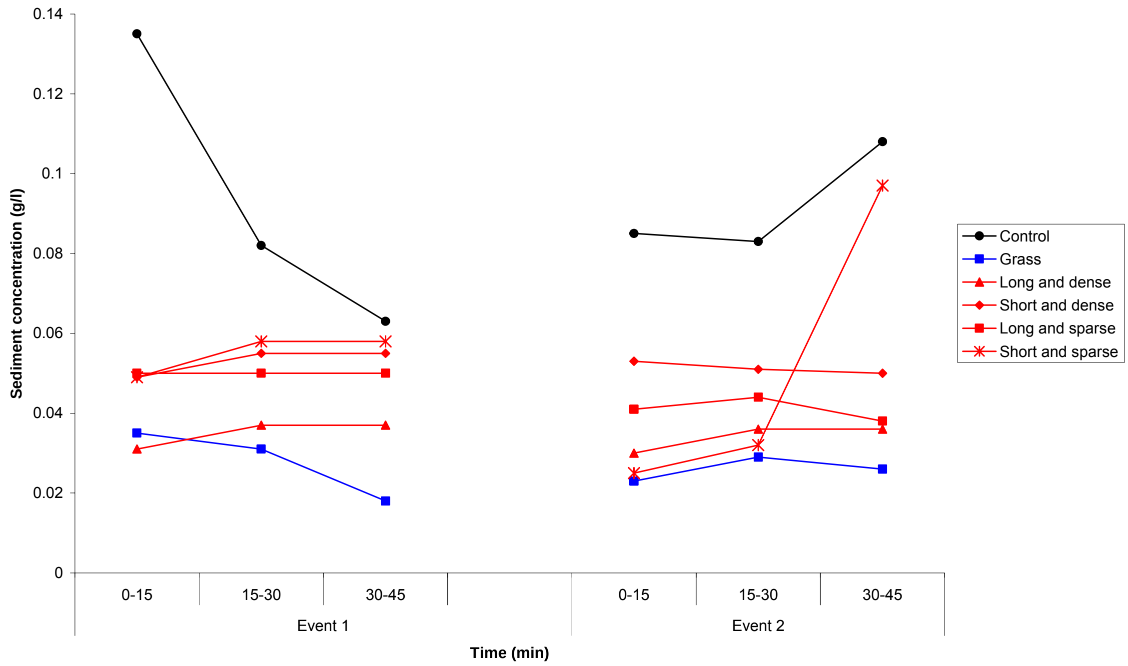
**Table 4.4 All results from lab experiments using Astroturf and grass turf.**




a) Overland flow

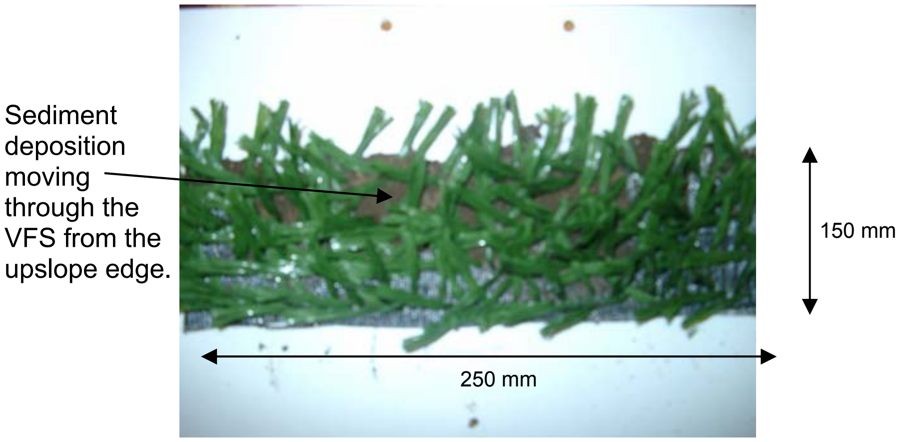



b) Through-flow

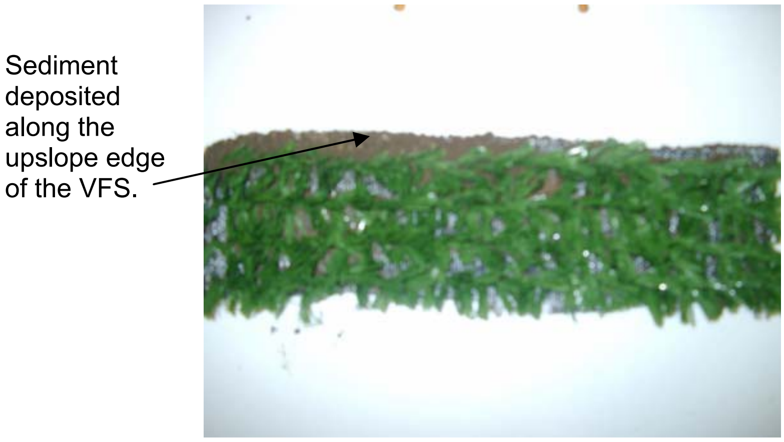



c) Sediment concentration

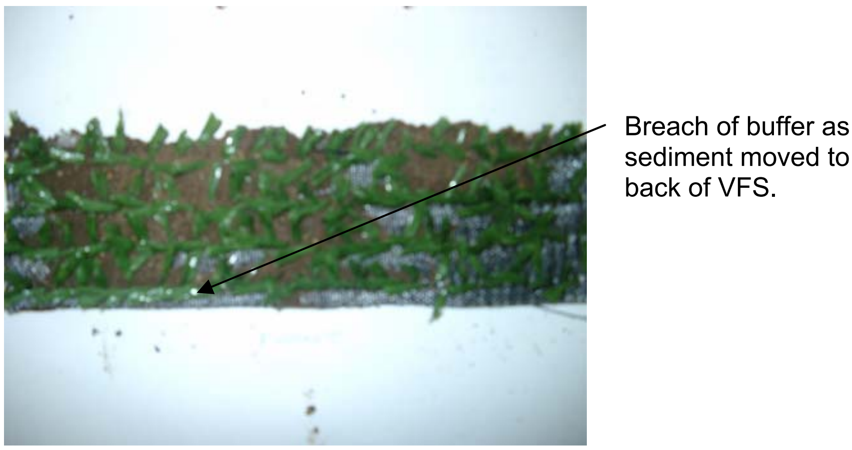
**Figure 4.6 Results of experiments testing the performance of Astroturf and grass turf for simulating VFS performance. (Values plotted are means.)**




a) Long and sparse




b) Short and dense




c) Short and sparse

**Figure 4.7 Plan view of Astroturf VFSs following two 45 minute rainfall events 24 hours apart.**

#### 4.3.1.4 Conclusions

As illustrated by Table 4.4 there was a high degree of variability in the results between the three replicates for each treatment. The effect of the different treatments is therefore difficult to interpret. However, the experiments proved that the experimental set up is effective in allowing the examination of two VFS parameters, density and height, at the selected scale. The time taken for preparation of the trays and wetting up of the soil was not unreasonable. Not enough sediment was generated for particle size analysis and so sediment concentration was the only sediment parameter used to compare the treatments. Higher soil loss may be achieved by increasing the rainfall duration, intensity, slope angle or by changing the soil type but this would result in very unrealistic conditions.

Astroturf was an effective test material due to its availability, low cost and ease of preparation. As shown in Figure 4.6, it is more useful for representing relative differences in treatments than for representing absolute values. Despite its ease of manipulation however, there were relatively few treatments that could be simulated. Infiltration could not be considered because the base of the Astroturf lay on top of the soil surface. It was difficult to place the Astroturf flush with the soil surface without disturbance and during the rainfall events some water was observed to flow between the soil surface and the base of the Astroturf.

The grass turf VFS was more effective in trapping sediment than any of the Astroturf treatments and therefore more likely to show any differences between buffered and bare soil plots. This may be due to the ground cover (i.e. that facing the flow at ground level) created by the different angles of the stems. However, similar problems were faced in placing the turf flush with the soil surface. Obtaining the turf pre-grown also reduces the number of treatments that can be created as well as removing any control over grass species, condition and stage of growth. Overall it was concluded that neither substrate would provide enough potential for varying VFS parameters. Nor would they indicate real life behaviour and variability. Hence live grass seeds were grown for the next set of experiments.

### 4.3.2 B) Testing vegetative parameters (grass)

#### 4.3.2.1 Introduction

Adaptations to the experimental design were incorporated into a further set of experiments and live seeded grass was grown in place of Astroturf and live turf. In order to increase eroded sediment, three rainfall events were carried out with a 48-hour gap between events.

In order to address the thesis hypotheses restated in Section 4.1, the aims of the experiments were a) to form the basis of recommendations on maintenance requirements to optimise VFS performance over time and b) to provide validation of the outputs from the modelling component. The objectives were, following the protocol of the previous experiments, to derive empirically-based relationships between biophysical parameters and buffer effectiveness.

### 4.3.2.2 Methods

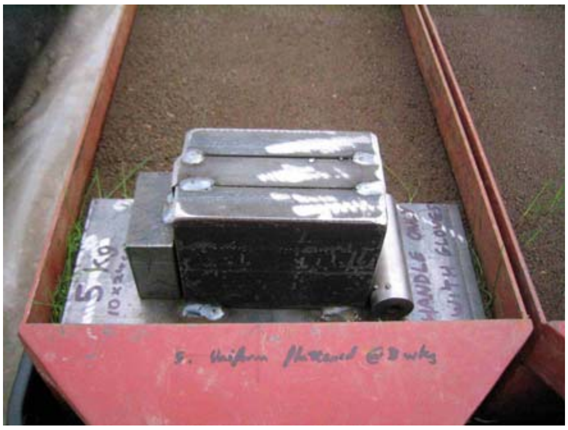
Treatments were designed in order to vary biophysical parameters but also to match the conditions observed in the field (Table 4.5). The trays were packed with soil as previously. Grass seeds were sown as 100 by 250 mm strips at the downslope edge of the trays and the trays left to stand in water in a greenhouse. The grass was a mix of cocksfoot (*Dactylis glomerata*), Timothy (*Phleum pratense*) and creeping red fescue (*Festuca rubra*) sown at 5000 seeds per $m^2$. Species selection was based on recommendations from various suppliers of grass seeds for agri-environment schemes. During the 6 weeks prior to the rainfall events the grass was treated according to Table 4.5. Figure 4.8 illustrates the experimental set up, three of the treatments and the process of grass flattening.

| *Factor* | *Properties* | *Treatments* | *Treatment ID* |
|---|---|---|---|
| Control | No VFS | Bare soil | BARE |
| Management / condition | Growth stage, condition, management practice. | Grass (6 weeks growth) | GRASS 6W |
| | | Grass (3 weeks growth) | GRASS 3W |
| | | Grass (1 year growth) | GRASS 1Y |
| | | Grass flattened at weeks 3, 4, 5 & 6 | FLATTENED |
| | | Grass cut at week 3 with cuttings left in | CUT |
| Structure | Vegetation type. | Woody stems inserted into soil | STEMS |
| | | Tussocks inserted into soil | TUSSOCKS |
| | | Grass plus woody stems | GRASS&STEMS |
| | | Grass plus tussocks | GRASS&TUSSOCKS |

**Table 4.5 Treatments used in live grass experiments.**

a) Trays in the greenhouse.

b) Uniform grass during rainfall event.




c) Flattening the grass prior to rainfall.


d) Grass after flattening at 3, 4, 5 & 6 weeks.

e) Tussocks after 40 minutes of rainfall.

**Figure 4.8 Photographs taken during preparation of the trays and during the rainfall events.**

#### 4.3.2.3 Results and discussion

The experiments did not yield a complete set of results. The number of replicates varied between 0 and 3 and in some cases only runoff, and no sediment, was collected (Table 4.6). Prior to the rainfall event, following transfer of the trays from the site of preparation to the rainfall tower, the soil surface lowered by up to 10 mm (Figure 4.11). This may have been due to wetting and drying of the soil surface, a change in temperature after removal from the greenhouse or to disturbance caused by the lifting and transporting of the trays. The result of this was ponding of runoff at the downslope edge of the tray followed by deposition of the transported sediment before it could leave the tray. Hence, the majority of the experiments produced very little, very diluted or no sediment discharge at all in the collecting vessels. Due to this insufficient replicates for statistical analysis were generated. Therefore only general observations can be made in place of statistical analysis.

The mean runoff and sediment values are presented in Figure 4.9 and Figure 4.10 (totals given where only one result was collected). Error bars are included where results for more than one replicate were collected. All of the VFSs, except the mix of grass and tussocks, reduced runoff compared with the bare soil plot. The increase in runoff with the mix of vegetation is surprising but is only one result and so may be uncharacteristic. The greatest decrease in runoff was achieved by the tussocks, followed by the cut grass, the uniform grass and then the woody stems.

The uniform grass and the cut grass reduced sediment compared with the bare soil plot whilst woody stems appeared to increase the amount of sediment generated. This may be due to loose soil being made available for transport when the stems were inserted. The cut grass treatment was effective because the cut stems filtered and trapped sediment on the soil surface, built up a barrier against the erect stems and may have also protected the soil within the buffer from further erosion.

A further set of experiments were attempted in order to increase the number of replicates. This time the grass seeds were sown in growing trays for later transplanting into a gap left at the downslope edge of each tray. However, baking of the soil surface, browning of the grass, weak and stunted grass swards and uneven growing patterns prevented an adequate amount of uniform grass from being grown in the time available. This may have been due to variable temperatures in the greenhouse, rusting of the tray base or nutrient deficiency.

| | | | | | *Runoff (ml)* | | *Sediment (g)* | |
|---|---|---|---|---|---|---|---|---|
| Treatment | Replicates | D (mm) | NV | Cover (%) | Mean | Std error | Mean | Std error |
| BARE | 2 | n/a | n/a | n/a | 5273 | 131 | 5.3509 | 0.833 |
| GRASS 6W | 3 | 0.9 | 14 | 48 | 3599 | 553 | 4.9577 | 2.30 |
| GRASS 3W | 0 | - | - | - | - | - | - | - |
| GRASS 1Y | 0 | - | - | - | - | - | - | - |
| FLATTENED | 0 | 0.9 | 14 | 55 | - | - | - | - |
| CUT | 2 | 0.9 | 14 | 95 | 3388 | 525 | 1.8350 | 0.8064 |
| STEMS | 1 | 1.6 | 12 | 5 | 5086 | - | 5.9426 | - |
| TUSSOCKS | 1 | 3.2 | 6 | 75 | 3213 | - | - | - |
| GRASS&STEMS | 0 | - | - | - | - | - | - | - |
| GRASS&TUSSOCKS | 1 | - | - | - | 6638 | - | - | - |

**Table 4.6 Vegetative measurements and runoff and sediment results collected from lab experiments using live seeded grass treatments.**

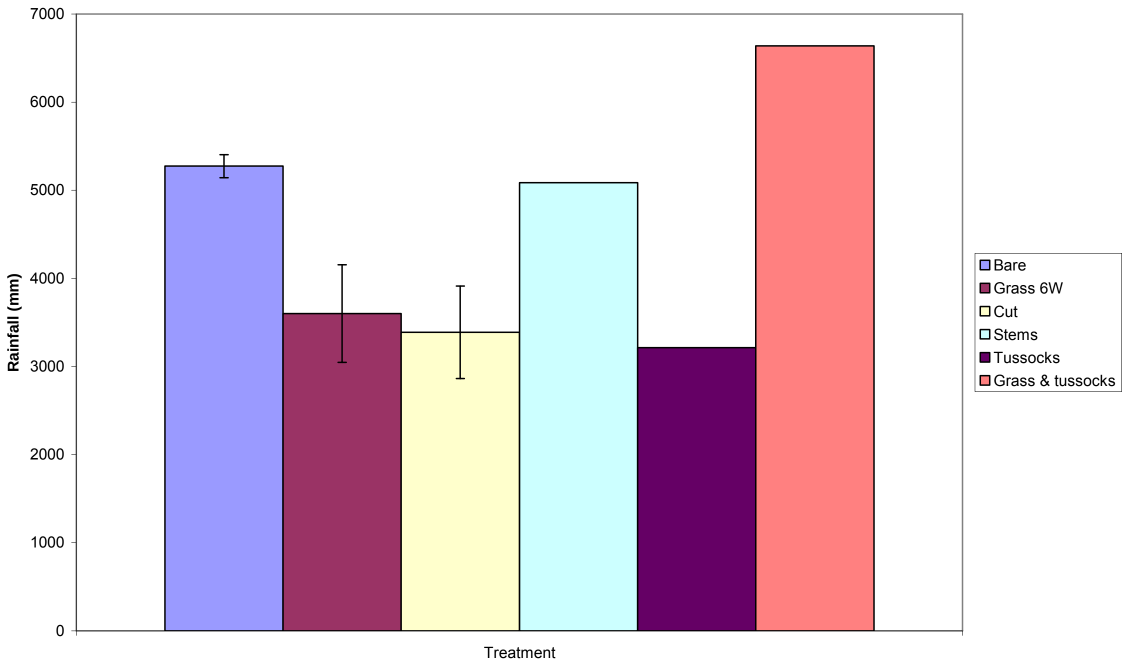



**Figure 4.9 Mean runoff for treatments.**

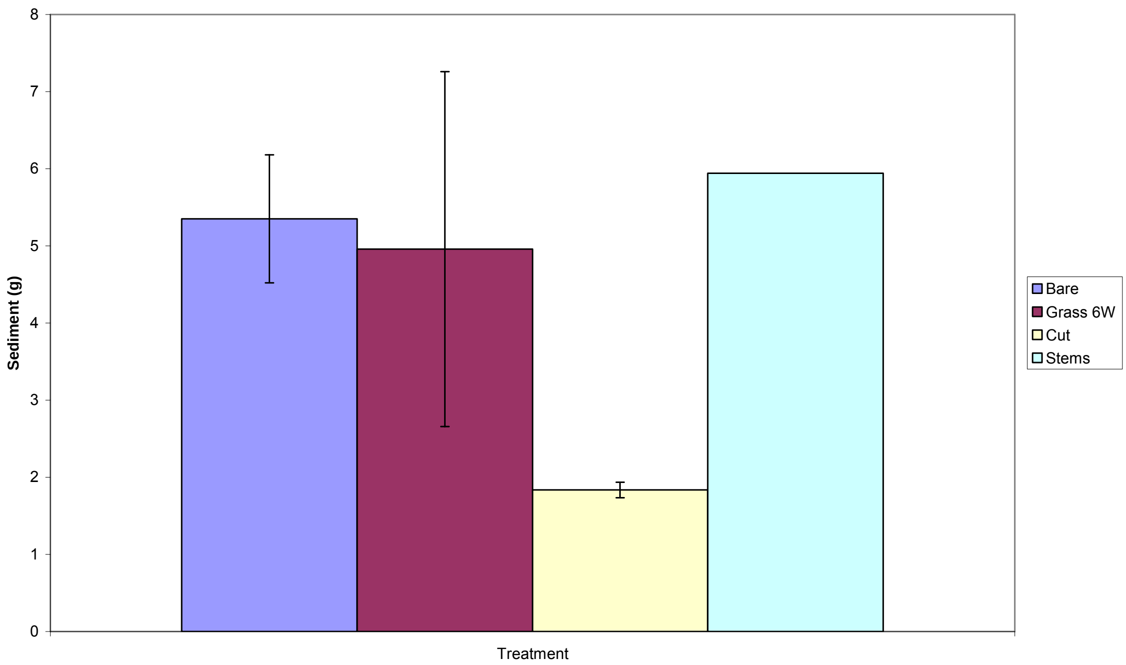
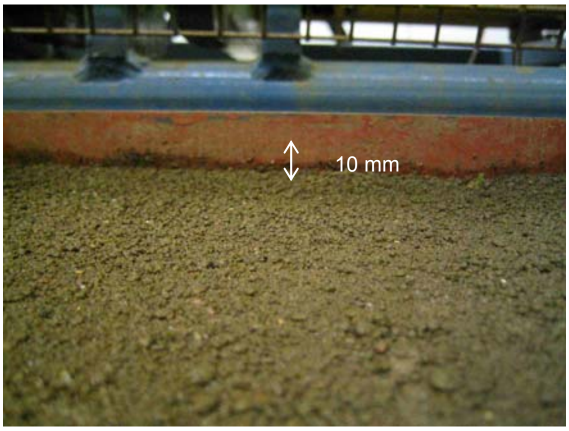
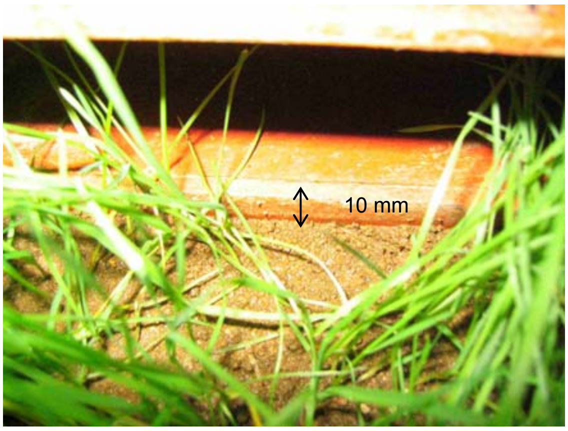


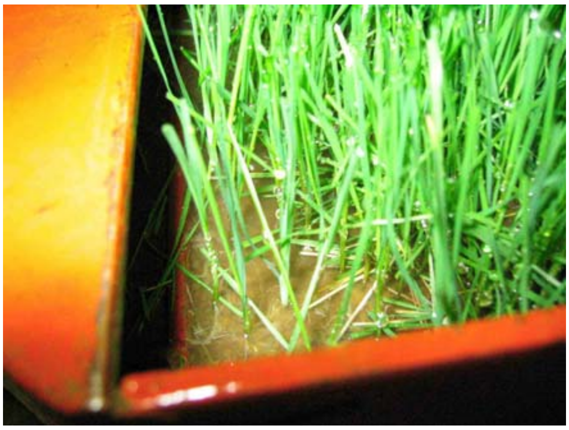

**Figure 4.10 Mean sediment for treatments.**




a) Slumping of bare soil surface.




b) Slumping of soil in VFS.



c) Ponding of water in front of funnel.


d) Accumulation of eroded sediment following ponding.

**Figure 4.11 Process of soil surface sinking and consequent ponding and deposition of sediment in front of the funnel.**

### 4.3.2.4 Conclusions

Experiments were carried out using live grass. Cut grass stems were the most effective treatment in reducing sediment loss compared with a bare soil plot. Cut grass and tussocks were the most effective treatments in reducing runoff. This is likely to be due to the extra resistance to flow provided by the cut stems and the bases of the tussocks i.e. an increased ground cover fraction. Live grass was found to be effective for creating different treatments but produced high variability and was difficult to replicate due to the requirements for consistent growing conditions.

A full set of results for all of the treatments and replicates was not achieved due to the soil surfaces of the trays sinking and preventing sediment from reaching the collection vessels. Therefore data for achieving the aims and testing the hypotheses were not obtained. Little information on biophysical requirements of VFSs was generated and neither were data for model validation.

## 4.3.3 C) Testing vegetative parameters (shrubs)

### 4.3.3.1 Introduction

Given the variability produced by the live grass treatments and the number of replicates feasible in the time available, it was difficult to isolate the effects of stem characteristics. In order to focus on testing stem diameter and plant density a set of experiments was conducted using nails and wooden dowel to represent shrub vegetation. Shrub vegetation and woody stems greater than 1 mm were observed in the field but no results have been published for laboratory experiments simulating shrub-like vegetation. As with the Astroturf nails and dowel had the disadvantage of being a proxy for real vegetation but in this case the advantages of time and control over stem characteristics outweighed the disadvantages. A further advantage over the Astroturf was that water could not flow beneath the turf strip. The protocol of the previous experiments was used without the growing time prior to rainfall events.

Ten treatments were achieved using different combinations of stem diameter and stem number and a range of ground cover fractions (Table 4.3). Each treatment was tested for its effect on total runoff, total throughflow and sediment concentration. Ground cover values were selected in order to get as wide a range as possible. Stem diameter values were selected to reflect those woody stems found in the field and stem numbers were based on the relative number required to give a certain ground cover, given the summed area for all stems used.

| *Treatment* | *Ground cover (%)* | *Stem diameter (mm)* | *Number of stems* |
|---|---|---|---|
| 0 | Bare | 0 | 0 |
| 1 | 75 | 3 | H |
| 2 | 50 | 3 | M |
| 3 | 25 | 3 | L |
| 4 | 75 | 6 | H |
| 5 | 50 | 6 | M |
| 6 | 25 | 6 | L |
| 7 | 75 | 10 | H |
| 8 | 50 | 10 | M |
| 9 | 25 | 10 | L |

**Table 4.7 Treatments used in the experiments.**

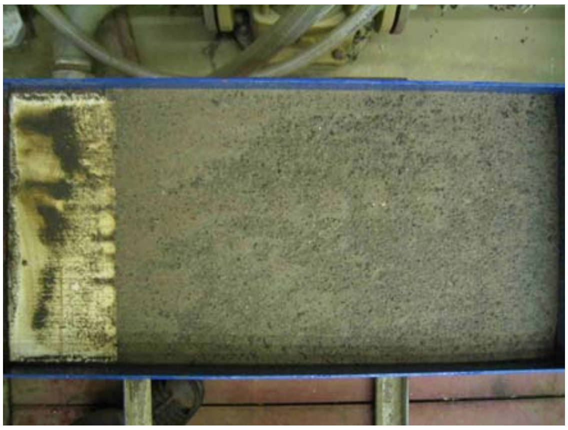
#### 4.3.3.2 Methods

Wooden dowel or nails were inserted into a purpose made wooden block which was inserted at the downslope end of the trays of soil. As not all of the ten treatments were tested at the same time the experiments were set up as an incomplete block design (Figure 4.12) with 30 experiments carried out over 5 days. The treatments are illustrated in ***Figure 4.13***.

| 1 | 2 | 3 | 4 | 5 |
|---|---|---|---|---|
| 4 | 9 | 1 | 5 | 5 |
| 7 | 8 | 9 | 7 | 9 |
| 0 | 2 | 8 | 3 | 7 |
| 6 | 3 | 4 | 0 | 4 |
| 5 | 6 | 3 | 6 | 8 |
| 1 | 0 | 2 | 2 | 1 |

**Figure 4.12 Incomplete block design consisting of 3 replicates of 6 treatments tested over 5 days (i.e. blocks of 5). Numbers across the top row are days and the rest are treatment numbers.**

a) Two trays before rainfall.



b) Control after rainfall.

c) 3 mm 75% cover after rainfall.

d) 6 mm 25% cover after rainfall

e) 10 mm 25% cover after rainfall.

f) 10 mm 50% cover after rainfall.

***Figure 4.13 Photographs of experiments using wooden dowel.***

## *4.4 Results and discussion*

The effects of ground cover, stem diameter and the combined effect of the two variables, on each of the three parameters, were not found to be significant at the 95% confidence level. Plots of the mean values and standard error show a highly variable pattern that is difficult to interpret (Figure 4.14). However, it is possible that:

1. the use of nails and wooden dowel failed to replicate the true behaviour of live stems. The uniformity, smoothness and regularity in spacing of the stems may have prevented sediment from being trapped;
2. the result is real and wider stems such as shrubs do not act as efficient VFSs; or
3. the VFSs failed due to the intensity of the rainfall event which was high considering its duration and the size of the plot. In this case erosion with the VFSs was always similar to that on bare soil.

It is likely that all three factors contributed. The main difference with the woody stems in the field is the lack of leaf litter and debris to build up against the barrier created by the stiff wooden stems. The importance of this is indicated by the effectiveness of the treatment in the set of experiments which used cut grass within the VFS stems. It is also possible that stems are only effective with large fluxes of coarse sediment, as witnessed at one of the field sites, where > 90 mm of sediment had built up against a hedge at the downslope edge of a grass filter strip. This raises questions over whether density, in terms of stem number and diameter, is more important than the roughness of the material between the stems. It is also not clear whether the density of the ground cover facing the flow is more important in VFS performance than the canopy cover.

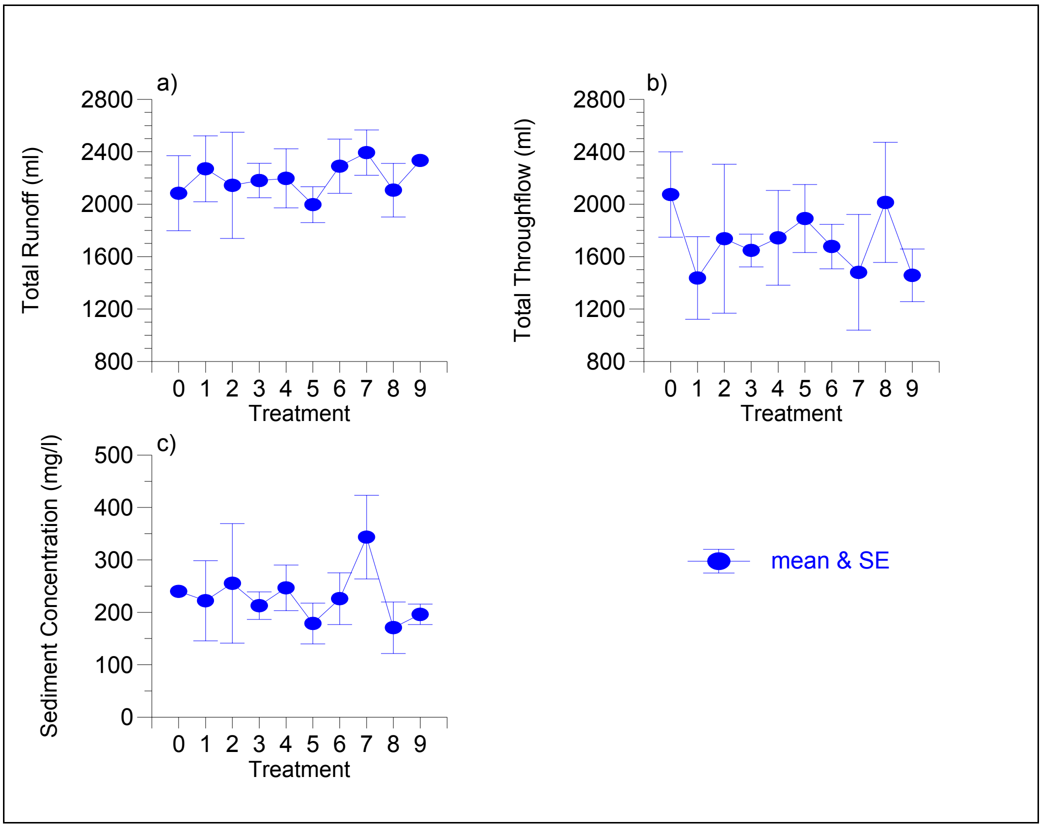



**Figure 4.14 Mean and standard error of the results for three replicates of each of ten treatments.**

## *4.5 Conclusions and further work*

The research hypotheses proposed in Chapter 1, and restated at the start of this chapter, were addressed through a series of objectives set out in Section 4.1. The extent to which each one of these has been met is detailed below.

1. To determine the architectural parameters of the VFSs. Parameters identified from observation of the VFSs are: height, density, branching, uniformity, roughness, stem diameter and stem number. The experiments tested height and density for a number of media. Further experiments examined stem diameter and number. Roughness and uniformity were not tested but may explain why no relationship was measured between the factors tested and VFS performance.

2. To identify which of these are important in controlling the filtering efficiency of the VFSs. In general, the high variability of the replicates, lack of sediment collected and limitations of the media used to create VFSs precluded relationships between biophysical parameters and VFS effectiveness. The second set of experiments indicated that material such as grass cuttings may be beneficial to VFS performance in building up a barrier against flow. The final experiments, which used wooden dowel to vary stem size, showed that, under the conditions tested, stem diameter and number were not

significant in controlling VFS performance. However, it is possible that factors absent from the laboratory set up, such as leaf litter, surface roughness and variability in stem patterns and measurements, may have precluded any relationships that may occur in the field between sediment trapping and stem characteristics.

3. To establish how trapping efficiency relates to the architectural parameters. Observations suggest that where there is a high ground cover, either provided by the diameter of stems, number of stems, tussocky vegetation or debris such as leaf cuttings, sediment movement through a VFS will be restricted. However, this was not proven statistically. In view of observations at the field sites it is probable that shrub vegetation, as simulated by the wooden dowel experiments, is important in providing a structure into which leaf litter and larger debris becomes lodged. The role of such vegetation may therefore be more important in preventing large fluxes of coarse debris from moving downslope than in filtering fine sediment.

4. To investigate the nature of the trapped material. Not enough data was collected for analysis of the trapped material.

5. To investigate when, where and in what form sediment gets deposited. As can be seen in Figure 4.7 and in ***Figure 4.13***, the sediment deposition patterns were variable. This appeared to be mostly controlled by undulations in the soil surface upslope of the VFS and suggests that, at the field scale, terrain in the contributing area will be very important in determining whether flow reaches a VFS in a uniform or concentrated pattern. Where flow was not so concentrated, sediment started building up at the front of the VFS before spreading to the downslope edge over time.

6. To generate data for use in evaluation of a VFS model. Not enough data was collected for use in model evaluation.

Whilst not meeting the proposed objectives in full, this work has provided a valuable insight into the use of various materials for simulating VFS performance under laboratory scales at the tray scale which should be useful in further work.

### 4.5.1 Simulating VFSs under laboratory conditions

Initial experiments involved testing in order to make recommendations on the use of vegetative and non-vegetative material for observing buffer feature performance. A protocol was derived for laboratory scale rainfall simulation experiments. Astroturf, live grass turf, live grass and wooden dowel were tested and revealed various benefits and limitations in terms of practical use and in yielding results. These are summarised in Table 4.8.

| ***Material*** | ***Benefits*** | ***Limitations*** |
|---|---|---|
| Wooden dowel | Easy to obtain.<br>Replicable.<br>Cheap.<br>No growing time.<br>Can be re-used.<br>Easy to isolate variables for testing e.g. height, density. | Difficult to achieve high densities.<br>Inserting directly into the soil can cause disturbance to surface.<br>Limited diameters available.<br>Prevents observation of biophysical parameters e.g. roots, leaves, canopy cover, hydrophobicity, resilience to bending.<br>No natural variability. |
| Artificial turf | No growing time.<br>Structure more similar to natural grass.<br>Easy to collect sediment deposited on the base within the strip for analysis.<br>Reasonably cheap.<br>Reasonably easy to obtain.<br>Can be re-used.<br>Some variables can be altered e.g. trimming to compare height and removing stems to compare density.<br>There is likely to be a limited height range. | Difficult to obtain enough different types to be able to hold some variables constant whilst varying others.<br>Prevents observation of roots, leaves, hydrophobicity.<br>Not possible to study infiltration of water due to plastic base.<br>Difficult to prevent water from flowing beneath, rather than across, the plastic base. |
| Live turf | No growing time.<br>Easy to obtain.<br>Provides natural biophysical parameters and natural variability.<br>Can observe infiltration.<br>Provides canopy cover.<br>Can be cut to different sizes. | Can be difficult to obtain turf consistent in age and condition and grown on the same soil type.<br>No control over seed species, growing density, soil type etc.<br>There will be an interface between the turf and soil surface of the experimental flume/tray.<br>Turf must be replaced for each replicate.<br>Turf must be kept alive throughout the experiments.<br>Natural variability means more replicates will be required.<br>Sward likely to be damaged during rolling and transportation. |

| | | |
|---|---|---|
| Live seeded grass | Have some control over factors such as age, height, density, species.<br>Can observe natural variability.<br>Can observe natural processes e.g. effects of canopy and infiltration.<br>Easier to compare with field results. | Growing time.<br>Consistent growing conditions required.<br>Space for growing required.<br>Can be difficult to transfer grass to experimental tray without damaging sward and breaking up strip.<br>Not easily replicable.<br>High natural variability therefore likely to require many replicates.<br>Grass must be kept alive throughout experiments.<br>Although can control to some extent, can be difficult over time to replicate grass of the same age and therefore uniform in flexibility and density.<br>There will be an interface between the grass strip and the soil in the experimental plot.<br>Difficult to isolate the influence of specific variables e.g. stem diameter, number of stems and resilience to bending may all increase with age of grass. |

**Table 4.8 Summary of the benefits and limitations of different materials used for testing buffer features.**

#### 4.5.1.1 Sediment deposition

The VFSs varied in their ability to reduce sediment loss compared with a bare soil plot (Table 4.9). Not enough material was collected to analyse the particle size distribution of the eroded sediment. It is likely that moving to a larger scale would be the most effective method for generating more sediment loss whilst maintaining the same slope and rainfall characteristics. This would however require longer preparation time. Sediment was observed to deposit initially at the front of the buffer and to move downslope through the VFS during each rainfall event. Given the problems encountered it was not possible, within the time of the project, to perform enough suitable replicates for use in the model evaluation and verification. This therefore required substitution with published results.

| *Experiment set* | *Treatment* | *Trapped sediment (%)* |
|---|---|---|
| A | Live grass turf | 71 |
| A | Astroturf: long and dense | 63 |
| A | Astroturf: short and dense | 44 |
| A | Astroturf: long and sparse | 51 |
| A | Astroturf: short and sparse | 43 |
| B | Grass (6 weeks) | 7 |
| B | Cut grass and cuttings | 66 |
| B | Grass and stems | -11 |

**Table 4.9 Fraction of sediment trapped by VFS compared with bare soil control.**

#### 4.5.1.2 Architectural parameters

The experiments examined vegetation height, density, ground cover fraction and number and diameter of stems. The inherent variability in testing live vegetation made it difficult to draw conclusions on how VFS trapping efficiency is related to plant parameters. However, the experiments using live turf and live grass indicate that the density of material facing the direction of flow, i.e. ground cover, is an important factor in improving the ability of the VFS to filter and trap sediment. The experiments using nails and wooden dowel explored this further but it was not possible to quantify the importance of stem diameter versus number of stems.

#### 4.5.1.3 Further work

An effective protocol would require overcoming 1) high variability between replicates; 2) slumping of soil in trays; and 3) the problem of the interface between the VFS and the upslope soil surface. Increasing the number of replicates may improve the observation of parameters but only if consistent growing conditions could be achieved. It would be advantageous to ensure that the variability between treatments is greater than that between replicates. In general, with experiments of this nature, a suitable testing time should be included in any research strategy, providing adequate time for growing denser, stronger vegetation and for performing trial runs from which to calculate the required number of replicates based on variability measured. Furthermore, given adequate growing time, different species could be compared so that differences, in for example height and density, are inherent rather than created. It would take several months for the different growing patterns of grass species to develop but would allow advice on optimum species to be transferred directly to VFS design.

Ideally, to address issue (2) and (3) experimental VFSs should be grown *in situ* under constant conditions conducive to growth, for example a controllable light source and temperature. Alternatively, the patterns displayed in Figure 4.6 suggest that artificial media could be calibrated to represent the relative patterns of different treatments in place of growing live turf. In order to minimise disturbance to the soil surface, it is important not to move the soil trays prior to rainfall. A further measure to address the soil slumping would be trays with a downslope edge that could be lowered to the level of the soil surface prior to rainfall.

Further work might include adding an overland flow facility to the experimental trays in order to provide better observation of runoff processes. To this flow could be added sediment of different textures in order to compare VFS performance for different particle sizes.

With an effective protocol, further vegetative parameters that could be tested include material behaviour such as bending stress, resilience over flow and roughness. Roughness may be quantified by Manning's $n$ in order to investigate whether this relates to the ability of VFSs to trap sediment. Below ground architecture may also be investigated such as root density, depth, branching and the effect on vegetation stability and subsurface flows. Once criteria are identified which provide the most effective filtering capacity, design guidance can be based on evidence of the relationships between architecture and performance. A further stage would be to match species to those criteria.

# Chapter 5 Field component

## *5.1 Introduction*

Chapter 4 described the laboratory component and this chapter discusses the field monitoring carried out in the Parrett catchment. Very little information exists on the quantities and distribution of sediment and associated pollutants trapped by VFSs in agricultural fields in the UK. Without appropriate field measurements and observations, it is impossible to evaluate the effectiveness of different types of VFS in the landscape or to develop or verify models of VFS performance and behaviour. This chapter describes the field testing of nine, and the monitoring of a further six, VFS features. The aims of the work were:

a) to determine the ability of the VFSs to trap sediment;

b) to identify where within the VFSs sediment is retained; and

c) to establish common causes of VFS behaviour.

Field site information is presented as recorded on an evaluation form (Appendix 2). Results are presented for data collected within the VFSs (Astroturf mats, sediment cores and depth profiles) and contributing fields (sediment cores and depth profiles) followed by a discussion, conclusions and implications.

*Related hypotheses:*

1. Stem diameter has a significant influence on the trapping efficiency of VFS.
2. The model can be used to predict the sediment trapping efficiency of established VFSs in the Parrett catchment.

## *5.2 Field research design*

Many factors cannot be properly simulated in the laboratory and, unless the laboratory facilities are very large, processes such as rill erosion cannot be reproduced either (Morgan, 2005). A field scale study was undertaken in order to build on the process understanding achieved in the laboratory and to provide verification of both the experimentally derived and the modelled data on VFS performance. Numerous field plot experiments investigating VFS performance have been reported by the literature but very few established VFSs have been monitored over time. The present study monitored fifteen established VFSs within the Parrett catchment, UK, over eighteen months in order to progress understanding of the macro-scale operation of VFSs in the landscape and how this might vary over time and space. Details of the study area and methods are presented in the following sections.

### 5.2.1 Study area

The River Parrett basin is located in southwest England (Figure 5.1). The basin occupies an area of 1665 $km^2$ and drains in a northwest direction into the Bristol Channel. It was selected for study because it has been identified as a basin with high sediment production on hillslopes and high sediment delivery to watercourses (McHugh et al., 2003) and, for the purpose of the Defra project (PE0205), because there are concerns over the phosphorus content of the river water (Murdoch and Culling, 2003). In addition, it has a long history of flooding due to low gradient reaches, poor drainage, a long tidal reach and low level land. Land use in the Parrett basin is predominantly grassland, followed by cereals and other crops and then woodland (Godwin and Dresser, 2003).

The average annual rainfall for the region is 800 to 1000 mm with as little as 700 mm in low-lying parts of the basin (Met Office, 2005). Rainfall is greatest in December and January with the driest months being April to July. July and August are the warmest months with mean daily maxima ranging from 19°C on the coast to 21°C inland. January is the coldest month with mean minimum temperatures between 1 and 2°C (Met Office, 2005). The geology of the Parrett catchment is predominantly Oxford Clay with a small band of Upper Greensand and Gault in the headwaters. There are also areas of Old Red Sandstone, Jurassic limestone and marls, and Lower Lias clays in the southern and western sections of the catchment (NERC, 2005).

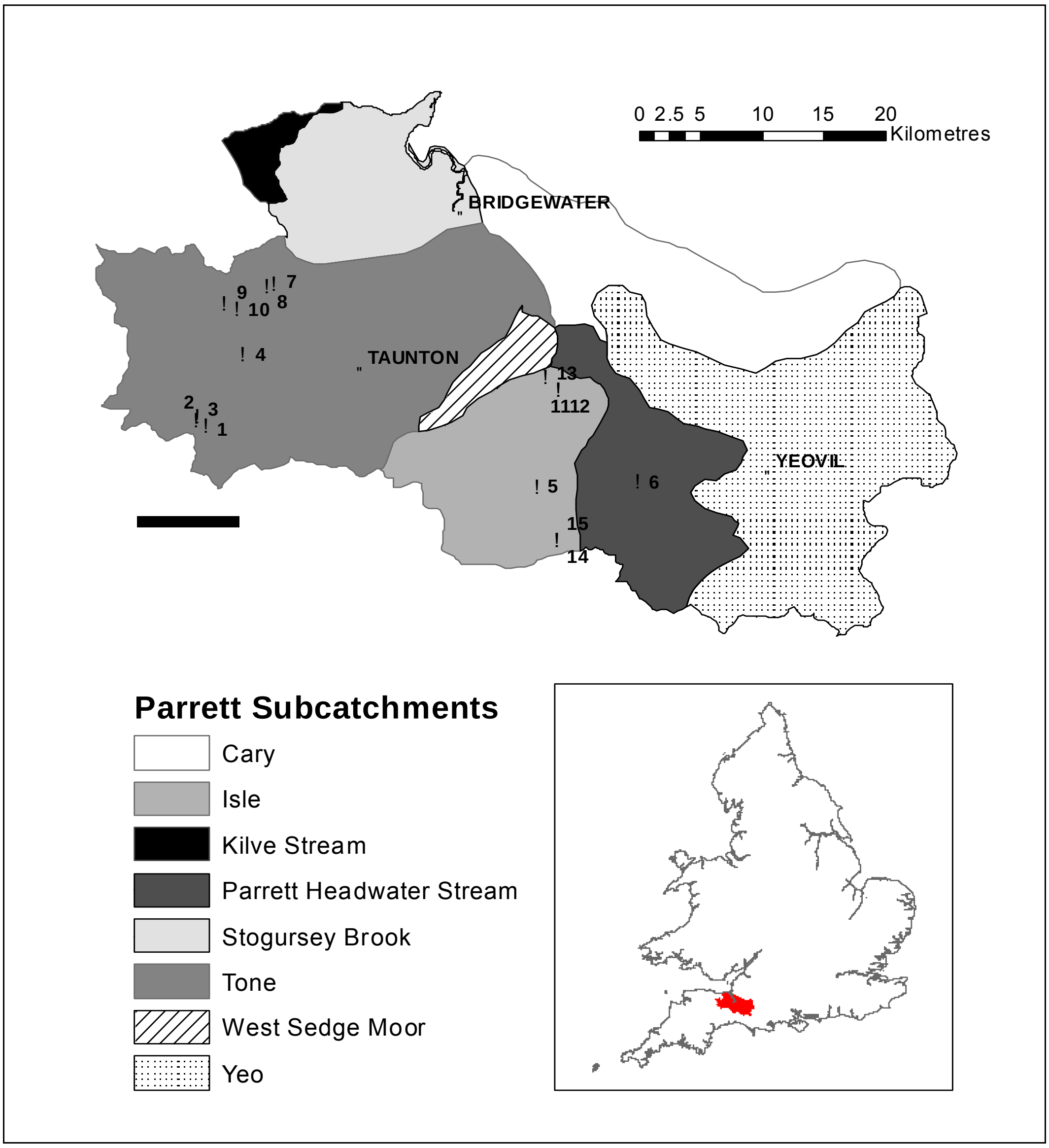



**Figure 5.1 Parrett catchment with subcatchments, town names and numbered field sites.**

### 5.2.2 Soils

Palmer (2002) described the soils of the Tone catchment and the Parrett (which comprises all of the other subcatchments in Figure 5.1). He categorised the catchments into landscape regions (excluding floodplain areas) which are presented in Table 5.1 and Table 5.2. The soil association names were taken from the 1:250,000 scale National Soil Map (NATMAP) (SSLRC, 1983).

| *Soil landscape* | *Main cropping systems* | *Soil association* |
|---|---|---|
| Rolling clayland formed in reddish Triassic and grey Liassic clays | Winter cereals<br>Grass | Worcester (431)<br>Evesham (411a)<br>Hallsworth (712e) |
| Loamy soils in drift over reddish mudstone | Winter cereals – oilseed rape<br>Grass<br>Maize | Whimple (572d&f)<br>Brockhurst (711b) |
| Deep soils over sandstone | Grass<br>Maize | Milford (541a)<br>Neath (541h)<br>Denbigh (541j) |
| Triassic soft siltstone and fine sandstone and river terrace deposits | Winter cereals – stubble<br>Grass | Bromsgrove (541b)<br>Crediton 541e)<br>Hodnet (572c)<br>Newnham (541w) |

**Table 5.1 Soil landscapes in the Tone catchment (Palmer, 2002).**

| *Soil landscape* | *Main cropping systems* | *Soil association* |
|---|---|---|
| Rolling clayland | Winter cereals - Beans<br>Grass | Evesham (411a, b &c)<br>Badsey (511h)<br>Oxpasture (572h)<br>Denchworth (712b)<br>Wickham (711f) |
| Limestone | Winter cereals<br>Grass | Sherborne (343d) |
| Soft siltstone and fine grained sandstone | Winter cereals<br>Top fruit | South Petherton (541m)<br>Curtisden (572i) |

**Table 5.2 Soil landscapes in the Parrett catchment (excluding the Tone) (Palmer, 2002).**

### 5.2.3 Site selection

Figure 5.1 shows the location of the field sites selected. In order to break up the catchment and identify broad areas of interest, subcatchments were targeted, based on reports and maps, before reconnaissance visits to individual sites. Palmer (2002 and 2003) identified degradation risk in the Parrett catchment based on soil structural conditions. Detrimental changes in topsoil and upper subsoil structure can interfere with water infiltration and increase run-off. Such changes are therefore a good indicator of the risk of soil erosion by water. In the Tone, severe and high levels of degradation were limited to the Soft-siltstone and Loamy-over-mudstone landscapes and were generally associated with the management of winter cereals or maize. The Parrett showed an increase in degradation between 2002 and 2003, particularly on the Soft-siltstone. This was mainly associated with winter cereals planted after root crops. Murdoch and Culling (2003) identified the Tone and the Upper Parrett, as well as the Isle, as the subcatchments of the Parrett having the highest phosphorus loads in terms of phosphate, agricultural loads per unit area, and export efficiency.

In order to ensure that sites on a range of soils were included, soil texture was mapped from National Soil Map Data (Figure 5.2 to Figure 5.4). This showed that areas west of the Upper Tone and north west of the Lower Tone consist largely of sandy soils whilst clay soils dominate the south east of the Lower Tone and across the West Sedge Moor, Fivehead and into the Isle catchment. The Upper Parrett consists mainly of loams and silts. The broad texture types (Figure 5.5) were overlaid onto sections of Ordnance Survey maps in order to match soil texture categories with areas on the ground (Figure 5.6).

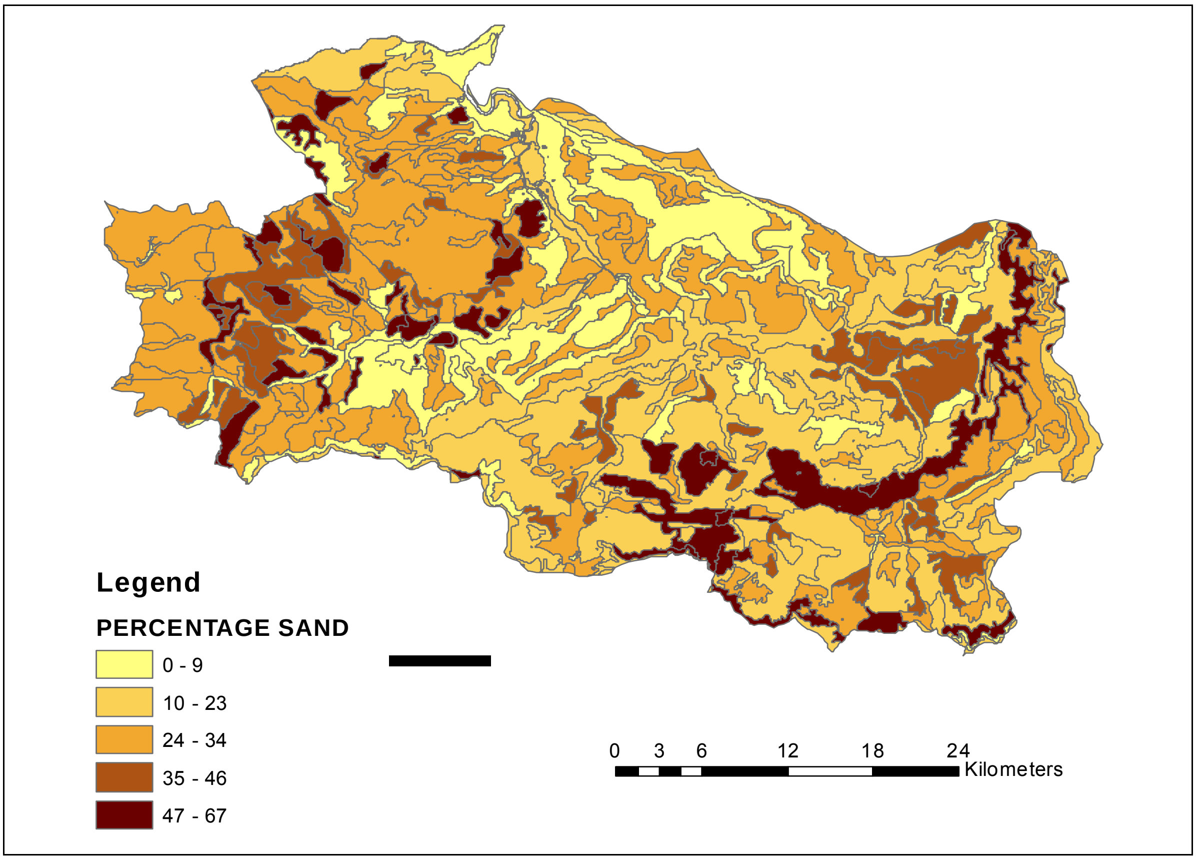



**Figure 5.2 Percentage of sand for soils of the Parrett catchment.**

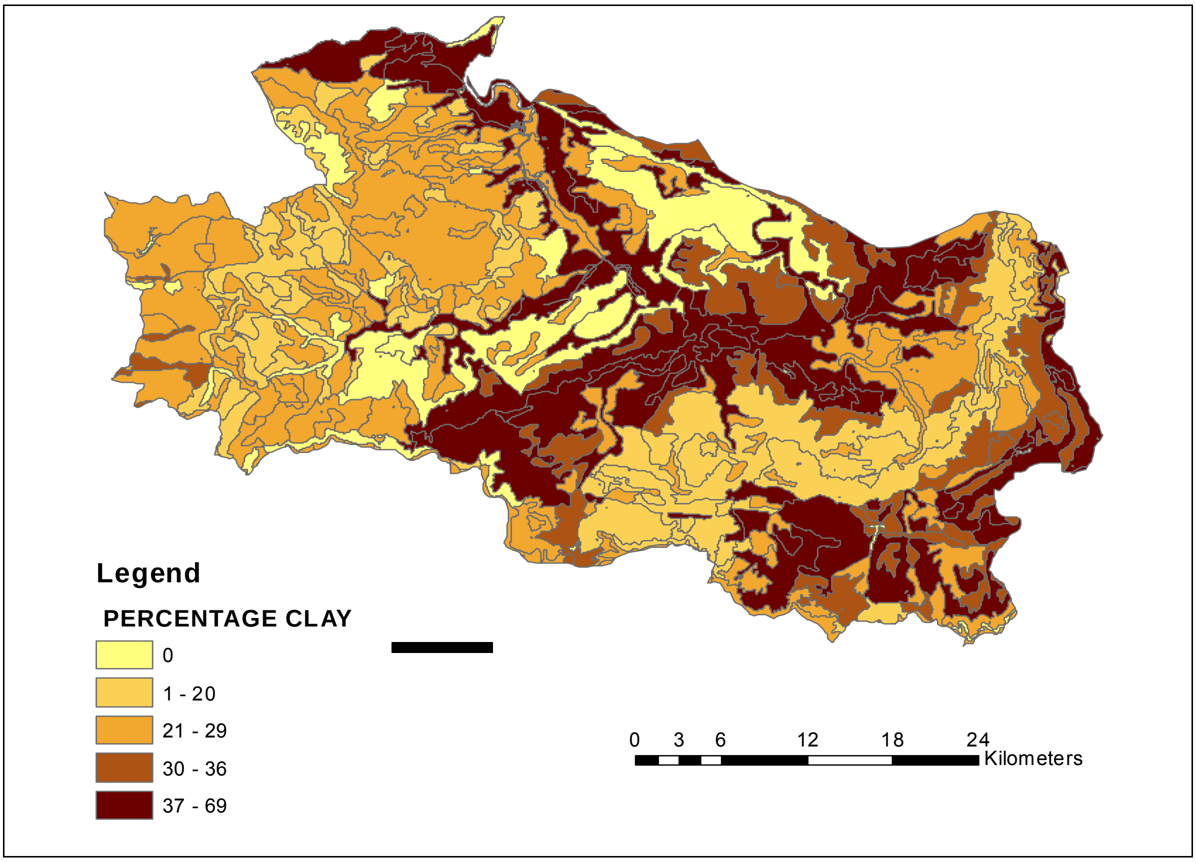



**Figure 5.3 Percentage of clay for soils of the Parrett catchment.**

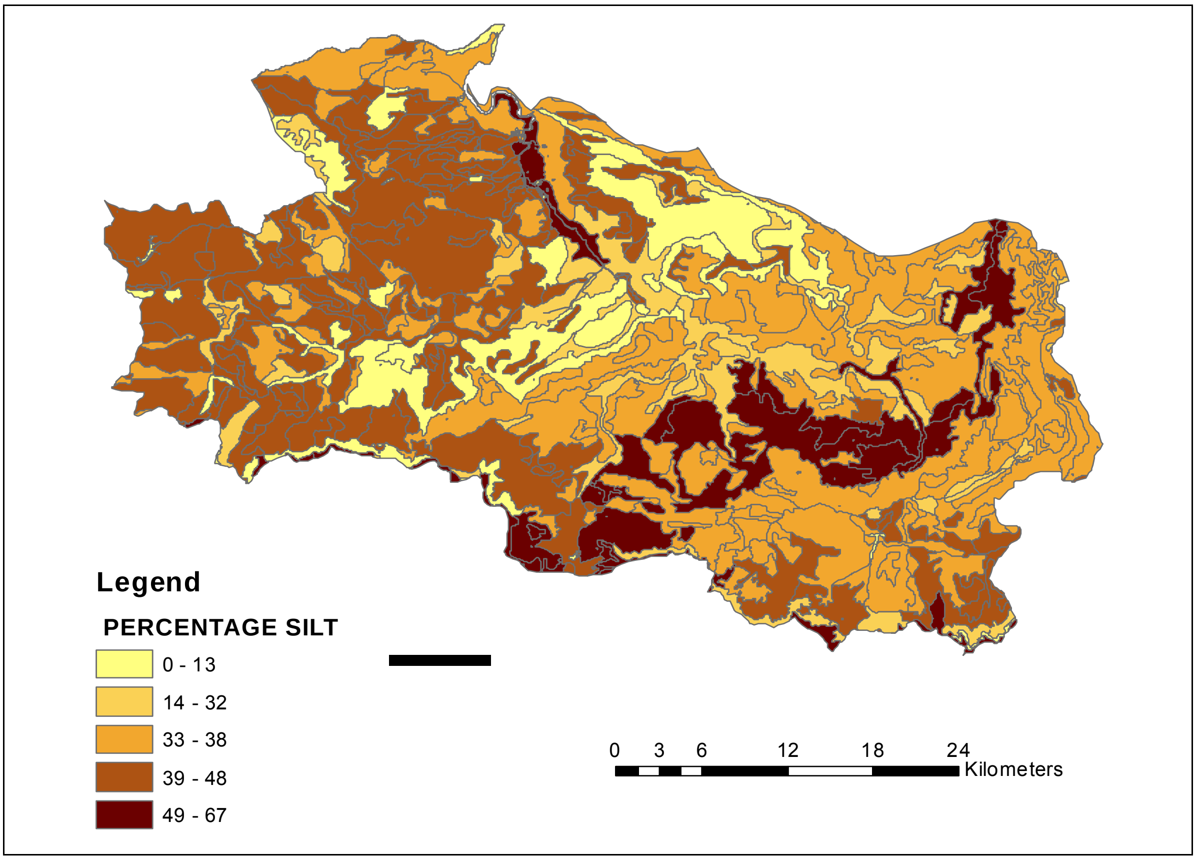



**Figure 5.4 Percentage of silt in soils of the Parrett catchment.**

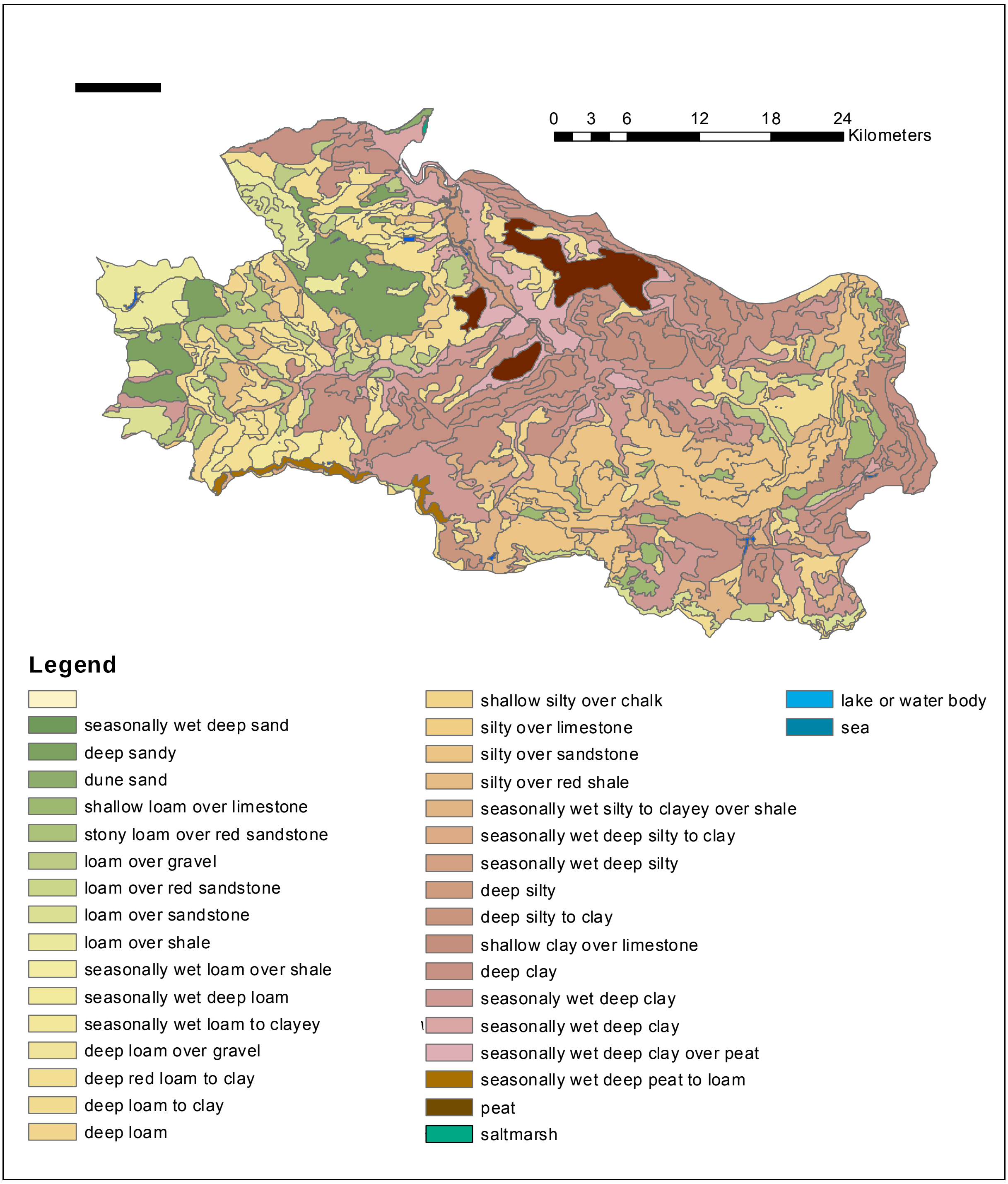



**Figure 5.5 Soil types of the Parrett catchment.**

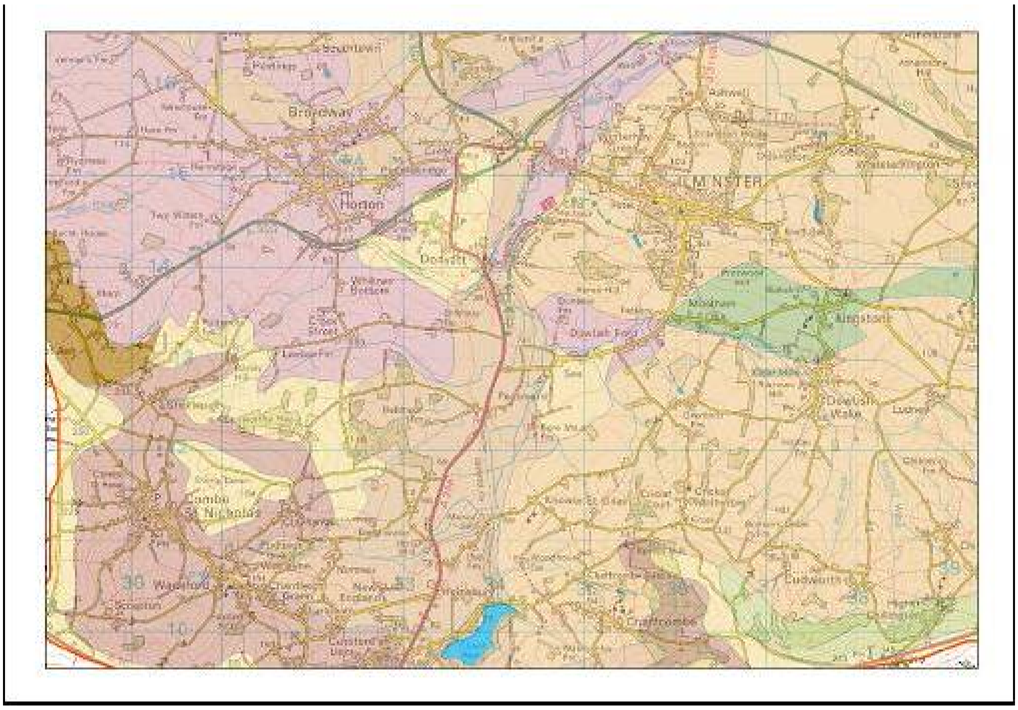


**Figure 5.6 Soil type data overlaid onto Ordnance Survey map (not to scale). For key to soil types see legend on Figure 5.5.**

Identification of soil, erosion risk and slope characteristics of the subcatchments enabled initial targeting. Within the subcatchments the advice of the Somerset Farming and Wildlife Advisory Group (FWAG) was used to find suitable features for monitoring. The following factors were considered in selecting sites.

a) Vegetative features were sought which had been established under an Agri-environment scheme (although not necessarily intended for the purpose of buffering sediment) or which occurred naturally but were known or suspected to perform a buffering function.
b) Accessibility i.e. permission granted by farmer or land owner and safe to access throughout the monitoring period.
c) Feasibility for instrumentation.
d) An attempt was made to select features that varied in land use of the upslope field, including arable crops, grassland and pig farming (although land use varied by rotation throughout the monitoring period).
e) Sites were chosen in order to include features on clayey as well as sandy soils.

Details of each feature are provided in Table 5.3.

| *Site ID* | *Slope (°)* | *Grid reference* | | *VFS feature* | *Scheme* | *Soil type* | *Land use* |
|---|---|---|---|---|---|---|---|
| | | Easting | Northing | | | | |
| 1 | 1 | 310186 | 120028 | Natural floodplain/wetland | Natural feature | Sandy clay loam | Outdoor pigs |
| 2 | 2 | 309449 | 120509 | Bottom of field. 6 m grass stewardship VFS | Countryside Stewardship | Sandy clay loam | Maize then beans |
| 3 | 2 | 309820 | 120770 | Bottom of field. 6 m grass strip (which became part of arable to grass reversion). | Countryside Stewardship | Sandy clay loam | Grass |
| 4 | 5 | 312865 | 126307 | Bottom of field. 7.5 m grass strip upslope of 1 m high bank. | Agri-environment scheme | Sandy loam | Wheat then oil seed rape |
| 5 | 3 | 337106 | 115117 | Mid-field. 3 m grass strip, 1 m hedge and 3 m grass strip. | Agri-environment scheme | Clay loam | Potatoes then wheat |
| 6 | 1 | 345200 | 115525 | Mid-field. 3 m grass strip, 1 m hedge and 3 m grass strip. | Countryside Stewardship | Silty clay loam | Barley then oil seed rape |
| 7 | 5 | 315786 | 131522 | Bottom of field. 9 m grass strip upslope of approx 9 m trees. | Agri-environment scheme | Sandy clay loam | Wheat then peas |
| 8 | | 315164 | 131314 | *Bottom of field. 6 m grass strip.* | Countryside Stewardship | | *Swede then peas* |
| 9 | 4 | 311746 | 129959 | Bottom of field. 6 m grass strip upslope of hedge. | Countryside Stewardship | Sandy clay loam | Maize then wheat |
| 10 | 3 | 312789 | 129496 | Grass riparian area | Natural feature | Sandy clay loam | Wheat |
| 11 | *3* | 338813 | 122938 | *Mid field. 6 m grass strip upslope of hedge and ditch.* | Countryside Stewardship | Sandy clay loam | *Oil seed rape then wheat* |
| 12 | *2* | 338813 | 122938 | *Mid field. 6 m grass strip upslope of hedge and ditch.* | Countryside Stewardship | Sandy clay loam | *Wheat* |
| 13 | *3* | 337767 | 124061 | *Set aside. 24 m grass strip.* | Countryside Stewardship | Sandy clay loam | *Wheat* |
| 14 | *4* | 338662 | 110802 | *Riparian. 6 m grass strip upslope of shrubs.* | Countryside Stewardship | Sandy clay loam | *Wheat* |
| 15 | *2* | 338662 | 110802 | *Set aside. 24 m grass strip.* | Countryside Stewardship | Sandy clay loam | *Oil seed rape then grass* |

**Table 5.3 Feature details for each site. Those in italics were monitored only using an evaluation form.**

### 5.2.4 Characterising VFS performance

The VFSs were investigated using an evaluation form. At nine of the fifteen sites deposited sediment was collected and additional on-site measurements were taken. The number of sites was based on the feasibility of being able to travel to, and sample at, all sites during each field visit and on the number of samples that could reasonably be processed following each visit. The sites were visited at 3 to 4 monthly intervals between December 2004 and May 2006.

#### 5.2.4.1 Evaluation form

Ducros and Joyce (2003) designed an evaluation form to assess whether buffer zones established, under the Water Fringe Option of the Habitat Scheme, to enhance stream habitats and improve water quality, had the potential to meet those objectives. Criteria for assessment focused on vegetation-related attributes as indicators of habitat quality and hydrological attributes that facilitate water quality improvements. Their study indicated that the Buffer Zone Inventory and Evaluation Form (BZIEF) was an effective tool for rapid field based assessment of buffer zones in this context.

Based on this concept a form was designed for the current study with a focus on VFSs for sediment retention. The design emerged through correspondence with the authors of the original BZIEF, selection of suitable criteria from the literature review, as well as field testing of a beta version by a group of staff from the National Soil Resources Institute, Bedfordshire, England. The form attempts to capture information on the VFS, landscape, soil, geomorphology and erosion and deposition features of the site. The form was completed during each field visit in order to capture seasonal variation. A copy of the form is included in Appendix 1.

#### 5.2.4.2 Collection of deposited sediment

In order to understand the sediment trapping processes active within the VFSs, information was required on the amount, rate and type of sediment transported to, trapped within and leaving the VFS. Astroturf mats (Figure 5.7) are an accepted method for assessing the rate of sediment trapped and deposited within river catchments. Lambert and Walling (1987) were the first to publish sedimentation rates from artificial mats, which replicated the natural turf-surface of the floodplain of the River Culm, Devon, UK. Since this first study, numerous investigations have employed similar artificial turf mats to quantify floodplain sedimentation rates and to determine grain size distributions of the deposited sediment (e.g. Walling and Bradley, 1989; Asselman and Middelkoop, 1995; Simm, 1995; Nicholas and Walling, 1996).

Steiger et al. (2003) assessed the use of artificial turf mats for investigating riparian sedimentation. They concluded that artificial turf has a number of benefits as a sediment trap including a rough surface, which retains sediment effectively, and a robust construction from a pliable material which allows the thorough removal of sediment for analysis. It also provides a sediment collection method that is replicable over time and space.

**Figure 5.7 Astroturf mat (300 x 200 mm)**

A minimum of twenty four mats were distributed throughout each site. Where possible at least six of the mats were positioned at the front, six in the middle and six at the back of each VFS. The section of the VFS to be instrumented was chosen based on consideration of the major flow pathways using both digital data and ground reconnaissance. Where possible, mats were placed across two sections of each VFS. One section was selected in order to avoid, and the other to intercept, any major pathways flowing perpendicular to the VFS. This meant that VFS performance could be captured for both concentrated and uniform flow. Figure 5.8 illustrates the mat layout and the processes that the field set up was designed to capture. Selection of the VFS monitoring sections was based mainly on field observation but was assisted by the use of digital data and Arc Hydro tools software.

Arc Hydro is an ArcGIS software that enables the creation and interrogation of hydrological networks of rivers and streams (Maidment, 2002). The software was used in order to generate a flow network for the Parrett catchment. A low stream order was used to define the pathways taken by water and sediment over the landscape. The generated flow network was then overlaid onto aerial photographs of the selected field sites in order to identify locations where flow paths were intercepted by VFSs. This was taken into the field and used to assist in the selection of appropriate locations for the Astroturf mats (Figure 5.9).

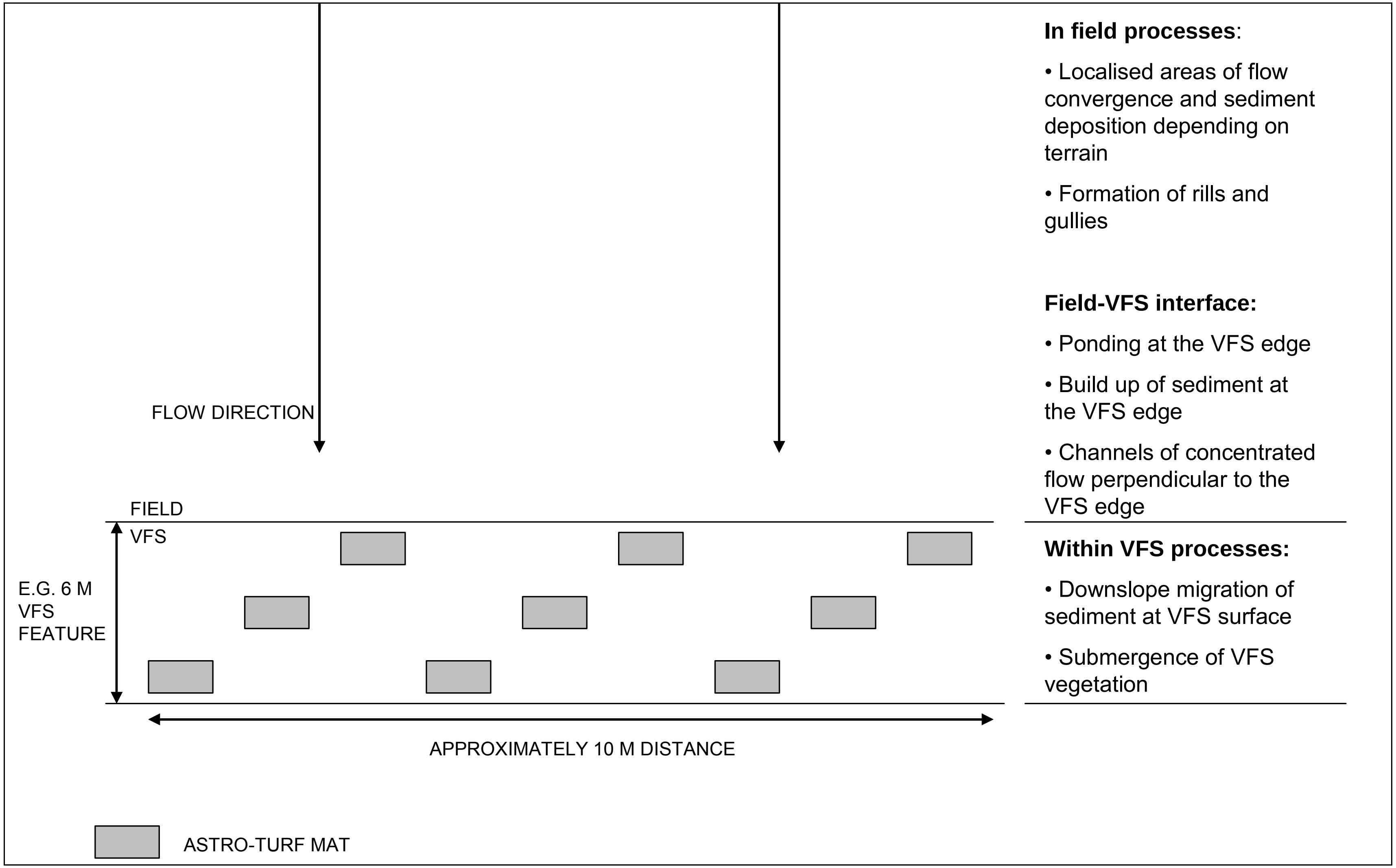



**Figure 5.8 Simplified plan of grass strip illustrating mat layout and processes that the field set up was designed to capture**

**Figure 5.9 Aerial photograph of an example site with lines representing the flowpath network of the subcatchment. X marks the monitoring site.**

### 5.2.4.3 Additional on-site measurements

In order to determine whether the material deposited in the VFSs is representative of that mobilised from the hillslopes, soil samples were collected from the VFS's contributing area. Samples were taken to measure changes in sediment particle size over space (transects) and over time (depth profiles):

*Transects* Cores of topsoil (ca. 0-50 mm depth) were taken along a transect perpendicular to the VFS using a bulk density core (Figure 5.10). Along the transect three cores were taken every 10 m. Where possible each transect included one set of three cores downslope of the VFS, two within the VFS and seven upslope of the VFS (Figure 5.11).

*Depth profiles* Soil samples were collected at 20 mm depth increments down to 200 mm at one location within the VFS and at one location within the field. The purpose of this was to compare the depth profiles of particle size between the field and the VFS.

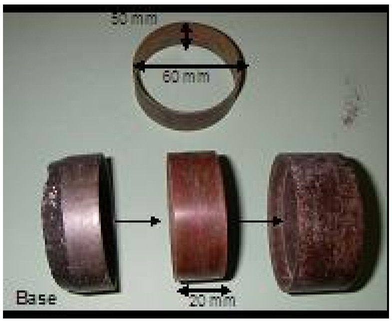



**Figure 5.10 Bulk density ring with 20 mm and 50 mm corers.**

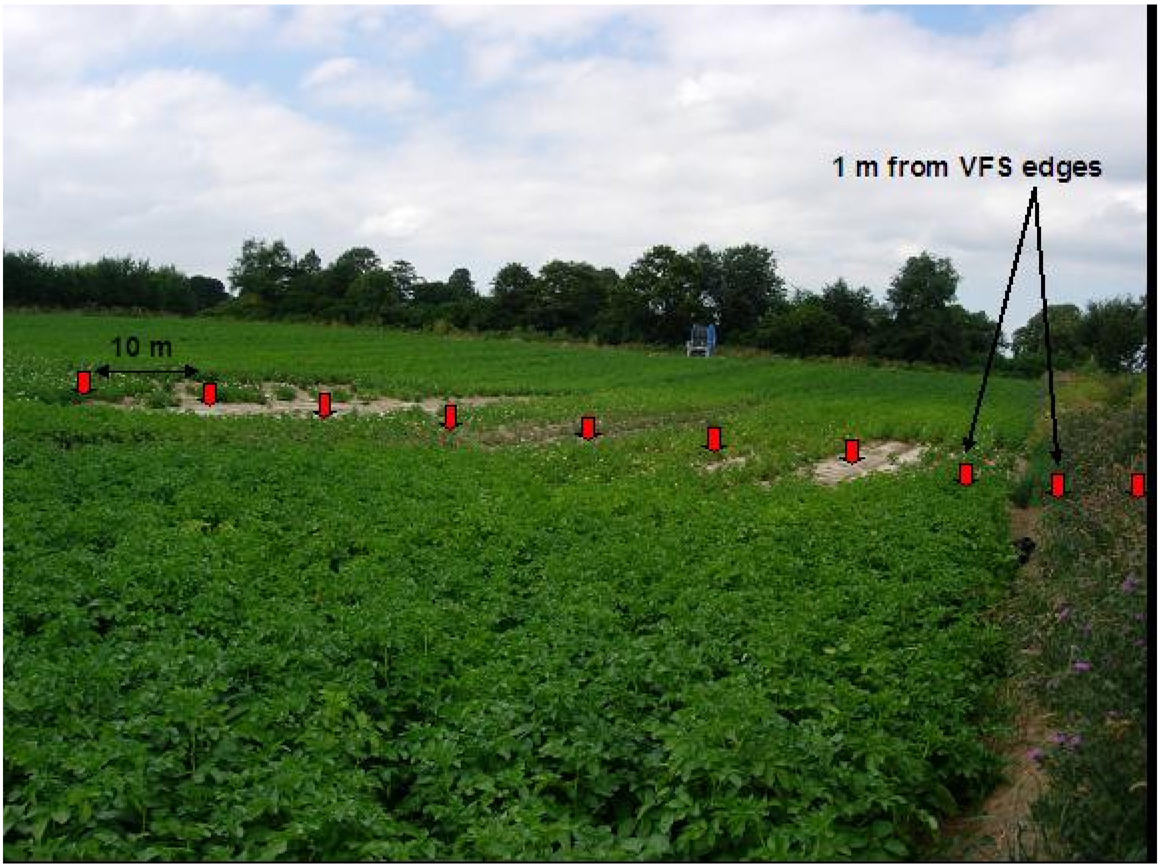



**Figure 5.11 Location of soil samples along a transect through a potato field (left) and VFS (right) – not to scale. (Three samples were taken at each of the arrow marked locations. Two transects were taken in each field.)**

Surface water traps (Figure 5.12) were installed within the VFSs, one at the upslope and one at the downslope edge, in order to collect water samples for analysis of suspended sediment. This instrumentation is designed to channel overland flow into a collecting vessel and was used successfully by McHugh et al. (2004) for analysis of dissolved nutrients. Following installation the trap is cleaned and emptied of any debris at the start of a rainfall event. At a predetermined time interval the runoff collected by the trap is removed for analysis. However, within this study the traps did not yield any samples due to consistently dry weather at the time of the field visits and so no results will be presented for this.

Figure 5.12 Metal (left) and plastic (right) surface water traps. Each design consists of two arms at a 90° angle facing upslope and a collection vessel. Water flows into the upslope edge of the arms and is channelled into a plastic container for collection.

## *5.3 Site details*

The instrumented sites included a range of land uses and soil types. Details on each site were summarised in Table 5.3. Figure 5.13 shows a typical field set up.


**Figure 5.13 Typical field set up (Site 7). Astroturf mats, surface water traps, 120 mm depth profiles and transects of 20 mm sediment cores were used at each site.**

## *5.4 Results*

### 5.4.1 Description of field sites (based on evaluation forms)

Evaluation forms were completed regularly (April 2005, July 2005, October 2005, January 2006, May 2006) by the author and, alongside detailed field notes, provided a record of observations at each field site. Rainfall data was obtained from the Met Office and heavier events were confirmed by local farmers. The following section describes the characteristics of each site and summarizes the main observations made during the sampling period.

#### 5.4.1.1 Site 1

Site 1 consisted of a small grass valley with a braided stream network at its base. The valley sides were used for grazing pigs. This site provided two types of potential buffering feature; patches of rough, shrub vegetation on the valley sides and a floodplain on the valley floor. The floodplain was 15 m wide and consisted of flat, coarse sandy banks broken by stream channels 0.2 to 1 m wide. In the winter there was very little vegetation but in the summer the banks were covered by a 60 to 80% vegetation cover consisting of mainly reeds and watercress. Rills and bare patches on the valley sides were observed on the first field visit. The rills were approximately 200 mm wide and 100 mm deep, spaced 0.6 m apart. This may have been a result of overgrazing by pigs which had recently been moved into an adjacent field. There were no further signs of erosion until the last visit when the pigs had been reintroduced to the field. Following this, channels and gaps in the vegetation became evident as well as signs of ponding and poaching. The farmer plans to establish trees and sediment ponds at this site to reduce further erosion of the grass slopes.

Mats were placed on the banks within the floodplain area as well as on the valley sides above and below the shrub vegetation. During the course of the field monitoring only three of the mats on the floodplain, and none of those on the slopes, showed evidence of sedimentation. It is likely that the mats on the floodplain collected sediment when the stream level rose and sank, suspending and depositing the coarser material.

#### 5.4.1.2 Site 2

The VFS at Site 2 was a 6 m grass strip at the lower edge of a 39000m$^2$ arable field, established under the Countryside Stewardship Scheme in 2001/2. The average length of the field upslope of the VFS was 195 m with a slope angle of approximately 2°. The soil is a sandy clay loam soil. The field was planted with maize followed by beans. The grass height within the VFS varied from 50 to 200 mm in winter to 300 to 500 mm during the summer months. The VFS length was continuous and a ground and canopy cover of 80 to 100% was maintained throughout the sampling period. There was some sign of trafficking but only following access for hedge trimming. Mats were placed at this site within a dip in the VFS.

#### 5.4.1.3 Site 3

The field was 17860 m$^2$ and bordered by a 6 m grass strip. Previously peas had been grown but during the course of the sampling period the field was sown with grass and used for grazing sheep. This is likely to have reduced the amount of sediment leaving the field as the farmer had previously observed sediment washing from the field and onto the adjacent road. Mats were placed in the corner of the field and across the VFS where the sediment had previously washed off the field but no sediment was collected. The VFS vegetation was generally very dense, short grass but bare patches had been formed by overgrazing at some locations (Figure 5.14).

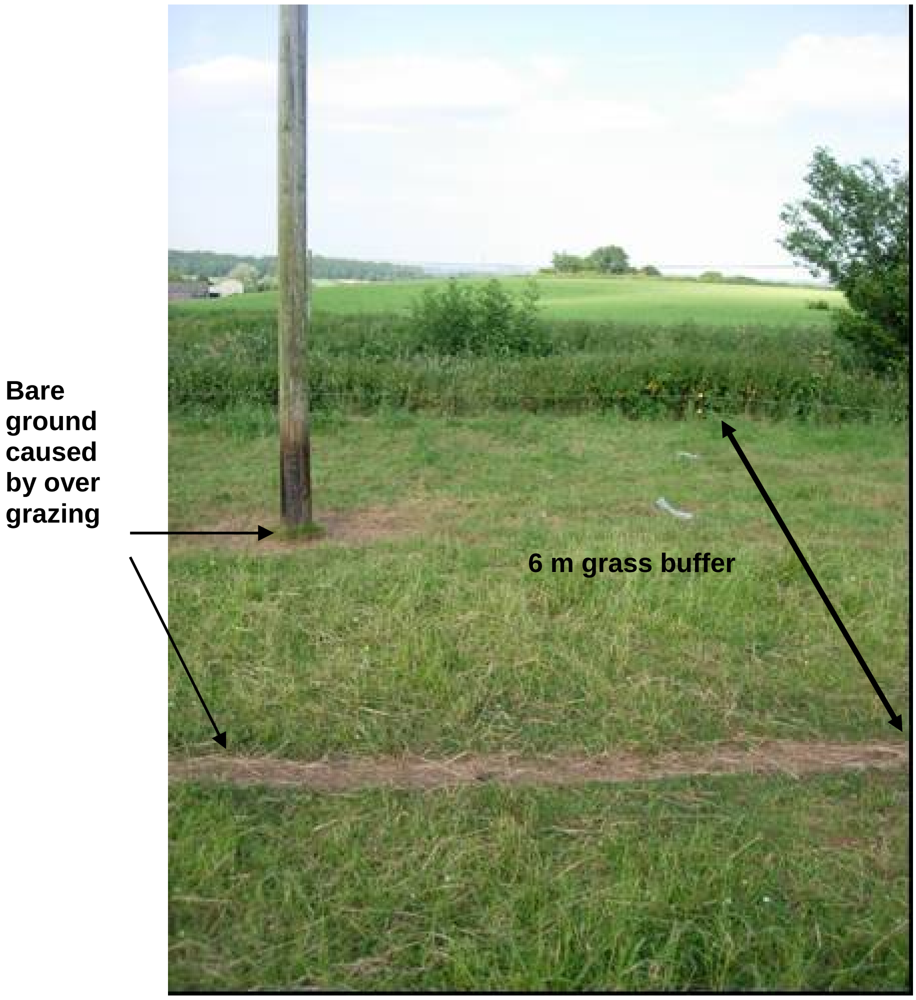



**Figure 5.14 Site 3 in July 2005. Bare ground caused by grazing.**

### 5.4.1.4 Site 4

Site 4 consisted of a 40000 m$^2$ field (wheat then potatoes) upslope of a 7.5 m grass VFS established under the Agri-environment scheme. At the bottom of the grass VFS was a 1 m high grass bank and then a gravel track. According to local residents, prior to installation of the VFS, the track regularly became covered in sediment that had washed off the field. The average length of the field above the VFS was 160 m and the slope angle was approximately 5°. The field was planted with wheat followed by oilseed rape. Along the length of the grass strip the vegetation was fairly uniform with a height of approximately 0.3 m in winter and 0.5 m in summer and a density of 80 to 100 %. The strip was occasionally used by dog walkers but no damage was evident. Along the front of the VFS a step of approximately 0.15 m down to the field had been created by ploughing activities (Figure 5.15). It is likely that both the step and the VFS vegetation prevented sediment from washing onto the VFS and track and therefore no sediment was collected from mats along this section.

At the field corner the VFS characteristics were different. Vegetation height varied in time and space between 0.25 and 1.35 m whilst density ranged from 20 to 100% cover. During the first visit the area appeared to have been used for vehicle turning. This had flattened the vegetation at the front of the VFS and created areas of bare ground in the field in front of it. During subsequent visits a network of rills formed here (approximately 0.45 m wide and 0.09 m deep). Crop growth was sparse in this area. Sediment delivered by the rills created areas of deposition, up to 30 mm deep, within the VFS vegetation. Being the lowest point along the VFS length, sediment transported along the edge of the step at the field edge was also deposited here. As the VFS vegetation became flattened and inundated by the flow the area of deposited sediment grew (Figure 5.16). This was particularly evident in the summer months following a period of heavy rain. The vegetation in the VFS at this site was also very tussocky and flow appeared to be channelled between the clumps of grass which measured 0.1 to 0.5 m in diameter.

The last two sampling visits occurred after the area of rills in the field had been levelled and planted with grass, creating a grass waterway feature running perpendicular to the VFS. This appeared to be effective in reducing the erosion and transport of sediment from the field. Sediment was deposited in the mats at the field corner on every visit until the establishment of the grass waterway. The mats that collected sediment were those at the front of the strip. The roughness of the vegetation seemed to prevent the sediment from being transported any further across the VFS.

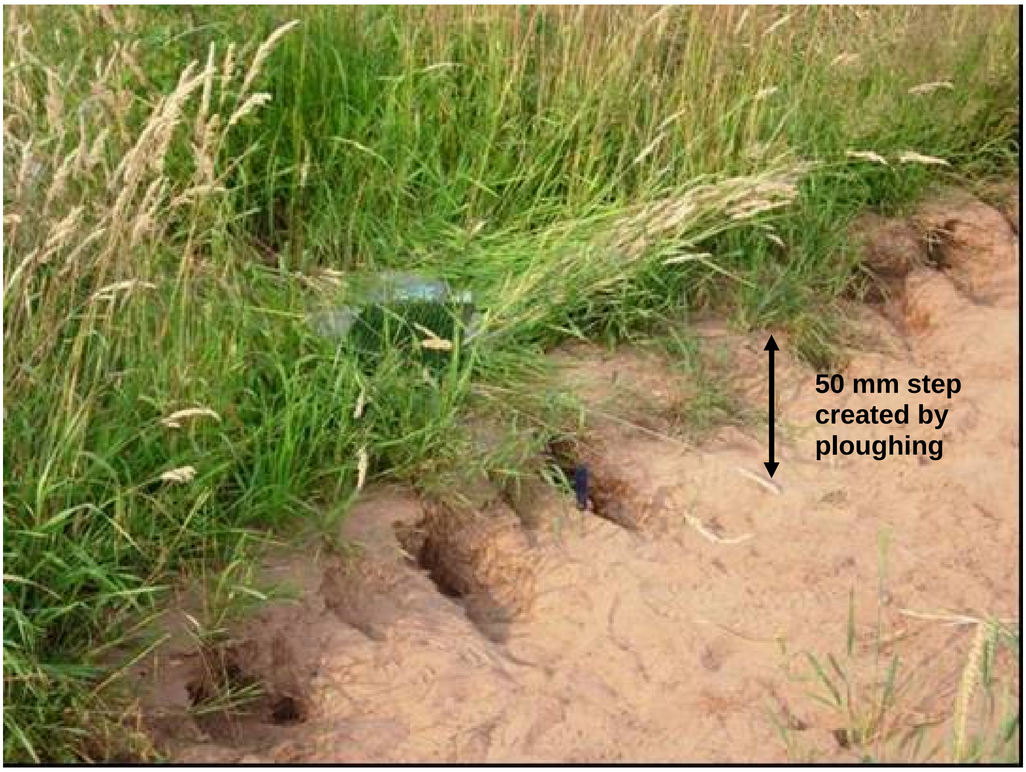



**Figure 5.15 Site 4: step with animal burrows at the VFS field interface.**

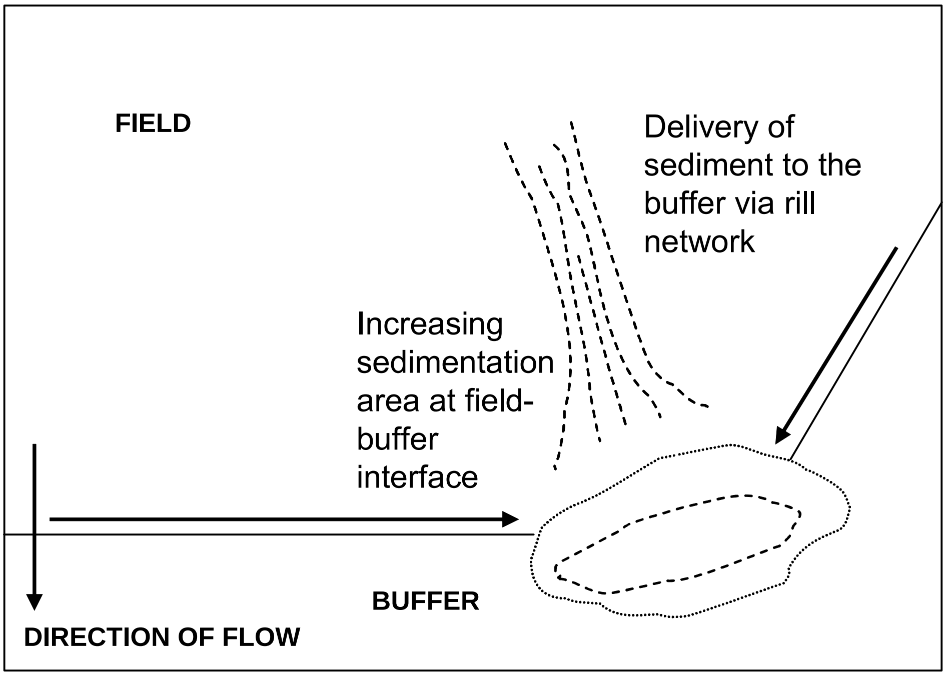



**Figure 5.16 Site 4: Schematic, flow & increasing deposition area (broken lines) at field-VFS interface.**

### 5.4.1.5 Site 5

At Site 5 a hedge, lined on either side by a 3 m grass strip, had been established to split what was once a large field into two smaller fields. The upslope field was 24700m$^2$ and during the sampling visits was observed as fallow, potatoes, fallow and then wheat. The field was on average 95 m long with an angle of 3°. The soil was a silty clay loam. The VFS vegetation maintained a 60 to 100% cover of grass, nettles and thistles. During the hot weather in the summer, when the crop was potatoes, the soil surface became very dry and cracked. At the lowest point in the field, within a wide, shallow channel running perpendicular to the VFS, a network of rills formed. The rills measured, on average, 0.72 m with an average depth of 0.35 m. They created a large sedimentation area in front of the VFS.

None of the mats at this site collected any sediment and no signs of sediment within the VFS were observed. This is due to a step measuring 0.5 m from the edge of the field up to the edge of the VFS which prevented any flow from moving over the VFS. Additionally, at the lowest point along the length of the VFS a large animal burrow had been created down to a depth of approximately 0.4 m (July 2005) to 0.5 m (May 2006) (Figure 5.17). It is likely that flow moving over the field and along the edge of the VFS drained through this hole.

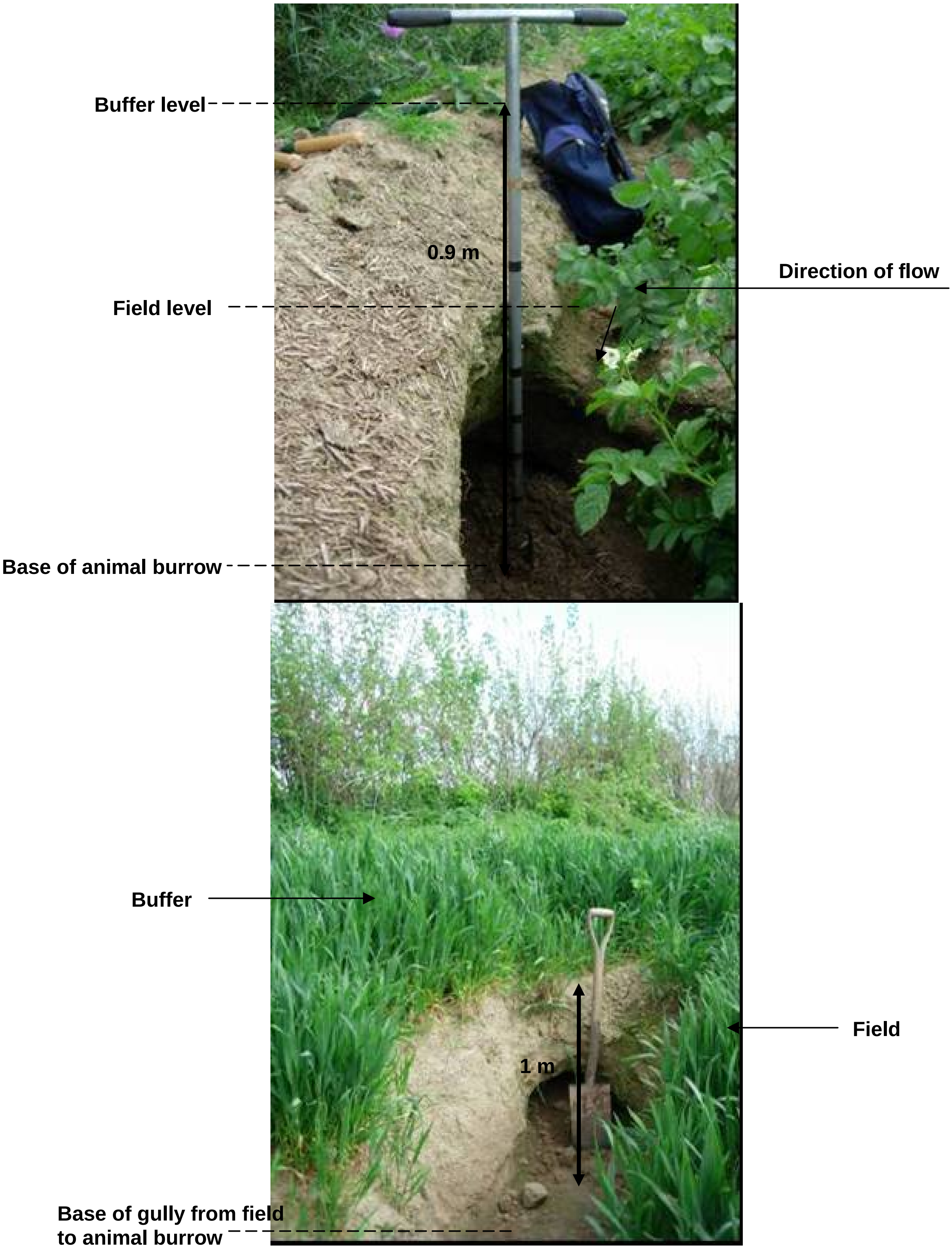



**Figure 5.17 Step and animal burrow at the VFS field interface at Site 5. (July 2005, May 2006).**

#### 5.4.1.6 Site 6

As at Site 5, the VFS at Site 6 consisted of a hedge lined upslope and downslope 3 m wide grass strips installed to split one large field into two. Prior to this the farmer had observed the formation of gullies and a build up of deposited sediment at the bottom of the field. This had not occurred following installation of the VFS under the Countryside Stewardship Scheme. The upslope field was 98550 $m^2$ and planted with barley followed by oilseed rape. The soil was a silty clay loam and the farmer was concerned about capping of the soil surface. The grass in the strip was consistently 150 to 200 mm high with a density cover of 80 to 100%. During the first field visit patches of compacted, bare ground suggested that the strip had been used by vehicles but the vegetation grew back in these areas.

Mats collected sediment within the upslope grass strip. Sediment was also deposited in the field upslope of the VFS, thereby creating a relatively smooth surface for sediment transport from the field into the VFS. However, the roughness of the vegetation appeared to prevent any of the material from moving to the back of the grass strip.

#### 5.4.1.7 Site 7

At Site 7 a 6.9 m grass strip had been established at the edge of a 100,000 $m^2$ field. There was a fence at the downslope edge of the VFS in front of a strip of shrubs, small trees and a large irrigation pond. Mats were placed within the grass strip. The field was bare during the first field visit and a large gully had developed perpendicular to the VFS. The average width of the gully was 1.07 m (1.5 m at the widest section) with a depth of 0.45 m (maximum depth of 0.65 m). At the base of the gully, covering the VFS from front to back, was a 33 m wide sedimentation area (Figure 5.18). The angle of the field sloping towards the VFS was approximately 7° but the field also converged towards the gully at an angle of 3 to 4°.

Sediment had also inundated the VFS at the field corner where the grass had become flattened by turning farm vehicles. Along the rest of the VFS the vegetation covered approximately 60 to 100% of the ground. There was a 50 mm step from the field surface up to the VFS edge. The field above the VFS was bare during the first field visit and then planted with wheat followed by peas. When the crop was present the gully was less deep and vegetation started to grow through the sedimentation area on the VFS. During the last two field visits a grass strip had been planted perpendicular to the VFS where the gully once existed. This appeared to prevent any further gully development and reduce transport of sediment. Mats located near to the base of the gully and in the field corner collected sediment but those along the field edge did not. No sediment was collected following the establishment of the grass waterway. Figure 5.18 to Figure 5.24 show the gully and resulting area of deposited sediment, re-growth of vegetation through the sedimentation area and establishment of a grass waterway.

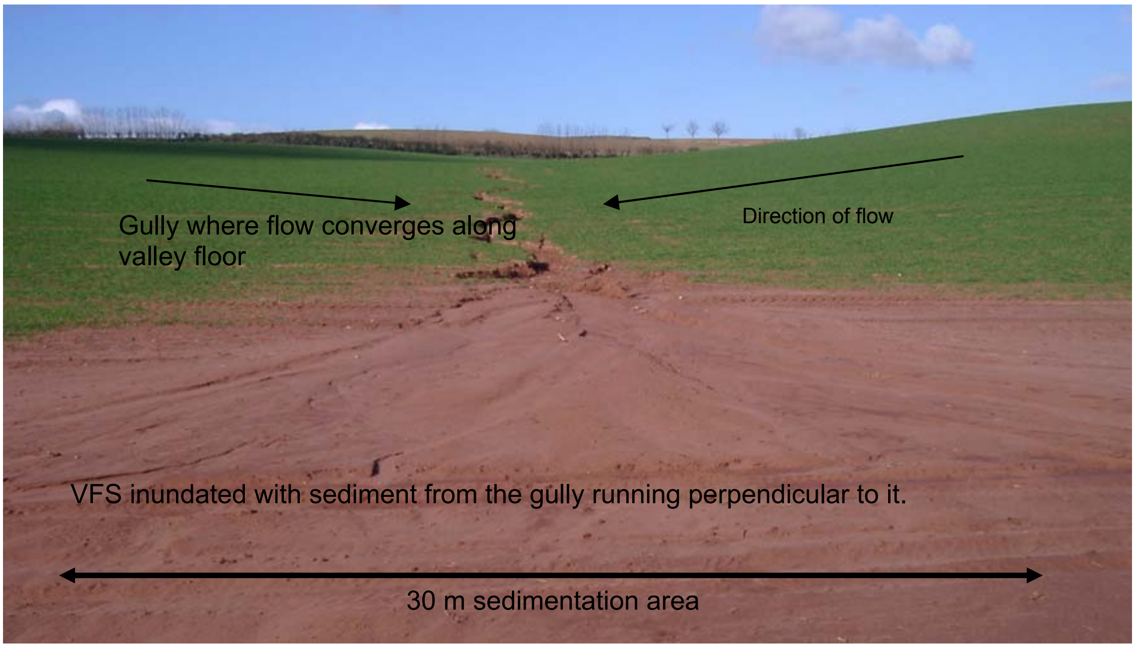



**Figure 5.18 Site 7: Sediment delivered by a gully to the VFS field interface (December 2004).**

**Figure 5.19 Site 7, looking upslope from the VFS: Protection of the field by a crop and vegetation growing through the area of deposited sediment visible in previous photograph (July 2005).**

**Figure 5.20 Site 7, looking cross slope: Sediment deposited along the VFS edge.**

**Figure 5.21 Site 7, looking upslope: Level ground in preparation for the establishment of a grass waterway (October 2005).**

**Figure 5.22 Site 7, looking upslope: Grass waterway (May 2006).**

**Figure 5.23 Site 7, looking cross slope: Grass waterway (May 2006).**

**Figure 5.24 Site 7, looking cross slope: Recovery of VFS vegetation following establishment of grass waterway (May 2006).**

### 5.4.1.8 Site 8

The field opposite Site 7 formed Site 8. The bottom of this wheat field was bordered by a 7 m Countryside Stewardship Scheme vegetated strip. The grass generally provided a 60 to 100% cover and a height of 100 to 300 mm in winter and 600 mm in summer. The field was gently sloping with an angle of 2 to 3°. It provided an even surface on which a dense crop cover was continuously maintained. No sediment was collected on the mats and there was no evidence of deposition in the VFS.

### 5.4.1.9 Site 9

Site 9 consisted of a 19975 m$^2$ field above a 6 m Countryside Stewardship grass strip and hedge. The field was planted with maize followed by wheat and sloped down to the VFS at an angle of approximately 4°. The grass within the front 3 m of the VFS was approximately 100 mm tall whilst that at the back was 200 to 500 mm tall. The majority of the VFS maintained an 80 to 100% cover. However, at isolated sections along the VFS-field interface sediment had spread onto the VFS. In July 2005 sediment spread from the front to the back of the VFS and across a width of 9.8 m at a significant "failure point". This followed a period of heavy rain in the early summer months. In this case it appeared that the sediment had built up against the hedge behind the grass strip so that the hedge was performing a buffering role. Figure 5.25 and Figure 5.26 show the width and depth of the area of deposition within the VFS.


**Figure 5.25 Site 9: VFS failure point with sediment building up against the hedge behind the grass strip.**


**Figure 5.26 Site 9: Depth of sedimentation in VFS.**

### 5.4.1.10 Site 10

Site 10 was a riparian strip consisting of 8.7 m of rough, dense vegetation (grass, nettles and thistles) at the base of a wheat field. The VFS vegetation maintained a 60 to 100% cover up to 1 m in height. This varied with season, as evident in Figure 5.27 and Figure 5.28. During February 2006 minor sedimentation areas were present within isolated areas along the field and VFS interface. Sediment was fairly evenly spread along the VFS and no rills were evident.

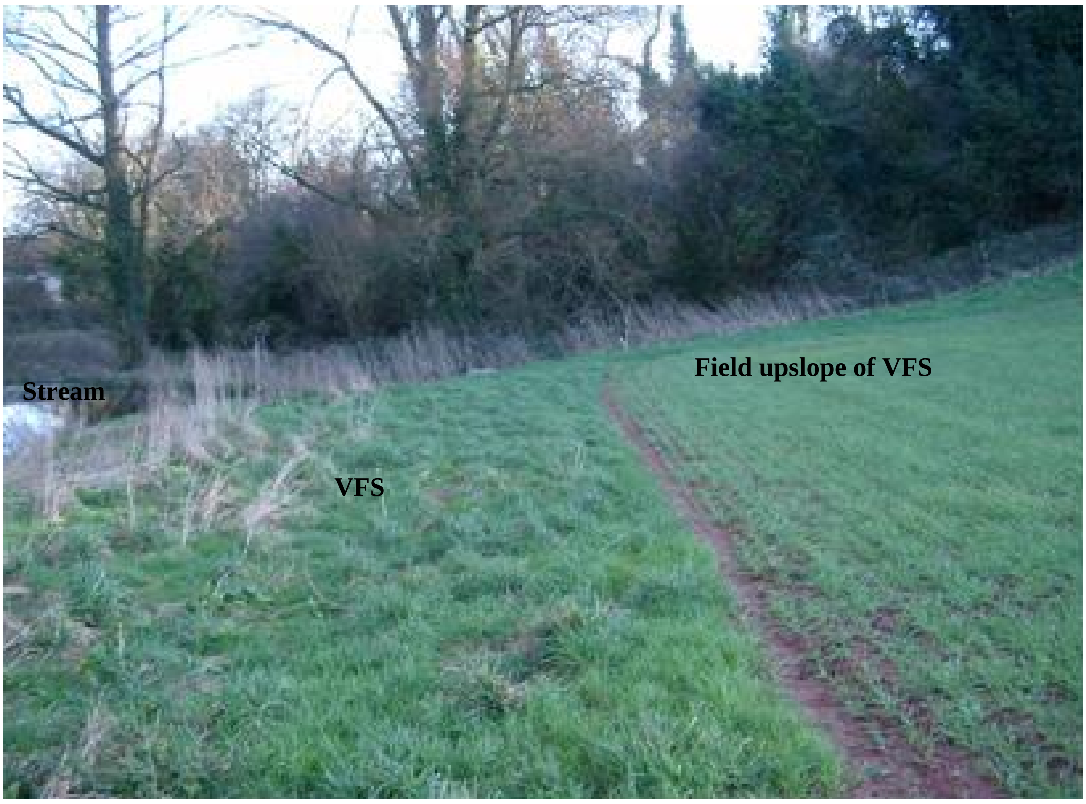



**Figure 5.27 Field Site 10 December 2004**

**Figure 5.28 Site 10 in July 2005.**

#### 5.4.1.11 Site 11 and Site 12

These two sites consisted of 6 m grass strips at the bottom (lower edge) of adjacent fields. They both maintained a vegetation density of 80 to 100% throughout the monitoring period. However, in both cases the exit to the field was at the lowest point along the VFS. Here sediment and concentrated flow converged from the field and was directed across the VFS. Points like this are potential routes of sediment transfer between fields.

#### 5.4.1.12 Site 13

An 18 m length of set aside at Site 13 had been established between a wheat crop and a road which had previously been periodically covered by sediment washing from the field. At the upslope edge of the set aside, deposited sediment occurred in isolated patches. However, no sediment deposition was evident further than approximately 8 m into the VFS. This suggests that the length of the area (and the vegetation density, which was maintained at 80 to 100 %) was effective in reducing transport of sediment to the road below.

#### 5.4.1.13 Site 14

The VFS at Site 14 was a 5 m riparian grass strip upslope of approximately 10 m of shrubs then trees. On average the grass cover maintained a density of 40 to 60% but there were a number patches of bare soil and channels running through the VFS from the field to the stream. This appeared to be caused by low points of elevation in the field directing concentrated flow to the VFS. The grass height was approximately 100 mm but was flattened in parts due to the VFS being used as a track for farm vehicles.

#### 5.4.1.14 Site 15

Site 15 consisted of approximately 30 m of riparian set aside; a constant 80 to 100% cover of grass and reeds. No erosion features or VFS failure points were observed at this site nor any movement or deposition of soil. The filtering capacity of the dense grass may have been enhanced by the resistance to bending of the reeds.

### 5.4.2 Site summary

Monitoring took into account VFSs at each site as well as the field contributing to the VFS. A number of characteristics, at various stages of sediment movement, influenced the performance of the features as VFSs. The processes and control variables directly observed or measured within the field study are summarised in Table 5.4.

| Stage | Process | Control variables & parameters | Related characteristics |
|---|---|---|---|
| Field upslope of buffer | Detachment of sediment | Rainfall | Amount |
| | | Crop | Density, row direction, distance between rows |
| | | Soil | Particle size distribution, compaction, |
| | | Slope | Angle, length, flow convergence |
| | | Erosion | Rill networks, gullies |
| | | In-field conservation | Grass waterways |
| Field-VFS interface | Transport of sediment to VFS | Terrain | Sedimentation area, difference in height between field & VFS, differences in height along length of VFS, animal burrows |
| VFS | Deposition of sediment in VFS | Vegetation | Roughness, density, height |
| | | Breach points | Channels, low points, gaps in vegetation, bare patches |
| | | Width | |
| | | Management | Damage by vehicles |

**Table 5.4 Factors observed at field sites that influenced the amount of sediment trapped by VFSs.**

### 5.4.3 Sediment deposition in VFS features (based on Astroturf mats)

#### 5.4.3.1 General

Sediment was observed and collected from Astroturf mats at seven of the nine instrumented sites (1, 2, 4, 6, 7, 9 and 10) (Table 5.5). At Site 3 no erosion or sediment movement was evident during any of the field visits. At Site 5 rill erosion took place but no eroded sediment passed through the VFS due to the step and animal burrow detailed in the site descriptions. Instead it was deposited at the leading edge of the VFS. Sampling took place at approximately three-monthly intervals following initial installation of the mats in December 2004/January 2005 (i.e. April 2005, July 2005, October 2005, January 2006 and May 2006). Within the following sections sediment amounts are presented in cm. This is to enable manageable, meaningful figures to be displayed rather than those that would result from a conversion to the SI units of mm or m.

| *Site* | *Number of mats installed* | *Sediment collected from mats (g cm$^{-2}$) averaged across mats that collected sediment* | | | | | |
|---|---|---|---|---|---|---|---|
| | | Jan to Apr 05 | Apr to Jul 05 | Jul to Oct 05 | Oct to Jan 06 | Jan to May 06 | *Average* |
| 1 | 12 | 0 | 0 | 0 | 1.17 (7) | 0 | *1.17* |
| 2 | 10 | 0 | 0 | 0 | 0.62 (5) | 0 | *0.62* |
| 3 | 12 | 0 | 0 | 0 | 0 | 0 | *0* |
| 4 | 15 | 5.79 (2) | 1.51 (4) | 0 | 0.01 (1) | 0 | *2.44* |
| 5 | 20 | 0 | 0 | 0 | 0 | 0 | *0* |
| 6 | 24 | 0.06 (2) | 0.07 (2) | 0 | 0.19 (1) | 0 | *0.11* |
| 7 | 12 | 0.86 (4) | 0.49 (11) | 0 | 0.42 (7) | 0.17 (4) | *0.49* |
| 9 | 11 | 0 | 4.21 (3) | 0 | 0.32 (1) | 0.15 (1) | *2.55* |
| 10 | 20 | 0 | 0 | 0 | 0.14 (2) | 0.70 (3) | *0.42* |

**Table 5.5 Number of mats that trapped sediment (in parentheses) and the average sediment deposition amount for each sampling period for the 60 mat that trapped sediment at the nine sample sites.**

Across all sites one hundred and thirty six mats were initially installed at positions covering the front, middle and back of the VFSs. Sediment was collected from thirty six of these mat positions. When a mat showed any signs of having collected sediment delivered by overland flow it was removed for analysis and another installed in its place. On many occasions the replacement mat collected sediment during the subsequent sampling period providing a total number of sixty mats that collected sediment during the field study.

None of the mats collected sediment during all periods. Sediment was collected from one or more mats in April and July 2005 and in January and May 2006. No samples were collected in October 2005. Over the eighteen month period, the typical number of mats to collect sediment, per site, was five to seven with exceptions being Sites 3 and 5 (no mats) and Site 7 (26 mats). The period between October 2005 and January 2006 was the period during which the greatest number (n = 24) of mats collected sediment, reflecting the numerous rainfall events and resulting erosion, during that autumn-winter period. The most sediment was collected at Site 7 where mats were collected on four out of the five sampling visits. The average amount of sediment deposition was greatest for mats collected in July

2005 and lowest for mats collected in May 2006. High sedimentation values for the period May to July 2005 reflect some intense storm events during this period and exposed topsoil, which caused severe erosion at Site 9. In contrast, in 2006 in-soil conservation measures dramatically reduced previously observed signs of erosion at Sites 4 and 7.

### 5.4.3.2 Mats with and without sediment (n = 136)

Table 5.6 presents average sediment deposition amounts for each site for the full 18 month sampling period. Values are based on all mats at each site so that mats that did not collect sediment are included in the calculations. Thus, cumulative sediment deposition was calculated for each mat over the full 18 month period. The average value for each site was calculated based on the number of mats installed at each site (i.e. column 2 of Table 5.5). As shown in Table 5.6, average sediment deposition for the sampling periods ranged from 0 g cm$^{-2}$ at Sites 3 and 5 to >1 g cm$^{-2}$ at Site 9 between April and July 2005. The average for all nine sites (i.e. based on all 136 mats) for a three month sampling period was 0.08 ± 0.02 g cm$^{-2}$, which is approximately 0.30 g cm$^{-2}$ year$^{-1}$.

| | *Sediment deposition (g cm$^{-2}$) averaged across all mats with and without sediment* | | | | |
|---|---|---|---|---|---|
| Site | Jan to Apr 05 | Apr to Jul 05 | Jul to Oct 05 | Oct to Jan 06 | Jan to May 06 |
| 1 | 0 | 0 | 0 | 0.68 (± 0.21) | 0 |
| 2 | 0 | 0 | 0 | 0.31 (± 0.11) | 0 |
| 3 | 0 | 0 | 0 | 0 | 0 |
| 4 | 0.77 (± 0.55) | 0.35 (± 0.28) | 0 | 0.001 (± 0.001) | 0 |
| 5 | 0 | 0 | 0 | 0 | 0 |
| 6 | 0.005 (± 0.004) | 0.006 (± 0.005) | 0 | 0.008 (± 0.008) | 0 |
| 7 | 0.288 (± 0.164) | 0.431 (± 0.154) | 0 | 0.248 (± 0.140) | 0.058 (± 0.043) |
| 9 | 0 | 1.150 (± 0.593) | 0 | 0.03 (± 0.03) | 0.014 (± 0.014) |
| 10 | 0 | 0 | 0 | 0.002 (± 0.012) | 0.105 (± 0.074) |

**Table 5.6 Average sediment deposition for all 136 mats installed with standard error in parentheses.**

### 5.4.3.3 Mats with sediment (n = 60)

Table 5.5 shows the average amount of sediment deposited at each of the sites per sampling period and also the average for all of the 60 mats collected. It is clear that there was considerable variation in the amount of sediment deposited between the seven sites. The average amount of deposited sediment ranges from 0.11 g cm$^{-2}$ with standard error (SE) 0.04 g cm$^{-2}$ (at Site 6 based on five mats) to 2.55 g cm$^{-2}$, SE 4.0 g cm$^{-2}$ (at Site 9 based on five mats). The average sediment deposition for all of the sixty mats that trapped sediment is 1.17 $cm^{-2}$, SE 1.5 $cm^{-2}$. Total sediment deposition for all sites is illustrated in Figure 5.17.

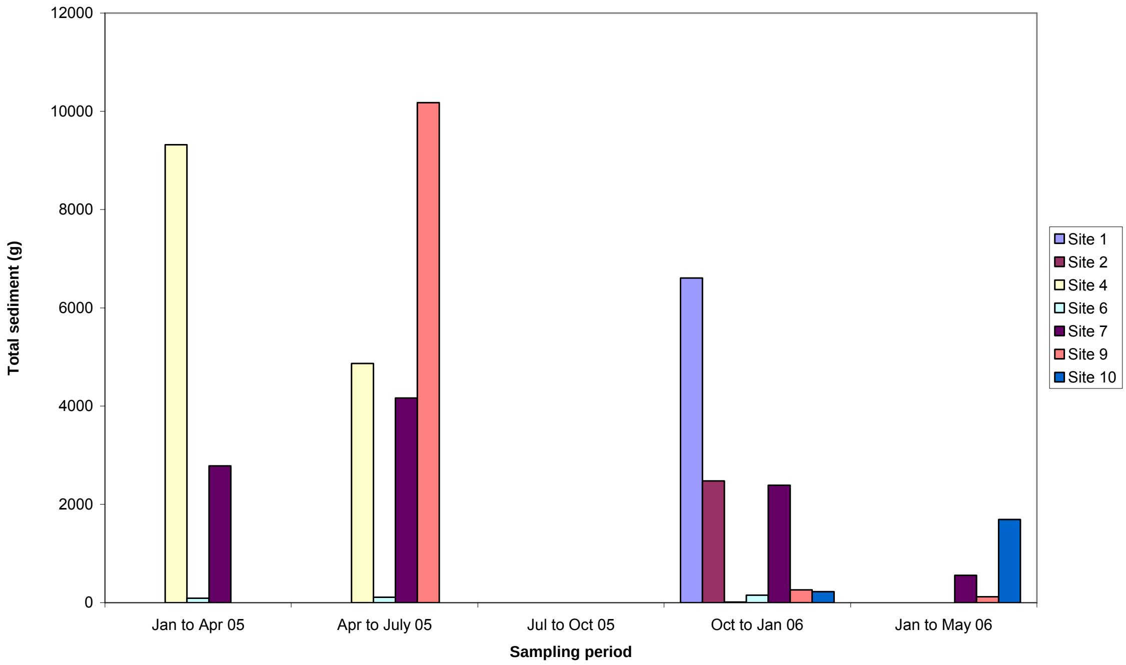



**Figure 5.29 Total sediment deposition at the sites during each sampling period as a sum of sediment collected from the mats.**

### 5.4.3.4 Particle size

Table 5.7 and Table 5.8 present average values of the particle size composition (% sand and $d_{50}$) of the deposited sediment. Generally, most of the sediment trapped by the mats was sand-sized material with values greater than 80% (except for Site 6 where approximately 50% of the trapped material is sand sized). The average value for all sites is 83 ± 14%. Average values for $d_{50}$ were more variable between sites and ranged between 78 ± 54 µm (Site 6) and 326 ± 19 µm (Site 2). The average value for all sixty mats was 195 ± 89 µm.

| *Site* | *Sand fraction (%) by sampling period* | | | | | |
|---|---|---|---|---|---|---|
| | Jan to Apr 05 | Apr to Jul 05 | Jul to Oct 05 | Oct to Jan 06 | Jan to May 06 | Average |
| 1 | - | - | - | 91 | - | 91 |
| 2 | - | - | - | 90 | - | 90 |
| 3 | - | - | - | - | - | |
| 4 | 93 | 96 | - | 89 | - | 93 |
| 5 | - | - | - | - | - | |
| 6 | 52 | 52 | - | 50 | - | 51 |
| 7 | 88 | 82 | - | 69 | - | 80 |
| 9 | - | 92 | - | 93 | 89 | 91 |
| 10 | - | - | - | 92 | 92 | 92 |

**Table 5.7 Average sand fraction for each sampling period for those 60 mats that trapped sediment at the nine sample sites.**

| *Site* | *$d_{50}$ (µm) by sampling period* | | | | | |
|---|---|---|---|---|---|---|
| | Jan to Apr 05 | Apr to Jul 05 | Jul to Oct 05 | Oct to Jan 06 | Jan to May 06 | Average |
| 1 | - | - | - | 250 | - | 250 |
| 2 | - | - | - | 326 | - | 326 |
| 3 | - | - | - | - | - | - |
| 4 | 144 | 151 | - | 218 | - | 171 |
| 5 | - | - | - | - | - | - |
| 6 | 69 | 94 | - | 63 | - | 75 |
| 7 | 130 | 142 | - | 137 | - | 136 |
| 9 | - | 306 | - | 310 | 279 | 298 |
| 10 | - | - | - | 323 | 305 | 317 |

**Table 5.8 Average $d_{50}$ for each sampling period for those 60 mats that trapped sediment at the nine sample sites.**

### 5.4.3.5 Position in VFS

Of all of the sixty mats that trapped sediment over the eighteen month period 50% were located at the front of the VFS with 20% and 30% at the middle and back of the VFS respectively. At most of the sites only a limited number of samples were collected (*n* ranges between five at Sites 2, 6, 9 and 10 and seven at Sites 1 and 4). However, some comparison at each site is possible.

Figure 5.30 shows that sediment is generally trapped at the front of the VFS. Site 2 is the only site at which the largest proportion of sediment was not collected from the front of the VFS. No sediment at all was collected from the back of Site 4. Table 5.9 to Table 5.11 compare the characteristics of the deposited sediment for each VFS position. Sites 2, 6 and 10 have a higher $d_{50}$ at the back of the VFS which indicates coarser material here. Additionally, with the exception of Site 9 all of the sites show lower clay and silt contents at the back of the VFS and all of the sites except Site 2 show lower clay contents at the back of the VFS. Although there is no significant difference between the clay content at the front and back of the VFSs the difference can be seen in Figure 5.31 indicating that the clay content is generally higher at the front of the VFSs.

At Site 9 there were enough samples for an assessment of the differences in the sediment collected by mats located at the front and back of the VFS for one sampling period (July 2005). This showed that there was a significantly greater amount of sediment deposited at the front of the VFS (Table 5.12). It also showed no significant difference in particle size between the front and back. However, in contrast to the other results, sediment deposited at the front was slightly coarser than that deposited at the back. Patterns in sediment characteristics are examined further within the section on soil core transects.

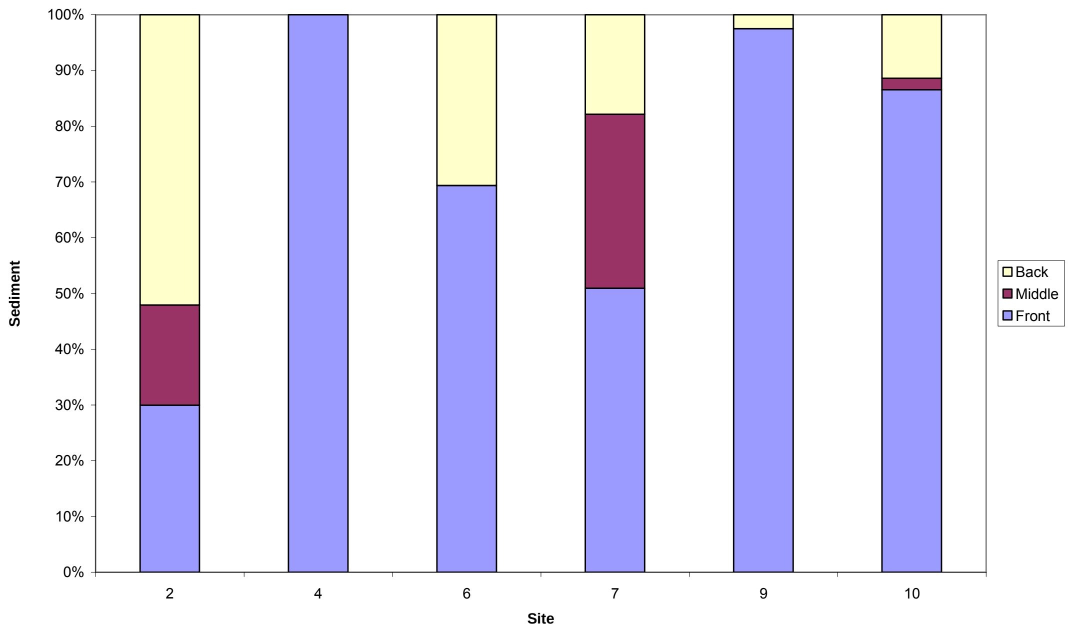



**Figure 5.30 Proportion of sediment at each VFS position, per site.**

| *Site* | *$d_{50}$ (μm)* | | |
|---|---|---|---|
| | *Front* | *Middle* | *Back* |
| 2 | 311 | 332 | 336 |
| 4 | 159 | - | - |
| 6 | 69 | - | 98 |
| 7 | 145 | 145 | 138 |
| 9 | 305 | - | 294 |
| 10 | 305 | 334 | 327 |

**Table 5.9 Average $d_{50}$ for all mat samples collected from each site during the 18 month sampling period.**

| *Site* | *Clay and silt (%)* | | |
|---|---|---|---|
| | *Front* | *Middle* | *Back* |
| 2 | 11.68 | 6.72 | 10.57 |
| 4 | 5.02 | - | - |
| 6 | 48.45 | - | 47.95 |
| 7 | 21.11 | 22.19 | 17.22 |
| 9 | 7.57 | - | 8.9 |
| 10 | 8.19 | 6.95 | 1.17 |

**Table 5.10 Average clay and silt content for all mat samples collected from each site during the 18 month sampling period.**

| *Site* | *Clay (%)* | | |
|---|---|---|---|
| | *Front* | *Middle* | *Back* |
| 2 | 2.76 | 1.92 | 3.03 |
| 4 | 1.05 | - | - |
| 6 | 8.50 | - | 7.90 |
| 7 | 3.21 | 4.13 | 2.46 |
| 9 | 4.00 | - | 1.30 |
| 10 | 1.37 | - | 1.17 |

**Table 5.11 Average clay content for all mat samples collected from each site during the 18 month sampling period.**

| *Location* | *N* | *Sediment deposition* ($g\ cm^{-2}$) | *Sand (%)* | *Clay (%)* | $d_{50}$ ($\mu m$) |
|---|---|---|---|---|---|
| Front | 5 | 0.66 (0.67)[a] | 85 (9) | 2.48 | 151 (53) |
| Back | 4 | 0.30 (0.11) | 79 (13) | 3.03 | 138 (48) |

[a] Average ± 1SD

**Table 5.12 Amount and particle size composition of the sediment deposited on the front and back mats at Site 9 collected in July 2005 (i.e. representing the period May to July 2005).**

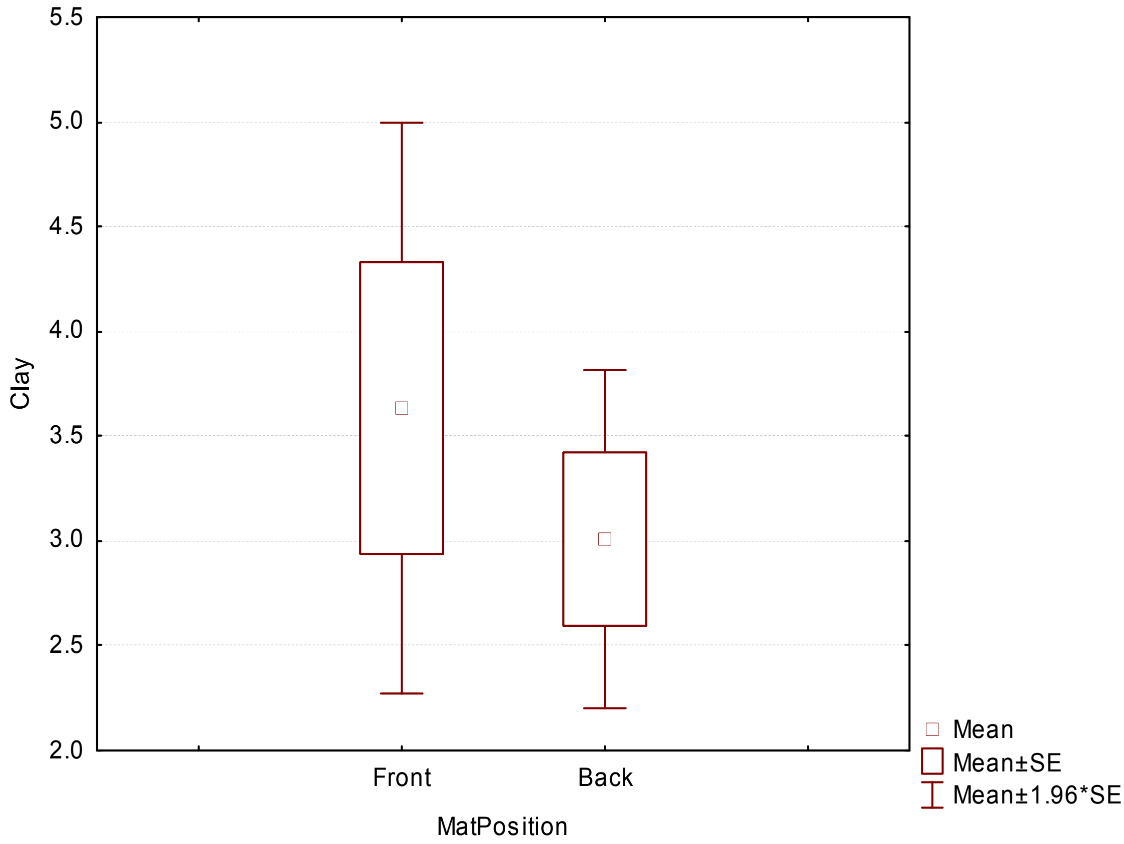



**Figure 5.31 Mean clay fraction at the front and back of the VFSs.**

## 5.4.4 Rainfall data

Rainfall data was taken from Met Office records from within the Parrett catchment (Yeovilton) and plotted against the total amount of sediment collected from the VFSs. A summary of the Met Office data is included in Table 5.13. A number of heavy rainfall events were confirmed by local observations, for example, over 25 mm of rainfall was observed by two farmers on 24th and 28th June 2005. This corresponds with sediment collected from Sites 7 and 9 in July 2005. Rainfall information is presented, for those sites at which sediment was collected by the Astroturf mats, in Figure 5.32 to Figure 5.38. At Sites 1, 2, 6 and 10 there appears to be a relationship between rainfall and sediment deposition with the majority of deposition occurring following the highest rainfall amounts (October 2005 to January 2006). At Sites 4, 7 and 9 a marked decrease in sediment deposition occurs following July 2005. This is likely to have been due to the introduction of a grass waterway at Sites 4 and 7 and due to a change in crop at Site 9. The change in crop from maize to wheat provided a denser cover. These details have been added to the graphs.

| Year | Month | Rainfall (mm) | Mean temp (degrees) | Rain days |
|---|---|---|---|---|
| 2005 | January | 95.8 | 6.3 | 15 |
| 2005 | February | 54.1 | 4.3 | 9 |
| 2005 | March | 65.6 | 7 | 10 |
| 2005 | April | 96.6 | 8.7 | 15 |
| 2005 | May | 58.6 | 10.8 | 10 |
| 2005 | June | 78.6 | 14.9 | 10 |
| 2005 | July | 78.1 | 16.4 | 10 |
| 2005 | August | 52.9 | 15.6 | 8 |
| 2005 | September | 76.9 | 14.7 | 13 |
| 2005 | October | 186.6 | 12.7 | 17 |
| 2005 | November | 147.1 | 6.5 | 14 |
| 2005 | December | 110.8 | 5 | 10 |
| 2006 | January | 49.8 | 4.5 | 8 |
| 2006 | February | 75.7 | 3.6 | 10 |
| 2006 | March | 119.6 | 5.3 | 17 |
| 2006 | April | 40.8 | 8.4 | 9 |
| 2006 | May | 138.9 | 11.4 | 15 |

**Table 5.13 Met Office data recorded within the Parrett catchment (Yeovilton) for rainfall, mean temperature and number of rain days.**

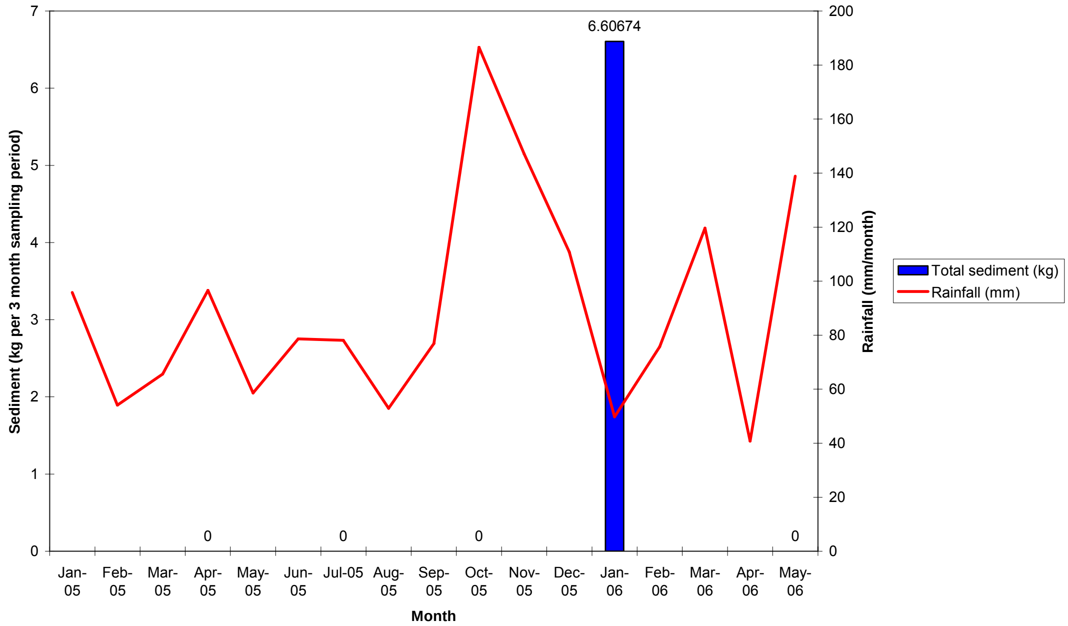



**Figure 5.32 Site 1: sediment collected from mats in VFS and rainfall**

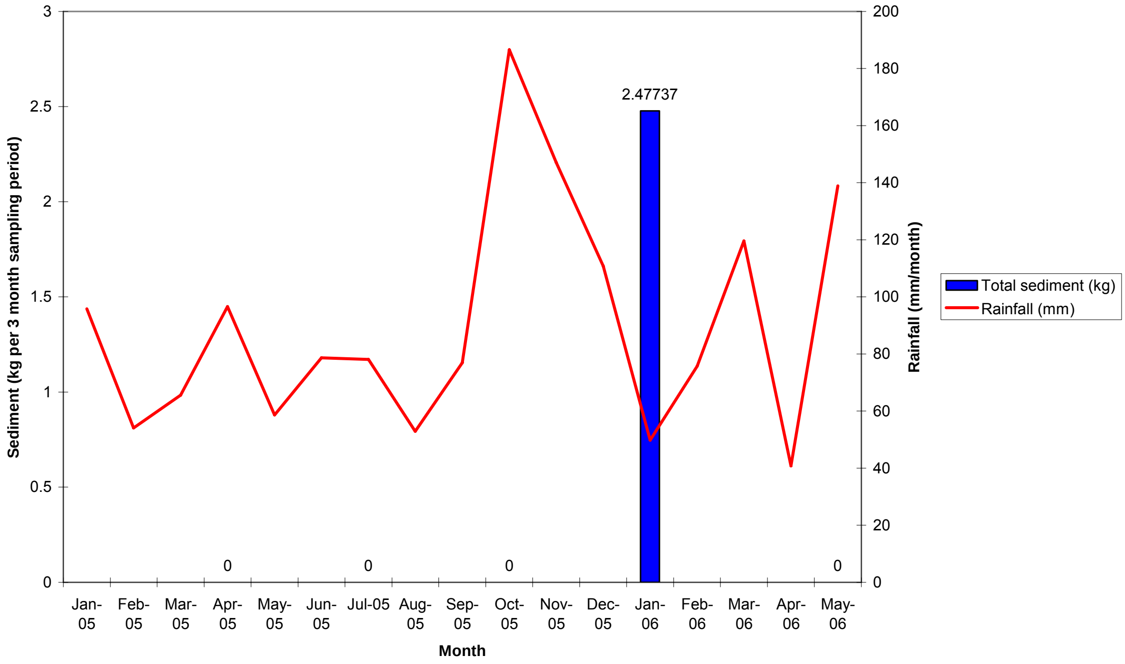



**Figure 5.33 Site 2: sediment collected from mats in VFS and rainfall**

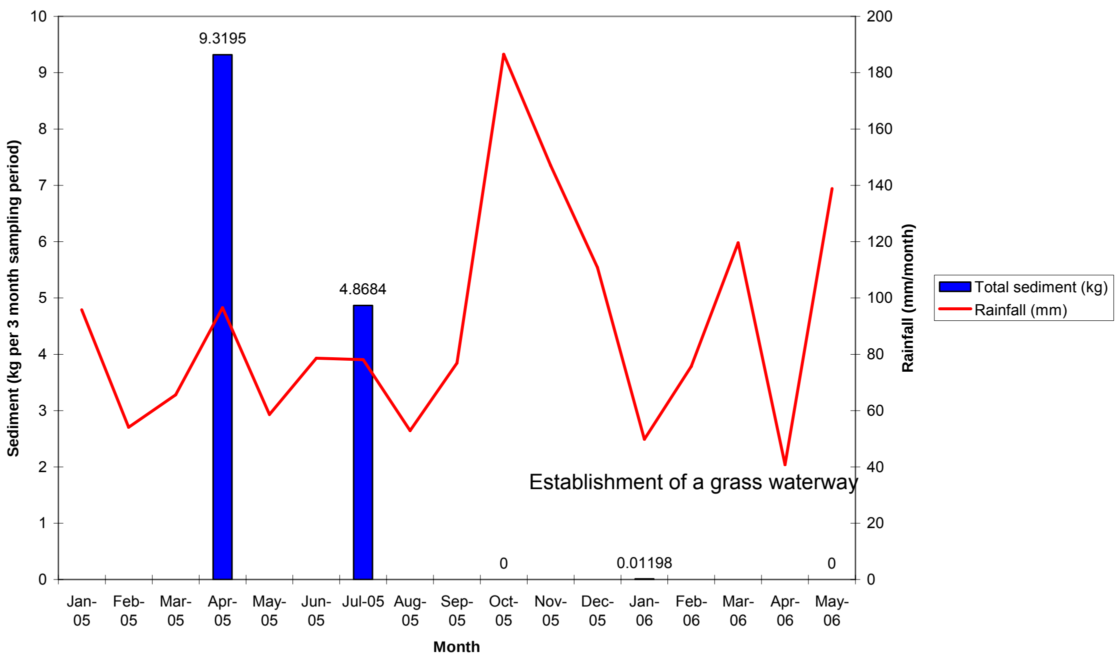



**Figure 5.34 Site 4: sediment collected from mats in VFS and rainfall**

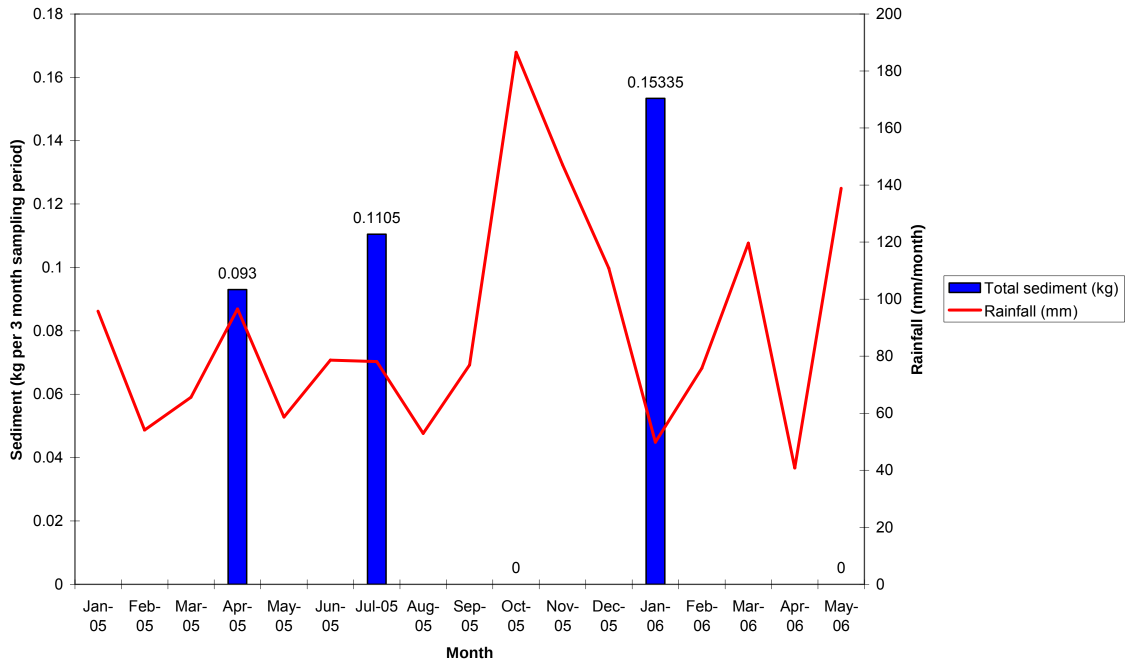



**Figure 5.35 Site 6: sediment collected from mats in VFS and rainfall**

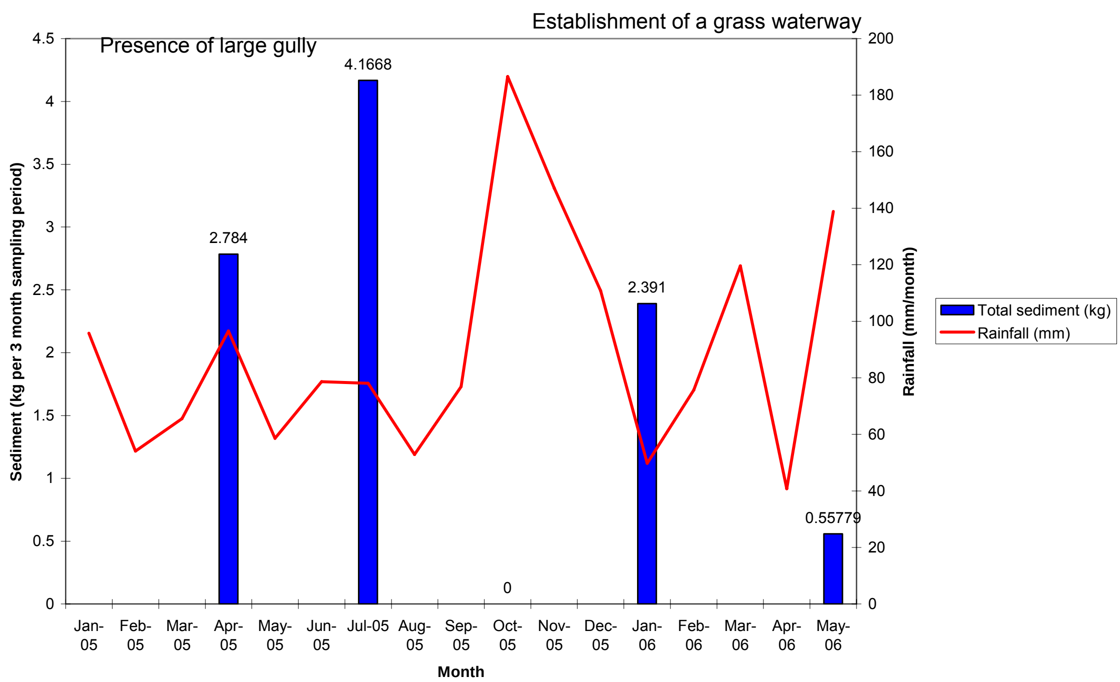



**Figure 5.36 Site 7: sediment collected from mats in VFS and rainfall**

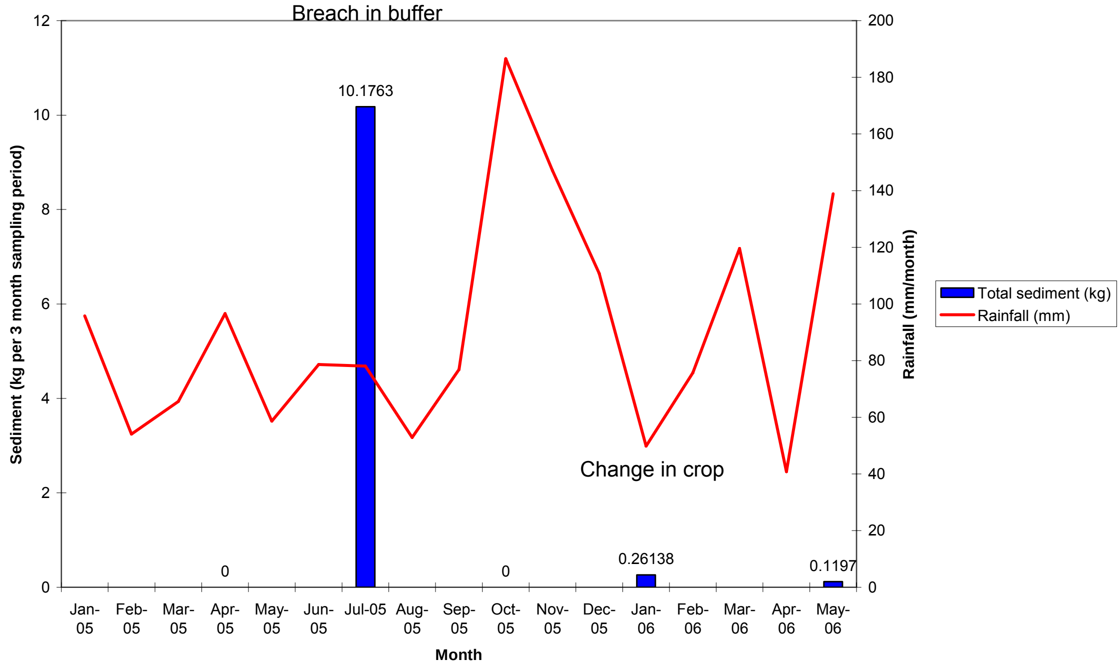



**Figure 5.37 Site 9: sediment collected from mats in VFS and rainfall**

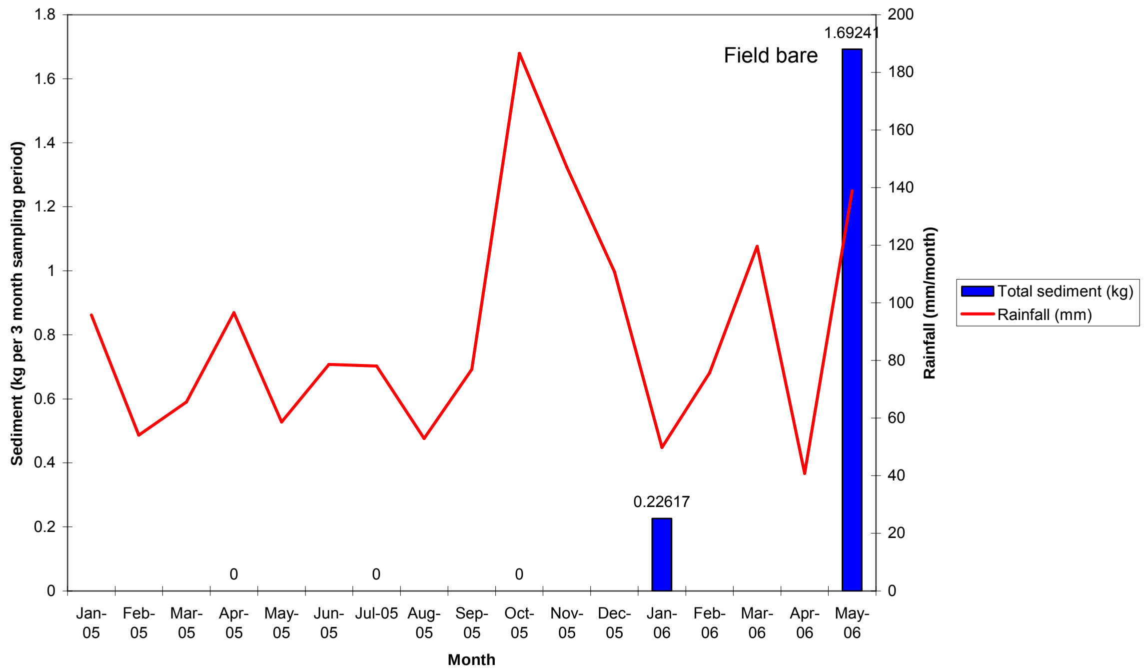



**Figure 5.38 Site 10: sediment collected from mats in VFS and rainfall**

### 5.4.5 Soil characteristics of upslope fields (based on depth profiles and transects)

#### 5.4.5.1 Depth profiles

The soil samples taken at the sites allow comparison of the soil in the fields with the sediment deposited in the VFSs. The layout of these samples is illustrated in Figure 5.39. Figure 5.40 illustrates the depth distribution of the <63 μm sediment fraction. Cores of 20 mm were taken in the VFSs and field down to a depth of 120 mm. It can be seen that at all sites, except Site 2 and Site 6, all of the cores from 0 to 120 mm depth are coarser in the VFSs than in the upslope fields. Site 6 follows the same pattern except for the top 30 mm where the soil in the field appears to be coarser than that in the VFS. There is no significant difference in the $d_{50}$ with depth or position. Neither is there a significant difference in clay content with depth but there is a difference between field and VFS profiles ($p = 0.001$). At every depth clay content is higher in the field than in the VFSs (Figure 5.41). This is consistent with the deposition of coarser sediment in the VFSs.

#### 5.4.5.2 Transects

Figure 5.42 illustrates the 0 to 20 mm samples taken within the VFSs and at 10 m intervals along transects through the upslope field. There is no trend for the silt and clay fraction with distance along the transect. Although not statistically significant, at Sites 4, 7 and 10 the <63 μm material (i.e. the silt and clay fractions) is generally lower in the VFSs compared to the contributing fields. Furthermore, it appears that the percentage of fine material is greatest at the upslope edge of the VFS and decreases both towards the back of the VFS and into the field. This within-VFS pattern is consistent with the material analysed from the mats. Figure 5.43 to Figure 5.48 show that the cores with the highest percentage of fine particles all occur either at the front, or in front (Sites 4, 7 and 10), of the VFS. Although there are not enough samples for this to be conclusive the pattern should be investigated further because it contradicts the expectation that fine material will move through the VFS.

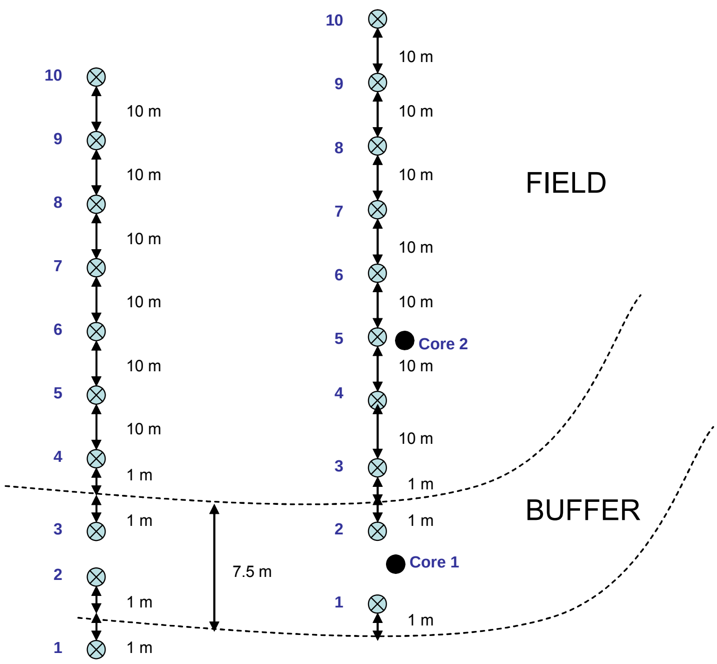



**Figure 5.39 Schematic plan of the location of surface cores (0-20 mm) along two transects and the location of two depth incremental cores (0-120 mm) at Site 4. The distance between the two transects is approximately 100 m. Distances are not to scale.**

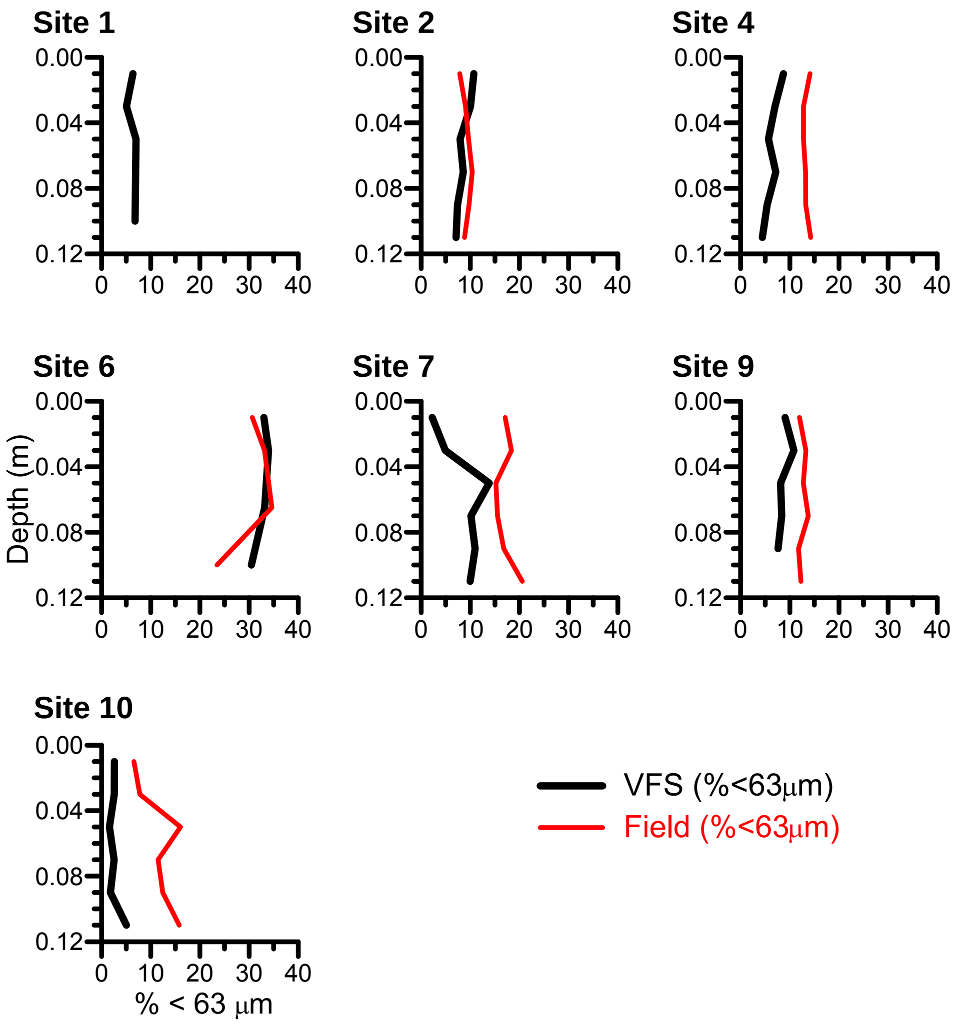



**Figure 5.40 Depth profiles of the % <63 µm (i.e. silt and clay) of the <1 mm fraction of soil for Sites. Two depth profiles were determined at each site; a profile for the VFS feature and one for the soil in the upslope contributing field.**

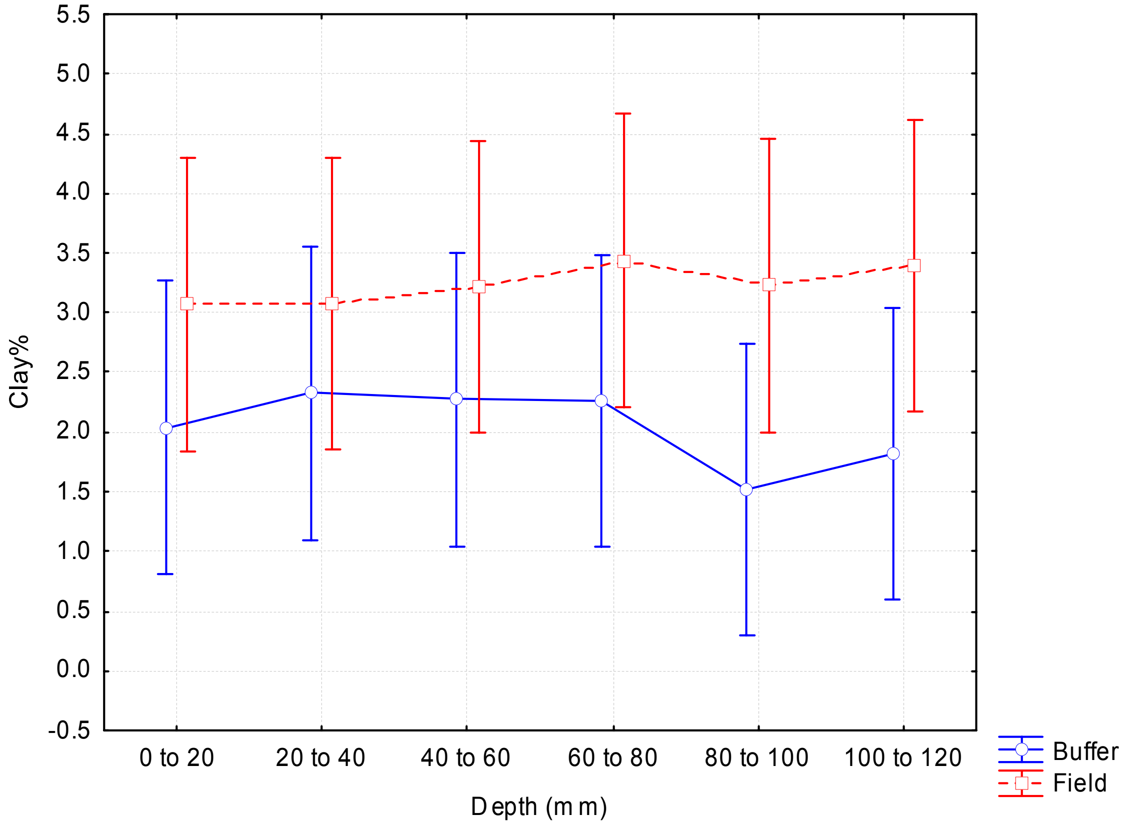



**Figure 5.41 Mean clay content for field and VFS at depths to 120 mm (all VFSs and fields).**

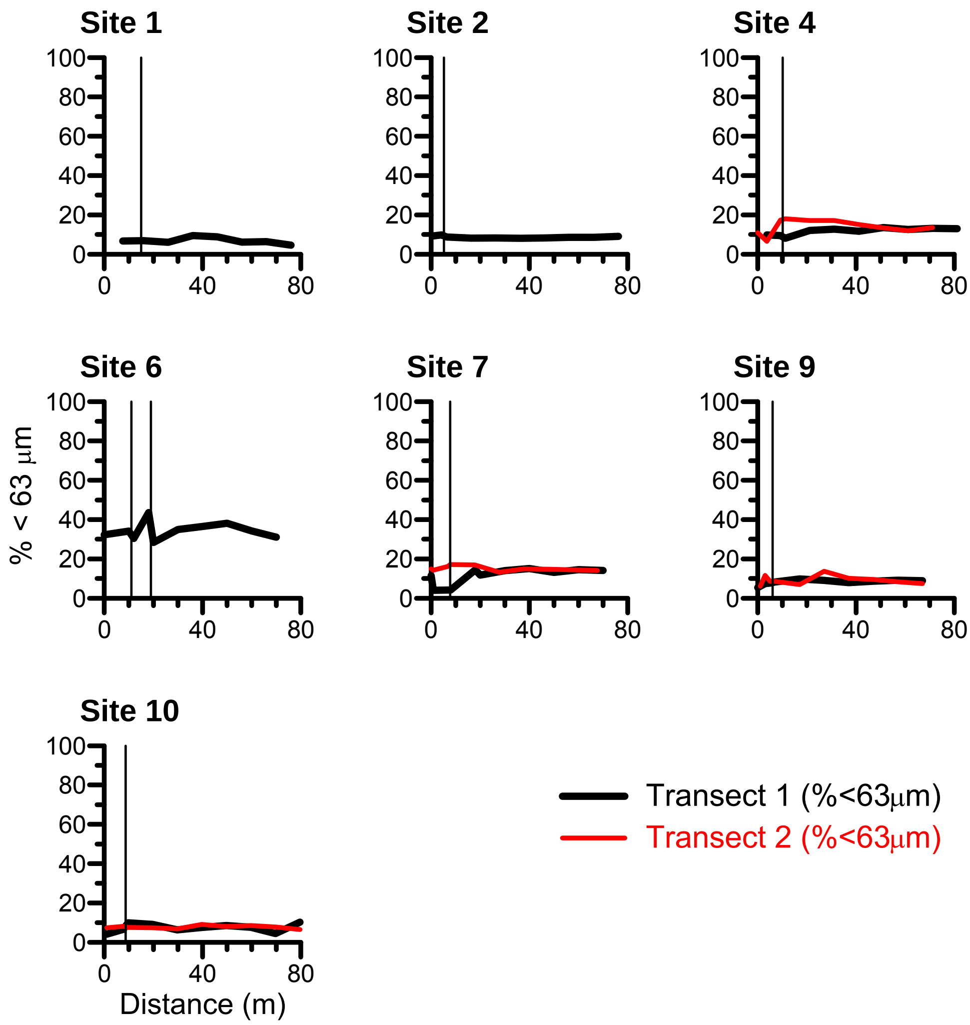



**Figure 5.42 Percentage of silt and clay (i.e. the % <63 µm content of surface (upper 20 mm) soil samples collected along the two transects at Sites 1, 2, 4, 6, 7, 9 and 10. The first sample represents the downslope edge of the VFS feature and the last sample was typically collected 60 to 70 m upslope of the leading edge of the VFS feature, which is represented by the thin vertical line.**

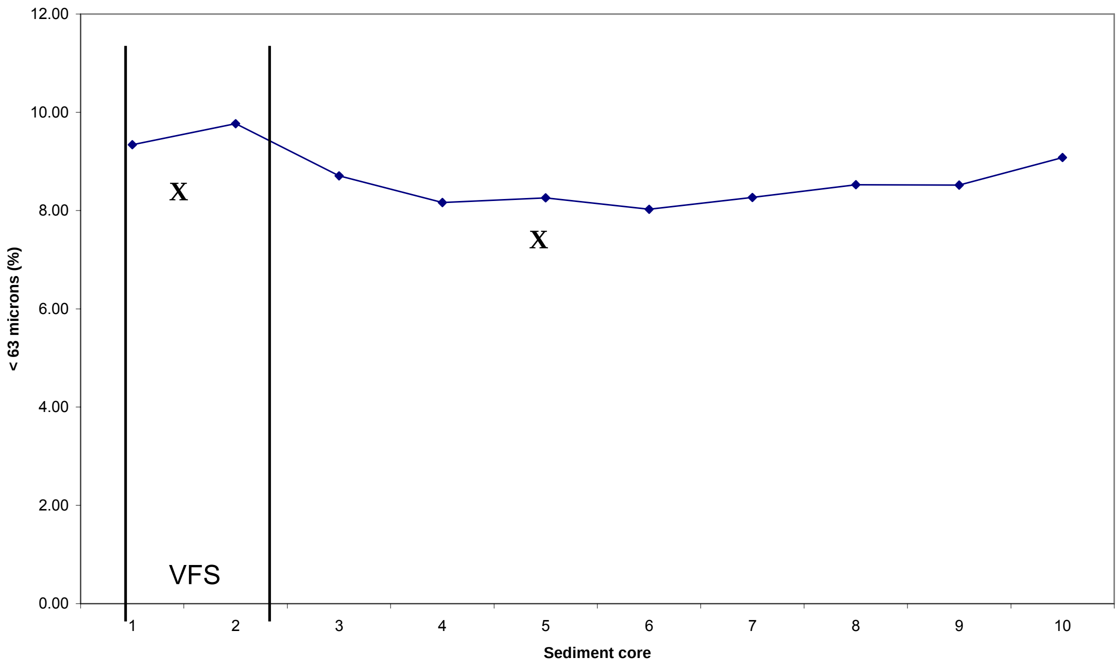



**Figure 5.43 Site 2: less than 63 micron fraction (i.e. silt and clay) for 10 sediment cores taken through the VFS and lower section of upslope field. X marks the locations where depth profiles were taken.**

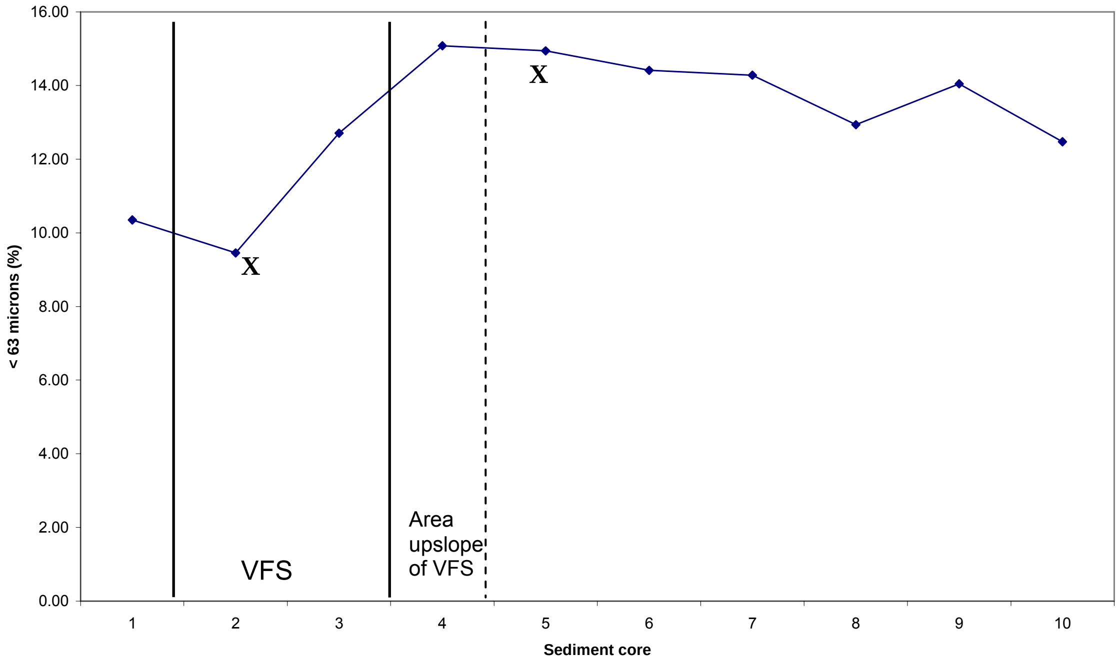



**Figure 5.44 Site 4: less than 63 micron fraction (i.e. silt and clay) for 10 sediment cores taken through the VFS and lower section of upslope field. X marks the locations where depth profiles were taken.**

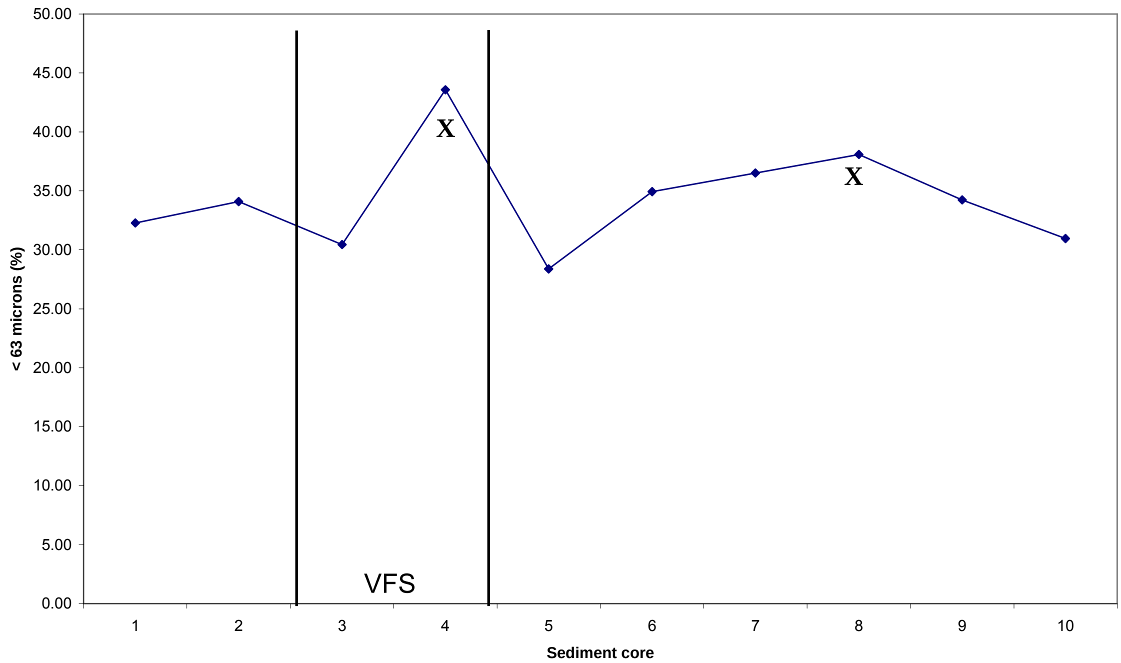



**Figure 5.45 Site 6: less than 63 micron fraction (i.e. silt and clay) for 10 sediment cores taken through the VFS and lower section of upslope field. X marks the locations where depth profiles were taken.**

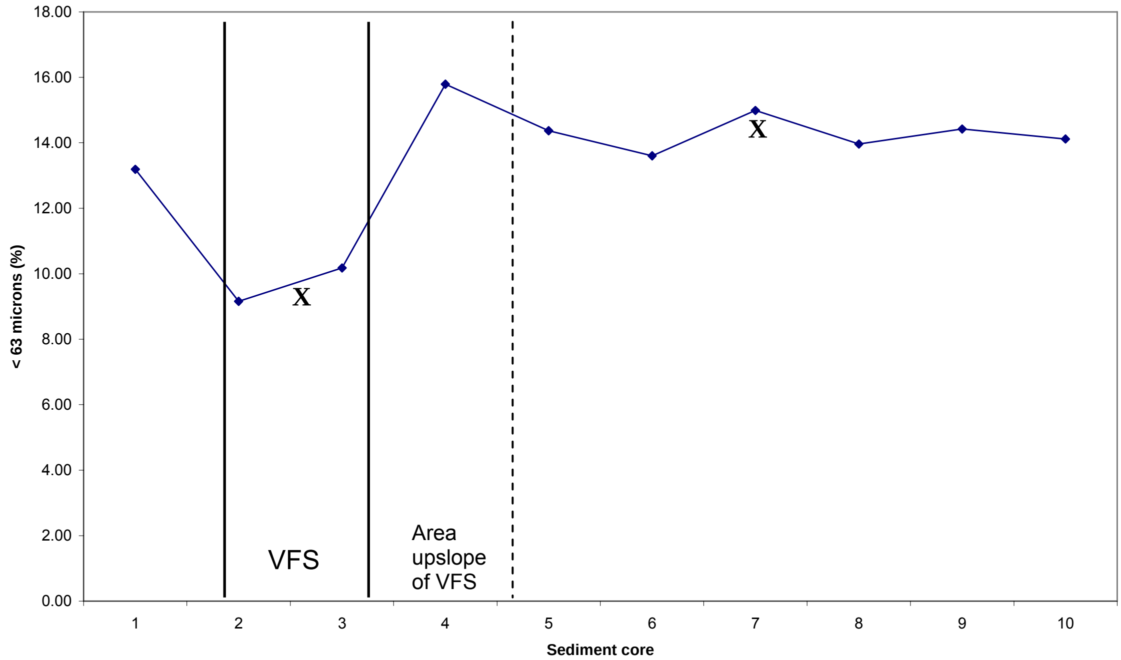



**Figure 5.46 Site 7: less than 63 micron fraction (i.e. silt and clay) for 10 sediment cores taken through the VFS and lower section of upslope field. X marks the locations where depth profiles were taken.**

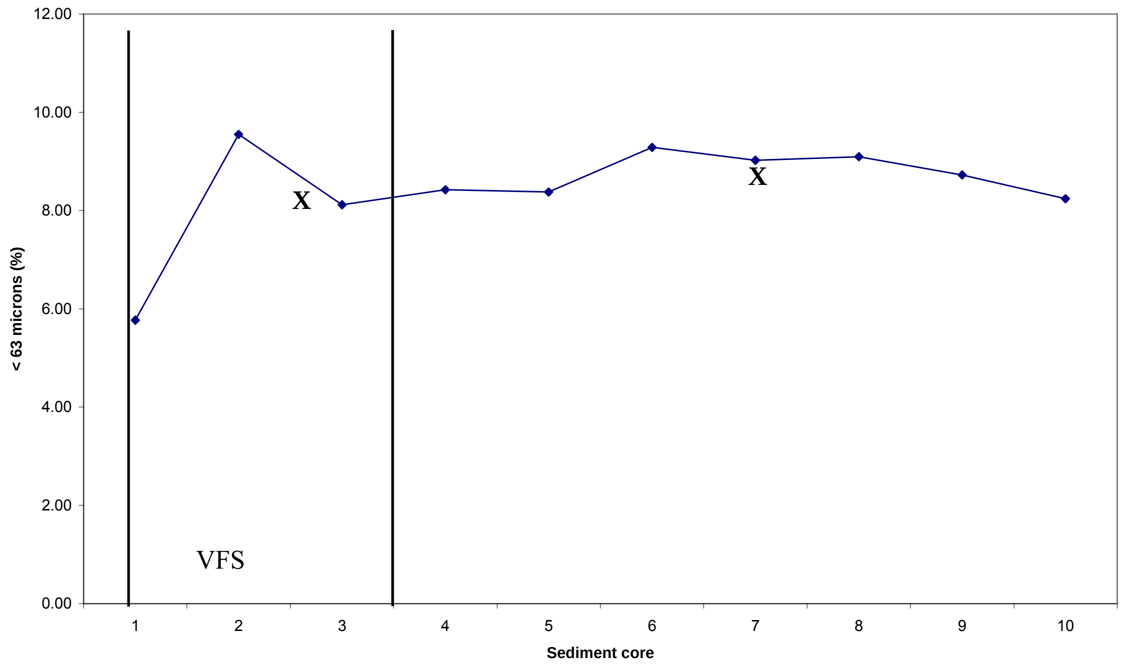



**Figure 5.47 Site 9: less than 63 micron fraction (i.e. silt and clay) for 10 sediment cores taken through the VFS and lower section of upslope field. X marks the locations where depth profiles were taken.**

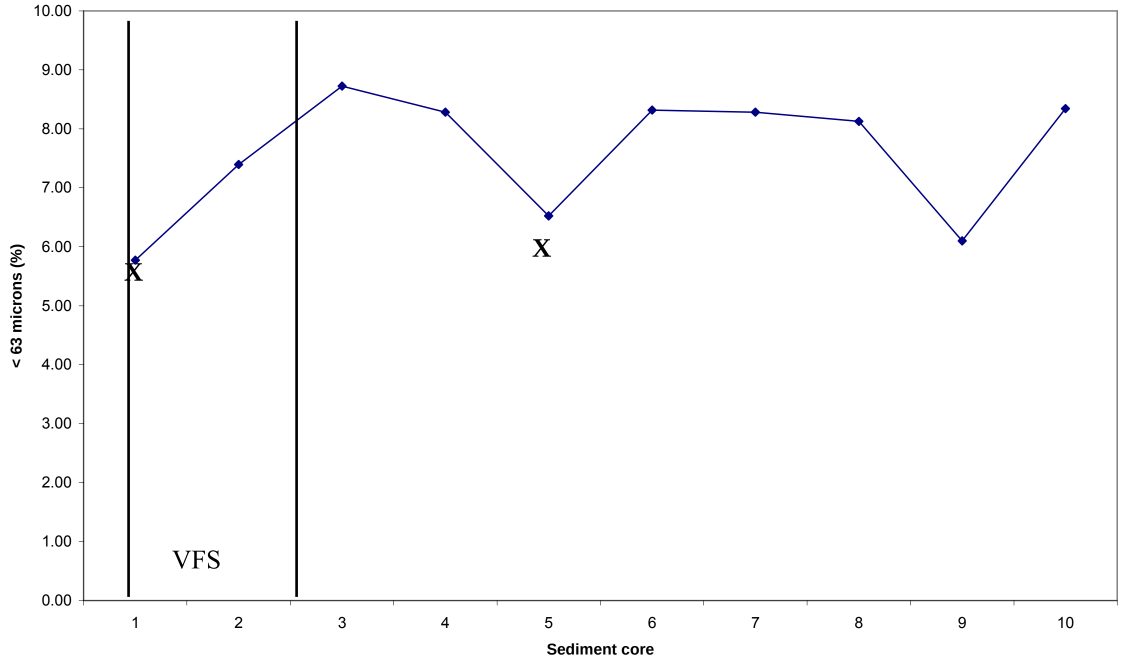



**Figure 5.48 Site 10: less than 63 micron fraction (i.e. silt and clay) for 10 sediment cores taken through the VFS and lower section of upslope field. X marks the locations where depth profiles were taken.**

### 5.4.6 Vegetation parameters

Table 5.14 shows the vegetation parameters recorded at each site and the results of a regression analysis performed to determine the influence of each parameter on sediment deposition at the site. The ground cover, canopy cover and height measurements are average values for the full sampling period. Stem diameter and number of stems were only recorded in July 2005. Figure 5.49 and Figure 5.50 show the average vegetation height, ground and canopy cover for all sites per month sampled.

| *Site* | *Ground cover (%)* | *Canopy cover (%)* | *Height (m)* | *Stem diameter (mm)** | *Number of stems (stems per $m^2$)** |
|---|---|---|---|---|---|
| 1 | 46 | 46 | 0.08 | - | - |
| 2 | 90 | 90 | 0.29 | 1.20 | 5728 |
| 3 | 90 | 82 | 0.11 | 0.87 | 15312 |
| 4 | 68 | 74 | 0.19 | 25.75 | 712 |
| 5 | 82 | 86 | 0.32 | 13.99 | 147.3 |
| 6 | 82 | 82 | 0.2 | 1.00 | 10128 |
| 7 | 82 | 82 | 0.18 | 1.00 | 6400 |
| 8 | 86 | 70 | 0.37 | 1.00 | 7776 |
| 9 | 82 | 86 | 0.23 | - | - |
| 10 | 74 | 78 | 0.21 | 23.68 | 1120 |
| 11 | 86 | 82 | 0.20 | 2.50 | 1136 |
| 12 | 86 | 78 | 0.20 | 1.20 | 1040 |
| 13 | 86 | 86 | 0.21 | 2.10 | 73 |
| 14 | 86 | 86 | 0.21 | - | - |
| 15 | 90 | 82 | 0.40 | - | - |
| *Pearson correlation coefficient* | *-0.371* | *-0.238* | *-0.191* | *0.380* | *-0.366* |
| *P value* | *0.325* | *0.537* | *0.623* | *0.401* | *0.419* |
| *$R^2$* | *0.1379* | *0.0568* | *0.0364* | *0.1441* | *0.134* |
| *Significant* | *No* | *No* | *No* | *No* | *No* |

* Only measured in July 2005

**Table 5.14 Average vegetation parameters for all sampling visits and results to regression analysis.**

Regression analysis showed that none of the vegetation parameters tested have a significant effect on the sediment deposition ($p > 0.05$). Together the parameters account for 69.3% of the variance of sediment deposition. Ground cover, canopy cover, height and stem number show a non-significant negative correlation with sediment deposition and stem diameter shows a non-significant positive correlation with sediment deposition. This is not surprising given the limited variation over time and between the sites, particularly for ground cover and canopy cover.

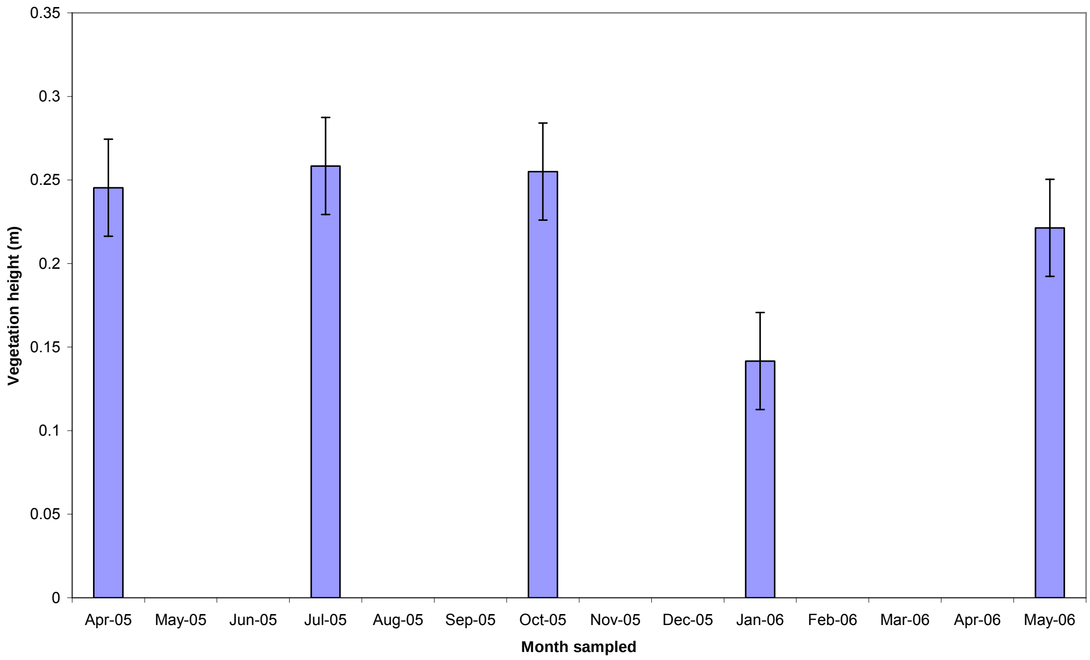



**Figure 5.49 Average vegetation height for all nine sites per month sampled.**

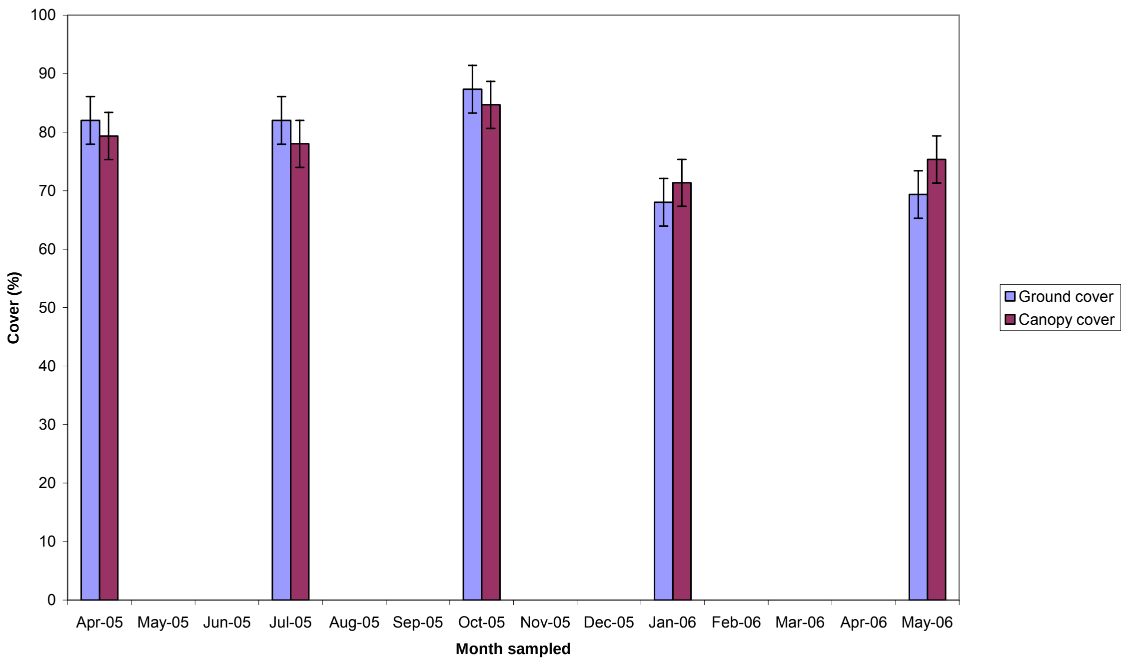



**Figure 5.50 Average ground and canopy cover per month sampled.**

## *5.5 Discussion*

The results collected during the 18 month sampling period inform on the sediment trapping capability of fifteen established VFSs in the Parrett catchment. Observations made throughout the sampling period indicate that a number of characteristics may enhance or compromise buffering capability. These include management, rainfall, terrain, in-field conservation, land use and soil type. The features are more complex than those reported in the literature on uniform simulated plots. Sampling and analysis of sediment deposited within the features showed a preferential trapping of coarse material, with typically a greater than 85% sand fraction, and a surprising increase in clay content in front of the buffer.

### 5.5.1 VFS effectiveness

The VFSs monitored for the study had all been established under Agri-environment schemes, as described in Chapter 2. The most common scheme was the Countryside Stewardship Scheme which included in its aims:

- Sustaining the beauty and diversity of the landscape
- Improving and extending wildlife habitats
- Conserving archaeological sites and historic features
- Improving opportunities for enjoying the countryside
- Restoring neglected land or features
- Creating new habitats and landscapes

The results of the field study show that the grass strip and hedge features installed under this, and similar schemes, have a further function; the potential to trap sediment and hence reduce diffuse pollution. This will, however, depend on the appropriate design and location of the feature.

There was a large variation in the amount of sediment trapped by the VFSs. The average amount of sediment collected during the 18 month sampling period ranged, at sites where sediment was collected, from 0.11 g $cm^{-2}$ (SE 0.04 g $cm^{-2}$) at Site 6 to 2.55 g $cm^{-2}$ (SE 0.04 g $cm^{-2}$) at Site 9. Taking into account all of the mats installed, across all nine sites, the average deposition recorded within the VFS features is 0.29 g $cm^{-2}$ $year^{-1}$. Based on a 6 m VFS this equates to 1.74 t $year^{-1}$ of material trapped from a field of 1 ha (i.e. 100 m x 100 m). This value was used to calculate the percentage of sediment trapped for a range of soil loss values reported for arable fields in the UK. The values are presented in Table 5.15 and suggest that a VFS of this size will trap between 3 and 1740% of the soil lost from a 1 ha field, assuming that all of the material leaving the field passes through the VFS. Values greater than 100% indicate a capacity to trap more sediment than is supplied, i.e. no need for a VFS.

| *Location* | *Soil type* | *Annual soil loss ($t\ ha^{-1}\ yr^{-1}$)* | *Author* | *Percentage trapped by VFS*[1] |
|---|---|---|---|---|
| Bedfordshire | Chalky | 0.6 - 21 | Morgan (1985) | 8 - 290 |
| Bedfordshire | Sandy loam | 0.6 - 24 | Morgan (1985) | 7 - 290 |
| Bedfordshire | Clayey | 0.3 - 0.5 | Morgan (1985) | 348 - 580 |
| Cambridgeshire/ Bedfordshire | Clay | 0.7 - 3.3 | Evans and Cook (1988) | 53 - 249 |
| Dorset | Clay | 0.9 - 11.8 | Evans and Cook (1988) | 15 - 193 |
| Hereford | Medium silt | 10 - 28.6 | Evans and Cook (1988) | 6 - 17 |
| Gwent | Medium silt | 2.2 - 13.8 | Evans and Cook (1988) | 13 - 79 |
| Shropshire | Sand/loam | 2.6 - 47.8 | Evans and Cook (1988) | 4 - 67 |
| Staffordshire/ Worcestershire | Sand/loam | 0.5 - 55.3 | Evans and Cook (1988) | 3 - 348 |
| Somerset | Light silt | 4.9 - 6.4 | Evans and Cook (1988) | 27 - 36 |
| Somerset | Light silt | 0.1 - 5.5 | Colborne and Staines (1986) | 32 - 1740 |
| Dorset | Clayey | 0 - 19.5 | Colborne and Staines (1986) | > 9 |
| Shropshire | Sand/loam | 8 - 13 | Reed (1986) | 13 - 22 |

[1] Based on all of the soil loss from a 1 ha field passing through a 6 m by 100 m VFS.

**Table 5.15 Calculated percentage sediment trapped by VFS based on a deposition rate of 0.29 g $cm^{-2}\ yr^{-1}$. Typical soil loss values reported for arable fields in the UK: list of studies compiled by Walling (1990).**

Observations during the study and information provided by farmers and landowners supplemented the data collected on the effectiveness of the VFSs. For example, prior to the establishment of the VFS at Site 6 the farmer had observed the formation of gullies and a build up of deposited sediment at the bottom of the field. This had not occurred following installation of the VFS under the Countryside Stewardship Scheme. No severe erosion was observed at this site during sampling. Figure 5.17 shows that it had a lower total sediment deposition than the VFSs at the other sites. However, this may be partly due to the soil type. It is not possible to compare this site with another of the same soil type because Site 5 did not yield any sediment deposition. Similarly, at Sites 2 and 4 the farmers and local residents had regularly witnessed sediment washing onto the road downslope of the field prior to establishment of the VFS. No sediment movement was observed during the study period but at Site 2 this may have been due to a change in land use to grassland.

In July 2005 the grass strip at Site 9 was severely breached and sediment was deposited from the front to the back of the VFS to a depth of 210 mm. It was evident that the hedge at the lower edge of the grass strip had prevented the bulk of the sediment from traveling any further downslope. Although the continuity of the base of the hedge was broken by gaps between stems, its height and resistance to bending, compared with the grass, was effective in trapping the coarser material.

## 5.5.2 Characteristics of the trapped material

The results demonstrate that VFSs primarily trap sand size material which is coarser than the topsoil in the contributing upslope fields. This might imply that the finer fraction is not deposited in a VFS but passes through to be delivered to water courses. However, the depth profiles suggest that the sediment in the VFSs was coarser further down from the soil surface. A possible explanation for this is earthworm activity. Under natural conditions, the seasonal activity of burrowing animals gives rise to considerable disturbance of the soil, with earthworms bringing upwards as much as 2 to 5.8 t/ha of fine material on agricultural land (Evans, 1948).

It may also be because fine material is deposited on the VFS surface and then washed downslope by further rainfall events i.e. the fine material may be deposited but not retained. Although no evidence was collected to support this it seems reasonable for coarse material to be preferentially deposited from moving water, as velocity slows, but for fine material to be deposited where water ponds (Figure 5.39).

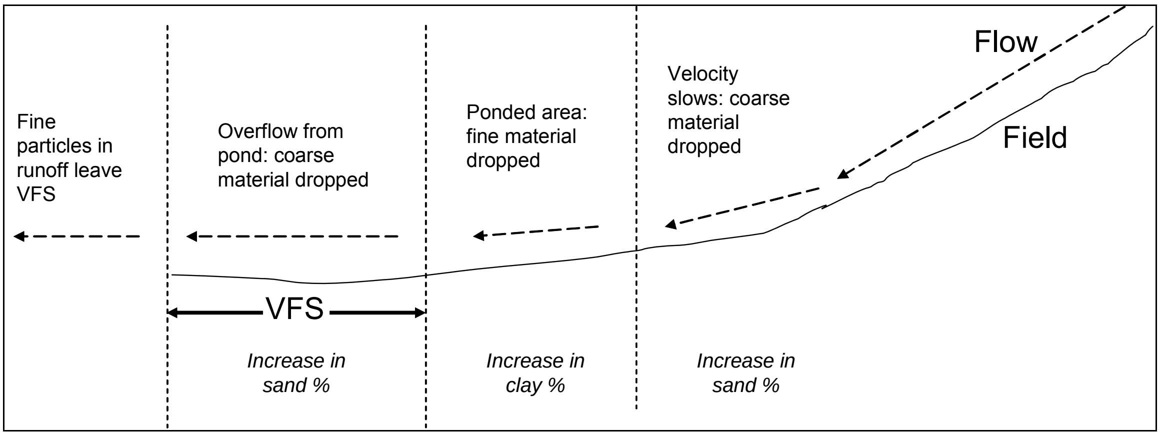



**Figure 5.51 Schematic illustration of preferential deposition of coarse material from moving water and fine material from ponded water.**

Furthermore, no fine material was collected from the mats downslope or at the back of the VFSs, which suggests that the fine material is deposited somewhere before it reaches the back of the VFS. The soil samples taken along transects indicate that this material may be deposited along the front edge of the VFS and, where ponding occurs, in front of the VFS. This should be investigated further with a denser sampling strategy. Deposition at the front of the VFS is consistent with Neibling and Alberts (1979) who observed that the majority of sediment deposition occurs just upslope and within the first metre of the VFS with subsequent deposition moving through the VFS in a wedge shape. This pattern has been observed by others (Chapter 2) but the particle size of the material has not been compared at different positions within VFSs.

For those sites where sediment samples were collected, approximately 50% were collected from the front of the VFSs, with 20% and 30% being collected from the middle and back of the VFS locations. This indicates that most of the sediment delivered to a VFS is retained within the first few metres. However, within the VFSs monitored (typically 2 to 9 m width) sediment was also trapped throughout the full VFS length on some occasions. For Site 4, where enough samples were collected to enable some preliminary comparison (Table 5.12) the mats located at the front of the VFS collected significantly more sediment than those located at the back.

Findings suggest that where VFSs intercept water and sediment pathways they are effective in trapping sediment. In most cases the majority of coarse sediment is trapped at the upslope edge of the VFS. However, when the VFS is severely breached, due to gully formation within the contributing upslope field or intense rainfall, sediment can travel much further across the VFS. In these cases, and for finer sediment, the length of the VFS will influence its effectiveness, although research has indicated that there is likely to be some point at which a further increase in length becomes less effective (Abu-Zreig, 2001).

## 5.5.3 VFS performance variables

### 5.5.3.1 Rainfall

Observations made during site visits, and the use of the evaluation form, suggest that sedimentation in the VFSs was associated with either individual or a small number of erosion events. Figure 5.32 to Figure 5.38 show the association of sediment deposition with rainfall. It is clear that the maximum sediment deposited at Sites 1, 2 and 6 occurred following the periods of most rainfall between October 2005 and January 2006. In this case the highest rainfall coincided with the season when the VFS vegetation had the lowest height and density and therefore potentially offered lower sediment filtering capability.

### 5.5.3.2 Vegetation

Stem diameter and number of stems varied between the nine field sites. Vegetation parameters such as height and ground and canopy cover varied between sites and with seasons. A dense grass cover appeared to be effective in filtering sediment except when the vegetation became inundated by sediment. Generally hedge vegetation was too sparse to trap sediment. However, at Site 9 when the VFS was severely breached the hedge downslope of the VFS prevented any mass transfer of sediment through the back of the VFS. No conclusions can be drawn on the influence of vegetation parameters on sediment deposition at the sites because the statistical tests performed showed no significant link between the measurements. However, this is likely to be due to the generally high percentage covers and low variability recorded between the sites and seasons. A more dense sampling strategy for the vegetation parameters, both in time and space, may reveal stronger relationships.

### 5.5.3.3 Soil type, slope and land use

Some sites were less effective at trapping sediment due to inherent variations in limiting factors such as soil type, slope and land use. The greatest rate of sediment deposition in a VFS was at Site 9 (1.43 ± 0.70 g $cm^{-2}$ $yr^{-1}$), which had sandy-clay loam soils, moderate to steep slopes ($4^o$) and maize crops. Similar values were obtained for Sites 4 (0.98 ± 0.69 g $cm^{-2}$ $yr^{-1}$) and 7 (1.23 ± 0.40 g $cm^{-2}$ $yr^{-1}$) which also had the greatest slopes (both $5^o$), sandy loam to sandy clay loam soils and wheat crops. Sites 4 and 7 also showed the most signs of erosion in the contributing fields. In contrast, Sites 3 (sandy clay loam, grassland and $2^o$ slope) and 5 (potatoes, $3^o$ slope) recorded no sediment deposition. In the case of Site 5 this mainly reflects the influence of burrowing animals creating subsurface pathways for runoff and mobilised sediment under the grass VFS. Site 6 had only minimal sediment deposition (0.02 ± 0.01 g $cm^{-2}$ $yr^{-1}$) and had silty-clay loam soils, gentle slopes and barley-wheat crops. Therefore, at the field sites, factors which determined the flow to the VFS appeared to be more important than soil type, slope and land use. These factors may have shown a greater influence had a wider range of conditions been tested. It is likely that guidance based on these factors may be applicable to the farm or catchment scale, but at the site specific scale, factors determining whether flow is uniform or concentrated will be of more importance.

### 5.5.3.4 Management

Specific land management operations may also influence VFS effectiveness. Figure 5.15Figure 5.16 shows deposition at the interface between the field and VFS at Site 4 due to a “step” created by tillage. Only when this difference in height is in-filled by sedimentation can the VFS (i.e. the vegetation) perform a sediment trapping function. The same process was observed at Sites 7 and 5. At the latter site the difference in surface height between the field and VFS was > 0.45 m. Additionally, at the lowest point along the VFS at Site 5 an animal burrow lowered the field level by a further 0.5 m. This provided a potential flow route beneath the VFS surface. (Figure 5.15 and Figure 5.17.)

At Site 4 and Site 7 field corners had been used to turn vehicles and this resulted in bare patches and compaction. Consequently these areas became inundated with sediment. VFSs were also used as routes for vehicles and this caused tracks of flattened grass and compaction. However, it is unlikely that this will affect sediment trapping so long as the vegetation and soil damage are not perpendicular to the VFS.

### 5.5.3.5 In-field erosion and conservation

Rills and gullies in the contributing upslope fields were a significant source of sediment to the VFSs (Figure 5.18). Breach points in the VFS were mostly located at the base of such erosion features due to the increased sediment transport capacity of the concentrated flow. Concentrated flow and erosion features and were caused by converging slopes (Sites 4, 5 and 7) and by crop rows (Site 9). In-field conservation measures enhanced VFS performance. At Sites 4 and Site 7 the establishment of a grass waterway along areas of previous erosion reduced the delivery of sediment to the instrumented VFSs. This prevented the VFS vegetation from becoming inundated and thus unable to perform its sediment filtering role. The decrease in sediment deposition within the VFSs following installation of a grass waterway can be seen in Figure 5.34 and in Figure 5.36.

## 5.5.4 Contributions to understanding VFS performance

The work described in this chapter is unique in investigating established VFSs in a UK catchment. It adds to previously reported plot scale studies (e.g. Dosskey et al., 2007 and Blanco-Canqui et al., 2006) by taking into account a range of variables at the field scale. Dillaha et al. (1989) and Dosskey et al. (2002) suggest that VFS performance may be compromised by concentrated flow. This appears to be confirmed at Sites 4, 7 and 9 where sediment was delivered to the VFSs by converging flow perpendicular to the VFS. At these locations sediment submerged the filter strip vegetation enabling sediment to be transported further downslope. Conversely, when rills were not present in the contributing areas of these sites, sediment did not travel as far across the VFSs. Linear VFS features like grass strips and hedges are only likely to be effective in trapping sediment at limited locations, defined by topography, hydrological pathways and the management and maintenance of the VFS. The work of others is confirmed (e.g. Haycock et al., 1997, Dabney et al., 2006) showing that targeted location and careful maintenance of VFS features, at key locations where flow converges, may be more effective than the widespread distribution of VFSs. The work points towards practices that may enhance VFS performance, e.g. in-field conservation, maintaining a dense VFS during seasons where erosion risk may be greatest, ensuring that sediment and overland flow is directed from the field to the VFS and does not meet a change in ground level or an alternative transfer route. Furthermore, it was found that a number of naturally occurring features in the landscape may have the potential to become “accidental” VFSs. Areas of dense vegetation and changes in ground level occur naturally and will intercept flow pathways and may cause sediment to be deposited in front or within them.

### 5.5.4.1 Length

Nine of the fifteen VFSs tested measured 6 m in the direction of flow. Of all of the 60 mats that trapped sediment over the 18 month period only 30% were located at positions at the back of the VFSs. This suggests that 70% of the sediment transported through the VFSs was deposited in the first 5 m. However, where concentrated flow occurs it is possible for a 6 m VFS to become breached, e.g. Site 7 and Site 9. At Site 9 any further downslope movement of sediment was prevented by a hedge at the back of the grass strip. This combination may be more efficient for sediment trapping of large losses than using longer grass strips which take more land out of production. This is especially true if, as suggested by the sediment core material, finer material is deposited at the front of the VFS.

#### 5.5.4.2 Slope

The VFSs observed were all on, and contributed to by, slopes less than 5°. Regardless of average slope angle, failure points occurred along sections of the VFSs where flow converged creating concentrated flow perpendicular to the VFS. For the VFSs observed in the study, no conclusions can be drawn between slope angle and VFS effectiveness. It was observed however that VFSs may be effective on slopes up to 5° and that terrain influences the transport of sediment to a VFS and may compromise VFS performance. More studies are needed to separate out the effect of slope of the land containing the VFS and slope of the land contributing runoff and sediment to the VFS.

**Figure 5.52 Breach points in the VFS at Site 9 caused by concentrated flow.**

### 5.5.5 Appraisal of field methods

Seven of the nine VFSs instrumented collected sediment during the 18 month sampling period. The lack of sediment at Sites 3 and 5 may be attributed to specific site variables such as land cover, a difference in ground level between the field and VFS and animal burrows. However, at the remaining sites only 39 out of 104 mats collected sediment. This suggests that either erosion at those sites was minimal or that overland flow by-passed the mats. Both of these effects were observed during the sampling period, confirming that much of the water and sediment flow through linear VFSs is concentrated in relatively narrow areas and over specific periods of time. The use of Astroturf mats may therefore be limited unless they are carefully located using appropriate tools. Even then the “best” location is likely to alter over time with changes in vegetation and the soil surface. A greater number of mats will increase the probability of trapping more sediment but this must be balanced against disturbance of the VFS and related processes.

In general the Astroturf mats proved to be an effective and least-intrusive method for collecting deposited sediment within grass VFSs. The mats were cheap to obtain, easy to install and could be left for long periods of time and easily removed for analysis when required. The rough surface trapped and retained sediment when placed flush with the VFS surface. One assumption that should be noted, but is easily overcome, is that all of the sediment deposited on a mat is transported via overland flow. In some cases sediment was moved onto the mats by field operations or vehicles/walkers passing over the mats and so careful observation of the sediment and surrounding processes is necessary prior to removal of the mat for analysis.

Use of the Astroturf mats did not inform on the material that moved through the VFSs. Use of surface water traps to collect samples of overland flow proved to be unsuccessful and this is a limitation of the sampling design. Results presented suggest that VFSs may be effective in trapping coarser material but no conclusions can be made on the material that was not deposited and continued to be transported further downslope. This is likely to include the finer sediment fractions.

### 5.5.6 Further study

The sampling period represents a relatively short time in terms of the lifespan of the VFSs. Sampling over several years would indicate whether VFSs are likely to become saturated with sediment and therefore a source rather than a sink for pollutants such as phosphorus. However, many of the VFSs had been established for several years and had not been inundated with sediment. Previous history, as well as the monitoring period, suggests that the VFSs have been effective except where breached by concentrated flow.

Monitoring of the grass waterways at Sites 4 and 7 would quantify their effect on the VFS performance. It is possible that, over time, the edges of the grass waterway may become channels for concentrated flow or that the rills and gullies may reform if the grass becomes inundated with sediment (Figure 5.53). In any case long term VFS studies are extremely scarce within the literature. In order to understand the sustainable use of VFSs for controlling diffuse pollution it is essential that this is carried out, with different land use scenarios, and ideally linked to simultaneous water quality monitoring.

**Figure 5.53 Sedimentation areas forming in the grass waterway at Site 7.**

**Figure 5.54 Bare soil at the edge of the grass waterway at Site 7.**

The current study has shown that differences exist in the vegetation characteristics of established VFSs, that these characteristics vary with changes in season and that they are likely to influence VFS performance. Due to the time available for monitoring, a limited number of quadrats were taken within each VFS at approximately three monthly intervals. Sampling at shorter intervals and taking more measurements may enable the characterization of the VFSs by a set of “ideal” vegetation characteristics which could be matched with optimum species. The role of soil fauna in the VFS could be investigated, such as animal burrows and bioturbation.

## 5.6 Conclusions and implications

The results discussed indicate that the features investigated in the Parrett catchment are effective VFSs for sediment control. Coarse sediment is trapped mainly within the first few metres of an established linear vegetated strip. It is possible that the finer sediment fraction passes through the VFSs and is delivered to surface waters, which has implications for surface water pollution by sediment associated pollutants such as soluble phosphorus. However, the main location for the deposition of fine sediment at the sites monitored was in front of the upslope edge of the VFS. An explanation posed for this is the deposition of fine material during ponding at the VFS-field interface caused by a decrease in flow velocity. This would suggest that optimum VFS designs should promote ponding, but further monitoring is required to confirm this trend and to determine how long the deposited fines may stay at this location following further rainfall events.

Soil, slope, land use, management and vegetation factors influence the amount of sediment that is mobilised and delivered to VFSs. This suggests that some VFSs may be required in some situations (e.g. downslope of fields with steep, erodible soils and periods of bare soil) and not in others (e.g. fields with gentle slopes, permanent grass cover and low erosion potential) (Owens et al., 2007). There is evidence in the study for an increase in sediment deposition with size of rainfall event. Further work should be carried out to investigate the threshold size of the event at which VFSs will fail.

A number of the VFSs were breached at "failure points". This is a problem where flow is concentrated and will limit the locations where linear features like grass strips and hedges are likely to be effective in trapping sediment. Strategic placement of VFSs and reinforcement should therefore be targeted. VFS performance may also be compromised by land use and management activities such as the creation of steps between fields and VFSs and the creation of sub-surface pathways by burrowing animals.

The effectiveness of VFSs for sediment associated pollutants in agricultural landscapes, such as pesticides, pathogens, metals, radionuclides and organic pollutants depends on their ability to trap and retain fine material. Therefore, in order to control the delivery of sediment and associated contaminants to watercourses, the strategic placement and appropriate design of VFS features is vital. This can most effectively be achieved through a comprehensive understanding of the sources, mobilization and transport pathways by which pollutants are delivered to watercourses and by a critical evaluation of any diffuse pollutant mitigation efforts.

The results discussed in this chapter provide data against which the model is tested in Chapter 9 and with which guidance tools are developed in Chapter 10. The next two chapters describe the selection and initial testing of a model for this purpose.

# Chapter 6 Erosion prediction model for evaluating VFS behaviour

## 6.1 Introduction

Chapter 1 proposed that a simple soil erosion prediction model can be used to predict the sediment trapping efficiency of laboratory simulated and field VFSs, to simulate the influence of different plant stem diameters on the sediment trapping efficiency of VFSs, and to derive the optimum design and location for VFSs. This chapter describes the selection of a suitable soil loss model. The concept of soil erosion modelling is first summarised and examples of published VFS studies utilising models are reviewed. The model selected for the current study is then presented and reasons are given for its suitability to the current work.

## 6.2 Modelling soil erosion

"Research on soil erosion and conservation is assisted by the use of a variety of models, which differ in their context, purpose, and degree of detail" (Rose et al., 2003). Researchers commonly use erosion prediction methods to inventory erosion for national assessments of the impact of erosion on crop productivity, off-site sedimentation and non-point source pollution. Erosion prediction methods are also useful tools in selecting conservation measures for specific fields and are convenient tools for extrapolating information where specific field situations have not been studied (Foster, 1987). There are a number of models available and they can be categorised in various ways. Before selecting a model for a specific purpose it is important to understand the advantages and limitations that the different types may offer. Models can be classified in different ways and so some key definitions and differences between model types are presented below. For a comprehensive review of erosion and sediment transport models see Merritt et al. (2003).

### 6.2.1 Deterministic or stochastic approach

Most models use deterministic representations and inputs (Grayson and Blöschl, 2001). This means that they have a single set of input parameters and values which are used to generate a single set of output. By contrast, in stochastic models, some or all of the inputs and parameters are represented by statistical distributions rather than single values. In the hydrological literature stochastic is generally used synonymously with "statistical" and suggests a random component to the model (Grayson and Blöschl, 2001).

### 6.2.2 Empirical models

Empirical models are primarily based on the analysis of observations and rely on the observed relationships between model inputs and output. An example is $runoff = a(rainfall)^b$ where $a$ and $b$ are derived via a regression between measured rainfall and runoff (Grayson and Blöschl, 2001). They generally have low computational and data requirements and integrate a small number of causal variables. They are useful where data and parameter inputs are scarce. However, such models may be limited by assumptions about the catchment. For example, they may not reflect the spatial variability of factors such as rainfall and soil type or the temporal variability in factors which may alter during the study period. They also tend not to be event-responsive. Empirical models do not inform on why erosion occurs but help to identify sources of sediment and nutrient generation. Generally, such models cannot be extrapolated to conditions beyond the range of the data used in their formulation.

### 6.2.3 Conceptual models

Conceptual models provide a general description of catchment processes and represent flow paths as a series of internal storages. The main difference between these and empirical models is that they reflect hypotheses about the processes at work and so may be used to provide an indication of the qualitative and quantitative effects of land use changes (Merritt et al., 2003). Although the parameters are, in principal, measurable physical quantities, in practice they can be difficult to determine (Grayson and Blöschl, 2001). Again the data requirements are fairly low.

### 6.2.4 Physically-based models

Physically-based models are based on the solution of mathematical equations to describe streamflow and sediment generation, incorporating the laws of conservation of mass and energy. They provide greater potential than empirical and conceptual models for predicting the spatial distribution of runoff and sediment over the land surface and for dealing with individual storms (Morgan, 2005). However, the assumptions underlying the mathematical equations may not necessarily be universally applicable. For example, it is not unusual for equations to be derived from small scale experiments and point source measurements and then applied to much larger scales and to continuous spatial and temporal data (Merritt et al., 2003). Yet, little evidence exists to support the viability of this up-scaling approach.

### 6.2.5 Distributed and lumped models

Models can be categorised according to how they represent the area to which the model is applied. Lumped models treat input parameters as aggregated over the area of analysis whilst distributed models reflect the spatial variability of the processes and outputs in a catchment (Merritt et al., 2003). The latter approach has advantages for decision making in catchment management such as the locating of diffuse pollution control measures. However, the data requirements for this approach are greater than for lumped models and require careful selection of cell resolution. Semi-distributed models provide a compromise by breaking the catchment into a group of subcatchments or other biophysical elements over which the model can be applied.

### 6.2.6 Event-based and continuous models

Event-based models are developed to provide an estimate of erosion by individual storms (Merritt et al., 2003). The output may be for an event of minutes or hours in duration. Alternatively models may be developed to reflect broad trends over time. In this case the output may be annual and may be used to explore changes in rainfall, land cover or land management. A further approach is to use a continuous time period broken into time steps which are responsive to processes that can be captured. Time steps are often daily, for example the development and recession of a saturated zone.

### 6.2.7 Selecting a modelling approach

In practice, there is a lot of overlap between the above categories of models, for example, event based models may be distributed or lumped and each of these may be empirical or physically based. Characteristically, erosion prediction methods are extrapolated beyond the range of the data used to derive them. The ability of a method to perform well when extrapolated is an important factor in the selection of a prediction method, especially where availability of baseline data is limited (Foster, 1988).

## *6.3 Applying models to VFS design and placement*

The main functions of a VFS were outlined in Chapter 2 and include enhancing infiltration, increasing sedimentation, increasing filtration of suspended material and increasing both the adsorption and absorption of solutes by vegetation. The usefulness of a soil erosion model in evaluating VFS performance will therefore depend on its ability to simulate these processes and the vegetation properties governing them. The following discussion provides a review of the representation of VFSs by soil erosion and sediment models. Particular attention is given to a) determining which processes are simulated, b) evaluating the role of vegetation and c) establishing how effective the models are in meeting their objectives when applied to VFS modelling. This provides the basis for selecting a model suitable for the current study.

### 6.3.1 Manning’s n

One approach to modelling VFSs is based on the fact that the stems and leaves of a plant reduce the velocity of runoff by the roughness that they impart on the flow. Roughness can be characterised by a parameter such as Manning's *n* and depends on the morphology of the plant and its density of growth, as well as the height of the vegetation in relation to the depth of flow (Coppin and Richards, 1990). Surface roughness is inversely related to both the velocity and quantity of runoff, as expressed by the following equations (Styczen and Morgan, 1995):

$$u = R^{0.667} S^{0.5} / n \qquad \textbf{Equation 6.1}$$

and

$$Q = R^{1.667} S^{0.5} / n \qquad \textbf{Equation 6.2}$$

where $u$ is the velocity of flow (m/s); $R$ is the hydraulic radius (m); $S$ is slope (m/m); $n$ is Manning's roughness coefficient and $Q$ is the quantity of runoff ($m^3$/m/s).

Difficulties with using this approach include assigning Manning's *n* values given the large range of values. For example, *n* values for grass range from 0.2 to 0.4 (Styczen and Morgan, 1995). The range reflects differences in plant height and density. Variation in values is also caused by differences in flow depth. During shallow flow the vegetation stands relatively rigid and roughness values are approximately 0.25 to 0.3 due to the distortion of the flow around the individual plant stems. With increasing flow depth, stems start to oscillate and the increased disturbance to flow increases values to around 0.4. When flow depth submerges the vegetation roughness values will decrease up to an order of magnitude because plants are flattened during the flow (Styczen and Morgan, 1995). Despite this it is used in a number of VFS models.

## 6.3.2 Riparian Buffer Delineation Equation

Phillips (1989a, 1989b) derived two equations to describe VFS performance, both of which calculate the effectiveness of a given buffer ($B_b$) to that of a reference buffer ($B_r$). The first, the Hydraulic Model, relates pollutant transport through a VFS to the energy of overland flow and is given by:

$$\frac{B_b}{B_r} = \left(\frac{n_b}{n_r}\right)^{0.6} \left(\frac{L_b}{L_r}\right)^{0.4} \left(\frac{K_b}{K_r}\right)^{0.4} \left(\frac{s_b}{s_r}\right)^{-1.3} \qquad \textbf{Equation 6.3}$$

where $K$ is the saturated hydraulic conductivity, $L$ is the length of the reach, $s$ is the sine of the slope angle relative to the horizontal and $n$ is the Manning roughness coefficient. The second, the Detention Time Model, assumes that VFS effectiveness is a function of the total contact time of both surface runoff and throughflow and is based on Darcy's law and Manning's equation. This equation is given by:

$$\frac{B_b}{B_r} = \left(\frac{n_b}{n_r}\right)^{0.6} \left(\frac{L_b}{L_r}\right)^{2} \left(\frac{K_b}{K_r}\right)^{0.4} \left(\frac{s_b}{s_r}\right)^{-0.7} \left(\frac{C_b}{C_r}\right)$$ **Equation 6.4**

where $C$ is the soil moisture storage capacity and the other terms are as previously defined.

It is not possible to assess the performance of the models because, as noted by Muscutt et al. (1993), no experimental verification was performed, nor were they field tested or calibrated. A further limitation of the approach is that the effectiveness of the equations relies on the reference buffer selected for comparison. Rather than real reference sites the parameters that Phillips used for reference buffers were based on typical recorded values from the regions under study. Indeed Phillips refers to some of his values as "somewhat arbitrary" (Phillips, 1989b). However, the model is still regarded by a number of researchers as a useful starting point for modeling buffer width and effectiveness (e.g. Wenger, 1999; Xiang, 1993, 1996; Xiang and Stratton, 1996).

### 6.3.3 Two models to determine VFS length

Nieswand et al. (1990) developed a model for determining appropriate VFS length (in the direction of flow) which rests on slope being the most important factor influencing VFS effectiveness:

$$W = k\left(s^{0.5}\right)$$ **Equation 6.5**

Here $W$ is the length of the VFS, $k$ is 15 m (constant), $s$ is the percent slope expressed as a whole number (e.g. 5% slope would be 5). The constant "15 m" is chosen based on common VFS recommendations, with the assumption that a 15 m VFS at 1% slope provides adequate protection to streams. The authors recommend that slopes greater than 16% and impervious surfaces are ineffective and should not be used in VFS width calculations.

A similar approach was taken by Mander et al. (1997) who proposed:

$$P = \left(tqfi^{0.5}\right) / \left(mK_i c\right)$$ **Equation 6.6**

where *P* is VFS length, *t* is a conversion constant (0.00069), *q* is the mean intensity of overland flow, *f* is either the distance between stream and watershed boundary or the ratio of catchment area to stream segment length, *i* is the slope, *m* is a roughness coefficient (not Manning's), $K_i$ is the water infiltration rate and *c* (is the soil adsorption capacity. There does not appear to be any validation of either model, and the 15 m constant selected by Nieswand et al. (1990) may be regarded as fairly arbitrary.

### 6.3.4 GRASSF

Researchers at the University of Kentucky (Tollner et al., 1977 and Barfield et al., 1979) developed and tested a model for filtration of suspended solids by artificial grass media. The model considers that vegetation, represented by Manning's *n*, causes an increase in hydraulic resistance that slows the flow of runoff. The model also represents the reduction of transport capacity and the resulting deposition of coarse material. The sediment trapped in this first part of the filter forms a wedge that moves downslope and varies depending on the depth of the sediment and the height of the vegetation. Media spacing is incorporated by distance apart in *x* and *y* directions (i.e. not diagonally) in units of length.

It should be noted that, for the measurements that provided the basis for the model development, metal rods and beads were used in place of grass stems and sediment. An attempt by Hayes et al. (1984) to test and validate the model on natural turf was only partially successful. Laboratory experiments carried out by the University of Aberdeen (Deletic, 1999, 2000) using artificial grass showed inaccuracies in model predictions, particularly for small particles and low concentrations of natural sediment. This was explained by the fact that the sediment used in the Aberdeen experiments was very fine whilst the GRASSF model was developed only for large particles (>27 μm) (Deletic, 2001). Despite this the model has been widely cited and forms the basis for a number of further models e.g. SEDIMOT II and VFSMOD.

### 6.3.5 SEDIMOT II

Wilson et al. (1981) incorporated GRASSF into a hydrology and sediment catchment model named SEDIMOT II. Modifications included the addition of an algorithm which calculates the outflow hydrograph and the ability to consider up to three different slope changes. However, Munoz-Carpena et al. (1999) noted the following limitations. The model does not handle time dependent infiltration, provide an accurate description of flow through the filter or describe changes in flow derived from sediment deposition during the storm event.

### 6.3.6 VFSMOD

Munoz-Carpena et al. (1999) combined the GRASSF algorithm for sediment transport with a hydrology submodel describing overland flow and infiltration. VFSMOD is regarded by Deletic (2000) as "the most advanced model available for assessment of water and sediment runoff through grass". Within the hydrology component soil surface and vegetation cover are represented by Manning's *n*. This is used, with a number of flow parameters including flow depth and flow direction, to calculate discharge per unit width. The overland flow model is coupled, at each time step, with an infiltration submodel that deals with the development of ponded conditions throughout a storm event. The hydrology component can be used to inform on the effect of soil type (infiltration), slope, surface roughness, filter length, storm pattern and field inflow on VFS performance.

The sediment component is described by the GRASSF model. Inputs to this component include a modified vegetation Manning's *n*, vegetation spacing and vegetation height. During model validation Munoz-Carpena (1999) used a Manning's *n* based on recommended values for the vegetation species tested. Vegetation spacing and height were measured at the experimental site. Sensitivity analysis of the model showed that vegetation spacing was the most important parameter controlling sediment outflow, particularly for finer sediment classes. Variations in the modified Manning's *n* and the vegetation height had relatively little effect on the output.

Agreement was obtained between observed and predicted values during field verification and the authors concluded that the model provides better sediment outflow predictions than previous attempts. Deletic (2001) however, noted that the sediment transport component is limited by predicting bulk sediment load and not particle size distribution. This is potentially a disadvantage if the model is to be developed further for assessment of sediment-bound nutrients which relies heavily on accuracy in the prediction of the particle size fraction with which they are associated.

Abu-Zreig (2001) further validated VFSMOD by testing its performance for a range of conditions. However, the influence of different vegetation covers was again limited to variations in Manning's *n*. Lower values represented sparse vegetation conditions whereas higher *n* values represented dense vegetation. Vegetation spacing and height were kept constant.

### 6.3.7 TRAVA

The TRAVA model was developed by researchers at the University of Aberdeen in order to tackle some of the difficulties and limitations of previous attempts at modelling water and sediment transport over grassed areas (Deletic, 2001). Similar to VFSMOD the model comprises two components. The generation of runoff represents infiltration, surface retention and surface runoff and is based on a similar model to that employed by VFSMOD. TRAVA attempts to improve the characterisation of grass roughness by providing both the Darcy-Weisbach and the Manning equation as options. It is unclear, however, how the Darcy-Weisbach actually improves model output. Whilst it may be technically a better descriptor of hydraulic roughness, few values are available as the value needs to be determined experimentally. The surface water retention process was also adapted specifically to take into account the higher retention expected for grassed surfaces compared with impermeable surfaces.

The sediment component aims to improve the assessment of sediment deposition in grass by using the results to experimental findings carried out by Deletic (1999, 2000). Particle deposition is based on the sediment trapping efficiency being a function of the particle fall number $N_{f,s}$, which is calculated as:

$$N_{f,s} = \frac{lV_s}{V} \qquad \textbf{Equation 6.7}$$

where $l$ is the grass length (m), $V_s$ is the Stokes' settling velocity of the particle $d_s$ (m s$^{-1}$) and $V$ is the average mean flow velocity between grass blades (m s$^{-1}$). The process of sediment transport is based on the assumption that sediment is transported as suspended sediment load rather than as bed load. Surface level and slope changes are also accounted for.

The model was successfully verified against a set of field data although the authors suggest that further development of the model is necessary to include modelling of ponding upstream of the filter strip, particle infiltration or resuspension and discharges of pollutants attached to sediment particles.

### 6.3.8 CREAMS

Williams and Nicks (1988) used the CREAMS (Chemical, Runoff, and Erosion from Agricultural Management Systems) field-scale soil erosion model to evaluate the effectiveness of grass filter strips for erosion control. They simulated the filter strips on the basis that vegetal retardance may be expressed in terms of Manning's *n* and that this depends upon the grass variety, growth stage and degree of submergence. They assumed a set of standard conditions; that the grass was fully developed and erect and that flow depth did not exceed the height of the grass. Simulations were then performed for a range of grass densities, represented by Manning's *n* values.

The study found that, as sediment load increases, filter strip width, or grass quality (based on density), or both need to be increased to maintain VFS performance. The simulations also indicated that there is a point beyond which increases in filter strip width have no impact on soil loss from the upslope field. These findings are consistent with those of other studies and it was concluded that CREAMS "can be a useful tool for evaluating filter strip effectiveness in reducing sediment yield". In general, the model overestimated erosion by 38%. These results were not tested against independently collected data. Dillaha and Hayes (1991) suggest that the applicability of CREAMS to sediment transport in VFS is questionable because the model does not simulate the principle processes involved.

### 6.3.9 AGNPS

Tim and Jolly (1994) integrated the AGricultural Non-Point Source (AGNPS) model with a GIS to evaluate the effectiveness of VFSs. The AGricultural Non-Point Source (AGNPS) model is an event-based distributed parameter model based on the concept of the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978) to estimate soil erosion and sediment yield in a watershed. USLE is given by:

$$A = R.K.LS.C.P \qquad \textbf{Equation 6.8}$$

where $A$ is the long term average annual soil loss, $R$ is rainfall, $K$ is a soil erodibility factor, $LS$ is a slope length-gradient factor, $C$ is a crop/vegetation and management factor and $P$ is a support practice factor.

In order to simulate different management strategies, parameters relating to the establishment of VFSs were modified accordingly. This included the USLE $C$ factor, USLE $P$ factor, a surface condition constant and Manning's $n$. The $C$ factor describes, for a certain stage of growth and plant density, the ratio of soil loss when vegetation is present to the amount lost on a bare soil. The $P$ factor reflects the effects of practices that will reduce the amount and rate of the water runoff and thus reduce the amount of erosion.

The study by Tim and Jolly (1994) provides an efficient framework for assessing the effectiveness of different land management strategies in reducing sediment loading to streams. It should be noted that the emphasis of the study was not on characterizing VFS performance but on illustrating the potential advantages of a GIS-based water quality modelling tool. On this basis, it is useful in dealing with catchment wide VFS implementation but it is not possible to determine how well the AGNPS tool describes the way VFSs work.

### 6.3.10 WEPP

The Watershed Erosion Prediction Project (WEPP) is a physically-based model intended to determine and assess the essential mechanisms controlling erosion by water. The processes simulated by the model can be classified into erosional, hydrological, plant growth and residue, water use, hydraulic and soil processes. Plant characteristics are extremely important in accurately describing the soil erosion and hydrological processes at the site (Merritt et al., 2003). These include canopy cover and height, above and below ground biomass of living and dead plant material, leaf area index and basal area and area estimated on a daily basis (Laflen et al., 1991).

The effectiveness of riparian buffers at 200 sites in the US was evaluated by Williams and Nicks (1988) using both CREAMS and WEPP (Water Erosion Prediction Project). The two models varied in their predictions. For example, in one case WEPP predicted an 85% reduction in soil loss while CREAMS predicted a 10% reduction. No field testing was carried out to verify the predictions.

### 6.3.11 EUROSEM

One of the key objectives of the European Soil Erosion Model (EUROSEM) is to evaluate the effectiveness of soil conservation measures appropriate to European conditions. Rickson (1994) applied EUROSEM to evaluating the effectiveness of a grass filter strip in reducing sediment and runoff from an agricultural field. The procedure involved incorporating the physical characteristics, and accounting for the indirect effects, of the grass strip. This was achieved by choosing input parameter values so as to characterise the grass strip and its effects. Manning's *n* was increased to represent the roughness imparted by a dense sward. Roughness was also accounted for in the across and up/downslope roughness ratios. The vegetation characteristics of percentage cover and basal area were increased. Maximum rainfall interception storage was increased to reflect the dense canopy of the grass strip and soil cohesion was increased to allow for increased cohesion due to the dense grass root mat. The model also allows for variations in leaf shape and plant height.

The study confirmed that input data to EUROSEM can be modified to represent grass filter strips. The model was able to simulate the expected effect of a filter strip in controlling sediment and runoff from a hypothetical slope. The study demonstrates the potential for adapting vegetative characteristics to reflect VFSs but further confidence in the model output can only be gained from field verification.

### 6.3.12 REMM

The Riparian Ecosystem Management Model (REMM) is the most detailed and representative VFS model developed to date. The processes simulated by the model include surface and subsurface hydrology; sediment transport and deposition; carbon, nitrogen and phosphorus transport, removal and cycling; and vegetation growth. It can deal with up to three zones of management; for example, undisturbed forest adjacent to the stream, managed woody vegetation and an herbaceous strip. The soil is characterised in three layers through which vertical and lateral movement of water and associated dissolved nutrients are simulated. This requires a large number of input parameters so that the buffer is characterised in a lot more detail than those of the previously discussed models. For example, the vegetation data file contains physical information on the plant, factors related to photosynthesis, transpiration characteristics, nutrient content of plant part pools, and the initial size of the plant pools. Up to six vegetation canopy layers can be allowed for in each zone including trees and shrubs.

Initial testing showed that REMM was accurate at predicting buffer function under many conditions, but at times appreciable error was observed (Lowrance et al., 1998). More detailed testing indicated that REMM provided good estimates of transport and nutrient cycling processes (Lowrance et al., 2000) but this was limited to a single site. REMM is also currently very data-intensive. During the time of the current study the model authors were in the progress of writing a number of publications evaluating the model.

### 6.3.13 MMF

The original version of the Morgan-Morgan-Finney (MMF) model, based on concepts described by Meyer and Wischmeier (1969) and Kirkby (1976), was designed to predict annual soil loss from field-sized areas on hillslopes (Morgan and Duzant, 2007). It was validated using erosion plot data for 67 sites in 12 countries and has been successfully used in a wide range of environments including Malaysia, Indonesia, Nepal and the Rocky Mountains of the USA. A revised version of the model (Morgan, 2001) has been validated on the original data set and with additional data from Denmark, Spain, Greece and Nepal and has been used to model catchment-scale erosion in Kenya and Tanzania.

At the time of the current study the model was being developed further. The revisions take into account recent approaches to erosion modelling and simulate the deposition of soil particles explicitly. They also consider the particle size of the soil and provide a particle size description of the material eroded from the hillslope. Changes are also made to the way that the effect of land cover (vegetation or crops) is simulated. The effect of the roughness of the soil surface, such as that induced by tillage practices, is also incorporated. The model concept is illustrated in Figure 6.1 and the input parameters and model equations are listed in Morgan and Duzant (2007) (Appendix 3). Further details on the revisions can be found in this paper which also presents an example application of the model to an area of vegetation in Bedfordshire. Their study presents the only attempt to use the updated version of the model for evaluating the effects of vegetation on flow. Validation of the model was concluded to be successful based on comparing predicted and observed soil loss values using a coefficient of efficiency. However, this applied only to one study area and the model was not tested on VFSs but on single vegetated or bare elements. Measurable plant parameters used in model simulations are canopy cover, ground cover, effective hydrological depth, height, stem density and stem diameter.

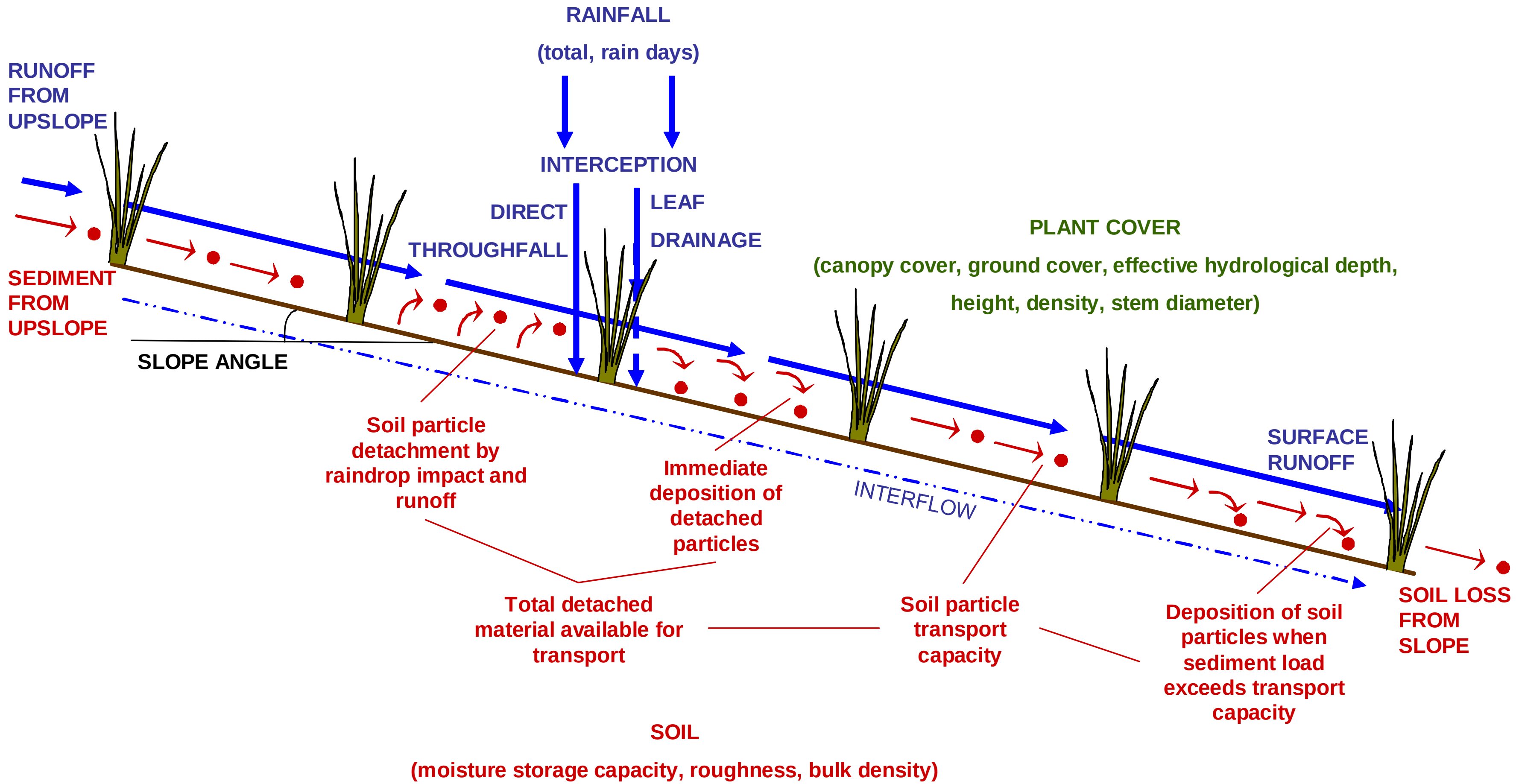



**Figure 6.1 Processes considered in the updated MMF model (Morgan and Duzant, 2007).**

## 6.4 Discussion

"Mathematical models present an alternative way to develop predictions of impacts that VFS installation will have on pollutant levels in streams" (Dosskey, 2002). In the preceding discussion a number of models were identified and discussed, some of which were developed for, and some adapted to, representing VFSs. Table 6.1 summarises the models and their applications. All of the models attempt to represent the filtration and sedimentation processes within a VFS. None of the models, except for REMM, deal with the adsorption and absorption of solutes by vegetation but this is mainly because those models discussed were primarily developed for sediment yield. Although the outputs obtained from the models appear to be promising, many of the models lack testing, particularly with a range of independently collected field data. They also differ from each other in objectives and therefore have not been tested on the same set of data so comparison between them is difficult.

| *Application* | *Model* | *References* |
|---|---|---|
| Designing in-field buffers and filter strips | CREAMS | Flanagan et al. (1989) |
| | RUSLE | Renard et al. (1997) |
| | VFSMOD | Munoz-Carpena et al. (2004) |
| | MMF | Morgan and Duzant (2007) |
| Predicting their impact on surface runoff from fields and surface runoff filtration | CREAMS | Hamlett and Epp (1994)<br>Williams and Nicks (1988) |
| | VFSMOD | Munoz-Carpena et al. (1999) |
| | EUROSEM | Rickson (1994) |
| | TRAVA | Deletic (1999, 2000,2001) |
| | SEDIMOT | Wilson et al. (1981) |
| | GRASSF | Tollner et al. (1977) Barfield et al. (1979) |
| | MMF | Morgan and Duzant (2007) |
| Scaling up predictions of individual VFS functions to watersheds | AGNPS | Tim and Jolly (1994) |
| | | Prato and Shi (1990) |
| Coupling surface and groundwater filtration functions of VFSs | REMM | Lowrance et al. (2000) |

**Table 6.1 Summary of applications of models in VFS research**

In terms of the factors that should be considered, from a bioengineering perspective, Coppin and Richards (1990) summarised the physical effects on soil and water transfer of a range of vegetation properties. Those processes relevant to VFSs have been selected and presented in Table 6.2. Based on this it can be proposed that at least ground cover, and stem and leaf density, should play a key role in modelling VFS performance, yet of the models discussed only EUROSEM uses these parameters directly. It was noted by Rickson (1994) that, due to the difficulties in obtaining site specific measurements, guide values are often used for important and sensitive parameters such as Manning's *n*.

Indeed the majority of the models reviewed use some form of coefficient, such as Manning's *n* or the *C* factor, in place of measurable plant properties. Styczen and Morgan (1995) suggest that using *C* factor values is a fairly simplistic approach to viewing the role of vegetation. They suggest that more emphasis should be put on understanding how vegetation operates within the erosion system and on the interaction between vegetation, climate, soil properties and hydrology. Values such as the *C* factor are derived experimentally and while they represent the conditions of a particular experiment they may not be readily used to predict the effect of the same or other vegetation in different climates and pedological conditions (Styczen and Morgan, 1995). Therefore models that do not take into account specific plant properties (e.g. GRASSF and its derivatives, TRAVA, CREAMS and AGNPS) are likely to be limited in the scope of their application.

Thompson and Robertson (1976) stated that in non-submerged vegetated overland flow, flow resistance is made up of the resistance due to surface roughness and the drag force due to the vegetative elements. However, Jin and Römkens (2001) proposed that in flow through dense vegetation it is the drag force that provides the dominant resistance to flow and that the flow velocity (*V*) is determined mainly by the surface slope (*S*) and vegetation density (*D*). This is expressed by Jin et al. (2000) as:

$$V = \left( \frac{2g}{C_d D} \right)^{1/2} S^{1/2}$$ **Equation 6.9**

where $C_d$ represents the drag coefficient of the vegetative elements and *g* is the acceleration due to gravity. Therefore, whilst roughness is an important factor a roughness coefficient alone may not be enough to express the vegetation properties of a VFS.

| *Vegetation* | | *Vegetation properties* | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Effect on | Influence | Ground cover (%) | Height | Leaf shape and length | Stem/leaf density | Stem/leaf robustness | Stem/leaf flexibility | Root depth | Root density | Root strength | Annual growth cycle |
| Soil surface | Soil detachment | ● | ● | ● | ● | | | | | | ● |
| | Mechanical strength | ● | ● | | ● | ● | | | ● | ● | ● |
| | Retarding/arresting | | ● | | ● | ● | ● | | | | |
| | Erosion | ● | | | ● | | | | | | ● |
| Surface water regime | Rainfall interception | ● | | ● | ● | | | | | | |
| | Overland flow/runoff | ● | | | ● | | | | | | |
| | Infiltration | | | | ● | | | ● | ● | | |
| | Subsurface drainage | | | | | | | ● | ● | | |
| | Surface drag | ● | ● | ● | ● | | ● | | | | ● |

**Table 6.2 Vegetation properties and their significance for soil and water transfer, adapted from Coppin and Richards (1990).**

Flannagan et al. (1989) demonstrated that, so long as a number of assumptions are met by a simplified version of the CREAMS model, it effectively simulates depositional processes in a VFS. These assumptions are: flow is fairly shallow and uniformly distributed along the upslope VFS edge, concentrated flow effects are minimal, the grass is not submerged and flattened by the flow, and previously trapped sediment does not affect future sediment delivery capacity (Wenger, 1999). These assumptions apply to the majority of the models reviewed, except for those which take into account the migration of a sediment wedge through the filter strip. This is likely to be particularly limiting in areas where concentrated flow dominates and causes submersion of the VFS vegetation.

Deletic (2000) suggested that the methods currently used for modelling of sediment transport through grass are very poor. This is attributed to previous work focusing on the overall performance of grass filter strips rather than the processes involved.

It should also be questioned as to whether any of the models define the erosivity of the flow once it leaves the grass strip. EUROSEM is based on the assumption that, if transport capacity has not been reached, then the flow is still potentially erosive. If all or much of the sediment has been deposited within or upslope of the filter then the flow's potential erosivity (i.e. transport deficit) is increased (Rickson, 1994). Considerable erosion was demonstrated by this process in laboratory experiments by Ligdi and Morgan (1995). Styczen and Morgan (1995) suggest that the foreslope of the sediment wedge (e.g. within the GRASSF model) is steeper than the ground slope. Therefore, once the foreslope of the wedge has migrated to the downslope edge of the filter, the velocity of the flow leaving the filter will be increased.

Despite the requirements to address limitations of current models there is general agreement that simple models, rather than precise parameter-intensive simulation models with large data requirements, will gain wider acceptance. Merrit et al. (2003) noted that model accuracy does not always increase with model complexity and there may be an accumulation of error through inaccuracies in the large number of parameters required. Wenger (1999) noted the dearth of models appropriate for wide-spread VFS implementation and stated the need for a new, simple formula. Dosskey (2002) proposed the need for models that enable useful approximations and that are easy to use. Perrin et al. (2001) demonstrated that very simple models can perform almost as well as those with more parameters and Merritt et al. (2003) warn of the errors introduced by over-parameterisation in complex models. Whilst REMM is the most advanced VFS model its use is restricted to trained users and requires access to large amounts of data. This suggests that improvements to models must be carefully targeted so the accuracy of simulating key processes, rather than general model complexity, is increased.

With the exception of TRAVA and EUROSEM, most VFS models have been developed in the US. When selecting a model it is important not to assume that a model developed and tested within a particular country will be reliable in another part of the world. For example, Rickson (1994) warns that whilst the USLE has been used by the United States Department for Agriculture (USDA) and the Soil Conservation Service (SCS) to evaluate the effectiveness of soil conservation measures, and has the advantage of simplicity, it should not be used unmodified in Europe due to its empirical foundations being based on US data. Furthermore, Brazier (2004) suggests that there is no 'universal' soil loss equation, as any empirically based model will always be limited, in scope, to the condition represented by the data it is built upon. Any predicted results, therefore, will only describe the range of experimental conditions for which they were obtained.

## 6.5 Conclusions

The models presented demonstrate that simulation of a buffering effect can be achieved by altering input variables and parameters which influence the differences in soil, plant and water relationships that result from a change in land management practice. However, limitations in application exist including an incomplete description of potentially important site and design variables and often a lack of field testing and verification.

Models applied to VFS simulation should be simple to use but should reflect the main processes involved when flow is intercepted by a barrier of vegetation. Chapter 2 outlined the importance of vegetative parameters in VFS performance and yet, within the models reviewed, there is a lack of consideration of measurable plant properties. Based on the literature reviewed in this chapter and in Chapter 2 it appears highly likely that model performance may be improved by using such parameters, rather than using coefficients such as Manning's *n* or a *C* factor. In particular, a model representing physically meaningful parameters was required to address the hypotheses posed in Chapter 1 and restated in Section 7.1. The preceding discussion provides the basis for model selection in the following section.

## 6.6 Selecting a suitable model for the study

For the purpose of this thesis a model was required that could be used to evaluate different VFS options in terms of design and placement. The specific objectives of the model were:

1. The model must be physically-based in order to offer greater potential than a conceptual or empirical approach in predicting the spatial distribution of runoff and sediment.
2. The model must be simple and easily understood without requiring a trained user if it is to be acceptable to a wide audience.
3. The model must estimate runoff, erosion and sediment deposition in order to allow evaluation of sediment and water retained by, and moving through, a VFS.
4. The model must take vegetative cover into account to allow scenario analysis for different vegetative parameters on which design guidance can be developed.

As well as meeting these criteria the WEPP, EUROSEM and MMF models take into account vegetation explicitly and are known to have been applied in the UK and Europe (Brazier, 2004). To assess these models further a more detailed list of desired capabilities was compiled, against which the models were rated.

The criteria were based, in part, on the discussion in the previous section and are presented, with the results, in Table 6.3. Each of the three models is scored against the original objectives (primary requirements) plus an extra ten criteria (secondary requirements). All criteria plus justification for the secondary requirements are listed in Table 6.3. The criteria were selected based on what was perceived to be useful for providing guidance, and supporting decisions, over VFS design and placement at the field or farm scale. For example spatial scale, timescale, cost and time for generating results, availability of data and ability to be combined with other management options. Consideration was also given to those factors identified as important in the model review, including ease of use and description of processes, particle size and vegetation characteristics. The information on each model was obtained through communication with the authors and developers of the models.

It can be seen that the MMF model (Morgan and Duzant (2007) version) not only meets the study requirements but scores highly against the extra criteria. It is also advantageous in that it can be adapted from its annual version to consider single events. This MMF version was used in the Defra project (PE0205).

In order to fully examine the model structure a sensitivity analysis was performed (Chapter 7) followed by further testing and development of the model to improve its application to VFS design and placement (Chapters 8 to 10). For ease of reference the Morgan-Duzant version of the MMF is referred to, in following chapters, as MMF-VFS.

| *Capability* | *Model* | | |
|---|---|---|---|
| | WEPP | EUROSEM | MMF |
| *Physically based (greater potential than other model approaches)* | *** | *** | *** |
| *Simple to use i.e. does not require trained user(acceptability to users)* | * | * | ** |
| *Describes soil erosion, runoff and deposition (evaluation of all material passing through VFS)* | * | * | *** |
| *Accounts for vegetative characteristics(advice provision for parameters and species)* | ** | *** | *** |
| Based on or modified with UK/European data (applicability to UK conditions) | ** | *** | *** |
| Describes particle size fractions (assessing VFS performance in relation to different size fractions and associated pollutants) | *** | * | *** |
| Annual/seasonal model (planning and monitoring over time) | *** | * | *** |
| Available or readily attainable data (acceptability to users) | ** | ** | *** |
| Enables routing over multiple elements comprising the field, farm and small catchment scale (planning over different scales) | *** | *** | *** |
| Can be combined with land cover and land management options (use in agri-environment scheme feature planning) | *** | *** | *** |
| Reasonable simulation time requirements (study resources and acceptability to users) | *** | *** | *** |
| Has been applied to VFS design (already an indication of model performance for the task) | ** | ** | * |
| Has been applied to VFS placement (already an indication of model performance for the task) | ** | ** | * |
| Low cost to obtain and use (study resources and acceptability to users) | *** | *** | *** |

Suitability scoring:

* = Low     ** = Moderate     *** = High

Definitions:

WEPP *Water Erosion Prediction Project*;     EUROSEM *European Soil Erosion Model*;   MMF *Morgan Morgan Finney*

**Table 6.3 Suitability criteria for model selection. Justification for criteria are shown in brackets and original objectives shown in italics.**

# Chapter 7 Sensitivity analysis

## *7.1 Introduction*

Chapter 6 introduced the MMF-VFS as the model selected for further study. This chapter describes the first stage in testing and developing the model before development of VFS guidance tools (Chapter 10). As introduced in Chapter 3, the objective of sensitivity analysis is to quantify the effects of parameter variations on calculated results. Sensitivity analysis is usually performed as an early stage in model testing and allows an examination of the model structure. It may eliminate errors in the model, indicate the range of conditions for which the model is most likely to be useful and ensure that changes in output with input are in the right direction (Pilgrim, 1975). Prior to calibration it can also be used to find out which of the parameters dominate the model response and therefore need the most critical parameterization (Grayson and Blöschl, 2001). During validation it may be used to quantify the effect that an error in the value of an input parameter has on the model output (Quinton, 1994). This chapter presents different methods of sensitivity analysis and the results of a sensitivity analysis carried out for the MMF-VFS. The results are discussed in terms of their implications for using the model for evaluating VFSs.

### 7.1.1 Objectives for the MMF-VFS sensitivity analysis

Chapter 3 proposed a modelling approach to the evaluation of laboratory and field VFSs. It was therefore essential to perform a sensitivity analysis on the selected model and this had the following objectives:

a) To establish which model parameters have the greatest control over the model output;
b) To provide information against which the rationale of the model structure could be evaluated, with the model responses being examined against measured trends;
c) To further the understanding of the model's behaviour with respect to its structure and operating equations;
d) To generate information that would be useful for assessing the likely error in model predictions.

A variety of statistics is available to describe model sensitivity. The following methods are suitable for the determination of the sensitivity of a single value output.

### 7.1.2 Absolute sensitivity

McCuen (1973) defined absolute sensitivity as the "rate of change in one factor with respect to change in another factor". In other words how the model output changes with changes in the model's input. It can be expressed as the linearised sensitivity equation (McCuen, 1973):

$$AS = \frac{[O_2 - O_1]}{[I_2 - I_1]}$$ **Equation 7.1**

where $O_2$ and $O_1$ are the values of the model output obtained with the values $I_2$ and $I_1$ respectively for an input parameter *I* and *AS* is the absolute sensitivity. Heatwole, Campbell and Botcher (1987) used this equation in their sensitivity analysis of the modified CREAMS model. However, a disadvantage of this method is that the absolute sensitivity equation gives more weight to large changes in model outputs when the absolute change in model inputs is small. It takes no account of the relative magnitudes of the values involved, e.g. that rainfall can vary from 0 to >5000 mm whereas bulk density is likely to vary only from 0.8 to 1.8.

This concept can be illustrated by considering some of the *AS* values generated for MMF model input parameters. Table 7.1 presents *AS* values calculated for rainfall (R), Effective Hydrological Depth (EHD), flow depth (Fd) and Manning's *n*. The *AS* values are derived from Equation 7.1 where model output is soil loss. The percentage change was then calculated from both the range of model input values used to perform the sensitivity analysis and from the corresponding output values generated. It is clear from this that the highest sensitivity (*AS*) value (Manning's *n*) is the result of a very small change in parameter input producing a large change in model output. The opposite occurs for R, with a very low *AS* resulting from a large change in input producing a small change in output. EHD and Fd have very similar *AS* values and in both cases the change in input value is approximately three times the change in output value.

| Model input parameter | AS | % change in input values | % change in output values |
|---|---|---|---|
| R | 0.02 | 400 | 9.1 |
| EHD | -121.75 | 120 | -47.14 |
| Fd | 155.31 | 5900 | 1885.6 |
| Manning's n | -844.76 | 2 | -47 |

**Table 7.1 Absolute sensitivity results for selected parameters and percent changes in input and output values.**

### 7.1.3 Relative sensitivity

In order to compare sensitivities it is necessary to normalise the input and output (McCuen, 1973). The following equation is given by McCuen (1973):

$$RS = \frac{\left[\frac{O_2 - O_1}{O}\right]}{\left[\frac{I_2 - I_1}{I}\right]} \qquad \textbf{Equation 7.2}$$

where $RS$ is the relative sensitivity, $I$ some base value lying between $I_1$ and $I_2$, $O$ the output for $I$, and $O_1$ and $O_2$ the outputs for $I_1$ and $I_2$. Lane and Ferrier (1980), Thomas and Beasley (1986), Ogunmokun (1990) and Ozara (1990) all use this approach in the analysis of their models. Relative sensitivity gives a dimensionless statistic which allows comparison between variables.

### 7.1.4 Average linear sensitivity

Nearing, Deer-Ascough and Chaves (1989) modified the relative sensitivity approach to give:

$$ALS = \frac{\left[\frac{O_2 - O_1}{\bar{O}}\right]}{\left[\frac{I_2 - I_1}{\bar{I}}\right]} \qquad \textbf{Equation 7.3}$$

where $\bar{I}$ is the mean of $I_1$ and $I_2$ and $\bar{O}$ is the mean of $O_1$ and $O_2$. The sensitivity parameter (ALS) represents a relative normalised change in output to a normalised change in input. Nearing, Deer-Ascough and Chaves (1989) state that this is a valid means of comparing sensitivities for different input parameters that have different orders of magnitude. An average linear sensitivity of zero indicates that the model output is insensitive to the input in question; a value of one indicates that the model output changes by relatively the same amount as the change in input (Heatwole, Campbell and Bottcher, 1987), and a value greater than 1 indicates that the relative change in output is greater than the change in the input.

A limitation of the ALS is that the linear form of the sensitivity parameter will not reflect sensitivity of the variable over the entire range of the parameter if the model response is non-linear (Nearing et al., 1990). It takes no account of the non-linearities present. If non-linearities are present, the value of ALS will depend upon the points between which the gradient is calculated. This is illustrated in Figure 7.1 whereby a small change in the value of *x,* when *x* is low, has little effect on the value of *y*; but when *x* is high a small change in *x* will have a large impact on *y*. An ALS of the whole data set (red dashed line) is therefore inappropriate. Calculating the ALS over a number of smaller sections (the blue lines) would help to indicate these non-linear relationships. However, performing the analysis on relatively narrow ranges of the input parameter values (indicated by the blue lines) and hence small datasets may be of little value in such an analysis because it may not reflect the entire dataset available.

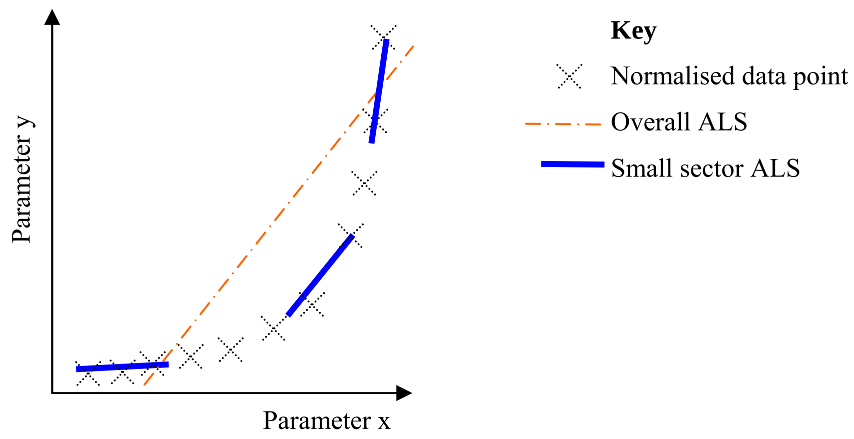



**Figure 7.1 ALS for non-linear relationships**

## 7.1.5 A deterministic sensitivity analysis approach

A deterministic sensitivity analysis approach was used by Deer-Ascough and Nearing, (1994) to measure model output during the evaluation of the WEPP model. This approach attempts to address the problem outlined in the previous section.

$$S = \frac{\left[\dfrac{O_b - O_1}{O_b}\right]}{\left[\dfrac{I_b - I_1}{I_b}\right]} \qquad \textbf{Equation 7.4}$$

Where $S$ is the sensitivity parameter, $O_b$ is the base output value, $O_1$ is the output for the current simulation run, $I_b$ is the base value for the current input parameter and $I_1$ is the value of the current input parameter. The base value is a reference value and may be derived from experimental or measured field values.

The sensitivity parameters reflect the linear response to a change in an input parameter with the change in the output response, all with respect to the base values. The parameter in this study reflects the linear response between the base value and the current input value. Deer-Ascough and Nearing (1994) propose that, while this *S* still reflects linear responses, it should provide a better examination (than the other methods of sensitivity analyses) of nonlinearity of responses between a series of input and output parameters. This is likely to be because an average *S* is calculated for a number of $I_I$s so that changes in *S* can be viewed as differences between $I_I$ and $I_b$ changes.

Within this approach the average sensitivity is an absolute value of the mean *S* value for each perturbation of a particular input variable. While the average sensitivity value may "mask" an input variable that is extremely sensitive for one perturbation, and not at all sensitive for another perturbation, the average provides a good means by which the model user may understand how sensitive a change in an input parameter is to a selected output response (Deer-Ascough and Nearing, 1994).

## 7.2 Sensitivity analysis of the MMF-VFS

The ALS method was selected for analysis of the MMF-VFS. The latter two methods (ALS and S) are based on developments of the other two methods (AS and RS). Both ALS and S have been reported in examples of the analysis of other soil erosion models. Of these two methods ALS is most commonly used in the literature and therefore is more likely to be the most understood and accepted. For comparison the deterministic sensitivity approach was also carried out but it was found that the results did not cause any difference to the categorising of results into high, medium and low sensitivity. Therefore the results in the following sections are derived using ALS.

Sensitivity analysis was carried out for an annual and for a single event version of the model. Input values for the rainfall and number of rain days were based on field and laboratory values for the annual and single event versions respectively. A further difference was the inclusion of evapotranspiration within the annual model (Table 7.2) given that the effect of this factor would only be important over time. For each version of the model sensitivity analysis was performed for both a bare soil and for a grassed element. In each case the model was run for a single soil-slope element with no routing of runoff and sediment between soil-slope elements.

### 7.2.1 Selection of base parameters

Commonly the number of parameters included in the sensitivity analysis is reduced, from the complete parameter set required to run the model, down to those which the modeller believes to be interesting (e.g. Lane and Ferrier, 1980; Nearing, Deer-Ascough and Laflen, 1990; and Ozara, 1990). Quinton (1994) suggests that this approach is unsatisfactory for the purposes of both code verification and corroboration because it may preclude unexpected results in need of further investigation.

In the sensitivity analysis of MMF-VFS all input parameters required to represent the physical description of the site being modelled are investigated. This meant considering 20 model parameters for a bare soil plot and 26 for a grassed plot.

### 7.2.2 Selection of base values

The ALS relies upon the value of all other descriptors (base parameters or variables) being held constant, while the value of the parameter or variable under examination is changed. There are different methods in the literature for choosing base values. Lane and Ferrier (1980) suggest selecting values on the basis that they give adequate predictions; Nearing, Deer-Ascough and Chaves (1989) select theirs on the basis of the availability of data and Ozara (1990) uses hypothetical parameter values. Morgan et al. (1987), Ogunmokun (1990) and Quinton (1994) selected base values close to the mid-point of the range encountered in the field. The latter approach was adopted for the sensitivity analysis of MMF-VFS.

The simulated runs for the sensitivity analysis of both versions of the MMF-VFS were based on data estimated for an arbitrary sandy loam soil, on a slope of 10° with a length of 15 m, a width of 3.5 m and a rainfall intensity of 45 mm/hr. These values relate to a soil type and field plot size typical of the literature. The slope angle and rainfall intensity are high for UK conditions but were chosen to encourage runoff and soil loss. They were also the conditions originally selected for the laboratory study, before adaptation based on the criteria discussed in Chapter 3. The model was run only for one element so the interflow (subsurface flow) component was not tested.

### 7.2.3 Variation in base values

A number of approaches to varying base parameter values have been reported in the literature. The type of approach chosen will depend largely upon the purpose of the analysis. In order to determine the effects of small measurement errors on model predictions the input parameter may be varied by an amount equal to the likely error. For example, Favis-Mortlock and Smith (1990) chose a value of 10% in their analysis of the EPIC model. McCuen and Snyder (1985) suggest that it is more important to consider the extremes of a range of likely values. This approach will check whether the model still behaves rationally under extreme conditions. In theory this should be the case but in practice the model is less likely to be used for extreme environments or events since these occur less frequently. Although climate change scenarios which predict an increase in extreme weather events may call for this. An alternative, and one used by a number of researchers (Thomas and Beasley, 1986; Heatwole, Campbell and Botcher, 1987; Nearing, Deer-Ascough and Chaves, 1989; Quinton, 1994), is to vary the base values according to their likely variation in the field.

### 7.2.4 Sensitivity indicators

Soil loss, runoff and particle detachment were selected as sensitivity indicators. These indicators represent the main model output and provide the most useful values for assessing the performance of the model against experimental results. The model input parameters for the annual model analysis of the grassed element are presented in Table 7.2. Those parameters marked with an asterisk were included in the analysis of the bare soil element. The values in parentheses are those used in the analysis of the single event version.

| Parameter | | | ***Base value*** | | |
|---|---|---|---|---|---|
| Mean annual rainfall (mm)* | 500 (10) | 1000 (20) | **1500 (30)** | 2000 (40) | 2500 (50) |
| Mean annual temperature (°)* | 5 | 10 | **15** | 20 | 25 |
| Rainfall intensity (mm/hr)* | 15 | 30 | **45** | 60 | 75 |
| Mean annual rain days* | 50 (1) | 100 (2) | **150 (3)** | 200 (4) | 250 (5) |
| Soil moisture (% w/w)* | 0.05 | 0.15 | **0.25** | 0.35 | 0.45 |
| Bulk density of top soil (Mg $m^{-3}$)* | 0.8 | 1 | **1.2** | 1.4 | 1.6 |
| Effective hydrological depth (m)* | 0.05 | 0.08 | **0.11** | 0.14 | 0.17 |
| Slope angle (°)* | 2 | 6 | **10** | 14 | 18 |
| Slope length (m)* | 5 | 10 | **15** | 20 | 25 |
| Element width (m)* | 0.5 | 2 | **3.5** | 5 | 6.5 |
| RFR (cm $m^{-1}$)* | 5 | 15 | **25** | 35 | 45 |
| PI (between 0 and 1) | 0.05 | 0.25 | **0.5** | 0.75 | 1 |
| Et/Eo (ratio of actual to potential)* | 1 | 3 | **5** | 7 | 9 |
| Canopy cover (between 0 and 1) | 0 | 0.25 | **0.5** | 0.75 | 1 |
| Ground cover (between 0 and 1) | 0 | 0.25 | **0.5** | 0.75 | 1 |
| Plant height (m) | 0 | 1 | **5** | 10 | 30 |
| Stem diameter (m) | 0.00001 | 0.1 | **0.5** | 1 | 3 |
| Stem density (elements per $m^{-2}$) | 0.00001 | 250 | **500** | 1000 | 2000 |
| Rainfall detachability (c)* | 0.1 | 0.4 | **0.7** | 1.1 | 1.5 |
| Rainfall detachability (z)* | 0.5 | 1.65 | **2.8** | 4 | 5.15 |
| Rainfall detachability (s)* | 0.15 | 1.1 | **2** | 3.1 | 4.15 |
| Runoff detachability (c)* | 0.02 | 0.11 | **0.2** | 1.1 | 2 |
| Runoff detachability (z)* | 0.016 | 0.08 | **0.16** | 0.88 | 1.6 |
| Runoff detachability (s)* | 0.015 | 0.08 | **0.15** | 0.83 | 1.5 |
| Manning's n* | 0.01 | 0.015 | **0.02** | 0.025 | 0.03 |
| Flow depth (m)* | 0.005 | 0.01 | **0.1** | 0.2 | 0.3 |

**Table 7.2 Input values for sensitivity analysis. An asterisk indicates an element used in the analysis of the bare soil element. Values in parentheses were used in the analysis of the single event version.**

## *7.3 Sensitivity analysis results: Annual model*

The results of the ALS, for both the bare soil and the grass element, are presented in Figure 7.2 and Figure 7.3 and summaries are provided in Table 7.3 and Table 7.4. Runoff appears to be the most sensitive model output showing high sensitivity to six of the eight input parameters tested for this sensitivity indicator. Soil loss and particle detachment are similar in their responses although soil loss is more sensitive to bulk density, effective hydrological depth and slope angle. The only parameters to which all three sensitivity indicators show high sensitivity are rainfall (bare slope and grass element) and number of rain days (grass element only).

A further difference between the bare slope element and the grass element is that the effect of the soil surface appears to be overridden when there is a vegetation cover. For example, sensitivity to soil roughness, Manning's *n*, bulk density and slope angle is lower on the grass element than on the bare soil element. Sensitivity to particle detachment also differs between the two elements with the grass element only showing sensitivity to the detachment of clay by rainfall. This is due to the vegetation cover protecting the soil surface from raindrop impact and reducing the flow velocity of runoff so that only the more mobile clay particles become detached. Although runoff does not appear to be sensitive to most of the vegetation parameters it is sensitive to effective hydrological depth (EHD). This is important to note otherwise it might appear that the runoff is insensitive to vegetation cover. In the field EHD would vary with root depth and development (Morgan and Duzant, 2007).

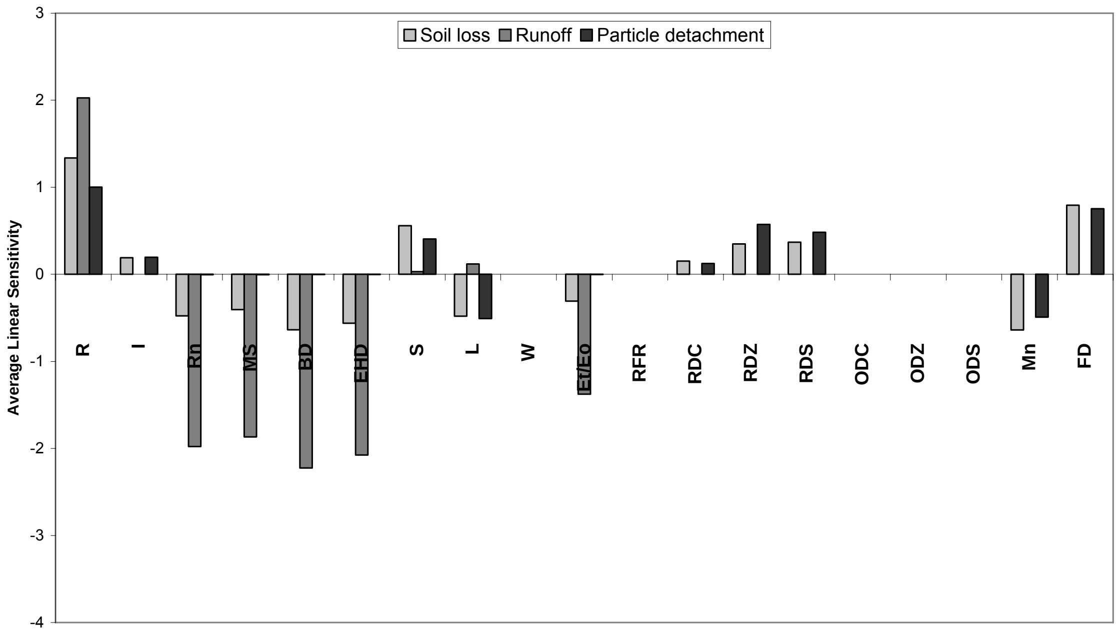



**Figure 7.2 ALS results for bare soil element, annual model.**

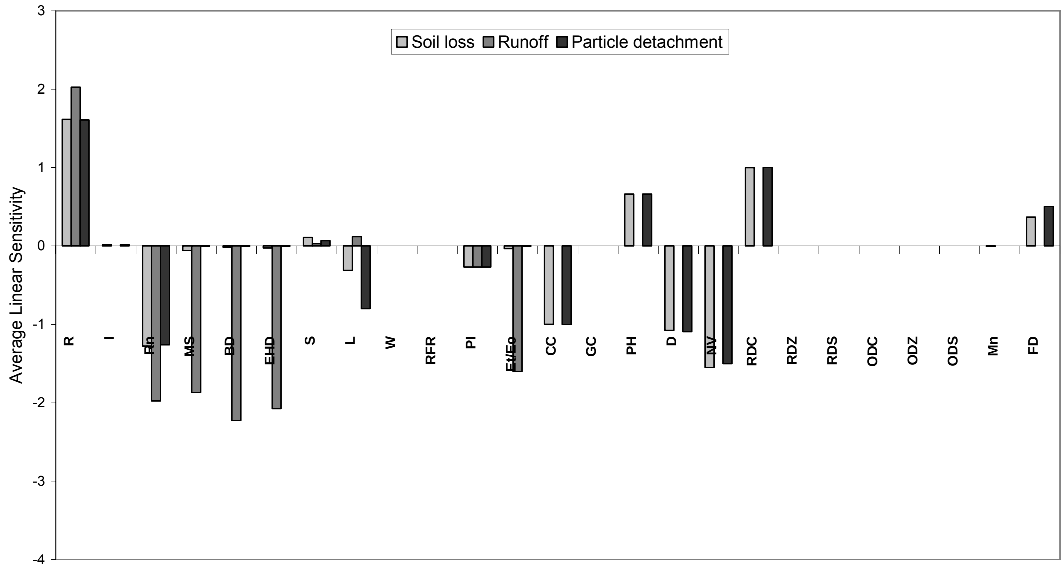



**Figure 7.3 ALS results for grass element, annual model.**

| *Parameter* | *Abbreviation* | *Soil loss* | *Runoff* | *Particle detachment* |
|---|---|---|---|---|
| Rainfall | R | *** | *** | *** |
| Temperature | T | | | |
| Intensity | I | * | | * |
| Rain days | Rn | * | *** | * |
| Soil moisture | SM | * | *** | * |
| Bulk density | BD | ** | *** | * |
| Effective hydrological depth | EHD | ** | *** | * |
| Slope angle | A | ** | * | * |
| Slope length | L | * | * | ** |
| Element width | W | | | |
| Evapotranspiration | ET | * | *** | * |
| Soil roughness | RFR | * | | * |
| Rainfall detachment C | RDC | * | - | * |
| Rainfall detachment Z | RDZ | * | - | * |
| Rainfall detachment S | RDS | * | - | * |
| Runoff detachment C | ODC | * | - | * |
| Runoff detachment Z | ODZ | | - | |
| Runoff detachment S | ODS | * | - | * |
| Manning’s n | Mn | ** | | * |

| | | | | |
|---|---|---|---|---|
| Flow depth | FD | ** | | ** |

**Table 7.3 Sensitivity classification of the MMF-VFS outputs for the bare soil element. *** indicates a highly sensitive response, ** moderate sensitivity and * low sensitivity. A blank cell indicates an insensitive response to the parameter tested. A dash means that the parameter was not tested.**

| *Parameter* | *Abbreviation* | *Soil loss* | *Runoff* | *Particle detachment* |
|---|---|---|---|---|
| Rainfall | R | *** | *** | *** |
| Temperature | T | - | - | - |
| Intensity | I | * | | * |
| Rain days | Rn | *** | *** | *** |
| Soil moisture | SM | * | *** | * |
| Bulk density | BD | * | *** | * |
| Effective hydrological depth | EHD | * | *** | * |
| Slope angle | A | * | * | * |
| Slope length | L | * | * | ** |
| Element width | W | | | |
| Soil roughness | RFR | * | | |
| Permanent interception | PI | * | * | * |
| Evapotranspiration | ET | * | *** | * |
| Canopy cover | CC | ** | | ** |
| Ground cover | GC | * | | * |
| Plant height | PH | ** | | ** |
| Stem diameter | D | *** | | *** |
| Stem density | NV | *** | | *** |
| Rainfall detachment C | RDC | ** | - | ** |
| Rainfall detachment Z | RDZ | | - | |
| Rainfall detachment S | RDS | | - | |
| Runoff detachment C | ODC | * | - | * |
| Runoff detachment Z | ODZ | | - | |
| Runoff detachment S | ODS | | - | |
| Manning's n | Mn | * | | |
| Flow depth | FD | * | | ** |

**Table 7.4 Sensitivity classification of the MMF-VFS outputs for the grass element (for key see caption under Table 7.3).**

## *7.4 Sensitivity analysis results: Single event model*

The results of the ALS, for both the bare soil and the grassed element, are presented in Figure 7.4 and Figure 7.5 and a summary is provided in Table 7.5 and Table 7.6. For the bare soil element both versions of the model show, for all three indicators, a high sensitivity to rainfall and a low response to rainfall intensity. They also show, for all three indicators, the same sensitivity to soil moisture, number of rain days, detachment of clay and sand (from runoff and rainfall) and flow depth. For the grassed element the level of response, for all three indicators, is the same for all parameters except Manning's *n*, to which only runoff was sensitive, and flow depth, to which only runoff was not sensitive.

Compared with the annual version of the model the sensitivity of soil loss to bulk density, effective hydrological depth and slope angle decreases from moderate to low on the bare soil element. Runoff becomes insensitive to slope angle and slope length. Particle detachment is more sensitive to Manning's *n* and to the detachment of silt by both rainfall and runoff. On the grass element soil loss and particle detachment are more sensitive to canopy cover. Soil loss decreases, while runoff increases, in sensitivity to Manning's *n*. These differences are discussed in later sections.

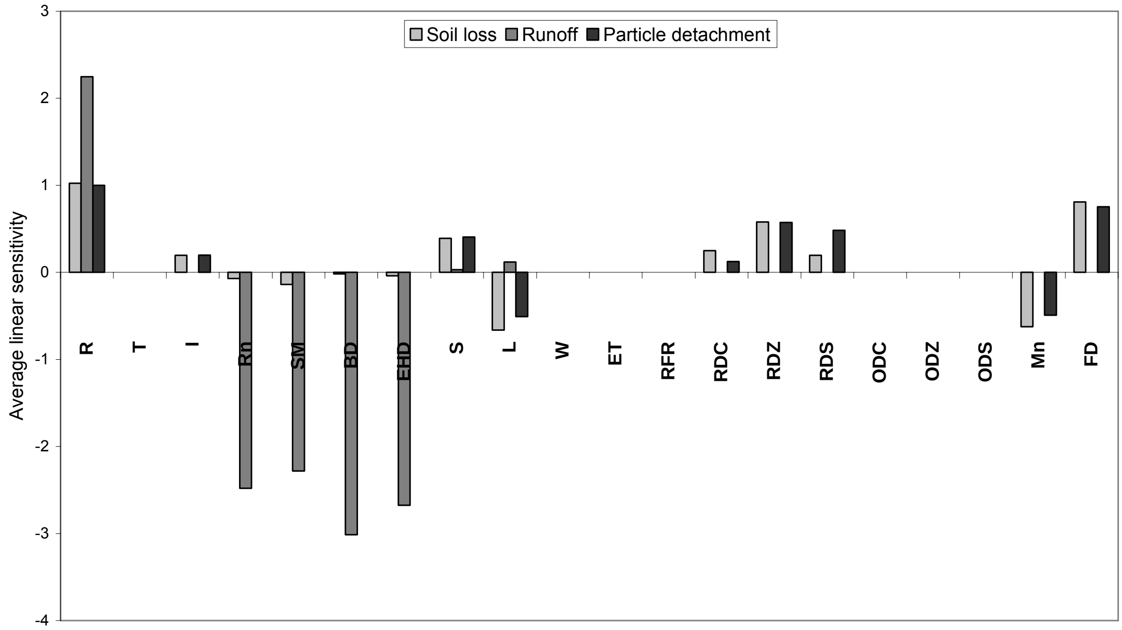



**Figure 7.4 ALS results for bare soil element, single event model.**

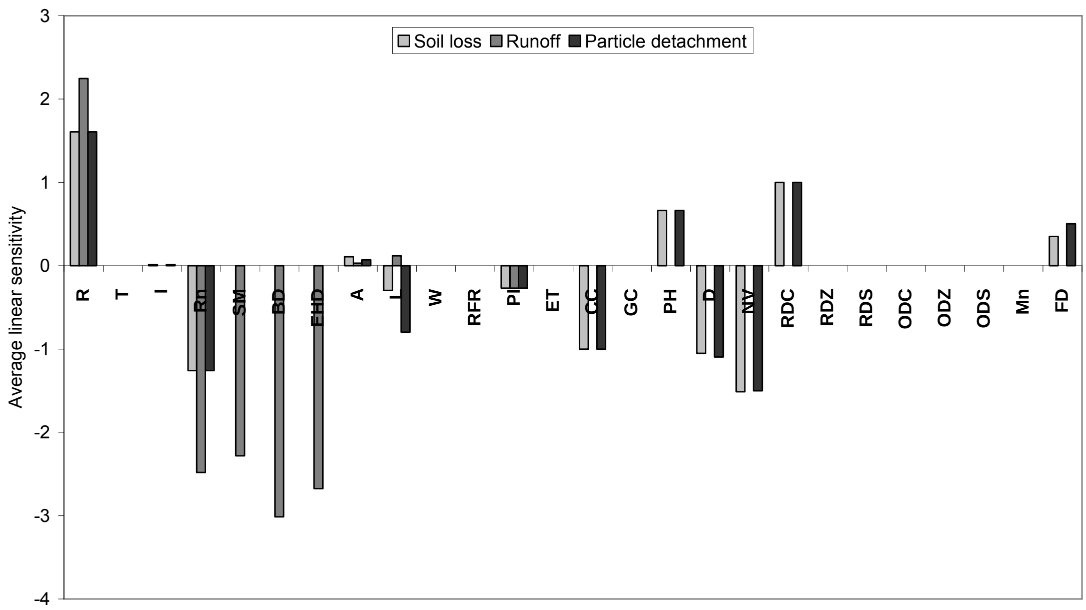



**Figure 7.5 ALS results for grass element, single event model.**

| *Parameter* | *Abbreviation* | *Soil loss* | *Runoff* | *Particle detachment* |
|---|---|---|---|---|
| Rainfall | R | *** | *** | *** |
| Temperature | T | | | |
| Intensity | I | * | | * |
| Rain days | Rn | * | *** | * |
| Soil moisture | SM | * | *** | * |
| Bulk density | BD | * | *** | * |
| Effective hydrological depth | EHD | * | *** | * |
| Slope angle | A | * | | * |
| Slope length | L | ** | | ** |
| Element width | W | | | |
| Evapotranspiration | ET | - | - | - |
| Soil roughness | RFR | | | |
| Rainfall detachment C | RDC | * | - | * |
| Rainfall detachment Z | RDZ | ** | - | ** |
| Rainfall detachment S | RDS | * | - | * |
| Runoff detachment C | ODC | * | - | * |
| Runoff detachment Z | ODZ | | - | * |
| Runoff detachment S | ODS | * | - | * |
| Manning's n | Mn | ** | | ** |
| Flow depth | FD | ** | | ** |

**Table 7.5 Sensitivity classification of the MMF Buffers outputs for the bare soil element. *** indicates a highly sensitive response, ** moderate sensitivity and * low sensitivity. A blank cell indicates an insensitive response to the parameter tested. A dash means that the parameter was not tested.**

| *Parameter* | *Abbreviation* | *Soil loss* | *Runoff* | *Particle detachment* |
|---|---|---|---|---|
| Rainfall | R | *** | *** | *** |
| Temperature | T | | | |
| Intensity | I | * | | * |
| Rain days | Rn | *** | *** | *** |
| Soil moisture | SM | * | *** | * |
| Bulk density | BD | * | *** | * |
| Effective hydrological depth | EHD | * | *** | * |
| Slope angle | A | * | * | * |
| Slope length | L | * | * | ** |
| Element width | W | | | |
| Soil roughness | RFR | | | |
| Permanent interception | PI | * | * | * |
| Evapotranspiration | ET | - | - | - |
| Canopy cover | CC | *** | - | *** |
| Ground cover | GC | * | - | * |
| Plant height | PH | ** | - | ** |
| Stem diameter | D | *** | - | *** |
| Stem density | NV | *** | - | *** |
| Rainfall detachment C | RDC | ** | - | ** |
| Rainfall detachment Z | RDZ | | - | |
| Rainfall detachment S | RDS | | - | |
| Runoff detachment C | ODC | * | - | * |
| Runoff detachment Z | ODZ | | - | |
| Runoff detachment S | ODS | | - | |
| Manning's n | Mn | | * | |
| Flow depth | FD | * | | ** |

**Table 7.6 Sensitivity classification of the MMF Buffers outputs for the grass element.**

## 7.5 Sensitivity of the parameters

It should be noted that the scales used on the *y*-axes of the Figures in this section vary depending on the values and therefore a quick visual comparison of sensitivity to changes from the base value, between the graphs, is not possible.

### 7.5.1 Rainfall

Soil loss, runoff and particle detachment are highly sensitive to mean annual rainfall (total rainfall amount in the single event version) (Table 7.3 and Table 7.4) with sensitivity to rainfall being higher for runoff than it is for soil loss and particle detachment (Figure 7.6). An increase in rainfall amount produces an increase in soil loss, particle detachment and runoff for both the bare soil (Figure 7.7) and grass (Figure 7.8) elements. This is expected as a correlation is typically observed when plotting runoff against rainfall depths. Whilst the relationship is the same the sensitivity is higher for the grass plot than for the bare soil plot (Figure 7.2 and Figure 7.3). These positive relationships, and the model's high sensitivity to rainfall, are unsurprising because rainfall provides the water volume necessary to initiate the detachment of soil particles by raindrop impact and runoff.

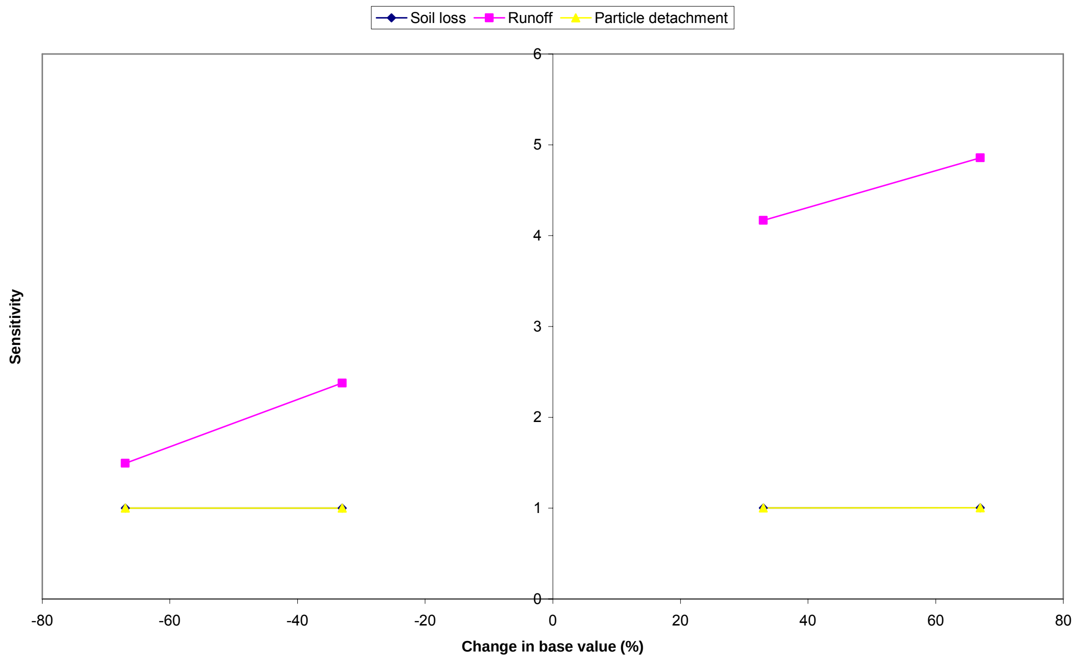



**Figure 7.6 Sensitivity of annual model to changes in mean annual rainfall amount on a bare soil.**

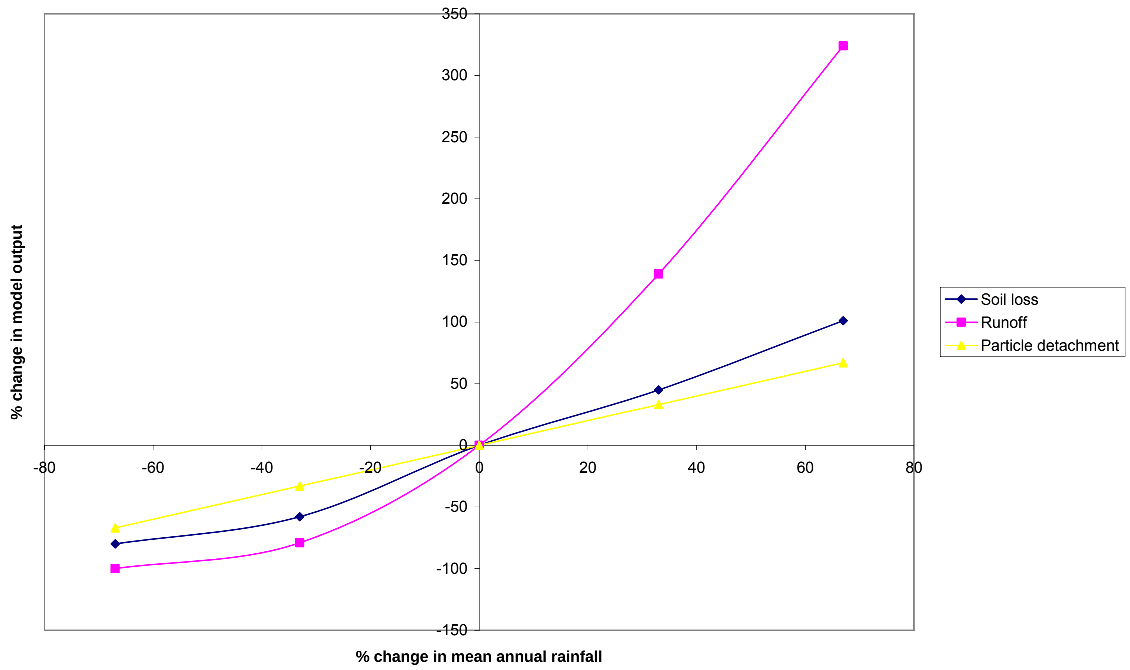



**Figure 7.7 Change in output response for mean annual rainfall amount, bare soil element**

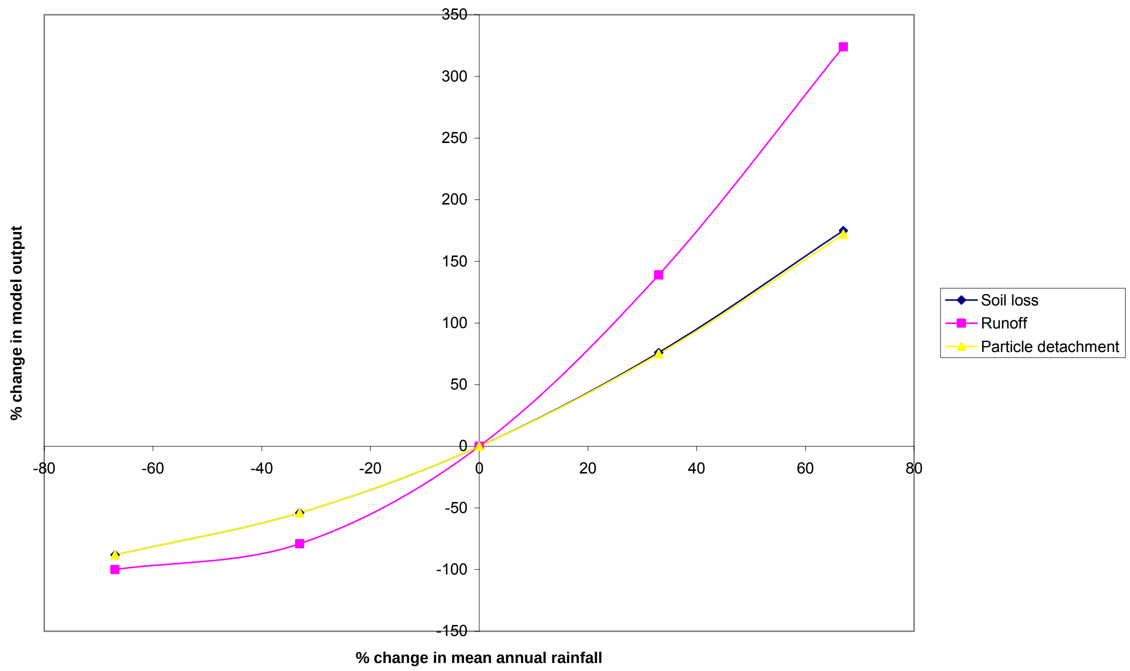



**Figure 7.8 Change in output response for mean annual rainfall amount, grass element**

### 7.5.2 Temperature

The effect of temperature was not tested by the sensitivity analysis because the model is only run for one element. Within the model, temperature only affects the interflow sub-routine through its effect on evapotranspiration. The model would, therefore, only be sensitive to temperature when run over two or more elements, with interflow being routed to the second element. In this case interflow would affect soil moisture storage and hence runoff generation on the second, and any further, elements.

### 7.5.3 Rainfall intensity

Soil loss and particle detachment show a low sensitivity, and runoff is insensitive, to rainfall intensity (Figure 7.10). The relationship with the former two indicators is positive, with an increase in rainfall intensity producing an increase in soil loss and in particle detachment. When rainfall is decreased by more than 33% (Figure 7.11) of the base value, which equates to a rainfall intensity of 30mm/hr, increases in soil loss and particle detachment become less on the bare soil and level off on the grass element. This is an artefact of the equation within the model which, at an intensity of 25-30mm/hr represents a decline in the rate of increase in kinetic energy with intensity. At an intensity of 75 mm/hr the rate of increase levels off (Hudson, 1981), as illustrated in Figure 7.9. Hence, up to 30 mm/hr an increase in drop size increases the kinetic energy of the rainfall, yet above 30 mm/hr the drops break up so that the energy does not further increase. This is not evident on the grass slope (Figure 7.12) because ground cover reduces the kinetic energy to very low levels and creates very similar levels of kinetic energy regardless of original intensity.

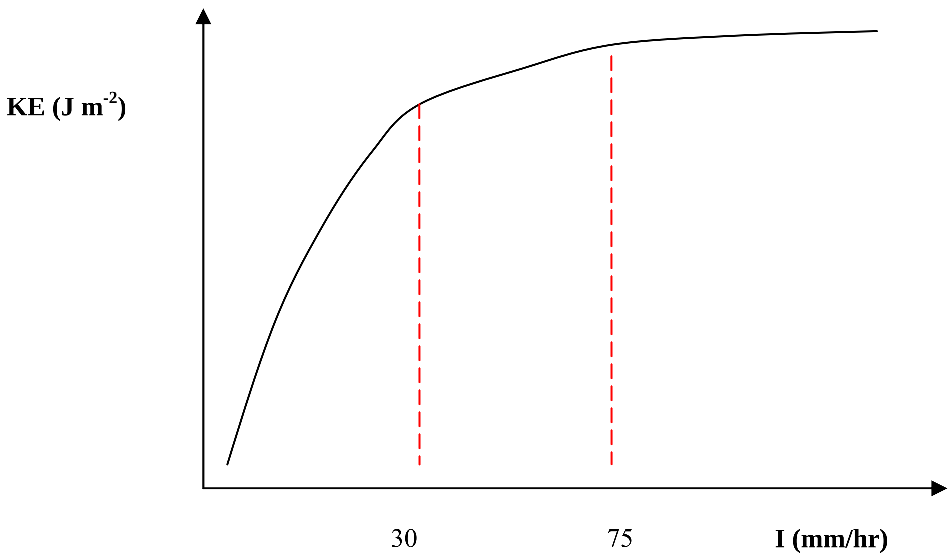



**Figure 7.9 Schematic illustrating relationship between kinetic energy (KE) and intensity (I).**

Sensitivity of runoff to rainfall intensity would be expected under natural field conditions as overland flow is inversely related to infiltration rate, which depends on the kinetic energy supplied by the rainfall to cause surface sealing (Ogunmoken, 1990). However, the lack of sensitivity shown by the model is not surprising because, within the model, intensity affects only kinetic energy and therefore particle detachment. Runoff, however, is a function of soil moisture storage. This is a weakness of storage capacity models as opposed to infiltration excess models for simulating runoff. However, it is less of a problem for application annually than it is for application to a single event because, in reality, differences in rainfall intensity are less important when averaged over a year.

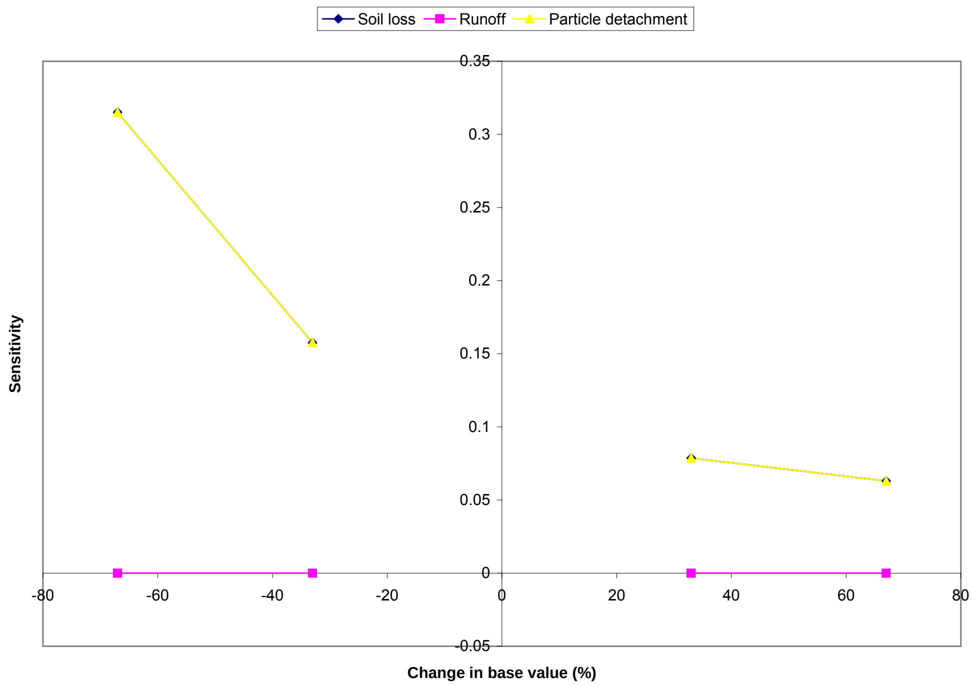



**Figure 7.10 Sensitivity of changes in rainfall intensity, bare soil element.**

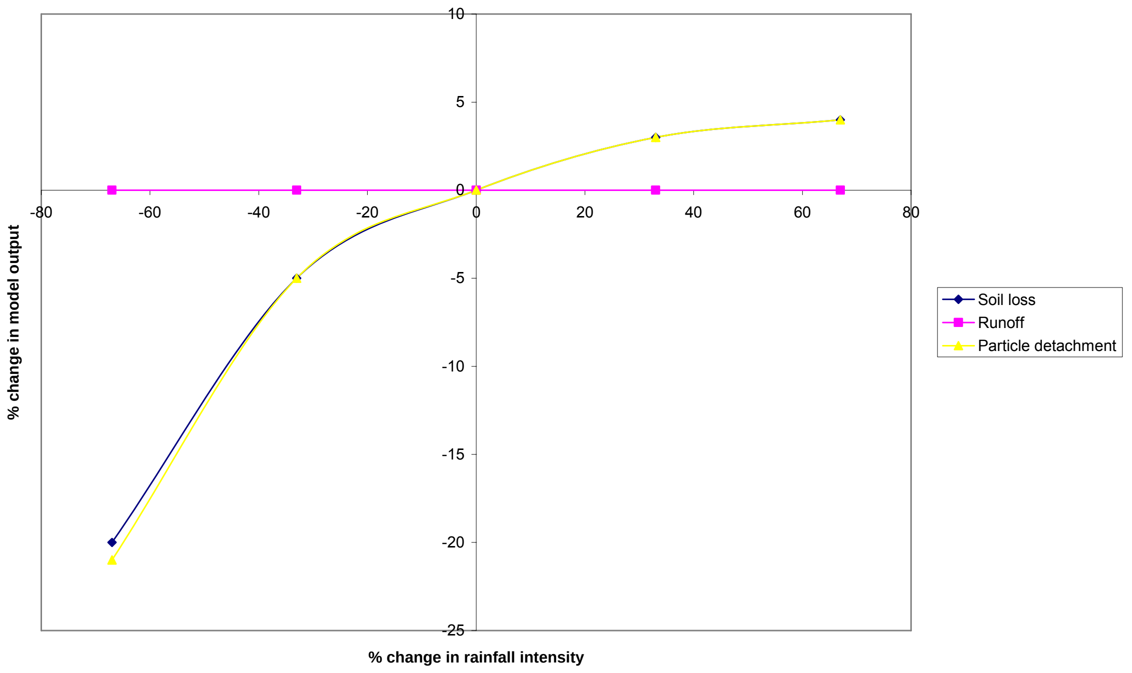



**Figure 7.11 Change in output response for rainfall intensity, bare soil element.**

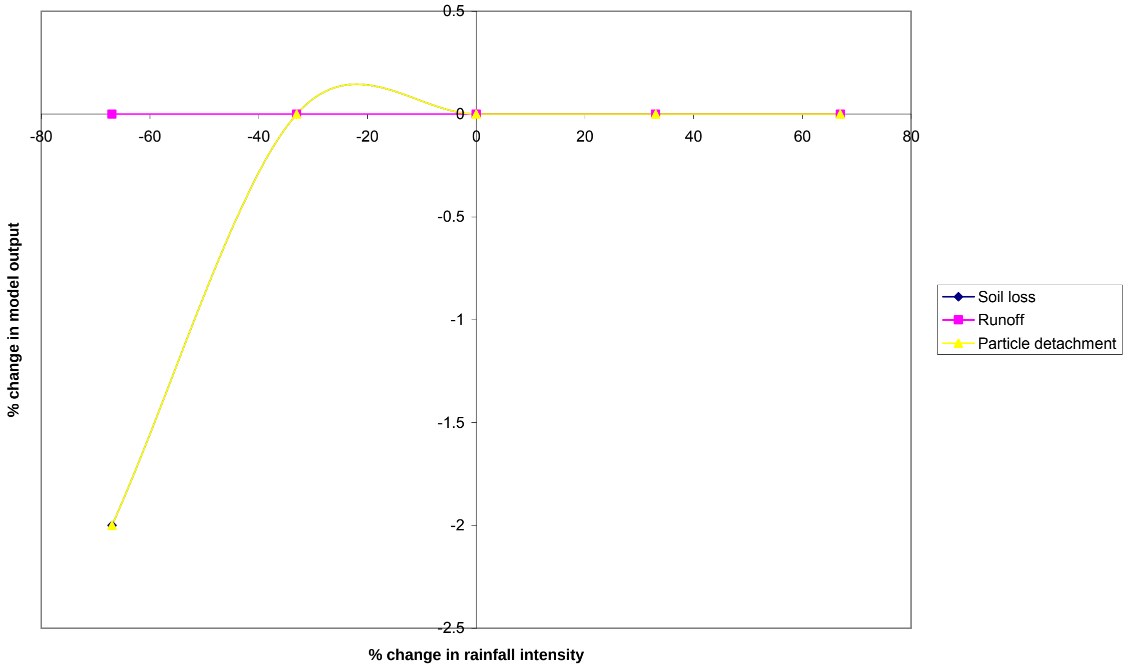



**Figure 7.12 Change in output response for rainfall intensity, grass element**

### 7.5.4 Number of rain days

With the bare soil simulation, runoff is highly sensitive to the number of rain days but soil loss and particle detachment show low sensitivity (Figure 7.13). With the grass simulation, soil loss, runoff and particle detachment are highly sensitive to the number of rain days (Figure 7.14). A decrease in the number of rain days produces an increase in soil loss, runoff and particle detachment. This is because the effective rainfall is divided between a lower number of events so that the amount of rain per event is higher.

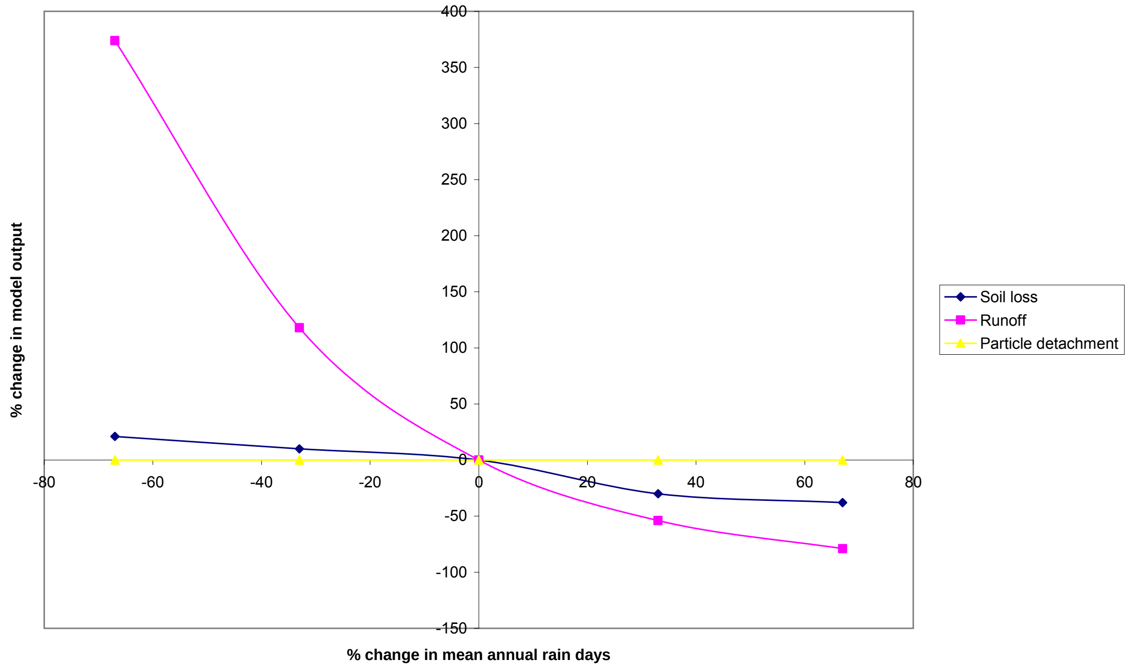



**Figure 7.13 Change in output response for number of rain days, bare soil element.**

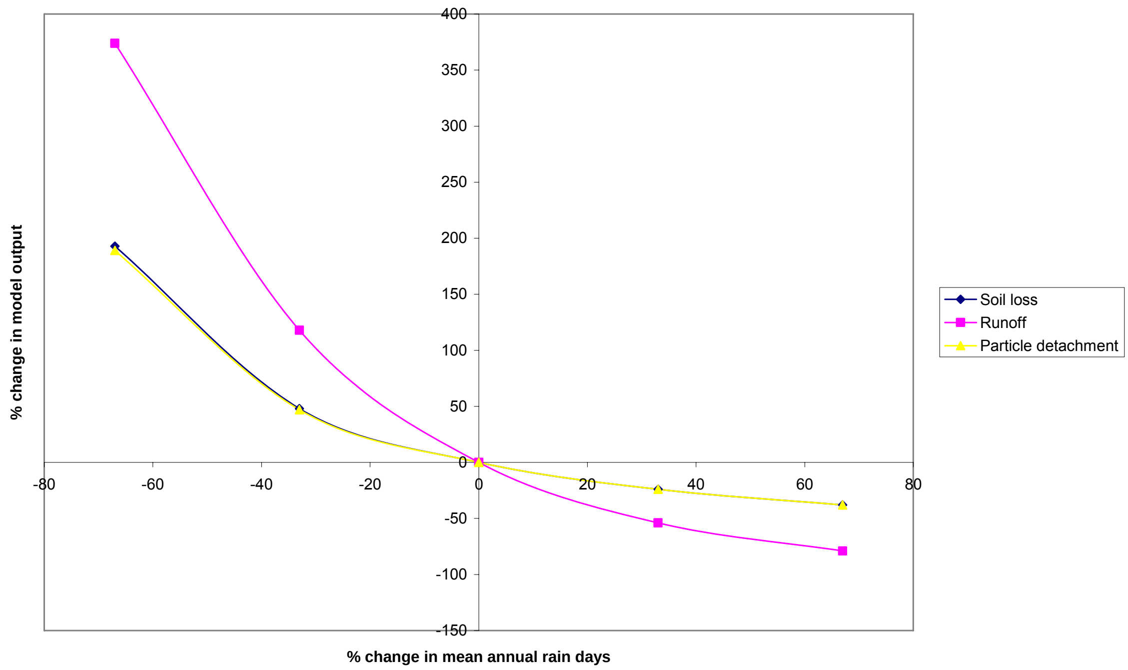



**Figure 7.14 Change in output response for number of rain days, grass element.**

## 7.5.5 Soil moisture content at field capacity, bulk density of the topsoil and effective hydrological depth

For the single event model, soil loss and particle detachment exhibit a low, and runoff a high, sensitivity to these three parameters. The only difference within the annual model is that it shows a moderate, rather than a low, sensitivity of soil loss to bulk density and effective hydrological depth on the bare soil plot. This is likely to be because event runoff is lower than annual runoff, and where soil loss is transport capacity limited, this would result in lower sensitivity at low runoff values.

The negative sign of the ALS (Figure 7.2 and Figure 7.3) indicates that an increase in soil moisture content at field capacity, an increase in bulk density and/or an increase in effective hydrological depth will decrease runoff. This is because these parameters are used within the model to calculate the moisture storage capacity of the soil. An increase in one or all of these parameters will increase the soil moisture storage capacity. For runoff to occur daily rainfall must exceed this capacity (Kirkby, 1976).

## 7.5.6 Slope angle

Sensitivity of both soil loss and runoff to slope angle was higher for the annual model on the grass plot. Both versions showed low sensitivity to slope angle for all indicators on the grass plot. Where a relationship does exist it is positive. Erosion would normally be expected to increase with increases in slope steepness as a result of respective increases in velocity and volume of surface runoff (Morgan, 2005). It appears that the model may underplay the effect of slope steepness. Where soil loss is detachment limited, slope angle has no effect. Where it is transport capacity limited the exponent in the sine relationship (Equations 18, 19 and 20, Appendix 2) with slope may be too low. This is an area for further model development work.

### 7.5.7 Slope length

The model output shows a decrease in soil loss with increasing slope length (in the direction of flow). Slope length is likely to increase soil loss due to an increase in runoff velocity and volume. However, there are a number of cases which state otherwise. Morgan (2005) suggests that without rills (which are not simulated within MMF-VFS) soil loss may not necessarily increase with distance downslope on slopes longer than 10 m. Gilley et al. (1986) propose that increasing the depth of overland flow downslope protects the soil from raindrop impact so that erosion becomes limited by the rate of detachment, which is decreasing with slope length. Meyer and Wischmeier (1969) found that once rills form, soil loss will increase with slope length, while Abrahams et al. (2001) observed a decrease in soil loss because, as the flow becomes concentrated, there is no longer sufficient flow on the interill areas to remove all the material detached by rainsplash. Parsons et al. (2004) proposed that sediment yield initially increases as plot length increases, but subsequently decreases. Results modelled by Parsons et al. (2006) showed the peak of sediment yield to lie somewhere between 4 and 14 m. Since the shortest slope length tested by the sensitivity analysis was 5 m it is possible that the analysis reflects the subsequent decrease found by Parsons et al. (2006).

In any case, low sensitivity to slope length suggests that the magnitude of any change with slope length is small. Soil loss was more sensitive and runoff less sensitive to slope length for the single event model on the bare soil element. On the grass plot the soil loss and runoff showed low sensitivity to slope length while the runoff was insensitive to slope length. These results are similar to those of Ogunmokun (1990) who found, with VERODEL that, in general, overland flow is insensitive to slope length, while only small changes of minor importance were noticed with soil loss. This is in agreement with the findings of Gilley et al. (1986) who concluded that runoff velocity and soil loss only vary slightly with variation in downslope distance when only interill erosion is taking place.

### 7.5.8 Element width

All indicators were insensitive to changes in element width. However, element width is more likely to influence model output when the model is used to route across a number of elements than when looking at a single element.

### 7.5.9 Evapotranspiration

Evapotranspiration was not tested within the single event version of the model. Within the annual model runoff was very sensitive to evapotranspiration on both the bare soil and grass element. Soil loss and particle detachment showed a low sensitivity on both elements. This is as expected as evapotranspiration is likely to have a direct effect on runoff amount.

### 7.5.10 Soil roughness

Soil roughness was not tested within the single event version of the model. For the annual model, the only conditions that showed any sensitivity to soil roughness were soil loss and particle detachment on the bare soil element. This seems reasonable because vegetation outweighs the effect of soil roughness on the grass covered element. In the model roughness affects only transport capacity. In reality, the situation is more complex because tillage-induced roughness causes micro-variation in local slope angle which may influence detachment. This is ignored by the model.

### 7.5.11 Permanent interception

Soil loss, particle detachment and runoff showed a low sensitivity to permanent interception (PI). An increase in PI produced a small decrease in the sensitivity indicators. This is due to the reduction in effective rainfall and therefore kinetic energy reaching the ground surface. The sensitivity may be low because the sensitivity parameter is a univariate parameter (Nearing et al., 1990). That is, the sensitivity parameter (ALS) requires that each parameter is varied in turn and therefore implies that there is no interaction between the variables. However, in natural conditions, it would be difficult for the permanent interception to alter without corresponding alterations to the other vegetation parameters e.g. canopy cover, ground cover, stem density and stem diameter. Also, PI is only going to influence soil loss directly when the erosion is detachment-limited; otherwise its effect is indirect through the effective rainfall contributing to runoff.

### 7.5.12 Canopy cover

Soil loss and particle detachment were very sensitive to canopy cover. Both outputs have a negative ALS which implies that an increase in canopy cover will produce a decrease in soil loss and particle detachment. This is not surprising since an increase in canopy cover should increase the protection of the soil surface. Conventionally, soil loss is considered to reduce in a linear fashion with increasing percentage cover (Wischmeier and Smith, 1978) and this relationship is represented by the model output (Figure 7.15).

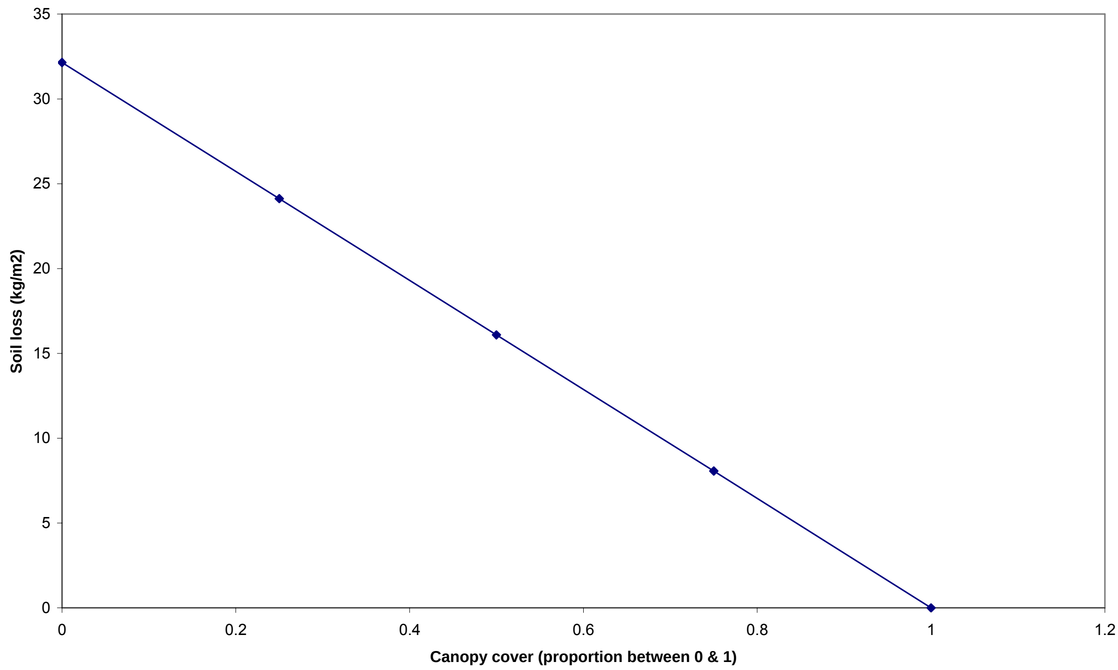



**Figure 7.15 Relationship between canopy cover and soil loss.**

## 7.5.13 Ground cover

Soil loss and particle detachment showed a low sensitivity to ground cover, with soil loss increasing slightly with a decrease in ground cover. The model uses ground cover to influence detachment by runoff. The interception of rainfall and consequent prevention of detachment by raindrop impact is accounted for by canopy cover. Plant height determines whether the canopy is on the ground or not.

An increase in ground cover should, under natural conditions, reduce flow velocity and hence increase the effectiveness of the plant cover in reducing erosion. There is little agreement on the nature of the relationship between soil loss and the extent of cover (Morgan, 2005). Elwell (1981) favoured an exponential decrease between soil loss with increasing percentage interception of rainfall energy and, therefore, increasing percentage cover. Such a relationship was suggested by Wischmeier (1975) as applicable to covers in direct contact with the soil surface and has been verified experimentally for crop residues (Hussein and Laflen 1982) and grass covers (Lang and McCaffrey 1984). The model exhibits a linear relationship but over a very small range (Figure 7.16) of soil loss values. It is difficult to find experimental evidence for the effect of ground cover as this parameter cannot easily be separated from the related changes to other vegetative parameters as discussed above.

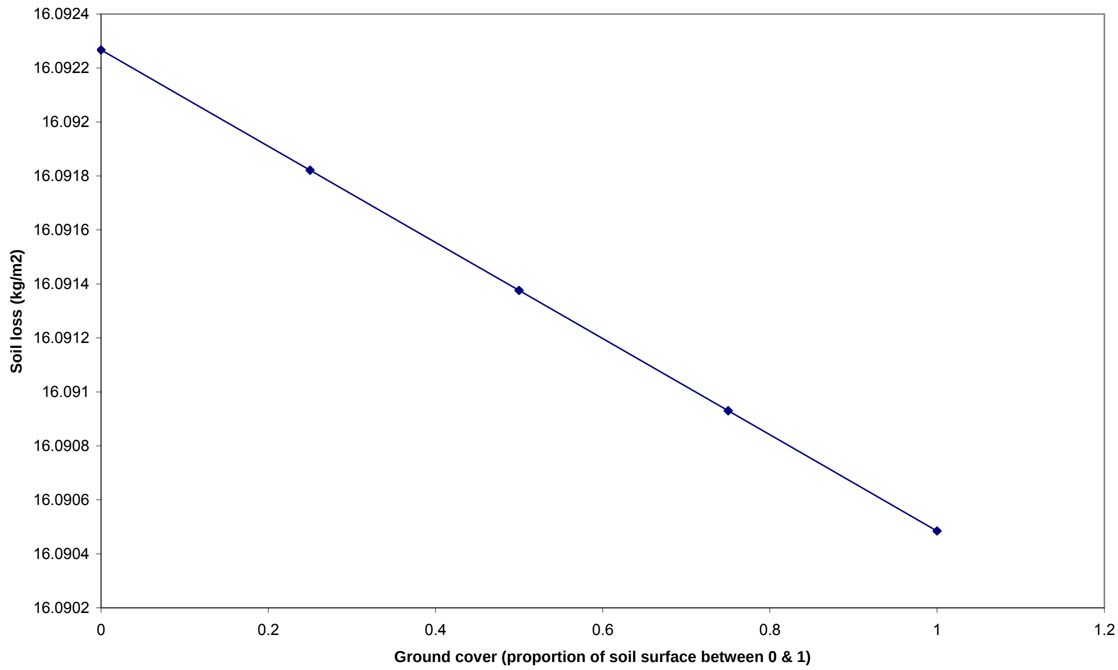



**Figure 7.16 Relationship between ground cover and soil loss.**

## 7.5.14 Plant height

Soil loss and particle detachment showed moderate sensitivity to plant height with an increase in plant height resulting in an increase in each sensitivity indicator. Plant height determines the height of fall of the raindrops. An increase in this will increase the kinetic energy reaching the soil surface and, therefore, the amount of detachment that takes place by leaf drip.

## 7.5.15 Stem diameter and density

Soil loss and particle detachment show a high sensitivity to stem diameter and density. An increase in either produces a decrease in soil loss and in particle detachment with these outputs being slightly more sensitive to stem density than to stem diameter. These results are unsurprising since dense plant stems slow runoff and reduce erosivity (Toy et al., 2002). Stems act to absorb some of the energy of falling raindrops and running water so that less is directed at the soil (Morgan, 2005). It is not possible to compare, for these factors, the sensitivity of the model with other models as they have not been used in VFS modelling. If this relationship can be validated with data from laboratory or field conditions this has important implications for modelling VFS performance.

### 7.5.16 Particle detachment by rainfall and runoff (clay, silt and sand)

On the bare soil element there was generally low sensitivity to the detachment factors for both soil loss and particle detachment. On the grass element the model was only sensitive to the detachment of clay by runoff and rainfall. Although, in the model and in reality, silt is the most detachable particle size, once cohesion is overcome and clay particles have become detached they then have a lower entrainmend than sand or silt. However, within the model the most important effects are through the particle fall number and its influence on deposition. Sands and silts have higher particle fall numbers than clay and will be more likely to be deposited. This is especially true with a vegetation cover where the fall numbers will be even higher because of the reduced velocity (Morgan, pers. comm., 2005). With the remaining suspended sediment (i.e. soil loss) being dominated by clay it is more likely to be sensitive to changes in the detachment of clay.

It is possible, within the model, for one of the particle size fractions (sand, silt and clay) to be transport-limited whilst the other fractions are detachment-limited and vice-versa. Examination of this provides a better understanding of the sensitivity of the model. On the bare soil element within the annual version of the model, for the majority of cases, the clay and silt fractions are transport-limited and the sand fraction is detachment-limited. This means that, for clay and silt, transport capacity has more influence than detachment i.e. the amount of material detached is less important than the amount that can be transported. It is, therefore, unsurprising that the model shows low sensitivity to differences in the detachment factors for clay and silt. However, it is surprising that the erosion of sand particles is detachment-limited since these are more difficult to transport than clay or silt and are deposited more easily. This was deemed to be due to the high detachability value for sand and, therefore, during model testing different detachment indices were tested (Chapter 8). This will also depend on the percentage of sand and silt in the soil and hence the quantities of both remaining for transport after the first phase of deposition.

On the grass element, within the annual model, clay is transport capacity limited and silt and sand are detachment-limited. In fact, the detached silt and sand particles are immediately deposited and so are not mobilised by the runoff i.e. everything is controlled by the particle fall number at this stage. Consequently, the model is insensitive to changes in their detachment factors. The clay fraction is the only mobile fraction and so is the only fraction to become transport-limited. The model shows a moderate sensitivity to rainfall detachment of clay and a low sensitivity to runoff detachment of clay. This sensitivity to the clay fraction, despite it being transport-limited, is due to it being the only size fraction to contribute to the total sediment detachment.

Within the single event version of the model, the only difference is an increased sensitivity of both soil loss (to rainfall detachment of silt) and of particle detachment (to runoff detachment of silt) on the bare soil plot. The transport of sediment is now transport capacity limited for clay, sand and silt.

### 7.5.17 Manning's n

The sensitivity of Manning's *n* varied with indicator, element cover (i.e. grass or bare) and version of the model. Sensitivity was highest for the soil loss on the bare soil plot. Runoff was only sensitive on the grass plot within the single event version of the model. Particle detachment was moderately sensitive on the bare plot within the single event model but showed low sensitivity within the annual model. Soil loss on the grass plot was only sensitive within the annual model and not within the single event model. It is not surprising that greater sensitivity was observed overall for the bare soil plot because flow velocity for bare soil elements is calculated within the model based on Manning's *n*. Flow velocity for grass elements, however, is calculated based on the equation of Jin et al., 2000 (Equation 7.5) which takes into account stem density and diameter and, therefore, Manning's *n* is not relevant for vegetated elements unless the flow has become channelled into rills, i.e. the lack of sensitivity is an artefact of the model. Roughness imparted by the vegetation is accounted for implicitly by D (m) and NV (elements $m^{-2}$).

$$V_v = \left( \frac{2g}{DNV} \right)^{0.5} s^{0.5} \qquad \textbf{Equation 7.5}$$

Where $V_v$ is the flow velocity of the vegetated element, $g$ is gravitational acceleration, $D$ is stem diameter, $NV$ is stem density and $s$ is slope.

### 7.5.18 Flow depth

On the bare soil plot both soil loss and particle detachment showed a moderate sensitivity to flow depth. Under grass the particle detachment showed a moderate, and the soil loss a low, sensitivity to flow depth. An increase in flow depth produced an increase in soil loss and in particle detachment. Within the model deeper flow increases soil loss and particle detachment by a) increasing flow velocity, which increases transport capacity, and b) decreasing the particle fall number, which in turn decreases the percentage of sediment deposited on the element. The model does not take into account the effect of water depth on particle detachment by rainfall. It is the condition of rainfall-impacted flow which initiates detachment and mobilisation of particles and hence is the state in which most erosion takes place. Further work may therefore determine how well modelling initial deposition simulates this process compared to other models.

# *7.6 Sensitivity analysis results*

## 7.6.1 Sensitivity analysis results: soil type

By varying each parameter in turn, whilst the values of the other parameters are held constant, artificial conditions are created that could not exist naturally in the field. For example if the value of the moisture storage of the soil is altered from 0.2 to either 0.05 or 0.45 it is no longer typical of a sandy loam soil. The soil particle distribution should also change which then means that the bulk density should be altered too. In order to go some way to addressing this, the model was assessed for its response to changes in soil type. Since the changes cannot be represented on a linear scale, the interpretation of *ALS* values is impossible. Instead the results are assessed based on their agreement with published trends.

The model was run for each soil type from the USDA soil texture classification. The base values from the ALS were used for all of the input parameters except for those parameters related to the difference in soil texture. Those parameters are percentage clay, silt and sand, moisture storage and bulk density and input values for these were taken from the MMF-VFS look up table (Table II in Appendix 2). (Saturated lateral permeability values would also vary with soil texture but this was not included in the analysis because only a single element was assessed.) The model output is presented in Figure 7.17.

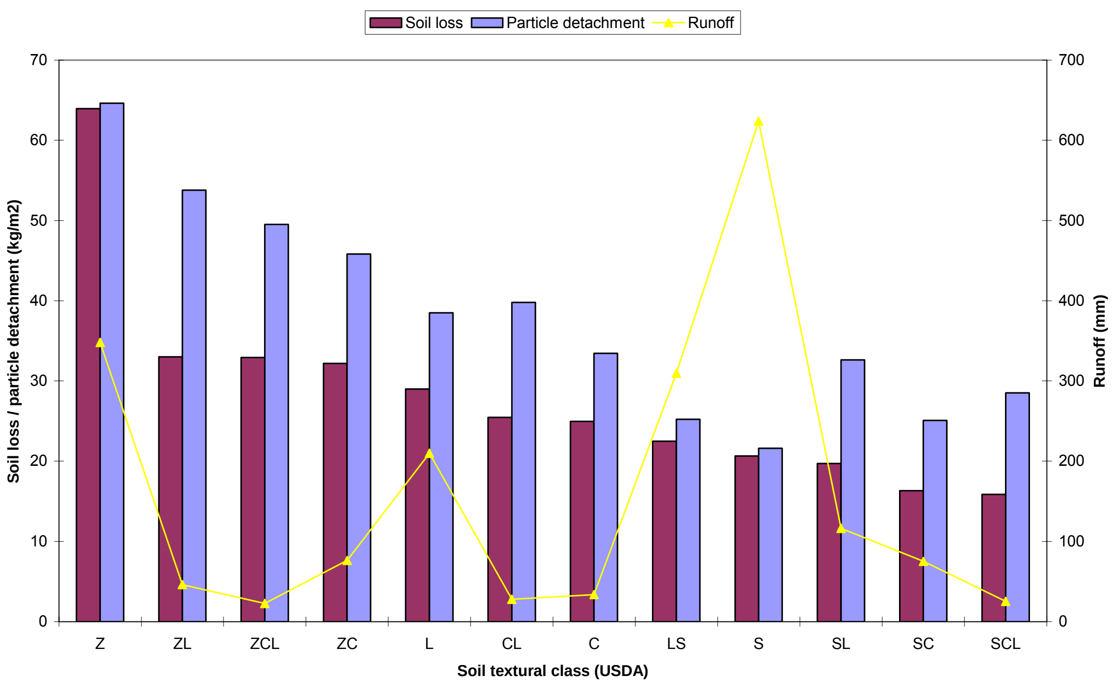



**Figure 7.17 MMF-VFS output for different soil textural classes (Z = silt, ZL = silty loam, ZCL = silty clay loam, ZC = silty clay, L = loam, CL = clay loam, C = clay, LS = loamy sand, S = sand, SL = sandy loam, SC = sandy clay, SCL = sandy clay loam).**

The results from Figure 7.17 are compared with the soil erodibility factor (K) used within the Universal Soil Loss Equation (USLE). K represents the average soil loss in mass per unit area for a particular soil. It is, therefore, a measure of the susceptibility of soil particles to detachment and transport by rainfall and runoff. Texture is the principal factor affecting K (although structure, organic matter and permeability will contribute). K values are presented in Table 7.7. Spearman's rank correlation of the ranked order of the soil loss values generated by the model and the ranked order of the published K factor values showed no difference with both sets of values showing increasing soil loss from the sands to the clays to the silts.

Caution should be exercised when comparing the model output with that of another model. A good correlation may be merely a reflection of the error in both models. However, in this case parameter values in the MMF-VFS (soil particle detachability by raindrop impact and by runoff) have been determined independently of K.

| **Textural class** | **Soil loss (tonnes per acre per unit of R where R is the rainfall factor of the USLE)** |
|---|---|
| Very Fine Sand | 0.43 |
| Loamy Very Fine Sand | 0.39 |
| Silt Loam | 0.38 |
| Very Fine Sandy Loam | 0.35 |
| Silty Clay Loam | 0.32 |
| Loam | 0.3 |
| Clay Loam | 0.3 |
| Silty Clay | 0.26 |
| Clay | 0.22 |
| Sandy Clay Loam | 0.2 |
| Fine Sandy Loam | 0.18 |
| Heavy Clay | 0.17 |
| Sandy Loam | 0.13 |
| Loamy Fine Sand | 0.11 |
| Fine Sand | 0.08 |
| Coarse Sandy Loam | 0.07 |
| Loamy Sand | 0.04 |
| Sand | 0.02 |

**Table 7.7 Soil loss values from the USLE based on measured values from the US Soil Conservation Services (US SCS, 2005).**

## 7.6.2 Sensitivity analysis results: land cover

Artificial conditions may also be created by adjusting individual plant parameters in isolation, for example, changes in the density of stems would normally occur with changes in stem diameter, canopy cover and plant height. In order to assess sensitivity to different groups of plant parameters the model was run for each of the land cover types within the MMF-VFS guide value table (Table III in Appendix 2). The input parameters that vary with land cover type are effective hydrological depth, permanent interception, evapotranspiration, canopy cover, ground cover, stem diameter and stem density. The soil loss, particle detachment and runoff output is presented in Figure 7.18.

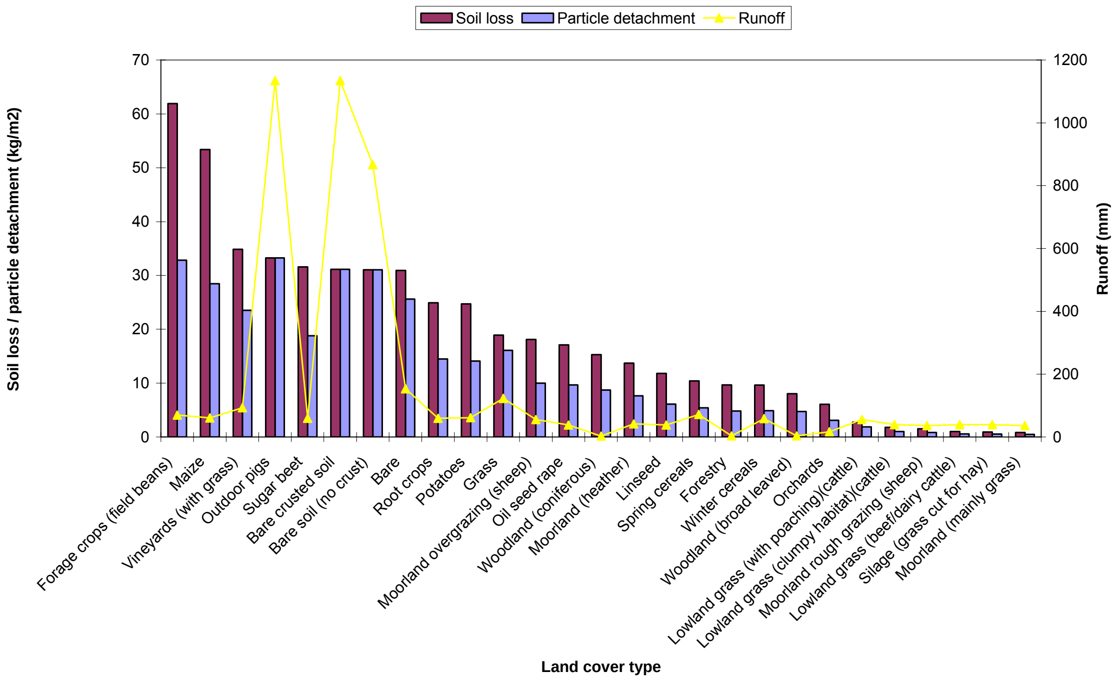



**Figure 7.18 MMF-VFS output for different land cover types**

The broad generic groups of bare soil, arable, grass and woodland, appear to be ranked as may be expected i.e. soil loss decreases from bare soil, through both spring and winter crops, to woodland and then grass. This corresponds with the level of cover that would be expected from these land cover types and hence the degree of erosion prevention. The extremes of land cover type also appear to be at the right end of the graph in Figure 7.18 with outdoor pigs exhibiting a high soil loss and moorland grass the lowest. Evans (2006) ranked crop types according to risk of erosion. This shows a poor comparison with the modelled categories (Table 7.8). However, the risk assigned by Evans (2006) takes into account factors such as crop growth on vulnerable soils and in ways that promote runoff. This is not taken into account by the model.

| *Crops ranked by erosion risk (Evans, 2006)* | *Crops ranked by model results* |
|---|---|
| Outdoor pigs | Outdoor pigs |
| Sugar beet | Field beans |
| Maize | Maize |
| Potatoes | Bare |
| Bare | Sugar beet |
| Spring cereals | Potatoes |
| Winter cereals | Oil seed rape |
| Field beans | Spring cereals |
| Oil seed rape | Winter cereals |

**Table 7.8 Land cover categories ranked in order of erosion risk based on Evans (2006) and on predicted results from the MMF-VFS model.**

## *7.7 Summary of sensitivity analysis results*

### 7.7.1 Implications for model use

Sensitivity analysis evaluates the importance of different parameters within a model to the prediction of one or more output values. In this case, the parameters which exhibit higher sensitivity to the prediction of soil loss will be more important in erosion control. It can be deduced from the analysis that:

a) For bare soil, runoff is highly sensitive (ALS > 1.0) to the parameters relating to rainfall, evapotranspiration and soil moisture storage. Soil loss is highly sensitive to rainfall amount and moderately sensitive (ALS $\geq$ 0.5 < 1.0) to soil moisture storage, through its effect on runoff, and to slope angle, Manning's *n* and flow depth.
b) For vegetated conditions, the sensitivity of runoff remains the same but soil loss is highly sensitive to rainfall, the number of rain days and the diameter and density of the plant stems. It is moderately sensitive to canopy cover, plant height and the detachability of the clay particles.

Alterations to input parameter values to which the model is highly sensitive are likely to produce the greatest changes to model output during the calibration stage of the model development. During the validation stage it is these factors that should be measured with the most accuracy since they will have the most impact on the model output. This accuracy may come in part from using measured values, where possible, instead of guide values. Grayson and Blöschl (2001) discuss a number of issues pertinent to the value of measured values in hydrology. The following section summarises the main points on problems associated with the scale and accuracy of measuring hydrological parameters including the error that may potentially be introduced by measurement.

### Measuring model parameters: Scale

Ideally measurements should be taken at a temporal and spatial scale that incorporates all of the natural variability that influences a particular hydrological feature (Grayson and Blöschl (2001)). In practice this is rarely logistically feasible. Figure 7.19 illustrates a) the problem of spacing too sparsely and hence not capturing the small-scale variability, and b) not extending the data far enough and hence losing the large-scale variability, which is instead translated into a trend in the data.

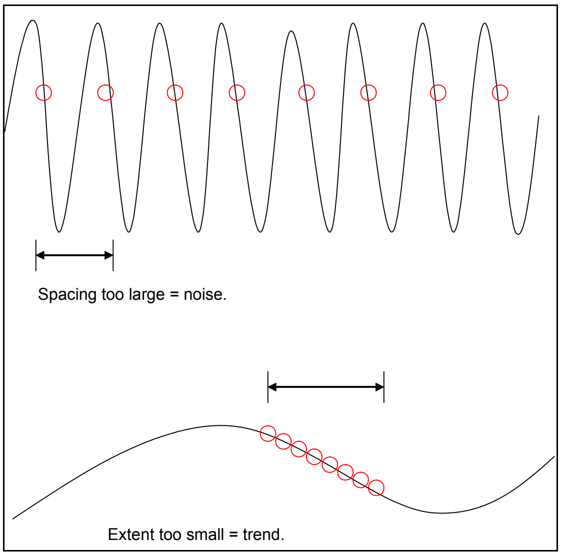



**Figure 7.19 Schematic illustrating the problem of spacing measurements too sparsely (i.e. missing small-scale variability) and too close (i.e. missing large-scale variability). The circles are the measurements and the line is the "true" hydrologic pattern. Adapted from Grayson and Blöschl (2001).**

The scale of the measurements and the spatial variability of the pattern of hydrological processes will determine how representative multiple point measurements are. If the process shows large-scale variability then patterns may be represented by a sparse sampling scheme. A more dense sampling method will be required where there is a lot of small-scale variability. “The key to a successful representation of spatial patterns in catchment hydrology is to maximise the number of sampling points in space, that cover an extent sufficient to capture the process of interest” (Grayson and Blöschl, 2001).

## 7.7.2 Measuring model parameters: Accuracy

Measurement errors may be systematic or random. A systematic error may be introduced by using an inappropriate experimental set up. It may be possible to correct for systematic errors where further, more accurate, data are available for comparison. A random error may be introduced by taking inaccurate readings or using faulty equipment. It is not possible to remove random errors but they may be reduced by taking multiple measurements of the same variable (Grayson and Blöschl, 2001).

### 7.7.3 Use of the MMF-VFS in meeting the study objectives

The model behaves rationally in effecting the expected changes in soil loss and runoff with alterations to the input parameters. This is true for both a bare soil and a vegetated element. Output from the sensitivity analysis shows that, for bare soil conditions, erosion is related to rainfall, soil and slope conditions. However, when a vegetation cover is simulated it outweighs soil and slope in influencing erosion. This supports the use of the model in evaluating vegetation-based erosion control strategies.

The analysis of the model output by soil type and by land cover provides further confidence in the model behaviour and its ability to represent land cover types typical of UK agriculture. As would be expected in natural field conditions the soil loss values showed a decrease from silt to clay to sandy soils. The ranked order of soil loss from the land cover categories showed broad agreement with those measured by other researchers. In relation to the thesis hypotheses set out in Chapter 1, the sensitivity analysis allowed enough confidence to be placed in the use of the model to proceed to model testing and evaluation. The following two chapters describe testing and further development of the model against independent datasets, and against the field data described in Chapter 5, before incorporation into tools for VFS design and placement (Chapter 10).

# Chapter 8 Model testing

## 8.1 Introduction

Following sensitivity analysis of the MMF-VFS model (Chapter 7) the next stage in preparing the model for use in developing VFS design and placement tools was testing and further development. This chapter demonstrates the setting up and running of procedures for parameter estimation and model testing with respect to evaluation of VFSs. Parameter estimation may be achieved by considering the rationale of the model when applied to different conditions or by analysis of observed input and output data. Once parameter values have been selected and calibrated the model needs to be tested or validated by comparing simulated and measured values. Again there are a number of possible techniques available (Pilgrim, 1975). Parameter estimation, selection and model testing were carried out for MMF-VFS with specific consideration for its application to evaluating and designing VFSs. This not only improved the model predictions, when compared with observed results, but also informed on the likely performance of the model for various situations.

## 8.2 Identifying parameters to be estimated

For conditions where measured values are not available a set of parameter guide values are available for the MMF-VFS model (Morgan and Duzant, 2007). However, it is always necessary to determine whether such guide values are suitable for the catchment or application in consideration (Pilgrim, 1975).

The sensitivity analysis provided one method of selecting those parameters to which the model was most sensitive and, therefore, accuracy in parameterisation would be most beneficial, for example rainfall characteristics, vegetation parameters and flow depth (Chapter 7). By using such information, any calibration attempt can proceed in a logical manner (Quinton, 1994). However, a problem with this may arise when a sensitive parameter is difficult to measure. If the parameter is based on a measurement that was taken with a degree of uncertainty then an element of error will be carried through from the model parameter to the model output. This should be considered when interpreting model results (Quinton, 1994). In order to reduce this effect the values selected for the sensitivity analysis were based on a range of data, both published and measured within this study. Parameter estimation also included those parts of the model unique to this version or that had been significantly altered since the previous version, for example the detachability values and, again, the vegetation component.

## 8.3 Methods of parameter estimation

Parameter estimation methods should be appropriate to the purpose of the model testing. In this case testing was required to ensure that the model could be applied to different scenarios and to inform model users on the most effective ways to set up and run the model. It was, therefore, not appropriate to calibrate the model to a particular set of conditions such that it would only be effective in that situation. For this reason parameter values were estimated directly.

### 8.3.1 Direct estimation

Parameter values can be directly estimated based on available knowledge of hydrological processes or on field or laboratory measurements. Pilgrim (1975) suggested that this method is logically satisfying to the user, utilizes all available information and is most likely to give valid results over a wide range of conditions. The method used for the current study consisted of obtaining parameter values from published data (e.g. erosivity indices, flow depth, settling velocities, detachability indices) and, where possible, direct measurement (e.g. vegetation parameters). Alternative techniques for parameter estimation may be based on analysis of observed input and output data for a specific catchment. Selection of the following techniques is dependent on the type of model.

### 8.3.2 Regression techniques

Regression enables one variable, $y$, to be estimated from another variable, $x$, or from several variables. Since the form of the relationship between the variables must be assumed, the method is generally restricted to linear relationships (Pilgrim, 1975). This method applies to regression models.

### 8.3.3 Theoretical relations

Parameters may be derived directly by application of their assumed theoretical relationships. Methods for this vary and selection is generally based on subjective judgement or from experience gained from testing over a wide range of conditions (Pilgrim, 1975).

### 8.3.4 Optimisation using search techniques

Search optimization methods are used systematically to find a set of parameter values giving the best possible reproduction by the model of the observed catchment output. This is used for complex models where direct derivation of parameters is not possible (Pilgrim, 1975).

## *8.4 Calibration*

Calibration describes the process of tuning the parameters so that the model response approximates the observed response. In other words the “best” set of parameter values is selected (Grayson and Blöschl, 2001). Generally, the calibration process is complex due to limitations of the model being tested, of the input and output data and of an incomplete understanding of catchment characteristics. As a result of these limitations a range of values may be derived for the model parameters of a given catchment. When selecting the "best" parameter values from this range it is important to consider what causes the differences between (model) simulated and (field) measured values.

Calibration may be qualitative but generally seeks to provide a quantitative measure of the error between simulations and observations. Grayson and Blöschl (2001) provide the following examples of methods applied to the comparison of observed and predicted streamflow, each of which emphasises a different flow component:

a) Sum of the squares of the difference between simulated and observed flows.
b) Sum of the absolute values of the differences between observed and simulated flows.
c) Sum of the ratio between observed and predicted flows.

Following calibration a model is typically tested against a validation dataset. Again observed and simulated values are compared but the calibration dataset is ideally independent of the validation dataset.

The laboratory experiments (Chapter 4) carried out within this study did not yield sufficient results to constitute a calibration dataset. Published data was obtained for comparison of the model output with an observed response but this was used in the next stage of model testing i.e. validation. Calibration of the model was therefore restricted to the procedure described in Figure 8.1 whereby parameter selection was based on evaluation of the model behaviour and not on the difference between simulated and observed responses. Had consistent model errors been identified at the testing stage parameters could have been adjusted i.e. a return to the parameter estimation stage. A more extensive and quantitative calibration approach would be more relevant if estimation of every parameter was required or if the aim was for the accurate simulation of a specific set of conditions. However, in this case the approach used enabled the evaluation of a range of scenarios and the identification of consistent errors and provided a great deal of insight into the model performance and limitations.

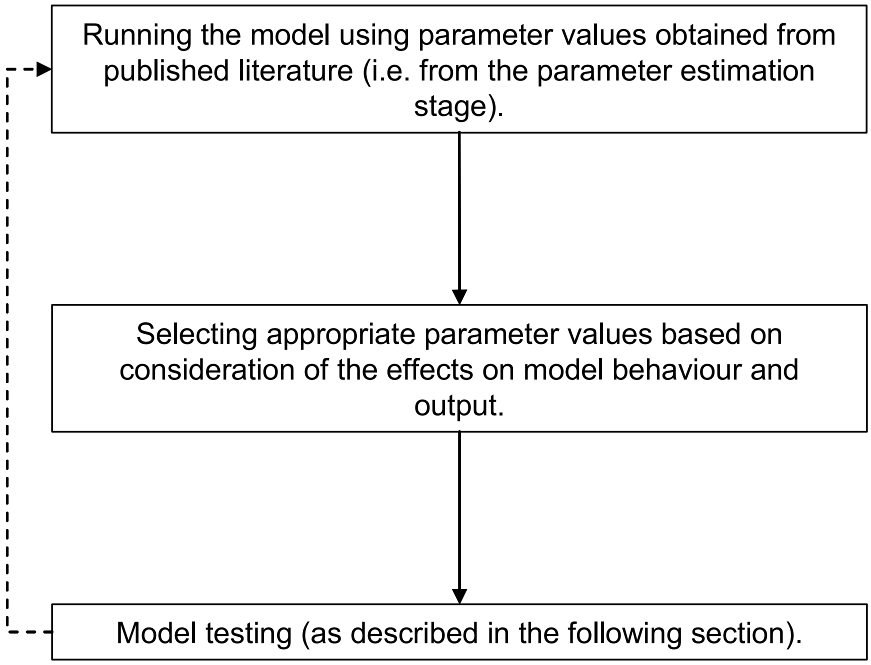



**Figure 8.1 Selecting parameter values. The dotted line represents the return to the parameter estimation stage in the case that the model exhibited consistent errors during testing.**

## *8.5 Testing of the fitted model*

Testing, or validating, of the parameterised model is necessary in order to evaluate its fit against reality. This process attempts to define the uncertainties in predictions or at least provides an idea of their magnitude (Grayson and Blöschl, 2001). A number of general approaches exist for testing or validating a fitted model after the parameter values are determined. The most common of these is 'split-sample testing'.

### 8.5.1 Split sample testing

This procedure involves dividing the available data into two halves, deriving parameter values from the first half (calibration) and using the second half to test the models ability to reproduce the observed data (validation). There are, however, limitations to this method. Pilgrim (1975) observed that both periods sample the same restricted population of conditions and correlations so that satisfactory reproduction of the second half of the data may give little information on the wider validity of the model. It also seems limiting not to use as many observations as available in the derivation of parameter values, rather than basing these values on only half of the data. Quinton and Morgan (1998) found, when testing EUROSEM, that it was difficult to obtain a true split of the data. For example, it was not as simple as taking data for one year for calibration and for another year for validation because of the differences between years in crop type and storm characteristics. This would mean that tillage roughness and crop cover differed for the data in the calibration year to that prevailing in the storms for the validation year. Ideally the split-sample method requires that all parameters are constant except for the parameter being tested.

### 8.5.2 Significance of parameter values

Significance levels may be used to define the validity of a fitted model. However, Pilgrim (1975) suggests that these values should only be used as aids or indicators in assessing validity. That is because statistical significance may not always correspond with physical significance. He recommends a more subjective use of this method alongside all available information and knowledge on the model behaviour.

### 8.5.3 Pattern of residuals

The pattern of residuals from derived relations will provide information on the performance of the model being tested. Patterns in the residuals will indicate inadequacies in the fitted model and may give guidance for the improvement of the model.

Further information on these and other methods is provided by Pilgrim (1975).

## *8.6 Model uncertainty*

Assessment of the calibration and validation steps will determine the confidence or predictive uncertainty that can be placed in simulations. Uncertainty may arise from

i. the input data due to sampling or interpolation errors;
ii. simulated responses due to model parameter errors;
iii. the hypotheses underlying the model and model structure; and
iv. the data against which the model output is calibrated and tested (Grayson and Blöschl, 2001).

Methods are currently being developed, for complex models, to quantify these sources of uncertainty and their effect on overall predictive capability. The most recent approaches make use of the Monte Carlo method. An example of one such approach is Generalised Likelihood Uncertainty Evaluation.

### 8.6.1 Monte Carlo method

The basis of this theory is that multiple simulations are produced using parameters or inputs taken randomly from a particular distribution (Grayson and Blöschl, 2001). The various sets of parameter values take into account spatial variation, and variation induced by parameterisation, and provide a large number of samples (Veihe and Quinton, 2000). Estimation techniques are then applied to the samples and distributions of the estimates are studied in relation to the true parameter values, and to theoretical expectations (Johnston, 1972).

### 8.6.2 Generalised Likelihood Uncertainty Evaluation (GLUE)

The GLUE methodology recognises the need for quantification of predictive error and provides a framework for estimating the predictive uncertainty of models. The GLUE approach, used by Brazier et al. (2001) to assess levels of uncertainty associated with the WEPP model, is summarised below.

The equifinality problem (Beven, 1993, 1996) recognises that many model structures or parameter sets, within a given model framework, will predict a required output to satisfy a chosen objective function. Therefore, no single optimum set of model parameters will be readily identifiable. The GLUE methodology targets this problem by assigning a likelihood to each model that it can predict the system and that the set of parameters provides an 'acceptable' simulation of the observed. A likelihood measure will assess the goodness of fit of a simulation to the observed data. The models (produced in this case using Monte Carlo simulations to sample the parameters) are weighted according to the cumulative likelihood measure for the set of parameters. Based on this, uncertainty bounds or confidence limits are chosen to represent uncertainty. The uncertainty bounds incorporate all of the errors associated with the sources of uncertainty listed above (in Section 8.6).

## *8.7 Testing of the MMF-VFS*

As described earlier, published data was obtained for model testing. Datasets were selected on the basis that input values were available for at least the most sensitive model parameters. Basic information on the selected datasets is provided in Table 8.1. However, the small number of conditions and replicates represented by the validation dataset precluded the use of the split-sample method. The method was therefore based on '8.5.2' and '8.5.3' above as well as a rational assessment of the processes occurring within the model and their representation of field processes.

Initially the main patterns in the measured data were examined and compared with the patterns shown by the predicted data. Analysis of variance was performed to see whether the same factors influencing the experimental results were influencing the model results. The magnitude of measured and predicted values were compared to give a coarse evaluation of the model data. The ranked order of values was compared to determine whether the patterns predicted by the model reflected those of the measured data. The means of the predicted and measured values were compared using paired t-test statistics and finally a Nash and Sutcliffe (1970) statistic was calculated to determine the efficiency of the model. The dataset was considered too small to test uncertainty using any of the methods described in the previous section.

| ***Dataset*** | ***Author*** | ***Max. plot size ($m^2$)*** | ***Slope angles (°)*** | ***Rainfall intensities (mm/hr)*** | ***VFS details*** | ***Variable(s) for testing*** |
|---|---|---|---|---|---|---|
| 1 | Boubakari (1992) | 2.5 | 12, 14, 16 | One simulated storm 30 | Live grass strips (Poa pratensis and Festuca ovina) | Soil loss & runoff |
| 2 | Dillaha (1989) | 150.7 | 3, 6, 9 | Six simulated storms 50 mm/hr | 4.6 and 9.1 m Orchard grass | Soil loss & runoff |
| 4 | Le Bissonnais et al. (2004) | 480 | 3 | Natural rainfall, measurements over 36 events with > 5 mm | 3 and 6 m Ray grass | Soil loss & particle size distribution |

**Table 8.1 Summary of datasets used for model testing.**

# *8.8 Results and discussion*

## 8.8.1 Parameter estimation

### 8.8.1.1 Energy of direct throughfall

The erosivity of a rainstorm is a function of its intensity and duration, and of the mass, diameter and velocity of the raindrops. To compute the erosivity of a rainfall event, therefore, requires an analysis of the drop size distributions of the rain (Morgan, 1995). There is, however, no constant relationship between the drop size characteristics and the intensity of the rainfall. The variation has been overcome in soil erosion research by deriving general relationships between kinetic energy and rainfall intensity. Table 8.2 gives six examples of different erosivity indices and the geographical region they have been based on.

| ***Erosivity index*** | ***Reference*** | ***Based on or considered suitable for*** |
|---|---|---|
| $E = 0.0119 + 0.0873\ log_{10} I$ | Wischmeier & Smith (1958) | Based on studies in Eastern USA. |
| $E = 0.0895 + 0.0844 \log_{10} I$ | Marshall & Palmer (1948) | Considered representative of a wide range of environments including the UK. |
| $E = 298(1-4.29/I)$ | Hudson (1965) | Based on measurements in Zimbabwe. Recommended for tropical rainfall. |
| $E = 0.0981 + 0.1125 \log_{10} I$ | Zanchi & Torri (1980) | Based on research in Italy. |
| $E = 0.0981 + 0.106 \log_{10} I$ | Onaga, Shirai & Yoshinaga (1988) | Based on work in Japan. |
| $E = 0.283(1-0.52e^{0.042I})$ | Van de Dijk et al. (2002) | Proposed as a universal relationship. |

**Table 8.2 Erosivity indices: relationship between kinetic energy of rain (KE, MJ $ha^{-1}mm^{-1}$) and rainfall intensity (I, mm $h^{-1}$). From Morgan (2005).**

Being widely representative, especially of UK conditions, the erosivity index derived by Marshall and Palmer (1948) was used for running the model at the field scale. In some cases results for the laboratory scale experiments could be improved slightly by using the equation obtained by Hudson (1965) but generally this made little difference. This could be because the rainfall intensities considered in this study were relatively low compared to those studied by Hudson (1965). He observed a change in the relationship between median drop size with rainfall intensity at 100 mm/hr whilst the highest intensity tested with the MMF-VFS was 60 mm/hr.

#### 8.8.1.2 Vegetative parameters

Being a recent addition to the model the vegetation component has had very little previous testing. Furthermore, the validity of this component is extremely important for application of the model to evaluating VFS performance. The effect of vegetation is modelled using permanent interception, ground cover, canopy cover, plant height, effective hydrological depth, stem diameter and number of vegetative elements.

#### 8.8.1.3 Defining vegetative parameters in the MMF-VFS

When using vegetative parameters it is important to clarify the measurements required. For example, where grass has tillered and grown out laterally stem diameter (D) refers to the base of the clump or tussock that it forms and not the individual stems (Figure 8.2). This is the measurement of interest because it is this that will intercept the flow of runoff through the vegetation. Similarly, the number of vegetative elements (NV) refers to the number of whole plants, as represented in Figure 8.2, and not the individual stems. Plant height (PH) should be measured as the height from which drops fall from the canopy to the ground surface. In multi-layered canopies, e.g. trees, it may be the height of the lowest canopy rather than the maximum height. Canopy cover (CC) refers to the proportion of the plant covering the ground as viewed from above whilst the ground cover (GC) is that proportion of the plant cover as viewed when facing the flow. The ground cover therefore refers to the percentage of the soil surface covered with some form of plant material which therefore imparts roughness to the flow, reduces flow velocity and protects the soil from particle detachment by surface runoff. This should include litter and low-growing canopies.

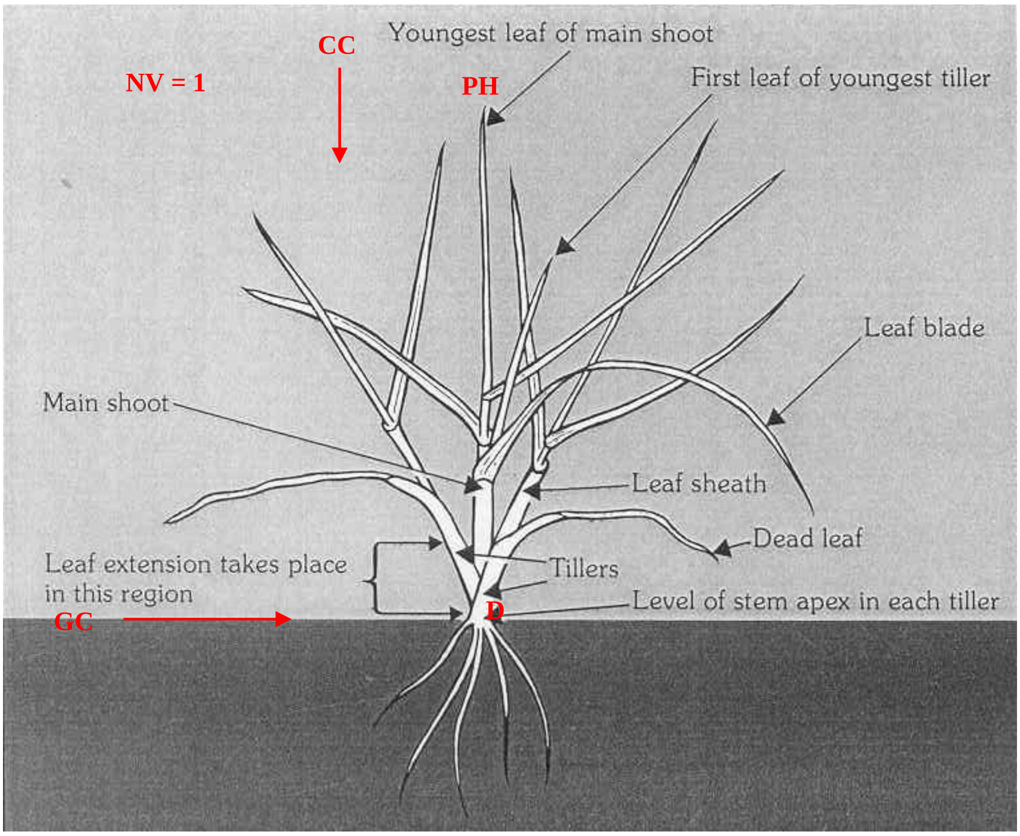



**Figure 8.2 Model vegetative parameters. Modified from Morgan and Rickson (1995).**

### 8.8.1.4 Measuring vegetative parameters

Some of the parameter values proved difficult to obtain or measure, particularly with the degree of accuracy that the sensitivity analysis implies is necessary, and many showed a high degree of natural variability. This may limit the use of such vegetative parameters in the model. For example, based on the definition of ground cover given above, the percentage should be calculated on a unit width rather than an area, for example taking a 1 m width of vegetation at right angles to the flow. In order to consider the complete VFS and not just the first row of vegetation, one approach would be to take a photograph of the VFS with a white board placed behind the vegetation (Figure 8.3). From this it would be possible to measure how many millimetres of white board and vegetation are visible respectively, and from this, the percentage bare ground is calculated. This approach realises that if flow impacts on a plant element somewhere in the VFS, roughness will be imparted and velocity will be reduced. However, as shown in Figure 8.3, the same result may be gained from both position X2 and X3. Due to the alignment of the stems, both of the VFSs would obscure a white board placed behind them yet, due to the number of rows, they have very different proportions of bare ground. This raises the question of how far into the vegetation barrier to place a white board, particularly if the area concerned is not a strip but a whole field. At some distance it is likely that most vegetation elements will have another plant in front of them and therefore extending the distance further will have little effect on the measured GC.

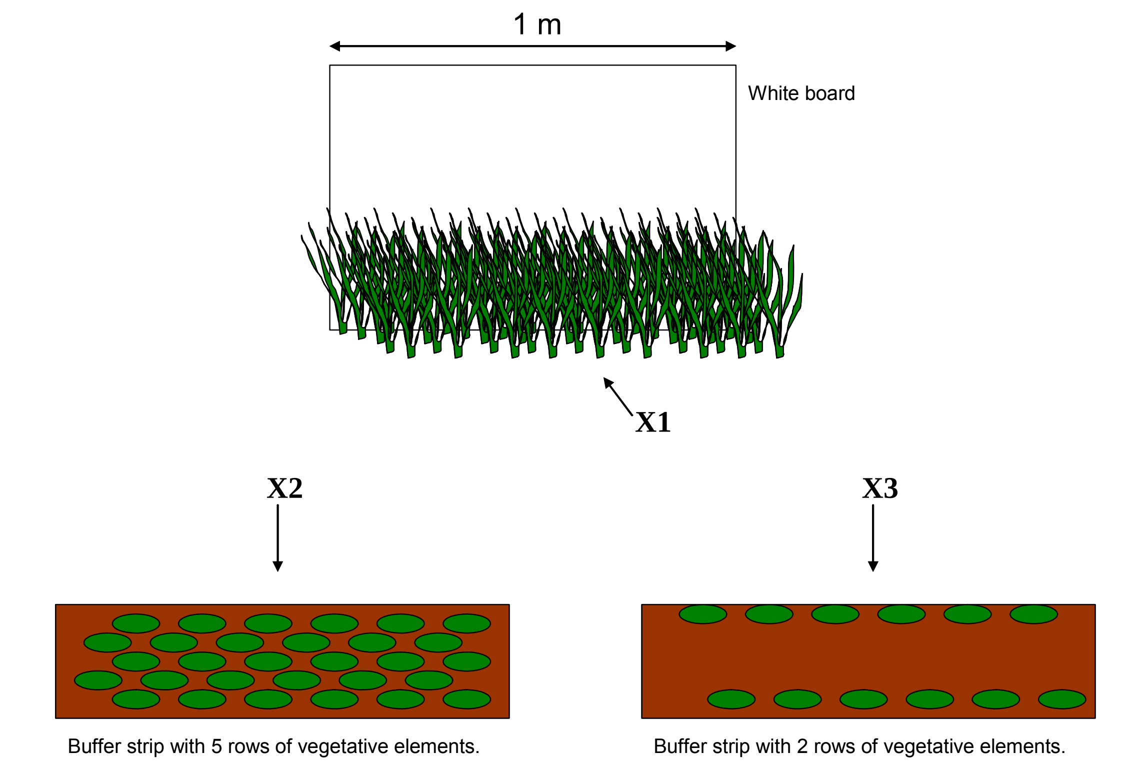



**Figure 8.3 Measuring ground cover as a unit width. From position X1 (section view) ground cover may be measured as the proportion of vegetative elements covering the white board. However, using this method from positions X2 and X3 (plan views) will provide the same result for different buffer strips.**

For one of the datasets used for model verification, photographs had been taken looking into the front of the VFSs. Ground cover was obtained from these by recording, every 5 mm along a transect, whether vegetation or soil was visible in order to calculate percentage cover.

The number of vegetative elements and average stem diameter had been measured for three of the laboratory datasets. It was not feasible to obtain, from published literature, typical measurements for species used in established field VFSs and so guide values were used for simulating the other datasets. However, the field component of this study revealed that there was a large amount of variation in stem characteristics between the VFSs investigated which makes it difficult to justify universally applicable values for NV and D. They are also likely to vary with age and condition of the VFS. Given that the sensitivity analysis showed the model to be sensitive to these parameters this could potentially limit the accuracy of model predictions where field measurements are not available.

### 8.8.1.5 Flow

Flow velocity calculations within the model take into account whether the element is vegetated and, for a bare soil, whether rills exist. For vegetated conditions flow velocity is estimated as a function of slope and the density of the vegetation. In theory this approach should be applicable to all plant types, rather than relying on broad estimates of Manning's *n* from tables where values exist for only a limited number of plant types.

Where rills exist they are modelled based on rill depth. Although, in reality, rill flow is concentrated into small channels, the model assumes that the flow is uniformly distributed across a slope. This is because of the difficulty in determining the number of rills in an erosion event. It is common in modeling applications to lump many single rills together in this way (Ogunmoken, 1990).

Where available, actual rill depth measurements were used in place of MMF guide values. Photographs taken during some of the experiments indicated that some VFS vegetation is more effective at preventing rill formation than others. Using this information generally improved the model predictions and therefore field data should be used where possible. This confirms the results of the sensitivity analysis which showed that flow depth was one of the most sensitive parameters.

### 8.8.1.6 Settling velocities

The values for settling velocity are different in the two stages of deposition in the model (Figure 6.1) because it was found during the model testing that not enough sediment was deposited at the second phase of deposition. Cheng (1997) and a number of others derived settling velocity equations from experiments carried out in sediment free water but Lovell and Rose (1991) found higher settling velocities in multi-particle settling. Whilst investigating entrainment theory during sediment deposition through a buffer strip Rose et al. (2003) observed an increased rate of deposition during a constant influx of sediment-laden water. They therefore applied a correction factor to Cheng's equation in order to increase the settling velocity. Whilst this provides some justification for the alterations made to settling velocity within this study it should be noted that the work of Rose et al. (2003) dealt with aggregates which are likely to be larger and heavier than individual particles.

Thonon et al. (2005) investigated sediment settling on floodplains and proposed

$$w_s = aD^b$$ **Equation 8.1**

where $w_s$ is the floc settling velocity (mm/s$^{-1}$), $D$ is the grain size (equivalent spherical diameter) (µm), $a$ = 2.7 x 10$^{-4}$ and $b$ = 1.57. This equation gives higher settling velocities and again is likely to be dealing with sediment-laden flow. Further settling velocities are provided in Table 8.3.

| *Particle size fraction* | *MMF (2005)* | *Thonon (2005)* | *Sha (1956)* | *Zhang (1989)* | *Van Rijn (1989)* | *Zhu & Cheng (1993)* |
|---|---|---|---|---|---|---|
| Clay | 0.00000225 | 0.0000008 | | | | |
| Silt | 0.002 | 0.0002 | 0.0022 | 0.0021 | 0.0029 | 0.0022 |
| Sand | 0.0225 | 0.001 | 0.017 | 0.02 | 0.02 | 0.017 |

**Table 8.3 Some settling velocities per particle size fractions (m $s^{-1}$) for sediment-laden flow. Table adapted from Cheng (1997).**

### 8.8.1.7 Detachability indices

The model includes guide values for detachability of soil by runoff and by raindrop impact taken from the work of Quansah (1982). Quansah produced values for sand, clay, clay loam and sandy loam. Estimates for other soils were based on relative differences in detachability obtained by Poesen (1985) and used in EUROSEM. It should be noted, however, that the absolute values given by Poesen (1985) are an order of magnitude higher than those measured by Quansah (1982). This may relate to experimental conditions such as soil preparation. Quansah's data for different soil types were used in the original MMF model and shown to work. However, the scientific basis for the values selected is small and Quansah's work was not validated in the field, so alternative values were tested for use in MMF-VFS.

Using the EUROSEM values for detachability by raindrops created too much detachment for the field data but improved predictions of the laboratory results. This may be because the soil in the laboratory experiments had already been sieved and disturbed. With its structure already broken down it was more likely to be readily detachable. A problem with the EUROSEM values, however, is that they apply to soil textural classes (e.g. sandy loams, silt loams, clays) rather than clay, silt and sand properties. Therefore, they do not conform to the generally accepted understanding that silt particles are the most detachable. Mean detachability values for clay, silt and sand can be derived from Poesen (1985) and these were tested. Generally those of Quansah gave the most accurate model predictions although "the detachability value for clay should be treated with care since, as shown by Poesen (1985) and Chisci et al. (1989), the detachability of clay particles has a very high variability depending on the aggregate stability of the soil and the type of clay" (Morgan and Duzant (2007).

# *8.9 Testing of the model*

## 8.9.1 Testing individual datasets:- Dataset 1: Boubakari (1992)

### 8.9.1.1 Measured data

Experiments were carried out by Boubakari (1992) on a 1 m wide, 2.5 m long indoor flume in order to test the effectiveness of *Festuca ovina* and *Poa pratensis* as VFSs for erosion control. Treatments consisted of both bare soil and buffered runs on slopes of 12°, 14° and 16°. Soil loss and runoff were collected from the bottom of the slope during three replicates of each treatment. Analysis of variance (ANOVA) was performed on the soil loss and runoff data in order to investigate the influence of slope and grass species. The soil was a sandy loam.

### 8.9.1.2 Comparing trends in measured and predicted values

The trends shown by the measured data were compared with those of the MMF-VFS predicted results for soil loss and runoff (Table 8.4 and Table 8.5). Both datasets showed a significant difference in the soil loss between grass species and between slope (Table 8.6). Both datasets showed a significant difference in the soil loss and runoff with treatment i.e. bare soil compared to buffered. Neither dataset showed a significant difference in the runoff data between the slopes. Only the predicted data set showed a significant difference in runoff between the grass species (Table 8.7). The following relationships were also evident within both datasets. Soil loss was greatest for the bare soil and least for the *F. ovina* treatment; soil loss increased with slope on all treatments, the rate of increase being greatest for the bare soil and least for the *F. ovina*; the soil loss increased more rapidly as slope increased from 14° to 16° than when it increased from 12° to 14°; with the grass barriers soil loss was greater on the 16° than on the 12° slope but the relationship with changing slope angle was less clear (Table 8.4).

| *Soil loss (kg/plot)* | Festuca ovina | | Poa pratensis | | *Bare soil* | |
|---|---|---|---|---|---|---|
| Slope (°) | Measured | Predicted | Measured | Predicted | Measured | Predicted |
| 12 | 2.24 | 2.32 | 2.72 | 3.4 | 7.76 | 7.72 |
| 14 | 5 | 2.36 | 2.84 | 3.48 | 12.48 | 8.2 |
| 16 | 4.92 | 2.4 | 7 | 3.52 | 20.56 | 8.64 |

**Table 8.4 Measured (average of three replicates) and predicted total storm soil loss.**

| *Runoff (mm)* | Festuca ovina | | Poa pratensis | | *Bare soil* | |
|---|---|---|---|---|---|---|
| Slope (°) | Measured | Predicted | Measured | Predicted | Measured | Predicted |
| 12 | 8.14 | 10 | 6.93 | 10 | 6.86 | 26 |
| 14 | 9.59 | 10 | 8.6 | 11 | 9.13 | 27 |
| 16 | 16.36 | 10 | 10.46 | 11 | 9.33 | 27 |

**Table 8.5 Measured (average of three replicates) and predicted total storm runoff.**

| *Variable* | *DF* | *F ratio* | *Probability* |
|---|---|---|---|
| Slope angle | 2 | 10.25 | <0.001 |
| Treatment (bare & grass) | 2 | 2812.43 | <0.001 |
| Grass species | 1 | 2755.6 | <0.001 |

**Table 8.6 Analysis of variance for predicted total storm soil loss data.**

| *Variable* | *DF* | *F ratio* | *Probability* |
|---|---|---|---|
| Treatment (bare & grass) | 2 | 796.000 | <0.001 |

**Table 8.7 Analysis of variance for predicted total storm runoff data.**

### 8.9.1.3 Comparing actual measured and predicted values

A Wilcoxon Matched Pairs test showed that there is no significant difference in the ranked order of the predicted and measured values ($P$ = <0.05). A paired t-test showed that there is no significant difference between the means of the predicted and measured values ($P$ = <0.05). An ANOVA on the difference between the actual and predicted results showed that the model performs equally as well for each of the slopes and treatments tested. The $r^2$ of the relationship between the predicted and actual results, 0.6, shows that the model can predict 60% of the variation in measured soil loss. The intercept value in the regression equation shows that the values are under-predicted by just over half. Also there is no 1:1 relationship. The regression equation is showed on Figure 8.4

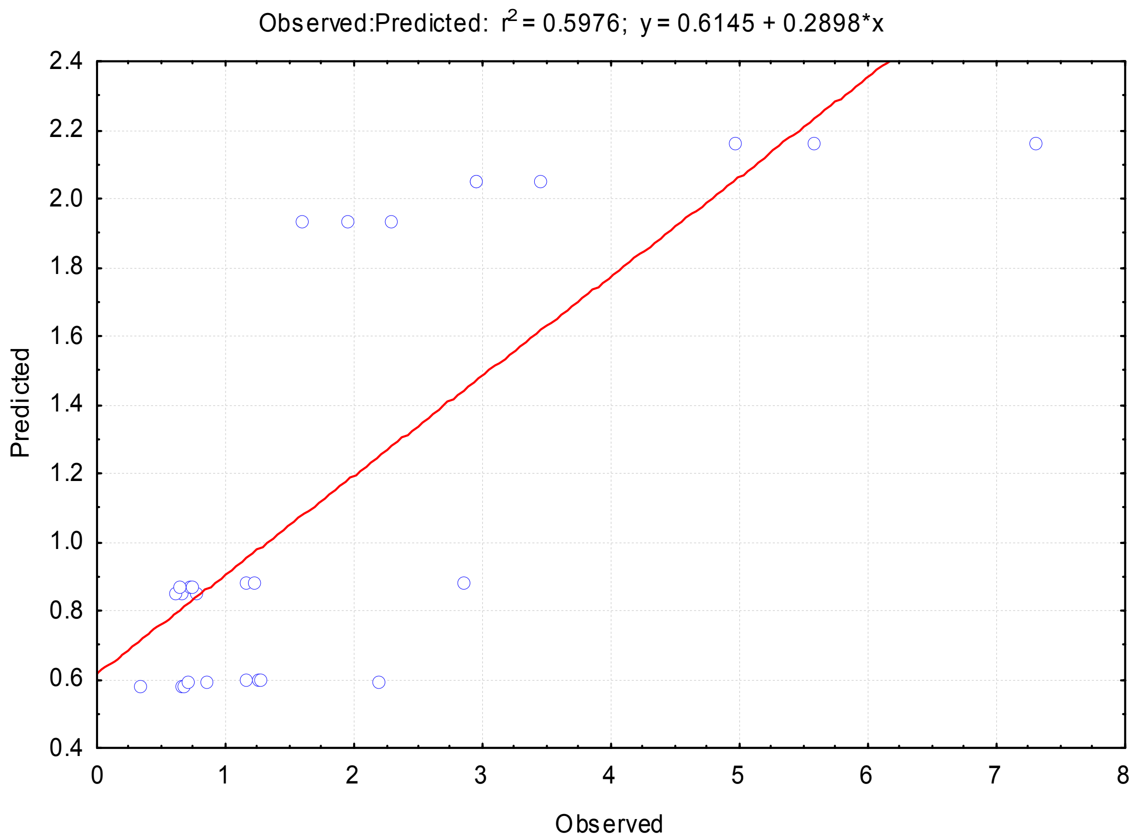



**Figure 8.4 Scatterplot of observed and predicted soil loss values (kg/plot) – dots represent different treatments (Boubakari dataset).**

## 8.9.2 Testing individual datasets :- Dataset 2: Dillaha et al. (1989)

### 8.9.2.1 Measured data

Experiments were conducted by Dillaha et al. (1989) on a silt loam soil on nine field plots near Blacksburg, Virginia, at 3, 6 and 9° slopes. Some of these plots had a VFS at the bottom consisting of an area either 4.6 or 9.1 m long and planted with orchard grass. Approximately 200 mm of rainfall was applied to the plots over one week, consisting of 6 events, at a rainfall intensity of approximately 50 mm/hr. Sediment, nutrient and water yield were recorded. Soil loss values are presented as total suspended solids (Mg/ha).

### 8.9.2.2 Comparing trends in measured and predicted values

The measured and predicted data are shown in Table 8.8 and Table 8.9. The model results do not appear to be as sensitive as the measured results to the change in VFS length but both sets of results appear to show an increase in soil loss with slope. The measured results showed an average sediment trapping efficiency for the two VFS lengths of 88, 92 and 62% for the 3, 6 and 9° slopes respectively. For the predicted results the trapping efficiencies are 95, 92 and 88%, also showing the VFS on the 3° slope to be the most efficient and that on the 9° slope less efficient.

Both datasets show higher runoff on the bare plots than on the buffered plots and both datasets show the same relative difference between the bare and buffered plots. For the predicted dataset runoff from the buffered slopes is approximately 36% of that from the bare slopes. For the measured dataset runoff from the buffered slopes is 27% of that from the bare slopes on the 3° slope. However, runoff is approximately the same for the bare and buffered plots on the 6 and 9° slopes.

| *Soil loss (t/ha)* | *Slope (°)* | | | | | |
|---|---|---|---|---|---|---|
| | 3 | | 6 | | 9 | |
| VFS length (m) | Measured | Predicted | Measured | Predicted | Measured | Predicted |
| 0 | 2.1 | 23.8 | 3.93 | 33.4 | 8.94 | 38.4 |
| 4.6 | 0.36 | 1.3 | 0.56 | 2.7 | 4.22 | 4.7 |
| 9.1 | 0.14 | 1.3 | 0.1 | 2.7 | 2.71 | 5.1 |

**Table 8.8 Measured and predicted soil loss.**

| *Runoff (mm)* | *Slope (°)* | | | | | |
|---|---|---|---|---|---|---|
| | 3 | | 6 | | 9 | |
| VFS length (m) | Measured | Predicted | Measured | Predicted | Measured | Predicted |
| 0 | 59.8 | 153.85 | 70.8 | 156.22 | 55.8 | 158.97 |
| 4.6 | 16.0 | 56.93 | 66.1 | 57.8 | 55.8 | 58.82 |
| 9.1 | 16.6 | 60.94 | 17.9 | 61.88 | 53.2 | 62.97 |

**Table 8.9 Measured and predicted total runoff.**

### 8.9.2.3 Comparing actual measured and predicted values

There is no significant difference between the ranked order of the measured and predicted soil loss values ($P$ = <0.05). The actual measurements of soil loss are, however, an order of magnitude greater for the predicted dataset than for the measured dataset. This suggests that the model is capable of comparing the relative effect of the different conditions measured by Dillaha et al. (1989) but not the absolute values. For the bare soil slopes this may be because the predicted runoff, and hence the transport capacity, is too high. However, this is not the case for the buffered slopes where the runoff values are more reasonable and, in any case, the soil loss is detachment limited. It is unlikely that the detachment values for raindrop impacts are too high because they are already low compared with those proposed by other researchers. The proportions of the sediment size fractions also seem reasonable so that it is not one size class that is controlling the predicted sediment load. The average detached sediment fractions are 23, 66 and 15% for clay, silt and sand respectively. The measured and predicted runoff values are significantly different with an $r^2$ of the relationship of 0.6 (Figure 8.5).

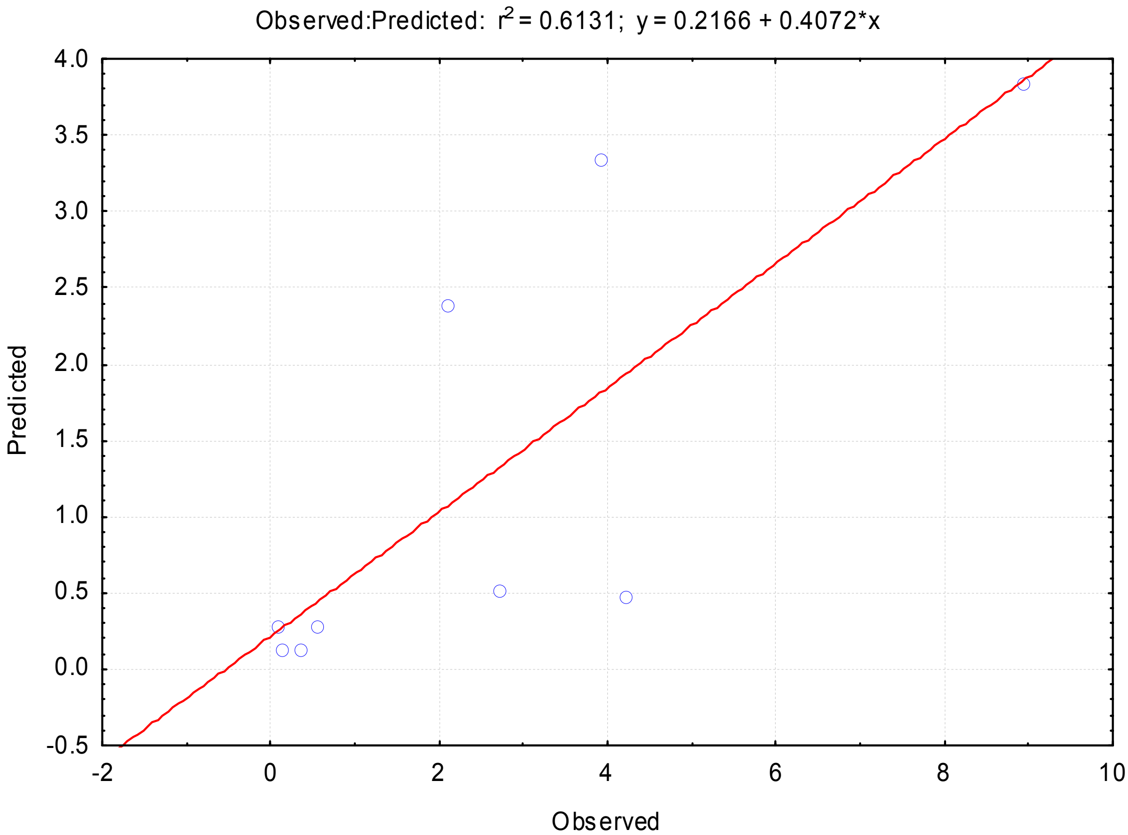



**Figure 8.5 Scatterplot and regression equation for observed and predicted soil loss (t/ha) (Dillaha dataset) – dotes represent different treatments.**

## 8.9.3 Testing individual datasets:- Dataset 3: Le Bissonnais et al. (2004)

### 8.9.3.1 Measured data

Experimental studies were conducted by Le Bissonnais et al. (2004) under natural rainfall on silt loam soils in the Pays de Caux (Normandie, France). Ray grass strips of 3 m and 6 m width were sown downslope of a 54 m long field. The slope gradient of the plots ranged from 3 to 5.8%. Annual rainfall for season 1 and season 2 was 653 mm and 727 mm respectively. Cumulative soil loss was measured over two seasons.

### 8.9.3.2 Comparing trends in measured and predicted values

Measured soil loss was reduced by 75% and by 98% for seasons 1 and 2 respectively (Table 8.10) for the 6 m VFS. Predicted soil loss was reduced by 67% and by 70% for the 6 m VFS. Le Bissonnais et al. (2004) report runoff reductions of 63% and 84% by the 6 m wide grass strip. Runoff reduction by the simulated 6 m strip is 64% in the first season and 74% in the second season. No experimental data was collected for the 3 m VFS in Season 1 so comparison is not possible.

| *Soil loss (t/ha)* | *Season 1* | | *Season 2 (cumulative soil loss)* | |
|---|---|---|---|---|
| Buffer (m) | Observed | Predicted | Observed | Predicted |
| 0 | 0.29 | 0.32 | 0.88 | 0.68 |
| 3 | No data | 0.12 | 0.17 | 0.24 |
| 6 | 0.07 | 0.12 | 0.02 | 0.21 |

**Table 8.10 Measured and predicted soil loss data (Le Bissonnais dataset).**

Le Bissonnais et al. (2004) present information on grain size entering and leaving the VFS (Table 8.11). It was observed that the clay and silt fraction of the sediment leaving the VFS doubled compared with the sediment entering the VFS. Hence the material deposited in the VFS consisted mainly of coarse silt and sand. In contrast, the model runs show the material deposited in the VFS to consist of approximately 50% clay, 14% silt and 3% sand. This suggests that the model is under predicting the sand component, possibly because it is still depositing too much material in the first stage of deposition. Another possible explanation is that the increase in clay leaving the VFS is due, in part to material being entrained by the flow through the VFS.

| *Grain size* | *0-2 µm* | *2-50 µm* | *>50 µm* |
|---|---|---|---|
| Entering VFS | 4.2 | 48.3 | 47.5 |
| Leaving VFS | 7.3 | 52.5 | 40.2 |
| *Amount of material leaving the VFS as a percentage of that entering.* | *174%* | *109%* | *85%* |

**Table 8.11 Sediment size distribution for the experiments of Le Bissonnais et al. (2004).**

### 8.9.3.3 Comparing actual measured and predicted values

There is no significant difference between the ranked order of the measured and predicted soil loss values ($P$ = <0.05). A t-test showed that there is no significant difference in the actual values ($P$ =

<0.05). The $r^2$ of the relationship between observed and predicted results is 0.95 suggesting that the model predicts the trends well but the intercept value is vastly different from 1 and the slope is not 1:1. This may be partly due to the very few data points available (Figure 8.6).

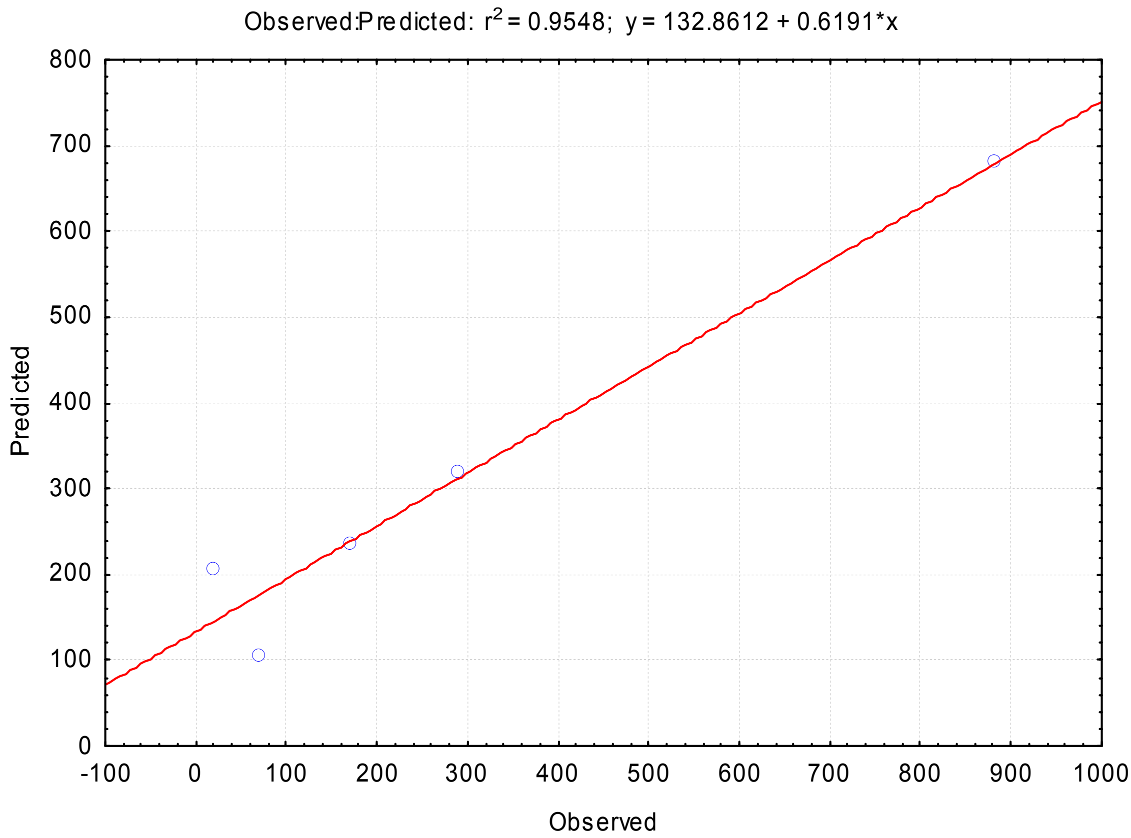



**Figure 8.6 Scatterplot and regression equation for observed and predicted soil loss (t/ha) (Le Bissonnais dataset) – dotes represent different treatments.**

## *8.10 Model efficiency*

The performance of a model can be calculated using an optimisation technique. Nash and Sutcliffe (1970) proposed that the efficiency of a model can be defined by the proportion of the initial variance accounted for by that model (Ef). The Nash-Sutcliffe criterion has been used extensively in hydrological modelling and is calculated by

$$Ef1 = 1 - \left( \frac{\sum (Q_{obs} - Q_{cal})^2}{\sum (Q_{obs} - \overline{Q_{obs}})^2} \right) \qquad \textbf{Equation 8.2}$$

where $Q_{obs}$ and $Q_{cal}$ are the observed and calculated values and $\overline{Q_{obs}}$ is the mean of the observed values. Perrin et al. (2001) suggested, however, that more than one criterion should be used to judge the simulation quality in model validation. Chiew and McMahon (1994) proposed the following, which is favoured by Perrin et al. (2001) as an "all purpose criterion", because they found the previous measure to emphasise large errors.

$$Ef2 = 1 - \left( \frac{\sum \left( \sqrt{Q_{obs} - Q_{cal}} \right)^2}{\sum \left( \sqrt{Q_{obs} - \overline{Q_{obs}}} \right)^2} \right) \qquad \textbf{Equation 8.3}$$

The two assessment criterion are based on the mean square model error (SE), defined as

$$SE = \frac{1}{n} \sum \left( Q_{obs} - Q_{cal} \right)^2 \qquad \textbf{Equation 8.4}$$

where *n* is the number of observations. Both methods were used to assess the model performance for the datasets described earlier. The measures vary between -∞ and 1 for perfect agreement and quantify the ability of a model to explain output variance (Perrin et al., 2001). An Ef value exceeding 0.5 is generally interpreted to mean that the model performs satisfactorily (Quinton, 1994) and a value exceeding 0.7 indicates that the model performs well (Nearing, 1998). An Ef value of zero means that the model predicts no better than using the mean value of the data.

Table 8.12 shows the model efficiency criterion of Nash and Sutcliffe (Ef1) and of Chiew and McMahon (Ef2), as well as the standard deviation, for each of the datasets tested. Ef2 exceeds 0.7 for the soil loss datasets of Dillaha and Le Bissonnais suggesting that the model works well in predicting the conditions tested in these datasets. The Ef2 value for the soil loss dataset of Boubakari is 0.4 which suggests that model performance is just below satisfactory. The negative values suggest that, for those datasets, the model predictions worse than those results gained from using the mean value of the data. Standard deviation (2σ) is presented to indicate the degree to which the model results will be correct 95% of the time. The lowest standard deviation calculated is for the soil loss dataset of Le Bissonnais and the highest for Dillaha. In general the results indicate that the model performs better for some datasets than it does for others.

| *Dataset* | *Variable* | *Ef1* | *Ef2* | *Std dev (2σ)* |
|---|---|---|---|---|
| Soil loss | Boubakari | 0.27 | 0.4 | 7.92 (t/ha) |
| | Dillaha et al. | -32.58 | 0.74 | 14.32 (t/ha) |
| | Le Bissonnais | 0.83 | 0.7 | 0.28 (t/ha) |
| Runoff | Boubakari | -15.42 | -10.61 | 18.72 (mm) |
| | Dillaha et al. | -7.27 | -2.82 | 81.44 (mm) |

**Table 8.12 Model efficiency criterion and standard deviation for each of the datasets.**

## *8.11 Summary and conclusions*

### 8.11.1 Model performance

Table 8.13 presents a summary of the main factors tested by the model. The highest slope angles tested in this chapter are 12, 14 and 16°. At 12° the model predictions were fairly accurate. However, although the simulated results increased with slope angle they did not increase as much as the measured results. The predicted soil loss values for the 14 and 16° slopes were approximately 50% of the measured values.

The model was able to simulate the differences in vegetation species and accurately predicted greater soil loss from the *Poa pratensis* than from the *Festuca ovina* buffered plots. Comparison with the Dillaha dataset tested the ability of the model to simulate lower slopes and different VFS lengths. Again the predicted soil loss increased with slope angle but the differences were not so pronounced as those for the measured dataset. Differences in the VFS length (3 m and 6 m) were only reflected in the simulated results for the 9° slope but not for the 3 and 6° slopes. Differences in VFS length were only picked up by the model for the second rainfall event in the dataset of Le Bissonnais.

The model accurately simulated the percent reduction of soil loss and runoff by the 6 m VFS in this dataset. Prediction of the sediment fractions were also tested within this dataset. The model appeared to under predict the sand fraction and over predict the clay fraction.

| *Variable tested* | *Conditions tested* | *Conditions for which model performed best* | *Further work/improvements to model* |
|---|---|---|---|
| Slope angle | 3,6 & 9°<br>5°<br>12,14 & 16° | 3 to 5°. Model showed low relative differences compared to the measured results. | The model includes a slope factor to increase sensitivity to slope. It is likely that the model will have an optimum range. 3 to 5° is that most likely to be used for UK agriculture. |
| VFS length | 3 & 6 m<br>4.6 & 9 m | 3 to 6 m. Differences only evident for slopes > 9°. | Differences in VFS length were only tested for single event storms and not for annual events. Differences may be more evident for annual conditions. |
| Storm | Single event & annual event | Not tested within the same dataset. | The model was more responsive to differences in VFS length for the annual model. |
| Size of plot | Laboratory & field. | Not tested within the same dataset. | Worked for both. Field testing will be more important from a management point of view and further testing should focus on this. |
| Soil type | Sandy loam & Silt loam. | Not tested within the same dataset. | Testing should be extended to other soil types e.g. clays. |
| Sediment size fractions | Sand, silt & clay. | Only 1 dataset included size fraction data. | The model overpredicted the clay fraction and underpredicted the sand fraction. Testing of further soil types is required to determine whether this is a generic problem. A range of available detachment factors were used within the initial model testing but none were specific to the measured datasets. Field measurement and determination of detachment factors by particle size may improve model predictions. This is an area of work likely to advance soil erosion prediction models. |

**Table 8.13 Summary of conditions tested, model performance and further model work.**

The model did not appear to perform so well in predicting values for runoff. However, limited data were available for this analysis. The importance of runoff is vital when the erosion is transport-capacity limited but is not important when it is detachment-limited. With VFSs, erosion will be transport-limited in the vegetative strip but what is delivered to the VFS may be either transport- or detachment-limited.

### 8.11.2 Limitations

The method of testing measured and predicted values does not allow for the fact that measurement errors may have occurred. The impact of this effect is uncertain particularly, as in this case, when few measurements are available and when little information is available on the measurement methods and accuracy of the data collected.

### 8.11.3 Conclusions

Despite limited data availability an approach to testing the MMF-VFS model was carried out. Foster (1987) stated that "The best method of validity is this: does the method serve its intended purpose?" The results of the sensitivity analysis and testing of the MMF-VFS suggest that the model is appropriate for the purpose of the current study. The model appears to behave rationally in simulating the detachment, transport and deposition of sediment on both bare and buffered slopes.

Although the dataset was small, a good correlation was found between the model output and the laboratory measured values of soil loss by Boubakari (1992). Although the simulated values for the experiments of Dillaha et al. (1989) were of the wrong magnitude, the model reflected the relative differences in soil loss. Model efficiency values exceeding 0.7 were obtained for the datasets of Dillaha et al. and Le Bissonnais et al. (2004) suggesting that the model is effective in explaining output variance.

Whilst the model works reasonably well for soil loss, the runoff predictions are poor. As mentioned earlier this could be due to measurement errors which cannot be accounted for when using data collected by others, for example, overflowing of runoff collecting vessels in experiments leading to under-recording of runoff. The fact that the model can provide satisfactory predictions of soil loss whilst under predicting runoff may be explained by the soil loss on the bare soil being detachment limited (Morgan et al., 1987). Erosion therefore depends on the kinetic energy of the rainfall and the detachability of the soil but not on runoff. In this case, detachment by raindrop impact is simulated reasonably well by the model and enough runoff is simulated to exceed the transport capacity required to carry the detached sediment.

Successful overall application of the model demonstrates that the MMF-VFS is able to simulate at least relative differences in soil loss and VFS performance for the sites and conditions simulated. In order to make further conclusions on the robustness of the model, testing should be performed over a wider range of conditions. From a management perspective the most important factor is whether the model can predict the performance of established VFSs in the field. The model is therefore validated with field data from the Parrett Catchment in Chapter 9. This chapter has addressed the second hypothesis set out in Chapter 1 and proven that a simple soil erosion prediction model can be used to predict the sediment trapping efficiency of laboratory simulated VFSs. The next stage is to address the fourth hypothesis, which states that the model can be used to predict the sediment trapping efficiency of established VFSs in the field.

# Chapter 9 Validation of model using field data

## *9.1 Introduction*

Following testing and further development of the model against published datasets (Chapter 8), this chapter presents validation of the model using data collected from the field sampling (Chapter 5) and demonstrates how to set up the model to simulate field conditions. A methodology for, and results of, the validation are presented. The results are discussed in the context of VFS design and placement guidance. Limitations to the validation method and areas of further work are suggested. This chapter tests the fourth research hypothesis, as set out in Chapter 1, that the model can be used to predict the sediment trapping efficiency of established VFSs in the field.

## *9.2 Validation*

The validation process involves comparing simulations with data that were not used in the calibration or development of the model (Grayson and Blöschl, 2001). The purpose of this testing is to determine whether the model is capable of making acceptable predictions (Refsgaard, 2001). Refsgaard provides a modified version of the scheme proposed by Klemeš (1986) who stated that a model should be tested to determine how effective it is for performing the task it was intended for. The modified scheme provides four types of test, dependent upon whether sufficient data are available for calibration and whether the catchment conditions are fixed or are dependent on the impact of a changing variable such as climate.

### 9.2.1 Types of model testing for validation

#### 9.2.1.1 Split sample test

As described in Chapter 8, this is applicable to cases where there is sufficient data for calibration and where the catchment conditions are stationary.

#### 9.2.1.2 Proxy-basin test

This approach may be taken when there is not enough data for calibration of a catchment. The example given by Refsgaard (2001) is where streamflow has to be predicted for an ungauged catchment Z. Two gauged catchments should be selected and each one validated on the other one. The two validation results should be used to evaluate the ability of the model to adequately simulate the streamflow of catchment Z.

#### 9.2.1.3 Differential split-sample test

This method may be applied when a model is to be used to simulate, for example, flows, soil moisture patterns or other variables under conditions different to those of the available data. Test variants should be defined depending on the nature of the study. For example, if the model is to be used to predict the effects of a change in climate then the model should be calibrated on a dry record of the historical records and validated on a wet segment.

#### 9.2.1.4 Proxy-basin differential split-sample test

This test may be applied to cases where there is no data available for calibration and where the model is directed to predicting non-stationary conditions. The example provided by Refsgaard (2001) is a case that requires simulation of hydrological conditions for a future period with a change in climate and for a catchment where no calibration data presently exits. This test is a combination of the latter two tests described above.

#### 9.2.1.5 Validation of the MMF-VFS

When designing validation tests, factors that should be taken into account include the scale of the measurements and the performance criteria to be used. Whilst the scales need not be identical, the field data and model results should be directly comparable (Refsgaard, 2001). Performance criteria may include capability to describe overall levels, capability to describe spatial patterns and capability to describe temporal dynamics (Refsgaard, 2001). Validation of the model within this study took place by comparing the deposition component of the model with the deposition measured in the field (Chapter 5). Although the model was calibrated on soil loss and runoff values it was beyond the scope of the project to collect soil loss and runoff data from the field. However, the process used for model testing assessed the rational behaviour of all aspects of model behaviour, which should be a good indication that different components of the model are working sufficiently. The model was validated at both the field scale and, by altering the model conditions appropriately, at the scale of a small plot which represented the flow paths at the field sites. Performance criteria included capability to describe absolute values, mean values and the ranked order of values. This was evaluated for annual deposition, deposition per sampling period and for the particle size distribution of the deposited material.

### 9.2.2 Model input parameters

Model input parameters were selected to represent the characteristics of the field sites. Where measured values were not available information was taken from other sources. The basis for each model input parameter is summarised in Table 9.1.

| *Parameter* | *Main source* | *Support information* |
|---|---|---|
| Rainfall, rain days, rainfall intensity, temperature | Met Office data from within the Parrett catchment (Yeovilton) | North Wyke weather station, Richard Huish weather station. |
| Soil information | Field measurements | Palmer 2002, NATMAP. |
| Vegetation information | Field measurements | Model guide values |
| Element size (fields and VFSs) | OS maps | |
| Slope angle | Field measurements | |

**Table 9.1 Data sources for model input parameters.**

The model was run for each field site and at each site the amount and type of material deposited in the VFS was considered. Each simulated site consisted of two elements, a field and a VFS downslope of the field. The total amount of deposition simulated by the model for each VFS was compared with the amount collected by the mats at that VFS. The particle size distribution of the deposited material was also compared. All of these characteristics were compared for the entire eighteen month sampling period as well as for each three month observation period.

## 9.3 Modelling the field set up

### 9.3.1 Approach 1: Simulating actual VFS area

Three approaches to modelling the field set up were taken. Approach 1 involved simulating the entire field and the entire VFS, hence the actual field and VFS area were represented. This assumes that all flow from the field entered the buffer and that it did so as a uniform shallow sheet flow (Figure 9.1). The approach was very effective for Sites 6 and 10 and fairly effective for Site 2 but the model under predicted the field results at the other sites by up to 92% (Table 9.2). A paired t-test showed that the means of the actual and predicted results were significantly different (95% confidence interval). For this reason two further methods of simulating deposition were tested.

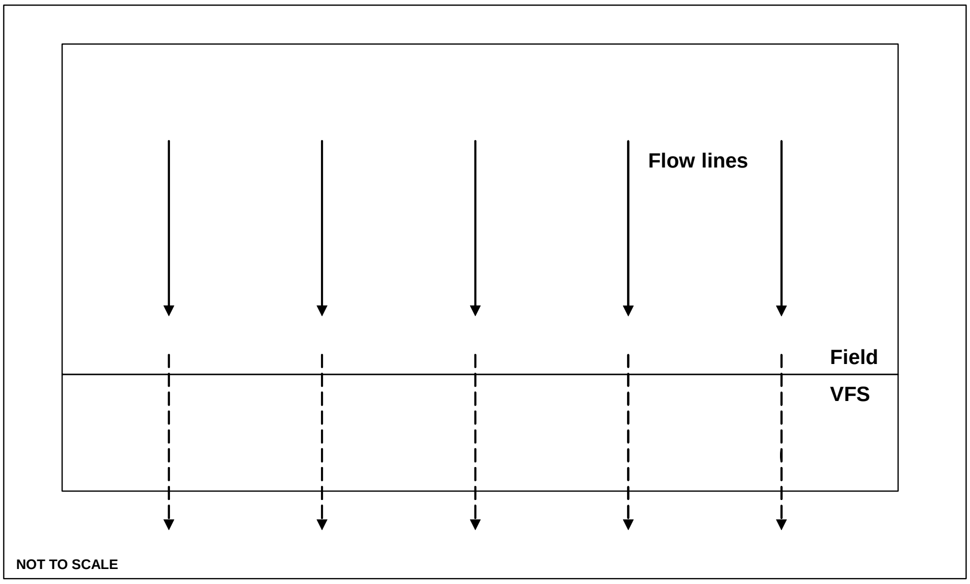



**Figure 9.1 Model representation of a buffered field site illustrating the assumption that flow towards the VFS is uniform and that it all passes through the VFS.**

| | *Deposition ($kg/m^2$)* | | |
|---|---|---|---|
| Site | Field (average of all mats) | Model | Model result as a percent of field result |
| 2 | 3.1 | 1.83 | 59 |
| 4 | 11.2 | 1.92 | 17 |
| 6 | 0.19 | 0.19 | 100 |
| 7 | 10.25 | 0.79 | 8 |
| 9 | 11.94 | 1.58 | 13 |
| 10 | 1.07 | 1.04 | 97 |

**Table 9.2 Field and model results when entire fields and VFSs are simulated, as shown in Figure 9.1.**

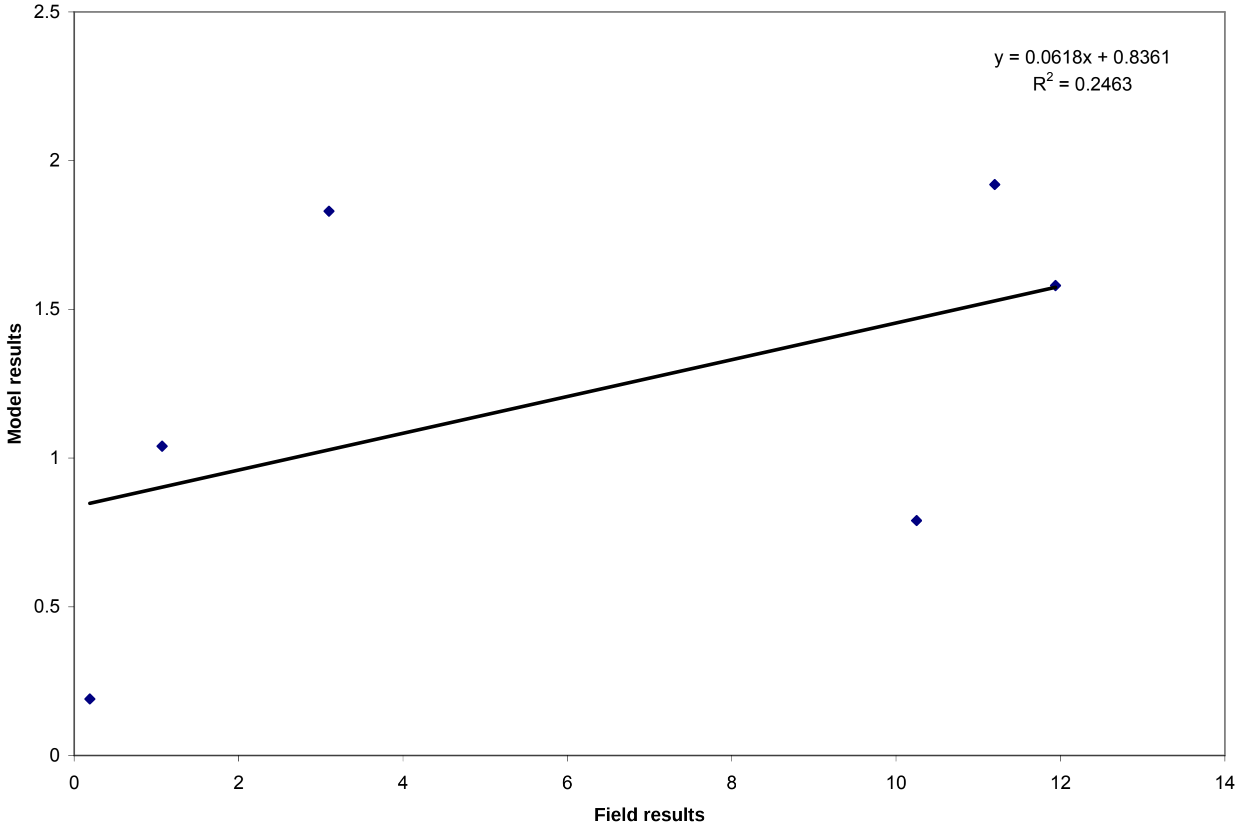



**Figure 9.2 Plot of field and model results (deposition in kg/m$^2$) when entire fields and VFSs are simulated, as shown in Figure 9.1.**

## 9.3.2 Approach 2: Simulating effective VFS area

It was noted in Chapter 5 that concentrated flow appeared to dominate the sediment deposition at a number of VFSs. It was at those sites that Approach 1 performed poorly i.e. Sites 4, 7 and 9. At Site 2 there was slight convergence of the upslope field, resulting in a section of the VFS that collected more sediment than the rest. Since the model was less effective for these sites a different approach to modelling them was taken. Instead of simulating flow into the entire width of the VFS the simulations were designed to recognise the areas of concentrated flow. It was decided that, based on field evidence, the mats at these sites were representative of the area subject to deposition and that deposition in the rest of the VFS was close to 0 kg/m$^2$. This takes into account the terrain at these sites by reflecting the convergence of flow into channels in the direction of the VFS. Using the measurements of VFS deposition at Sites 2, 4, 7 and 9 it was determined that deposition takes place selectively over 50, 13, 10 and 9 % of the VFSs respectively i.e., along the lines of concentrated flow. For example, at Site 4 the model was run for a field 250 m wide flowing into a VFS 20 m wide. "Forcing" the flow into a narrower element increased the deposition in the receiving element, i.e. the VFS, per unit area and Table 9.3 shows that this improved the model results. For this approach a paired t-test showed no significant difference between the predicted and actual results (95% confidence interval). For Sites 4 and 9, the flow lines of the contributing areas, and the sections of the VFSs simulated as the receiving elements, are represented on top of aerial photographs of the sites (Figure 9.4 and Figure 9.5). Although evidence for the effective VFS area was taken from field observations and measurements of deposition areas, in-field channels and depressions are visible on the aerial photographs which gives some indication of flow patterns.

| | *Deposition (kg/m$^2$)* | | |
|---|---|---|---|
| Site | Field (average of all mats) | Model | Model result as a percent of field result |
| 2 | 3.1 | 3.65 | 85 |
| 4 | 11.2 | 9.88 | 88 |
| 6 | 0.19 | 0.19 | 100 |
| 7 | 10.25 | 7.78 | 76 |
| 9 | 11.94 | 12.15 | 102 |
| 10 | 1.07 | 1.04 | 97 |

**Table 9.3 Field and model results when effective VFSs are simulated.**

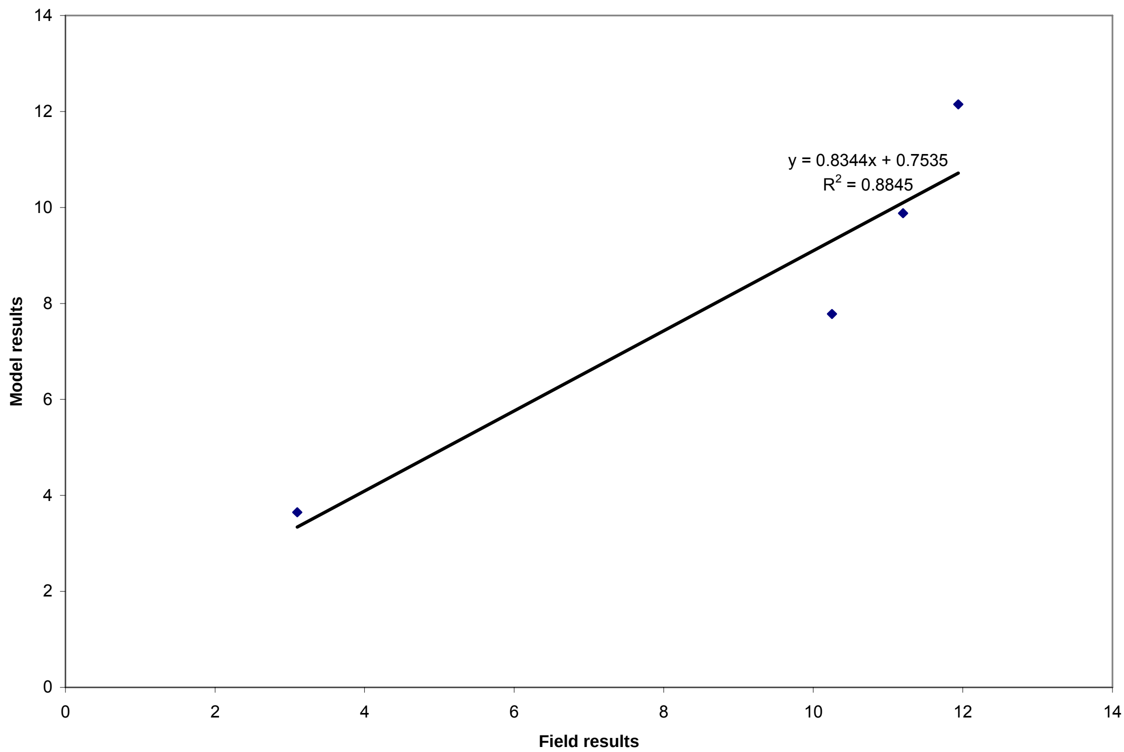



**Figure 9.3 Plot of field and model results (deposition in kg/m$^2$) when effective VFSs are simulated.**

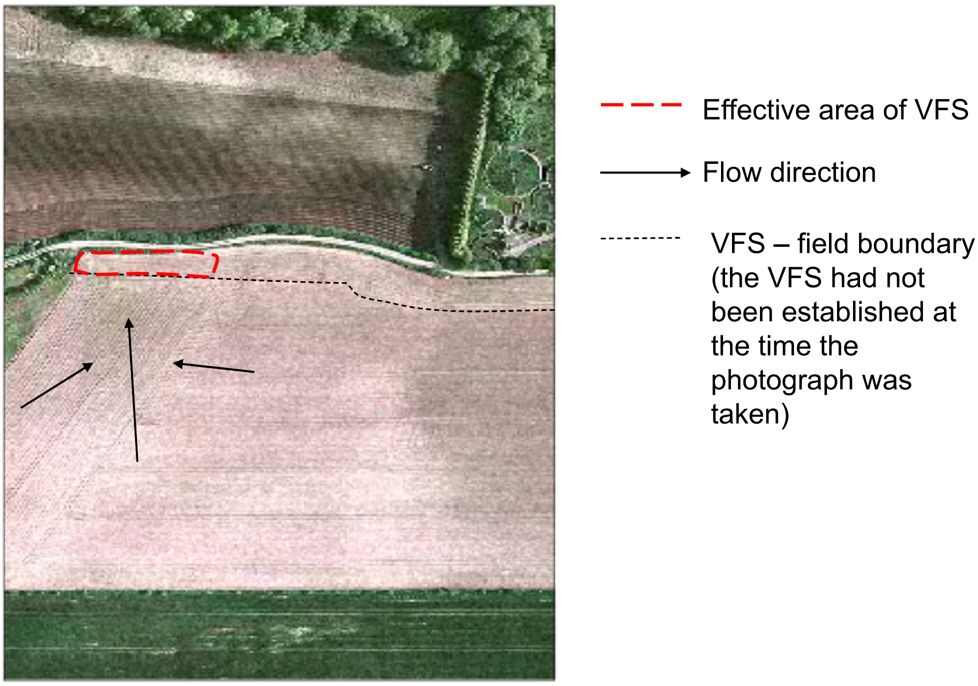



**Figure 9.4 Site 4 flow direction and simulated area of VFS based on field observation. A dark area above the marked effective VFS area represents a lower area of the field where flow collected before entering the VFS.**

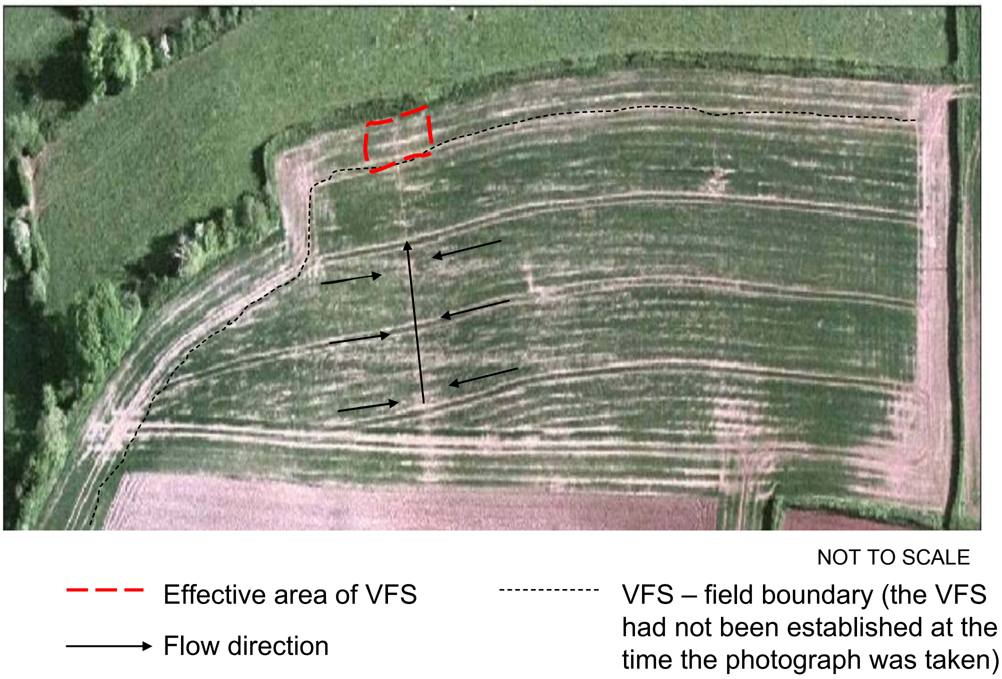



**Figure 9.5 Site 9 flow direction and simulated area of VFS based on field observation. A channel is visible running through the field to an area of the VFS where sediment deposition was observed.**

### 9.3.3 Approach 3: Simulating flow paths

In order to recognise further the differences in deposition, and contributing area, characteristics along the VFSs, a third approach was taken. For Sites 4, 7 and 9 flow paths in the field were identified from evidence of rills (recorded on the evaluation forms) and the length and width of the these used as element dimensions. For Sites 2, 6 and 10 where no obvious flow paths could be identified, the characteristics of the area immediately upslope of a section of the VFS was simulated (this is similar to treatment of the field area in Approach 2). In all cases the model results were compared only with the mats within the area simulated (Table 9.4) e.g. where only the front row of mats on a 6 m VFS collected sediment only the first 1 m was modelled. The model requires width and length measurements to be input for the modelled elements i.e. the field and VFS. Therefore in some cases the field element modelled was wider than the VFS element. The dimensions of the elements are presented in Table 9.5. For this approach a paired t-test showed no significant difference between the predicted and actual results (95% confidence interval). The following discussion describes how each of the simulations were set up for this approach.

| | *Deposition ($kg/m^2$)* | | |
|---|---|---|---|
| Site | Field (average of mats directly downslope of flow path) | Model | Model result as a percent of field result |
| 2 | 6.21 | 6.35 | 102 |
| 4 | 73.1 | 6.95 | 10 |
| 6 | 0.32 | 0.32 | 100 |
| 7 | 4.85 | 4.29 | 88 |
| 9 | 23.4 | 8.22 | 35 |
| 10 | 1.07 | 0.79 | 74 |

**Table 9.4 Model and field results when flow paths are simulated.**

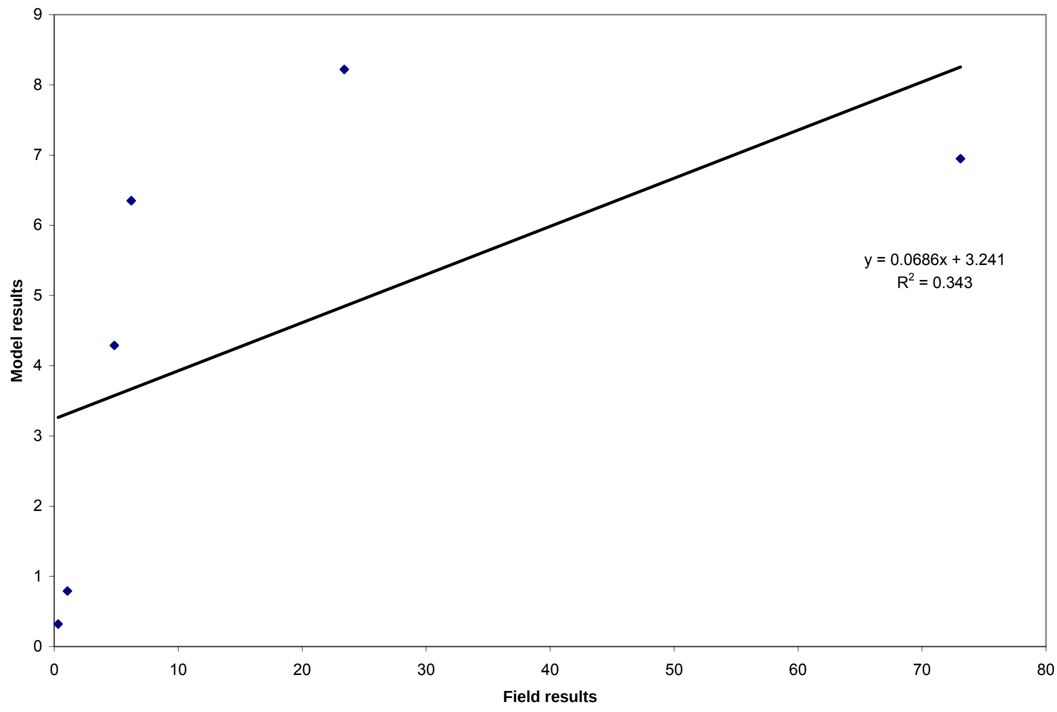



**Figure 9.6 Plot of model and field results (deposition in kg/m$^2$) when flow paths are simulated.**

| ***Site*** | ***2*** | ***4*** | ***6*** | ***7*** | ***9*** | ***10*** |
|---|---|---|---|---|---|---|
| **Length of field element (m)** | 20 | 10 | 20 | 60 | 10 | 20 |
| **Width of field element (m)** | 20 | 6 | 20 | 10 | 20 | 20 |
| **Length of VFS element (m)** | 6 | 1 | 3 | 8 | 3 | 6 |
| **Width of field element (m)** | 20 | 3 | 3.5 | 10 | 9 | 20 |
| **Field observations** | No rills observed upslope of VFS. | Rills upslope of the VFS leading edge to a distance of 10 m. | Areas of bare ground and sediment deposition upslope of the VFS. | Gully running for 60 m upslope from the VFS edge. | Deposition, rills and areas of bare ground upslope of the VFS. | No evidence of rills in the field above the VFS. |
| **VFS observations** | Mats collected sediment across a 20 m width of the VFS and from the front to the back. | Sediment only collected on the front mats. | Deposition on mats only on the upper 3 m grass strip within the VFS. | Mats collected sediment across 10 m of the VFS width. | Front mats collected sediment but this spread across the VFS over 9 m. | Mats collected sediment at the front, middle and back of the VFS. |

**Table 9.5 Element dimensions modelled (length in direction of flow downslope and width intercepting flow downslope) and field and VFS measurements on which the dimensions were based.**

#### 9.3.3.1 Site 2

*Field conditions*

The field was mostly under maize except for January 2006 when bare soil was observed. January 2006 was the only period during which sediment was collected from the mats at this site indicating the importance of maintaining a field cover. Although no concentrated flow or channels leading to the VFS were observed at this site the field dipped slightly towards the centre so it is possible that flow was increased here and then directed towards the section of the VFS where the mats were installed. Ten mats were placed at the site, of which just five collected sediment; two from the front, two from the back and one from the middle of the VFS.

*Simulation*

To account for the lower section of the field, the model was run for an area 20 m by 20 m. The characteristics of the field were not altered. No rills were observed within the VFS, therefore the flow depth was reduced to that of an un-channelled flow (0.005 m). The total deposition measured in the field was averaged only over the mats in the 20 by 20 area.

#### 9.3.3.2 Site 4

*Field conditions*

During the sampling period the field was planted with wheat followed by oil seed rape. The VFS consisted of 7.5 m of grass, within which fifteen mats were placed. Sediment was only collected from those mats located at the front of the VFS. Deposition occurred on the mats in May 2005, June 2005, July 2005 and January 2006.

*Simulation*

The mats that collected sediment at this site were clustered in an area of the VFS that received flow from the convergence of two slopes in the field above. This area was fairly bare compared to the rest of the field and was rilled. The depth of flow was based on the depth of the rills measured and the field element was given the characteristics of a bare plot to represent the sediment deposition and lack of crops. Sediment was only collected from the front mats and so only the first 1 m of the VFS was modelled. At this site the modelled deposition result did not give a good agreement with the field result, with the model result being just 10% of the field result. Deposition on the mats in the field was relatively high at this site and sediment collected to a depth of 30 mm. This may be due to the high fraction of loose, coarse material or to local variations in the terrain, e.g. a difference in height between the field and VFS.

### 9.3.3.3 Site 6

*Field conditions*

Crops during the sampling period included barley and oil seed rape. The VFS consisted of two 3 m grass strips either side of a 1 m hedge between two fields. Rills were observed leading into the VFS with an average depth of 0.9 m. Flow depth was therefore increased to 0.9 m within the model. Twelve mats were placed within the grass strips and the hedge. Mats were collected from the front of the VFS in May 2005, July 2005 and January 2006.

*Simulation*

A 20 by 20 m area upslope of the VFS was modelled as shown in Table 9.5. Only the mats downslope of this area were taken into account in comparing the field deposition with the modelled deposition amount.

### 9.3.3.4 Site 7

*Field conditions*

During the sampling period the field was planted with wheat. A gully ran down the centre of the field perpendicular to the VFS. At the base of this the VFS was covered in sediment. During the last sampling period the gully had been levelled out and the strip down the middle of the field planted with grass. Eighteen mats were placed within the VFS at this site. Sediment was collected from mats at the front, middle and back of the site during all sampling periods except for October 2005.

*Simulation*

The model simulation took account of the area in the field where two slopes converged to form a gully perpendicular to the VFS. This element was modelled based on the depth, width and length of the gully. Sediment from only the mats within this area were compared with the simulated deposition result. The model gave good agreement with the field result at this site. The set-up is however, somewhat artificial as it ignores runoff from side slopes contributing to the gully. Further work may involve developing a multiple element set-up to represent runoff coming from different directions, e.g. Figure 9.7. This would involve flow entering the VFS element from different directions.

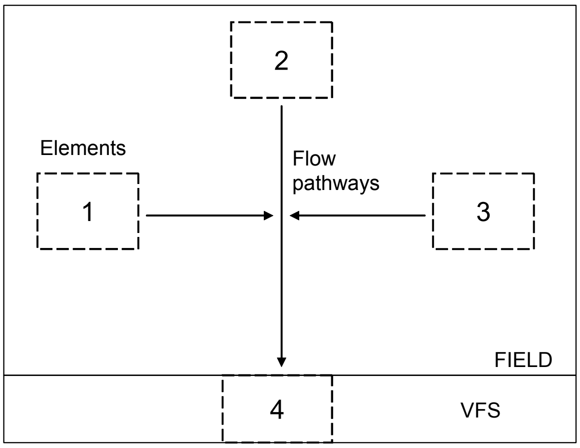



**Figure 9.7 Possible multi -element set up to take into account runff from side slopes.**

## 9.3.3.5 Site 9

*Field conditions*

During the sampling period the field was planted with maize followed by wheat. The VFS consisted of 6 m of grass. Fifteen mats were placed within the VFS, from which sediment was only collected from those located at the front of the VFS. Deposition occurred on the mats in May 2005, June 2005, July 2005 and January 2006. The high deposition observed in the field in January was due to a large area of breaching within the VFS which became an area of deposition. Flow in the contributing area became channelled, deposited sediment spread both laterally and through the VFS and the filter strip vegetation became inundated and flattened. The depth of deposited material increased as coarse sediment built up against a hedge at the downslope edge of the VFS.

*Simulation*

The simulation represented the convergence of flow in the field and the mats within the corresponding section of the VFS. The area above the VFS was modelled as bare soil to represent the lack of crops and deposition that had already taken place here. The model value for deposition was approximately 30% of the field value for deposition. The high field result is likely to be due to the depth of coarse material that built up when the VFS failed. The result for Site 9 (Table 9.4) is worse then with simulating effective VFS area, suggesting that reducing the size of the contributing area is not appropriate in this case.

## 9.3.3.6 Site 10

*Field conditions*

During the sampling period the field was planted with wheat, followed by a period of bare ground and then wheat again. The VFS consisted of 6 m of grass. Twenty mats were placed within the VFS at this site. Sediment was only collected from three mats at the front of the VFS in May 2006.

*Simulation*

In the absence of obvious flow paths a 20 by 20 m area was modelled upslope of a 20 m wide section of the VFS. This approach reduced the deposition prediction. This is likely to be due to the shorter flow length. The result for Site 10 (Table 9.4) is worse than with simulating effective VFS area, suggesting the reducing the size of the contributing area is not appropriate for these conditions.

### 9.3.4 Summary

Table 9.6 shows, for each approach, the modelled deposition values as a percentage of the field deposition measurements for each site. At Site 2, simulating the flow paths into the VFS was the most effective approach with a 2% difference between the field and model result. Simulating both the actual VFS area and the flow paths was effective for Site 6. This may be because this was the simplest site in terms of terrain. The slope was planar and did not undulate along the VFS width, hence this is the site most likely to generate close to uniform flow into the VFS. It may also be because the soil at this site was different; it is possible that the model is more effective at simulating silt loams than sandy clay loam and sandy loam soils.

Simulating the individual flow paths is the most time consuming method as it requires measurement of the flow path characteristics and the areas contributing to VFS deposition; measurements which are likely to introduce a degree of subjectivity. It is possible that more elements are necessary to accurately represent the flow paths. This approach also focuses on isolated sections of the VFS and is therefore possibly more useful for investigating specific processes and groups of variables. From a management perspective the most useful approach is likely to be a combination of the first two. In this case a field assessment would be required to determine the fraction of the VFS over which deposition takes place. This information could then be used in designing the simulations.

| | *Model deposition value as a percentage of field deposition value* | | |
|---|---|---|---|
| Site | Approach 1: Simulating actual VFS area | Approach 2: Simulating effective VFS area | Approach 3: Simulating flow paths |
| 2 | 59 | 85 | 102 |
| 4 | 17 | 88 | 10 |
| 6 | 100 | 100 | 100 |
| 7 | 8 | 76 | 88 |
| 9 | 13 | 102 | 35 |
| 10 | 97 | 97 | 74 |

**Table 9.6 Difference between field and model deposition values for all approaches.**

### 9.3.5 Simulating individual time periods

For every site each individual sampling period was simulated, using the effective area approach, and compared to the field results. The model was run over a three month time period. Characteristics were altered to reflect the number of rain days, amount of rainfall, temperature and crop characteristics. This also included altering the number of days in the interflow Equation 1. This increased interflow (*IF*) so that the runoff (*Rc*) was reduced and therefore rainfall (*R*) was more likely to exceed *Rc*. This had the effect of increasing the amount of runoff and soil loss. The field results were obtained by taking the total amount of deposition for the sampling period, averaging it over the number of mats installed and converting this to a unit area amount of deposition. The results are presented in Table 9.7.

$$IF = \frac{(R - E - Q).LP.\sin S}{365}$$ **Equation 9.1**

Where $IF$ is interflow, $LP$ is lateral permeability (25 m/day), $S$ is slope (°), $Q$ is runoff (mm), $R$ is rainfall (mm), $E$ is evaporation (mm).

There is good agreement between the model and field results for the total amount of sediment deposited over all sampling periods. However, the model generally spreads the deposition over the sampling periods more than the field results show. This may reflect the greater sensitivity of the field results to individual storm events than the model. As the model is an annual model, over three-month periods it is likely to simulate the aggregate effect of storms better than for an individual storm. Discrepancies may also be caused by the length of time between field visits. This means that the evidence of field and storm characteristics on which the model is set up is based only on one assessment every three months. This is likely to be fairly generalised and it is possible that the model predictions could be improved with more detailed information on the time in between the field assessments.

| *Site* | *2* | | *4* | | *6* | | *7* | | *9* | | *10* | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Sampling period | Model ($kg/m^2$) | Field ($kg/m^2$) | Model ($kg/m^2$) | Field ($kg/m^2$) | Model ($kg/m^2$) | Field ($kg/m^2$) | Model ($kg/m^2$) | Field ($kg/m^2$) | Model ($kg/m^2$) | Field ($kg/m^2$) | Model ($kg/m^2$) | Field ($kg/m^2$) |
| Jan to Apr 2005 | 0.03 | 0 | 1.11 | 7.77 | 0.19 | 0.05 | 1.29 | 2.9 | 0.97 | 0 | 0.31 | 0 |
| Apr to Jul 2005 | 0.09 | 0 | 1.23 | 4.06 | 0.11 | 0.06 | 1.43 | 4.34 | 1.08 | 11.56 | 0.18 | 0 |
| Jul to Oct 2005 | 0.18 | 0 | 2.78 | 0 | 0.06 | 0 | 2.4 | 0 | 5.32 | 0 | 0.07 | 0 |
| Oct to Jan 2006 | 0.08 | 3.1 | 3.6 | 0.008 | 0.04 | 0.08 | 3.17 | 2.49 | 2.04 | 0.29 | 0.04 | 0.14 |
| Jan to May 2006 | 0.05 | 0 | 4.17 | 0 | 0.05 | 0 | 0.31 | 0.58 | 1.35 | 0.14 | 0.05 | 1.06 |
| *Total* | *0.43* | *3.1* | *12.89* | *11.84* | *0.45* | *0.19* | *8.6* | *10.31* | *10.76* | *11.99* | *0.65* | *1.20* |

**Table 9.7 Model and field results for individual time periods.**

| *Site* | *2* | | *4* | | *6* | | *7* | | *9* | | *10* | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Size fraction | Model (%) | Field (%) | Model (%) | Field (%) | Model (%) | Field (%) | Model (%) | Field (%) | Model (%) | Field (%) | Model (%) | Field (%) |
| Clay | 24 | 3 | 2 | 2 | 91 | 9 | 17 | 4 | 6 | 4 | 37 | 1 |
| Silt | 76 | 7 | 91 | 8 | 9 | 37 | 83 | 16 | 58 | 8 | 55 | 8 |
| Sand | 0 | 90 | 7 | 90 | 0 | 54 | 0 | 80 | 36 | 88 | 8 | 91 |

**Table 9.8 Model and field results for size fraction of the deposited material.**

### 9.3.6 Simulating particle size

Table 9.8 shows the model and field results for the particle size distribution. A paired t-test was carried out for the silt and clay fractions of all sites in order to compare the model and field results. This is illustrated by Figure 9.8 and Figure 9.9. (The values are percentages and therefore it was only necessary to carry out the analysis for two of the three fractions.) The results of the test show that at a confidence level of 95% there is a significant difference between the clay values but not the silt values. The main reason for the difference in results is that the model under predicts the sand fraction.

The model shows that, where sand is eroded from the field and transported to the VFS, it is deposited in the VFS, as observed at the field sites. The difference in the field and model results arises from the situation whereby too much sand is deposited in the initial stage of deposition and not enough is transported to the VFS.

In general, the model results suggest that a very high amount of sand is deposited close to the point of detachment and not transported down slope. The amount of sand predicted to be detached from the soil is high for runoff and raindrop impact but the total sand going into transport in the flow is very low (<36%) since most is deposited straight away at the site of detachment. For the buffered element, detachment of sand is lower than it is for clay or silt. Transport capacity is greater than detachment for sand, but not for clay and silt, therefore the sand is detachment limited whilst the clay and silt are transport capacity limited. Within the VFS, the sediment is transport capacity limited. The model consequently predicts a lower amount of sand deposition than was observed in the field.

One explanation for the higher proportion of sand particles in the field results is the ponding effect described in Chapter 5. If the fine material is deposited in front of the VFS due to the change in velocity then the material deposited in the VFS will have a high sand fraction, as observed at the field sites. The model does not take into account the ponding effect or the likely subsequent deposition of fine material in front of the VFS. Therefore it is possible that the model reflects the relative size fractions had no ponding occurred. This would explain why the total amount of sediment deposited in the VFS is well predicted by the model but the relative size fractions of the deposited sediment are not.

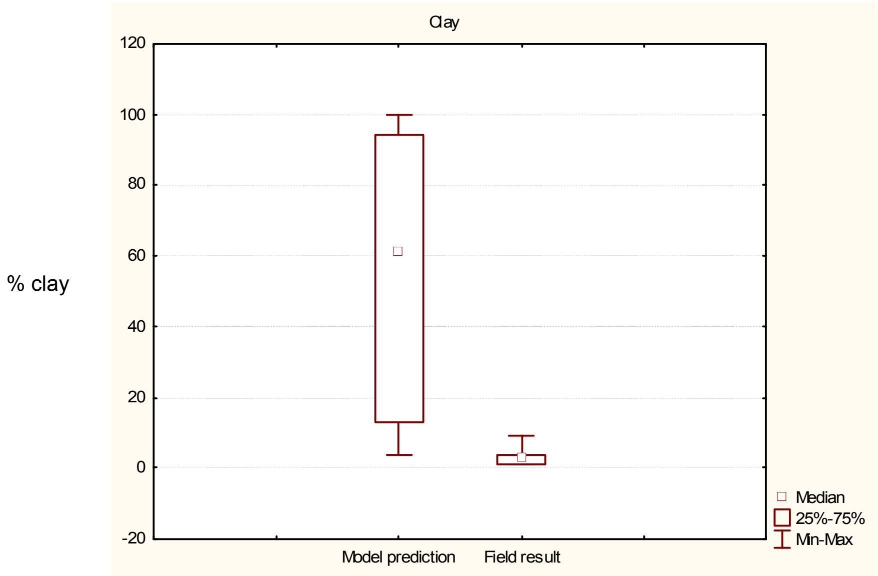



**Figure 9.8 Box and whisker plot showing model and field values for deposited clay fraction (%).**

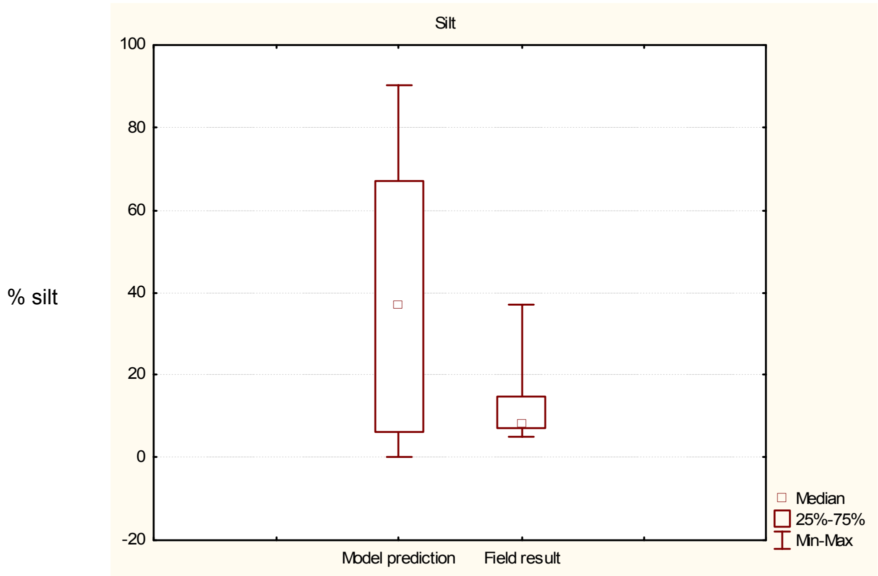



**Figure 9.9 Box and whisker plot showing model and field values for deposited silt fraction (%).**

## *9.4 Summary and conclusions*

In general there is a good agreement between the model results and the field data for deposition. This is proven by the t-test which, for Approaches 2 and 3 showed no significant difference between the measured and predicted results. For sites without obvious lines of concentrated flow there was less than 5% difference between the model results and field data. However, where flow to the VFS and consequent deposition was more irregular, model results were improved by simulating the utilised or effective VFS area.

### 9.4.1 Simulating effective VFS area

Better agreement between the model results and field data was obtained when the model was run for the entire field flowing into just a section of the VFS, thus representing concentrated flow. The VFS section was selected on the basis of field evidence of deposition in the VFS. This was taken as evidence that the VFS was effective in performing a filtering role along that section. It may be argued that these sections of the VFSs are not in fact effective in as much as, in many cases, sediment was deposited right through the VFS from front to back. Hence, in Chapter 5 these areas were referred to as VFS 'failure points'. However, the sections do represent an area where deposition has, at some time, taken place, and may continue to occur, demonstrating that the VFS section has the potential to prevent sediment from travelling further downslope. This approach recognises the suggestion by a number of researchers, detailed in Chapter 2, that the majority of flow reaching a VFS does so as concentrated flow and that flow is not uniform along the field-VFS interface.

Dosskey et al. (2002) proposed, for riparian buffers, that gross VFS area is the area that field runoff would contact if flow were dispersed across the entire VFS area and effective VFS area is the actual pathway that field runoff travels to the stream. This is a very similar concept to that proposed for the sites measured within the current study. Figure 9.10 depicts the relationship proposed by Dosskey et al. (2002) as well as the approach taken to modelling effective VFS area in the current study.

Dosskey et al. (2002) concluded that, on four farms observed, effective VFS area averaged 6, 12, 40 and 81% of the gross VFS area. The difference that this made to VFS performance at the sites was quantified by comparing the sediment retention efficiency for gross and effective VFS area. Estimated gross VFS sediment retention efficiency for the four farms was 99, 67, 59 and 41%, whilst that for the effective VFS was just 42, 15, 23 and 34%. They concluded that sediment trapping efficiency of VFS based on gross VFS area may greatly overestimate actual performance.

This concept may be translated into the development of the model for design and placement guidance in two ways. It is recommended that the model is always run with field, rather than guide, values wherever possible. Therefore, part of the field collection could include a visual or measured assessment of the proportion of the VFS that is being utilised, i.e. the effective VFS area. This may be based on areas of deposition within or in front of the VFS, on areas of rills and gullies in front of the VFS and on areas of converging flow within the field upslope of the VFS. It may then be suggested that all the predicted deposition takes place selectively over, for example, 10, 15 or 20% of the model run for this as described earlier. Dosskey et al. (2002) estimated effective VFS-area by a similar method based on visual observations of microrelief, sediment and debris deposition and orientation and erosion patterns on the ground surface. Alternatively, in the absence of a field assessment prior to running the model, the simulated results could be adjusted accordingly based on the proportion of effective VFS. The proportion of effective VFS is likely to vary depending on the field conditions but the average for the sites in this study, based on field observations, is 21%.

### 9.4.2 Simulating flow paths

The simulations based on flow paths indicate that more work is required to ensure that all of the contributing area is accounted for. This may be achieved by running simulations with more than two elements. Such an approach would require a detailed field survey and is perhaps beyond that expected in a simple evaluation approach.

### 9.4.3 Conclusions

"A model is validated for a particular application and a successful validation in one example does not imply that the model is validated for universal use." (Grayson and Blöschl, 2001). The validation performed informs on the ability of the model to predict deposition at established VFSs in the Parrett catchment. The results to the validation suggest that the model may be used effectively to predict deposition for similar conditions but exemplify the site-specific nature of the processes occurring at field-VFS interfaces. It is recommended that when running the model, parameters are always based on field data, where possible, and that this data collection includes an assessment of the proportion of the VFS that is being utilised i.e. the effective area.

The model predicts the particle size distribution of the deposited material poorly. This was not improved by any of the alternative approaches taken to modelling the field scenarios. In all cases this is because the model deposits most sand at the site of detachment and does not transport it whereas, in reality, most of it is transported. This is an area of further development for the model. Modification of the model may be possible to better represent the detachment and deposition stages of sand. To prevent as much sand being deposited would require lower fall velocities than that conventionally recognised for sand. Therefore the model may be improved following experimental work into particle-size-specific fall velocities.

The fourth research hypothesis proposed that the model could be used to predict the sediment trapping efficiency of established VFSs in the field. This chapter has explored this culminating in recommendations for using the model to represent field scenarios. On this basis guidance for using the model in VFS design and placement can now be developed. Methods for applying this are presented in Chapter 10.

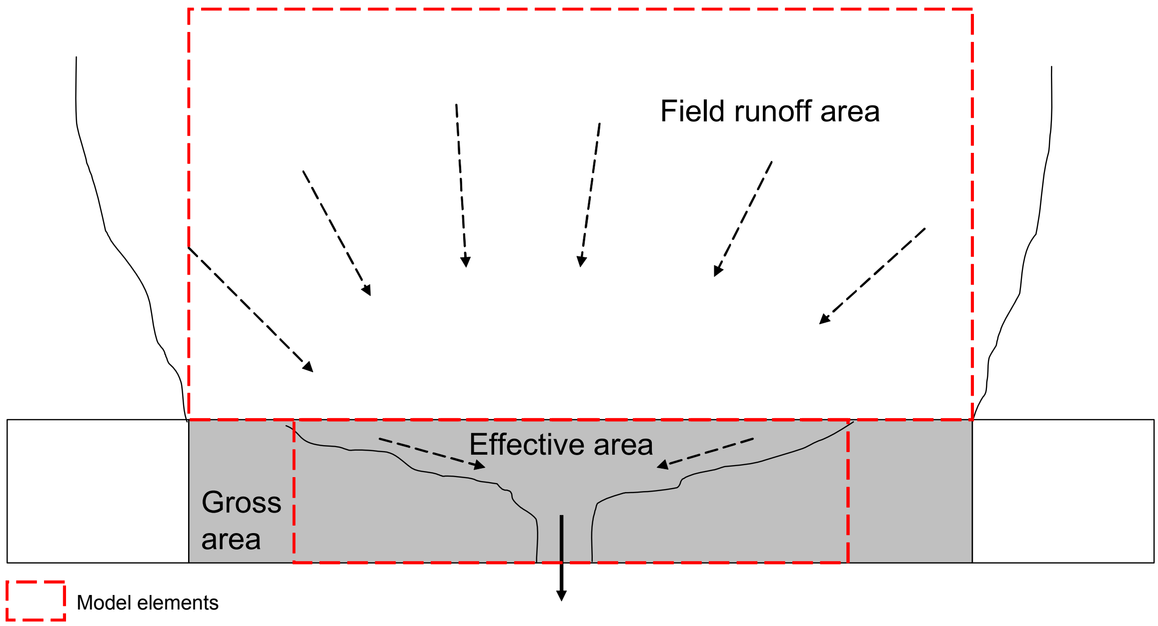



**Figure 9.10 Modified version of a diagram by Dosskey et al. (2002) showing the general relationship between field runoff area, gross VFS area and effective VFS area. The approach used in the current study to modelling effective VFS areas has been superimposed in the form of red lines representing the elements simulated. The effective area modelled was based on field evidence of the area subject to deposition.**

# Chapter 10 Potential applications of the research

Following successful testing of the MMF-VFS model (Chapters 8 and 9), this chapter explores methods for applying the research. This addresses the fifth research hypothesis which proposes that the model can be used to derive the optimum design and location for VFSs in the field. It also targets the gap identified in the literature in terms of tools to support decisions on VFS design and placement. The chapter is split into the following sections:

1. Decision support system for VFS design and placement

2. Design of VFS architecture

3. Catchment planning

## *10.1 Decision support system for VFS design and placement*

### 10.1.1 Introduction

For the purpose of the Defra project (PE0205) a system was required that farmers, land owners and policy makers can use to aid in VFS decision making. The preferred system was one that could be used in the field rapidly and without extensive data requirements. A paper-based system was, therefore, developed within the current study that provides advice on the most appropriate solutions for specific sites.

### 10.1.2 Developing the system

The system was developed by evaluating a range of VFS conditions that might be expected within a typical landscape and including conditions that were not observed in the Parrett catchment study area. The model was run for the range of conditions to generate data to provide information about the importance of ground cover, slope angle and VFS length in relation to soil type.

It was assumed that, within the field, slope angle and soil type cannot be changed, whilst VFS type, percentage cover and VFS length can be altered or managed. The system, therefore, needed to take into account the existing conditions and to support decisions on how much to alter the other variables in order to either improve the effectiveness of an existing VFS or to design an effective new VFS. An effective VFS was judged to be one which could reduce the net soil loss from a 1 ha field to less than 2 t $ha^{-1}$.

### 10.1.2.1 Model runs

Runs were performed for 10 categories of percentage cover, 5 VFS lengths, 6 soil types and 3 slope angles. All contributing fields were considered to be 100 m by 100 m (1 ha). All runs assumed a worst case scenario by simulating a bare soil with minimal roughness. Each model run consisted of a field and VFS combination. The soil loss leaving each pair of elements used to identify effective VFS designs i.e. combinations of slope, VFS length and percentage cover which maintained soil loss at < 2t $ha^{-1}$ $yr^{-1}$.

Evaluation of the model (Chapter 7) concluded that the trapping efficiency of a VFS is sensitive to stem diameter (D) and the number of plants (NV). A "standard" VFS was modelled using, for D and NV, the average of the vegetation parameters measured at the field sites. Further permutations of vegetative parameters were also modelled based on the variability in the field data set.

### 10.1.2.2 Percentage cover

Table 10.1 shows the vegetation measurements taken in the field component. Values in bold are the quartile ranges when the data are ranked according to D whilst those underlined represent the quartile ranges when the data are ranked according to NV. These values were determined in order to provide a quantitative description of the variability in parameters. It is apparent from Table 10.1 that at the upper quartile for D, a ground cover fraction of 80% may be achieved by very few large stems (e.g. Site 14), whilst at the lower quartile for D, this may be achieved by a large number of small stems (e.g. Site 21). This provides two 'classes' in terms of D and NV for achieving an 80% ground cover fraction across the field sites; Class One, based on a high NV and low D, i.e. 6400 x 1 mm stems per $m^2$ and Class Two, based on a low NV and high D, i.e. 148 x 14 mm stems per $m^2$. In order to derive parameter values for 'classes' in between these two extremes, the two D values (1 mm and 14 mm) were kept constant whilst NV was varied proportionally to derive values for GC and CC. The resulting combinations (Table 10.2) were used in the model runs.

| *Site* | *Average stem diameter (mm)* | *Number of stems ($1m^2$)* | *Ground cover (%)* | *Canopy cover (%)* |
|---|---|---|---|---|
| 2 | 0.9 | 15312 | 90 | 82 |
| 5 | 1 | 10128 | 80 | 77 |
| **6** | **1** | **6400** | **82** | **82** |
| 7 | 1 | 7776 | 80 | 86 |
| 1 | 1.2 | 5728 | 90 | 90 |
| 10 | 1.2 | 1040 | 85 | 80 |
| 11 | 2.1 | 73 | 74 | 70 |
| 9 | 2.5 | 1136 | 85 | 82 |
| **4** | **14** | **148** | **82** | **78** |
| 8 | 24 | 1120 | 74 | 78 |
| 3 | 26 | 712 | 63 | 74 |

**Table 10.1 Average vegetation parameters for 11 field sites, ranked by stem diameter.**

| ***Constants based on measured field values*** | ***Derived from combinations measured in the field by varying parameters proportionally*** | | |
|---|---|---|---|
| **D (m)** | **NV (elements per m$^2$)** | **GC (%)** | **CC (%)** |
| 14 | 18.4 | 10 | 10 |
| 14 | 36.8 | 20 | 20 |
| 14 | 55.1 | 30 | 30 |
| 14 | 73.5 | 40 | 40 |
| 14 | 91.9 | 50 | 50 |
| 14 | 110.3 | 60 | 60 |
| 14 | 128.7 | 70 | 70 |
| **14** | **147** | **80** | **80** |
| 14 | 165.4 | 90 | 90 |
| 14 | 183.8 | 100 | 100 |
| 1 | 800 | 10 | 10 |
| 1 | 1600 | 20 | 20 |
| 1 | 2400 | 30 | 30 |
| 1 | 3200 | 40 | 40 |
| 1 | 4000 | 50 | 50 |
| 1 | 4800 | 60 | 60 |
| 1 | 5600 | 70 | 70 |
| **1** | **6400** | **80** | **80** |
| 1 | 7200 | 90 | 90 |
| 1 | 8000 | 100 | 100 |

**Table 10.2 Model input parameters representing ranges in canopy and ground cover.**

#### 10.1.2.3 Slope, soil type and VFS length

Slopes of 1º, 5º and 10º were used within the model runs to represent a range of slopes. Six soil types (clay, silty clay, silty clay loam, sandy clay loam, silty clay and sandy loam) were used, characterizing those used by Defra (2005b). This was to facilitate cross referencing and integration of the system with other Defra field systems. Broad classes are appropriate because rapid field assessment is unlikely to enable soil texture to be determined (or simulated) to any greater precision. Based on the VFSs recommended by the current Environmental Stewardship Schemes, and on those measured in the field, the VFS lengths tested were 2 m, 4 m, 6 m, 10 m and 24 m.

#### 10.1.2.4 Model output

The key model output was a soil loss value representing the net soil loss from each field and VFS combination. This output enabled pairs of tables to be populated for 2 m, 4 m, 6 m, 10 m and 24 m VFSs (Table 10.3 to Table 10.7). Each pair includes one table representing soil loss through a VFS with 1 mm grass stems and one table representing soil loss through a VFS with 14 mm grass stems. Together they indicate the upper and lower soil loss values within which a VFS of a certain percentage cover is likely to function. The highlighted cells are those with values less than 2 t $ha^{-1}$ $yr^{-1}$ i.e. where effective control can be achieved.

| FIELD CLASSIFICATION | | | | PERCENTAGE COVER (%) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | | | |
| **Class** | *Soil* | **Slope** | *Degrees* | *<10%* | *10-20%* | *20-30%* | *30-40%* | *40-50%* | *50-60%* | *60-70%* | *70-80%* | *80-90%* | *90-100%* |
| **HEAVY** | *C* | **Gentle** | *0 to 2* | **10.75** | **9.53** | **8.61** | **7.65** | **6.7** | **6.4** | **5.85** | **5.27** | **4.81** | **4.36** |
| | | **Moderate** | *3 to 6* | **11.03** | **9.79** | **8.86** | **7.89** | **7.24** | **6..64** | **6.08** | **5.5** | **5.03** | **4.58** |
| | | **Steep** | *7 to 12* | **11.19** | **9.94** | **9.01** | **8.04** | **7.38** | **6.77** | **6.21** | **5.63** | **5.16** | **4.71** |
| **HEAVY** | *ZC* | **Gentle** | *0 to 2* | **7.7** | **6.8** | **6.12** | **5.41** | **4.93** | **4.49** | **4.08** | **3.65** | **3.31** | **2.98** |
| | | **Moderate** | *3 to 6* | **7.99** | **7.07** | **6.39** | **5.67** | **5.18** | **4.73** | **4.32** | **3.89** | **3.54** | **3.21** |
| | | **Steep** | *7 to 12* | **8.14** | **7.21** | **6.52** | **5.8** | **5.31** | **4.86** | **4.44** | **4.01** | **3.66** | **3.33** |
| **MEDIUM** | *ZCL* | **Gentle** | *0 to 2* | **5.78** | **5.1** | **4.59** | **4.06** | **3.7** | **3.37** | **3.06** | **2.74** | **2.48** | **2.23** |
| | | **Moderate** | *3 to 6* | **5.99** | **5.3** | **4.79** | **4.25** | **3.88** | **3.55** | **3.24** | **2.91** | **2.65** | **2.4** |
| | | **Steep** | *7 to 12* | **6.1** | **5.4** | **4.88** | **4.34** | **3.97** | **3.64** | **3.33** | **3** | **2.74** | **2.49** |
| **MEDIUM** | *SCL* | **Gentle** | *0 to 2* | **5.13** | **4.57** | **4.15** | **3.72** | **3.43** | **3.16** | **2.91** | **2.65** | **2.45** | **2.25** |
| | | **Moderate** | *3 to 6* | **5.16** | **4.6** | **4.18** | **3.75** | **3.46** | **3.19** | **2.94** | **2.68** | **2.47** | **2.27** |
| | | **Steep** | *7 to 12* | **5.21** | **4.64** | **4.22** | **3.79** | **3.49** | **3.23** | **2.98** | **2.72** | **2.51** | **2.31** |
| **LIGHT** | *ZL* | **Gentle** | *0 to 2* | **2.41** | **2.12** | **1.91** | **1.69** | **1.54** | **1.4** | **1.27** | **1.14** | **1.03** | **0.93** |
| | | **Moderate** | *3 to 6* | **2.5** | **2.21** | **1.99** | **1.77** | **1.62** | **1.48** | **1.35** | **1.21** | **1.11** | **1** |
| | | **Steep** | *7 to 12* | **2.54** | **2.25** | **2.04** | **1.81** | **1.66** | **1.52** | **1.39** | **1.25** | **1.14** | **1.04** |
| **LIGHT** | *SL* | **Gentle** | *0 to 2* | **1.84** | **1.64** | **1.49** | **1.33** | **1.23** | **1.13** | **1.04** | **0.95** | **0.88** | **0.81** |
| | | **Moderate** | *3 to 6* | **1.87** | **1.68** | **1.51** | **1.36** | **1.25** | **1.16** | **1.07** | **0.97** | **0.9** | **0.83** |
| | | **Steep** | *7 to 12* | **2.06** | **1.85** | **1.7** | **1.52** | **1.42** | **1.32** | **0.23** | **1.12** | **1.04** | **0.97** |

| FIELD CLASSIFICATION | | | | PERCENTAGE COVER (%) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | | | |
| **Class** | *Soil* | **Slope** | *Degrees* | *<10%* | *10-20%* | *20-30%* | *30-40%* | *40-50%* | *50-60%* | *60-70%* | *70-80%* | *80-90%* | *90-100%* |
| **HEAVY** | *C* | **Gentle** | *0 to 2* | **11.76** | **10.55** | **9.61** | **8.58** | **7.88** | **7.23** | **6.62** | **5.97** | **5.45** | **4.94** |
| | | **Moderate** | *3 to 6* | **12.06** | **10.83** | **9.89** | **8.85** | **5.45** | **7.5** | **6.88** | **6.23** | **5.7** | **5.19** |
| | | **Steep** | *7 to 12* | **12.23** | **11** | **10.05** | **9.01** | **8.3** | **7.65** | **7.03** | **6.38** | **5.85** | **5.34** |
| **HEAVY** | *ZC* | **Gentle** | *0 to 2* | **8.43** | **7.53** | **6.84** | **6.08** | **5.56** | **5.08** | **4.62** | **4.14** | **3.75** | **3.38** |
| | | **Moderate** | *3 to 6* | **8.54** | **7.83** | **7.13** | **6.36** | **5.83** | **5.35** | **4.89** | **4.41** | **4.01** | **3.64** |
| | | **Steep** | *7 to 12* | **8.89** | **7.98** | **7.28** | **6.5** | **5.98** | **5.49** | **5.03** | **4.55** | **4.15** | **3.77** |
| **MEDIUM** | *ZCL* | **Gentle** | *0 to 2* | **6.32** | **5.65** | **5.13** | **4.56** | **4.17** | **3.81** | **3.47** | **3.11** | **2.81** | **2.53** |
| | | **Moderate** | *3 to 6* | **6.55** | **5.87** | **5.34** | **4.76** | **4.37** | **4.01** | **3.67** | **3.3** | **3.01** | **2.72** |
| | | **Steep** | *7 to 12* | **6.66** | **5.98** | **5.45** | **4.87** | **4.48** | **4.11** | **3.77** | **3.4** | **3.11** | **2.82** |
| **MEDIUM** | *SCL* | **Gentle** | *0 to 2* | **5.6** | **5.05** | **4.63** | **4.17** | **3.86** | **3.57** | **3.29** | **3.01** | **2.77** | **2.55** |
| | | **Moderate** | *3 to 6* | **5.64** | **5.09** | **4.66** | **4.2** | **3.89** | **3.6** | **3.33** | **3.04** | **2.8** | **2.58** |
| | | **Steep** | *7 to 12* | **5.68** | **5.13** | **4.7** | **4.24** | **3.93** | **3.64** | **3.36** | **3.08** | **2.84** | **2.62** |
| **LIGHT** | *ZL* | **Gentle** | *0 to 2* | **2.63** | **2.35** | **2.14** | **1.9** | **1.74** | **1.59** | **1.44** | **1.29** | **1.17** | **1.05** |
| | | **Moderate** | *3 to 6* | **2.73** | **2.45** | **2.23** | **1.99** | **1.82** | **1.67** | **1.53** | **1.38** | **1.25** | **1.14** |
| | | **Steep** | *7 to 12* | **2.78** | **2.49** | **2.27** | **2.03** | **1.87** | **1.71** | **1.57** | **1.42** | **1.3** | **1.18** |
| **LIGHT** | *SL* | **Gentle** | *0 to 2* | **2** | **1.81** | **1.66** | **1.49** | **1.38** | **1.28** | **1.18** | **1.08** | **0.99** | **0.91** |
| | | **Moderate** | *3 to 6* | **2.04** | **1.84** | **1.69** | **1.52** | **1.41** | **1.3** | **1.21** | **1.1** | **1.02** | **0.94** |
| | | **Steep** | *7 to 12* | **2.24** | **2.03** | **1.88** | **1.69** | **1.58** | **1.47** | **1.37** | **1.25** | **1.16** | **1.08** |

**Table 10.3 Total soil (t ha$^{-1}$ yr$^{-1}$) transported through a 2 m VFS from a 1 ha field. The top table indicates the soil losses through a VFS with 1 mm stems and the lower table with 14 mm stems. The highlighted areas show values less then a 2 t ha$^{-1}$ yr$^{-1}$ threshold.**

| FIELD CLASSIFICATION | | | | PERCENTAGE COVER (%) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | | | |
| **Class** | *Soil* | **Slope** | *Degrees* | *<10%* | *10-20%* | *20-30%* | *30-40%* | *40-50%* | *50-60%* | *60-70%* | *70-80%* | *80-90%* | *90-100%* |
| **HEAVY** | *C* | **Gentle** | *0 to 2* | **9.34** | **8.09** | **7.2** | **6.33** | **5.74** | **5.22** | **4.74** | **4.26** | **3.88** | **3.51** |
| | | **Moderate** | *3 to 6* | **9.59** | **8.23** | **7.42** | **6.54** | **5.94** | **5.41** | **4.93** | **4.45** | **4.06** | **3.69** |
| | | **Steep** | *7 to 12* | **9.73** | **8.45** | **7.54** | **6.66** | **6.06** | **5.53** | **5.04** | **4.56** | **4.16** | **3.79** |
| **HEAVY** | *ZC* | **Gentle** | *0 to 2* | **6.68** | **5.77** | **5.12** | **4.48** | **4.04** | **3.66** | **3.31** | **2.95** | **2.67** | **2.4** |
| | | **Moderate** | *3 to 6* | **6.94** | **6** | **5.34** | **4.69** | **4.25** | **3.86** | **3.5** | **3.14** | **2.86** | **2.58** |
| | | **Steep** | *7 to 12* | **7.07** | **6.13** | **5.46** | **4.8** | **4.36** | **3.96** | **3.61** | **3.24** | **2.95** | **2.68** |
| **MEDIUM** | *ZCL* | **Gentle** | *0 to 2* | **5.01** | **4.33** | **3.84** | **3.36** | **3.03** | **2.74** | **2.48** | **2.21** | **2** | **1.8** |
| | | **Moderate** | *3 to 6* | **5.2** | **4.5** | **4** | **3.52** | **3.19** | **2.89** | **2.63** | **2.36** | **2.14** | **1.94** |
| | | **Steep** | *7 to 12* | **5.29** | **4.59** | **4.09** | **3.6** | **3.26** | **2.97** | **2.7** | **2.43** | **2.21** | **2.01** |
| **MEDIUM** | *SCL* | **Gentle** | *0 to 2* | **4.46** | **3.89** | **3.48** | **3.09** | **2.82** | **2.58** | **2.36** | **2.15** | **1.97** | **1.81** |
| | | **Moderate** | *3 to 6* | **4.49** | **3.91** | **3.51** | **3.11** | **2.84** | **2.6** | **2.39** | **2.17** | **1.99** | **1.83** |
| | | **Steep** | *7 to 12* | **4.53** | **3.95** | **3.54** | **3.14** | **2.87** | **2.63** | **2.42** | **2.2** | **2.02** | **1.86** |
| **LIGHT** | *ZL* | **Gentle** | *0 to 2* | **2.09** | **1.8** | **1.6** | **1.4** | **1.26** | **1.14** | **1.03** | **0.92** | **0.83** | **0.75** |
| | | **Moderate** | *3 to 6* | **2.17** | **1.87** | **1.67** | **1.46** | **1.33** | **1.21** | **1.09** | **0.98** | **0.89** | **0.81** |
| | | **Steep** | *7 to 12* | **2.21** | **1.91** | **1.7** | **1.5** | **1.36** | **1.24** | **1.13** | **1.01** | **0.92** | **0.84** |
| **LIGHT** | *SL* | **Gentle** | *0 to 2* | **1.6** | **1.39** | **1.25** | **1.11** | **1.01** | **0.92** | **0.85** | **0.8** | **0.71** | **0.65** |
| | | **Moderate** | *3 to 6* | **1.63** | **1.42** | **1.27** | **1.13** | **1.04** | **0.95** | **0.87** | **0.79** | **0.73** | **0.67** |
| | | **Steep** | *7 to 12* | **1.85** | **1.63** | **1.48** | **1.32** | **1.22** | **1.13** | **1.05** | **0.95** | **0.89** | **0.83** |

| FIELD CLASSIFICATION | | | | PERCENTAGE COVER (%) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | | | |
| **Class** | *Soil* | **Slope** | *Degrees* | *<10%* | *10-20%* | *20-30%* | *30-40%* | *40-50%* | *50-60%* | *60-70%* | *70-80%* | *80-90%* | *90-100%* |
| **HEAVY** | *C* | **Gentle** | *0 to 2* | **10.52** | **9.29** | **8.83** | **7.43** | **6.79** | **6.21** | **5.67** | **5.11** | **4.66** | **4.23** |
| | | **Moderate** | *3 to 6* | **10.79** | **9.69** | **8.77** | **7.67** | **7.16** | **6.57** | **6.02** | **5.45** | **4.87** | **4.44** |
| | | **Steep** | *7 to 12* | **10.95** | **9.55** | **8.62** | **7.81** | **7.02** | **6.43** | **5.89** | **5.33** | **5** | **4.56** |
| **HEAVY** | *ZC* | **Gentle** | *0 to 2* | **7.54** | **6.63** | **5.96** | **5.26** | **4.79** | **4.35** | **3.96** | **3.54** | **3.21** | **2.89** |
| | | **Moderate** | *3 to 6* | **7.82** | **7.03** | **6.35** | **5.51** | **5.15** | **4.71** | **4.31** | **3.77** | **3.43** | **3.11** |
| | | **Steep** | *7 to 12* | **7.96** | **6.89** | **6.21** | **5.63** | **5.03** | **4.59** | **4.19** | **3.84** | **3.55** | **3.22** |
| **MEDIUM** | *ZCL* | **Gentle** | *0 to 2* | **5.6** | **4.97** | **4.47** | **3.94** | **3.59** | **3.27** | **2.97** | **2.65** | **2.4** | **2.17** |
| | | **Moderate** | *3 to 6* | **5.65** | **5.27** | **4.75** | **4.13** | **3.86** | **3.53** | **3.22** | **2.82** | **2.57** | **2.33** |
| | | **Steep** | *7 to 12* | **5.86** | **5.17** | **4.66** | **4.22** | **3.77** | **3.44** | **3.14** | **2.91** | **2.65** | **2.41** |
| **MEDIUM** | *SCL* | **Gentle** | *0 to 2* | **5.02** | **4.45** | **4.04** | **3.62** | **3.33** | **3.06** | **2.82** | **2.57** | **2.37** | **2.17** |
| | | **Moderate** | *3 to 6* | **5.05** | **4.52** | **4.11** | **3.64** | **3.39** | **3.13** | **2.88** | **2.6** | **2.39** | **2.2** |
| | | **Steep** | *7 to 12* | **5.09** | **4.49** | **4.07** | **3.68** | **3.35** | **3.09** | **2.85** | **2.63** | **2.43** | **2.23** |
| **LIGHT** | *ZL* | **Gentle** | *0 to 2* | **2.35** | **2.07** | **1.86** | **1.64** | **1.5** | **1.36** | **1.24** | **1.11** | **1** | **0.9** |
| | | **Moderate** | *3 to 6* | **2.44** | **2.2** | **1.98** | **1.72** | **1.61** | **1.47** | **1.34** | **1.18** | **1.07** | **0.97** |
| | | **Steep** | *7 to 12* | **2.49** | **2.15** | **1.94** | **1.76** | **1.57** | **1.43** | **1.31** | **1.21** | **1.11** | **1.01** |
| **LIGHT** | *SL* | **Gentle** | *0 to 2* | **1.8** | **1.59** | **1.45** | **1.29** | **1.19** | **1.1** | **1.01** | **0.92** | **0.85** | **0.78** |
| | | **Moderate** | *3 to 6* | **1.83** | **1.85** | **1.69** | **1.32** | **1.41** | **1.31** | **1.22** | **0.95** | **0.87** | **0.8** |
| | | **Steep** | *7 to 12* | **2.06** | **1.63** | **1.48** | **1.51** | **1.22** | **1.12** | **1.04** | **1.11** | **1.04** | **0.97** |

**Table 10.4 Total soil (t $ha^{-1}$ $yr^{-1}$) transported through a <u>4 m VFS</u> from a 1 ha field. The top table indicates the soil losses through a VFS with 1 mm stems and the lower table with 14 mm stems. The highlighted areas show values less then a 2 t $ha^{-1}$ $yr^{-1}$ threshold.**

| FIELD CLASSIFICATION | | | | PERCENTAGE COVER (%) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | | | |
| **Class** | *Soil* | **Slope** | *Degrees* | *<10%* | *10-20%* | *20-30%* | *30-40%* | *40-50%* | *50-60%* | *60-70%* | *70-80%* | *80-90%* | *90-100%* |
| **HEAVY** | *C* | **Gentle** | *0 to 2* | **8.4** | **7.14** | **6.27** | **5.46** | **4.91** | **4.43** | **4** | **3.58** | **3.25** | **2.93** |
| | | **Moderate** | *3 to 6* | **8.62** | **7.34** | **6.46** | **5.64** | **5.08** | **4.6** | **4.17** | **3.74** | **3.4** | **3.08** |
| | | **Steep** | *7 to 12* | **8.75** | **7.46** | **6.57** | **5.75** | **5.18** | **4.7** | **4.26** | **3.83** | **3.48** | **3.17** |
| **HEAVY** | *ZC* | **Gentle** | *0 to 2* | **6.01** | **5.19** | **4.45** | **3.86** | **3.46** | **3.11** | **2.79** | **2.48** | **2.23** | **2.01** |
| | | **Moderate** | *3 to 6* | **6.24** | **5.3** | **4.65** | **4.05** | **3.63** | **3.28** | **2.96** | **3.64** | **2.39** | **2.16** |
| | | **Steep** | *7 to 12* | **6.36** | **5.41** | **4.57** | **4.14** | **3.73** | **3.37** | **3.04** | **2.73** | **2.47** | **2.24** |
| **MEDIUM** | *ZCL* | **Gentle** | *0 to 2* | **4.51** | **3.81** | **3.34** | **2.89** | **2.59** | **2.33** | **2.09** | **1.86** | **1.68** | **1.5** |
| | | **Moderate** | *3 to 6* | **4.68** | **3.97** | **3.48** | **3.03** | **2.72** | **2.46** | **2.22** | **1.98** | **1.79** | **1.62** |
| | | **Steep** | *7 to 12* | **4.76** | **4.05** | **3.56** | **3.1** | **2.79** | **2.52** | **2.28** | **2.04** | **1.85** | **1.67** |
| **MEDIUM** | *SCL* | **Gentle** | *0 to 2* | **4.02** | **3.43** | **3.03** | **2.66** | **2.41** | **2.19** | **2** | **1.8** | **1.65** | **1.51** |
| | | **Moderate** | *3 to 6* | **4.04** | **3.46** | **3.06** | **2.69** | **2.43** | **2.21** | **2.02** | **1.82** | **1.67** | **1.53** |
| | | **Steep** | *7 to 12* | **4.08** | **3.49** | **3.09** | **2.71** | **2.46** | **2.24** | **2.04** | **1.85** | **1.69** | **1.55** |
| **LIGHT** | *ZL* | **Gentle** | *0 to 2* | **1.88** | **1.59** | **1.39** | **1.21** | **1.08** | **0.97** | **0.87** | **0.78** | **0.7** | **0.63** |
| | | **Moderate** | *3 to 6* | **1.95** | **1.65** | **1.45** | **1.26** | **1.14** | **1.02** | **0.92** | **0.83** | **0.75** | **0.67** |
| | | **Steep** | *7 to 12* | **1.99** | **1.69** | **1.48** | **1.29** | **1.16** | **1.05** | **0.95** | **0.85** | **0.77** | **0.7** |
| **LIGHT** | *SL* | **Gentle** | *0 to 2* | **1.44** | **1.23** | **1.09** | **0.95** | **0.86** | **0.79** | **0.72** | **0.65** | **0.59** | **0.54** |
| | | **Moderate** | *3 to 6* | **1.47** | **1.26** | **1.12** | **0.98** | **0.89** | **0.81** | **0.74** | **0.67** | **0.61** | **0.56** |
| | | **Steep** | *7 to 12* | **1.7** | **1.49** | **1.34** | **1.18** | **1.09** | **1.01** | **0.94** | **0.84** | **0.78** | **0.73** |

| FIELD CLASSIFICATION | | | | PERCENTAGE COVER (%) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | | | |
| **Class** | *Soil* | **Slope** | *Degrees* | *<10%* | *10-20%* | *20-30%* | *30-40%* | *40-50%* | *50-60%* | *60-70%* | *70-80%* | *80-90%* | *90-100%* |
| **HEAVY** | *C* | **Gentle** | *0 to 2* | **9.7** | **8.46** | **7.56** | **6.67** | **6.06** | **5.52** | **5.03** | **4.52** | **4.12** | **3.73** |
| | | **Moderate** | *3 to 6* | **9.95** | **8.69** | **7.78** | **6.88** | **6.27** | **5.72** | **5.22** | **4.72** | **4.31** | **3.92** |
| | | **Steep** | *7 to 12* | **10.1** | **8.83** | **7.91** | **7.01** | **6.39** | **5.84** | **5.34** | **4.83** | **4.42** | **4.03** |
| **HEAVY** | *ZC* | **Gentle** | *0 to 2* | **6.95** | **6.03** | **5.38** | **4.72** | **4.27** | **3.87** | **3.51** | **3.14** | **2.84** | **2.55** |
| | | **Moderate** | *3 to 6* | **7.2** | **6.27** | **5.61** | **4.94** | **4.49** | **4.08** | **3.71** | **3.34** | **3.03** | **2.75** |
| | | **Steep** | *7 to 12* | **7.34** | **6.4** | **5.73** | **5.06** | **4.6** | **4.19** | **3.82** | **3.44** | **3.14** | **2.85** |
| **MEDIUM** | *ZCL* | **Gentle** | *0 to 2* | **5.21** | **4.52** | **4.03** | **3.54** | **3.2** | **2.9** | **2.63** | **2.35** | **2.13** | **1.92** |
| | | **Moderate** | *3 to 6* | **5.4** | **4.7** | **4.2** | **3.7** | **3.36** | **3.06** | **2.78** | **2.5** | **2.27** | **2.06** |
| | | **Steep** | *7 to 12* | **5.5** | **4.79** | **4.29** | **3.79** | **3.44** | **3.14** | **2.86** | **2.58** | **2.35** | **2.13** |
| **MEDIUM** | *SCL* | **Gentle** | *0 to 2* | **4.63** | **4.06** | **3.65** | **3.25** | **2.97** | **2.73** | **2.5** | **2.28** | **2.09** | **1.92** |
| | | **Moderate** | *3 to 6* | **4.66** | **4.09** | **3.68** | **3.27** | **3** | **2.75** | **2.53** | **2.3** | **2.12** | **1.94** |
| | | **Steep** | *7 to 12* | **4.7** | **4.12** | **3.71** | **3.3** | **3.03** | **2.78** | **2.56** | **2.33** | **2.14** | **1.97** |
| **LIGHT** | *ZL* | **Gentle** | *0 to 2* | **2.17** | **1.89** | **1.68** | **1.47** | **1.33** | **1.21** | **1.1** | **0.98** | **0.89** | **0.8** |
| | | **Moderate** | *3 to 6* | **2.25** | **1.96** | **1.75** | **1.54** | **1.4** | **1.28** | **1.16** | **1.04** | **0.95** | **0.86** |
| | | **Steep** | *7 to 12* | **2.29** | **2** | **1.79** | **1.58** | **1.44** | **1.31** | **1.19** | **1.07** | **0.98** | **0.89** |
| **LIGHT** | *SL* | **Gentle** | *0 to 2* | **1.66** | **1.45** | **1.31** | **1.16** | **1.06** | **0.98** | **0.9** | **0.82** | **0.75** | **0.69** |
| | | **Moderate** | *3 to 6* | **1.69** | **1.48** | **1.34** | **1.19** | **1.09** | **1** | **0.92** | **0.84** | **0.77** | **0.71** |
| | | **Steep** | *7 to 12* | **1.94** | **1.72** | **1.57** | **1.4** | **1.3** | **1.21** | **1.12** | **1.02** | **0.95** | **0.89** |

**Table 10.5 Total soil (t ha$^{-1}$ yr$^{-1}$) transported through a 6 m VFS from a 1 ha field. The top table indicates the soil losses through a VFS with 1 mm stems and the lower table with 14 mm stems. The highlighted areas show values less then a 2 t ha$^{-1}$ yr$^{-1}$ threshold.**

| FIELD CLASSIFICATION | | | | PERCENTAGE COVER (%) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | | | |
| **Class** | *Soil* | **Slope** | *Degrees* | *<10%* | *10-20%* | *20-30%* | *30-40%* | *40-50%* | *50-60%* | *60-70%* | *70-80%* | *80-90%* | *90-100%* |
| **HEAVY** | *C* | **Gentle** | *0 to 2* | **7.1** | **5.82** | **4.98** | **4.25** | **3.75** | **3.33** | **2.97** | **2.62** | **2.34** | **2.1** |
| | | **Moderate** | *3 to 6* | **7.29** | **5.99** | **5.13** | **4.39** | **3.88** | **3.46** | **3.08** | **2.73** | **2.45** | **2.2** |
| | | **Steep** | *7 to 12* | **7.4** | **6.08** | **5.22** | **4.47** | **3.96** | **3.53** | **3.15** | **2.8** | **2.52** | **2.26** |
| **HEAVY** | *ZC* | **Gentle** | *0 to 2* | **5.07** | **4.15** | **3.54** | **3** | **2.65** | **2.33** | **2.07** | **1.82** | **1.62** | **1.44** |
| | | **Moderate** | *3 to 6* | **5.27** | **4.32** | **3.69** | **3.15** | **2.78** | **2.46** | **2.19** | **1.93** | **1.73** | **1.54** |
| | | **Steep** | *7 to 12* | **5.37** | **4.41** | **3.78** | **3.22** | **2.85** | **2.53** | **2.25** | **1.99** | **1.79** | **1.6** |
| **MEDIUM** | *ZCL* | **Gentle** | *0 to 2* | **3.81** | **3.11** | **2.65** | **2.25** | **1.98** | **1.75** | **1.55** | **1.36** | **1.21** | **1.08** |
| | | **Moderate** | *3 to 6* | **3.95** | **3.24** | **2.77** | **2.36** | **2.08** | **1.85** | **1.64** | **1.45** | **1.29** | **1.16** |
| | | **Steep** | *7 to 12* | **4.03** | **3.3** | **2.83** | **2.41** | **2.13** | **1.89** | **1.69** | **1.49** | **1.34** | **1.2** |
| **MEDIUM** | *SCL* | **Gentle** | *0 to 2* | **3.4** | **2.8** | **2.41** | **2.08** | **1.84** | **1.65** | **1.48** | **1.32** | **1.19** | **1.07** |
| | | **Moderate** | *3 to 6* | **3.42** | **2.82** | **2.43** | **2.09** | **1.86** | **1.66** | **1.49** | **1.33** | **1.2** | **1.09** |
| | | **Steep** | *7 to 12* | **3.45** | **2.85** | **2.45** | **2.11** | **1.88** | **1.68** | **1.51** | **1.35** | **1.22** | **1.1** |
| **LIGHT** | *ZL* | **Gentle** | *0 to 2* | **1.59** | **1.3** | **1.1** | **0.94** | **0.82** | **0.73** | **0.65** | **0.57** | **0.5** | **0.45** |
| | | **Moderate** | *3 to 6* | **1.65** | **1.35** | **1.15** | **0.98** | **0.87** | **0.77** | **0.68** | **0.6** | **0.54** | **0.48** |
| | | **Steep** | *7 to 12* | **1.68** | **1.38** | **1.18** | **1.01** | **0.89** | **0.79** | **0.7** | **0.62** | **0.56** | **0.5** |
| **LIGHT** | *SL* | **Gentle** | *0 to 2* | **1.22** | **1** | **0.87** | **0.74** | **0.66** | **0.59** | **0.53** | **0.47** | **0.43** | **0.39** |
| | | **Moderate** | *3 to 6* | **1.25** | **1.03** | **0.89** | **0.77** | **0.69** | **0.62** | **0.56** | **0.5** | **0.45** | **0.41** |
| | | **Steep** | *7 to 12* | **1.5** | **1.28** | **1.14** | **0.99** | **0.9** | **0.83** | **0.77** | **0.68** | **0.64** | **0.59** |

| FIELD CLASSIFICATION | | | | PERCENTAGE COVER (%) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | | | |
| **Class** | *Soil* | **Slope** | *Degrees* | *<10%* | *10-20%* | *20-30%* | *30-40%* | *40-50%* | *50-60%* | *60-70%* | *70-80%* | *80-90%* | *90-100%* |
| **HEAVY** | *C* | **Gentle** | *0 to 2* | **8.55** | **7.29** | **6.42** | **5.6** | **5.04** | **4.56** | **4.12** | **3.69** | **3.35** | **3.03** |
| | | **Moderate** | *3 to 6* | **8.78** | **7.49** | **6.61** | **5.78** | **5.22** | **4.73** | **4.28** | **3.85** | **3.5** | **3.18** |
| | | **Steep** | *7 to 12* | **8.91** | **7.61** | **6.72** | **5.89** | **5.32** | **4.82** | **4.38** | **3.94** | **3.59** | **3.26** |
| **HEAVY** | *ZC* | **Gentle** | *0 to 2* | **6.12** | **5.2** | **4.56** | **3.96** | **3.55** | **3.2** | **2.88** | **2.56** | **2.31** | **2.07** |
| | | **Moderate** | *3 to 6* | **6.35** | **5.41** | **4.76** | **4.15** | **3.73** | **3.37** | **3.05** | **2.72** | **2.47** | **2.23** |
| | | **Steep** | *7 to 12* | **6.47** | **5.52** | **4.86** | **4.25** | **3.83** | **3.46** | **3.13** | **2.81** | **2.55** | **2.31** |
| **MEDIUM** | *ZCL* | **Gentle** | *0 to 2* | **4.59** | **3.9** | **3.42** | **2.97** | **2.66** | **2.4** | **2.16** | **1.92** | **1.73** | **1.55** |
| | | **Moderate** | *3 to 6* | **4.76** | **4.05** | **3.57** | **3.11** | **2.8** | **2.53** | **2.28** | **2.04** | **1.85** | **1.67** |
| | | **Steep** | *7 to 12* | **4.85** | **4.13** | **3.64** | **3.18** | **2.87** | **2.59** | **2.35** | **2.1** | **1.91** | **1.73** |
| **MEDIUM** | *SCL* | **Gentle** | *0 to 2* | **4.08** | **3.5** | **3.1** | **2.73** | **2.47** | **2.25** | **2.05** | **1.86** | **1.7** | **1.55** |
| | | **Moderate** | *3 to 6* | **4.11** | **3.52** | **3.12** | **2.75** | **2.49** | **2.27** | **2.07** | **1.88** | **1.72** | **1.57** |
| | | **Steep** | *7 to 12* | **4.14** | **3.56** | **3.15** | **2.78** | **2.52** | **2.3** | **2.1** | **1.9** | **1.74** | **1.59** |
| **LIGHT** | *ZL* | **Gentle** | *0 to 2* | **1.91** | **1.62** | **1.43** | **1.24** | **1.11** | **1** | **0.9** | **0.8** | **0.72** | **0.65** |
| | | **Moderate** | *3 to 6* | **1.98** | **1.69** | **1.49** | **1.3** | **1.17** | **1.05** | **0.95** | **0.85** | **0.77** | **0.7** |
| | | **Steep** | *7 to 12* | **2.02** | **1.72** | **1.52** | **1.33** | **1.19** | **1.08** | **0.98** | **0.88** | **0.8** | **0.72** |
| **LIGHT** | *SL* | **Gentle** | *0 to 2* | **1.46** | **1.25** | **1.11** | **0.98** | **0.89** | **0.81** | **0.74** | **0.67** | **0.61** | **0.56** |
| | | **Moderate** | *3 to 6* | **1.5** | **1.29** | **1.14** | **1.01** | **0.91** | **0.83** | **0.76** | **0.69** | **0.63** | **0.58** |
| | | **Steep** | *7 to 12* | **1.77** | **1.55** | **1.4** | **1.23** | **1.14** | **1.06** | **0.98** | **0.88** | **0.83** | **0.77** |

**Table 10.6 Total soil (t ha$^{-1}$ yr$^{-1}$) transported through a <u>10 m VFS</u> from a 1 ha field. The top table indicates the soil losses through a VFS with 1 mm stems and the lower table with 14 mm stems. The highlighted areas show values less then a 2 t ha$^{-1}$ yr$^{-1}$ threshold.**

| FIELD CLASSIFICATION | | | | PERCENTAGE COVER (%) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | | | |
| **Class** | *Soil* | **Slope** | *Degrees* | *<10%* | *10-20%* | *20-30%* | *30-40%* | *40-50%* | *50-60%* | *60-70%* | *70-80%* | *80-90%* | *90-100%* |
| **HEAVY** | *C* | **Gentle** | *0 to 2* | **4.54** | **3.24** | **2.45** | **1.86** | **1.45** | **1.13** | **0.87** | **0.65** | **0.47** | **0.32** |
| | | **Moderate** | *3 to 6* | **4.66** | **3.33** | **2.53** | **1.92** | **1.5** | **1.17** | **0.9** | **0.67** | **0.49** | **0.34** |
| | | **Steep** | *7 to 12* | **4.33** | **3.39** | **2.57** | **1.96** | **1.53** | **1.2** | **0.92** | **0.69** | **0.5** | **0.35** |
| **HEAVY** | *ZC* | **Gentle** | *0 to 2* | **3.24** | **2.31** | **1.74** | **1.32** | **1.02** | **0.79** | **0.1** | **0.45** | **0.33** | **0.22** |
| | | **Moderate** | *3 to 6* | **3.37** | **2.4** | **1.82** | **1.38** | **1.08** | **0.84** | **0.64** | **0.48** | **0.35** | **0.24** |
| | | **Steep** | *7 to 12* | **3.44** | **2.45** | **1.86** | **1.41** | **1.1** | **0.86** | **0.66** | **0.49** | **0.36** | **0.25** |
| **MEDIUM** | *ZCL* | **Gentle** | *0 to 2* | **2.43** | **1.73** | **1.31** | **0.99** | **0.77** | **0.59** | **0.45** | **0.34** | **0.24** | **0.17** |
| | | **Moderate** | *3 to 6* | **2.53** | **1.8** | **1.36** | **1.03** | **0.81** | **0.63** | **0.48** | **0.36** | **0.26** | **0.18** |
| | | **Steep** | *7 to 12* | **2.57** | **1.84** | **1.39** | **1.06** | **0.83** | **0.64** | **0.49** | **0.37** | **0.27** | **0.18** |
| **MEDIUM** | *SCL* | **Gentle** | *0 to 2* | **2.17** | **1.56** | **1.19** | **0.91** | **0.71** | **0.56** | **0.43** | **0.32** | **0.24** | **0.16** |
| | | **Moderate** | *3 to 6* | **2.18** | **1.57** | **1.2** | **0.92** | **0.72** | **0.56** | **0.44** | **0.33** | **0.24** | **0.17** |
| | | **Steep** | *7 to 12* | **2.2** | **1.58** | **1.21** | **0.93** | **0.73** | **0.57** | **0.44** | **0.33** | **0.24** | **0.17** |
| **LIGHT** | *ZL* | **Gentle** | *0 to 2* | **1.01** | **0.72** | **0.54** | **0.41** | **0.32** | **0.25** | **0.19** | **0.14** | **0.1** | **0.07** |
| | | **Moderate** | *3 to 6* | **1.05** | **0.75** | **0.57** | **0.43** | **0.34** | **0.26** | **0.2** | **0.15** | **0.11** | **0.07** |
| | | **Steep** | *7 to 12* | **1.07** | **0.77** | **0.58** | **0.44** | **0.34** | **0.27** | **0.21** | **0.15** | **0.11** | **0.08** |
| **LIGHT** | *SL* | **Gentle** | *0 to 2* | **0.78** | **0.56** | **0.43** | **0.33** | **0.26** | **0.2** | **0.16** | **0.12** | **0.09** | **0.06** |
| | | **Moderate** | *3 to 6* | **0.81** | **0.59** | **0.46** | **0.35** | **0.28** | **0.23** | **0.18** | **0.14** | **0.11** | **0.08** |
| | | **Steep** | *7 to 12* | **1.11** | **0.89** | **0.75** | **0.61** | **0.54** | **0.48** | **0.43** | **0.36** | **0.33** | **0.3** |

| FIELD CLASSIFICATION | | | | PERCENTAGE COVER (%) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | | | |
| **Class** | *Soil* | **Slope** | *Degrees* | *<10%* | *10-20%* | *20-30%* | *30-40%* | *40-50%* | *50-60%* | *60-70%* | *70-80%* | *80-90%* | *90-100%* |
| **HEAVY** | *C* | **Gentle** | *0 to 2* | **6.27** | **4.98** | **4.16** | **3.47** | **3.01** | **2.62** | **2.29** | **1.99** | **1.75** | **1.54** |
| | | **Moderate** | *3 to 6* | **6.43** | **5.12** | **4.28** | **3.59** | **3.11** | **2.71** | **2.38** | **2.07** | **1.83** | **1.61** |
| | | **Steep** | *7 to 12* | **6.53** | **5.2** | **4.35** | **3.65** | **3.17** | **2.77** | **2.43** | **2.12** | **1.87** | **1.65** |
| **HEAVY** | *ZC* | **Gentle** | *0 to 2* | **4.49** | **3.55** | **2.96** | **4.26** | **2.12** | **1.84** | **1.6** | **1.38** | **1.21** | **1.06** |
| | | **Moderate** | *3 to 6* | **4.66** | **3.7** | **3.08** | **2.58** | **2.23** | **1.94** | **1.69** | **1.47** | **1.29** | **1.13** |
| | | **Steep** | *7 to 12* | **4.75** | **3.77** | **3.15** | **2.64** | **2.28** | **1.99** | **1.74** | **1.51** | **1.33** | **1.17** |
| **MEDIUM** | *ZCL* | **Gentle** | *0 to 2* | **3.37** | **2.67** | **2.22** | **1.84** | **1.59** | **1.38** | **1.2** | **1.04** | **0.91** | **0.79** |
| | | **Moderate** | *3 to 6* | **3.49** | **2.77** | **2.31** | **1.93** | **1.67** | **1.45** | **1.27** | **1.1** | **0.97** | **0.85** |
| | | **Steep** | *7 to 12* | **3.56** | **2.83** | **2.36** | **1.97** | **1.71** | **1.49** | **1.3** | **1.13** | **1** | **0.88** |
| **MEDIUM** | *SCL* | **Gentle** | *0 to 2* | **2.99** | **2.39** | **2.01** | **1.69** | **1.47** | **1.29** | **1.14** | **0.99** | **0.88** | **0.78** |
| | | **Moderate** | *3 to 6* | **3.01** | **2.41** | **2.02** | **1.7** | **1.48** | **1.3** | **1.15** | **1.01** | **0.89** | **0.79** |
| | | **Steep** | *7 to 12* | **3.04** | **2.43** | **2.04** | **1.72** | **1.5** | **1.32** | **1.16** | **1.02** | **0.9** | **0.8** |
| **LIGHT** | *ZL* | **Gentle** | *0 to 2* | **1.4** | **1.11** | **0.92** | **0.77** | **0.66** | **0.57** | **0.5** | **0.43** | **0.38** | **0.33** |
| | | **Moderate** | *3 to 6* | **1.46** | **1.15** | **0.96** | **0.8** | **0.7** | **0.61** | **0.53** | **0.46** | **0.4** | **0.35** |
| | | **Steep** | *7 to 12* | **1.48** | **1.18** | **0.98** | **0.82** | **0.71** | **0.62** | **0.54** | **0.47** | **0.42** | **0.37** |
| **LIGHT** | *SL* | **Gentle** | *0 to 2* | **1.07** | **0.86** | **0.72** | **0.61** | **0.53** | **0.46** | **0.41** | **0.36** | **0.32** | **0.28** |
| | | **Moderate** | *3 to 6* | **1.11** | **0.89** | **0.75** | **0.64** | **0.56** | **0.49** | **0.44** | **0.38** | **0.34** | **0.31** |
| | | **Steep** | *7 to 12* | **1.43** | **1.2** | **1.06** | **0.9** | **0.82** | **0.75** | **0.7** | **0.61** | **0.57** | **0.53** |

**Table 10.7 Total soil (t ha$^{-1}$ yr$^{-1}$) transported through a 24 m VFS from a 1 ha field. The top table indicates the soil losses through a VFS with 1 mm stems and the lower table with 14 mm stems. The highlighted areas show values less then a 2 t ha$^{-1}$ yr$^{-1}$ threshold.**

### 10.1.2.5 VFS design and placement

Values from Table 10.3 to Table 10.7 can be used to consider any threshold soil loss value. They also enable interrelationships between ground cover fraction, soil type, VFS lengths and soil losses to be extracted and plotted. By inference this may be applied to relationships with pollutants e.g. particulate phosphorus. Figure 10.1and Figure 10.2 illustrate, based on the values in the tables, the relative effects of soil type, ground cover and slope gradient on VFS efficiency for VFSs of 2 m and 24 m respectively. (Only two illustrative examples are used because soil loss is shown in Table 10.3 to Table 10.7 to be proportional to VFS length.) For a given width, soil type is the factor which has the most influence on soil loss, followed by ground cover fraction. The effect of slope is minor, as represented by the upper and lower bar (10° and 1° slopes).

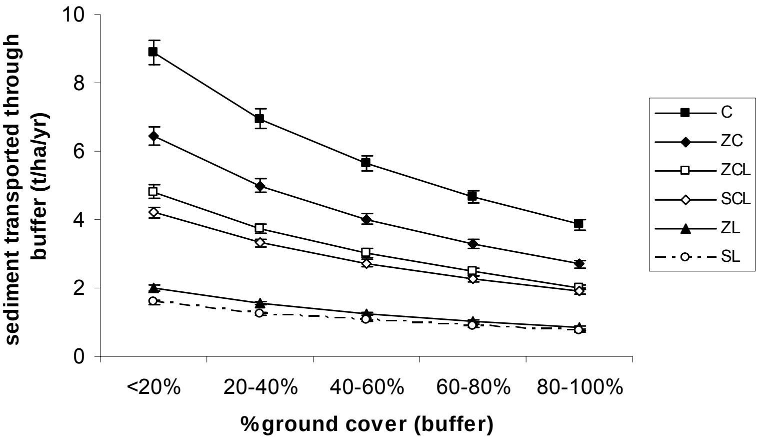



**Figure 10.1The relationship between soil type, ground cover fraction and predicted soil loss for a 4 m VFS. The upper bars represent a 10° slope and the lower bars a 1° slope.**

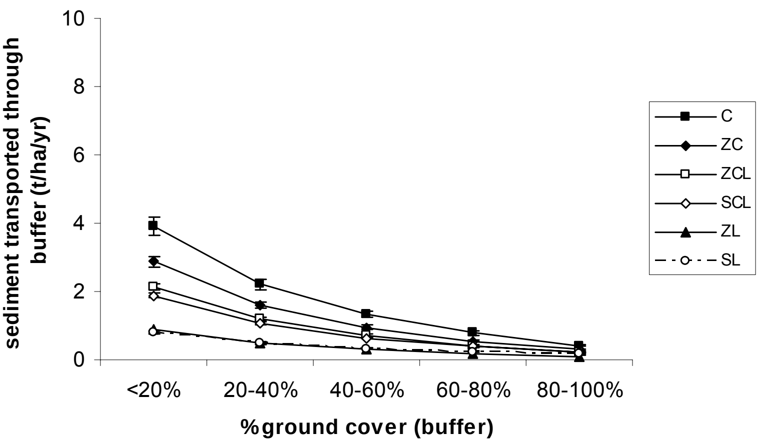



**Figure 10.2 The relationship between soil type, ground cover fraction and predicted soil loss through a 24 m VFS. The upper bars represent a 10° slope and the lower bars a 1° slope.**

## *10.2 Evaluating the system*

### 10.2.1 VFS selection table

A full version of the Decision Support System (DSS) is presented in the Defra report PE0205 (Wood et al., 2007) which includes using the system to first to identify fields where erosion is known to be a problem or a risk. This may be based on local knowledge and experience or may be assessed by an agronomist. Methods for controlling sediment transported from the field are presented including in-field sediment control, ponds, barriers such as fences and stone walls as well as VFSs.

Working through the DSS, diffuse pollution risk is then identified by taking into account the connectivity of the risk fields with local water courses via flow pathways. Again this may be based on a combination of local knowledge, maps and field assessment. Connectivity is important because a field where erosion is severe may be localised and contained, whereas a field with a lower rate of erosion may drain straight into a nearby watercourse, thus, preventing a greater risk of pollution. Additionally, any boundaries intercepting the flow pathways are identified at this stage.

In conditions where flow is in the form of shallow sheetwash, where multiple flow pathways need to be intercepted, where a particular feature needs to be buffered or where a VFS already exists that can be altered or managed, linear VFSs may be selected as the most effective mitigation measure. In order to design a VFS appropriate to the existing conditions a simplified VFS selection table was presented based on Table 10.3 to Table 10.7. The table provides a means of determining the necessary VFS length to control soil loss from a field, based on the soil type and VFS ground cover. This may require adaptation of an existing feature or development of a new VFS feature.

Table 10.8 shows, in green, VFSs that are likely to be acceptable to UK farmers based on the maximum VFS length advocated under the Environmental Stewardship Scheme. It can be seen that on a light sandy loam (SL) soil a VFS of 2 m will be sufficient to reduce soil loss, from a 1 ha field, to less than 2 t $ha^{-1}$ $yr^{-1}$. However, if the percentage cover is less than 20%, for example in the initial stages of vegetation establishment, it will not be as effective. On a sandy clay loam a VFS of 6 m with a greater than 80% cover will be as effective in maintaining soil loss below the threshold as a 24 m VFS with only a 20 to 40 % cover.

Regardless of soil type, slope angle was shown by the intermediary tables to affect VFS performance by less than ±5% and so it was not included in the final version of the selection table. The VFS lengths represent those required for the lower stem diameter value (1 mm) in order to protect against the worst case scenarios. Lengths up to 24 m are rounded to the nearest 2 metres and for lengths over 24 m are rounded to 1 metre.

Six soil classes are presented in the table. Differences between the modelled output suggest that these classes could be further grouped into light, medium and heavy categories. However, whilst SL is similar to ZL in the lighter category, clays and silty clays produced different results and, therefore, differentiation should perhaps be maintained. It should be noted that the higher lengths required for clay soils relate to the need to deposit clay particles, as the model simulates separate particles. In reality, soil loss on clay soils in the UK will rarely exceed 2 t $ha^{-1}$ $yr^{-1}$ so VFSs may not be required. Also, on soils where erosion is most likely the lengths are not too onerous.

Comparing, for any given soil, the <20% cover class with 80-100% shows that soil loss from a VFS is sensitive to ground cover by over a factor of two. Therefore, estimates of GC will have a large influence on whether a VFS is deemed effective or not. For example, estimating ground cover at 20% or 40% for a 24 m VFS on a clay soil will determine whether the 2 t $ha^{-1}$ $yr^{-1}$ threshold is met. This illustrates the importance of making accurate field estimates.

| **FIELD CLASSIFICATION** | | **PERCENTAGE COVER WITHIN VFS (%)** | | | | |
|---|---|---|---|---|---|---|
| Class | Soil | <20% | 20-40% | 40-60% | **60-80%** | 80-100% |
| HEAVY | C | 137 m | 63 m | 47 m | **30 m** | 24 m |
| HEAVY | ZC | 99 m | 53 m | 28 m | **20 m** | 14 m |
| MEDIUM | ZCL | 67 m | 29 m | 18 m | **12 m** | 8 m |
| MEDIUM | SCL | 51 m | 24 m | 14 m | **10 m** | 6 m |
| LIGHT | ZL | 8 m | 4 m | 2 m | **2 m** | 2 m |
| LIGHT | SL | 4 m | 2 m | 2 m | **2 m** | 2 m |

**Table 10.8 Buffer selection table developed for the DSS. The widths represent the buffer width required to reduce net soil loss from the field to less than 2 t $ha^{-1}$ $yr^{-1}$.**

## 10.2.2 Other VFS selection methods

A similar table was produced by Prosser and Karssies (2005) for riparian filter strips in Australian conditions (Table 10.9). This table requires a knowledge of the soil loss from the field before an appropriate VFS length can be selected. In contrast to those derived for the DSS, the guidelines presented by Prosser and Karssies (2005) are based on slope and do not take into account soil type or ground cover fraction. This may be a limitation given the importance of percentage cover, displayed by Table 10.8, in influencing appropriate VFS lengths. Cover is also a factor that will vary with season and age of the VFS and one that can be managed and manipulated unlike soil type and slope. Where similar benefits may be achieved it is likely to be more economically feasible to increase percentage cover on a narrow VFS than to increase the VFS length and take more land out of production.

| soil loss (t/ha/yr) | filter strip slope (%) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 |
| 1 | 2 m | 2 m | 2 m | 2 m | 2 m | 2 m | 2 m | 2 m | 2 m | 2 m |
| 2 | 2 m | 2 m | 2 m | 2 m | 2 m | 2 m | 2 m | 2 m | 2 m | 2 m |
| 5 | 2 m | 2 m | 2 m | 2 m | 3 m | 3 m | 3 m | 4 m | 4 m | 4 m |
| 10 | 2 m | 2 m | 4 m | 5 m | 6 m | 6 m | 7 m | 7 m | 7 m | 7 m |
| 20 | 3 m | 9 m | 11 m | 12 m | 12 m | 13 m | 13 m | 13 m | 13 m | 14 m |
| 30 | 9 m | 15 m | 17 m | 18 m | 19 m | 19 m | 19 m | 20 m | 20 m | 20 m |
| 40 | 15 m | 21 m | 23 m | 24 m | 25 m | 25 m | 26 m | 26 m | 26 m | 26 m |
| 50 | 22 m | 28 m | 30 m | > 30 m | > 30 m | > 30 m | > 30 m | > 30 m | > 30 m | > 30 m |
| 60 | 28 m | > 30 m | > 30 m | > 30 m | > 30 m | > 30 m | > 30 m | > 30 m | > 30 m | > 30 m |
| 70 | > 30 m | > 30 m | > 30 m | > 30 m | > 30 m | > 30 m | > 30 m | > 30 m | > 30 m | > 30 m |

**Table 10.9 Recommended grass filter lengths for typical values of annual soil loss and filter gradient under conditions of dispersed overland flow in Australia. From Prosser and Karssies (2005).**

The output generated by MMF-VFS did not cover the range of soil loss values included in the table by Prosser and Karssies (2005) so there are only limited conditions that can be compared. For those conditions where comparison is possible, Table 10.10 presents the recommendations of Prosser and Karssies for a 10% (5.7°) slope and the equivalent recommendations based on the MMF-VFS output used to derive the VFS selection table. It should be noted that the recommendations of Prosser and Karssies are for trapping all sediment being delivered from a 2 ha field into a 100 m VFS. Table 10.10 relies on existing knowledge of the magnitude of soil loss from a field so it is not so useful for rapid field assessment.

| *Soil loss ($t\ ha^{-1}\ yr^{-1}$)* | *Prosser & Karssies (2005) recommendation* | *VFS selection table recommendation* |
|---|---|---|
| 1 | 2 m | 2 m (ZL) |
| 2 | 2 m | |
| 5 | 4 m | 10 m (ZCL) 24 m (ZC) |
| 10 | 7 m | 6 m (SCL) 24 m (C) |
| 20 | 14 m | 2 m (SL) |
| 30 | 20 m | |
| 40 | 26 m | |
| 50 | > 30 m | |
| 60 | > 30 m | |
| 70 | > 30 m | |

**Table 10.10 Recommendations by Prosser and Karssies (2005) for a 10 % (5.7°) slope and equivalent recommendations based on the VFS selection table.**

The USDA Natural Resources Conservation Service (NRCS) also uses a system based on slope angle and VFS length. This guidance is presented with the equivalent recommendations given by the VFS selection table in Table 10.11. The slopes presented are similar to those in the UK. Some comparison with the other systems is possible, for example, NRCS recommends, on a 6º slope, a minimum of 4.5 m, whilst Prosser and Karssies recommend between 2 m and > 30 m for the same slope. For a soil loss of 10 t $ha^{-1}$ $yr^{-1}$ Prosser and Karssies recommend a 7 m VFS whilst MMF-VFS recommends 6 to 24 m, depending on soil type. The NRCS system covers very few situations.

| *NRCS recommendation* | | *VFS selection table recommendation* | | | | | |
|---|---|---|---|---|---|---|---|
| | | Soil type | | | | | |
| Field slope ° | Minimum width | C | ZC | ZCL | SCL | ZL | SL |
| < 0.6 | 3 m | 24 m | 24 m | 6 m | 6 m | 2 m | 2 m |
| 0 - 6 | 4.5 m | 24 m | 24 m | 10 m | 6 m | 2 m | 2 m |
| 6 - 11 | 6 m | 24 m | 24 m | 10 m | 6 m | 2 m | 2 m |
| 11 - 17 | 7.6 m | - | - | - | - | - | - |

**Table 10.11 VFS selection table recommendations for minimum filter strip length based on VFS selection table and NRCS recommendations for minimum filter strip length based on a maximum field to filter area ratio of 30:1 (30 acres of cropland draining to 1 acre of grass filter) USDA (1988).**

The most comprehensive guidance for UK conditions is provided by Hilton et al. (2003) which presents a protocol for identifying best management practices for reducing diffuse pollution to rivers. The general approach is summarised in Figure 10.3. Detailed guidance and case studies are provided on identifying problem areas and matching the most appropriate measures. However, descriptions of the individual measures do not provide guidance on specific design and placement. Therefore, the DSS designed in the current study extends the selection guidance provided by this system to feature design and management.

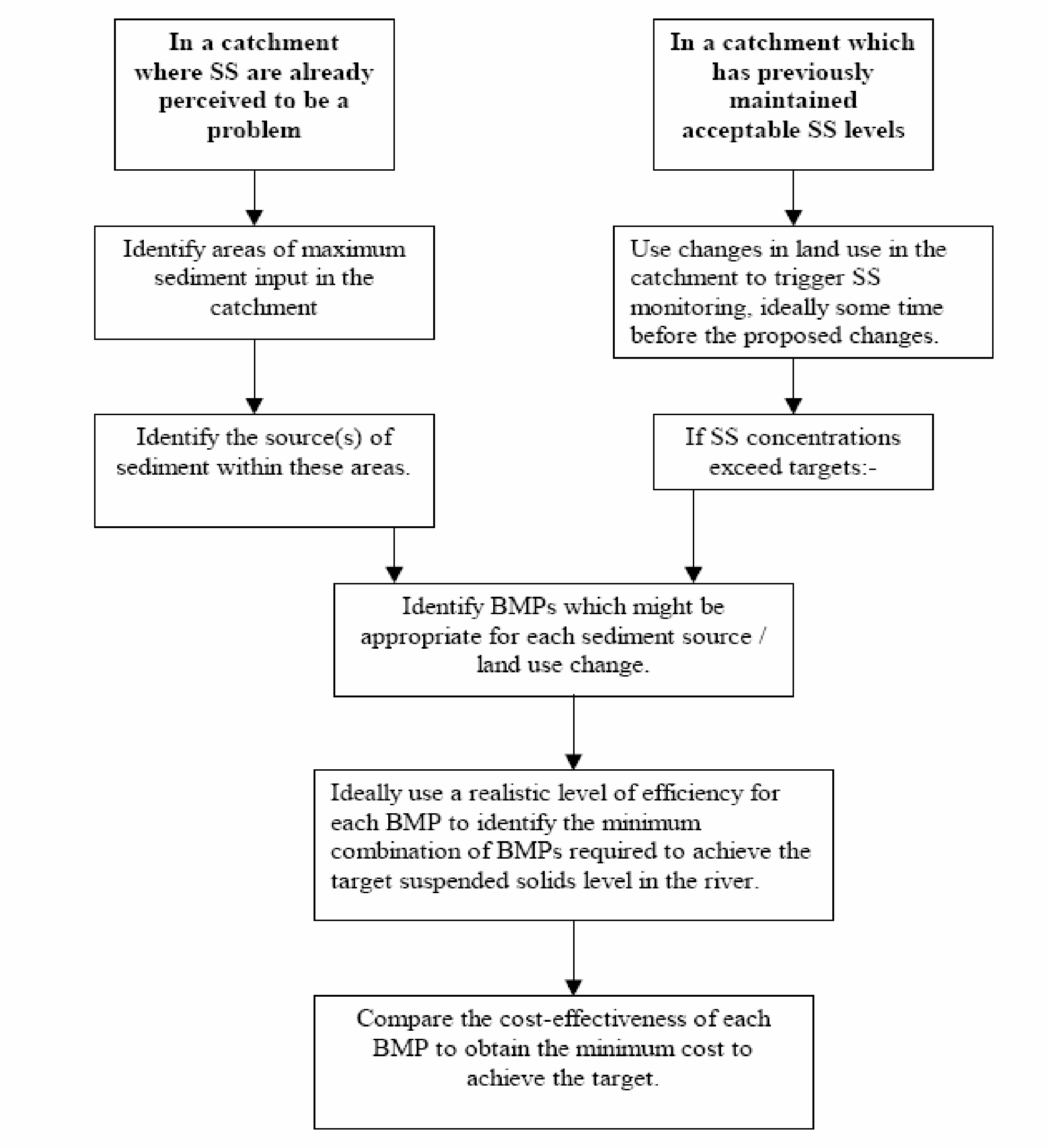



**Figure 10.3 Flow chart of the process to identify appropriate Best Management Practices (BMPs) to reduce suspended solids in a specific catchment (Hilton, Hornby and Moy, 2003). SS = suspended sediments.**

## 10.2.3 Discussion

### 10.2.3.1 Design and placement advice

The system developed in this thesis is the first to incorporate soil type and vegetation cover within guidance on VFS design and placement for UK conditions. It is quick and simple to use in the field and does not have large data requirements. Both soil type and vegetation cover were found, within both published literature and the current study, to impose limitations on VFS performance. Their use in the system is also practical because they can be quickly and easily assessed without detailed measurement or specialist training. The final VFS selection table adopts six soil classes which reflect those used by Defra, although it could be grouped further into light, medium and heavy classes as used in some Defra guidelines (e.g. Defra, 2005a). The recommended lengths match those required by the current Environmental Stewardship Scheme (Defra, 2005a). This provides the potential for integration with other systems, for example the protocol designed by Hilton, Hornby and Moy (2003) or use alongside other Defra guides on soil erosion and soil management (e.g. Defra 2005a and 2005b).

#### 10.2.3.2 Limitations

The main and intermediary tables categorise continuous criteria such as slope and soil texture into relatively few conditions. The system relies on perceived erosion risk and an appreciation of the connectivity of the catchment. Not all of the modelled scenarios were ground truthed with field monitoring.

#### 10.2.3.3 Further development

The system could be developed through further field testing and validation. For example, the system could be used to design field plot experiments from which data could be collected to validate the information presented in the final table. Additional development might include the integration of non-physical factors such as the cost effectiveness of different VFSs. The system could also be linked to recommendations for optimum species based on, for example, suitability to soil type or for achieving a desired ground cover.

## *10.3 Design of VFS architecture*

### 10.3.1 Introduction

The model has the potential to be used as a “soil loss calculator” to evaluate VFS design expressed by simple architectural parameters. This would be of value where it is not favourable to perform extensive field and laboratory testing and where data is required, for example, to target resource management or to inform policy decisions, although, wherever possible, some form of validation should be carried out. This method could also be applied to investigating a specific variable or VFS characteristic. In the following example the model is used to test the parameters stem number (NV) and stem diameter (D). Work on the development of the DSS, described in Section 10.1, showed that altering NV and D will affect the reduction of soil loss even when the same percentage cover is maintained. This section tests this further with the following hypotheses:

1. Increasing stem number is more important than increasing stem diameter.
2. There will be an optimum combination of stem number and diameter for maximum sediment reduction.

## 10.3.2 Method

The model was run ninety times in order to allow for 10 replicates of each combination of variables. In order to introduce some variability and hence create replicates, for each run, values for NV and D were selected at random from the range of values measured in the field. The other input parameters remained constant and were based on a 6 m VFS at the base of a 94 m field with a 5° slope angle. Rainfall and temperature the Parrett catchment were used. Density was calculated by multiplying the number of elements (NV) by stem diameter (D). The lowest density value was 392 mm/m$^2$ and the highest 340564 mm/m$^2$. Based on this range the values were grouped into seven categories (very low, low, medium low, medium, medium high, high and very high) which are presented in Table 10.12.

| ***Category*** | ***Density range (mm/m$^2$)*** |
|---|---|
| Very low | 392 – 48973 |
| Low | 48974 - 97554 |
| Medium low | 97555 – 146135 |
| Medium | 146134 – 194716 |
| Medium high | 194717 – 243297 |
| High | 243298 – 291878 |
| Very high | 291879 - 340564 |

**Table 10.12 Categories and range of density values.**

NV values ranged from 98 to 14701 stems per m$^2$ and D values ranged from 1 to 20 mm. These were each categorised as low, medium and high by splitting the range into 3 groups of 30 (Table 10.13) in order to get an equal number of values in each category.

| ***Category*** | ***NV range (elements per m$^2$)*** | ***D range (mm)*** |
|---|---|---|
| Low | 98 – 4499 | 1 – 8 |
| Medium | 4500 – 9868 | 9 – 16 |
| High | 9869 - 14701 | 17 - 25 |

**Table 10.13 Categories and range of NV (number of vegetative elements) and D (diameter) values.**

Combinations of NV and D values to be used in the model runs were based on Table 10.14 e.g. the top corner "HH" represents high NV and high D whilst the middle right "LM" represents low NV and medium D. For each combination 10 replicates were created from the values in Table 10.13 e.g. for the "LL" category ten low NV values were paired with ten low D values. Soil loss was calculated within the model for each of the resulting 90 pairs of values. Taking the average for each 10 replicates gave an average soil loss for each of the combinations in Table 10.14. The resulting combinations are not realistic based on what could physically exist in the field but were created to give a wide spread of values for model input.

| | *NV* | | |
|---|---|---|---|
| D | HH | HL | HM |
| | LH | LL | LM |
| | MH | ML | MM |

**Table 10.14 Combinations for which soil loss was calculated.**

## 10.3.3 Results and discussion

The values presented in Table 10.15 show that the highest soil loss is generated by a combination of low stem diameter with a low number of stems. The lowest soil loss, and hence the most effective VFS, is achieved with a high stem diameter and a high stem number, as expected. However, a low stem diameter and high number of stems yields less erosion than a low number of stems with a high stem diameter suggesting that increasing stem number is more important than increasing stem diameter.

This confirms that density may comprise various stem number and diameter combinations and that the combination will influence the filtering capability of a VFS. The results indicate that species should be selected, and practices encouraged, which increase stem number rather than stem diameter. This might include species that grow individual stems rather than tussocky growth patterns, dense sowing rates and regular mowing. It suggests that young grass which is likely to include narrow, sparse stem coverage is likely to be less effective than older grass so long as stem number has been maintained.

| *Soil loss (kg m$^{-2}$ yr $^{-2}$)* | | *D (stem diameter)* | | |
|---|---|---|---|---|
| | | *High* | *Medium* | *Low* |
| NV (number of stems) | *High* | 0.59 | 0.64 | 0.84 |
| | *Medium* | 0.64 | 0.72 | 0.98 |
| | *Low* | 0.99 | 0.94 | 1.4 |

**Table 10.15 Soil loss values for each stem diameter and stem number combination.**

The graphs in Figure 10.4 to Figure 10.9 show the variation in predicted soil loss with variation in NV and D. The points plotted are grouped into categories of density (very low to very high) based on the combination of NV and D values. In general there is a decrease in soil loss with an increase in either stem diameter (D) or number of stems (NV). The pattern becomes more evident with an increase in either parameter i.e. the points in Figure 10.4 and Figure 10.9, representing low D and NV respectively, are more scattered than those of the other graphs.

With a low stem diameter, (Figure 10.4) soil loss stabilizes at approximately 0.75 kg $m^2$, when the number of stems is greater than 8000 per $m^2$. With a medium stem diameter, soil loss stabilizes at approximately 0.6 kg $m^2$ with 10,000 stems per $m^2$ or more. With high a stem diameter, soil loss levels out at approximately 0.5 kg $m^2$ when the number of stems is 8000 per $m^2$. As stated in the previous section, this combination of values is not physically possible but was created to represent a wide and continuous range of input values for NV and D.

With a low number of stems, soil loss starts to stabilize at approximately 0.75 kg $m^{-2}$ when stem diameter reaches 0.015 m. With a medium number of stems, increasing stem diameter to greater than 0.01 m gives a slight improvement in soil loss up to a diameter of 0.02 m, after which it stabilizes at approximately 0.6 kg $m^{-2}$. With a high number of stems, soil loss stabilizes at approximately 0.5 kg $m^{-2}$ once a stem diameter of 0.01 m is reached.

From each graph legend the level of density represented by each point can be determined. Figure 10.4 suggests that with a low stem diameter, density can be increased and hence soil loss reduced by increasing the number of stems. In contrast, Figure 10.7 suggests that with a low number of stems not much improvement can be gained by increasing the diameter of the stems. From a VFS management perspective this suggests promoting a dense growth of individual stems and avoiding grass that grows in tussocks.

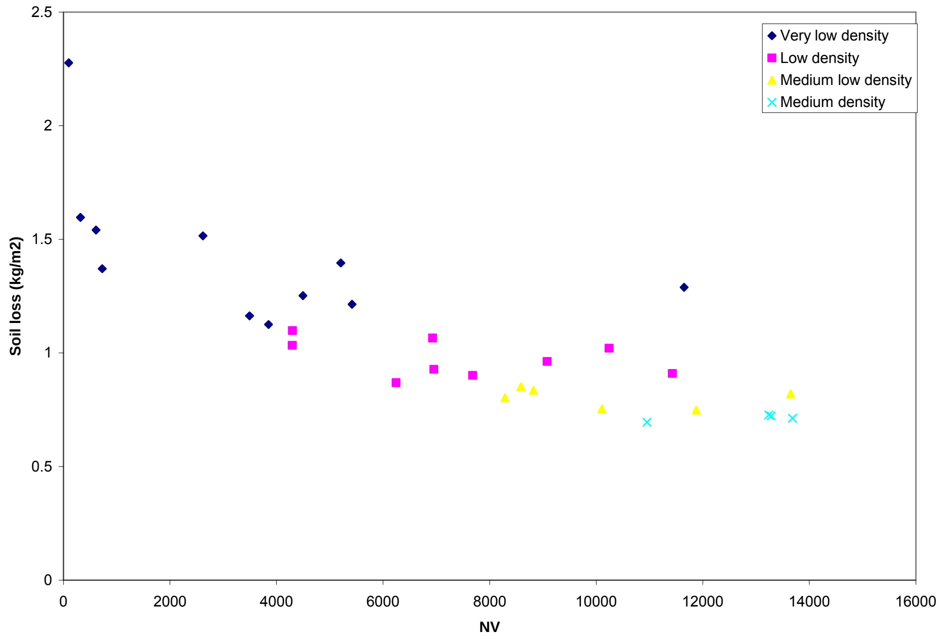



**Figure 10.4 Predicted soil loss plotted against NV (number of vegetative elements per m$^2$). All stems from the "low diameter" category i.e. < 8 mm. Stem density is D x NV.**

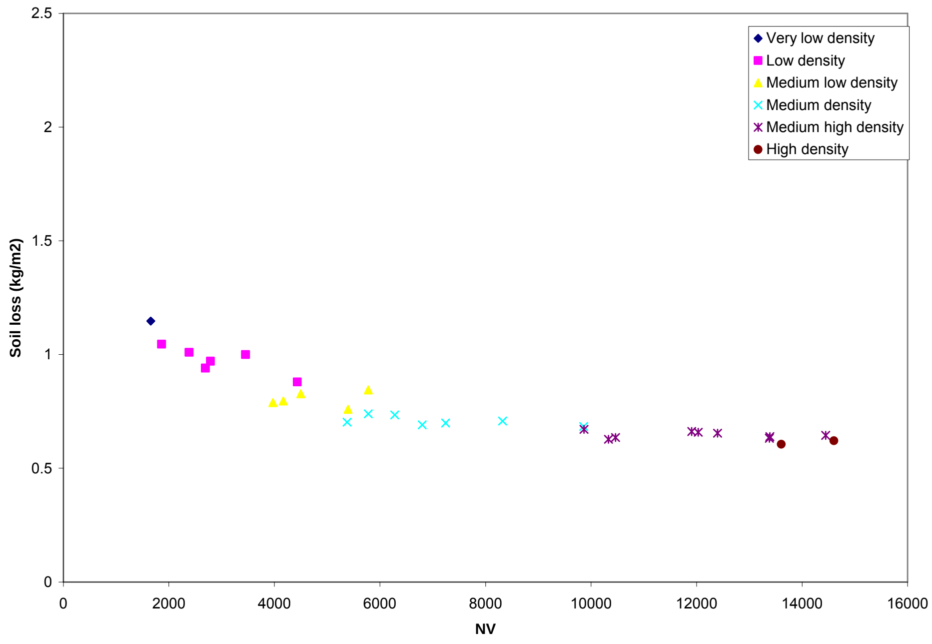



**Figure 10.5 Predicted soil loss plotted against NV (number of vegetative elements per m$^2$). All stems from the "medium diameter" category i.e. 9-16 mm. Stem density is D x NV.**

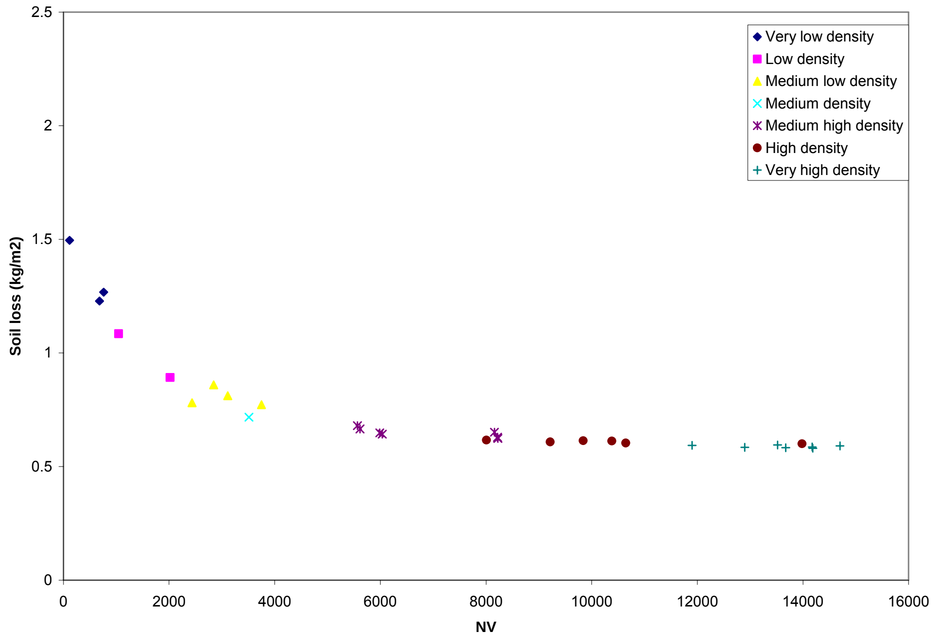



**Figure 10.6 Predicted soil loss plotted against NV (number of vegetative elements per m$^2$). All stems from the "high diameter" category i.e. > 17 mm. Stem density is D x NV.**

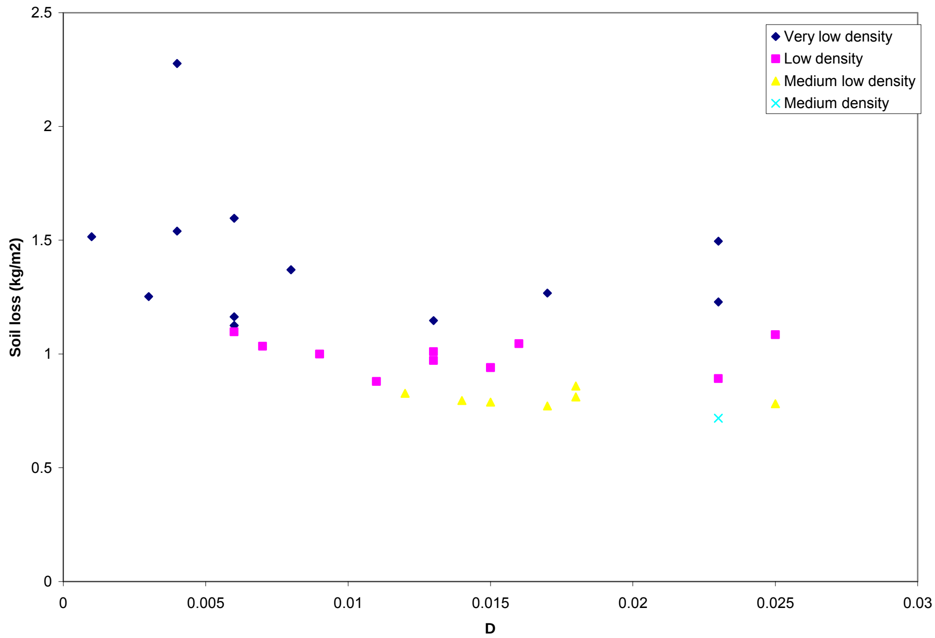



**Figure 10.7 Predicted soil loss plotted against D (stem diameter, m$^2$). All stems from the "low number of stems" category i.e. < 4499 mm. Stem density is D x NV.**

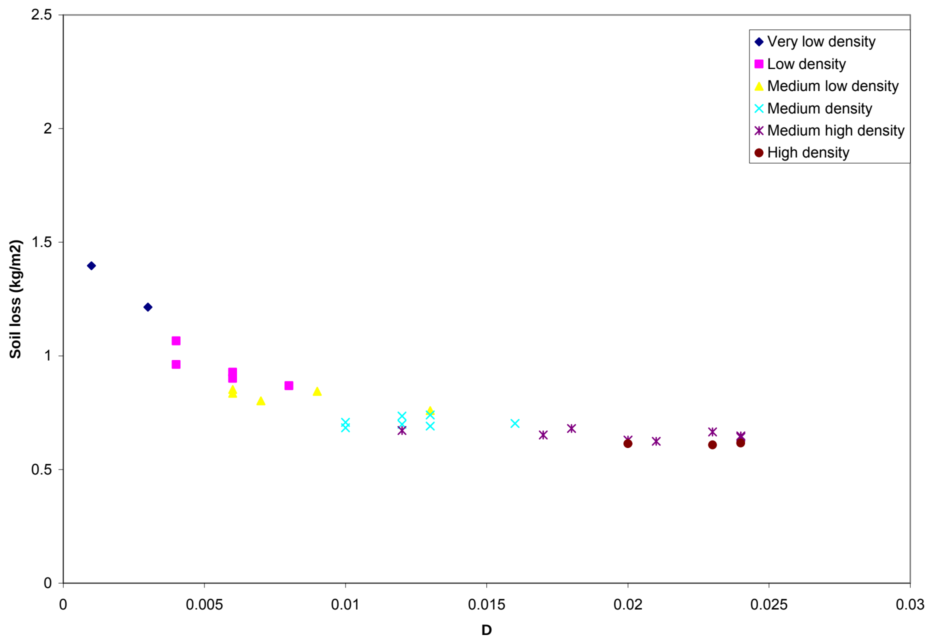



**Figure 10.8 Predicted soil loss plotted against D (stem diameter, m$^2$). All stems from the "medium number of stems" category i.e. 4500 - 9868 mm. Stem density is D x NV.**

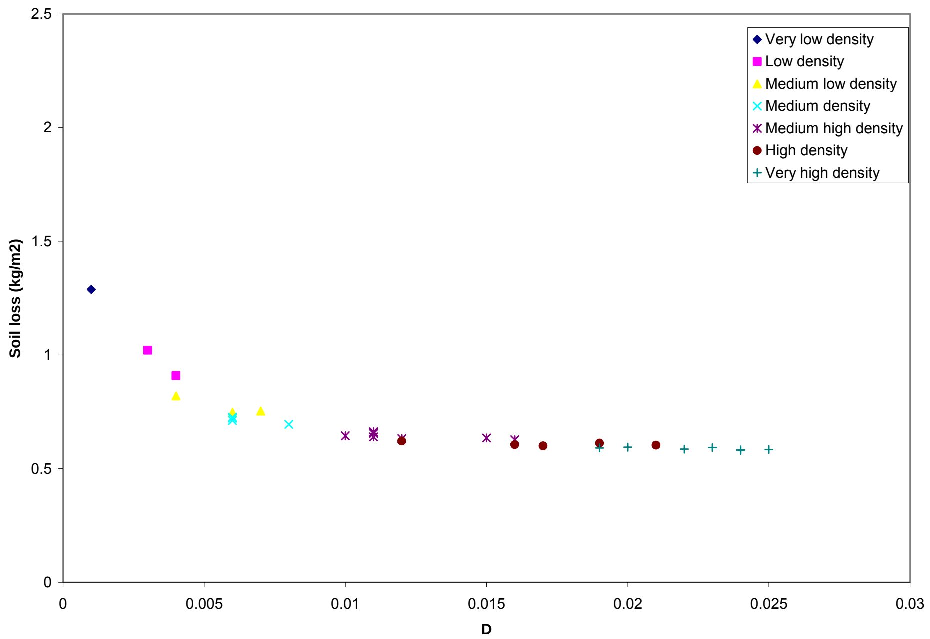



**Figure 10.9 Predicted soil loss plotted against D (stem diameter, m$^2$). All stems from the "high number of stems" category i.e. > 9869 mm. Stem density is D x NV.**

### 10.3.4 Conclusions

The model was used to generate soil loss values for field and VFS scenarios with different vegetation densities. Density was altered by using different combinations of stem diameter and number of stems. The results showed that greater improvements in trapping performance are more likely to be associated with increasing the number of stems than increasing the stem diameter. The greatest reductions in soil loss, 0.62 $m^2/m^2$, were achieved with combinations that included a stem diameter of 0.01 m and that included 8000 stems per $m^2$. The next stage would be to validate these results with field or laboratory experiments designed to test the results. Simulated scenarios may be assessed not only on soil loss but using any combination of soil loss, sediment retention, clay enrichment, runoff reduction or grass strip factor which compares soil loss with a VFS to soil loss without a VFS. It is always favourable to input measured values but in the absence of these, model parameters may be taken from paper maps, farm plans, farmer or land owner knowledge and model guide values.

#### 10.3.4.1 Model use

The model was useful in allowing the creation of combinations of parameter values. However, the values used were taken from limited field measurements and it was not possible to determine how much a parameter is likely to vary with the variation of a different parameter. Further model development may include writing rules for the combinations of vegetation parameters that are likely to be found in the field. This would require extensive field measurements and validation to determine, for example, how canopy cover is likely to vary with ground cover and to be able to derive sensible stem diameter and density values for different ground cover and canopy cover values.

#### 10.3.4.2 Flexibility of output

The work shows that the model can be easily set up to test the effect of different parameter values on soil loss. Testing of vegetation parameters could be carried out for a range of conditions including different slopes, soils and rainfall characteristics. The output presented can be used to derive relative trends in VFS performance caused by varying the stem number and diameter. However, deriving absolute values would require validation of a range of parameter combinations as described above.

## *10.4 Catchment planning*

### 10.4.1 Introduction

At the catchment scale, the main benefit of a GIS system is the integration of spatial data. Spatial characteristics identified in this study as being useful for VFS placement include slope angle, soil type, land use and flow pathways for runoff and suspended sediment. The benefits of GIS to VFS design and placement include catchment scale planning and assessment, evaluating combinations of VFSs and other conservation measures and considering the impacts of VFS placement on landscapes and stakeholders (Tomer et al. 2003). Fried et al. (2004) suggested that the ideal GIS model would "prioritize locations for [VFS] installation by both type and magnitude of potential non-point source pollution problem and likelihood of [VFS] effectiveness, operate at a high enough resolution to guide decisions about efficient action on individual parcels and be parsimonious in its requirements for data and skilled analysts."

### 10.4.2 Integration of MMF and GIS

#### 10.4.2.1 General approach

Advantages of incorporating the MMF-VFS approach into a GIS include wider coverage (for example, catchment scale integration of several layers of spatial data) evaluation of VFSs and other conservation measures as well as the generation of sediment production and deposition maps for use in policy and management decisions. Figure 10.10 presents one approach to incorporating the model into a GIS structure. This concept uses each of the input parameters as a spatial layer of data plus a representation of the hydrological connectivity of the area. In this approach, the model input data is exported to an Excel spreadsheet where the calculations are performed and the output, for example soil loss, runoff, particle detachment and clay content are imported back into the GIS. Maps can then be created based on the model output of interest and used in making decisions over VFS placement.

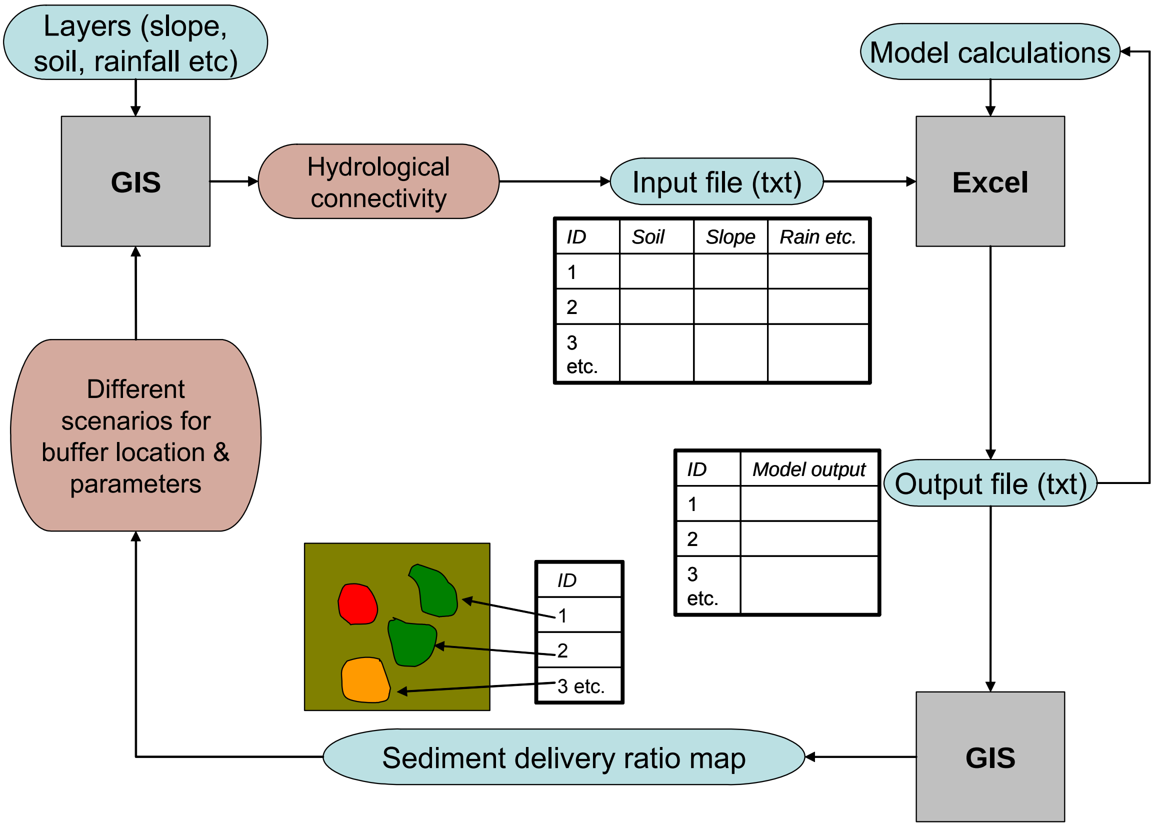



**Figure 10.10 Flow diagram for development of a GIS system.**

### 10.4.2.2 Hydrology network

A conceptual network of flow pathways (hydrology network) was devised by De Luca (2004) which is capable of accounting for landscape factors that influence sediment production and transfer, including VFS features. The process required production of a land use map and a VFS features map (Figure 10.11 and Figure 10.12). Sediment production maps indicate how much sediment is delivered to each section of the flow network by its contributing area (which is based on subcatchment delineation from a Digital Elevation Model (DEM)) (Figure 10.13). Finally, the amount of sediment generated at points along the network is calculated with and without VFSs present (Figure 10.14).

The network was successfully devised and implemented but the author notes a number of areas for development. Work carried out within the current study may be used to progress this development in three key areas.

a) The buffering potential of the features used in the network was not precise due to a lack of information available. The field information collected, and model runs performed, within the current study would enable the classification and quantification of the sediment trapping capability of features such as those presented in Figure 10.12.

b) The USLE was applied to the network in order to generate sediment production, transfer and interception maps. De Luca (2004) states that use of this model as a sediment production estimation algorithm required some assumptions to be made, mainly regarding the connectivity of the elements modelled and the accuracy of the modelled sediment transfer. The MMF-VFS should overcome much of these issues as it is designed to route sediment over multiple elements and allows greater parameterisation than USLE, explicitly taking into account vegetation parameters and providing the potential to consider different particle size classes.

c) Buffering capability was incorporated by each feature acting as a valve which allowed a certain fraction of sediment to pass through. This could be revised to take into account design and site factors as evaluated within the current study. For example, percentage ground cover, presence of in-field conservation features and likelihood of concentrated flow. This would mean that meaningful values based on measurable factors would determine the sediment transferred through a VFS.

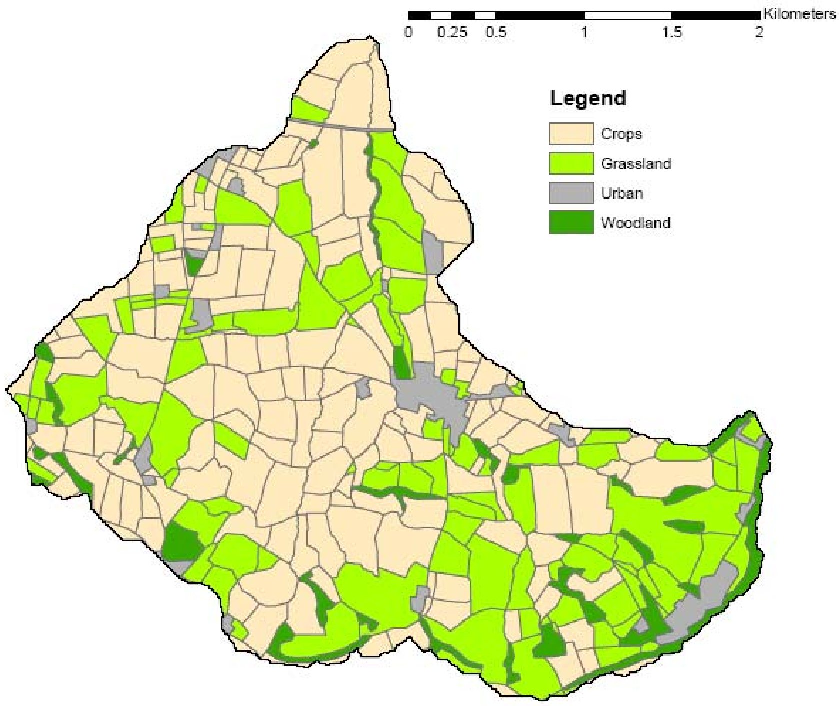



**Figure 10.11 Land use map of an undefined subcatchment of the Parrett catchment (De Luca, 2004).**

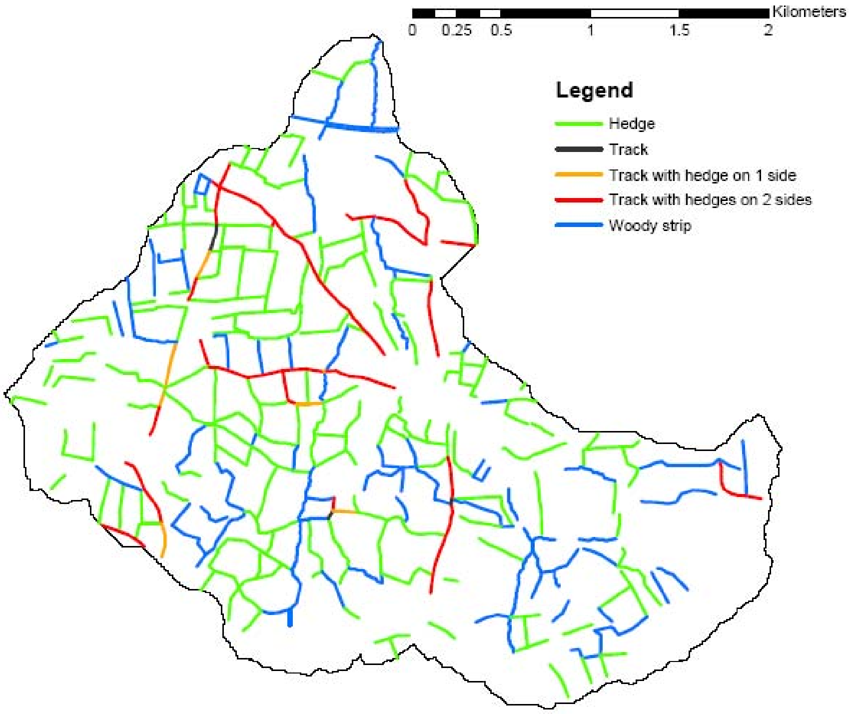



**Figure 10.12** **VFS features map of an undefined subcatchment of the Parrett catchment (De Luca, 2004).**

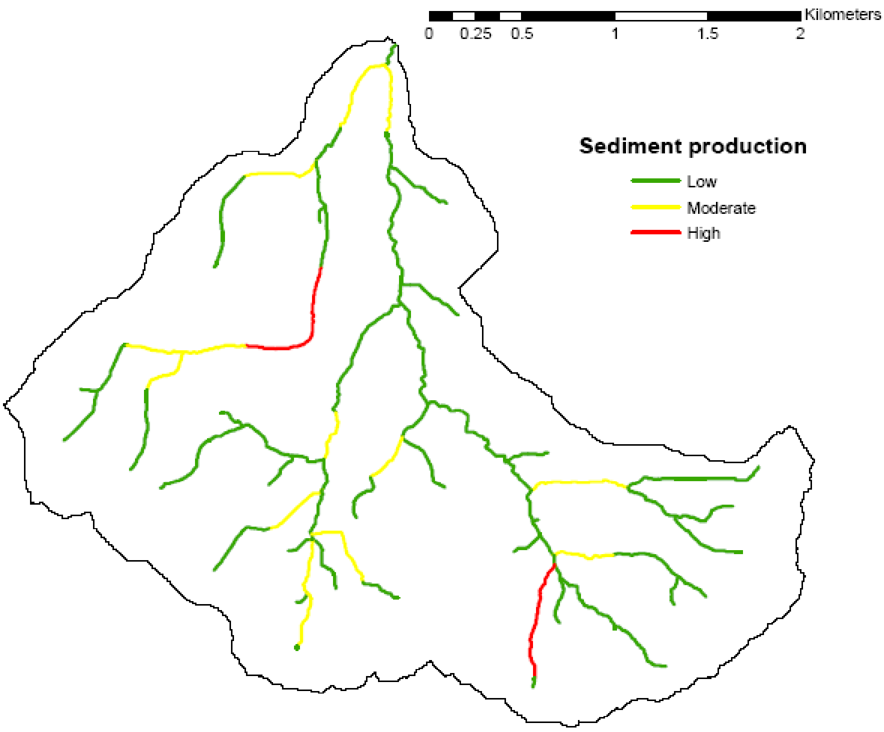
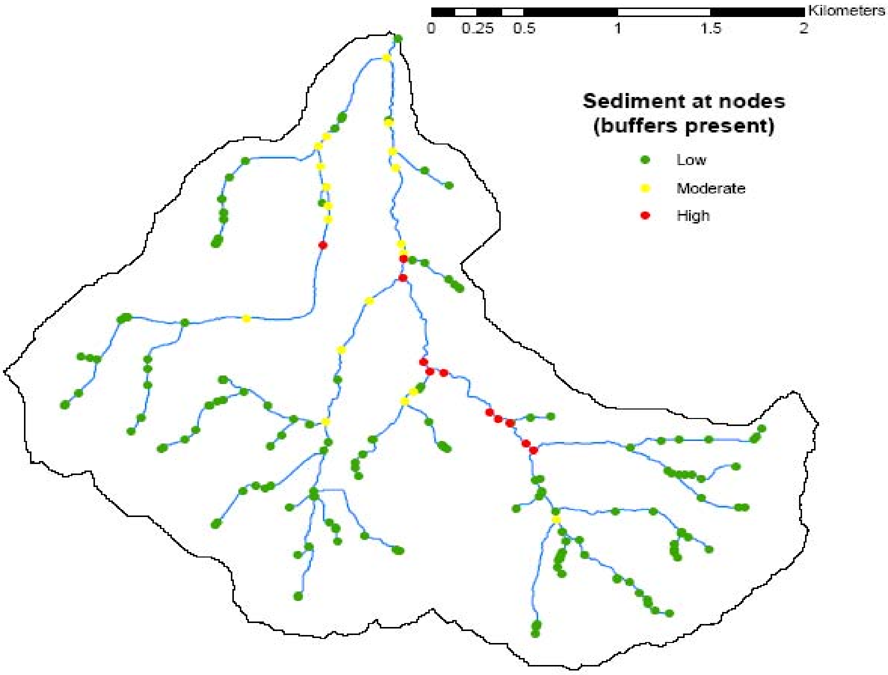



**Figure 10.13** **Sediment production map of an undefined subcatchment of the Parrett catchment (De Luca, 2004).**



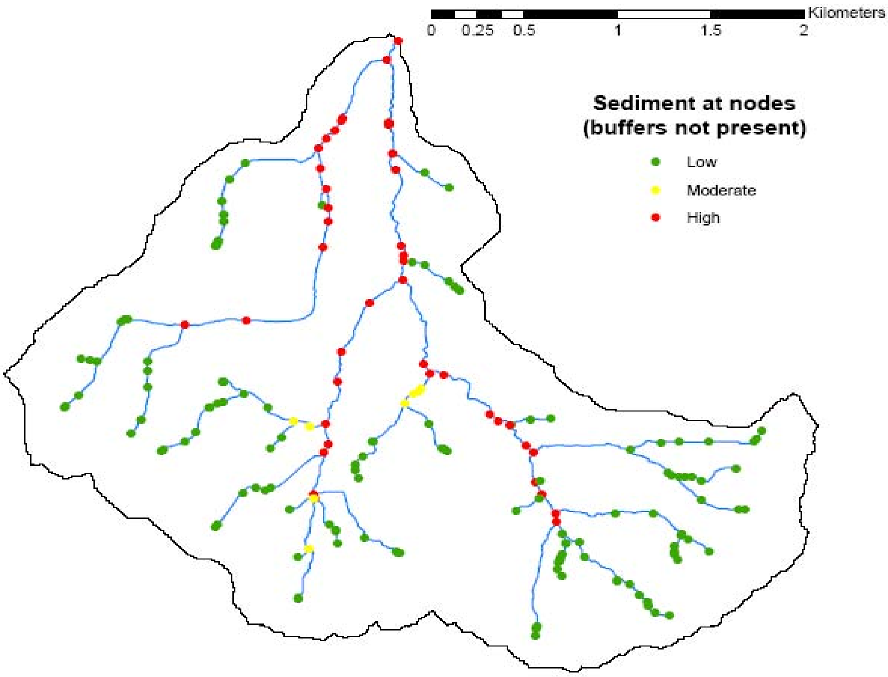




**Figure 10.14 Sediment measured at nodes with (top) and without (bottom) VFSs for an undefined subcatchment of the Parrett catchment (De Luca, 2004).**

## *10.5 Summary and conclusions*

### 10.5.1 Design and placement tools

This chapter demonstrates the application of the MMF-VFS model to three approaches to making decisions on VFS design and placement. The Decision Support System represents a practical field method but relies on running the model for a set of pre-determined conditions. This may limit application to a wider set of conditions, although tables for other scenarios (e.g. rainfall, contributing field areas) could be developed in further work.

The second application of the model was effective in generating data to inform VFS design. This represents a useful tool as it can be adapted to simulate and answer questions on a range of conditions. The next stage would be to match effective combinations of parameter values with species. However, unlike the DSS this approach requires running the model and, therefore, some understanding of the requirements for selecting parameter values. Validation of the generated data would also be necessary, particularly for deriving absolute values rather than trends.

An outline is presented for a GIS method. The model has the potential to simulate catchment scales by routing sediment and flow over multiple elements whilst a GIS would enable spatial data to be pulled out for different areas. Development of this would provide greater capability in geographic data interrogation and representation of more complex landscape scenarios. This development should include testing the capability of the model to deal with multiple elements as the scenarios in this study have focused on one VFS element downslope of one field element. The number of elements needed would depend on the complexity of the landscape to be modelled. Modelling multiple elements would allow different flow paths to be incorporated i.e. flow reaching the VFS from different angles. Field validation of a range of landscape conditions and VFS combinations would also be necessary but could be carried out using a version of the evaluation form used in Chapter 5.

### 10.5.2 Further development

As described, at the catchment scale, a GIS system would be useful for producing maps showing the suitability of land for VFS installation and identifying areas where the most benefits could be gained. The field results in Chapter 5, however, show that local features (e.g. burrows and ploughed steps) and differences in terrain, which are not likely to be picked up in a GIS, exert a strong influence on VFS performance. VFS behaviour is very site specific and so design and placement planning is likely to require some in-field information. Placement tools such as this may be more useful in the US where in, some areas, blanket installation of all perennial streams is undertaken (Dillaha et al. 1989). In the UK, widespread installation of VFSs over large areas is not compulsory so site scale assessments are likely to provide more value.

To account for localised terrain variation, the system would need a proviso that uniform flow and VFSs in good condition are assumed. Also, following the production of a map of potential VFS locations, the best locations should ideally be visited to determine, from field assessment, the likely effective VFS area (which is likely to change with time). Results could then be corrected by this percent, which may then require re-identification of the best VFS locations from the original map.

Related to the differences in VFS performance with uniformity of flow, a further application for a GIS-MMF-VFS system would be determining the effect of VFS failure. For example, development to enable it to determine the damage in terms of amount and location of deposited sediment, on land and in surface waters, with different amounts of concentrated flow through a VFS would be useful particularly useful in Agri-environment scheme feature planning, particularly if a cost could be attributed to the damage. Again this would require extensive field validation of VFSs with failure points, ideally linked to sediment tracing and river sampling.

In general, a modelling approach is useful where long term monitoring does not exist, as is the case with VFS studies. At this stage in its development, and without further validation, the MMF-VFS is probably most useful for providing relative rather than absolute amounts, unless the output can be corrected for the effective VFS area. This would be useful in catchment planning for VFS combinations and types. A further stage would be to build in the effect of ponding in front of the VFS which may improve the prediction of the particle size fractions being deposited and transported through the VFS. In all cases, it should be noted that the condition of the VFS and uniformity of the flow reaching the VFS are of paramount importance in accurate modelling of real world VFS behaviour.

### 10.5.3 Advancing VFS design and placement

The field monitoring (Chapter 5) was particularly useful in furthering practical advice on VFS design and placement because it enabled observation, over time, of established VFSs of varying condition and performance. The modelling work allowed extrapolation of trends and creation of a wider range of conditions. The practical advice that can be extracted from this, and how this furthers current understanding of VFS design and placement, is illustrated in Table 10.16. This table was used in Chapter 2 to summarise design and placement guidance from published literature and is now presented with further guidance developed from the work presented in this thesis. This chapter has taken up the fifth research hypothesis which proposed that the model can be used to derive the optimum design and location for VFSs in the field. Three applications of the MMF-VFS have been proposed which address a gap in VFS planning, design and management.

| *Section* | *Factor* | *Level of guidance* | *Guidelines provided by available literature in black. Guidance provided/supported by this study provided in red.* |
|---|---|---|---|
| Design | Height | General agreement on optimum range but this may only reflect range of heights tested | 100 to 150 mm. Depends on depth of sediment deposition and flexibility of stems. |
| | Density | No quantitative guidance but agreement that higher is better | High density. High density should be achieved by increasing stem number rather than diameter. Wide stems for trapping leaf litter to provide a barrier to large fluxes of coarse sediment. |
| | Species / vegetative properties | No guidance on species and very little on properties | 1. Uniform, erect grass with a range of stem & leaf angles<br>2. Dense rooting system |
| | Age of vegetation | No quantitative guidance | Older grass. Also depends on season. |
| | Length | General agreement that performance improves with lengths up to 10 m but contradictory results on optimum length | 6 to 10 m for sediment and particulate pollutants, longer for dissolved pollutants. Depends on the size of material trapped and retained during ponding upslope of the VFS. |
| | Other | | 1. Contributing flow lengths < 45 m<br>2. Water table close to the ground surface<br>3. Permeable soils<br>4. VFS maintained in good condition i.e. no failure points or sediment inundated vegetation, high ground cover. |
| Placement | Critical areas | Different approaches to defining critical areas including VSA approach | Consider: erosion rate, distance to nearest watercourse, onsite evaluation, present conservation status, terrain in contributing area and field/VFS interface. |
| | Soil type | Not enough studies to derive guidance | 1. Permeable, un-compacted soils<br>NB Shallow soils most likely to need protection. On the most erodible soils combine VFS with in-field measures. |

| | | | |
|---|---|---|---|
| | Slope angle | General agreement the performance declines with increasing slope gradient but differences in recommendations | Average maximum recommendation of 7.5° |
| Cost | Economic cost | Very few studies of relative costs for different designs & locations | Dependent on soil type, land opportunity costs, tillage practices, field size, VFS size and relative size of field to buffer |
| Management | Time | Effects uncertain depending on ability of vegetation to grow through sedimentation.<br>Lack of long term >4 year performance monitoring | Regular mowing to remove sediment build-up from VFS and to provide leaf litter which can build up against hedge features at the downslope edge of grass VFSs and form a barrier to coarse sediment. |
| | Runoff characteristics | Studies confer that VFS are most efficient when flow is slow, shallow and diffuse | 1. Minimise concentrated flow<br>2. Entry runoff velocity < 0.45 m/sec |
| | Other | | 1. Minimise total sediment load to VFS<br>2. Avoid trafficking on VFS<br>3. Farm along contours<br>4. Remove steps caused by field operations which alter the height of the ground surface at the field/VFS interface.<br>5. Reduce the effect of features which cause flow to bypass the VFS e.g. farm gates and burrows.<br>6. Locate in-field conservation measures to disperse concentrated flow. |

**Table 10.16 Summary of design and placement guidance derived from published literature and the current study.**

# Chapter 11 Contributions to knowledge and recommendations for further work

## 11.1 Contributions to knowledge

The hypotheses posed at the beginning of the study have been assessed in the preceding chapters. Table 11.1 presents the chapters relevant to each hypothesis.

| ***Hypotheses*** | ***Chapter*** |
|---|---|
| 1. Stem diameter has a significant influence on the trapping efficiency of VFS. | Chapter 4, Chapter 5, Chapter 10. |
| 2. A simple soil erosion prediction model can be used to predict the sediment trapping efficiency of laboratory simulated VFS. | Chapter 6, Chapter 7, Chapter 8. |
| 3. The model can be used to simulate the influence of different plant stem diameters on the sediment trapping efficiency of VFS. | Chapter 10. |
| 4. The model can be used to predict the sediment trapping efficiency of established VFSs in the Parrett Catchment. | Chapter 6, Chapter 7, Chapter 8, Chapter 9. |
| 5. The model can be used to derive the optimum design and location for VFS in the Parrett catchment. | Chapter 10. |

**Table 11.1 Hypotheses and relevant chapters.**

### 11.1.1 Hypotheses One and Three

Model results showed that greater improvements in VFS trapping performance are more likely to be associated with increasing the number of stems than increasing the stem diameter. The greatest reduction in soil loss was achieved with combinations that included a stem diameter of 0.01 m and that included 8000 stems per $m^2$.

Validation of these results was not achieved using laboratory or field experiments due to variability in the experimental results. The laboratory and field studies did, however, show the importance of a dense barrier which could be formed from mulch or cut grass and not necessarily the number or diameter of stems. In laboratory experiments simulated shrub stems were not effective at trapping sediment alone, but in the field were observed to be important in providing a rigid barrier against which leaf litter and debris could build up to intercept further flow.

Further work is required to validate the modelled relationships between stem diameter, stem number and VFS trapping performance. Also, it is not yet clear whether the relationship between sediment retention and cover is linear or a negative exponential, or whether there is a threshold cover beyond which further increases in stem number have little effect. In the laboratory these investigations may be achieved by using a greater number of replicates but only if constant growing conditions could be achieved, whilst in the field a more dense sampling strategy is necessary. Whilst Hook (2003) suggests that practical field indicators of a site's potential for sediment retention should emphasize major differences in vegetation that can be evaluated visually, rather than accurate, quantitative measurements, getting to this point is likely to require detailed studies with appropriate measurements. The next stage would be to match optimum vegetative properties with appropriate species.

### 11.1.2 Hypothesis Two

Model output was compared with measured results from a range of laboratory and plot scale studies. Model efficiency values exceeding 0.7 (a score out of 1) were obtained for the results of Dillaha et al. (1989) and Le Bissonnais et al. (2004) suggesting that, for these datasets, the model is effective in explaining output variance. For other datasets the relative differences were more accurate than the absolute numbers. This demonstrates that the MMF-VFS is able to simulate at least relative differences in soil loss and VFS performance for the sites and conditions simulated.

The model was able to simulate differences in soil loss from a buffered plot due to vegetation species. Differences in soil loss due to variations in slope angle and VFS length were not so pronounced in the model results as in the measured results. For the deposited material, sand was under-predicted and clay over-predicted but this could be because the model deals with particles and not aggregates. Limited data was available for testing overland flow prediction. In order to make further conclusions on the robustness of the model, testing should be performed over a wider range of conditions.

### 11.1.3 Hypothesis Four

The model was successfully applied to predicting VFS behaviour in the Parrett catchment. This was achieved by simulating the portion of the VFS which was observed to be active in sediment trapping by taking into account observed paths of concentrated flow. Based only on the field observations made in this study, the average proportion of a VFS active in trapping sediment is 21%. Verstraeten et al. (2006) found a similar effect whereby simulations of a field plot with a straight slope produced a sediment reduction of 70% whilst at the catchment scale, when overland flow convergence and bypassing of VFSs was taken in account, sediment was reduced by only 20%.

Where VFSs intercept water and sediment pathways they are effective in trapping sediment. The field results suggest that a 6 m VFS will trap an average of 1.74 t year$^{-1}$ of material from a field of 1 ha.

Based on various soil loss values reported for arable fields in the UK, this equates to a trapping efficiency of between 3 and 1740%. Values greater than 100% indicate a capacity to trap more sediment than is supplied, i.e. a VFS is likely to be unnecessary. In most cases the majority of coarse sediment is trapped at the upslope edge of the VFS with typically >85% sand. This is consistent with the results of Le Bissonnais et al. (2004), McKergow et al. (2004) and Daniels and Gilliam (1996). However, when the VFS is severely breached, due to gully formation within the contributing upslope field or intense rainfall, sediment can travel much further across the VFS.

In contrast to other reported results, field monitoring results suggests an increase in clay content in front of the VFS. This may be due to fine material being deposited at the VFS edge and then being washed downslope by further rainfall events i.e. the fine material may be deposited but not retained. However, no evidence was collected to support this and this should be investigated with a denser sampling strategy as it holds implications for VFS design, particularly length. Although Neibling and Alberts (1979) observed that most sediment was deposited upslope of a VFS and within the first metre, no-one has reported particle size results at different VFS positions.

Of considerable importance to VFS performance is management. This was extremely variable at the sites monitored. The effects, on flow convergence and sediment trapping, of ploughed field edges, animal burrows, farm vehicles and in-field erosion and conservation should be taken into account in VFS design and maintenance.

### 11.1.4 Hypothesis Five

The model was used within three different approaches to designing and locating VFSs. A paper based Decision Support System is practical for use in the field but relies on running the model for a set of predetermined conditions. This means that application is limited to those conditions for which the model has already been run. The model has potential to be used as a VFS calculator to simulate and answer questions on a range of conditions. In previous studies there has been little systematic study of VFS lengths and many studies have had to use existing VFS lengths rather than derive optimal lengths experimentally (Parkyn, 2004). The model provides the potential to take this recommended approach and to test optimal parameters rather than existing parameters on which to base design guidance. To further develop this potential, work is needed to improve the ability of the model to describe particle size fraction and runoff. Catchment simulation may be achieved by linking the model to a GIS. This would provide greater capability in geographic data interrogation, and representation of more complex landscape scenarios. Further work on this should include testing the capability of the model to deal with multiple elements and complex flow pathways.

Despite the potential development of such tools, the site specific nature of VFS behaviour may, in reality, preclude their value. In particular, placement tools may be more useful in the US where blanket installation along all perennial streams is sometimes undertaken e.g. Dillaha et al. 1989. Overall, it is difficult to derive general design guidance. It is likely that VFS design should always be considered specific to the site characteristics and potential sediment loading. Based on this and the approaches developed in this study, a suggested protocol for using the model for assessing VFS performance is to first measure the effective area for a landscape or existing VFS, then to model that area. Alternatively, following model simulation, results may be corrected, following a field visit, to take into account only the effective VFS area.

## 11.2 Summary and further work

This study recognises that currently, published literature does not provide sufficient information or understanding of VFS performance to enable design and placement decisions to be made in UK agri-environments. Understanding of VFS performance has been furthered by addressing five specific hypotheses. Further work may comprise the following.

a) Further long term monitoring of established VFSs is necessary to determine what happens when areas of deposition are subject to significant storm events and whether they then become sources rather than sinks for sediment and associated pollutants. This should include the effect of in-field conservation measures on VFS performance and whether combinations of mitigation approaches are sustainable over several years.

b) Very little fine material was collected in the VFSs compared with that of the source soil. It is unlikely that all of this passed straight through the VFS as mats at the back, and downslope, of the VFSs did not collect fine sediment. More work is required to determine the mechanisms by which this material is deposited upslope of the VFSs, at the VFS leading edge, as indicated by the soil samples taken through the VFSs and into the upslope fields.

c) It is proposed that sediment trapping efficiency based on actual VFS area may overestimate VFS performance. Field and model results show that as much as 90% of the VFS may not be effective, i.e. does not receive sediment by flow from the field upslope. This proves suggestions by other researchers that concentrated flow substantially limits sediment trapping by VFSs. VFS design, placement and management should include a consideration and quantification of the area of the VFS that is effectively trapping sediment rather than assuming uniform trapping along the entire VFS-field interface.

d) The laboratory component of the study provides an evaluation of experimental design for simulating VFSs at small scales. Further experimental testing at this scale should include biophysical parameters such as bending stress, resilience over flow and roughness. Below ground architecture may also be investigated such as root density, depth and branching and the effect of roots on stability and subsurface flows. Conducting paired experiments at the lab and field scale would inform on the effect of scale when examining the influence of biophysical parameters on sediment trapping. However, live vegetative experiments are difficult to control and should be carried out with adequate time and space to enable suitable replication. An alternative may be to calibrate artificial media as Chapter 4 shows that artificial turf is effective in representing relative patterns between treatments when compared with live grass.

e) The study presents a model capable of predicting VFS efficiency in terms of soil loss and deposition. Further validation of the model would enable it to be applied to a wider range of conditions and should include additional soil types. Further development of the model should include testing of a wider range of particle fall numbers, based on experimental evidence. Such experimental evidence is currently limited.

f) The Decision Support System provides a field tool for guiding VFS design and placement. Tools such as this provide a mechanism for effectively targeting diffuse pollution mitigation measures so that they can provide optimum reduction of the transport of sediment and associated pollutants to watercourses. The more such tools are validated, the more effective they are likely to be, particularly given the variable nature of VFS performance with location.

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# Appendix 1

# Description of buffer type

# A1 Buffer types

## *A1.1 Grass features*

### A1.1.1 Riparian buffer zones

"A buffer feature may be considered as a permanently vegetated area of land most likely, but not exclusively, adjacent to a watercourse and managed separately from the rest of a field or catchment" (Muscutt A.D. et al., 1993). Indeed the most widely reported buffer feature in the literature is the riparian buffer zone. Occurring along the edges of objects to be protected, like rivers and reservoirs, these zones aim to protect water quality and to function as a nutrient filter (Nieswand et al., 1990). They do not necessarily consist of only grass but stands of native vegetation including mature woody vegetation (Correll, 2005) and may be from 20 to over 100 m long in the direction of flow downslope.

Muscutt et al. (1993) outlined the major water quality abatement functions of riparian buffers as:

- surface runoff reduction;
- surface runoff filtration;
- groundwater filtration;
- bank erosion reduction; and
- in-stream filtration.

Additional benefits from a riparian zone may include stabilising channels, preventing stock access to waterways, removing soluble nutrients, providing terrestrial and aquatic habitat, providing corridors for the movement of native fauna and flora between geographically separate areas, maintaining invertebrate communities (Parkyn, 2004) and moderating stream temperature and light (Lee et al., 2004).

A number of reviews of the scientific literature on riparian buffer zones exist (Barling and Moore, 1994; Muscutt et al., 1993; Wenger, 1999; Hickey and Doran, 1994; Parkyn, 2004; Correll, 2005) and the literature of Correll (2005) may be found in an annotated and indexed bibliography at http://www.riparian.net/. It should be noted that the majority of the literature cited relates to the USA and only one of the reviews referenced above (Muscutt et al., 1993) focuses on the UK. This illustrates a gap in the literature since it is likely that it may not always be possible to apply information to other climates and agricultural landscapes. The following review focuses on the UK and considers whether non-UK literature can be directly applied.

### A1.1.2 Grass strips

The filter processes of grass strips are similar to those in riparian zones (Van Dijk et al., 1996) but grass strips usually cover less area than riparian zones, varying between 1 and 25 m in the direction of flow. Grass strips can consist of permanent vegetation, but may also be part of the crop rotation cycle and may exist at the bottom or middle of a field. They are applied for both the filtration of sediment and the removal of nutrients from runoff. The strips change the hydraulic characteristics of the runoff (Dillaha et al., 1987) so that, (a) infiltration is enhanced, (b) sedimentation increases when the flow velocity and transport capacity decrease, (c) filtration of suspended material by vegetation increases, (d) adsorption of solutes to plants and the soil surface increases, and (e) absorption of solutes by vegetation increases (Dillaha et al., 1987).

In England and Wales Defra promote various grass strip features in agricultural landscapes for a range of purposes (e.g. Defra, 2005) including:

a) Beetle banks, which are tussocky grass ridges, generally about 2 m wide, that run from one side of a field to the other whilst still allowing the field to be farmed. They provide habitat for ground nesting birds, small mammals and insects (including those which feed on crop pests). When carefully placed across the slope such banks can help reduce run-off and erosion (Defra, 2005a);
b) Buffer strips, which may be 2 m, 4 m, 6 m or 10 m in the direction of flow, require maintaining a grassy strip by controlled cutting, not applying fertilisers or manure and not using the strip for access (Defra, 2005a). They are intended to protect sensitive features such as wetlands and woodlands from fertilisers and pesticides as well as providing wildlife habitat;
c) Field corner management refers to the provision of a grassy area less than 1 ha, for example, one that is awkward to reach with machinery or of lower productivity, in order to provide habitat for insects and birds (Defra, 2005a);
d) Conservation headlands are 6 to 24 m wide strips along the edge of cereal crops where the careful use of sprays allows populations of broadleaved weeds and their associated insects to develop (Defra, 2005a).

### A1.1.3 Grass hedges

Like grass strips, grass hedges (sometimes termed ‘grass barriers’) are set out along contour lines and are separated by strips of arable land. They consist of 0.3 to 1 m strips of permanent, erect grass such as Vetiver, which disperses concentrated runoff, retains sediment and, in the longer term, leads to the development of terraces (Dabney et al., 1995). Essential features for suitable grass species are stiff stalks, which do not bend under the pressure of typical water flow, and a high stalk density in order to reduce the flow velocity and cause ponding (Van Dijk et al., 1996). Runoff that accumulates against the barrier is released gradually to the field down-slope. Sedimentation mainly takes place in ponded areas upslope of, and immediately adjacent to, the vegetative barriers. The runoff itself is affected in three ways by the barriers (Van Dijk et al., 1996):

1) The catchment discharge is delayed as a result of water storage in the ponds along the barriers and the subsequent delivery of this water. Peak discharge will also be reduced.

2) If barriers follow the slope contours accurately, shallow overland flow will disperse when it meets the barrier, thus reducing the risk of gully erosion. This may not be so effective for concentrated flow.
3) The combination of these two effects will (assuming good soil structure and appropriate soil types) lead to an increase in infiltration of water into the soil.

Gilley et al. (2000) described substantial reductions in runoff and soil loss, under both tilled and no-till conditions, with grass hedges covering only 7% of the total experimental plot area. The main advantage of grass hedges is their relatively small spatial extent (Young, 1997). However, the narrow structure makes them vulnerable to breaching by water breaking through, often as a result of animal-burrows that facilitate subsurface flow and tunnelling, which strongly increases the erosion hazard (Van Dijk et al., 1996). The risk of breakthrough can be reduced by a careful layout of the strips (following the slope contours and a well-founded distance between the barriers) and by conscientious maintenance (Van Dijk et al., 1996).

The erosion control potential of agroforestry systems is generally recognized (Bregman, 1993) and examples include the use of hedges. Observing leucaena hedges in India over a nine year period Narain et al. (1997) found a reduction in soil loss and runoff which they attributed in part to the barrier effect and also to the formation of micro-terraces by sediment deposition along the contours. They promoted the method as one which offers productivity as well as protection in areas with gentle slopes. Hedges can hence contribute to sustainable agriculture and rural development by providing a conservation measure with the added opportunity for some harvesting (Hellin and Larrea, 1997).

## *A1.2 Hedgerows and related features*

### A1.2.1 Hedgerows

In a UK context, a hedge can be defined as “a more or less continuous line of woody vegetation that has been subject to a regime of cutting in order to maintain a linear shape” (Defra, 1993). Commonly hawthorn (*Crataegus monygna*) or blackthorn (*Prunus spinosa*), they are likely to offer both a different structure and a lower stem density to the grass hedges described previously. The term hedgerow is used to include the vegetation within, above and alongside the hedge itself including hedgerow bottom flora, hedgerow trees and any adjacent field margins (Barr and Gillespie, 2000). Outside the UK hedgerows are frequently much bigger and less subject to management regimes (for example, fencerows in the US and Canada and roadside vegetation in Australia and South Africa).

It is not always clear within the literature on buffers which type of hedge is being referred to. Generally though, in the UK, hedgerows do not appear within the literature on buffer features and it is not clear how much of the published information specifically on grass hedges in other climates can be directly transferred. However, a well maintained hedgerow planted along the contour does constitute a porous barrier crossing the paths of concentrated flow channels which suggests a buffering effect in slowing runoff, causing temporary ponding upslope of the barrier and allowing time for the settling of suspended sediment. Furthermore, Owens et al. (2007) describes sediment deposition immediately upslope of hedgerows which reduced sediment transfers from hillslopes to watercourses in Devon, UK.

Herzog (2000) is one of the few authors to use the term hedgerows in the context of buffering, suggesting that these linear landscape elements are powerful tools in soil conservation and water quality improvement because they have the potential to control the fluxes of matter and energy in landscapes while requiring only a relatively limited surface. Ryszkowski (1992) suggested that hedgerows and shelterbelts root deeper than annual crops and have higher evapotranspiration which enables them to function as “ecological water pumps”. At the same time they intercept nutrients contained in lateral flows of water in the subsoil (Herzog F., 2000).

### A1.2.2 Ditches

Hedges in UK agricultural environments are often sited adjacent to a ditch and ditches are, therefore, considered here as hedgerow-related buffer features. Agricultural drainage ditches are constructed primarily to facilitate removal of excess surface and shallow ground water (Dabney et al. 2006). They range in size from small intermittently flooded ditches draining individual fields to higher order, permanently flooded channels with very large capacities. Moore et al. (2001) demonstrated that when these ditches are vegetated, they may significantly attenuate the movement of pollutants through them. The mechanism of this processing is not clear but is likely to be associated with leaves, stems and roots providing additional surfaces for deposition, adsorption, absorption and the activity of associated micro-organisms. Properly managed vegetated ditches can function as a special class of constructed wetlands, and both of these landscape features can function as buffers (Dabney et al. 2006).

### A1.2.3 Banks

Where sediment accumulates at the base of hedgerows and builds up a bank, runoff will be intercepted and forced to flow in front of, and parallel to, the hedgerow. The role of the bank in preventing runoff from flowing downslope may be similar to some forms of terrace (see Management Practice section).

## *A1.3 Trees and woodland*

Trees and woodland may exist either within riparian buffer zones or as in-field trees, woodland fences and woodland edges. Such features are protected in UK agriculture as important landscape features as well as for their role in habitat maintenance and soil erosion control. Wooded riparian buffer zones are often remnants of former river plain forests with willows (*Salix sp.*), alder (*Alnus glutinosa*) and a variety of hardwood trees (*Fraxinus excelsior, Ulmus sp., Acer sp., Quercus robus*) (Herzog F., 2000). Forested or planted native trees will provide a buffering function in providing water quality benefits

(Parkyn, 2004). In forestry systems buffer zones generally consist of production trees left beside the stream when the surrounding area is harvested (Parkyn, 2004). Fruit and nut trees or high value native tree species that can be selectively harvested may provide an ecological function and a mechanism to remove nutrients such as P from the riparian zone. Forest vegetation in particular can shade streams and lower stream temperatures (Parkyn, 2004.) and trees provide organic matter inputs in the form of leaves and woody debris, creating a diversity of food resources and habitats for in-stream fauna.

Forested buffer zones that are sufficiently dense may also improve water quality by restricting the access of livestock to streams, thereby both reducing inputs of nutrients and bacteria (associated with livestock faeces) and reducing erosion resulting from streambank trampling (Barling and Moore, 1994; Muscutt et al, 1993). For example, Correll (1997) recommended that woody vegetation, especially forest, may be more effective than grass at removing nitrate from groundwater and more effective at providing organic matter in the deeper subsoils, where it is needed for effective denitrification in groundwater. Generally however, woody buffer vegetation appears to be limited in its role in sediment control. For example, McKergow et al. (2004) compared riparian rainforest buffers with grass buffers and found that the former performed poorly due to low vegetation density and a lack of under-storey. Although sediment was deposited during storm events the material was not permanently trapped and was re-suspended during subsequent events, creating a sediment source area. In a further study (McKergrow et al., 2006) higher suspended sediment concentrations in a tree buffer, compared with a grass buffer, were attributed to the lack of understorey vegetation and hence surface cover. In other studies tree riparian buffers have been sediment source areas, with more sediment leaving the riparian buffer than entering (Daniels and Gilliam, 1996; McKergrow et al., 2004).

Verchot et al. (1997) found that on North Carolina Piedmont sites, forested buffers might be either sources or sinks of nutrients in surface runoff depending on season. Forest buffers were ineffective during the winter and spring when water-filled pore space exceeded 25 to 35% and infiltration was low. Conversely, Daniels and Gilliam (1996) found that during the dry season forested ephemeral channels had little vegetation and were effective sediment sinks but were ineffective during large storm events because there was little resistance to flow. The examples cited suggest that the lack of surface vegetation in wooded buffers leaves any deposited sediment vulnerable to remobilisation, and these buffers appear to perform poorly in conditions when the likelihood of erosion is highest. Their potential is likely to be limited further in the UK by the lack of widespread, dense forested areas in agricultural landscapes.

## *A1.4 Water retention features*

“Water diversions reduce the speed of runoff, and therefore erosion, by intercepting water flow across the slope and diverting it to existing water structures or proposed structures such as retention/water basins. They are generally applied to soils subject to severe erosion due to soil type or slope or climate in order to reduce rill and gully formation" (Hilton et al., 2003).

### A1.4.1 Grass waterways

Grass waterways (GWW) are broad, shallow channels often located within large fields, with the primary function of draining surface runoff from farmland and preventing gullying along natural drainage ways (Atkins and Coyle, 1977). Their purpose is to reduce runoff volumes from agricultural catchments due to their comparably high infiltration rates and the reduction in runoff velocity that prolongs the potential infiltration time (Fiener P. & Auerswald K., 2003). Vegetation within the channel may act as a physical filter in removing some of the sediment (Hilton et al., 2003). They are commonly used on roadside margins in sustainable urban drainage systems where they can handle large flow rates, but are also used in large sloping ranges in the USA.

Runoff volume is reduced when adjacent fields produce runoff while the rain intensity does not exceed the infiltration rate in the GWW itself. Sedimentation is mainly controlled by:

a) a decrease in transport capacity caused by reduced runoff velocity;
b) the sieving of particles by dense vegetation and litter; and
c) the infiltration of sediment-laden runoff.

There is usually a selection of fast growing grass sown in the GWW which is mowed frequently to prevent build up of sediment deposits inundating and damaging the sward (Fiener P. & Auerswald K., 2003). The cross section of waterway depends on slope, soil texture and area to be drained (Manyatsi, 1998). Example GWW dimensions are 600 m long and 10 m wide in a 34 ha catchment and 650 m long and 10 to 25 m wide in a 23 ha catchment (Fiener and Auerswald, 2003). Urbonas (1994) suggests that they are effective on slopes less than 0.03 m/m. The performance of a GWW in reducing runoff volume will depend on the length of the side-slopes, the shape of its cross-section in the area of concentrated flow and on the sediment settling due to decreased runoff velocity and the infiltration of sediment-laden runoff. Sediment settling takes place primarily during sheet flow on the side slopes, where Reynolds numbers are small (<200). Most of the settling is expected to occur in the first few metres of the grass filter (Fiener P. & Auerswald K., 2003). Comprehensive design procedures can be found in Hudson (1981).

### A1.4.2 Retention ponds

Ponds have been installed in many countries in Europe; their main purpose is water supply to agriculture but they aid hydropower, irrigation and flood control and some are constructed specifically for capturing sediment to prevent the pollution of rivers downstream (Verstraeten G. & Poesen J., 2000). In this case, ponds reduce the velocity of runoff and therefore help to limit downstream sediment losses (Hawkins and Scholefield, 2003), as well as allowing some treatment of dissolved pollutants (Hilton et al., 2003). Since they require either a reliable base-flow or a surface catchment of at least 5 ha to stop them drying out, they are generally, but not universally, located lower down in the catchment, such as the low point of a drainage slope, ditch or surface runoff area such as a farmyard or car park (Hilton et al., 2003). Their effectiveness may be increased by adding inlet and outlet sumps, by forming a sequence of ponds and/or by aeration of the water and application of oxidants (Hilton et al., 2003). However, their performance in nutrient retention is varied, especially in the case of P, being particularly dependent on water residence time, and their capacity for storage of nutrients and organic matter being finite (Hawkins and Scholefield, 2003). The useful life of ponds is therefore very short unless they are frequently dredged and carefully maintained (Verstraeten G. & Poesen J., 2000).

## A1.4.3 Detention basins

Like retention ponds, detention basins retain storm runoff in order to delay and reduce the runoff peak. However, whilst retention ponds are designed to retain some water at all times detention ponds retain runoff for only short periods and remain dry during dry weather. They perform this purpose following a storm in order to reduce the flow rate, allow settlement of solids, retain the first flush of pollutants, or dilute them before draining into a watercourse, usually through a flow reducer. Detention basins are located close to the runoff source and can be combined with other practices, such as water diversions, ditch management, grass waterways, grass hedges, roof and farmyard runoff intersection and porous pavements, which aim to direct run-off and erosion away from water bodies (Hilton et al., 2003).

## A1.4.4 Wetlands

Riparian wetlands and floodplains have proven to be important depositional zones for sediment and nutrients (Barling and Moore, 1994). Constructed wetlands, like buffer zones and ponds, have often been established to slow down runoff water from agriculture, enhance infiltration and facilitate the sorption of phsophorus to soil and vegetation, thereby trapping sediment and nutrients (Uusi-Kämppä J. et al., 2000).

Wetlands consist of permanently or semi-permanently flooded land, generally, but not necessarily, adjacent to a river, which can slow down the rate of runoff through temporary storage and develop microbial communities in the soil that help break down pollutants (Hilton et al., 2003). An approximate ratio of wetland to catchment area of 0.01 to 0.1 is required.

Uusi-Kämppä et al. (2000) observed that several processes, in addition to assimilation of nutrients, may be active in constructed wetlands:

1. A local reduction of turbulence may be important in reducing water velocity.
2. Vegetation increases retention time under high water velocities compared with basins without vegetation.

3. Vegetation mitigates resuspension of sediments in shallow waters. This is clearly seen in constructed wetlands as vegetation cover increases.
4. Leaves and stems create local deposition surfaces.

### A1.4.5 Floodplains

Floodplains have two potential buffering functions. They may act either as a conduit or a barrier to water movement and associated sediment and solute transport from hillslopes to the river channel (Burt T.P., 1997). They may also retain sediment and associated pollutants following overbank flooding, i.e. the river rather than the hillslope is the source of water and sediment. The ability of a floodplain to act as a pollution buffer between farmland and the river depends fundamentally on the floodplain sediments providing opportunity for deposition of suspended sediment load from surface runoff (as well as processes such as denitrification), which is dependent on their hydrological properties (Burt, 1997). In small to medium-sized (< 10,000 $km^2$) catchments floodplains usually cover a maximum width of 1 km (Burt T.P., 1997). Cooper et al. (1987) used $^{137}Cs$ to map the areal extent and thickness of sediment to determine the amount of deposited sediment in riparian areas during the previous 20 years. Although only a thin layer of sediment had been deposited in the floodplain swamp, the large area available made it an important depositional zone. Additionally, Walling and Owens (2003) reported that sedimentation on river floodplains due to overbank flooding can result in significant reduction of the suspended sediment and pollutant load transported by a river. This has two important implications. First, the accumulation of pollutants may result in enhanced levels of contamination and the potential for future remobilisation back into the system (Walling and Owens, 2003). Secondly, an apparent reduction of the nutrient or contaminant flux at the catchment outlet may lead to a significant under-estimate of the total mass of contaminant mobilised within the catchment (Walling and Owens, 2003).

## *A1.5 Infrastructure*

No reference is made within the literature to the value of stone walls and fences as buffer features. However, some suggestions are given below on their potential buffering roles.

### A1.5.1 Stone walls

Defra encourages the protection and maintenance of stone walls on agricultural land, in England and Wales, for stock management and as landscape and historic features (Defra, 2005b). Since they fit Herzog's (2000) definition of a linear landscape feature it does not seem unreasonable to assume that they might perform a role in preventing the transfer of runoff and sediment. However, whilst walls along contours should trap sediment, their impermeable nature might encourage the flow of water along the wall edge until it meets a gate or opening. Flow through this is likely to be deeper and more concentrated, possibly causing more damage than it might have done as a shallow dispersed flow. This will depend on the porosity of the stone wall.

### A1.5.2 Fences

Fences may have a similar role to stone walls but will usually be of a more open structure. This will allow some sediment to flow through the fence until a bank of transported soil and debris builds up against the fence. The bank might then promote a buffering role by reducing runoff velocity and volume and encouraging grass growth.

## *A1.6 Management practice*

Brief definitions are provided for field management practices which aim to reduce run-off water speeds by reducing the angle or length of slope. Whilst they perform a buffering function little detail is provided here because the focus of the review is the implementation of vegetated features. More detailed descriptions are provided by Hilton et al. (2003), from which the following information is summarised.

### A1.6.1 Strip cropping

The practice of strip cropping alternates, along the contour, bands of row crops with closely growing crops such as grass and clover (Manyatsi, 1998). The row crops limit erosion and the vegetative strips in between trap any detached sediment. For example, in Swaziland strip cropping is encouraged in maize fields. Grass strips of approximately 2 to 4 m are grown between cropped areas of approximately 15 to 45 m (Manyatsi, 1998). Generally, the size of the strip depends on the slope angle and the machinery used. Tippett and Dodd (1995) suggested that strip cropping is similar in effectiveness to VFSs although they do not report efficiency data for either practice.

### A1.6.2 Contour cropping

This practice involves sowing crops or extending seed lines along the slope contours and may be combined with contour cultivations and strip cropping. Erosion and runoff are reduced by the removal of natural drainage channels which occur down slope with conventional cropping. Instead, the crops provide a perpendicular barrier to runoff which, when reduced, improves infiltration and reduces soil erosion. However, this practice is probably not very effective against concentrated flow.

### A1.6.3 Contour cultivation

Cultivating along the contour directs surface runoff and any erosion across the slope rather than down existing seedbeds and tramlines. As a consequence of the reduction in runoff velocity, infiltration and sedimentation will be improved. Soil type and slope gradient and shape will determine the effectiveness of this practice. Again this practice has limited capacity as it is unlikely to be able to deal with concentrated flow.

### A1.6.4 Terracing

Terracing describes "a series of regularly spaced embankments across a slope that form a channel on the up-slope side of the embankment to retain water behind the bank, improving infiltration and sedimentation" (Hilton et al., 2003). Morgan (2005) classifies three main types of terrace:

a) Diversion terraces aim to intercept runoff and divert it across the slope to a suitable outlet. They are therefore placed at an angle to the contour and are suitable for slopes up to 7°.
b) Retention terraces store water on the hillslope and are therefore used where water conservation is required. They are recommended for slopes up to 4.5°.
c) Bench terraces consist of a series of alternating shelves and risers and are used on cultivated slopes up to 30°.

Terraces are typically accompanied by contour planting, strip planting or both and act as a check on any contour row failure (Tippett and Dodd, 1995). The risk of significant loss of useable area means that it tends to be focused on areas where erosion is severe. As the practice is not used in the UK it is not discussed further in this review.

### A1.6.5 Soil berms/contour bunds

These features intercept and divert runoff and eroded material away from watercourses. A raised berm may be created by deep ploughing into a riparian edge. Alternatively, strengthened soil structures may be engineered around a retention pond, detention basin or across a field slope to reduce runoff velocity.

# Appendix 2

# Evaluation form

**General**

Name of surveyor:________________________________ Date: ___________________________

Sub-catchment:___________________________ Grid ref. / Site no.:_________________

Site details *(e.g. access, parking)*:_____________ Section:________________________

Weather *(e.g. current and recent)*:_____________ Element: BUFFER FIELD

Date and magnitude of latest rainfall event: ____________________________________________

Estimated distance upslope to watershed boundary (m)*: __________________________

Estimated distance downslope to watershed boundary (m)*: ____________________________

Stream type:________________Stream width:______________Stream depth:_________

Field dimensions (m)*:____________________________________ ** May be easier to measure on a map*

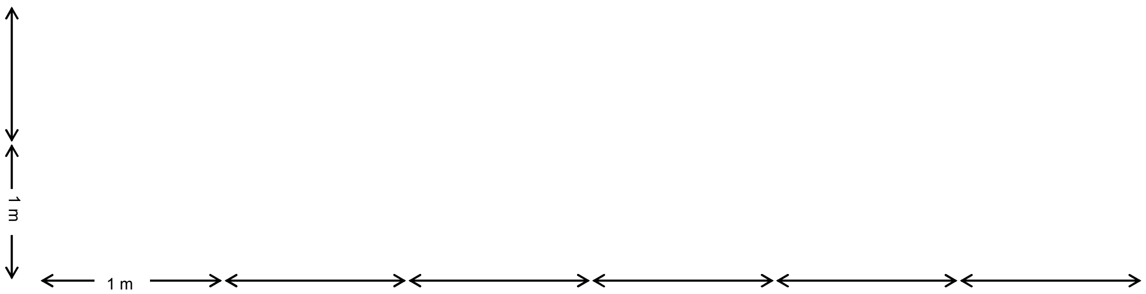
**Buffer description**

Buffer feature(s): SHRUB HEDGE GRASS BANK WOODLAND DETENTION POND

MISCANTHUS OTHER______________________________________________________________

Buffer position: MID-FIELD RIPARIAN Distance from stream (m*) __________

Sketch of feature:




Nature of feature: AGRI-ENVIRONMENT COUNTRYSIDE STEWARDSHIP

NATURAL FEATURE OTHER ___________________

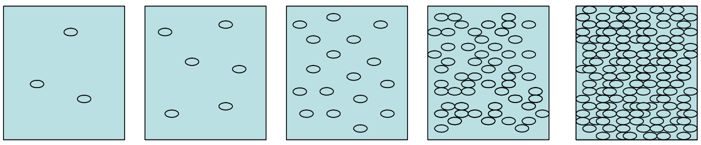
Buffer width (m) *(max.)*:_____________________Buffer width (m) *(min.)*:______________

Buffer width (m) *(modal/average.)*:_________________________________________________

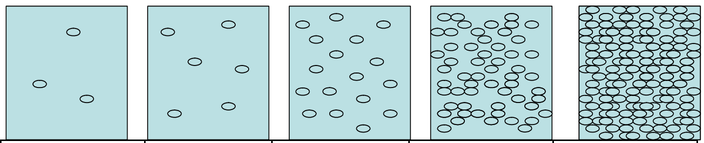
Buffer height (m) *(actual)*: ___________________Buffer length *(overall)*: ____________

Buffer length (m): BROKEN CONTINUOUS CHANGES IN ASPECT

Buffer density *(on ground and at canopy)*:



| 0-20% | 20-40% | 40-60% | 60-80% | 80-100% |
|---|---|---|---|---|



| 0-20% | 20-40% | 40-60% | 60-80% | 80-100% |
|---|---|---|---|---|

Dominant vegetation type: _____________________________________________________

Average plant diameter (mm):________________Sampling method:_____________________

Plant diameter measurements:_______________Mean: ___________________________

| | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | |
| | | | | | | | | | | | |
| | | | | | | | | | | | |

Vegetative element: _________________Unit area:__________________Number per area:___________

Vegetative element:__________________Unit area:__________________Number per area:___________

Vegetative element:__________________Unit area:__________________Number per area:___________

**Landscape**

Landuse: GRASSLAND ARABLE ROUGH GRAZING MIXED PIG FARMING OTHER___________

Crop type: MAIZE WHEAT __________________Width between crop rows (m):________

Direction of cropping: WITH CONTOUR WITH SLOPE OTHER ______________________

**Geomorphology**

| (m/°) | Above buffer | Below buffer | Within buffer |
|---|---|---|---|
| Ave. slope length (overall i.e. field) | | | |
| Ave. slope length (contributing to buffer) | | N/A | N/A |
| Max. slope length (overall i.e. field) | | | |
| Max. slope length (contributing to buffer) | | N/A | N/A |
| Ave. slope angle (overall) | | | |
| Ave. slope angle (contributing to buffer) | | N/A | N/A |
| Max. slope angle (overall) | | | |
| Max. slope angle (contributing to buffer) | | N/A | N/A |
| Slope form (overall i.e. field) | | | |
| Slope form (contributing to buffer) | | N/A | N/A |

Field surface: SMOOTH EVEN SURFACE FEW DEPRESSIONS MANY DEPRESSIONS
Contributing area: SMOOTH EVEN SURFACE FEW DEPRESSIONS MANY DEPRESSIONS
Total number of rills over section of buffer: ____________________________________________

| Observation | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | Ave. |
|---|---|---|---|---|---|---|---|---|---|---|
| Rill width | | | | | | | | | | |
| Observation | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | Ave. |
| Rill depth | | | | | | | | | | |
| Observation | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | Ave. |
| Rill interval | | | | | | | | | | |

**Soil**

| | Sample taken | Field | Sample taken | Buffer |
|---|---|---|---|---|
| Soil texture class (topsoil) | | | | |
| Organic matter content | | | | |
| Soil moisture | | | | |
| Permeability | | | | |
| Bulk density | | | | |
| Soil surface cohesion | | | | |

**Erosion & deposition**

Presence of a sedimentation area: YES NO Dimensions of area (m):_______
If yes: WITHIN BUFFER WITHIN FIELD
ISOLATED LOCALISED AREAS CONSTANT ALONG BUFFER LENGTH
Thickness of sedimentation area:

| Observation | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | Ave. |
|---|---|---|---|---|---|---|---|---|---|---|
| Thickness (cm) | | | | | | | | | | |

Please tick:

| | None | Little | Moderate | Many |
|---|---|---|---|---|
| Channels in buffer | | | | |
| Gaps in buffer vegetation | | | | |
| Ponding | | | | |
| Trafficking | | | | |
| Poaching | | | | |
| In-field soil conservation | | | | |
| Buffer management | | | | |
| Litter in buffer | | | | |
| Concentrated flow | | | | |

Notes:

# Appendix 3

# Photos and schematics of field sites (CD ROM)

CD ROM containing:

- Photos of field sites.
- Site details used in initial site selection.
- Schematics representing transects taken through fields and VFSs.
- Schematics representing mat layout at sites.

# Appendix 4

# Model Description

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# Modified MMF (Morgan–Morgan–Finney) model for evaluating effects of crops and vegetation cover on soil erosion

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## Abstract

**Modifications are made to the revised Morgan–Morgan–Finney erosion prediction model to enable the effects of vegetation cover to be expressed through measurable plant parameters. Given the potential role of vegetation in controlling water pollution by trapping clay particles in the landscape, changes are also made to the way the model deals with sediment deposition and to allow the model to incorporate particle-size selectivity in the processes of erosion, transport and deposition. Vegetation effects are described in relation to percentage canopy cover, percentage ground cover, plant height, effective hydrological depth, density of plant stems and stem diameter. Deposition is modelled through a particle fall number, which takes account of particle settling velocity, flow velocity, flow depth and slope length. The detachment, transport and deposition of soil particles are simulated separately for clay, silt and sand. Average linear sensitivity analysis shows that the revised model behaves rationally. For bare soil conditions soil loss predictions are most sensitive to changes in rainfall and soil parameters, but with a vegetation cover plant parameters become more important than soil parameters. Tests with the model using field measurements under a range of slope, soil and crop covers from Bedfordshire and Cambridgeshire, UK, give good predictions of mean annual soil loss. Regression analysis of predicted against observed values yields an intercept value close to zero and a line slope close to 1·0, with a coefficient of efficiency of 0·81 over a range of values from zero to 38·6 t ha$^{-1}$. Copyright © 2007 John Wiley & Sons, Ltd.**

**Keywords: soil erosion; erosion modelling; vegetation**

## Introduction

The last decade has witnessed considerable interest in the use of vegetation to control the movement of sediment and associated pollutants by runoff over the land surface. Vegetation-based strategies are increasingly preferred to engineering structures such as bunds and terraces on sloping agricultural land and on engineered slopes such as cuttings and embankments. Erosion control is also an important component of land restoration projects involving revegetation of disturbed slopes such as pipeline rights of way. In order to improve design procedures and evaluate the effects of different types of vegetation, erosion models need to be adapted to include a more explicit treatment of vegetation based on measurable parameters of plant density and architecture, which are known to affect their ability to control erosion. Where vegetated buffer strips are used to control the transport of sediment and sediment-laden phosphorus, they are most likely to trap the coarser material but only a proportion of the finer particles. It is therefore important to know the quantity of clay material passing through any buffer and into adjacent water courses.

Whilst detailed process-based models such as WEPP (Nearing *et al.*, 1989b) and EUROSEM (Morgan *et al.*, 1998) can simulate the effects of vegetation on erosion in individual storms, they are often too complex and data-hungry to be used as simple management tools and as screening models. Simpler, empirically based annual models may be more appropriate and give as good, if not better, results (De Roo, 1996; Tiwari *et al.*, 2000). The Morgan–Morgan–Finney model (Morgan *et al.*, 1984; Morgan, 2001) is an annual model that has been used successfully at plot, hillslope and small catchment scales in a wide range of environments including Malaysia (Morgan and Finney, 1982), Indonesia (Besler, 1987), Nepal (Shrestha, 1997), the Rocky Mountains of the USA (Morgan, 1985), Nepal (Morgan, 2001) and Kenya and Tanzania

(Vigiak *et al.*, 2005). The model has been incorporated within a GIS to evaluate erosion in Mediterranean Europe (De Jong, 1994; Paracchini *et al.*, 1997; De Jong *et al.*, 1999). This paper describes modifications made to the model to allow the effects of vegetation and crop cover on erosion to be expressed through measurable properties of plant architecture instead of empirically derived coefficients. The modifications are based on recent research that has improved our understanding of the way different plant properties influence the erosion process. Since one important effect of vegetation is to promote sediment deposition and this is a particle-size selective process, changes are made to the model to provide a more explicit treatment of deposition. The processes of detachment, transport and deposition are simulated separately for clay, silt and sand particles. An important objective of the work was to make these modifications but, at the same time, retain the simplicity of the model and, in particular, its ability to be used with readily available input data. For this reason, the treatment of particle-size selectivity was not extended to soil aggregates because aggregate-size distributions of soil need to be obtained specifically and are not routinely available from national soil survey databases. The modified model is evaluated using data from field measurements of erosion in south-east England.

## Model Operation

When used to predict erosion in catchments, the model simulates the movement of water and sediment over the landscape from source to delivery to the river system. The routing procedure used requires dividing the catchment into elements, each element being reasonably uniform in its soil type, slope and land cover. The user needs to define the pathways taken by water and sediment from one element to another. This requires assigning each element a number (e.g. element 1, element 2 etc) and then determining for that element the number of the element that will deliver runoff and sediment to it and the number of the element into which it will discharge water and sediment. For example, element 3 may receive water and sediment from element 2 and discharge it to element 4. Routing is based on calculations of the water and sediment budgets for each element, taking account of inputs from and outputs to adjacent elements. In this paper, a simple description of the landscape is used, whereby each element receives water and sediment from only one upslope element and discharges to only one downslope element. The model can be applied to more complicated arrangements, whereby discharge is proportioned between two or more downslope elements and individual elements receive inputs from two or more upslope elements, provided the user sets the proportions involved in advance.

Elements may be defined as grid cells on a raster basis or as polygonal discrete landscape units, based on soils, slope and land cover. Figure 1 shows the structure of the model as applied to a hillslope. The input data required for each element to run the model are given in Table I.

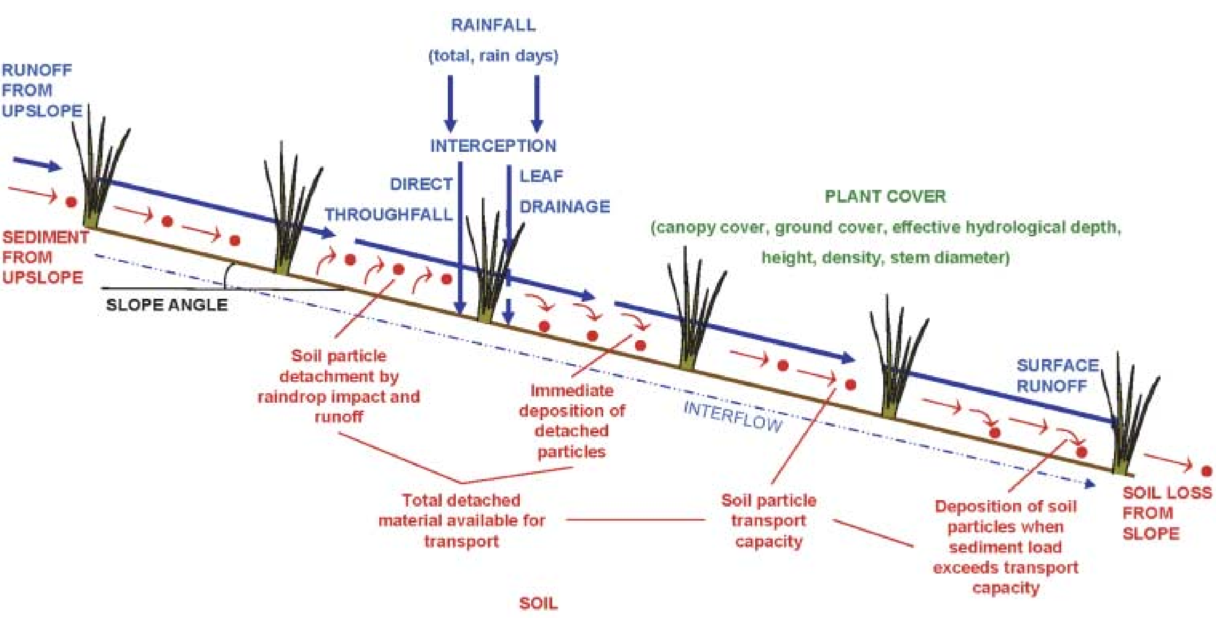


**Figure 1.** Schematic representation of Morgan–Morgan–Finney model (Morgan–Duzant version). This figure is available in colour online at www.interscience.wiley.com/journal/espl



DOI: 10.1002/esp

**Table I.** Input parameters for the MMF model (Morgan–Duzant version)

| Factor | Parameter | Definition |
|---|---|---|
| Climate | $R$ | Mean annual rainfall (mm) |
| | $T$ | Mean annual temperature (°C) |
| | $R_n$ | Mean annual number of rain days |
| | $I$ | Typical intensity of erosive rain (mm $h^{-1}$). Use 10 for temperate climates, 25 for tropical climates and 30 for strongly seasonal climates (e.g. Mediterranean type or monsoon) |
| Soil | %c | Percentage clay |
| | %z | Percentage silt |
| | %s | Percentage sand |
| | ST | Percentage rock fragments on the soil surface |
| | MS | Soil moisture at field capacity (% w/w) |
| | BD | Bulk density of the top soil layer (Mg $m^{-3}$) |
| | EHD | Effective hydrological depth of the soil (m) |
| | RFR | Roughness of the soil surface (cm $m^{-1}$) |
| Slope | $S$ | Slope steepness (°) |
| | $L$ | Slope length (m) |
| | $W$ | Slope width (m) |
| Land cover | PI | Permanent interception expressed as the proportion (between 0 and 1) of rainfall |
| | $E_t/E_0$ | Ratio of actual to potential evapotranspiration |
| | CC | Canopy cover expressed as a proportion (between 0 and 1) of the soil surface protected by the vegetation or crop canopy |
| | GC | Ground cover expressed as a proportion (between 0 and 1) of the soil surface protected by vegetation or crop cover on the ground |
| | PH | Plant height (m), representing the effective height from which raindrops fall from the crop or vegetation cover to the soil surface |
| | $D$ | Average diameter (m) of the individual plant elements (stems, leaves) at the ground surface |
| | NV | Number of plant elements per unit area (number $m^{-2}$) at the ground surface |

The model can be applied to a single slope element or to a number of elements arranged in a logical sequence to reflect the direction of flow of runoff and sediment over the landscape. Each element should be reasonably uniform in its soil type, slope and land cover. It is recommended that element lengths should be between 10 and 50 m.

## The Modified (Morgan–Duzant) Version

Since the modifications to the model include not only new routines but also changes to some of the existing operating equations, the model is described in its entirety in its new version.

### Estimation of rainfall energy

The model starts by taking mean annual rainfall ($R$; mm) and calculates the effective rainfall (Rf; mm) after allowing for permanent interception (PI; proportion between zero and unity) by the vegetation cover. An additional term is added to allow for the effect of slope angle ($S$; degrees) on the quantity of rain received per unit area.

$$\mathrm{Rf} = R(1 - \mathrm{PI})\frac{1}{\cos S} \tag{1}$$

The effective rainfall is divided into direct throughfall (DT; mm), i.e. that reaching the soil surface directly through gaps in the vegetation cover, and leaf drainage (LD; mm), i.e. that reaching the soil surface as flow or drips from the leaves and stems of the vegetation.

Leaf drainage is directly dependent upon the proportion of the effective rainfall intercepted by the canopy cover (CC; proportion between zero and unity):

$$\mathrm{LD} = \mathrm{Rf}\,\mathrm{CC} \tag{2}$$

Direct throughfall then becomes

$$\mathrm{DT} = \mathrm{Rf} - \mathrm{LD} \tag{3}$$

The kinetic energy (J $m^{-2}$) of the direct throughfall is a function of the intensity of the erosive rain ($I$; mm $h^{-1}$) and the amount of direct throughfall. Since the relationship between kinetic energy and intensity varies with the drop-size distribution of the rainfall, different relationships exist for different geographical regions of the world (see Table 3.2 in the work of Morgan (2005)). For the UK, the rainfall is assumed to have a drop-size distribution similar to that measured by Marshall and Palmer (1948). Therefore,

$$\mathrm{KE(DT)} = \mathrm{DT}(8{\cdot}95 + 8{\cdot}44 \log_{10} I) \tag{4}$$

Guide values are used for the intensity of the erosive rain according to geographical location; typically these are 10 mm $h^{-1}$ for temperate climates, 25 for tropical climates and 30 for strongly seasonal climates (e.g. Mediterranean, tropical monsoon). A value of 10 is suitable for the UK.

Since the drop-size distribution of leaf drainage is reasonably uniform, regardless of the type of plant cover (Brandt, 1989), the kinetic energy of the leaf drainage is a function of the plant height (PH; m), which determines the height of fall of the raindrops. Based on the work of Brandt (1990),

$$\text{for PH} < 0{\cdot}15 \quad \mathrm{KE(LD)} = 0 \tag{5}$$

$$\text{for PH} \geq 0{\cdot}15 \quad \mathrm{KE(LD)} = (15{\cdot}8 \times \mathrm{PH}^{0{\cdot}5}) - 5{\cdot}87 \tag{6}$$

Equation (5) is used to prevent the kinetic energy of the leaf drainage from becoming negative when Equation (6) is extrapolated to very low plant heights.

The total kinetic energy of the effective rainfall is

$$\mathrm{KE} = \mathrm{KE(DT)} + \mathrm{KE(LD)} \tag{7}$$

## Estimation of runoff

Based on the work of Kirkby (1976), runoff occurs when the daily rainfall exceeds the soil moisture storage capacity of the soil ($R_c$; mm). The moisture storage capacity depends upon the soil moisture content at field capacity (MS; w/w); the bulk density of the soil (BD; Mg $m^{-3}$); the effective hydrological depth (EHD; m), defined as the depth of soil within which the moisture store controls the generation of runoff, and the loss of water from the soil through evapotranspiration, described by the $E_t/E_0$ ratio of the land cover. In this version of the model an additional term is added to allow for the input of water to the soil by interflow (IF; mm) from the contributing element (CE) upslope. Therefore,

$$R_c = (1000\mathrm{MS} \times \mathrm{BD} \times \mathrm{EHD} \times (E_t/E_0)^{0{\cdot}5}) - \mathrm{IF(CE)} \tag{8}$$

Guide values for MS and BD are given in Table II. Values of EHD can be varied to take account of the different depths of rooting of the vegetation/crop cover and the presence or absence of surface crusting (Table III).

In climates with low intensity precipitation and non-seasonal rainfall regimes, the daily rainfall amounts approximate an exponential frequency distribution and the annual runoff generated on the element (Qe; mm) can be predicted from (Kirkby, 1976)

$$\mathrm{Qe} = \mathrm{Rf}\exp(-R_c/R_0) \tag{9}$$

where $R_0$ is the mean rain per rain day (i.e. $R/R_n$, where $R_n$ is the mean annual number of rain days). Previous experience with the model shows that Equation (9) is reasonably robust and will yield acceptable predictions over a wide range of climatic conditions (Morgan *et al.*, 1984).

The total runoff calculated in Equation (9) is appropriate for slopes about 10 m long (Carson and Kirkby, 1972) and needs to be adjusted for the actual slope length. Failure to do this makes the model very sensitive to the number of elements over which routing of runoff takes place. Without the correction, very different results are obtained if a slope is simulated as a single element compared with dividing the slope into three or more elements. This is because the total runoff on an element is the summation of that generated on the element and that derived from the contributing element upslope $Q$(CE)). The total runoff ($Q$) thus becomes



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**Table II.** Typical values for soil parameters for the MMF model (Morgan–Duzant version)

| Soil type | % clay | % silt | % sand | MS | BD | LP |
|---|---|---|---|---|---|---|
| Sand | 4 | 4 | 92 | 0·08 | 1·5 | 230 |
| Loamy sand | 6 | 11 | 83 | 0·15 | 1·4 | 117 |
| Sandy loam | 10 | 25 | 65 | 0·28 | 1·2 | 50 |
| Loam | 20 | 35 | 45 | 0·20 | 1·3 | 25 |
| Silt | 5 | 89 | 6 | 0·15 | 1·3 | 20 |
| Silt loam | 15 | 66 | 19 | 0·35 | 1·3 | 14 |
| Sandy clay loam | 28 | 14 | 58 | 0·38 | 1·4 | 8 |
| Clay loam | 36 | 35 | 29 | 0·40 | 1·3 | 4 |
| Silty clay loam | 36 | 55 | 7 | 0·42 | 1·3 | 3 |
| Sandy clay | 42 | 5 | 53 | 0·28 | 1·4 | 2 |
| Silty clay | 48 | 45 | 7 | 0·30 | 1·3 | 2 |
| Clay | 64 | 18 | 18 | 0·45 | 1·1 | 1 |

Guide values for % clay, silt and sand are based on mid-point values on the triangular soil texture graph used by USDA.
Values for MS (% w/w) are based on those for clay, clay loam, sandy loam and sand given by Withers and Vipond (1974).
Values for bulk density (Mg $m^{-3}$) are based on values for clay, clay loam, sandy loam and sand given by Hall (1945), with interpolations for other textures. Bulk density values may be increased for crusted or compacted soils by 0·1 and for heavily crusted or heavily compacted soils by 0·2.
Values of LP (m/day) are based on a value of 25 for a loam soil. Other values are calculated relative to this, based on relative differences in saturated hydraulic conductivity as used in the EUROSEM model (Morgan *et al.*, 1993). These values should be treated with caution until they can be verified by further research.

$$Q = (\mathrm{Rf} + Q(\mathrm{CE}))\exp(-R_c/R_0)\left(\frac{L}{10}\right)^{0\cdot 1} \tag{10}$$

The right-hand term in this equation is an empirical adjustment for slope length, which yields reasonable results when comparing the results for slope lengths up to 50 m simulated by a single element with those for the same slope represented by up to five separate elements.

The amount of interflow (IF; mm) generated on the element is simulated as a function of the water balance, which requires information on rainfall ($R$), runoff ($Q$) and evaporation ($E$), the saturated lateral permeability of the soil (LP; m/day) and the slope angle ($S$). In order to calculate annual evapotranspiration by a simple procedure that uses readily available data, the Turc function (Turc, 1961) is used:

$$E = \frac{R}{\sqrt{0\cdot 9 + \dfrac{R^2}{Z^2}}} \tag{11}$$

where

$$Z = 300 + 25T + 0\cdot 05T^2 \tag{12}$$

where $T$ is the mean annual temperature (°C).

The volume of interflow, based on the work of Kirkby (1976), is

$$\mathrm{IF} = \frac{(R - E - Q) \times \mathrm{LP} \times \sin S}{365} \tag{13}$$

Kirkby (1976) set LP at 25 m/day. Assuming this is typical of a loam soil and that values of saturated hydraulic conductivity can be used as indicative of interflow volume, values of LP are estimated for different soil texture classes, based on the relative differences in saturated hydraulic conductivity used in the EUROSEM model (Morgan *et al.*, 1993) (Table II). At present, these values should be considered unverified and should be used with caution.

## Detachment of soil particles

The detachment of soil particles by raindrop impact ($F$; kg $m^{-2}$) is a function of the kinetic energy of the effective rainfall, the detachability of the soil ($K$; J $m^{-2}$) and the stone cover (ST; expressed as a proportion between zero and



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**Table III.** Land cover parameters for the MMF Model (Morgan–Duzant version)

| Cover | EHD | PI | $E_t/E_0$ | CC | GC | PH | *V* | *D* |
|---|---|---|---|---|---|---|---|---|
| Woodland (broad leaved) | 0·20 | 0·20 | 0·95 | 0·98 | 1·0 | 30·0 | 0·6 | 2·0 |
| Woodland (coniferous) | 0·20 | 0·30 | 0·95 | 0·95 | 0·95 | 25·0 | 1·2 | 1·5 |
| Moorland (mainly grass) | 0·12 | 0·30 | 0·90 | 0·98 | 1·0 | 0·5 | 500 | 0·01 |
| Moorland (heather) | 0·12 | 0·20 | 0·90 | 0·75 | 0·30 | 0·5 | 20 | 0·12 |
| Moorland rough grazing (sheep) | 0·12 | 0·30 | 0·90 | 0·95 | 0·8 | 0·2 | 400 | 0·01 |
| Moorland overgrazing (sheep) | 0·11 | 0·25 | 0·86 | 0·60 | 0·50 | 0·4 | 200 | 0·015 |
| Lowland grass (beef/dairy cattle) | 0·12 | 0·30 | 0·86 | 0·90 | 0·6 | 0·1 | 200 | 0·01 |
| Lowland grass (clumpy habitat) (cattle) | 0·12 | 0·30 | 0·86 | 0·80 | 0·5 | 0·1 | 100 | 0·04 |
| Lowland grass (with poaching) (cattle) | 0·11 | 0·25 | 0·86 | 0·80 | 0·6 | 0·1 | 80 | 0·01 |
| Silage (grass cut for hay) | 0·12 | 0·30 | 0·86 | 0·90 | 0·6 | 0·07 | 200 | 0·015 |
| Spring cereals | 0·12 | 0·30 | 0·58 | 0·80 | 0·3 | 1·0 | 200 | 0·04 |
| Winter cereals | 0·12 | 0·40 | 0·60 | 0·80 | 0·3 | 1·5 | 250 | 0·05 |
| Forage crops (field beans) | 0·12 | 0·20 | 0·65 | 0·60 | 0·6 | 1·0 | 20 | 0·02 |
| Orchards | 0·15 | 0·25 | 0·70 | 0·98 | 0·4 | 4·0 | 0·2 | 1·5 |
| Carrot | 0·12 | 0·15 | 0·70 | 0·60 | 0·2 | 0·3 | 100 | 0·01 |
| Parsnip | 0·12 | 0·15 | 0·50 | 0·65 | 0·2 | 0·4 | 12 | 0·02 |
| Cabbage | 0·12 | 0·25 | 0·70 | 0·60 | 0·2 | 0·4 | 3·3 | 0·02 |
| Brussels sprouts | 0·12 | 0·15 | 0·70 | 0·60 | 0·2 | 0·4 | 2·3 | 0·02 |
| Peas | 0·12 | 0·20 | 0·60 | 0·60 | 0·2 | 0·9 | 80 | 0·01 |
| Onions | 0·10 | 0·10 | 0·55 | 0·30 | 0·2 | 0·3 | 80 | 0·02 |
| Maize | 0·12 | 0·25 | 0·68 | 0·65 | 0·5 | 2·0 | 10 | 0·05 |
| Potatoes | 0·12 | 0·12 | 0·75 | 0·80 | 0·4 | 0·8 | 4·5 | 0·10 |
| Sugar beet | 0·12 | 0·15 | 0·75 | 0·60 | 0·4 | 0·5 | 7·5 | 0·15 |
| Oil seed rape | 0·12 | 0·40 | 0·80 | 0·70 | 0·4 | 1·0 | 100 | 0·02 |
| Linseed | 0·12 | 0·40 | 0·35 | 0·70 | 0·4 | 0·7 | 700 | 0·01 |
| Vineyards (with grass) | 0·12 | 0·25 | 0·30 | 0·80 | 0·80 | 1·5 | 0·5 | 0·15 |
| Bare soil (no crust) | 0·09 | 0·00 | 0·05 | 0·0 | 0·0 | 0·0 | 0·0 | 0·0 |
| Bare crusted soil | 0·05 | 0·00 | 0·05 | 0·0 | 0·0 | 0·0 | 0·0 | 0·0 |

These guide values represent approximate average values for the growing season, in the case of crops, or for the year, in the case of non-seasonal land cover. Wherever field data are available, measured values should be used instead.

Where terracing is used, add 0·01 to EHD to take account of the increase in water storage.

Values for NV are taken from the work of Harper (1983), Salunkhe and Kadam (1998) and Wiseman *et al.* (1993).

Values for $E_t/E_0$ are taken from the work of Withers and Vipond (1974). Where no values are given in this source, the values proposed are those for a closely analogous land cover.

Values for *D* are taken largely from photographs. Since they refer to the base of the plant at the soil surface, they relate, for example, to the 'clump' of a grass rather than to the diameter of the individual tillers or stems. This may be up to an order of magnitude greater than the diameter of a single stem.

Parameter values for sheep, cattle and pigs are estimates for the grass cover prevailing under different levels of management. Although variations occur on sward height for improved (lowland) grass between sheep (usually 0·04–0·06 m high) and cattle (usually 0·06–0·10 m high), the model does not take account of this, because when the height is less than 0·15 m the kinetic energy of the leaf drainage is taken as zero. For overgrazed upland soils under sheep and for poached soils under cattle, EHD has been reduced by 0·01 to reflect lower soil moisture storage on the more compacted soil. A greater reduction is not proposed because of the offsetting effects of greater roughness and depression storage of the surface soil.

unity). The latter will act like a mulch and protect the soil from detachment. In order to allow for the particle-size distribution of the soil, the effective rainfall is proportioned according to the proportion of clay (c), silt (z) and sand (s) particles in the soil. The detachment of clay, silt and sand is calculated respectively as follows:

$$F_c = K_c \times \%c/100 \times (1 - ST) \times KE \times 10^{-3} \quad (14)$$

$$F_z = K_z \times \%z/100 \times (1 - ST) \times KE \times 10^{-3} \quad (15)$$

$$F_s = K_s \times \%s/100 \times (1 - ST) \times KE \times 10^{-3} \quad (16)$$

Based on data from Quansah (1982), initial values of $K_c$, $K_z$ and $K_s$ are taken respectively as 0·1, 0·5 and 0·3 g J$^{-1}$. The value of $K_c$ should be treated with care since, as shown by Poesen (1985) and Chisci *et al.* (1989), the detachability of clay particles has a very high variability depending on the aggregate stability of the soil and the type of clay, both of which influence soil cohesion. Laboratory experiments with rainfall simulation could be carried out on the separate clay, silt and sand fractions of a specific soil to obtain measured values. Detachability is also likely to be reduced by the additional cohesion given to the soil by plant roots.

Total detachment by raindrop impact is

$$F = F_c + F_z + F_s \tag{17}$$

The detachment of soil particles ($H$; kg m$^{-2}$) by runoff is a function of the volume of runoff ($Q$), the detachability of the soil by runoff (DR; g mm$^{-1}$), the slope angle ($S$) and the proportion of the soil covered by vegetation (GC) and stones (ST). Proportioning the effect of the runoff by particle size gives

$$H_c = \mathrm{DR}_c \times \%\mathrm{c}/100 \times Q^{1\cdot5} \times (1 - (\mathrm{GC} + \mathrm{ST})) \times \sin^{0\cdot3} S \times 10^{-3} \tag{18}$$

$$H_z = \mathrm{DR}_z \times \%\mathrm{z}/100 \times Q^{1\cdot5} \times (1 - (\mathrm{GC} + \mathrm{ST})) \times \sin^{0\cdot3} S \times 10^{-3} \tag{19}$$

$$H_s = \mathrm{DR}_s \times \%\mathrm{s}/100 \times Q^{1\cdot5} \times (1 - (\mathrm{GC} + \mathrm{ST})) \times \sin^{0\cdot3} S \times 10^{-3} \tag{20}$$

and

$$H = H_c + H_z + H_s \tag{21}$$

Values for DR are 1·0, 1·6 and 1·5 for the clay (c), silt (z) and sand (s) fractions respectively, based on data from laboratory experiments carried out by Quansah (1982). As with the values for $K$, they should be treated as provisional and open to replacement if the model user has measured data for specific soils.

## Immediate deposition of detached particles

Only a proportion of the detached sediment will be delivered to the runoff for transport; the remainder will be deposited close to the point of detachment. Deposition at this point can be considered as the first phase of the depositional process. The amount of detached soil delivered to the flow ($G$) is therefore the result of a balance between the amount of soil detached by raindrop impact ($F$), the amount of soil detached by the runoff ($H$) and the amount of detached soil which is deposited (DEP).

The deposition of soil particles is modelled as a function of the probability that a detached particle will fall to the soil surface rather than be entrained in the runoff. This probability is related to a particle fall number ($N_f$; Tollner *et al.*, 1976), which depends upon the length of the element ($l$), the fall velocity of the particles ($v_s$), the flow velocity ($v$) and the flow depth ($d$). It is therefore necessary first to calculate the velocity of flow.

Flow velocity calculations are made for four possible conditions:

(a) a standard condition related to unchannelled overland flow over a smooth bare soil ($v_b$);
(b) the actual condition, which can take account of whether or not the flow has become channelled into rills ($v_a$);
(c) a condition taking account of the effects of the vegetation cover ($v_v$) and
(d) a condition taking account of the roughness of the soil surface, for example that resulting from tillage ($v_t$).

Calculations are made only for those conditions that exist for the study area where the model is being applied. Thus, if there is no vegetation or crop cover, $v_v$ is not calculated. For grassland, rough pasture or trees, a value of $v_t$ can be calculated, based on the roughness of the soil surface, even though the land is not tilled.

(a) For the standard bare soil condition, flow velocity is estimated from the Manning equation:

$$v_b = 1/n\, d^{0\cdot67} S^{0\cdot5} \tag{22}$$

where $n$ is Manning's roughness coefficient and $d$ is the flow depth (m). Values of $n = 0{\cdot}015$ and $d = 0{\cdot}005$ are recommended.



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(b) For the actual flow velocity ($v_a$), the same equation is used, with $d = 0{\cdot}005$ for unchannelled flow, $d = 0{\cdot}01$ for shallow rills and $d = 0{\cdot}25$ for deeper rills. An additional term is added to account for the effects of the stone cover (ST) (Poesen, 1992). The rill depths may be based on field measurements of what typically occurs. The equation used is

$$v_a = 1/n\, d^{0{\cdot}67}\, S^{0{\cdot}5}\, e^{-0{\cdot}018ST} \tag{23}$$

(c) For vegetated conditions, flow velocity is estimated as a function of slope and the density of the vegetation; the latter depends upon the diameter of the plant stems ($D$) and the number of stems per unit area (NV) (Jin *et al.*, 2000):

$$v_v = \left(\frac{2g}{D\,\mathrm{NV}}\right)^{0{\cdot}5} s^{0{\cdot}5} \tag{24}$$

(d) For the effect of tillage, $v_t$ is calculated from the Manning equation using $d = 0{\cdot}005$ and a value of $n$ dependent upon the surface roughness (cm cm$^{-1}$) obtained by the tillage implement and defined by the index $(L_a - L_0)/L_a$, where $L_0$ is the straight-line distance between two points along a transect over the soil surface and $L_a$ is the actual distance between the points measured over all the microtopographic irregularities. This is the roughness term used in the European Soil Erosion Model (Morgan *et al.*, 1998). The following procedure was used to obtain an appropriate equation.

Gilley and Fincker (1991) showed a relationship from experiments between $n$ and the random roughness (RR) of the soil surface:

$$n = \frac{0{\cdot}172\mathrm{RR}^{0{\cdot}742}}{\mathrm{RE}^{0{\cdot}282}} \tag{25}$$

where RR is the standard deviation of a series of surface height measurements and RE is the Reynolds number of the flow. Using this equation, values of $n$ were calculated for various values of RR for RE = 100 000, a typical value for combined unchannelled and rill flow over the landscape (Grayson and Moore, 1992). The values of RR were then converted into values of RFR using the relationship developed by Auerswald (personal communication):

$$\mathrm{RFR} = -1{\cdot}77 + 9{\cdot}25 \ln \mathrm{RR} \tag{26}$$

Best-fit regression analysis for the values of $n$ and RFR yielded the following equation ($r^2 = 0{\cdot}9999$) for use in the model:

$$\ln n = -2{\cdot}1132 + 0{\cdot}0349\mathrm{RFR} \tag{27}$$

Guide values for RFR are given in Table IV.

**Table IV.** Values of surface roughness parameter (RFR)

| **Implement** | **Roughness (RFR) (cm m$^{-1}$)** |
|---|---|
| Mould-board plough | 33–48 |
| Chisel plough | 17–26 |
| Cultivator | 6–15 |
| Tandem disc | 18–26 |
| Offset disc | 38–51 |
| Paraplough | 10 |
| Spike tooth harrow | 8–15 |
| Spring tooth harrow | 18 |
| Rotary hoe | 12–13 |
| Rototiller | 15 |
| Drill | 10–12 |
| Row planter | 5–13 |

Values taken from the work of Morgan (2005).



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The particle fall number ($N_f$) is determined separately for each soil particle class from

$$N_f(c) = \frac{l\,v_s(c)}{v\,d} \tag{28}$$

$$N_f(z) = \frac{l\,v_s(z)}{v\,d} \tag{29}$$

$$N_f(s) = \frac{l\,v_s(s)}{v\,d} \tag{30}$$

where $v$ is that for either the bare soil or vegetated condition and $d$ is the depth of flow, taking the value used in Equation (23). Fall velocities are estimated from

$$v_s = \frac{1/18 \times \delta^2(\rho_s - \rho)g}{\eta} \tag{31}$$

where $\delta$ is the diameter of the particle, $\rho_s$ is the sediment density (= 2650 kg m$^{-3}$), $\rho$ is the flow density (typically 1100 kg m$^{-3}$ for runoff on hillslopes (Abrahams *et al.*, 2001)), $g$ is gravitational acceleration (taken as 9·81 m s$^{-2}$) and $\eta$ is the fluid viscosity (nominally 0·001 kg m$^{-1}$ s$^{-1}$ but taken as 0·0015 to allow for the effects of the sediment in the flow). Applied to particle sizes of 0·000 002 for clay, 0·000 06 for silt and 0·0002 for sand, this gives respective $v_s$ values of 0·000 002 m s$^{-1}$ for clay, 0·002 m s$^{-1}$ for silt and 0·02 m s$^{-1}$ for sand.

The percentage of the detached sediment that is deposited is estimated from the relationship obtained by Tollner *et al.* (1976) but calculated separately for each particle size:

$$\mathrm{DEP}(c) = 44{\cdot}1(N_f(c))^{0{\cdot}29} \tag{32}$$

$$\mathrm{DEP}(z) = 44{\cdot}1(N_f(z))^{0{\cdot}29} \tag{33}$$

$$\mathrm{DEP}(s) = 44{\cdot}1(N_f(s))^{0{\cdot}29} \tag{34}$$

These equations can yield values of DEP > 100, but since it is physically not possible to deposit more material than has been detached, a maximum value is set at DEP = 100 when this occurs.

## Delivery of detached particles to runoff

The number of detached particles ($G$) going into transport in the flow is calculated separately for clay, silt and sand, taking account of detachment by raindrop impact ($F$) and detachment by runoff ($H$) on the element, deposition of the detached material (DEP) on the element and the input of material in the runoff from the contributing element upslope (SL(CE)):

$$G(c) = (F(c) + H(c)) \times (1 - (\mathrm{DEP}(c)/100)) + (\mathrm{SL(CE)}(c) \times W(\mathrm{CE})/W) \tag{35}$$

$$G(z) = (F(z) + H(z)) \times (1 - (\mathrm{DEP}(z)/100)) + (\mathrm{SL(CE)}(z) \times W(\mathrm{CE})/W) \tag{36}$$

$$G(s) = (F(s) + H(s)) \times (1 - (\mathrm{DEP}(s)/100)) + (\mathrm{SL(CE)}(s) \times W(\mathrm{CE})/W) \tag{37}$$

where $W$(CE) is the width of the upslope contributing element and $W$ is the width of the element in question. Adjustment by the ratio $W$(CE)/$W$ is necessary because, unless the routing is based on grid cells of a regular size, the widths of the two elements are unlikely to be the same and runoff and sediment will be either spread out or concentrated, depending on whether the receiving element is wider or narrower than the contributing element.

The total detached material going into transport in the flow is then

$$G = G(c) + G(z) + G(s) \tag{38}$$

## Transport capacity of the runoff

The transport capacity of the runoff (TC; kg m$^{-2}$) is determined separately for the clay, silt and sand components as a function of the volume of runoff on the element, the slope steepness and the effect of the plant cover. In the previous



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version of the MMF model, the plant cover effect was allowed for by using the *C*-factor value of the universal soil loss equation. In this version, this is replaced by a new term, defined as the ratio of the flow velocities with the actual conditions of rills (if any), vegetation and surface roughness to the flow velocity in the standard condition. This approach reflects the concept that transport capacity depends on the product of velocity and slope. Thus

$$\mathrm{TC(c)} = ((v_a v_v v_t)/v_b)\ (\%\mathrm{c}/100)\ Q^2 \sin S\ 10^{-3} \tag{39}$$

$$\mathrm{TC(z)} = ((v_a v_v v_t)/v_b)\ (\%\mathrm{z}/100)\ Q^2 \sin S\ 10^{-3} \tag{40}$$

$$\mathrm{TC(s)} = ((v_a v_v v_t)/v_b)\ (\%\mathrm{s}/100)\ Q^2 \sin S\ 10^{-3} \tag{41}$$

Where there is no vegetation or crop cover, $v_v$ is omitted. Where the land is not under arable cultivation, $v_t$ can be omitted unless the roughness of the 'natural' soil surface is accounted for.

## Sediment balance

The transport capacity (TC) is now compared with the amount of material available for transport (*G*). Two conditions are possible.

(1) If TC is greater than or equal to *G*, all of *G* is transported from the element and the soil loss (SL) from the element equals *G*.

$$\text{If } \mathrm{TC(c)} \geq G(\mathrm{c}), \quad SL(\mathrm{c}) = G(\mathrm{c}) \tag{42}$$

$$\text{If } \mathrm{TC(z)} \geq G(\mathrm{z}), \quad SL(\mathrm{z}) = G(\mathrm{z}) \tag{43}$$

$$\text{If } \mathrm{TC(s)} \geq G(\mathrm{s}), \quad SL(\mathrm{s}) = G(\mathrm{s}) \tag{44}$$

(2) If the transport capacity (TC) is less than *G*, material will need to be deposited from *G* until TC = *G*. In many process-based erosion models, deposition of sediment from runoff is described as a function of the settling velocity of the particles and difference between the actual sediment concentration in the flow and the concentration that can be carried at transport capacity. Whilst this approach is feasible for modelling over very short time periods, it is difficult to conceptualize and apply in terms of an annual model. Further, sediment concentration may not be so relevant when describing deposition rates averaged over longer time periods. Studies of sedimentation in farm ponds by Verstraeten and Poesen (2001) showed that, whilst deposition in individual events was related to the sediment concentration of the inflow, long-term deposition rates were independent of sediment concentration. They were, however, related to a parameter incorporating particle settling velocity, length of the pond, inflow velocity and settling depth, namely the terms included in the particle fall number ($N_f$) described in Equations (28)–(30). When applying these equations to deposition from runoff on hillslopes, the settling velocities used earlier to describe deposition of particles immediately following their detachment by raindrop impact and runoff need to be modified because the effective settling velocities for mixed particle sizes falling out of runoff in a depositional environment are often much higher than those in sediment-free water (Zaneveld *et al.*, 1982; Lovell and Rose, 1991; Rose *et al.*, 2003). They approach the maximum value of the distribution of settling velocities for the particle size distribution of a given soil (Misra and Rose, 1991). A simple increase in value of the settling velocities by an order of magnitude is used here to bring them in line with those obtained experimentally for multi-particle sediments (Lovell and Rose, 1988; Hogarth *et al.*, 2004).

Deposition of excess sediment from the flow is therefore calculated by determining the particle fall number ($N_f$) from Equations (28)–(30) using particle settling velocities of 0·000 02 m s$^{-1}$ for clay, 0·02 m s$^{-1}$ for silt and 0·2 m s$^{-1}$ for sand; calculating deposition (DEP) from Equations (32)–(34); and applying the following equations to determine the sediment balance:

$$\begin{gathered}\text{If } \mathrm{TC(c)} < G(\mathrm{c}), \text{ calculate } G(\mathrm{c1}) = G(\mathrm{c})\ (1 - (\%\mathrm{DEP(c)}/100)) \\ \text{If } \mathrm{TC(c)} \geq G(\mathrm{c1}),\ \mathrm{SL(c)} = \mathrm{TC(c)}; \text{ if } \mathrm{TC(c)} < G(\mathrm{c1}),\ \mathrm{SL(c)} = G(\mathrm{c1})\end{gathered} \tag{45}$$

$$\begin{gathered}\text{If } \mathrm{TC(z)} < G(\mathrm{z}), \text{ calculate } G(\mathrm{z1}) = G(\mathrm{z})\ (1 - (\%\mathrm{DEP(z)}/100)) \\ \text{If } \mathrm{TC(z)} \geq G(\mathrm{z1}),\ \mathrm{SL(z)} = \mathrm{TC(z)}; \text{ if } \mathrm{TC(z)} < G(\mathrm{z1}),\ \mathrm{SL(z)} = G(\mathrm{z1})\end{gathered} \tag{46}$$

$$\begin{gathered}\text{If } \mathrm{TC(s)} < G(\mathrm{s}), \text{ calculate } G(\mathrm{s1}) = G(\mathrm{s})\ (1 - (\%\mathrm{DEP(s)}/100)) \\ \text{If } \mathrm{TC(s)} \geq G(\mathrm{s1}),\ \mathrm{SL(s)} = \mathrm{TC(s)}; \text{ if } \mathrm{TC(s)} < G(\mathrm{s1}),\ \mathrm{SL(s)} = G(\mathrm{s1})\end{gathered} \tag{47}$$



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The mean annual soil loss (SL; kg m$^{-2}$) from the element is

$$SL = SL(c) + SL(z) + SL(s) \quad (48)$$

## Sensitivity Analysis

A sensitivity analysis was carried out to test that the model behaved rationally and to determine which input parameters had most effect on the predictions of runoff and soil loss. Sensitivity was analysed using the average linear sensitivity (ALS) approach (Nearing *et al.*, 1989a), which expresses a relative normalized change in output to a normalized change in input:

$$ALS = \frac{\left[\dfrac{O_2 - O_1}{\bar{O}}\right]}{\left[\dfrac{I_2 - I_1}{\bar{I}}\right]} \quad (49)$$

where $O_1$ and $O_2$ are the values of the model output obtained with the values of $I_1$ and $I_2$ for input parameter $I$ and $\bar{I}$ and $\bar{O}$ are the means of the two input and two output values respectively. This approach is appropriate for comparing the sensitivities of input parameters with values of different orders of magnitude. Although it does not deal well with sensitivity where the output of the model is related non-linearly to an input, this issue can be addressed by examining how the value of ALS changes as the input is varied over small ranges.

The calculation of ALS relies upon the value of all other parameters being held constant while the value of the parameter under examination is changed. Generally, base values are selected to represent the mid-point of the range likely to be encountered in the field. Different approaches have been adopted to select the range of values over which a parameter is varied. Favis-Mortlock and Smith (1990) proposed varying the base value by ±10%, whereas McCuen and Snyder (1986) considered it more important to examine extreme values. A more practical approach is to vary the values over the ranges likely to occur in the field (Thomas and Beasley, 1986; Heatwole *et al.*, 1987; Nearing *et al.*, 1989a) and this is the procedure adopted here (Table V). The analysis is carried out for bare soil and vegetated

**Table V.** Values used in the average linear sensitivity analysis

| Parameter | Low value | Base value | High value |
|---|---|---|---|
| Mean annual rainfall (mm) | 500 | 1500 | 2500 |
| Mean annual temperature (°C) | 5 | 15 | 25 |
| Rainfall intensity (mm h$^{-1}$) | 15 | 45 | 75 |
| Mean annual number of rain days | 50 | 150 | 250 |
| Soil moisture (% w/w) | 0·05 | 0·25 | 0·45 |
| Bulk density (Mg m$^{-3}$) | 0·8 | 1·2 | 1·6 |
| Effective hydrological depth (m) | 0·05 | 0·11 | 0·17 |
| Slope angle (°) | 2 | 10 | 18 |
| Slope length (m) | 5 | 15 | 25 |
| Slope width (m) | 0·5 | 3·5 | 6·5 |
| Soil surface roughness (cm m$^{-1}$) | 5 | 25 | 45 |
| Permanent interception (proportion) | 0·05 | 0·5 | 1·0 |
| $E_t/E_0$ ratio | 0·1 | 0·5 | 0·9 |
| Canopy cover (proportion) | 0 | 0·5 | 1 |
| Ground cover (proportion) | 0 | 0·5 | 1 |
| Plant height (m) | 0 | 5 | 30 |
| Plant stem diameter (m) | 0·000 01 | 0·5 | 3 |
| Plant stem density (number m$^{-2}$) | 0·000 01 | 500 | 2000 |
| Detachability of clay particles by rain (RDC) | 0·1 | 0·7 | 1·5 |
| Detachability of silt particles by rain (RDZ) | 0·5 | 2·8 | 5·15 |
| Detachability of sand particles by rain (RDS) | 0·15 | 2·0 | 4·15 |
| Detachability of clay particles by runoff (ODC) | 0·02 | 0·2 | 2·0 |
| Detachability of silt particles by runoff (ODZ) | 0·016 | 0·16 | 1·6 |
| Detachability of sand particles by runoff (ODS) | 0·015 | 0·15 | 1·5 |
| Manning's *n* (Mn) | 0·01 | 0·02 | 0·03 |
| Flow depth (m) (FD) | 0·005 | 0·1 | 0·3 |

Detachability values in g J$^{-1}$ for rainfall and g mm$^{-1}$ for runoff.



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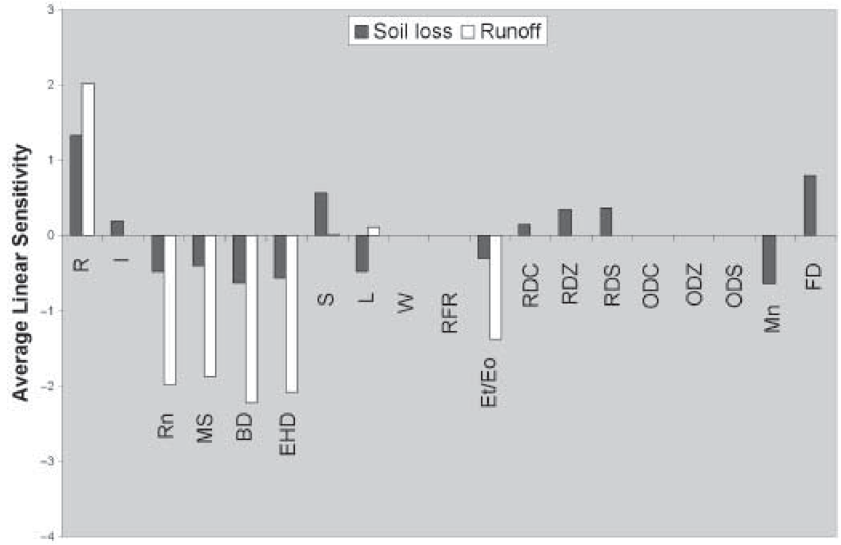


**Figure 2.** Values of average linear sensitivity for selected input parameters for bare soil conditions.

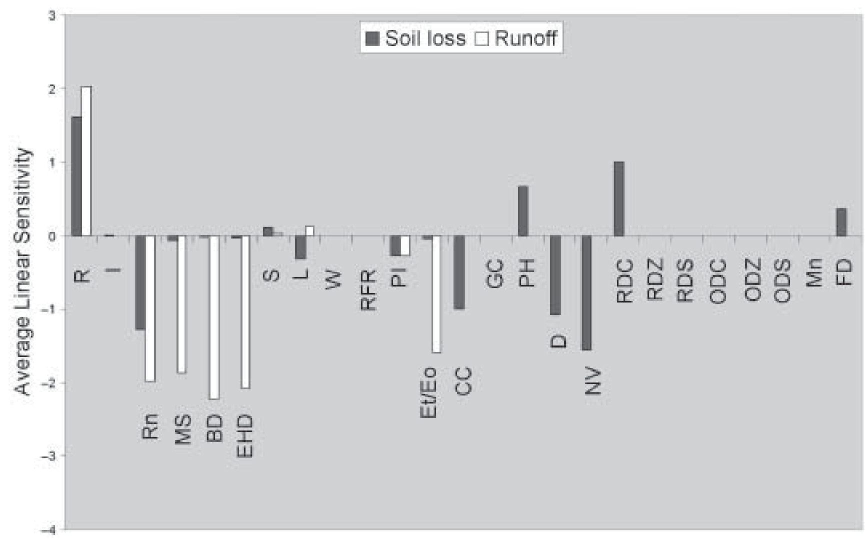


**Figure 3.** Values of average linear sensitivity for selected input parameters for vegetated conditions.

conditions in order to investigate whether the inclusion of a plant cover has an effect on the relative importance of the input parameters. The analysis was conducted for a single slope element, which means that the effects of routing runoff, interflow and sediment over the landscape were not examined.

Figures 2 and 3 show the results of the sensitivity analysis. For bare soil, runoff is highly sensitive (ALS > 1·0) to the parameters related to rainfall, evapotranspiration and soil moisture storage. Soil loss is highly sensitive to rainfall amount and moderately sensitive (ALS ≥ 0·5 < 1·0) to soil moisture parameter, through its effect on runoff, and to slope angle, Manning's *n* and flow depth. For vegetated conditions, the sensitivity of runoff remains the same but soil loss is highly sensitive to rainfall, the number of rain days and the diameter and density of the plant stems; it is moderately sensitive to canopy cover, plant height and the detachability of the clay particles. These results are broadly as expected in that they are rational and conform to previous sensitivity analysis of the MMF model (Morgan *et al.*, 1984).

## Model Application

The model was applied to an area of Bedfordshire and Cambridgeshire, UK, where runoff, soil particle detachment and soil loss were measured over a range of soil and vegetated types between May 1973 and July 1980 (Morgan *et al.*, 1987). Measurements were made with 0·5 m wide Gerlach troughs, which were placed in pairs at three different positions along convexo-concave slopes with maximum angles between 7 and 11° on arable land and 20° under woodland. Soils ranged from sands to sandy loams and clay loams. Land covers included bare soil, grass ley, winter-sown and spring-sown cereals and mixed coniferous and deciduous woodland. Mean monthly temperatures range from 3 °C in January to 16 °C in July. Mean annual rainfall is 555 mm, evenly distributed throughout the year. The mean annual number of rain days is 150. Mean annual runoff, soil particle detachment and soil loss values were calculated from the measured data.

Measured values were used for the input parameters where available; otherwise guide values (Tables II and III) were used. Adjustments were then made to a small number of the input parameters identified above as sensitive in order to improve the predictions. These were made only where they could be kept within the range of likely field values and where the change could be justified by knowledge of the local conditions. No attempt was made to calibrate the model, partly because of the limited data available but also because in models of this type many different combinations of parameters can give acceptable fits with measured values, which renders calibration a meaningless exercise. Further, as indicated by De Roo (1996), calibration of a process-based model generally compromises its physical base. For the bare soil site at Silsoe, the value of MS was increased from 0·08 to 0·15 and that of EHD from 0·05 to 0·12 in order to reduce the value of predicted runoff. The change in MS was within the likely range encountered in field soils and that in EHD was made to reflect the effects of the management practice of repeated rotovations of the soil in order to remove any vegetation and maintain the bare condition. It was therefore reasonable to assume that EHD matched that of a tilled soil rather than a compacted bare soil. This change had no effect on the predicted soil loss, because this remained limited by detachment rate rather than transport capacity. For the sites under arable, NV was increased to 250 for spring cereals and 300 for winter cereals, both reflecting that the farmers might have used a higher planting density than the guide value. For the woodland site, a value of 5000 was chosen for NV to take account of the high density of different plant elements (grasses, bracken) at ground level, and a value of 1·5 was chosen for $D$ to reflect the diameter of plant element with the dominant effect on flow, namely the trees. Since all the slopes in the study area were broadly convexo-concave in form, simulations were carried out by dividing the slope into upper, middle and lower portions and routing runoff and sediment from top to bottom of the slope over the three elements.

The results of the model simulations (Table VI) show that the model generally predicts soil loss reasonably well. The runoff predictions are less satisfactory. Despite the adjustments to the input parameter values, runoff was substantially overpredicted for the Silsoe bare soil site. In contrast, it was underpredicted for the clay soils under arable. Soil loss predictions were closer to the measured values, although there was a tendency to underprediction. These results indicate that for clay soils it is important to improve the value used for the detachability of clay particles by raindrop impact. More research is needed to enable appropriate values to be determined taking account of aggregate stability.

The linear fit between predicted ($P$) and observed ($O$) values was expressed using the reduced major axis line (Kermack and Haldane, 1950; Till, 1973), which allows for likely variability in the observed values due to measurement errors. The resulting relationships are for runoff

$$P = -7{\cdot}0064 + 2{\cdot}1015O \tag{49}$$

and for soil loss

$$P = -0{\cdot}1953 + 0{\cdot}9617O \tag{50}$$

Respective values of $r^2$ are 0·9873 and 0·8973. The results for soil loss show an intercept value close to zero and a line slope close to 1·0, which indicates an acceptable relationship between the measured values and model output. The results for runoff are poorer, despite the higher value of the coefficient of determination. As noted above, for this data set, the model overpredicts the runoff for the bare soil conditions and underpredicts the clay soils under cereals.

The performance of the model can also be indicated by values of the efficiency coefficient (EC) of Nash and Sutcliffe (1970):

$$\mathrm{EC} = 1 - \frac{\sum (X_{\mathrm{obs}} - X_{\mathrm{pred}})^2}{\sum (X_{\mathrm{obs}} - X_{\mathrm{obs}})^2} \tag{51}$$



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**Table VI.** Comparison of observed and predicted values for mean annual runoff and soil loss for sites in Bedfordshire and Cambridgeshire, England

| | | Runoff (mm) | | Soil loss (t/ha) | |
|---|---|---|---|---|---|
| **Site** | **Slope (°)** | **Observed** | **Predicted** | **Observed** | **Predicted** |
| Silsoe: sandy soil; bare | 9 | 58 | 110 | 11·29 | 0·84 |
| | 11 | 67 | 132 | 38·58 | 31·53 |
| | 9 | 63 | 135 | 13·63 | 25·39 |
| Silsoe: sandy soil; ley grass | 9 | 11 | 11 | 0·80 | 0·18 |
| | 9 | 17 | 11 | 2·27 | 0·34 |
| | 5 | 7 | 11 | 0·55 | 0·46 |
| Woburn: sandy loam soil; winter-sown cereals | 6 | 13 | 16 | 0·70 | 0·40 |
| | 7 | 11 | 17 | 0·60 | 2·18 |
| | 4 | 7 | 17 | 0·90 | 1·07 |
| Meppershall: clay soil; winter- and spring-sown cereals | 10 | 6 | 0 | 0·50 | 0·84 |
| | 6 | 9 | 0 | 0·37 | 0·17 |
| Pulloxhill: clay soil; spring-sown cereals | 10 | 1 | 0 | 0·69 | 0·63 |
| | 7 | 0 | 0 | 0·30 | 0·84 |
| Ashwell: chalky soil; winter-sown cereals | 10 | 5 | 3 | 0·70 | 0·56 |
| | 5 | 2 | 3 | 0·65 | 0·74 |
| Maulden: sandy loam soil; mixed coniferous and deciduous woodland | 15 | 2 | 3 | 0·00 | 0·01 |
| | 16 | 5 | 3 | 0·01 | 0·02 |
| | 20 | 2 | 3 | 0·00 | 0·03 |

Values of observed runoff and soil loss are based on data collected between the following dates: Silsoe – 2 May 1973 to 16 August 1979; Woburn – 8 March 1977 to 21 March 1979; Meppershall – 10 February 1977 to 26 April 1979; Pulloxhill – 9 December 1976 to 17 April 1979; Ashwell – 3 February 1977 to 2 August 1979; and Maulden – 2 May 1973 to 16 August 1979.

Values for EC are –0·51 for runoff and 0·81 for soil loss. The value for soil loss indicates that the model performs well. Values greater than 0·5 are generally interpreted to mean that a model performs satisfactorily (Quinton, 1997). Whilst, in theory, EC can equal 1·0, in practice values rarely exceed 0·7 (Nearing, 1998). The negative value of EC indicates that the model predictions have a higher variation than the observed values. If the bare soil slope at Silsoe is excluded from the calculations, the value of EC for runoff becomes 0·03, an improvement but still an unsatisfactory outcome.

Since the model treats the clay, silt and sand fractions of the soil separately, its output includes predictions of soil loss for each fraction as well as the total. From this the percentage of each fraction in the material removed from each element of the hillslope can be calculated. For the sites at Ashwell, Maulden, Meppershall, Pulloxhill and the site at Silsoe under grass, 99·9% of the material leaving the bottom of the slope is predicted as clay. This compares with the clay contents of the respective soils of 30, 10, 41, 45 and 8% of soils. In contrast, for the sandy loam soil under cereals at Woburn, the sediment discharging from the slope is predicted to comprise 11% clay, 10% silt and 79% sand, compared with 10, 5 and 85% respectively in the original soil. That at Silsoe on the bare soil comprises 7% clay, 13% silt and 80% sand; the original soil contained 8% clay, 9% silt and 83% sand. Unfortunately, data on the particle-size distribution of the eroded material were not collected when the field measurements were made, so it is not possible to validate this output. The above results are therefore presented only as an indicator of the type of model output that can be obtained. If sub-routines were added linking particulate-bound phosphorus to the clay content of the soil, the model would therefore have the potential for evaluating the transport of sediment-laden phosphorus from hillslopes to watercourses. Since the model expresses the effects of vegetation cover explicitly, it also has the potential for assessing the effects of different vegetation strategies, such as live buffer strips, in controlling sediment movement and, by implication, phosphorus transfer.

## Discussion

The modifications made to the Morgan–Morgan–Finney model show that it is feasible to represent crops and vegetation cover in relatively simple, annual erosion models by explicit measurable parameters of the plant architecture:



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namely plant height, canopy cover, ground cover, stem diameter and stem density. Output from the sensitivity analysis shows that under bare soil conditions erosion is related to rainfall, soil and slope conditions, but that when a plant cover exists it outweighs soil and slope in importance. Although few studies exist of the interactions between the various factors influencing soil erosion, the result accords with field experience that vegetation-based strategies are effective ways of reducing erosion over all types of soil. It is also in line with previous studies that show that soil loss is less sensitive to slope in the presence of a plant or mulch cover (Lal, 1976; Quinn *et al.*, 1980).

When output of the model is compared to measured data for soil loss, the values obtained for the correlation coefficient and the efficiency coefficient are similar to those achieved for previous versions of the model (Morgan *et al.*, 1984; Morgan, 2001). The poor performance in predicting runoff also matches that found with the previous versions, which, although generally satisfactory when tested against world-wide data sets, gave poor predictions for the field sites in southeast England used in this study. One possible reason for the discrepancy between observed and predicted values is that the measured values are in error. The Gerlach troughs used to collect runoff and sediment often overflowed during large events, which meant that an unknown but probably substantial under-recording of runoff occurred. The ability of the model to yield reasonable predictions of erosion on these sites whilst underpredicting the runoff is easily explained by the soil loss on the bare soil being detachment limited (Morgan *et al.*, 1987) and therefore dependent upon the kinetic energy of the rainfall and the detachability of the soil but independent of runoff. Under these conditions the model appears to simulate the detachment process, at least by raindrop impact, reasonably well, and always predicts more than sufficient runoff to exceed the transport capacity required to carry the detached sediment. No changes have been made to the procedures used to describe soil particle detachment by runoff, which, as observed by Vigiak *et al.* (2005), may be under-represented.

## Conclusion

Modifications made to the Morgan–Morgan–Finney model to simulate plant cover effects more explicitly have been shown to behave rationally. They also indicate that plant cover effects are more important than soil properties in controlling erosion, which accords with reality. When the model was run as it might be applied by a user without calibration and with a combination of measured and guide values as input parameters, acceptable results were obtained, a judgement based on comparing predicted and observed values using the coefficient of efficiency. Improved performance could be obtained if better input data were available for the detachability of clay particles. Research is also needed to evaluate the proposed guide values for daily saturated lateral permeability of the soil. The results show that it is feasible to simulate the effects of plant cover on soil erosion by water using measurable parameters of plant architecture instead of global plant cover coefficients.

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