Cranfield University

School of Applied Sciences

The National Soil Resources Institute

Doctor of Philosophy

Academic Year: 2006-2007

SOPHIE E COOPER

The role of conservation soil management on soil and water protection at different spatial scales

Supervised by Dr R. J. Rickson

November 2006

This thesis is submitted in partial fulfilment of the requirements for the degree of doctor of philosophy

Abstract

Agriculture has a direct impact on the soil environment, altering soil properties, surface characteristics and erosion risk. This has led to a move away from conventional tillage to the adoption of conservation practices, aiming to minimise soil disturbance and reduce erosion. The effectiveness of this has been shown in studies from the USA, but equivalent research in Europe is limited.

The present study investigated losses of soil, water, nutrients and carbon from different tillage regimes. Two UK sites were used – Loddington (Leicestershire, on heavy clay) and Tivington (Somerset on sandy clay loam). Three cultivations were applied - conventional (deep, inversion ploughing), and two forms of conservation tillage - SOWAP (non-inversion, shallow tillage), and Farmer Preference (non-inversion, deep tillage). Treatment effects were investigated at three spatial scales, ranging from field based erosion plots (0.05 ha), to micro-plots (1.5 m^2), to soil aggregate tests.

Results from 2004 to 2006 showed that adoption of conservation tillage did not consistently reduce losses of soil, water, nutrient and carbon, due to high temporal variability. Notable differences were found between sites. Runoff coefficients ranged from 0.39-0.46% at Loddington, and 2.43-3.82% at Tivington. Soil losses at Loddington were below 2 t ha^{-1} y^{-1}, but higher at Tivington (3.47 t ha^{-1} y^{-1}). Conservation tillage led to notable changes in soil properties and surface characteristics, including a decrease in bulk density and increases in organic matter, micro-topography and residue cover.

Absolute values of erosion from small scale investigations could not be extrapolated directly to field scale results. Relative treatment ranks gave better comparisons, although results were not consistent for all small scale methods, due to high levels of variability. Caution should be used when extrapolating between spatial scales.

Further work is required to understand the links between temporal and spatial fluctuations in soil, surface and rainfall characteristics and erosion processes.

Acknowledgements

Firstly, I would like to thank Dr Jane Rickson, for her supervision, hard work and eternal patience. Thank you.

I wish to recognise Robert Read for his assistance and Kathryn Allton for her encouragement over the course of this project.

Acknowledgement must be given to Sygnenta for providing the funding to allow this investigation to be carried out.

Finally, I wish to thank my family, Lucie Grange and in particular Mike Needs, all of whom gave me great support during the last three years. I could not have done this without them.

Table of Contents

List of Tables

List of Figures

List of Plates

List of Equations

Glossary

Abbreviation	Definition
ANOVA	Analysis of variance
AS	Aggregate stability
C	Conventional tillage treatment
DEFRA	Department for the Environment, Food and Rural Affairs
EA	Environment Agency (England and Wales)
EU	European Union
F	Farmer's Preference tillage treatment
LSD	Least significant difference
MWD	Mean weight diameter (mm)
NGO	Non-government organisation
NPK	Nitrogen, phosphorus and potassium
RC	Runoff coefficient
RI	Rainfall intensity
S	SOWAP tillage treatment
SOC	Soil organic carbon
SOM	Soil organic matter

1 Overview

1.1 Introduction

There have been numerous studies quantifying the effects of different land management practices on runoff generation and sediment production. Rarely are these effects explained in terms of changes to soil properties and surface characteristics. This present study will investigate runoff and soil loss generation under different land management practices and use measurements of soil and surface properties to explain these results. In addition, this study will investigate nutrient and carbon loss associated with generated surface runoff and eroded soil.

Research into soil erosion and erodibility has been carried out at a variety of different spatial scales, from whole catchments (several hectares) to individual soil aggregates (<5mm). At the smallest scale soil aggregates are used as indicators of soil quality and susceptibility to erosion (Bryan 1968). The methods used are relatively cheap, quick and replicable compared to field or catchment scale erosion studies and hence a great deal of research has been undertaken in this area. Erosion studies have been carried out at other spatial scales where erosion rather than erodibility is quantified. Micro-plots, usually one square metre or less are used in conjunction with rainfall simulators. Field erosion plots (of several hundred square metres), sub-catchments and catchments (hectares) are used, relying on natural rainfall events. These larger scales investigations are costly, in terms of set up times and financial requirements for instrumentation and maintenance. Consequentially, it is rare to find a study which encompasses more than one spatial scale. One exception to this was a project based in Sri Lanka that looked at four different scales, from plots to an entire catchment (Hudson 1981). However, such an approach is uncommon, especially in Europe. Of the few studies to have looked at erosion at different spatial scales, only a small proportion considered the impact of different tillage or cropping regimes on erosion rates or processes.

The present study is part of an EU Life Environment project, SOWAP (SOil and WAter Protection in Northern and Central Europe). The SOWAP project represents collaboration between partners from industry, NGO's, academic institutions and farmers from four European countries. The SOWAP project sets out to assess the effects of conventional and conservation agriculture on the environment, in terms of soil and water loss, terrestrial, avian and aquatic ecology and socio-economic indicators, such as crop yields. This type of information is lacking from European studies (Holland 2004). Through the SOWAP project, this current study aims to investigate runoff and soil erosion at three different spatial scales of investigation; field plots (0.05 hectare), micro-plots (1.5m^2) and soil aggregates (3.35-5mm).

There will be several outcomes from this study:

I whether a move from conventional to conservation soil management practices will reduce soil erosion, runoff and associated nutrient and carbon losses

II whether micro-plot scale assessment of erosion can be used to indicate treatment differences of erosion at the field scale

III whether small scale assessment of soil erodibility can be used as a quick, reliable tool to indicate soil erosion risk at a farm scale in the UK. The implication of this is especially important for those farms involved in current European stewardship schemes e.g. the Entry Level Scheme and Cross-Compliance.

1.2 Background and Literature Review

The following section aims to set this research in context with regard to the effect of land management on erosion. This includes discussion of the global importance of soil and soil erosion (1.2.1), an overview of the present situation in the EU (1.2.2), England and Wales (1.2.3), an examination of the mechanisms and processes of soil erosion (1.2.4 and 1.2.5) and a review of how soil erosion is quantified (1.2.6). Summation of the literature is then given to identify the research gaps that this current research will address.

1.2.1 Global importance of soil and implication of soil erosion

Soils are vital to a sustainable existence. They have six main functions:

I supporting ecological habitats and biodiversity

II providing a platform for infrastructural developments

III providing raw materials including water, minerals and other natural resources such as peat

IV producing food and fibre requirements for the human population

V facilitating crucial environmental interactions between the atmosphere, the earth's geology, water and land (EA 2004a)

VI protecting the world's cultural heritage

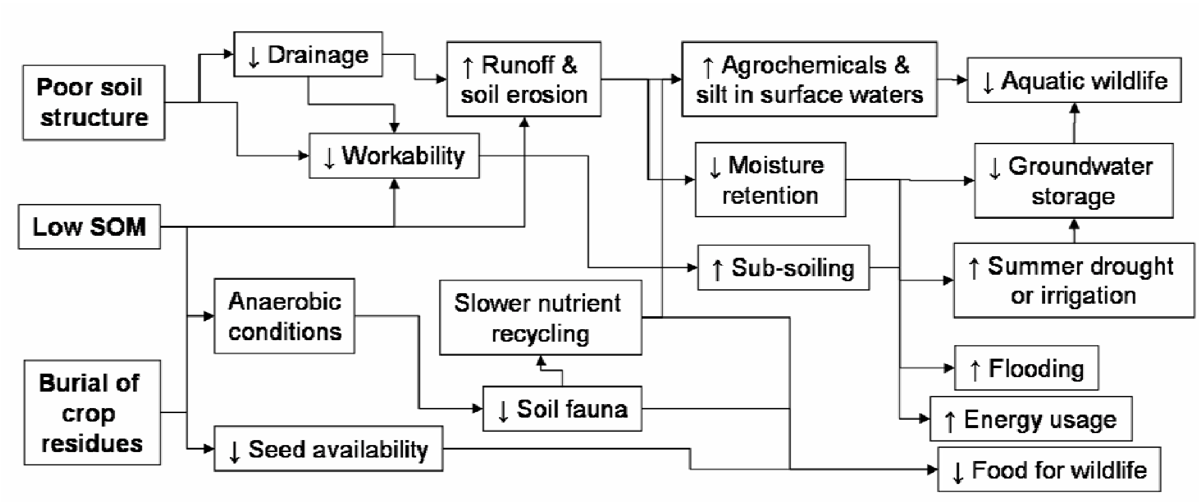


Figure 1.2-1 Environmental effects of soil degradation. Reproduced from Holland (2004).

Soil erosion is a natural phenomenon, eroding and forming at a sustainable rate. Human interference, primarily through agriculture, has increased soil erosion rates above those of soil formation (Hudson 1995). The ability of soil to sustain its functions is at risk due to erosion in particular, as a result of land management. An increase in soil loss above its formation rate will lead to a decline in soil 'health' (EA 2004b). The impact of degraded soils on the environment has been summarised in Figure 1.2-1.

Soil damage from agriculture is a global problem. Increases in agricultural food production are still needed to meet the demands of an estimated 98 countries with 800 million malnourished people (Benites & Vaneph 2001). The Environment Agency for England and Wales (EA 2004) estimated that globally there has been a 23% degradation of available usable land. Desertification affects nearly half of Africa, and severe water and wind erosion affect parts of Asia, China and India. Human-induced soil degradation affects 15% of the total land surface (Oldeman et al. 1991). This has been broken down into major terrain divisions (Table 1.2-1).

Table 1.2-1 Worldwide human-induced soil degradation: major terrain division of the GLASOD map. Modified from Oldeman et al. 1991

Terrain Divisions	Human-Induced Soil Degradation		Total Land Surface
	million hectares	%	million hectares
Africa	494	16.66	2966
Asia	748	17.58	4256
South America	243	13.74	1768
Central America	63	20.59	306
Northern America	95	5.40	1885
Europe	219	23.05	950
Australia	103	11.68	882
World	1964	15.09	13013

It is not just the impact of soil and water loss that is of global importance, agriculture has a direct impact on the quality of its surrounding water courses. The US Environmental Protection Agency or EPA reported in 1992, that agriculture was one of the primary contributors of surface water pollution from siltation and pollution from nutrients and pesticides (Hill & Mannering 1995).

Risk of water pollution from nutrients (specifically nitrogen and phosphorus) is especially high from agricultural land just after the surface application of fertilisers. Ground water supplies of drinking water are polluted by nutrients via leaching from agricultural fields. Ground water contamination by nitrate is of particular concern. In 2002, over half of the total area of England (55%) was designated as Nitrate Vulnerable Zones (NVZs), with 60% of the nitrate coming from agricultural fields (DEFRA 2004b). Costing each farm an estimated £200 per year to rectify, resulting in a country wide combined total cost of £20 million per annum (DEFRA 2003). Nitrate loss from agricultural occurs via runoff and nitrate enriched eroded sediment (Zheng et al. 2005).

Phosphorus loss is another cause for concern, being identified as a primary cause of eutrophication (Plate 1.2-1), a problem especially in Europe (Miller 2004). Loss of phosphorus occurs when dissolved concentrations are high at the soil surface. Rapid transfer may occur into surface runoff or enriched sediment which is lost via erosion (Zheng et al. 2005). Between 1931 and 1991 England and Wales saw a fourfold increase in phosphorus pollution of water (EA 2004a), with over 50% of the rivers containing artificially high phosphate concentrations. Zheng et al. (2005) found that loss of phosphorus associated with enriched sediment could be mitigated by a reduction in the amount of tillage undertaken on a field.

The presence of high concentrations of nutrients (in particular nitrogen and phosphorus) within a water body are causes of eutrophication. Natural water courses generally have low nutrient contents and are described as being oligotrophic. They become eutrophic when nutrients no longer become limiting to the production of biological material. In water bodies this results in the growth of phytoplankton most easily seen as algal blooms (Plate 1.2-1). This results in depleted dissolved oxygen levels within the water body, and in extreme cases, hypoxia which is detrimental to oxygen dependant aquatic organisms, such as fish.

Plate 1.2-1 Eutrophication of an English river (Buckinghamshire)

On-site losses of nutrients from agricultural fields are also of great concern to the farmer. Nitrogen and phosphorus are both essential nutrients to plant growth, as well as potassium. The loss of these primary nutrients from the soil environment directly impacts crop productivity and yield. Nitrogen is important for rapid plant growth, leaf quality and seed and fruit formation. Phosphorus is essential in plant energy reactions allowing the conversions of light into chemical energy. Potassium maintains plant turgidity and is involved with enzyme reactions. Potassium is also important in boosting plant resistance to disease and drought. Reducing nutrient loss by runoff and/or sediment saves the farmer from having to apply increased amounts of NPK fertilisers.

Although not a primary nutrient soil organic carbon (SOC) is vital for crop productivity (Charman & Murphy 2000) and soil health. Increases in tillage regimes and associated erosion have led to a decline in SOC. This has left a deficiency within many soil systems, with an estimated 18% of organic carbon lost in English soils between 1980 and 1995 (DEFRA 2004). More recently, work by Bellamy et al. (2005) has shown that between 1978 and 2003 soil carbon was lost at a mean rate of 0.6% yr^{-1} in England and Wales. Where carbon levels are low, farmers can apply additional organic matter or incorporate previous crop residues (Charman & Murphy 2000). Soil organic carbon has also been linked to an increase in biological activity and with the stability of soil aggregates (Charman & Murphy 2000). The importance of SOC has been recently recognised, having been proposed as a soil quality indicator in Action 11 within the first Soil Action Plan for England (DEFRA 2004a). However, there is little known as to the relationships between SOC and soil, water and plant variables under different land managements and climates, due to temporal and spatial variation in SOC (Verheijen et al. 2005).

Currently a topical subject, the ability of a soil to store carbon, known as sequestration, is important in the reduction of the green house gas, carbon dioxide. Autotrophs or primary producers initially fix carbon dioxide from the atmosphere, leading to the eventual storage of carbon within the soil environment (Killham 1994). This is part of the carbon cycle. It is estimated that 10 billion tonnes of carbon is stored within UK soils (EA 2004). Soil carbon sequestration may mitigate against one-third of yearly increases in atmospheric carbon dioxide (Smith 2004). This effect is estimated at being limited to 20-50 years as soils reach equilibrium (i.e. reach saturation point) and may be reversed with land use change (Smith 2004). Paustian et al. (1997) identified that some management practices in temperate climates can increase soil carbon content. Practices included the retention of crop residues and, related to this, an increase of soil organic matter (Holland 2004) and a reduction in tillage operations. Owens et al. (2002), observed that a reduction in tillage may reduce carbon transport by lowering concentrations within eroded sediment. However, a reduction in loss of eroded sediment was more effective at minimising carbon movements than decreases in carbon concentration.

In addition to carbon dioxide, soil is important in the emission of other trace greenhouse gases such as methane (CH_4) and nitrous oxide (N_2O) (Smith & Conen 2004). In particular the emission of nitrous oxide can be reduced by improving diffusion of the gas throughout the soil matrix, by maintaining drainage and avoiding anaerobic conditions (Holland 2004). This could be achieved by the long term adoption of conservation based tillage practices, although short term emissions would be expected to increase until the soil structure had become well established (Holland 2004).

Conservation of soil biodiversity is important in the maintenance of soil health and resistance to erosion. There is a positive feedback mechanism by which conserving soil and soil biota is essential for soil processes such as nutrient cycling, organic matter and carbon breakdown, improvement of soil structure and increase in aggregate stability (Brady & Weil 2002). These changes in soil properties can lead to a reduction in soil erosion (and thus conservation of soil biodiversity) by increasing the resistance to breakdown, increasing drainage and therefore reducing the risk of overland flow and sediment generation. Soil ecology is also fundamental to numerous food chains.

A move towards better agricultural land management practices should alleviate some of the threats to the environment (Benites & Vaneph 2001) including the world's soil and water resources, as mentioned previously. Other outcomes may include long term benefits of higher crop yields coupled with a reduction in input of labour and capital due to fewer required field operations. However, short term decline in crop yield can be found and increases in capital needed due to the purchasing of specialised equipment.

1.2.2 Soil Protection in Europe

In the past, soil was not recognised as a vital, non-renewable resource, but a better understanding of natural capital has led to a recent change within the EU (Gobin et al. 2004). The EA (2004) estimate that within the EU 16% of land is affected by soil degradation, while Oldeman et al. (1991) calculated 23% degradation representing 1,964 million hectares (Table 1.2-1). Of the 23% degraded land Oldeman et al. (1991) calculated that 12% was attributable to water erosion and just over 4% from wind erosion. In Europe the average formation rate of soil is 0.5 to 1 Mg ha^{-1} (Troeh & Thompson 1993), however average losses on cultivated arable land are generally greater than the formation rates. Losses are highly variable e.g. it is estimated that in the average UK rates of erosion range between 0-20 t ha^{-1} per annum (Morgan 2005; Gobin et al. 2004). These erosion rates are not sustainable.

In 2002 the European Parliament and Council adopted the Sixth Environment Action Programme 'Our Future, Our Choice', first published in 2001 (EC 2001). This programme runs until 2012, by which time the European Commission must have prepared seven Thematic Strategies covering different areas of the environment; one of which is soils (EUROPA 2006). The objectives of annex two within the erosion mandate are:

I soils and their functions are to be protected against erosion, in relation to the viability of agricultural land

II information on soil is to be synchronised throughout Europe by use of data networks and soil surveys

III a sustainable, EU soil protection policy on erosion should be developed, based on the concepts of prevention and precaution

IV soil protection should be integrated within important EU policies

V differences in local and regional needs should be recognised, and integrated into soil protection policies (Van-Camp et al. 2004 & 2004b).

The use of conservation agriculture has been extensively researched in North American (Uri et al. 1998), in Canada (McLaughlin & Mineau 1995) and Australia (Vere 2005). In the EU however, conservation agriculture as a concept has involved much less study. Successful European adoption of conservation agriculture may be constrained by several factors, including insufficient support for the farming community regarding access to sufficient information about new techniques, the financial implications and the lack of support groups with other farmers (Nelson 2006). Appropriate incentives for following conservation orientated farming practices are paramount to successful adoption across the EU. It is important that farmers are educated in the wider benefits of conservation agriculture, not just in terms of crop productivity but in the control of soil and water loss, and the improvement to local biodiversity and surrounding environment. Legislation is already in place giving farmers incentives for following best management practices. The common agricultural policy (CAP) was the primary subsidy programme running in the EU which was originally based on production. Since the 1990s CAP has undergone multiple reforms. It is now based on single payment schemes to farmers who are encouraged to employ more environmentally based practices (EUROPA 2003). Member states employ their own agricultural schemes. In England, DEFRA introduced in 2005 the new environmental stewardship schemes, designed to address the conservation of wildlife, the protection of historic features and natural resources, and promote public awareness (DEFRA 2006). There are three types of environmental stewardships schemes. Entry level scheme (ELS) open to all farms, organic ELS specific to organic farms, and higher level stewardship (HLS) which targets priority areas requiring more complicated management regimes. Both ELS and organic ELS address the need to management soil and nutrients (DEFRA 2006). The adoption of conservation tillage practices could be used as a management tool in the control of nutrients and soil loss.

1.2.3 Soil and water protection in England and Wales

In the past farmers in England and Wales considered soil loss through water erosion to be relatively unimportant, unless crop productivity was affected (EA 2004). Across England and Wales 17% of soils show signs of erosion, and annually 2.2 million tonnes of arable topsoil is lost (EA 2004a). Although crop productivity decline can be masked by the addition of fertilisers for example, the off-site impacts of this erosion are immense. Soil erosion within the UK is reported to be mainly associated with water rather than wind erosion (Robinson & Blackman 1989), and is concentrated in areas with low clay content. Other processes of soil loss such as tillage erosion and soil co-extracted on root crops have been reported by Owens et al. (2002, 2006). It is reported by the Environment Agency (EA 2004) that the annual costs of remedying soil related problems are extremely high. In 1995 sediment clean up within urban drainage systems cost between £50 and £60 million, and £20 million within rivers. The remediation costs of agricultural related environmental impacts from soil erosion were in 2000, estimated to be £90 million (EA 2004). Increasingly in the UK, soil erosion is having a more direct impact on society through devastating floods causing considerable damage to people's homes and livelihoods. In 2004, for example, the area of Boscastle in Cornwall, received over 1,400 million litres of rain fell in two hours across a 23km^2 area (NCDC 2006) causing extensive damage.

Adoption of better land management practices is aimed at minimising risk from erosion. An increasing number of projects have been set up to study this on working farms. The Parrett Catchment Project is one such study, where the impact of soil management on the water environment is investigated. This project also aims to disseminate its findings directly to farmers through demonstration days and trials (SCC 2006). The Allerton Project also aims to demonstrate to farmers how to integrate conservation land management and profitable farming (Leake 2005).

Despite these on-going projects, the EA (2004) states that across England and Wales there is still a distinct lack of good quality information on soil protection, restricting the creation and implementation of effective soil protection programmes and policies. Clearly further research is required in England and Wales on current soil susceptibility to erosion, and the effects of different land management practice.

1.2.4 Processes of soil erosion

Erosion is a natural process, often referred to as geological erosion. Human intervention and manipulation of soil has lead to an increased amount of erosion, known as accelerated erosion. Soil is made up of structural units containing planes of weakness. When stress is applied, breakdown occurs along these planes, producing soil fragments more stable than the applied stress. As more energy is applied, fragmentation increases. The rate of this degradation is linked to the structural stability between aggregates (Diaz-Zorita et al. 2002). In its simplest form erosion by water is the transformation of soil into sediment (Brady & Weil 2002) and occurs by a three step process; detachment, transport and finally deposition.

There are multiple ways in which detachment takes place, usually following the breakdown of soil aggregates. Raindrops dominate the process of surface aggregate breakdown and are thus the primary detaching agent. As a raindrop hits the surface of an exposed soil aggregate, the mechanical energy from the water droplet dissipates, causing the aggregate to deform or shear. Small particles are detached from the main aggregate body and are projected vertically and horizontally from the point of impact. The severity of this action is dependant upon multiple factors including the energy or erosivity of the rainfall and the susceptibility or erodibility of the soil. Detached soil particles are transported in the trajectory jets or ‘splash’ effect of the impacting raindrops, to distances as much as 0.7m vertically and 2m horizontally (Brady & Weil 2002), exacerbated by windy conditions.

Once aggregates are broken down, detached soil particles are then moved via transporting agents; water based processes only are discussed here. There are two main transporting agents (Morgan 2005) - rain-splash (as described previously), which transports detached soil over a uniform area of infinite width, and overland flow. The latter initiates either a) when soils are saturated or near-saturated, and infiltration capacity is close to zero, or b) when soil infiltration rates are exceeded by rainfall intensity, as may occur when surface seals or caps are present. A crust or seal is defined as an impermeable layer developed by the reconstruction of surface aggregates. Seal formation is driven by rainfall or irrigation, whereas crusts are the results of soil drying. Crust and seal formation are affected by:

I soil properties in particular clay and organic matter content (Morgan 2005) and aggregate stability (Robinson & Phillips 2001);

II surface conditions such as surface roughness and cover (Linden et al. 1988); and

III the properties of rainfall itself (Brady & Weil 2002).

Plate 1.2-2 Rill erosion in Somerset. Source www.sowap.org

If rainfall input is greater than the infiltration rate ponds will form in the soil surface depressions. The depth and size of surface ponds is dependent on the soil properties and surface conditions present at the time. If the sides of a pond break or if multiple ponds connect, then surface water flows down slope as surface runoff (also termed overland flow). This movement of water can be as sheet or inter-rill flow; a smooth thin single layer of water that carries rainsplash-detached particles down slope. Inter-rill flow is unlikely to detach soil particles, due to low hydraulic energy available for detachment by flow (Morgan 2005). Also, it is rare that inter-rill flow remains as a continuous water layer, due to the presence of surface irregularities such as soil microtopography, stones, crop residues and vegetation. Such surface irregularities concentrate the flow into channels of various sizes, forming other transporting agents, including micro-rills, rills and gullies. The velocity of such concentrated flow is relatively higher than for inter-rill flow, so not only are eroded particles transported effectively, but detachment of soil also occurs (Brady & Weil 2002). Rills, which by definition can be removed during ploughing or by subsequent rainfall events cut into the soil mass (Plate 1.2-2) and may start to retreat up slope via the process of undercutting (Morgan 2005). As this process becomes accelerated, larger erosion features such as gullies are created. Large quantities of sediment are moved within gullies and once formed, they are extremely difficult to eradicate, as, by definition, they are usually deeper than the depth of ploughing operations.

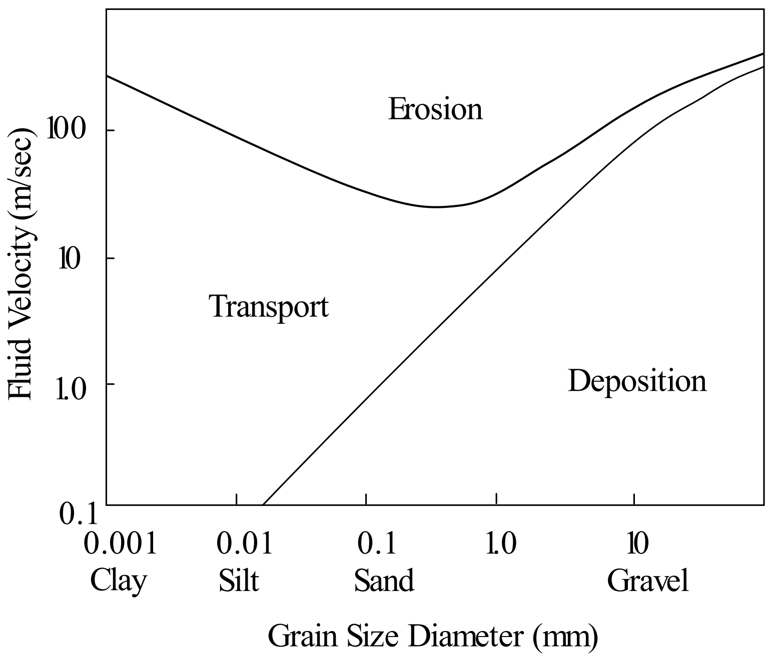


Figure 1.2-2 Energy requirements of soil erosion, transport and deposition in water (based on Hjulström's Curve)

The transportation of aggregates or soil particles requires energy. If energy levels fall below this threshold point soil particles are deposited. The deposition rates of soil are related to the size and mass of soil particles. This can be summarised in Figure 1.2-2 based on Hjulström's curve (originally based on channel velocity associated with rivers), showing the energy levels required for the processes of erosion, transport and deposition of particles in water (Hjulström 1935). Similar findings have been found by Poesen (1992) of the relationship between particle size and the kinetic energy required for detachment.

The breakdown, detachment, transport and deposition of soil are influenced by soil properties and soil surface characteristics present at the time. These are in turn affected by soil management. Soil management practices have been shown to be an important influence in the development of gullies and other erosion features (Oygarden 2003).

1.2.5 Factors affecting erosion

Any investigation of soil erosion and soil conservation must consider the factors affecting erosion, to aid design and development of soil erosion control practices. It should be understood that not all factors can be changed by human intervention or management.

1.2.5.1 Rainfall

All other factors being equal, the intensity and volume of rainfall received at a soil surface is closely associated with the amount of runoff and soil erosion generated there. Seasonal fluctuations in rainfall and the condition of the soil will affect these links. The impact of a low intensity storm over a long duration may have more or less of an impact than a high intensity storm over a short period of time. The effect of seasonal changes in rainfall is currently a topic of discussion in relation to global climate change. Areas which previously experienced multiple low intensity storms may start to get one or two intense storms per year and vice versa. The condition of the soil when it rains also affects erosion rates.

Morgan (2005b) showed that work by Fournier (1972) measured a storm of 19.3mm which fell on dry ground resulting in only 25% runoff, the remaining water infiltrating the soil. The following day 13.7mm of rainfall fell, which lead to a loss of 66% runoff and a trebling of soil lost. Rainfall erosivity is used as an index of the ability of rainfall to cause erosion, taking into consideration the total volume and intensity of rainfall as well as the kinetic energy and drop size of the rain itself. An increase in rainfall erosivity is likely to generate higher erosion rates. It should be noted that not all rainfall is erosive. Hudson (1965) estimated that only rainfall intensities of over 25 mm h^{-1} were erosive, which represents only 5%, of British rainfall for example. However, Morgan (1980) suggested that for the UK rainfall intensities of 10 mm h^{-1} and above are erosive, and Reed (1979) stipulated that rainfall intensities as low as 1mm h^{-1} would be sufficient to initiate erosion as long as a total of 10mm of rain fell during that storm.

In terms of controlling erosion, little can be done with regard to rainfall received at a site. However, the fate of that rainfall (as infiltration or runoff) is strongly affected by the adopted management practices at a site.

1.2.5.2 Soil Properties

Soil erodibility is the susceptibility of soil to be detached and transported (Brady & Weil 2002). Soil erodibility is affected by a variety of soil properties, the most important of which have been stated as being the structural stability and infiltration capacity (Brady & Weil 2002). A soil with a higher infiltration rate is at less risk of surface ponding and eventual overland flow. However, the presence of surface ponds reduces the energy of impacting raindrops (Palmer 1964) and reduces soil detachment. Soil stability is also important, affecting the resistance of aggregates to breakdown via raindrop impact. Both infiltration rates and soil stability change with soil properties. In this review the key properties are discussed below. What is of particular relevance to the present study is to ascertain whether it is feasible to modify any of these properties by management practices.

1.2.5.2.1 Texture

Soil comprises of three main fractions, sand, silt and clay. The proportions of each within a soil affect water retention, pore space, permeability, soil strength and adsorption capacity of nutrients and carbon (Charman & Murphy 2000). Soil texture was identified by Wischmeier & Smith (1978) as an important inherent soil property affecting soil erodibility, and as such was incorporated into the K factor of the Universal Soil Loss Equation. Clay has been shown to have a positive correlation to the stability of soil aggregates (Holland 2004; Le Bissonnais et al. 2002; Levy & Mamedov 2002) due to the strong cohesive forces between clay particles. It has been stated by the European Environment Agency that soils with low clay contents are at greater risk of soil loss during erosion (Gobin et al. 2004). Increased clay content within soil has also been linked to greater risk of surface sealing and capping (Ferry & Olsen 1975). Clay particles have a high specific surface area which makes them highly adsorptive, meaning that soils with increased clay content retain higher concentrations of nutrients and carbon. Consequently when this fraction is eroded, there is preferential loss of nutrients and carbon on-site, and enrichment of these properties in water and sediment off-site.

1.2.5.2.2 Organic Matter & Carbon

Soil organic matter (SOM) and soil organic carbon (SOC) are interrelated. This is unsurprising as organic matter is the result of the decay of carbon based biologically derived material. Organic matter is said to be one of the most important soil factors (Holland 2004) affecting soil structure and stability and a useful indicator of soil sustainability (King et al. 2005). Research has shown erosion is likely to initiate when SOC levels fall below 2% (Greenland et al. 1975; Evans 1996). Wischmeier & Smith (1978) identified organic matter as an important inherent soil property affecting soil erodibility, and as such was incorporated into the K factor of the Universal Soil Loss Equation. Aggregate stability has been shown to be strongly linked to the organic matter content within the soil (Robinson & Phillips 2001; Le Bissonnais et al. 2002), affecting other soil characteristics such as seal and crust formation, degree of cohesion between soil particles; and reduction of bulk density. The presence of SOM also influences soil biological activity and nutrient balances (Holland 2004; Fullen & Catt 2004). Organic matter and carbon content have also been linked to the wettability of soil aggregates. An organic matter film around an aggregate makes it more water repellent (hydrophobic), thereby lowering wettability (Ellerbrock et al. 2005), and reducing erodibility. Eynard et al. (2006) believes that SOC is the most effective tool at regulating soil wettability.

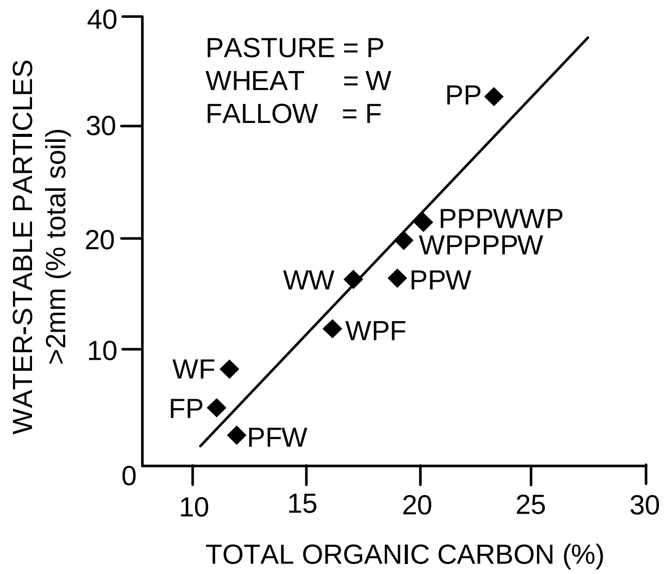


Figure 1.2-3 Reproduced from Tisdall & Oades 1980; as cited in Charman & Murphy 2000

1.2.5.2.3 Moisture Content

The moisture content of soil aggregates can affect the severity of aggregate breakdown from rainfall. As rain falls onto dry aggregates, water is forced into air filled pores. If air becomes trapped within the pores, it is released suddenly under pressure, forcing soil particles apart – a process known as slaking. An increase in moisture reduces this explosive response. However, as soil moisture decreases, cohesion between particles increases and inorganic cements (e.g. carbonates) are concentrated, drawing together clay platelets (Kemper & Rasenau

1984; Kemper & Rasenau 1986). Infiltration rates are also affected by soil moisture, as shown above. Water infiltrates through the soil matrix, driven by gravitational forces and the suction from dry aggregates (Charman & Murphy 2000). The effect of the latter will decline if moisture content increases within the soil matrix. Previous research has shown that soil infiltration rates were greater in wet soils (not saturated) compared to dry soils (Le Bissonnais & Singer 1992). Soil moisture content also affects the strength of surface seals (Bennett et al. 1964; Callebaut et al. 1985) which will also affect infiltration and the potential for runoff generation. The links between moisture content and erosion processes is reflected in the fact that moisture content is used as an indicator of soil erodibility in the Morgan-Morgan-Finney model (Morgan 1995).

1.2.5.2.4 Bulk density

A soil with a high bulk density indicates that the soil matrix has been consolidated with few pore spaces. As bulk density increases, infiltration rates decrease and drainage is impeded. If water cannot infiltrate through the soil, it will collect on the soil surface and will contribute to overland flow. Therefore, the risk of erosion increases as the bulk density increases. The links between bulk density and susceptibility to erosion are reflected in the Morgan-Morgan-Finney model (Morgan 1995).

1.2.5.2.5 Soil Biota

One soil property affecting erodibility that is often overlooked is the role of soil biota. This is rarely considered explicitly in soil erosion models. With the increase in research in this area, it is becoming more apparent that the soil biota have an important affect on soil structure, aggregation and therefore erosion (Jastrow & Miller 1991; Kandeler & Murer 1993; Kiem & Kandeler 1997). Biological processes play a great role in the aggregation of macro (0.3mm) soil particles (Brady & Weil 2002; Rowell 1994). Larger aggregate formation or formation in sandy soils (with little clay content) relies substantially on biological processes. Primary soil particles are bound together to form aggregates through biotic stabilising agents (Brady & Weil 2002; Stuttard 1985). Worms (especially earthworms), certain arthropods (e.g. termites) and plant roots force soil particles into aggregates through the action of burrowing. Earthworm activities have been found to be important in the formation of macro- and micro-aggregates (Six et al. 2004) and the increase in soil aggregate stability (Pulleman et al. 2005). Plant roots (especially root hairs) and fungal hyphae bind soil particles together through the formation of sticky networks of organic compounds. This process aids the formation of macro-aggregates. The production of organic glues from plant roots, bacteria and other microbes bind soil particles together; this production of organic glues are prevalent where organic matter accumulation occurs. Water resistant glues are important in the long term stability of aggregates (Brady & Weil 2002). Work by Kiem & Kandeler (1997) showed that the resistance of aggregates to slaking increased with microbial biomass. This work confirmed similar findings by Kandeler & Murer (1993). The importance of this relationship was found to decline as soil clay content increased (Kiem & Kandeler 1997). Aggregate stability is intimately linked to the generation of soil erosion and runoff (section 1.2.5.2.6).

1.2.5.2.6 Aggregate stability

Surface soil aggregates are formed through the aggregation or arrangement of primary soil particles into structural units of different sizes. These sub-units are bound together through environmental, abiotic (physical-chemical) and biotic processes (Stuttard 1985). Environmental processes drive the formation of micro-aggregates, defined by Brady & Weil (2002) as being <0.3mm driven by temperature and water cycles (i.e. freeze/thaw and drying/wetting actions). This is particularly relevant on smectite dominated soil e.g. vertisols (Brady & Weil 2002; Stuttard 1985). Abiotic processes also drive micro-aggregation dominating aggregates <1mm (Stuttard 1985). Abiotic processes involve aggregation through stabilising agents including cohesive force between clay and water, flocculation of clay platelets into clay domains, and inorganic cements (calcium and iron or aluminium sesquioxides). The final process (biotic) involves actions from roots and micro-organisms in the generation of aggregates (Brady & Weil 2002; Rowell 1994; Jastro & Miller 1991).

Soil erosion processes are influenced by aggregate stability, the most important of which is the resistance of aggregates to rainfall impact (Legout et al. 2005). Stable aggregates resist breakdown, and thus are less prone to the associated processes of surface crust and seal formation, reduction in infiltrate rates and the generation of runoff and sediment (Le Bissonnais & Arrouays 1997; Tisdall & Oades 1982). Aggregate stability is increasingly being considered as an important indicator of soil erodibility due to the linkages with soil properties (Six et al. 2000; Bryan 1968). Bryan (1968) found the stability of aggregates to be the most important soil property affecting soil susceptibility to erosion.

Agriculture also plays an important role in the breakdown of aggregates via mechanical manipulation by modifying the soil properties mentioned above, and altering the aggregate size distribution. It was found that smaller aggregates were more susceptible to wetting and as a result led to increased soil losses (Teixeira & Misra 1997). However, work by Abu-Hamdeh et al. (2006) found a positive relationship between clod size and splash erosion due to a reduction in tensile strength with clod size increase. It should be noted that these studies used different ranges of aggregate sizes and as shown by Poesen (1992, Figure 1.2-4) and Hjulström (1935, Figure 1.2-2) would have affected the required energies in detaching and transporting soil particles. The work by Hjulström (1935) was based on stream flow velocities but is still valid in highlighting the relationship between energy and particle size detachment. Both Poesen (1992) and Hjulström (1935) show that very small and coarse particles results in the greatest resistance to detachment, due to strong adhesive or chemical bonding in small particles and the affect of increased weight of coarser particles (Morgan 2005).

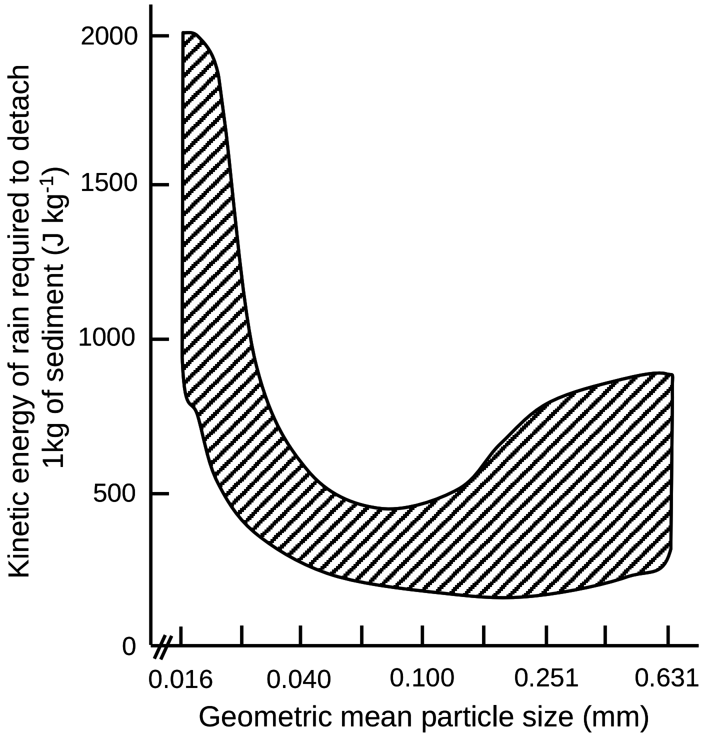


Figure 1.2-4 Relationship between detachment energy and particle size. Hatched area representing range of experimental values Reproduced from Morgan (2005) on work by Poesen (1992).

1.2.5.3 Soil Surface Characteristics

In this review the key surface properties affecting soil erosion and runoff generation will be discussed. These include the surface cover and roughness, both of which can be modified by soil management practice.

1.2.5.3.1 Surface Cover

Surface cover may include the main crop itself, a sown cover crop, weeds, crop residues, stones or an erosion control blanket. The latter is outside the context of this research and shall not be discussed.

The presence of a surface cover physically protects the soil from direct rainfall impact, minimising soil detachment (Morgan 2005). The greater the percentage of soil covered the smaller the area exposed to rainfall, thereby reducing the susceptibility of soil to detachment and associated formation of a crust, seal or pond (Robinson & Philips 2001). The application of residues is a common way of increasing the surface cover, and this is a distinctive characteristic of conservation agriculture as opposed to conventional tillage practices. Residue application is associated with an increase of organic matter (Robinson & Blackman 1989) and related to this, aggregate stability.

The effect of surface cover on soil erosion processes was well documented in work carried out by Hudson (1957) in Zimbabwe where experimental plots were compared. The treatments consisted of bare plots and plots with an artificial covering (gauze). It was found that the presence of surface cover reduced soil loss by up to 100 times to that generated from bare plots (Hudson 1995). Other research has also found reductions in erosion rates from soils with surface cover compared to bare plots (Laflen & Colvin 1981; Ngatunga et al. 1984; Isensee & Sadeghi 1999). The impact of a surface cover is clearly presented in Table 1.2-2, taken from a report by the USDA on the effect of conservation tillage on water quality.

Table 1.2-2 Taken from (Hill & Mannering 1995): effects of surface residue cover on runoff and soil loss

Residue Cover %	Runoff (% of rain)	Sediment in Runoff (% of runoff)	Soil Loss (tons/acre)
0	45	26	12.4
41	40	14	3.2
71	26	12	1.4
93	0.5	7	0.3

The effects of surface cover on protecting the soil is changeable as residues are broken down overtime and vegetation growth alters crop height, stem length, leaf shape area and rigidity (Morgan & Rickson 1988). The presence of most surface covers are likely to increase infiltration into the soil and lead to a reduction in available water for overland flow (Charman & Murphy 2000). The presence of surface stones has also been linked to a reduction in surface runoff and sediment concentration (Mandal et al. 2005; Poesen & Lavee 1994; Poesen et al. 1990; Cerdá 2001).

1.2.5.3.2 Surface Roughness

Surface roughness, as affected by surface cover (see above) and/or soil management practices can have a significant impact on soil erosion and runoff, by altering the direction and velocity of runoff flow (Takken et al. 2001). Surface roughness is a major controlling factor in the generation of overland flow (Darboux et al. 2001); however, this effect is reduced after every rainfall event (Mwendnera & Feyen 1994). A soil with a rough surface will have an initially higher infiltration rate compared to a smooth surface, due to presence of soil surface depressions. These fill during periods of rain when infiltration rates are lower than rainfall input. Water then collects at the surface and retained in these surface depressions. Once the storage capacity of these depressions is reached or the walls are breached, runoff is initiated. Overland flow occurs as other overfilled depressions connect (Darboux et al 2001). A rough surface also reduces the risk of soil loss by increasing the hydraulic resistance, disrupting and dissipating the energy within surface runoff (Foster 1982; Einstein & Barbarossa 1951; Abrahams & Parsons 1991). This reduction in available energy within the water flow reduces the risk of soil detachment and transport. The presence of depressions also allows sedimentation of soil entrained within overland flow. These effects can be represented by parameters such as the Manning's n coefficient, which shows the degree, to which surface roughness (imparted by vegetation and /or the micro topography) reduces the velocity of flow, and thus soil detachment and transport (see Figure 1.2-2).

1.2.5.4 Site Topography

Slope length and gradient affect erosion rates. As the effective catchment area of rainfall increases there is a positive relationship between slope length and overland flow volume. As slope length increases so too does the velocity of runoff as gravity accelerates momentum. However, the relationship between slope length and soil loss is not linear, partly due to the greater influence of surface irregularities on flow characteristics, and increased opportunity for infiltration and deposition to take place on longer slopes. Many workers have tried to find this relationship (Morgan 2005). If rills are the dominant erosion process operating on a slope, then slope length has been stated as having a greater influence on runoff generation than slope gradient (Brady & Weil 2002). Similarly with slope gradient, the general theory is that erosion rates increase on steeper slopes due to higher energy of overland flow (Potential energy = mass x gravity x height). Also, the distance soil particles are transported during rain splash erosion is greater as the slope gradient increases, resulting in a greater net movement of soil down slope.

1.2.5.5 Soil management

Soil management practices can modify the factors affecting erosion as detailed above. Agriculture has a direct impact on the erodibility of soil as it alters the organic and chemical content, aggregate stability, size distribution and infiltration capacity (Morgan 2005). Traditional or conventional forms of agriculture involve primary and secondary soil cultivation. Soil is tilled and inverted, large clods of earth are broken up to produce an even seed bed for plant growth. This action inevitably alters the aggregate size distribution affecting soil erosion, transport and deposition as shown by Hjulstrom's curve (Figure 1.2-2) and work by Poesen (1992, Figure 1.2-4). Aggregate stability is altered through management and through the alteration of other soil and surface characteristics such as organic matter, the soil biota (i.e. earthworm populations, microbial community structure and mass), surface cover and roughness. Organic matter and the organic glues and gums produced from soil biota act as binding agents increasing aggregate stability. This stability is lost during tillage as the soil is inverted burying organic matter and soil surface biota. Soil inversion also results in a loss of surface cohesive strength (Brady & Weil 2002). Tillage can lead to compaction problems (Holland 2004) which results in increased bulk densities and higher runoff volumes. Compaction can occur with conventional and conservation practices but can be mitigated by sub-soiling (Larink et al. 2001). Mwendera & Feyen (1994) showed that bulk density increased after rainfall but that this effect was the least on untilled treatments compared to harrowing regimes.

Agriculture also modifies soil surface characteristics, through tillage, applications of residues and sowing of different crops. An increase in tillage can reduce the surface roughness which has been shown to lead to higher erosion rates. Work by Mwendera & Feyen (1994) showed this related to four different tillage treatments, ploughed, harrowed and rolled. Despite the surface roughness being reduced on all treatments after rainfall, it was found that a ploughed soil would maintain a higher water storage capacity when compared to a rolled surface because the smoother surface would generate more runoff and soil loss. A part of land management is the control of ground cover in terms of type and application rates. Ground cover is important to erosion rates (as discussed previously 0). In the main the application of residues are associated with a decline in runoff and sediment generation (Laflen & Colvin 1981). It has been reported by Wischmeier & Smith (1978) that the alteration of the cover and management factor, or C factor, as described in the USLE is a common way of farmers to move towards conservation practices.

These effects of agricultural practice on erosion rates have been shown by many workers. Ngatunga et al. (1984) investigated several soil management treatments, two of which were mulch tillage (around 30% cover) and one ploughing. It was shown that for the steepest slope tested (22%), the annual soil loss and runoff coefficients (i.e. the ratio between rainfall received to runoff) were 23.5 tonnes ha^{-1} and 5.1% for a ploughed soil, and 0.18 tonnes ha^{-1} and 0.5% for the mulched soil.

1.2.6 Erosion monitoring and assessment

Monitoring and assessment of erosion processes and the factors affecting these processes can be done through 1) direct measurement of erosion though field and/or laboratory work; or 2) the use of models which combine multiple soil and site characteristics to simulate erosion processes and / or identify erosion risk.

1.2.6.1 Direct measurements

Direct measurements of soil erosion can be done in a variety of ways including surveying and physical collection. Simple erosion surveys may concentrate on a pre-defined area of land, in the field or on aerial photographs of a suitable scale (e.g. 1:10,000). Here, point locations are systematically surveyed for the presence/absence of erosion features (e.g. micro-rills, rills, gullies, sheet erosion and deposition fans). The spatial distribution and density of the erosion features is taken as an indicator of erosion severity in the area. A more detailed, quantified approach would measure the dimensions of these features, in order to calculate the amount of soil lost from that area. In addition, present erosion activity can be addressed including stages of development (e.g. active, dormant, and senescent). These in-depth surveys allow identification of spatial variations in erosion severity across a larger area. This technique can also be used to identify areas likely to be affected in the future, so that soil conservation management can be employed at these locations. However, this method does raise some problems. Areas that have already suffered erosion may be too degraded for soil conservation to be cost-effective. Also, socio-economic factors are rarely included in the mapping because of the complexity, yet are vital for soil conservation.

Direct measurements of soil erosion can be done in the field or in the laboratory. Field measurements can be achieved by setting up a collection system into which eroded sediment and runoff from a defined area of land can discharge. Large scale (catchment or sub-catchment) collections utilise natural drainage lines.

Field scale experiments demarcate an area for study by the use of artificial boundaries or natural borders i.e. ditches, earth bunds or man-made material (e.g. metal sheeting). At a small scale erosion plots are still used and are termed micro-plots (Morgan 2005). At the larger scale (catchment to field) natural rainfall is mainly used while at the micro-plot scale, simulated rainfall systems are set up. In either case, the amount of rainfall is measured and the amount of runoff and/or soil loss is measured, giving a direct measurement of erosion for that particular area. Laboratory experiments are also carried out, using simulated rainfall to measure erosion or erodibility. Relatively undisturbed soil samples may be removed from the field and placed under a rainfall simulator and a variety of measurements can be taken, including surface runoff, sub-surface runoff, infiltration rates, and changes in soil properties. Artificially packed soil with known physical properties i.e. soil texture, bulk density, nutrient composition and vegetation type, are used to investigate the role of certain soil properties or surface characteristics on soil erosion. Single soil aggregates can be tested to indicate their resistance to breakdown. Such tests include wet sieving, dropping single raindrops onto the aggregate or exposure of the aggregates to simulated rainfall.

1.2.6.2 Factorial scoring, classification and erosion models

Factorial scoring involves the identification of factors that affect soil erosion including rain erosivity, soil erodibility, slope of the land, land cover and the impact of human occupation of the area. Each factor is given a score from low to high risk at specific survey points across the area. The scores are totalled and areas of high risk can be identified. Although this method is fast, relatively simple and gives clear spatial differences, it is a subjective method and does not consider the interactions of the selected factors in terms of erosion processes operating.

Soil erosion models are used to predict soil erosion risk or rates of erosion for a specified area. The inputs of such models can be modified to simulate erosion risk or rates for pre-defined scenarios for example, changing climatic or land use conditions or varying land management practice. There are many erosion prediction models in existence but some notable ones include the Universal Soil Loss Equation (USLE), Water Erosion Prediction Project (WEPP; WEPP 2006), European Soil Erosion Model (EUROSEM) and the Morgan-Morgan-Finney model (MMF). All erosion models require input data which may include rainfall (erosivity, volume and, intensity), soil (erodibility, properties such as moisture content, bulk density, cohesion strength, depth and surface depression storage), slope steepness and length, cropping regimes, land management, and land cover (Morgan 1995). With any model it is important that it should be validated by measured results. One limitation of erosion prediction models is that the input factors are often represented as a single figure which does not take into account their inherent spatial variability. Validation of these models is limited, and there is still a great deal of doubt as to the accuracy of them.

1.2.7 Present study

Investigating soil erosion on agricultural fields is a challenging task due to the existence of natural variability within fields and between farming practices. Previous work has been carried out at a variety of spatial scales from whole catchments down to individual aggregates, but very few studies encompass more than one or two spatial scales within the same study. This may be due to the absence of a standard methodological approach to tackle multiple spatial scale investigations. There are also financial implications associated with larger scale collection of data through direct measuring.

The present study is unique in that it considers three spatial scales concurrently within the same project – at the field scale, the micro-plot scale and on individual aggregates. At the same time, unlike other erosion projects, the present study considers the effects of different soil management treatments at these three spatial scales simultaneously. This thesis will attempt to explain the results observed at each scale by considering the changes in soil properties and surface characteristics which result from soil management practice applied. Finally, this study will test whether small scale assessments of soil susceptibility to erosion can be used to indicate field scale erosion. These areas of investigation have been specifically identified in Volume 6 of the Soil Thematic Strategy as being essential in future soil erosion research (EUROPA 2006b).

1.2.8 Summation

Soils are an important global resource, under threat from misuse and over-use. This is primarily because of over food production, due to population increase pressures leading to an increase in agriculture. Soil erosion is a costly worldwide problem resulting in the diffuse pollution of waterbodies, sedimentation of water courses, desertification (especially in arid and semi-arid areas) and loss of productive land. UK annual costs of soil erosion currently stand at an estimated £90 million (EA 2004; Morgan 2005). This has led countries world wide to adopt policies to aid in the protection of this vital resource. Europe adopted the Sixth Environment Action Programme in 2002, which will run until 2012 by which, time the European Commission will have prepared seven Thematic Strategies - one of which is solely concerned with the protection and conservation of soils. As part of this soil thematic strategy a framework directive was proposed. Contained in which, were principles allowing member states to utilise soil resources in a sustainable and protective manner (EUROPA 2006c)

Agriculture has a direct impact on the soil environment, altering soil structure, organic matter, water content, nutrient composition and the soil biota. These changes in soil properties and soil surface characteristics affect losses of soil, water, nutrients and carbon through erosion. Traditional soil cultivation (conventional tillage) inverts the soil using tillage practices such as mouldboard ploughing, and the creation of fine seed beds for crop drilling, often resulting in soil erosion and runoff. The use of conventional agricultural practices such as continuous tillage, crop residue incorporation, burning, and intensive preparation of the seed bed have lead to a decline in soil fertility, loss of available water to the crop and a reduction in biodiversity (Benites & Vaneph 2001). In extreme cases, desertification can occur as happened in the mid West of the United States, with the creation of the dustbowl, starting a drive to adopt conservation agriculture. Other countries also affected by desertification are Brazil and Australia (Benites & Vaneph 2001). This has led to a recent shift towards practicing conservation tillage, which aims to minimise soil disturbance, and thus reduce erosion. Conservation based agricultural practices aim to minimise soil erosion, and where possible maintain or improve soil properties e.g. texture, organic matter, moisture content, bulk density and the soil biota. Conservation agriculture aims to improve soils surface conditions by increasing surface cover and surface roughness.

Since 1990, in the US, the adoption of no-till conservation systems has increased by over 25% (Hill & Mannering 1995) and is increasing each year. A case study in Worcestershire on heavy clay loam showed a move from plough and power/harrow-drill to a reduced cultivation/direct drilling. The 320ha farm showed soil structure improvements, significant worm number increase and improvement in soil fertility. For a farmer this is positive, but there are financial implications in adopting conservation agriculture, as new equipment is required, new techniques to be learned and there is a fear that crop productivity would be reduced. This case study showed that as well as soil improvements overall crop establishment cost was reduced by over 45% (SMI 2005a).

A further concern of farmers implementing conservation agriculture is the risk of pest resistance and damage to crops. Weed and pest problems were controlled in conventional agriculture by mechanical means; breaking the soil and inverting it, killing pests, breaking weed roots and burying weed seeds. As conservation agriculture minimises tillage practices, farmers were concerned about having to apply increased amounts of herbicides at additional costs. However, this is not always the case, as a study in Wiltshire showed that a move towards reduced cultivation can still control herbicide resistant black grass and slugs. This study found that ploughing had not controlled black grass but increased the problem by rotating the seeds annually. The use of reduced cultivation improved crop emergence and slug activity. As with the previous case study there was an unexpectedly low crop establishment cost (SMI 2005b). This could be set against any costs incurred for additional herbicide applications, if required.

Any move towards wider adoption of conservation regimes needs to be supported by fundamental research as to the effectiveness of these practices on processes such as soil erosion and erodibility. This can be done through a variety of ways including direct measurement or empirical or physical based models. The present study uses direct measurements to assess the impact of conservation and conventional practices on soil erosion and erodibility at a variety of spatial scales in relation to a variety of soil properties and soil surface characteristics.

1.3 Aim and Objectives

1.3.1 Aim

The aim of this thesis is to investigate the impact of conventional and conservation soil management practices on runoff, soil, nutrient and carbon losses, aggregate stability, inherent soil properties and surface characteristics across different spatial scales, at two sites in the UK.

1.3.2 Objectives

The following objectives were set to address research gaps as identified in the literature review.

I To assess whether observed runoff volume and soil loss from a conventional soil management treatment differed from conservation based practices. This assessment will be made at a field and a micro-plot scale, and explanation of results will be made with regard to relevant environmental variables.

II To analyse the losses of nutrients and carbon associated with surface runoff and eroded sediment at the field scale, identifying and explaining any differences that may exist between conventional and conservation based soil management.

III To identify if differences in soil erodibility exist between conventional and conservation practices by testing the stability of surface aggregates obtained from each treatment, using three different methodologies. The results will be evaluated critically to assess if the technique employed affects the results obtained.

IV To evaluate whether field scale erosion losses can be extrapolated from the micro-plot scale, and whether relative treatment rank in terms of soil loss is consistent for all 3 spatial scales (field, micro-plot and soil aggregate).

1.4 Thesis structure

The proceeding chapters will address the following:

Chapter 2 introduces the methodologies employed to meet the four research objectives. This chapter includes site locations and layouts, soil management treatment descriptions and an overview of the methods used at each spatial scale.

Chapter 3 presents the research carried out at the field plot scale, measuring runoff, sediment, nutrient and carbon losses. The outcome of this chapter addresses the questions set out in objectives I and II.

Chapter 4 presents the background and results from the micro-plot scale where runoff volume and soil loss were measured and percentages of surface seals and ponds were also quantified. The results of this chapter form part of objective I.

Chapter 5 addresses the small scale assessment of soil erodibility through quantification of aggregate stability. The background and specific details of the methods involved in each technique are presented. The results of this chapter refer to objective III.

Chapter 6 focuses on objective IV, in the integration of results from each spatial scale (using the results presented in chapters 3-5).

Finally, Chapter 7 presents the conclusions from this research project, reflecting the overall findings from each of the previous chapters. The limitations and implications of this project are presented, as well as future research directions. The findings from the present research are put into context with regard to soil erosion research, policy drivers and the farming community.

2 Methodology

In order to meet the previously defined aims and objectives of this research, the following methodology was employed.

2.1 Experimental site design

The site locations and soil management treatments were pre-determined by the pan-European demonstration project, SOWAP. The following section describes the location and layout of each site, and the treatments employed.

2.1.1 Site location

Two locations in the UK were chosen to undertake this study. The first site was situated in central England, on the Loddington estate in Leicestershire, spanning over 300 hectares of arable cropland owned and managed by the Allerton Research and Educational Trust[1], established in 1992. The second site is in Somerset, in the south west of England. The farm is located at Tivington, managed by a local farmer and set within the Holnicote Estate covering over 5,000 hectares of the Exmoor National Park. The locations of these sites in relation to the rest of the UK can be found in Figure 2.1-1.

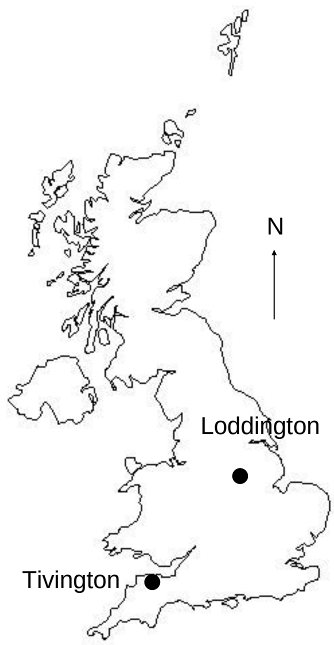


Figure 2.1-1 UK site location

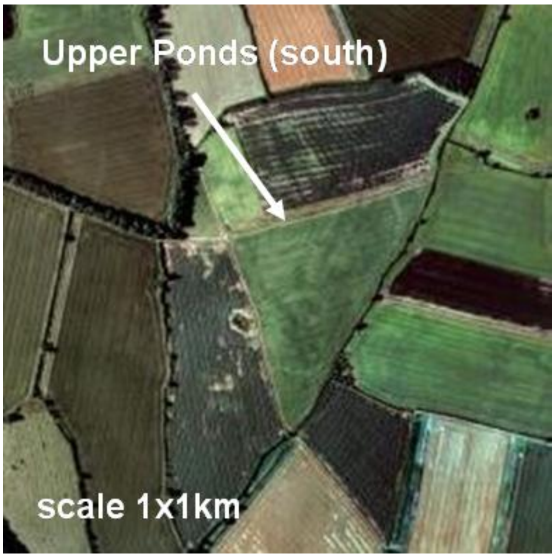
At Loddington, the experimental site is located on Upper Pond Field (south), which is approximately 8.5 hectares in size (Plate 2.1-1). The field elevation is between 135 and 140 metres. The grid reference of the field is SK7901. The field has a mean slope of 3.5%, measured using a clinometer. The experimental site at

[1] More information can be found at their website http://www.allertontrust.org.uk

Tivington is located on Pit Field, which is around 3.7 hectares (Plate 2.1-2). The field elevation is between 90 and 100 metres. The grid reference is SS9344. The field has a mean slope of 7% as measured by a clinometer. Aerial photographs of both site locations can be found in Figure 2.1-2.

Plate 2.1-1 Loddington field site

Plate 2.1-2 Tivington field site



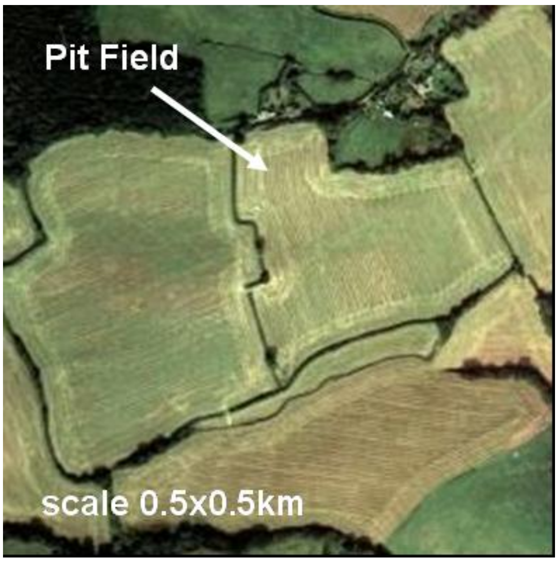


Figure 2.1-2 Aerial photographs of Loddington (left) and Tivington (right) field sites

Baseline soil samples were taken before the experimental site was constructed or treatments applied; the baseline results are presented in Table 2.1-1. The soil at Loddington is part of the Hanslope and Denchworth clay series, and a soil profile can be found in Figure 2.1-3. Hanslope soils are calcareous pelosols developed in Chalky-Jurassic Boulder clay. The topsoil consists of slightly calcareous brown clay, slowly permeable but is seldom waterlogged. The subsoil is chalky and calcareous, slightly mottled, which increases with depth. It is naturally compact with a poor structure remaining wet over winter. This soil series is associated with the East England. Denchworth soils are pelo stagnogleys developed over Jurassic and Cretaceous clay. The topsoil consists of stoneless, mottled dark brown heavy clay with a greyish, stoneless clay subsoil. The soil at Tivington is part of the Worcester series, a clayey soil, reddish colour with moderate permeability and free to imperfect drainage with a minimal risk of erosion by water (Hodgson 1997). This soil was formed over Triassic marl (calcium carbonate mudstone) and clay shale. The main differences between the sites in terms of soil texture is that the Loddington site (calssified as a clay soil) has over 20% more clay than the soil from the Tivington site (classified as a sandy clay loam). However, the former contains over 4% more organic matter.

Table 2.1-1 Baseline soil properties

Soil Parameter	Loddington	Tivington
Organic Matter (%)	5.2	0.84
Sand 2.00-0.063mm (%)	32	53.8
Silt 0.063-0.002mm (%)	23.5	26.6
Clay <0.002mm (%)	44.5	19.6
Textural Class	Clay	Sandy clay loam

Figure 2.1-3 Baseline soil profile at Loddington. Source www.sowap.org

2.1.2 Soil Management Treatments

Three soil management treatments were employed in this study - two conservation oriented treatments and one conventional treatment. The conservation treatments are based on non-inversion, conservation tillage; more specifically this tillage practice does not use a mouldboard plough. Instead conservation tillage uses many different techniques aimed at reducing mechanical manipulation of the soil ranging from heavy discs to zero-tillage. The two types of conservation treatments reflect the best management practice as defined firstly by the parent project SOWAP (SOWAP or S), and secondly by the farmer at each site (Farmer's Preference or F). Both conservation treatments applied surface residues. The conventional treatment (C) utilised the traditional mould board plough, inverting the soil, applying both primary and secondary cultivations. No surface residues were applied as part of this treatment. More detailed information on the specific operations involved for each treatment can be found for Loddington and Tivington in Table 2.1-2 and Table 2.1-3, respectively.

Table 2.1-2 Loddington treatment descriptions

Season & Dates	Crop	Treatment	Tillage
Pre-season one	Mustard	SOWAP	Cover crop drilled to 2.5-4cm depth
Season One October 2003 to September 2004	Winter Wheat – Solstice variety	Conventional	Plough to a depth of 15-20cm Cultipress to 5-10cm depth Drill to a depth of 2.5-4cm Roll
		SOWAP	Simba Solo to 12.5-20cm depth Cultipress to 5-10cm depth Drill to 2.5-4cm depth Roll
		Farmer's Preference	Simba Solo to 12.5-20cm depth Cultipress to 5-10cm depth Drill to 2.5-4cm depth Roll
Season Two September 2004 to March 2005		Conventional	Soil left from previous harvest with complete surface cover
		SOWAP	Simba Solo to 12.5-20cm depth Drilled Mustard-rye cover crop 2.5-4cm depth
		Farmer's Preference	Simba Solo to 12.5-20cm depth
Season Three March 2005 to September 2005	Spring Beans - Quatro variety	Conventional	Plough to a depth of 15-20cm Power harrow to 10-15cm depth Drill to a depth of 2.5-4cm
		SOWAP	Drill to 2.5-4cm
		Farmer's Preference	Power harrow to 10-15cm depth Drill 2.5-4cm depth
Season Four October 2005 to July 2006	Winter Wheat – Solstice variety	Conventional	Plough to a depth of 15-20cm Cultipress to 5-10cm depth Another Cultipress to 5-10cm depth Drill to a depth of 2.5-4cm Roll

		SOWAP	Subsoil to a depth of 35-45cm depth Cultipress to 5-10cm depth Drill to 2.5-4cm Roll
		Farmer's Preference	Subsoil to a depth of 35-45cm Cultipress to 5-10cm depth Drill to 2.5-4 cm depth Roll

Table 2.1-3 Tivington treatment descriptions

Season	**Crop**	**Treatment**	**Tillage**
One August 2003 to July 2004	Winter Oil Seed Rape – variety Winner	Conventional	Plough to a depth of 15-18cm Vaderstad Carrier to a depth of 2.5cm Drill to a depth of 2cm Roll
		SOWAP	Vaderstad Carrier to a depth of 2.5cm Drill to 0.5cm depth Roll
		Farmer's Preference	Subsoil to a depth of 20-30cm Drill to 0.5cm depth Roll
Two September 2004 to August 2005	Winter Wheat – variety Claire	Conventional	Plough to a depth of 15-20cm Vaderstad Carrier to 2.5-3.5cm depth Drill to a depth of 2cm Roll
		SOWAP	Subsoil to a depth of 20cm Drill to a depth of 2cm Roll
		Farmer's Preference	Vaderstad Carrier to 2.5-3.5cm Repeated again one month later Drill to 2cm depth Roll
Three November 2005 to August 2006	Beans – variety Wizard	Conventional	Plough to a depth of 15-18cm Subsoil to 25cm depth Drill to a depth of 7-8cm
		SOWAP	Subsoil to a depth of 25cm Drill to 7-8cm
		Farmer's Preference	Subsoil to a depth of 25cm Drill to 7-8cm

2.1.3 Site Layout

At both sites, the three treatments were installed adjacent to one another; the conventional treatment being placed at one side, the SOWAP conservation treatment in the middle and the other conservation treatment, Farmer's

Preference to the other side. For each treatment, land was set aside for two, duplicate erosion plots, the rainfall simulation trial plots, soil sampling and terrestrial ecology sampling (the latter being outside the scope of the present study). At Loddington, each erosion plot was 70 metres in length and 9 metres wide, encompassing an area of 630m^2. Above each erosion plot an area of 90m^2 was set aside for soil sampling and the rainfall simulation plots. The layout of this area can be seen in Figure 2.1-4. The boundaries of the erosion plots (see Erosion Plot Layout Section 2.1.5) were removed before any field operations (e.g. harvesting) were undertaken and reinstalled after.

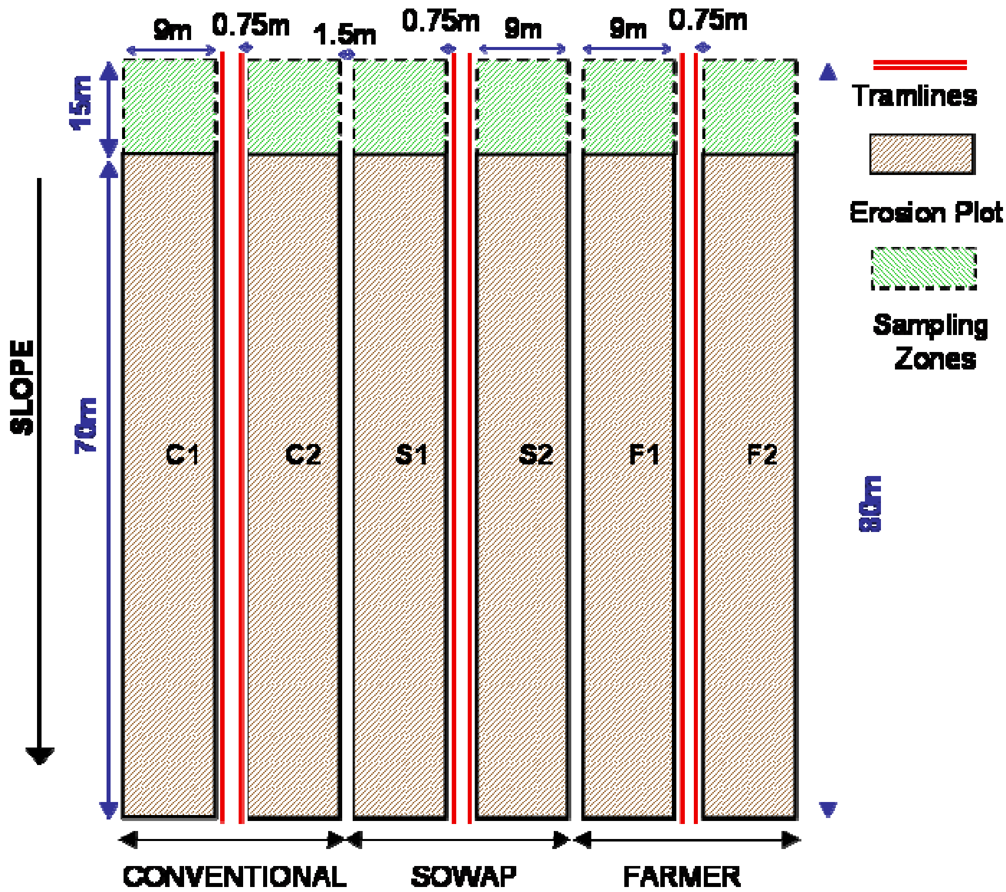


Figure 2.1-4 Loddington site layout

At Tivington the erosion plots were 55 metres in length and 10 metres wide, covering an area of 550m^2. However, the first replicate for the conventional treatment (C1) varied in size for each season, depending on where the farmer placed the tramlines and plot location relative to the field boundary (Figure 2.1-5). During season 1 the plot area was 500m^2, season 2 it was 455m^2 and in season 3 it was 461.5m^2. For each treatment there was a piece of land of equal size to the erosion plots located parallel with them. This land was left for the soil sampling and rainfall simulation plots. The layout of this area can be seen in Figure 2.1-5. The boundaries of the erosion plots (see Erosion Plot Layout Section 2.1.5) were removed before any field operations were undertaken and reinstalled after.

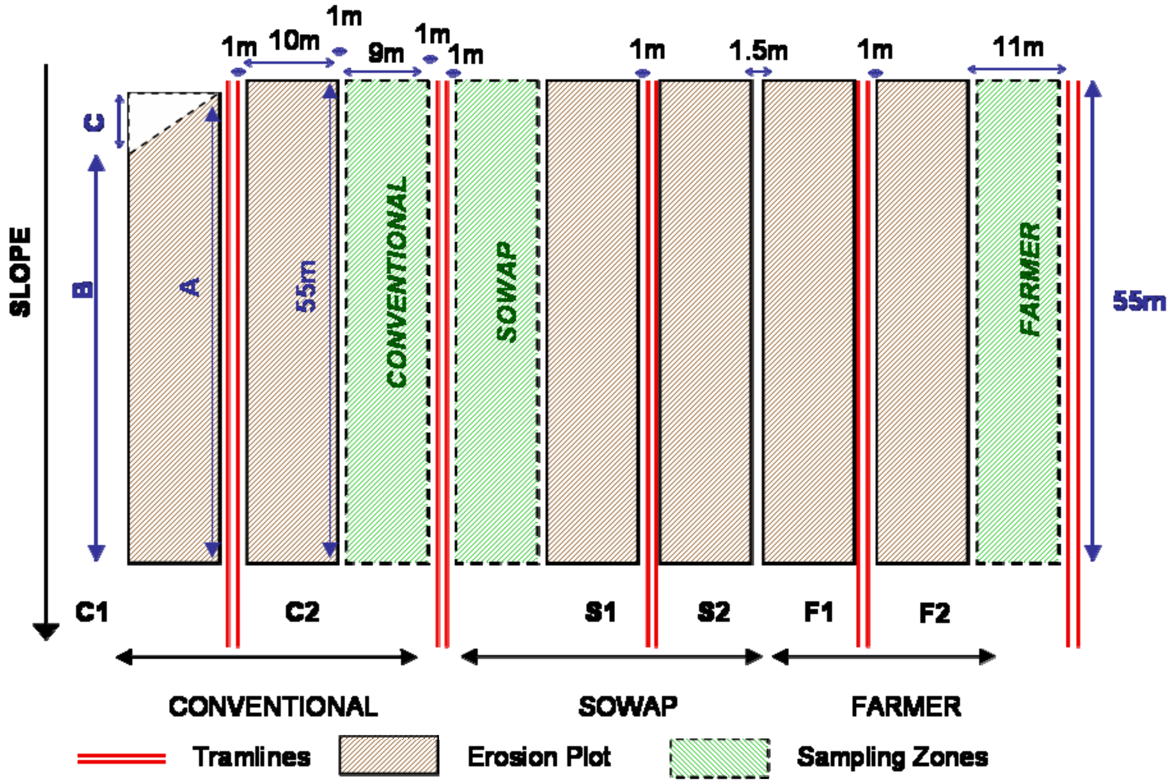


Figure 2.1-5 Tivington site layout

Weather stations at both sites recorded a variety of meteorological parameters, including daily precipitation (mm), wind speed (m s^{-1}), wind direction (i.e. NSEW), solar flux energy (W m^{-2}), air temperature (°C), soil temperature (°C), and relative humidity (%). These parameters were recorded every 5 minutes throughout the day, and were used in other studies within the SOWAP project, from soil microbiology to avian ecology. The primary parameter used in the present research was precipitation. The information was recorded, relayed and updated daily onto the SOWAP website (www.sowap.org), allowing remote access. The weather station at Loddington can be seen in Plate 2.1-3.

Plate 2.1-3 Loddington weather station

2.1.4 Crop Regime

Different cropping regimes were used at the two sites, thus direct comparison of results between sites was not possible. However, for each site, in any given season, the same crop was grown on all treatments. Details of the cropping management can be found for Loddington (Table 2.1-4) and Tivington (Table 2.1-5).

Table 2.1-4 Crop management at Loddington

Season	Crop Regime	Crop Dates	Bunding Dates		No. of tank clearances
1	Winter Wheat	September 2003 to September 2004	IN OUT	22/3/04 18/6/04	1
2	No crop sown C: left as stubble for late ploughing * S: tilled and cover crop sown F: ploughed and residue left	September 2004 to March 2005	IN OUT	27/09/04 9/3/05	4
3	Spring Beans	March to August 2005	IN OUT	22/5/05 7/8/05	2
4	Winter Wheat	September 2005 to July 2006	IN OUT	2/11/05 21/7/06	4

IN = installation and OUT = removal of the erosion plot boundaries. C, S and F represent the different treatments. * not typical of this treatment, but bad weather forces late ploughing

Table 2.1-5 Crop management at Tivington

Season	Crop Regime	Crop Dates	Bunding Dates		No. of tank clearances
1	Winter Oil Seed Rape	September 2003 to August 2004	IN OUT	23/3/04 20/5/04	2
2	Winter Wheat	September 2004 to July 2005	IN OUT	13/9/04 15/7/05	8
3	Winter Beans	November 2005 to July 2006	IN OUT	6/12/05 4/8/06	9

2.1.5 Erosion Plot Layout

Each erosion plot was 9-10 metres wide and between 45 and 70 metres in length, depending on the site location (section 2.1.3). This gave a plot size of approximately 0.05 of a hectare (Plate 2.1-4).

Plate 2.1-4 Erosion plots (Loddington)

The plots were delineated by sheets of galvanised metal, (each measuring 2.5m long, 0.3m wide and 1mm thick) which were overlapped, clipped and hammered into the ground to a depth of at least 15cm (Plate 2.1-5) to limit lateral sub-surface flow in and out of the plot area. At the top of the erosion plots, the metal sheets were bent into right-angles for the corners. At the bottom edge of each plot the metal sheets were angled across the slope towards large funnels made of galvanised metal (2m wide opening). A layout of the funnels can be found in Appendix A. These funnels directed any runoff and soil loss generated on the plots, through plastic piping (15cm diameter) and into the system of collecting tanks (Plate 2.1-6). Wire mesh was placed on the face of the collection funnels to prevent small animals and debris entering the collection tanks and contaminating the collected runoff and sediment.

Plate 2.1-5 Installation of erosion plots

Plate 2.1-6 Funnel and pipes

The collection system (Plate 2.1-7) consisted of a primary collection tank, which received all runoff and sediment generated from the erosion plots. Once this tank was full, a slot device at the top of the tank's backwall allowed 1/9th of the overflow to spill into the 2nd tank, which was located down slope of the first tank. A 3rd overspill tank (receiving 1/9th of the overspill from tank 2) was installed for each plot at Tivington, because of the greater rainfall and runoff predicted for this site.

The primary tank for each plot (2 plots per treatment) contained a floating sensor which was connected to the on-site weather station (Plate 2.1-3). Runoff levels were recorded and could be accessed remotely. As previously stated, other site parameters were recorded by the weather station, including air and soil temperature, wind direction, sunlight, and most importantly for erosion studies, rainfall depth. This system was updated every 5 minutes and could be accessed remotely. The tanks were set within a large reinforced pit below ground level, which had its own drainage system to a local water course to prevent flooding during heavy rainfall and tank emptying operations (section 3.1 Plate 3.1-2). Each tank had a lid to prevent direct rainfall entering the tank, and to limit evaporation of the retained runoff. The lids were removable for the purpose of tank clearing (Plate 2.1-8). At the base and to the side of each tank was an outlet pipe used for tank emptying and cleaning.

Plate 2.1-7 Collection system

Plate 2.1-8 Inflow pipe, collection tank and tank lid

The tanks were emptied as soon as logistically possible after a runoff event had been recorded by the tank sensors, and relayed to the SOWAP website. During periods of intense rainfall multiple rainfall events may have occurred before a collection could be made. Hence, tanks were not necessarily emptied on an event basis. Measurements of the runoff and sediment included total runoff depth (mm) and volume (l), total soil loss (kg), and associated with these were nutrient (NPK) and carbon concentration and loadings. A list of the analysis undertaken can be found in Appendix B.

2.2 Statistical Analysis

All results in this study were analysed using the statistical computer package, Statistica version 7.0 (StatSoft Inc, Bedford, UK). All data analysis compared treatment means across different temporal scales; a list for each chapter is summarised in Table 2.2-1. At each spatial scale simple correlations were performed using data from the entire sampling period, between the runoff and sediment data with soil properties and surface characteristics.

Table 2.2-1 Summary of temporal analysis performed on treatment data

Chapter	Temporal scale
Three: field erosion plots	overall, season, tank clearance
Four: micro-plot rainfall simulations	overall, season, simulation trial
Five: aggregate stability	overall, season, simulation trial

All analysis was performed using an analysis of variance (ANOVA) and subsequent post hoc comparison through a Fisher LSD test. To satisfy the assumptions made by ANOVA, all data were tested for normality (Skewness and Kurtosis) and equal variance (Levene's test for homogeneity of variance). If data were identified as having a non-normal distribution and/or unequal variance then mathematical transformations were carried out. Initially all types of transformations were applied and are listed in Table 2.2-2 and the one that gave the greatest normality and equal variance was chosen.

Table 2.2-2 Types of transformation

Reference	Transformation
Type I	Square root
Type II	Natural log plus one
Type III	Natural log
Type IV	Inverse of the square root plus one

Data that had undergone transformation were then back-transformed (i.e. the mean outputs were then calculated with the inverse to the original transformation) Where transformation was necessary the back-transformed means are presented within the text. Due to some data sets having unequal replications, pooled standard errors could not be calculated. Therefore instead of error bars, or numerical notation, letters have been used to denote significant difference (the same letters are statistically the same). All tests were performed at 95% confidence.

3 Plot scale erosion assessment

3.1 Introduction

Hudson (1957) claims that field erosion plots probably give the most reliable results on soil loss per unit area. They are used widely in erosion research for the collection of runoff and sediment (and associated losses of nutrients, agro-chemicals or soil carbon etc.), or for the study of specific erosion features, such as rills. The way in which field erosion plots are used is dependent on their size. Although not applicable to all research Stoosnijder (2005) summarised the different sizes of erosion plot used for various studies. According to Stoosnijder's definition, the plots being used in the current study fall between hill slope sized plots (<500m^2) and field plots (<1ha), which Stoosnijder argues are appropriate for sediment deposition and channel studies, respectively (Table 3.1-1). However, Hudson (1995) would class the plots used within this study as being small-scale plots rather than field plots, which he described as approximately one hectare in size.

Table 3.1-1 Spatial scales as defined by Stoosnijder (2005)

Spatial scale	Size
The point scale for interrill (splash) erosion	1 square metre
The plot for rill erosion	100 square metres
The hillslope for sediment deposition	500 square metres
The field for channels	1 hectare

There are advantages and disadvantages associated with the use of erosion plots, and it is important to understand these fully before developing a research methodology based on field erosion plot results. When using erosion plots of a large size (>100m^2) it becomes difficult to simulate realistic, reliable and uniform rainfall, so runoff and soil loss generation tend to rely on natural rainfall events. Whilst this reflects the natural field conditions it also makes any data generated effectively non replicable. Soil loss and runoff are often only generated by larger rainfall events. Hudson (1995) states that erosive rainfall only occurs >25mm h^{-1}, whereas Morgan (2005) estimated this to be >10 mm h^{-1} for temperate climates. The unpredictability of natural rainfall can also cause problems with regard to staffing the plots for runoff and sediment collection (Hudson 1995).

Field erosion plots consist of various components. Runoff and sediment generated on each plot runs into the collection system located at the downslope edge of the plot. Research projects have used a variety of collection systems, such as automated flumes and large collection tanks, capable of collecting all runoff and soil generated on the plot. Hudson (1993) has shown some of the problems caused by ill conceived collection systems. These have included insufficient tank storage, leading to runoff overflowing and the collection tanks floating away during a heavy storm. The collection system used in this study comprises of a sequence of collection tanks – 2 for each plot at Loddington (Plate 3.1-1), and 3 for each plot at Tivington. The runoff and soil loss generated on a plot is directed into the first tank. Once this tank is full, a slot device at the top of the tank's backwall allows 1/9th of the overflow to spill into the 2nd tank, which is located down slope of the first tank. A 3rd overspill tank (receiving 1/9th of the overspill from tank 2) was installed for each plot at Tivington because of the greater rainfall and runoff predicted for this site. The tanks are set within a large reinforced pit below ground level, which has its own drainage system to a local water course to prevent flooding during heavy rainfall and tank emptying (Plate 3.1-2).

Another very important aspect of field erosion plots is whether they should be bounded or unbounded. There is no standard, as to whether or not a plot should be bounded, and if so what material should be used. Unbounded plots have the advantage that they do not incur "edge effects" (see below) and normal farming practices can be undertaken across the plots, with no additional staffing required to remove or reinstall the boundary material. Unbounded plots represented an “open system”, which allows natural functioning of hydrological pathways and allows the plots to be replenished with eroded soil from up slope. However, the natural variability of slope morphology in the field means that unbounded plots (as determined by the natural topography), cannot be replicated in terms of space. Also, tillage operations may change the slope topography and thus alter the catchment area of each plot.

Plate 3.1-1 Tank layout at Loddington **Plate 3.1-2 Pit drainage channel (Tivington)**

Alternatively, field erosion plots are often bounded (by compacted earth bunds, sheet metal, plastic strips etc. – see below) to demarcate precisely the catchment area contributing the runoff and sediment collected at the downslope edge of the plot. These physical plot boundaries may however, interfere with the erosion processes being observed (known as “boundary” or “edge” effects). For example, a compacted earth bund may contribute more runoff to the plot area than would occur naturally. Metal sheets inserted into the soil as plot boundaries may create preferential pathways for vertical flow so reducing surface runoff. Hudson (1995) suggested that to minimise boundary effects, field erosion plot size should be at least 5 m wide and 20 m long. However, the standard plot size used for the USLE plots of Wischmeier & Smith (1978) is stated as being 1.8m wide and 22m long (Morgan 2005). This indicates that there is still confusion and uncertainty as to the appropriate size of plot to use.

The size of field erosion plots used in research is varied; with many studies using the USLE defined size of 1.8m wide by 22m wide (Basic et al. 2004; Hartanto et al. 2003; Romero-Diaz et al. 1999) but just as many using their own defined sizes. A few examples are shown of the different sizes used in research on soil management practice effect on runoff and soil loss. The eight experimental plots used in the research carried out at Rothamsted, Woburn, were 24 by 36 metres or 0.086ha (Rickson 1994; Quinton et al. 2006), where two contrasting soil management treatments were investigated. In work by Andraski et al. (1985) where four tillage treatments were compared (3 conservation based, 1 conventional), field erosion plots were installed at 4.6m wide and 22.1m long. Research carried out by Boix-Fayos et al. (in press) used erosion plots sized 15m by 5m and 10m by 3m. Ollesch & Vacca (2002) investigating different land use on three different hill slopes, used 18 field erosion plots of 20m^2; 10m long by 2m wide. Williams (2004) studied field hydrology under five different soil management treatments; these were duplicated, equalling 10 plots, each measuring 12m by 40m (0.05ha).

The field erosion plots being used in this study are of sufficient size to overcome any boundary effects. The plot sizes differ slightly between sites; at Loddington the mean plot size was 70m long and 9m wide and at Tivington 55m long and 10m wide. It is assumed that these are effectively the same spatial magnitude, and that the same scale of erosion process is operating at both sites. The plot dimensions were chosen to maximise size but still allow duplication per treatment within the field space available.

Another problem of bounded plots is that the plot becomes depleted of fresh soil. As soil erodes within a bounded plot and is collected, there is no new soil from upslope to replace it. Over time there is a risk that the measured erosion rates do not reflect the natural system as erosion rates decrease with time due to less available soil to erode. In work carried out by Boix-Fayos et al. (in press) the period of soil exhaustion was between four to seven years. This period is dependent on the surface characteristics, the local climatic conditions, size of the field erosion plots (Boix-Fayos et al. in press), depth of soil, and erosion / formation rate of soil. The risk of exhaustion or depletion within this study is not a factor, due to the sufficient depth of topsoil, the relatively low erosion rates experienced in the UK, and the fact that local replenishment of soil by natural processes is possible when the plot boundaries are removed each season to allow tillage operations.

The benefit of using bounded plots is that runoff and soil loss can be measured over a given area which allows losses to be compared with other treatments within one study, or with the results from other studies. There is no set way in which a field plot should be bounded. Work undertaken at the Rothamsted Experimental Station at Woburn for example (Rickson 1994) used grassed earth bunds (1m wide and 30cm high) to demarcate the erosion plot area. These had the advantage that farming machinery could cross them during field operations. Earth bunds have also been used in work by Barton et al. (2004) and Quinton et al. (2006). In the experiments carried out by Hudson (1957) field erosion plots were bounded by asbestos planks, which were removed and reinstalled to allow farming operations. As well as the hazardous health and safety aspects of this material, these planks could not be sunk into the ground to limit water movement in and out of the plots. Other materials and ways of bounding plots include wood (Hartanto et al. 2003), brick or concrete (Correchel et al. 2006), sheet metal (Basic et al. 2004; Burney & Edwards 1994; Hartanto et al. 2003; Ollesch & Vacca 2002) and drains/ditches (Romero-Díaz et al. 1999; Williams 2004). In this study, galvanised metal sheets (2.5m long, 0.31m wide and 1mm thick) were inserted into the soil to a depth of at least 15cm to limit lateral water movement and define plot catchment area (Plate 3.1-3). The sheets were removed to allow for field operations, including drilling, spraying and harvesting, and then reinstated once the operations were completed.

Plate 3.1-3 Bunding installation. Source www.sowap.org

As with many field experiments, data errors and variability are associated with field erosion plots and must be taken into account. Replication is therefore important. Where erosion plots are used for demonstration purposes only, generally only one plot is used per treatment (Hudson 1995). For more rigorous, scientific research it is advised that at the very least two replicates should be used (Morgan 2005). In this study two duplicates were installed per treatment at both UK sites. The limited number of replicates is a concern for this study, but is justified by the fact that a) the SOWAP Project is primarily a field-scale demonstration project (see Hudson's comment above), and b) increasing the number of replicates would reduce individual plot size, given the finite space available for the field experimental layout. The consequence of this is that the plots would not be large enough to be representative of the field scale, and boundary effects would be more significant (see above) on the smaller plots.

Variation between replicate plots is common in erosion research (Boix-Fayos et al. in press &. 2006). Work by Nearing et al. (1999) investigated results from replicated plot pairs for over 2000 storm events under a variety of soil conditions and site locations. This work showed that the variation of soil loss between replicates reduced as the magnitude of measured soil loss increased. A variation of just under 15% was found for a soil loss of 20 kg ha^{-1}, compared to 150% variation for soil loss less than 0.01 kg ha^{-1}. This variability between replicates may be the result of human error during sampling and measuring, especially in the assessment of sediment concentration (Zobisch et al. 1996), as well as the natural field variation in the erosion plots themselves. Zobisch et al. (1996) studied the accuracy of runoff and soil loss measurements using a field collection system in the laboratory with 5 different members of staff. This work showed that human error contributed to the variation in measurements of runoff and soil lost from field erosion plots, with no automated sampling systems. Despite these studies, the variability between replicate erosion plots is not completely understood (Gómez et al. 2001).

In research involving replicate field erosion plots, it is often assumed that the replicates are from the same population. In reality there are always inherent differences between plots (Nearing et al. 1999) due to natural fluctuations in the underlying parent material, slope, depressions, water table, and soil texture. These are compounded by human induced differences due to inconsistencies in sampling technique and/or farming operations. An analysis of the variation of soil loss among 40 identical replicated field erosion plots by Wendt et al. (1986), suggested that only a small proportion of the observed variation could be explained by measured soil properties. Using the same field erosion plot data and combining this with a numerical model, Gómez et al. (2001) suggested that 50% of the variation in runoff was attributable to the hydraulic conductivity, depth to clay pan and surface storage. Although not stated, the remaining 50% of variation could be due to human induced error and unaccountable variability related to spatial differences in field properties/condition. The research undertaken above highlights the inherent variability associated with field erosion plot studies. This is an important consideration, when embarking on erosion assessment using field plots.

In brief, rainfall variability, plot size, method of bounding, number of replicates and inherent site variability are all important factors that contribute to the variation and reliability of scientific data generated on field erosion plots. In this study, all of the above factors have been taken into account in an attempt to minimise error and gain a better understanding of soil erosion processes and severity at a field scale.

3.2 Aim, Objectives and Hypotheses

3.2.1 Aim

The aim of this chapter is to investigate whether different soil management practices can improve the conservation of soil and water, and minimise losses of nutrients and carbon in runoff and sediment. The practices under investigation are a conventional tillage treatment and two forms of conservation tillage treatment (2.1.2). The treatments were applied at two sites in the UK (Somerset and Leicestershire).

3.2.2 Objectives

The aims of the chapter will be met by quantifying and analysing runoff volume and soil loss under natural rainfall for three different soil management treatments – conventional tillage, and two forms of conservation tillage (SOWAP and Farmer's Preference). Nutrient and carbon losses associated with the runoff and sediment are also quantified for the three different soil management treatments. Explanations for the observed results will be given, on the basis of supporting field evidence.

3.2.3 Hypotheses

3.2.3.1 Hypothesis One

It is expected that runoff volume and the amount of soil lost from the field plots will be highest from the conventional tillage treatment when compared to the conservation tillage treatments (SOWAP and Farmer's Preference). All other factors being equal (rainfall, slope gradient and length, etc.), runoff volume is influenced by the properties of the soil and the land surface, which in turn are affected by the soil management treatments.

In general, conventionally tilled soil will have primary and secondary cultivations, compared to just one cultivation operation on conservation treatments. The increased number of field operations on the conventional treatment will increase mechanical breakdown of soil aggregates, leading to soil compaction and therefore increased bulk density. Work from Smith (1987), has shown that dry bulk densities increase after the passage of vehicle wheels. This increased bulk density is the result of consolidation of the soil matrix, resulting in impeded drainage and infiltration rates, leading to increased runoff volume at the surface.

The use of primary cultivation for the conventional tillage treatment also means that the soil is inverted, and an increase in the mechanical manipulation of the soil. This has dramatic effects on a variety of soil properties. Cutting through the soil with a plough (used in primary cultivation) mechanically breaks established pathways of preferential flow within the soil profile. The numbers of earthworms in particular deep burrowing species (anecic) and other organisms such as mites can be reduced (Brady & Weil 2002; Chan 2001). These larger organisms have multiple effects on the soil, one of which is the effect on infiltration rates. Earthworms in particular create preferential pathways of flow by burrowing through the soil profile. A reduction in infiltration rates can lead to increased volumes of overland flow. Earthworms have also been found to influence soil fertility and productivity (Brady & Weil 2002). The soil biota also increases aggregate formation via burrowing action from plant roots and organisms such as earthworms, and through biotic stabilising agents (Brady & Weil 2002; Stuttard 1985) i.e. organic exudates and physical action through sticky root hairs and fungal hyphae. The presence of soil organisms leads to increase aggregate stabilisation, consequentially reducing soil erodibility.

Organic soil aggregate connectors such as plant roots and fungal hyphae are destroyed by ploughing, as well as the alteration of microbial communities. Parallel to this study, the SOWAP project is also studying the effect of tillage on the microbial community (Allton 2006). This alteration in the soil biota can affect soil properties adversely, such as soil structure, nutrient cycling, and soil aggregate stability, as discussed above (Brady & Weil 2002; Kladivko 2001). As organic cements and organic connectors (plant roots and fungal hyphae) are destroyed by the increased soil mechanical manipulation and inversion, aggregate stability is reduced. Soil surface caps and seals form to a greater extent, reducing infiltration rates (Robinson & Phillips 2001); soil loss and runoff will also increase. Where inversion and increased mechanical manipulation of the soil occur (as with conventional tillage), it is expected that soil loss and runoff will be greater, compared to a conservation treatment where tillage operations involve no inversion and less mechanical manipulation of the soil.

The soil surface is also expected to differ between treatments. The omission of primary cultivation on the conservation treatment means that the surface soil has not been inverted, and there has been less mechanical manipulation and breakdown during a secondary cultivation operation. As a result surface aggregates from the conservation treatment have a wider range of aggregate sizes and a rougher soil surface. An increase in surface roughness can increase infiltration rates as there is a larger surface area over which infiltration can occur. Increase in surface roughness also promotes the formation of surface ponds which occur when water is prevented from flowing either vertically or horizontally. Surface ponds form as rainfall input becomes greater than infiltration rates, as may occur where the surface soil has become capped or sealed. The rougher the soil surface, the longer the residence time of runoff on the surface or within a pond, and flow velocity is reduced. Sediment entrained within the surface runoff falls out of suspension and settles on the soil surface, thereby reducing the amount of soil transported by the flow. Eventually the pond may completely fill with deposited sediment and retained runoff. When this happens the ponds will overflow initiating runoff. However, the presence of a rough surface reduces the hydraulic energy of this surface water (as expressed in parameters such as Manning's n), so reducing its potential energy to detach and transport soil (Foster 1982; Abrahams & Parsons 1991; Einstein & Barbarossa 1951). Meyer & Wischmeier (1969) showed that flow detachment rates vary with the square of the runoff velocity; and flow transport rates vary with the fifth power of velocity. Hence any small reduction in runoff velocity (by surface roughness for instance) has significant effects on flow detachment and transport.

Another important difference between the two forms of tillage treatment is the application of residues or the growth of a cover crop. In conventional tillage, no residues or surface cover are present after sowing. Until the new crop is established, the soil surface is completely bare and the soil is at the highest risk of erosion (Morgan 2005). On soils with conservation tillage, residues from the previous crop or a cover crop (e.g. mustard) are present before the establishment of the main crop. The amount of cover depends on the type of cover and site-specific agronomy and soil management, but the general estimate when using residues is to cover approximately 30% of the surface (Uri 1999). The presence of a residue or cover crop affects soil erosion and runoff generation. Surface cover physically protects the soil from rainfall impact, reducing soil detachment and the subsequent processes of soil surface capping and sealing, which can lead to increased runoff generation or overland flow. In addition to this, residues can act as a physical barrier to overland flow, reducing the hydraulic energy of flow thereby encouraging deposition of any soil in suspension, lowering soil losses. This is why mulch has been shown to be successful in the reduction of soil erosion and runoff generation (Fiener et al. 2005; Lal 1976; Laflan & Colvin 1981). The impedance to flow from surface cover also increases the depth of runoff, creating a protective buffer against raindrop impact on the soil surface (Palmer, 1965). Residues also reduce soil evaporation and soil drying, which improves water availability to the crops. This leads to a healthier crop stand which is able to protect the soil from erosion and runoff generation. Minimising soil drying also has an important role in aggregate wettability and reduction in macro-pore formation. Finally, the increases in soil organic matter content associated with surface residues (Robinson & Blackman 1989) are known to increase soil biota (Six et al 2004) and improve the stability of soil aggregates (see chapter 5).

3.2.3.2 Hypothesis Two

It is expected that nutrient and carbon losses in the runoff and sediment will be less from the plots subjected to the conservation tillage treatment, compared to the conventional treatment. As stated in section 3.2.3.1 runoff volume and soil loss is expected to be higher from the conventionally tilled soil because of factors such as increased mechanical manipulation of the soil, and subsequent aggregate breakdown, reduced infiltration rates, reduction in surface ponding and sediment deposition, the lack of protective surface residues and a reduction in soil organic content and thus aggregate stability.

It is particularly important that erosion of the soil's clay fraction and organic matter should be minimised. The colloids of both clay particles and humus (derived from organic matter decay) have very large surface areas resulting in high adsorptive capacity of cations and to a lesser extent, anions. In the context of this research the presence of clay and humus colloids increases nutrient chelation. The negative charge of colloids means that cations (in this study refers to potassium) are readily adsorbed. Anions (which, in this study include nitrate and phosphate) are also adsorbed but to a lesser extent – adsorption decreasing with an increase in pH, which is the opposite for cation exchange (Brady & Weil 2002). Compared to cations, anions exist in soil solution for an increased period of time, and as a result are most at risk from leaching or loss via runoff.

The loss of nutrients and carbon in eroded sediment and runoff from agricultural fields has on- and off-site consequences. Both nutrients and carbon are vital to crop productivity and yield. As previously mentioned in (section 1.2.5.2) nitrogen is important for rapid plant growth and seed/fruit formation; phosphorus allows plants to convert light into chemical energy; potassium is essential in plant enzyme reactions and resistance to environmental stresses (drought and disease); and carbon is vital for crop productivity. Losses nutrients and carbon therefore force farmers to apply additional supplies to replenish the system.

The loss of nutrients in runoff and sediment also causes off-site impacts. Nitrate contamination of drinking water is of great concern due to of the health implications, such as the contraction of the blood condition which affects young babies called methaemoglobinaemia, alternatively known as ‘blue baby syndrome’ (DEFRA 2003). The mobilisation of nitrogen and phosphorus from agricultural sources to water bodies is one of the primary causes of eutrophication (DEFRA 2003). Eutrophication (over-enrichment of water bodies) is an environmental concern for freshwater and marine ecosystems, where phosphorus and nitrogen are the limiting nutrients, respectively (Brady & Weil 2002). Input of these nutrients into the aquatic environment can lead to the rapid utilisation of existing levels carbon and potassium within the system by aquatic biota, leading to excessive growth aquatic vegetation and phytoplankton.

3.3 Methodology

Runoff and soil loss at the plot scale were studied at two UK sites - Loddington, Leicestershire and Tivington, Somerset (section 2.1.1). At these two sites, three soil management treatments were applied; one conventional (C) tillage treatment, and two conservation tillage treatments – SOWAP (S) and Farmer’s Preference (F). More details of these treatments can be found in section 2.1.2. At each site, a demonstration field was identified in advance of the present study (summer 2003), where two erosion plots for each treatment were located. A detailed layout of the field erosion plots at each site can be found in sections 2.1.3 and 2.1.5.

3.3.1 Analysis of collected runoff and sediment

The procedure for analysing the collected runoff and sediment followed that already in use on the PROTERRA project, which studied runoff and soil loss from field erosion plots under different Mediterranean agricultural systems (Llewellyn, 2006). As soon as possible after a rainfall event (as indicated by the weather station read-outs on the SOWAP dedicated webpage[2]) the site was visited for measurements to be taken. The depths of collected runoff and of sediment in the tank were measured from the base of the tank. Several depth measurements were taken at various random points in the tank to account for any variability in runoff and sediment depth within the tank. Three 1 litre water samples were taken from the collected runoff, without any agitation of the tank contents. Once these samples had been taken the remaining runoff in the tank was either pumped or drained off, and released onto the surrounding grassed area, thereby allowing slow infiltration towards the local water course. Care was taken not to drain or pump away any of the sediment retained at the bottom of the tank. The depth of these sediments was measured at several points to calculate a mean depth of retained sediment. Total sediment was removed from the tanks using plastic scoops and placed into buckets. The last traces of sediment were scraped from the bottom of the tank using a squeegee wiper. The weight of each bucket was noted, and the collected sediment was re-distributed on the field, away from the field erosion plots. A scoop of sediment (around 100ml) from each filled bucket was taken and put aside, to be used in later analysis (see below). The minimum amount of sediment required for analysis was approximately 1kg. Once the tank had been emptied of runoff and sediment, the inside was washed and cleaned ready for the next runoff event.

The samples of runoff were analysed for soluble nutrients (N, P and K), carbon and total suspended sediment. The sediment samples were analysed for moisture

[2] www.sowap.org

content, organic matter, associated nutrients (N, P and K), carbon and particle size. A list of these measurements can be found in Table 3.3-1. These parameters were measured as they were considered to be indicators of soil quality or they affected soil erodibility. More detailed information about these parameters can be found in 1.2.5).

Table 3.3-1 Analysis of runoff and sediment in collection tanks.

Analysis	Runoff	Sediment
N	yes - soluble nitrate	yes - total N
P	yes – soluble phosphate	yes - total P
K	yes – soluble potassium	yes - total K
TOC (total organic carbon)	yes	yes
suspended sediments	yes	
moisture content		yes
total sediment mass (kg) (= mass of suspended sediment in runoff + mass of sediments deposited on bottom of tank)		yes
organic matter		yes
particle size (textural analysis)		yes

3.3.2 Additional soil measurements

Additional surface characteristics and soil properties were measured adjacent to and within the field erosion plots to provide supporting information in order to explain the differences in runoff and sediment observed for the different tillage treatments.

Within the field erosion plots, crop and weed surveys were carried out. Without walking on the plot itself, three quadrats ($1m^2$) were placed at the bottom, middle and top of each field erosion plot, at approximately 0.5m inside the plot (to avoid boundary effects). Within each quadrat the percentage surface cover due to crop and weed cover, residues and stones was calculated. Also within each quadrat the presence of any erosion features was noted and the surface roughness measured. Surface roughness was calculated by taking a small ball (5mm) linked chain, 1 metre in length and draping it across the soil surface. The ratio of distance covered to actual length of chain is an indicator of surface roughness (Morgan et al. 1988). Hence, on a perfectly smooth surface the 1m long chain would measure 1m in distance, whereas the same chain would cover a much smaller distance on a rough surface, as it conforms to micro-topographic irregularities i.e. surface troughs and ridges.

Outside of the field erosion plots, the following soil properties were measured in the top 5cm of soil, taken to be representative of the surface soil - soil texture (particle sizes), organic matter content, moisture content (gravimetric and volumetric), bulk density, organic carbon and nutrients (total nitrogen, phosphorus and potassium) These soil properties were taken biannually in conjunction with the micro-plot, rainfall simulation experiments in the spring and autumn (chapter 4) of each season. These measurements may provide explanation of the magnitude of and variations in runoff and soil losses observed from the two spatial scales of erosion plots studied in this thesis.

3.4 Results

This results section will show all data relating to the field erosion plots in relation to the previously set out objectives (section 3.2). Results were standardised to per unit area, to allow for the variable plot size. Soil and runoff losses have been expressed as losses per hectare or concentrations per litre. The runoff and soil loss results are presented for each site separately. For each site, the runoff and soil loss data and associated nutrient and carbon losses are analysed for treatment differences at 3 temporal scales:

- The mean loss for the entire sampling period (the mean of each treatment over all tank clearances)
- The mean loss for each cropping season (the mean of each treatment over all tank clearances within each season)

- The mean loss for individual tank clearances (the mean loss for each treatment for each individual tank clearance)

Statistical analysis could be carried out on the results from duplicate plots due to the temporal replication. Where results were not normally distributed, and had unequal variance, the data were transformed to allow ANOVA to be performed. Details of the statistical analysis undertaken in this chapter can be found in section 2.2. Where statistically significant differences between treatments have been found, these have been highlighted. Cumulative results are also shown; however, no statistical analysis could be carried out.

3.4.1 Loddington

3.4.1.1 Hypothesis to be tested

The mean runoff, soil loss and associated nutrient and carbon losses over all tank clearances will be higher from the field erosion plots where conventional tillage has been applied, compared to the conservation treatments (SOWAP and Farmer's Preference). To test the hypotheses specifically, more emphasis is placed on comparing differences *between treatments* at each of the different temporal scales, rather than comparisons of individual treatment results between seasons or sites.

3.4.1.2 Runoff

3.4.1.2.1 Runoff volume

The runoff volumes over the entire sampling period were analysed for each treatment. The runoff volume data were normalised (Type I) to allow ANOVA to be performed. The normalised treatment means were then back transformed to the original scale. The overall mean runoff loss for the entire sampling period was 1157 l ha^{-1} from the conventional treatment and 1617 and 2244 l ha^{-1} from the conservation treatments, SOWAP and Farmer's Preference, respectively. Graphical outputs for runoff volume over the entire sampling period and for each season can be found in Figure 3.4-1. Statistical analysis of the normalised data showed no significant differences between treatment means at any of the three temporal scales – over the entire sampling period, at each season or for each tank clearance. Despite this, the overall mean results show a (non-significant) treatment trend of runoff volumes being lowest from the conventional treatment, and highest from the Farmer's Preference conservation treatment. This trend was also found during the third and fourth season (Figure 3.4-3).

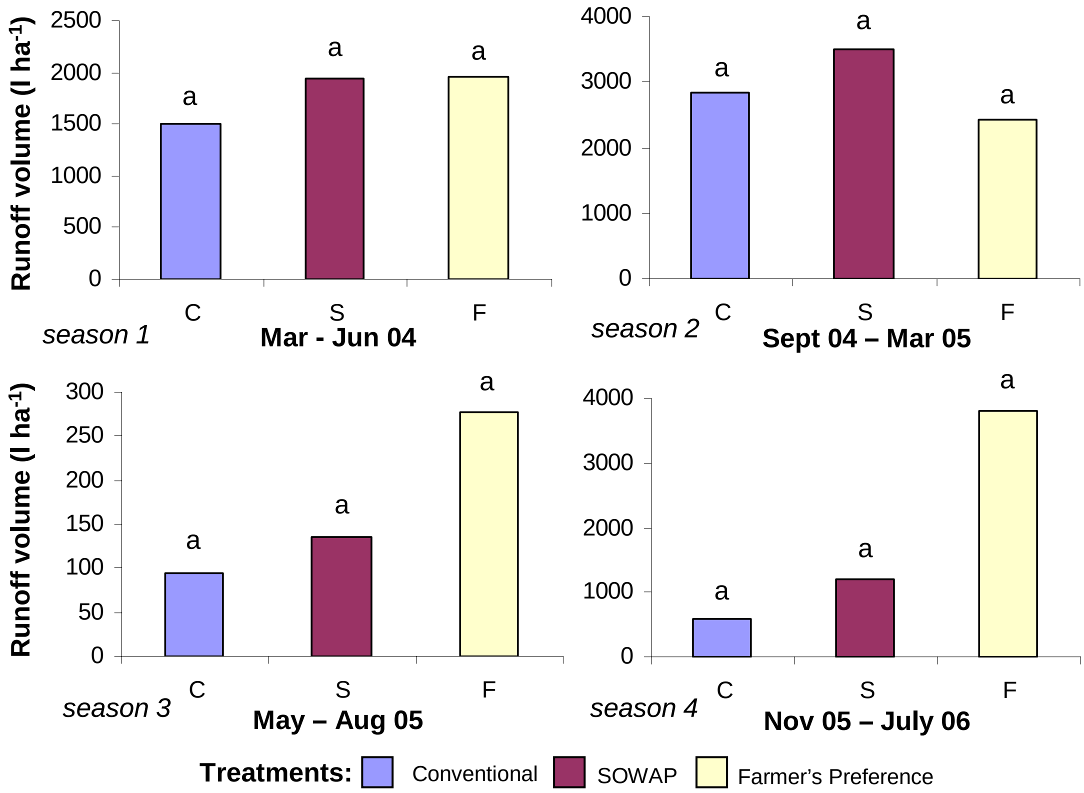


Figure 3.4-1 Loddington: runoff volume for each season. Lettering signifies statistical differences.

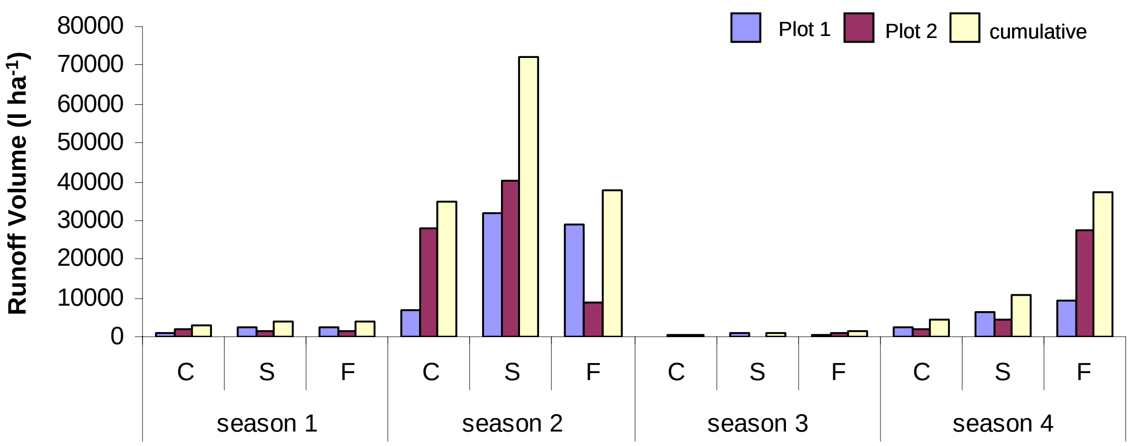
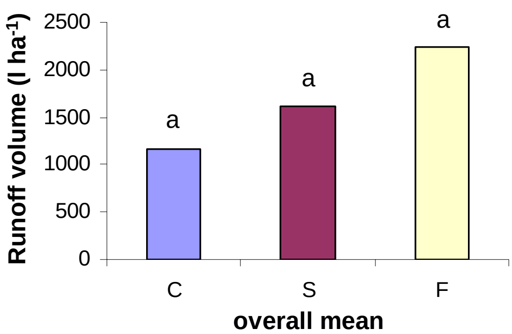


Figure 3.4-2 Loddington: runoff volume for each erosion plot and the combined cumulative results for each season. Treatments where C=conventional, S=SOWAP and F=Farmer's Preference.



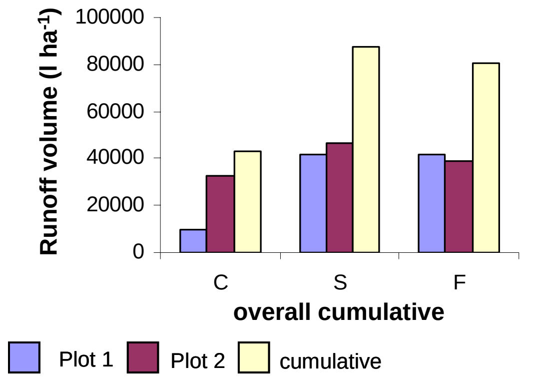


Figure 3.4-3 Loddington: runoff volume over the entire sampling represented as mean (left) and cumulative (right) results. Lettering signifies statistical differences

The cumulative results for each season (Figure 3.4-2) and over the entire sampling period (Figure 3.4-3) the variation between the duplicate plots for each treatment. Overall the conventional treatment shows the greatest variation between plots in terms of runoff generation. The seasonal cumulative results show the relative contribution of each season to the total runoff generated. The second season produced the highest runoff volumes representing 68.5% of the total runoff generated over the entire sampling period (Table 3.4-1). Results for both the conventional and SOWAP treatments generated the most runoff during the second season. The runoff volumes from the Farmer's Preference during the second and fourth season were similar. These results might be related to rainfall patterns in each season, although Figure 3.4-4 shows that the fourth season received the most rainfall, yet season two generated the greatest amount of runoff.

Table 3.4-1 Loddington: relative percentage of overall cumulative results (from both duplicates) for each treatment and for each season compared to the total runoff generated.

%	Conventional	SOWAP	Farmer's Preference	Total
Season 1	1.48	1.85	1.87	5.20
Season 2	16.43	34.24	17.85	68.52
Season 3	0.32	0.45	0.70	1.47
Season 4	2.14	5.07	17.62	24.82
Total	20.36	41.60	38.03	**100.00**

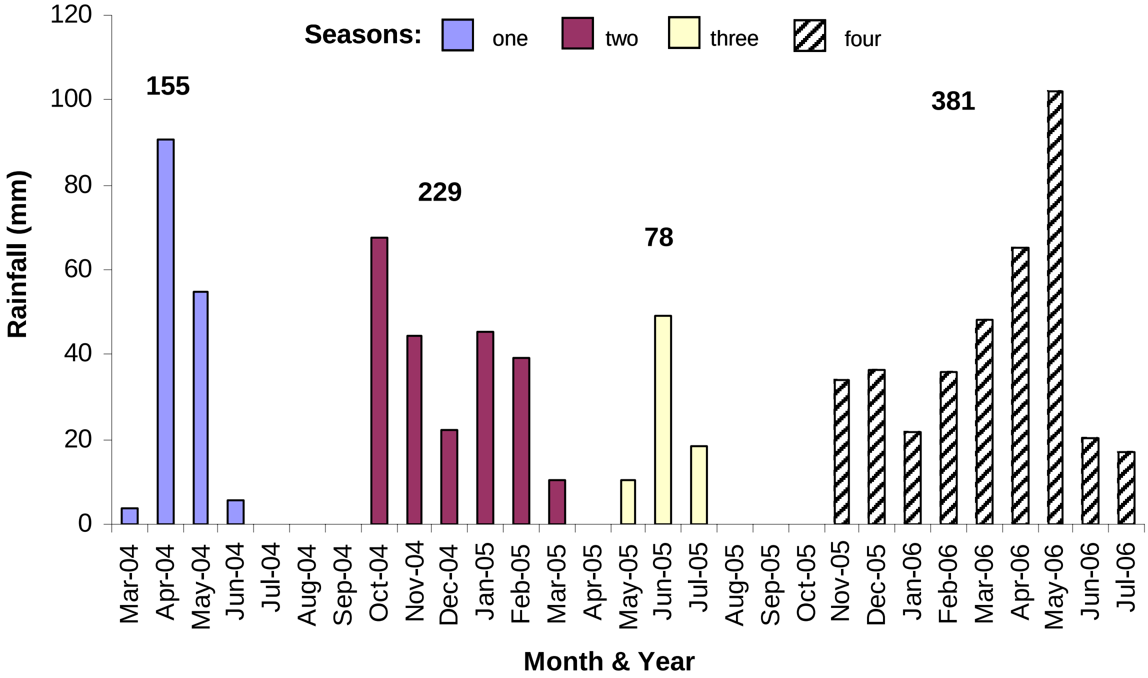


Figure 3.4-4 Loddington: precipitation data for each season. The data shown represents rainfall from installation to removal of the erosion plots. Floating numbers above each season indicate total rainfall received for that time period.

3.4.1.2.2 Nutrient Loss

Runoff generated from the field erosion plots was analysed for nutrient concentrations and total loads. Nutrient loads were a function of runoff volume.

The nutrients tested for were nitrogen, phosphorus and potassium. Each nutrient will be addressed separately in the following text.

Nitrogen

Collected runoff was analysed for nitrate, and the results are expressed as soluble nitrogen concentration, and total load (as a function of the runoff volume). The data for both concentrations and loadings of N were normalised through a Type I transformation (section 2.2) to allow ANOVA to be performed. The mean N concentration (over the entire sampling period) in the runoff for the three treatments ranged between 4.5 and 5.1 mg l^{-1} and are presented in Table 3.4-2. Statistical analysis was carried out on nitrogen concentrations over the entire sampling period, where no significant differences were found. This was also the case when treatment means were analysed over all tank clearances. Seasonal analysis showed that there was a significant treatment difference (p<0.005) during the first season (Figure 3.4-5), March to June 2004. Here, nitrogen concentrations were greatest in runoff generated from the Farmer's Preference treatment in comparison to the other conservation treatment (SOWAP) and the conventional treatment.

Table 3.4-2 Loddington: runoff associated mean nitrogen over the entire sampling period

Treatment	Concentration (mg l^{-1})	Loading (g ha^{-1})
Conventional	4.53	4.42
SOWAP	4.70	7.71
Farmer's Preference	5.12	15.87

The mean loading of nitrate associated with the runoff over the entire sampling period are presented in Table 3.4-2. Statistical analysis was carried out on the normalised nitrate loads. There were no significant treatment differences when analysed at any of the 3 temporal scales - over the entire sampling period, seasonally or across the different tank clearances.

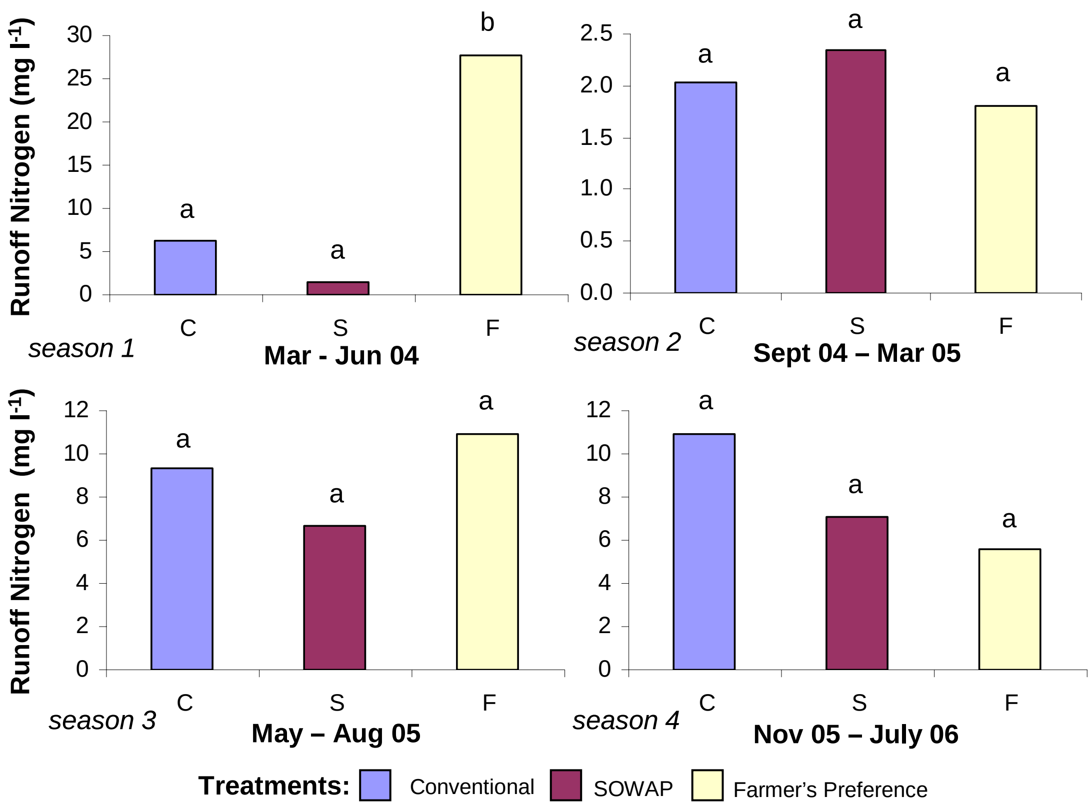


Figure 3.4-5 Loddington: nitrogen concentration associated with the runoff. Lettering signifies statistical differences

Phosphorus

Collected runoff was analysed for phosphate and the results expressed as soluble phosphorus concentration and loading. The results for both mean P concentration and loadings were normalised (Type I). The mean P concentration was 0.8 mg l^{-1} in runoff from the conventional treatment and 0.1 mg l^{-1} from both conservation treatments. Statistical analysis showed a significant difference between treatment means over the entire sampling period. Phosphorus concentration was significantly higher from the conventional treatment (Figure 3.4-6) compared to both conservation treatments (p=0.003). There were no significant differences in P concentration between treatment means when analysed seasonally or over all tank clearances.

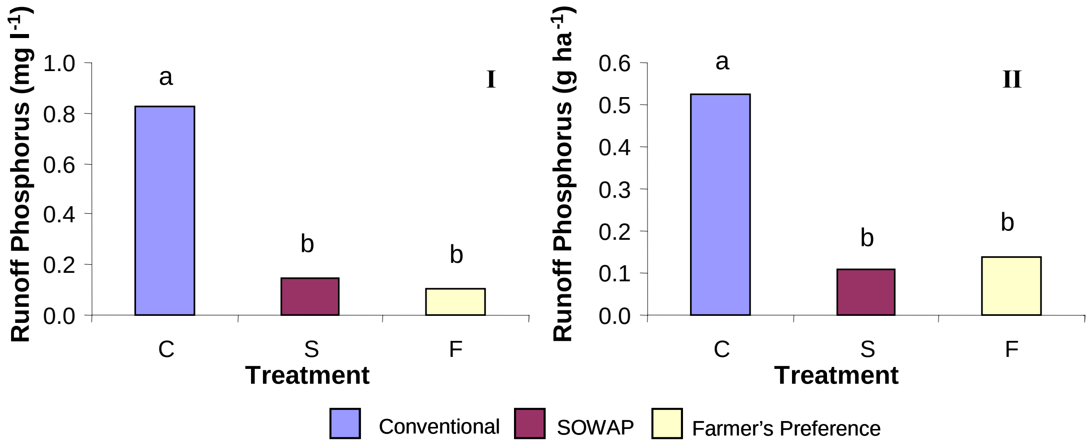


Figure 3.4-6 Loddington: I) phosphorus concentration in the runoff; II) phosphorus load in the runoff. Lettering signifies statistical differences

Statistical analysis of treatment means over the entire sampling period of phosphorus loading in the runoff were significantly different (p=0.04); the result from the conventional treatment was significantly higher than both conservation treatments (Figure 3.4-6). The overall means were 0.53 g ha^{-1} from the conventional plot and 0.11 and 0.14 g ha^{-1} from the SOWAP and Farmer's Preference treatments, respectively.

Potassium

Runoff was analysed for concentration and loading of potassium. Both data sets were normalised (Type II) to allow ANOVA to be carried out. The treatment means of both potassium concentration and loads were statistically analysed over the entire period, seasonally and for each separate tank clearance. Analysis showed no significant difference between treatments at any of the temporal scales. The mean potassium concentrations and loads can be found in Table 3.4-3.

Table 3.4-3 Loddington: runoff associated mean potassium over the entire sampling period

Treatment	Concentration (mg l^{-1})	Loading (g ha^{-1})
Conventional	32.38	26.24
SOWAP	18.49	20.30
Farmer's Preference	16.93	19.12

3.4.1.2.3 *Carbon Loss*

Runoff generated from the field erosion plots from each treatment was analysed for concentration and loading of total organic carbon. The data for both concentrations and loadings were not normally distributed, and had unequal variance, so the data was transformed (Type II) to allow ANOVA to be performed. The treatment means of total organic carbon concentration and loading in the runoff were analysed over the entire sampling period, seasonally and for each separate tank clearance. Differences between treatment means were not statistically significant at any of the temporal scales investigated. The overall mean of runoff associated carbon concentration and loads can be found in Table 3.4-4.

Table 3.4-4 Loddington: runoff associated mean total organic carbon over the entire sampling period

Treatment	Concentration (mg l^{-1})	Loading (g ha^{-1})
Conventional	11.34	9.55
SOWAP	7.39	9.28
Farmer's Preference	6.85	10.24

3.4.1.3 Soil Loss

3.4.1.3.1 *Sediment*

Soil losses at the three temporal scales were investigated – over the entire sampling period, on a seasonal basis and for each tank clearance. The data were not normally distributed and so were normalised (Type I) to allow ANOVA to be undertaken. Statistical analysis showed there was a significant difference (p=0.04) between treatments when compared on a tank clearance basis (Figure 3.4-7). However, LSD outputs showed this to be the case for the second tank clearance only. The soil loss from this tank clearance (carried out on the 22nd October 2004) was much higher when compared to all other 10 clearances, and so had to be presented on a separate graph (Figure 3.4-9). The soil lost from the

SOWAP treatment during this tank clearance was significantly higher than that generated by the other two treatments (conventional and Farmer's Preference).

Soil loss results from Loddington were also analysed on a seasonal basis; no significantly difference were found. The results are presented in Figure 3.4-10. Although not significant it can be seen that in season one and two the treatment trends were similar, with losses being the highest from the SOWAP treatment. During season three no sediments were collected from the base of the tank. Results represent only the mass of suspended sediment in the runoff, which is why the results are much lower than the other seasons, when sediment was also found on the bottom of the tank. There was also no significant difference between treatments during season four.

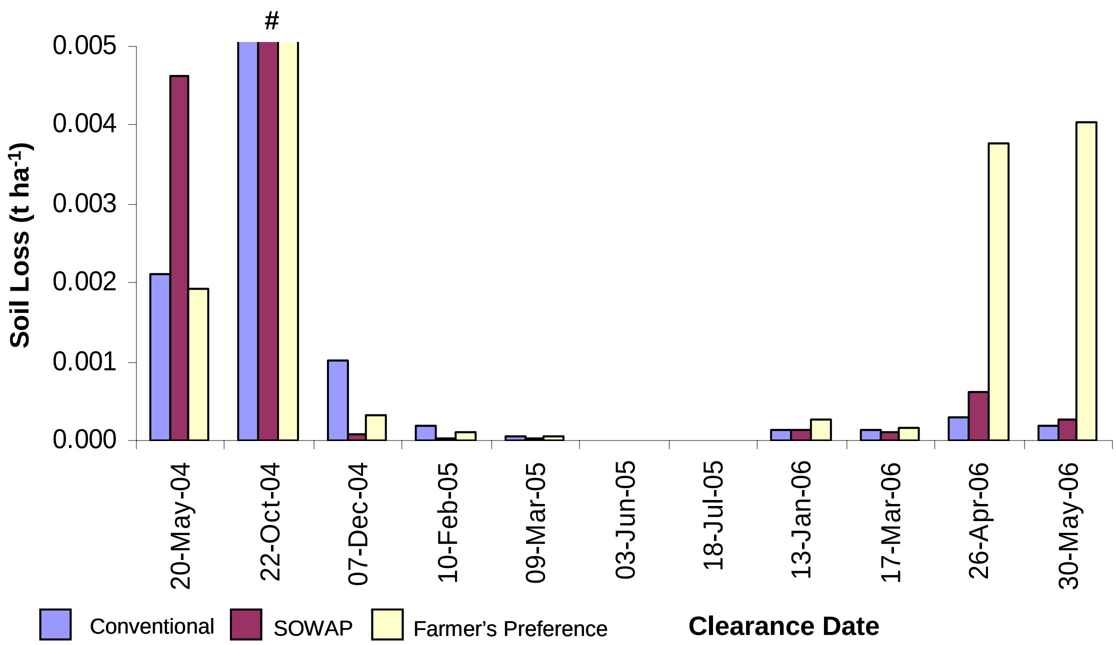


Figure 3.4-7 Loddington: soil loss for every tank clearance. # soil loss results too high to show on this graph (see Figure 3.4-8).

The results from the second tank clearance were unusually high and so were removed to identify the significance of this result on the remaining dataset. Once removed, the results from the Farmer's Preference treatment during season four became significantly higher than the other two treatments (p=0.01). The reasoning behind this unusually high clearance can be found in the discussion. All further presentation of results was done using the full dataset.

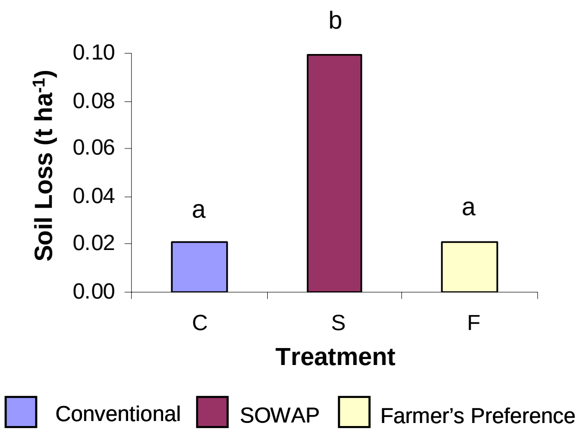


Figure 3.4-9 Loddington: soil loss from the October 2004 tank clearance. Letters represent significant differences

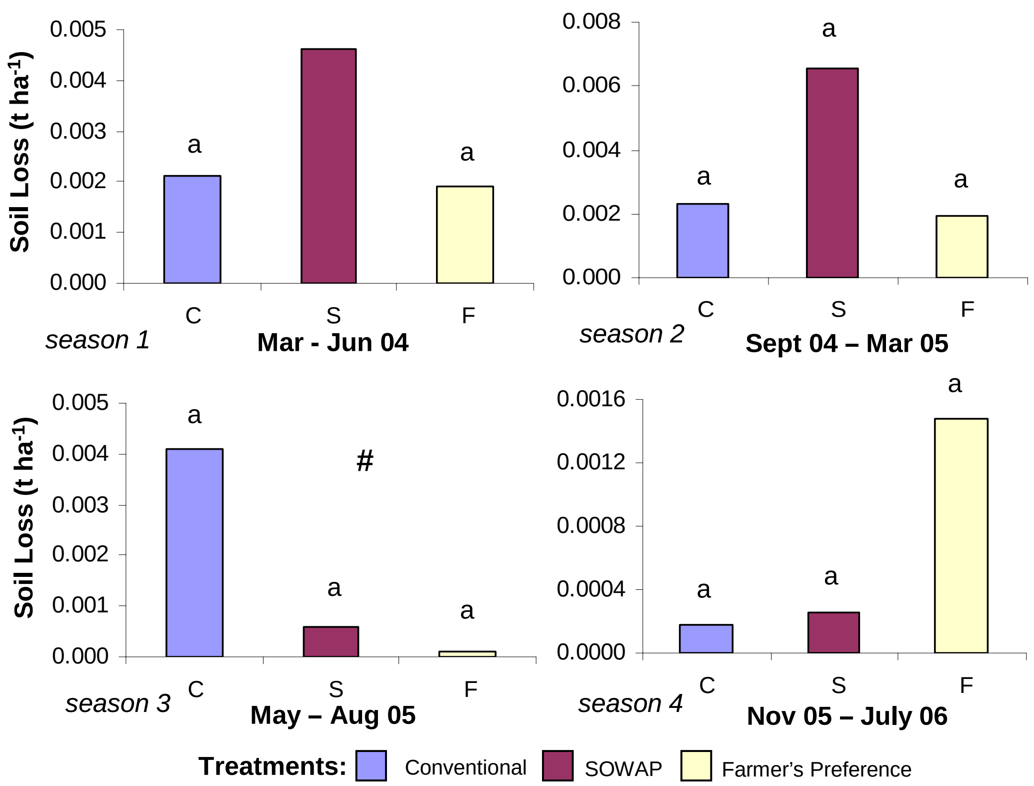


Figure 3.4-10 Loddington: soil loss for each season. Lettering signifies statistical differences. # = units expressed are 10^{-6}. Lettering signifies statistical differences

The last temporal scale analysed was the mean soil loss over the entire sampling period. Statistical analysis showed no significant difference between treatments with the inclusion (Figure 3.4-11) or exclusion of October 2004 tank clearance. The overall and seasonal cumulative soil loss results from each treatment were plotted (Figure 3.4-12 and Figure 3.4-11). The overall cumulative results showed the same treatment trends as the overall mean. The cumulative soil loss was higher than the other two treatments (conventional and Farmer's Preference). The cumulative results (overall and seasonal) show the variation that existed between erosion plots of the same treatment. The two plots from the SOWAP treatment showed the least amount of variation.

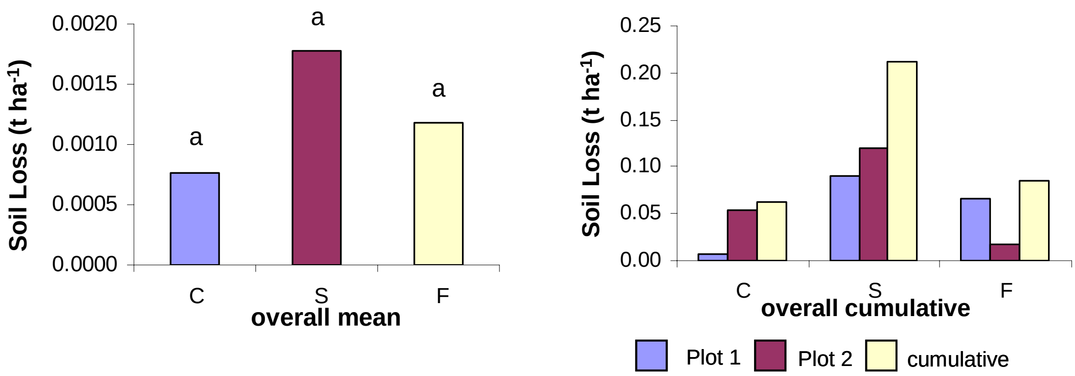


Figure 3.4-11 Loddington: soil loss over the entire sampling represented as mean (left) and cumulative (right) results. Lettering signifies statistical differences

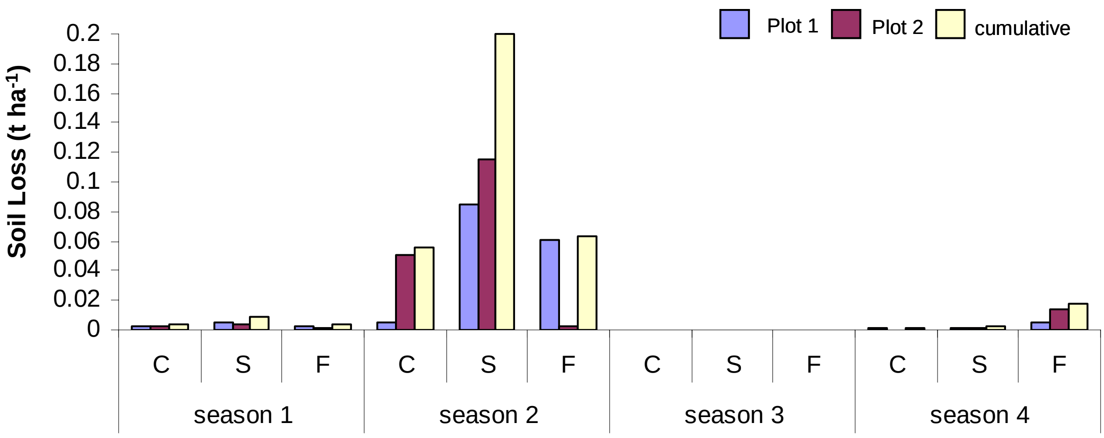


Figure 3.4-12 Loddington: cumulative soil loss for each season. Treatments were – C=conventional, S=SOWAP and F=Farmer's Preference

The majority of soil loss occurred during the second season for all treatments, clearly seen in Figure 3.4-12, making the treatment trends found during this season important for the results overall. Calculations were made as to the percentage of soil lost from each treatment during each season compared to the total amount over the entire sampling period. These results are presented in Table 3.4-5. Soil losses from season two represent over 89% of the total amount lost. The SOWAP treatment contributed to 59% of the total soil loss compared to 24% from the Farmer's Preference and 17% from the conventional treatment.

Table 3.4-5 Loddington: relative percentage of overall cumulative results from each treatment and for each season compared to the total soil loss.

%	Conventional	SOWAP	Farmer's Preference	Total
Season 1	1.18	2.59	1.08	4.85
Season 2	15.71	55.79	17.57	89.07
Season 3	0.00	0.00	0.00	0.00
Season 4	0.38	0.65	5.06	6.08
Total	17.27	59.02	23.71	**100.00**

3.4.1.3.2 Nutrient Loss

Soil lost from the field erosion plots from each treatment was collected and analysed for nutrient concentration and total loads. Nutrient loads were a function of soil loss. Nutrient concentration and loads could only be analysed for the entire sampling period, and over the first and second season due to insufficient sediment being retained in the collection tanks for nutrient analysis. Comparisons for each tank clearance were not possible, again due to insufficient sediment generation. However, where there was sufficient sediment for analysis, the nutrients analysed were total nitrogen, phosphorus and potassium. Each nutrient will be addressed separately in the following text.

Nitrogen

Soil lost from the field erosion plots was analysed for nitrogen concentration and total load. The nitrogen concentration data were normally distributed with equal variance. The mean nitrogen concentration did not exceed 6500 mg kg^{-1} from any of the treatments. Statistical analysis of the treatment means over the entire sampling period showed no significant differences. This was also the case when results were compared seasonally.

The total nitrogen load in the sediment was normalised (Type II) to allow analysis of variance to be carried out. Statistical analysis of treatment means over the entire sampling period and on a seasonal basis showed no significant differences. The mean nitrogen load associated with the eroded sediment ranged between 16 and 76 g ha^{-1} for the 3 treatments.

Phosphorus

The phosphorus concentration and loads in the sediment were not normally distributed, so were transformed (Type II) allowing statistical analysis to be undertaken. When phosphorus concentrations were analysed over the entire sample period (Figure 3.4-13), they were found to be significantly higher in the sediment from the conventional treatment compared to both conservation treatments (p=0.012). However, seasonal comparison of treatment means showed no significant differences in phosphorus concentrations.

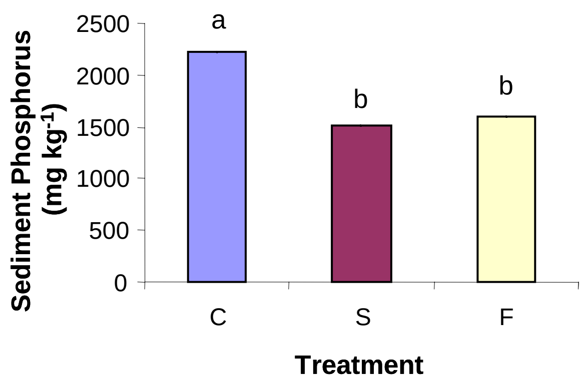


Figure 3.4-13 Loddington: sediment associated phosphorus concentration. Letters denote significant differences.

Phosphorus loadings associated with the sediment were statistically analysed over the entire sampling period and seasonally for differences between treatment means. There were no significant treatment differences in phosphorus loads at either temporal scale. The mean load of phosphorus in the sediment ranged between 5 and 31g ha^{-1} for the three treatments.

Potassium

The results of potassium concentration in the sediment were normally distributed with equal variance. When statistically analysed over the entire sampling period, there were no significant differences between treatment means. The mean concentration of potassium in the sediment did not exceed 5500 mg kg^{-1} from any of the treatments (Table 3.4-6). The treatment means were also compared across the seasons; the treatment means did not significantly differ.

The potassium loading in the sediment was transformed (Type II) to allow ANOVA to be carried out, as the original data was not normally distributed. Statistical analysis over the entire sampling period and on a seasonal basis showed no significant differences between treatment means. The mean loading of potassium for all treatments can be found in Table 3.4-6

Table 3.4-6 Loddington: overall mean sediment potassium over the entire sampling period

Treatment	Concentration (mg kg^{-1})	Loading (g ha^{-1})
Conventional	5490	17.7
SOWAP	5483	110.5
Farmer's Preference	5019	13.4

3.4.1.3.3 Carbon Loss

Soil lost from the field erosion plots was collected and analysed for total organic carbon concentration and total load. Carbon concentration and load could only be analysed over the entire sampling period and not on a seasonal or tank clearance basis due to insufficient sediment being retained in the collection tanks for carbon analysis.

Carbon concentrations in the sediment were normally distributed and were analysed using analysis of variance. No significant difference between treatment means was found overall. The loading of carbon in the eroded soil had to be normalised (Type I). Statistical analysis showed no significant difference between treatment means. The mean carbon concentration and load for all treatments are presented in Table 3.4-7.

Table 3.4-7 Loddington: overall mean sediment carbon over the entire sampling period

Treatment	Concentration (m g kg^{-1})	Loading (g ha^{-1})
Conventional	46900	99.34
SOWAP	33600	154.23
Farmer's Preference	34000	114.07

3.4.1.4 Additional analysis

Additional analyses were carried out on the sediment generated from the field erosion plots, including organic matter content and particle size. Additional field soil properties and surface characteristics were also quantified with measurements taken adjacent to and within the field erosion plots to give supporting information for observed treatment differences. These have been referred to previously in section 3.3.2.

3.4.1.4.1 Correlations

Utilising data from the entire sampling period the volume of runoff and mass of eroded soil were correlated against each other to identify if a relationship existed. A simple correlation of normalised (both Type I) runoff and soil loss data showed a significant positive relationship (r=0.94), therefore, as runoff increased so did the amount of soil lost (Figure 3.4-14).

Sediment was analysed for soil particle size (texture) and organic matter content. These results were then correlated to sediment concentration of nutrients and carbon to identify possible mechanisms of loss. No significant relationships were found. Correlations were also carried out between runoff- and sediment-associated nutrients and carbon in relation to a variety of parameters, the results of which are presented in Table 3.4-8.

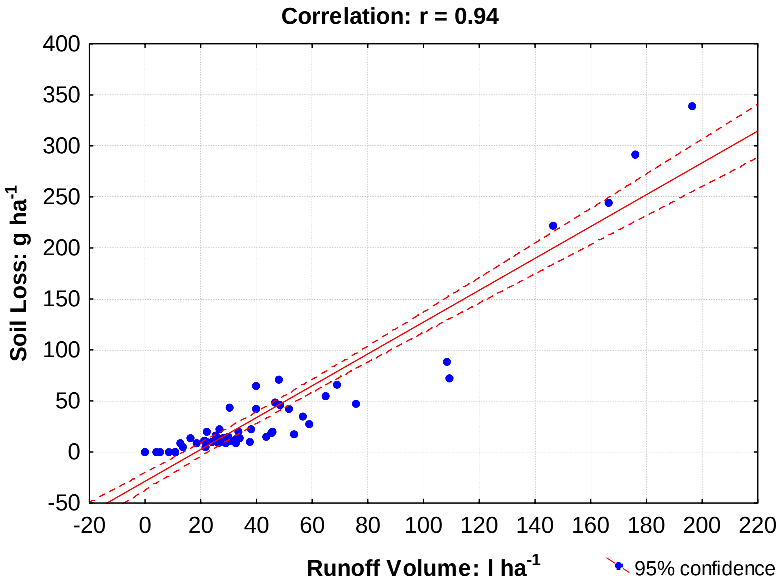


Figure 3.4-14 Loddington: correlation between normalised (Type I) runoff volume and soil loss (n = 59, p<0.05)

Variables	Runoff	Soil Loss	Suspended Sediment	S-O.M. %	S-Silt %
Runoff	n/a	0.94	0.47	x	-0.64
Soil Loss	0.94	n/a	0.40	x	x
R-Nitrogen conc.	x	x	x	x	x
R-Phosphorus conc.	-0.39	-0.34	x	x	x
R-Potassium conc.	x	0.32	x	x	x
R-Organic Carbon conc.	x	0.30	x	x	-0.52
S-Nitrogen conc.	x	x	x	0.73	x
S-Phosphorus conc.	x	x	x	x	x
S-Potassium conc.	x	x	x	x	x
S-Organic Carbon conc.	x	x	x	x	x
S-O.M. %	x	x	0.47	n/a	x
S-Sand %	x	x	x	x	n/a
S-Silt %	-0.64	x	-0.84	x	n/a
S-Clay %	x	x	0.61	x	n/a

Table 3.4-8 Loddington: table of significant correlations (p<0.05). S- before a variable denotes it is associated with the sediment e.g. S nitrogen is the nitrogen

associated with the sediment. R- variable associated with the runoff. conc. = concentration. x = not significant. No signifcant correlations existed in relation to sand and clay content.

3.4.1.4.2 Field Soil Properties

The soil properties which were measured were soil particle size (texture), organic matter content, total organic carbon, moisture content (gravimetric and volumetric) and bulk density. These properties were measured biannually in the spring and autumn for each treatment. The properties have been measured in order to explain any observed differences in runoff volume and soil, nutrient and carbon loss. Specific explanation as to the effect these soil properties have on erosion can be found in section 1.2.5.

Textural Analysis

It is expected that an increase in clay content will contribute to a reduction in erosion and concurrent loss of nutrient and carbon due to the strong cohesive forces between clay particles. Soils with higher silt content have been linked to increased formation of surface seals or caps which would contribute to an increase in erosion.

The data for percentage clay in the field soil were normally distributed with equal variance and were statistically analysed for difference in treatment means over the entire sampling period and on a seasonal basis. Over the entire sampling period there was a significant difference between treatments ($p<0.001$). The clay content of soil from all 3 treatments was different with the lowest content from the conventional treatment and the highest from the Farmer's Preference treatment (Figure 3.4-16). When compared seasonally differences in treatment means were also significant ($p<0.01$). Clay content was lowest for the conventional treatment for all four seasons, but the Farmer's Preference treatment had the highest clay content only in the first and fourth season (Figure 3.4-15).

The percentage silt data were not normally distributed and so had to be transformed before analysis of variance could be carried out. Statistical analysis of the normalised (Type III) data over the entire sampling period (Figure 3.4-16), showed that percentage silt in the soil from the conventional treatment was significantly lower than from both conservation treatments, and that silt content on the Farmer's Preference treatment was significantly higher than the SOWAP treatment (p<0.001). Treatment means were compared seasonally, but there were no significant differences.

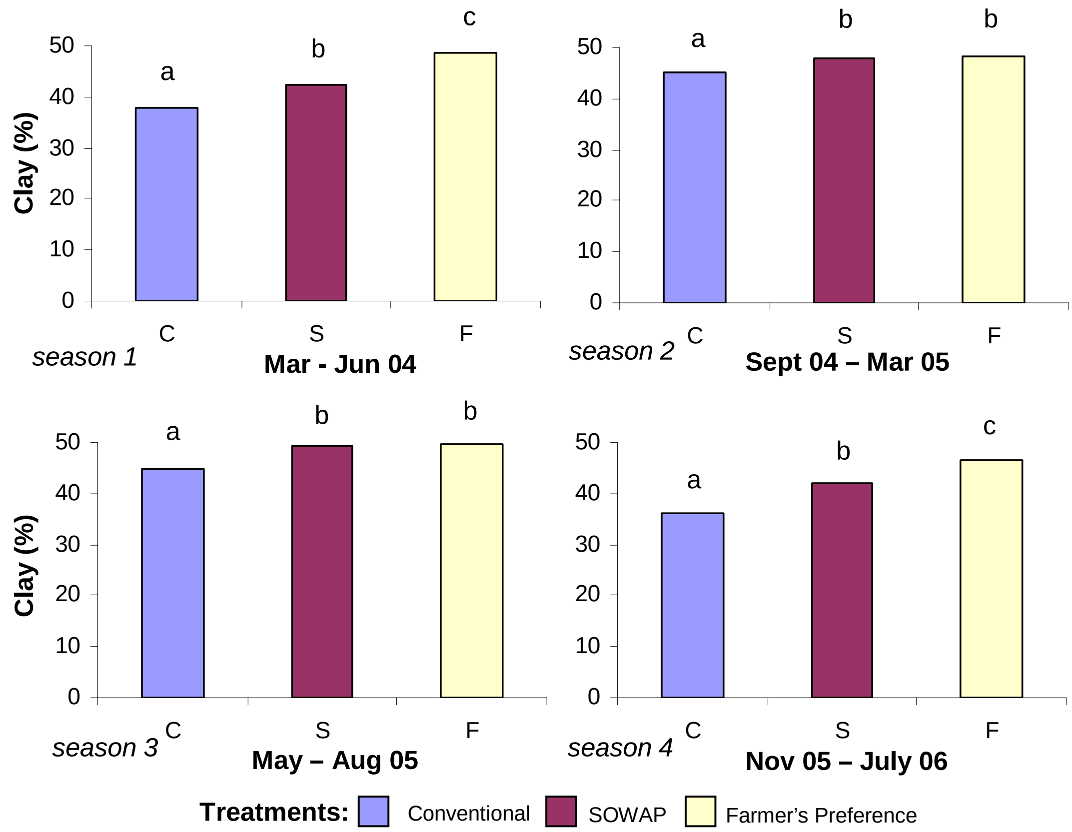


Figure 3.4-15 Loddington: clay content in field soil. Letters indicate significant differences

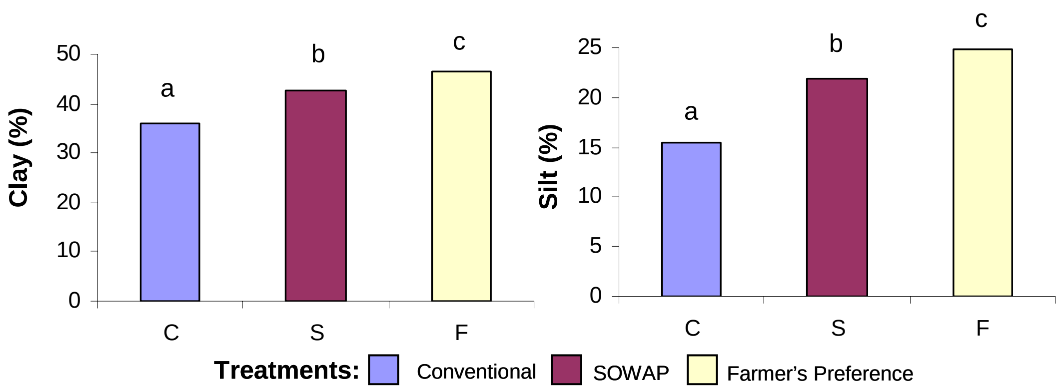


Figure 3.4-16 Loddington: overall mean clay content (left) and silt content (right) in field soil. Letters indicate significant differences.

Organic Matter

The presence of organic matter is expected to increase aggregate stability and reduce seal and crust formation, contributing to a reduction in erosion. Soil organic matter is expected to influence nutrient balances and is a source of organic carbon.

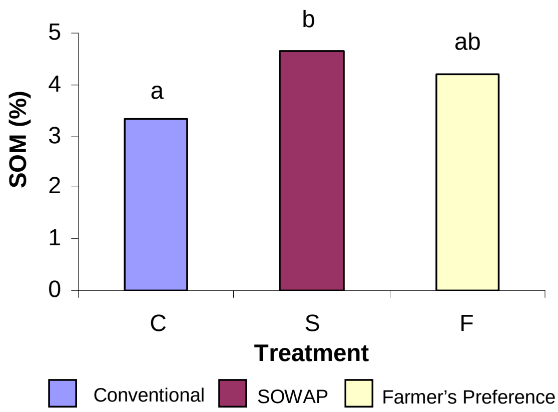


Figure 3.4-17 Loddington: organic matter in field soil. Letters indicate significant differences

The organic matter results were normally distributed with equal variance, allowing statistical analysis to take place. Results were compared over the entire sampling period (Figure 3.4-17) where the soil organic matter content of the conventional treatment was found to be significantly lower than the SOWAP treatment only (p=0.03). Comparison between treatment means was also carried out on a seasonal basis, however, no significant differences were found.

Moisture Content

Moisture content has been linked to the resistance to raindrop impact and infiltration rates. It is expected that differences between treatments may help explain observed erosion results. Soil moisture content was measured in two ways. The first, gravimetric moisture content represents the wetness of a soil i.e. the mass of water per mass of soil. The second, volumetric moisture content also represents soil water content as the volume of water per volume of soil.

The mean gravimetric moisture content was compared between treatments over the entire sampling period and across seasons. At both temporal scales there were significant differences between treatments ($p<0.001$), where gravimetric moisture content was the lowest for the conventional treatment compared to both conservation treatments. Season four and the overall means showed all three treatments were significantly different from one another (Figure 3.4-19). Only in season 2 (October 2004 to March 2005) were significant differences not found (Figure 3.4-18).

Analysis of volumetric moisture content was done for the entire sampling period and across seasons. At both scales significant treatment differences were found. Over the entire sampling period volumetric moisture content was found to be significantly lower for the conventional treatment compared to the Farmer's Preference treatment ($p=0.025$). When compared seasonally, significant differences were found in all seasons except for the second season ($p<0.001$). Graphical output of these results can be found in Figure 3.4-20 and Figure 3.4-19.

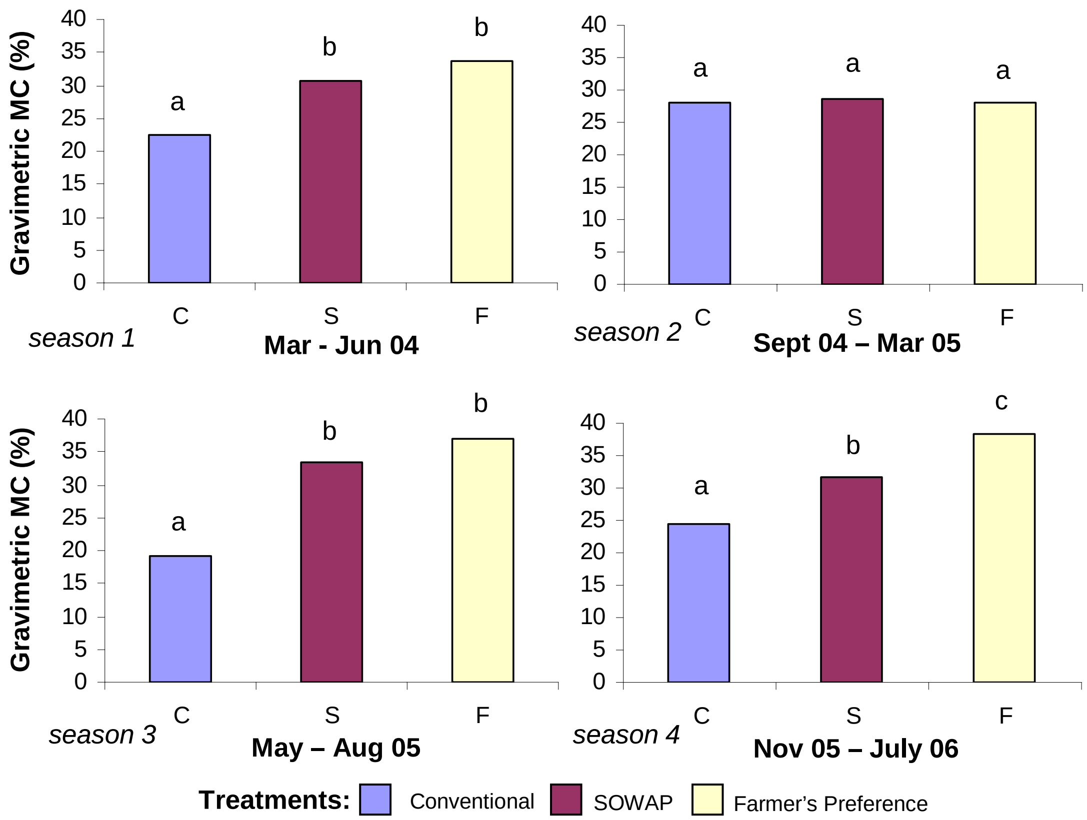


Figure 3.4-18 Loddington: gravimetric moisture content of field soil. Letters indicate significant differences.

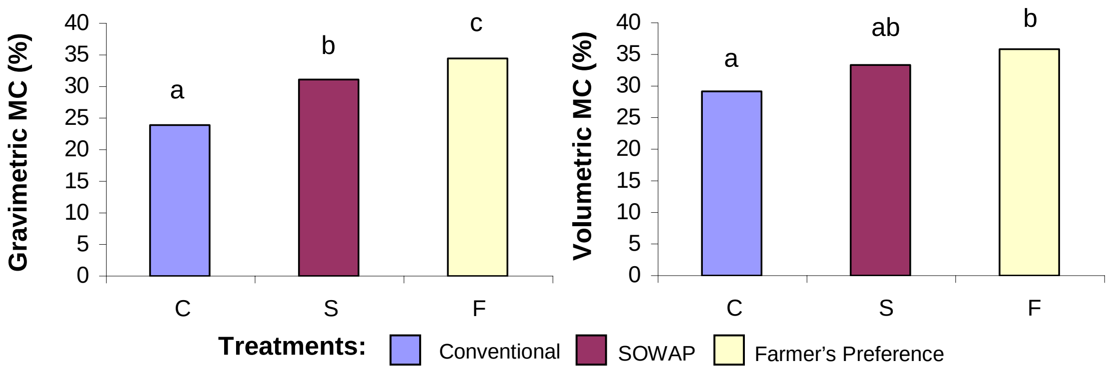


Figure 3.4-19 Loddington: overall means of moisture content, gravimetric (left) and volumetric (right). Letters indicate significant differences.

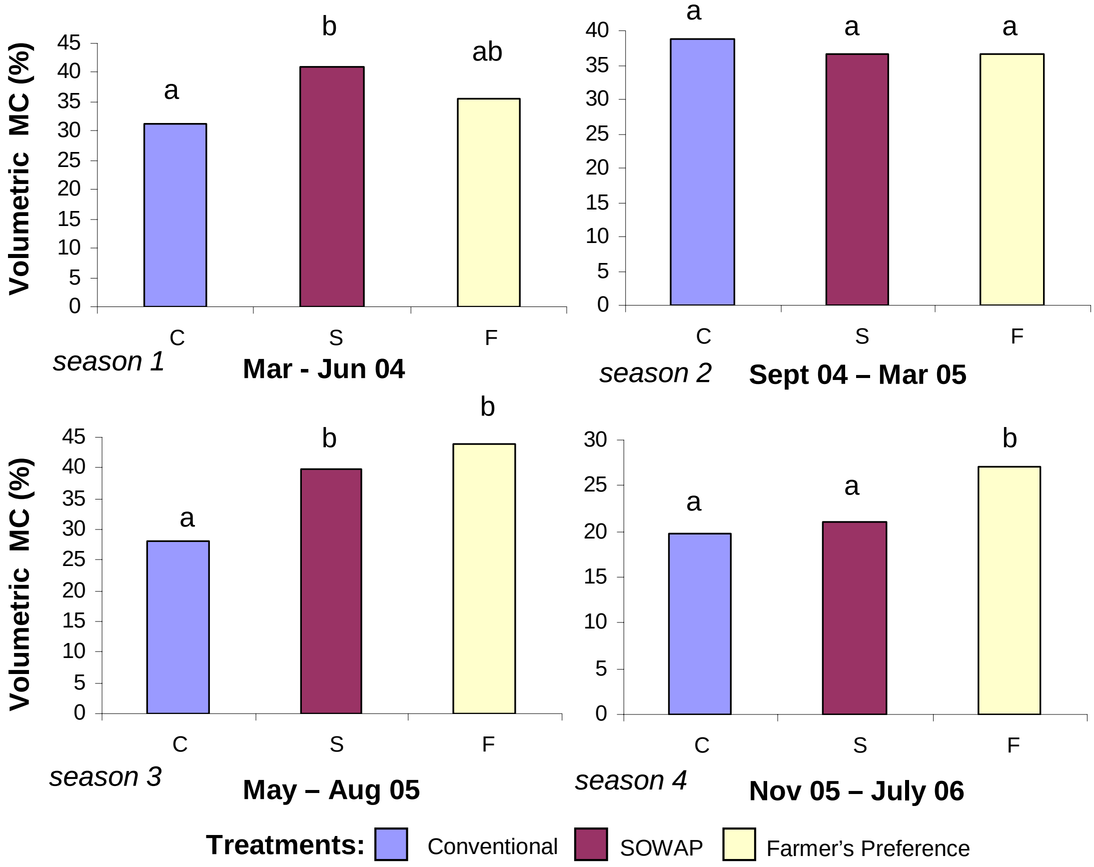


Figure 3.4-20 Loddington: volumetric moisture content of field soil. Letters denote significant differences.

Bulk Density

It is expected that soils with a higher bulk density will have decreased infiltration rates and impeded drainage, contributing to overland flow. The results of bulk density in the field soil were normally distributed with equal variance. Means were statistically analysed over the entire sampling period, and there were no significant differences between treatments. However analysis on a season basis showed significant differences in bulk density between the treatments in 3 out of the 4 seasons ($p=0.02$). In the first season, soil bulk density was lowest for the Farmer's Preference treatment compared to the other two. In season 3, May to August 2005, soil bulk density was significantly higher for the conventional treatment compared to both conservation treatments. In the last season, bulk density was again highest for the conventional treatment, but only when compared to the SOWAP treatment (Figure 3.4-21).

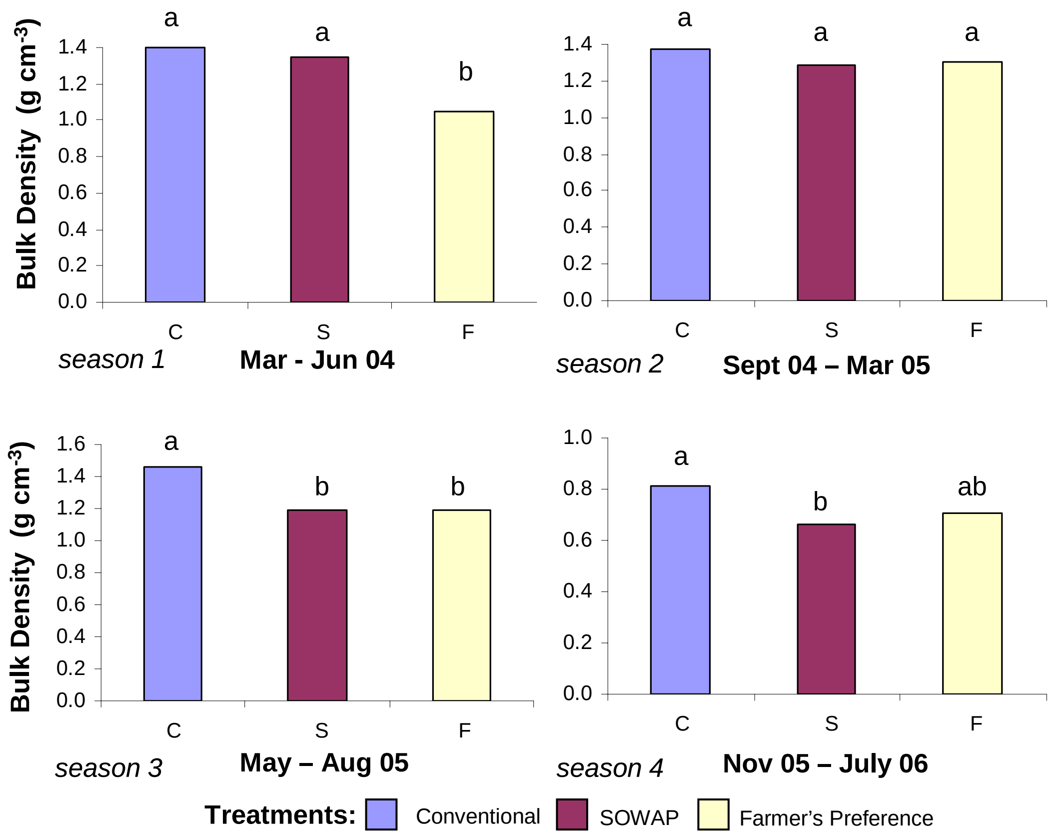


Figure 3.4-21 Loddington: bulk density in field soil on a seasonal basis. Lettering indicates significant differences

Total Organic Carbon

Total organic carbon (TOC) has been shown to have positive relationships with aggregate stability and reducing erosion. The data on TOC in the field soil were normally distributed, with equal variance allowing analysis of variance to undertaken. Means were compared over the entire sampling period where the organic carbon content of soil from the conventional treatment was significantly lower than both conservation treatments (p=0.01). This was also the case when means were compared on a seasonal basis (p<0.001); organic carbon content was lowest from the conventional treatment compared to both conservation treatments except in season four where no significant differences were found (Figure 3.4-22).

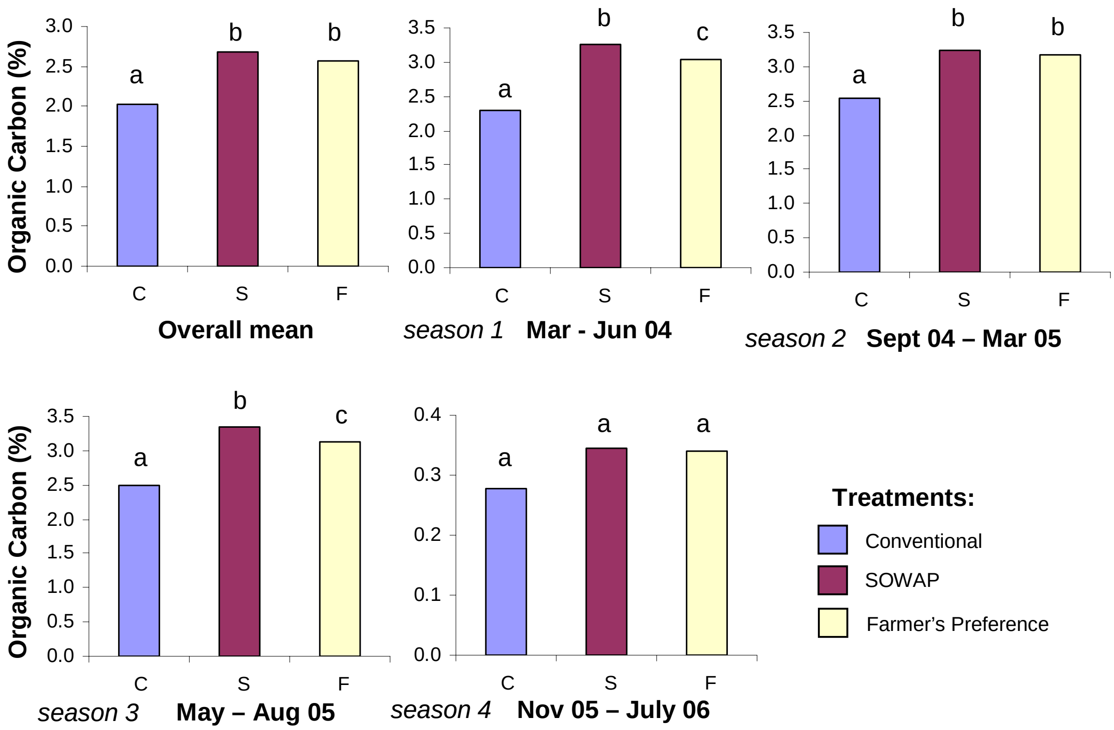


Figure 3.4-22 Loddington: organic carbon content of field soil on a seasonal basis. Lettering indicates significant differences

Field Soil Surface Characteristics

Soil Cover

Soil cover was assessed by measurements of the percentage of bare or exposed soil, the percentage residue, weeds and stones. The presence of any surface cover physically protects the soil surface and aggregates from direct rainfall impact, minimising soil detachment. It is therefore expected that an increase in cover will contribute to a reduction in erosion. The results of which will be presented in the following text.

The percentage of bare soil was statistically analysed over the entire sampling period. No significant treatment differences where found. However, when treatment means were compared on a seasonal basis, there was a significant difference between treatments ($p<0.001$), but only in the second season (October 2004 to March 2005), where the conventional treatment had significantly less percentage bare soil (more ground cover) in comparison to both conservation treatments (Figure 3.4-23).

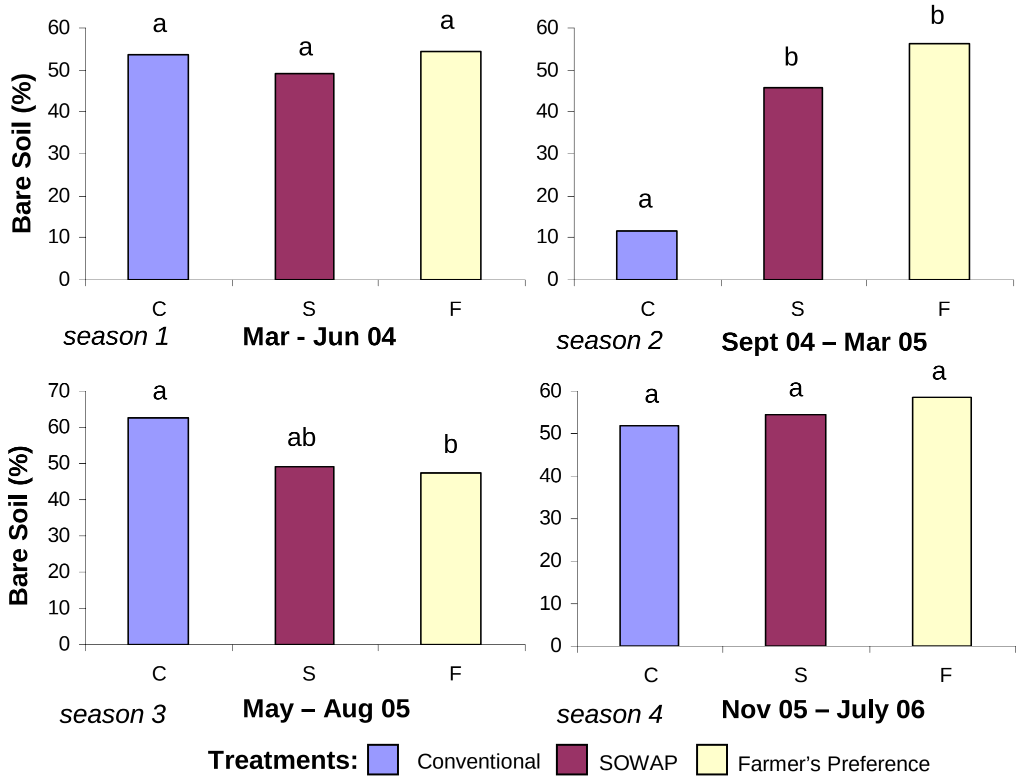


Figure 3.4-23 Loddington: percentage bare soil. Lettering indicates significant differences

The results of percentage residue cover were not normally distributed and so were transformed (Type II) to allow ANOVA to be carried out. When the results were analysed for differences between treatment means over the entire sampling period, none were found. However, when compared seasonally, significant differences between treatments were found ($p<0.001$), but only during season two; where all treatments differed significantly, as shown in Figure 3.4-24.

The percentage weed cover was analysed over the entire sampling period and on a seasonal basis; there were no significant differences between treatment means at either temporal scale. The mean weed cover on all three treatments ranged from 3.7 and 5.0 percent.

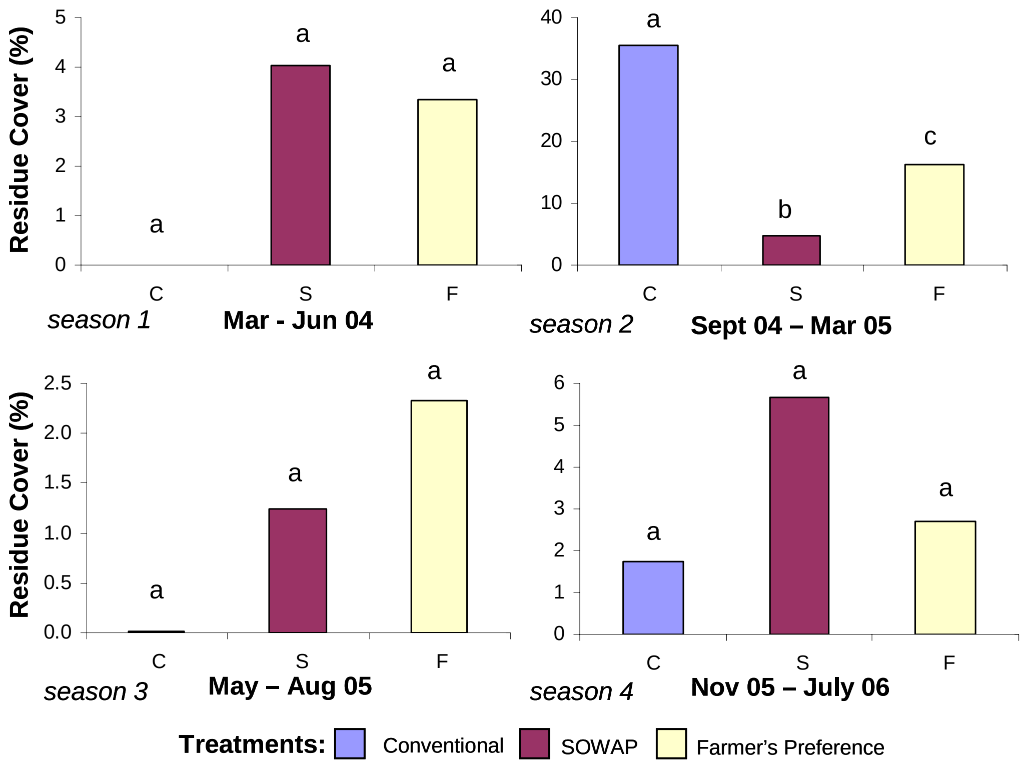


Figure 3.4-24 Loddington: percentage residue cover. Lettering indicates significant differences

The percentage of surface cover from stones was measured and statistical analysis was carried out over the entire sampling period where no significant difference between treatment means was found. When compared seasonally there were significant differences between treatments in the second and last season (p=0.01). During the second season the percentage of stones was significantly less on the conventional plots compared to the Farmer's Preference treatment. In the last season stone percentage was the highest for the conventional treatment and lowest for the SOWAP conservation treatment (Figure 3.4-25).

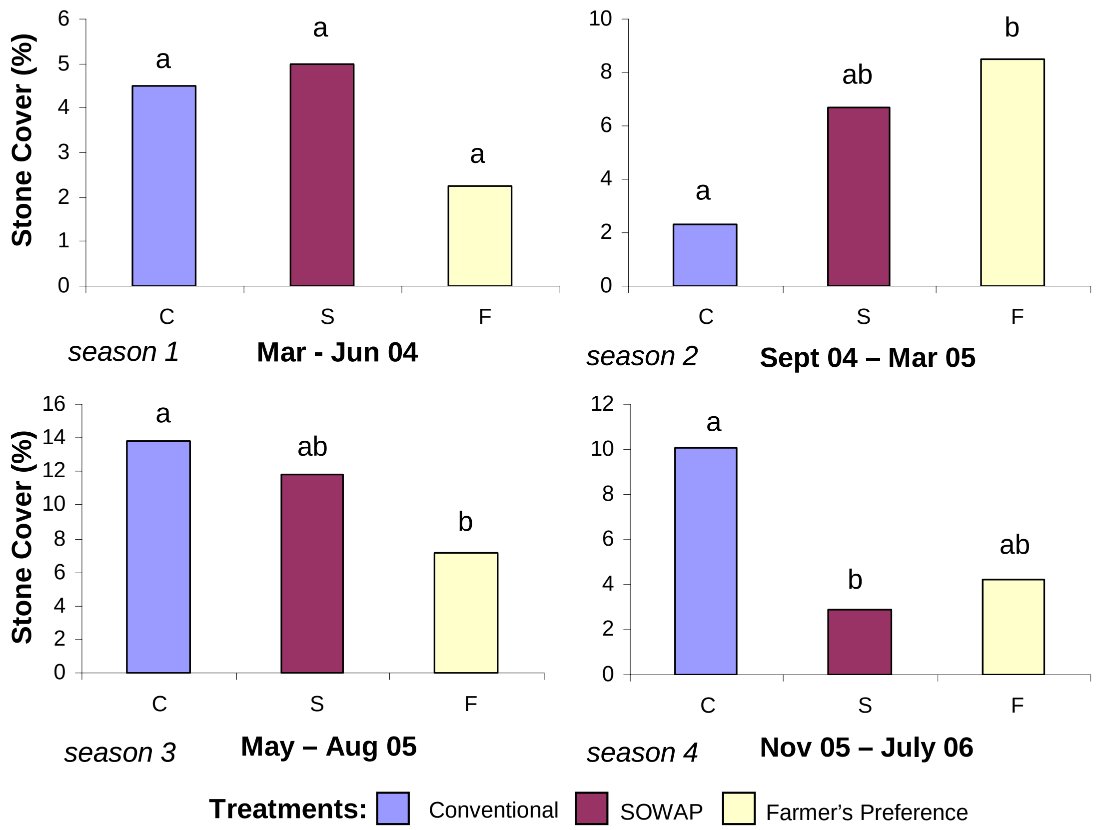


Figure 3.4-25 Loddington: stone cover percentage. Lettering indicates significant differences

Surface Roughness

It is expected that an increase in surface roughness will reduce the generation of overland flow and loss of sediment. The results of surface roughness were normally distributed and were analysed over the entire sampling period. The soil surface on the Farmer's Preference plots was significantly rougher than the other two treatments (Figure 3.4-26). When compared seasonally there were no significant differences between treatments. There were no clear patterns of surface roughness decline over time.

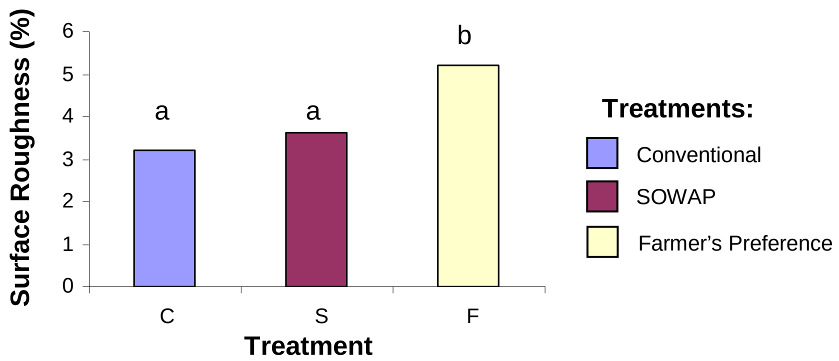


Figure 3.4-26 Loddington: soil surface roughness. Lettering indicates significant differences

3.4.2 Tivington

3.4.2.1 Hypothesis to be tested

The mean runoff, soil loss and associated nutrient and carbon losses over all tank clearances will be higher from the conventionally tilled plots in comparison to the conservation treatments (SOWAP and Farmer's Preference).

3.4.2.2 Runoff

3.4.2.2.1 Runoff volume

The volume of runoff generated from the field erosion plots was measured and treatments compared. The data were not normally distributed, so were transformed (Type I) to allow analysis of variance to be undertaken. Analysis was carried out on the overall mean runoff volume. No statistically significant differences were found between treatments (Figure 3.4-27). Despite this there appears to be a trend with runoff volumes from the SOWAP treatment being lower than the other two treatments. This was trend was reflected during season one and three (Figure 3.4-27). However, when the data were compared seasonally there were no significant differences between treatments. Analysis over all tank clearances also produced no statistically significant differences between treatments.

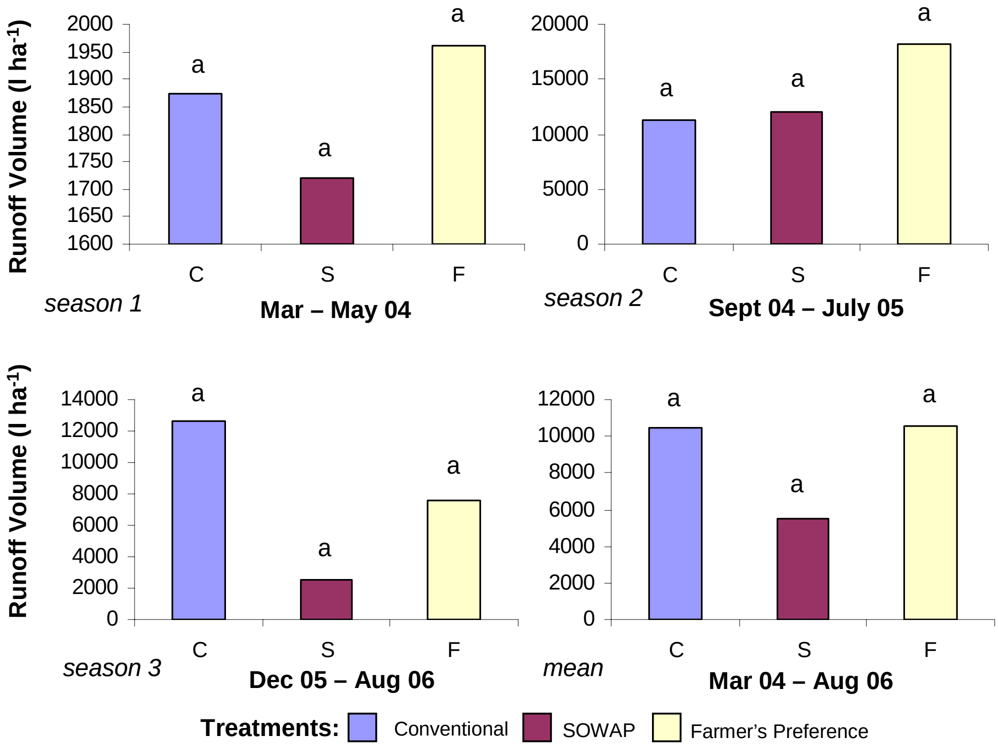
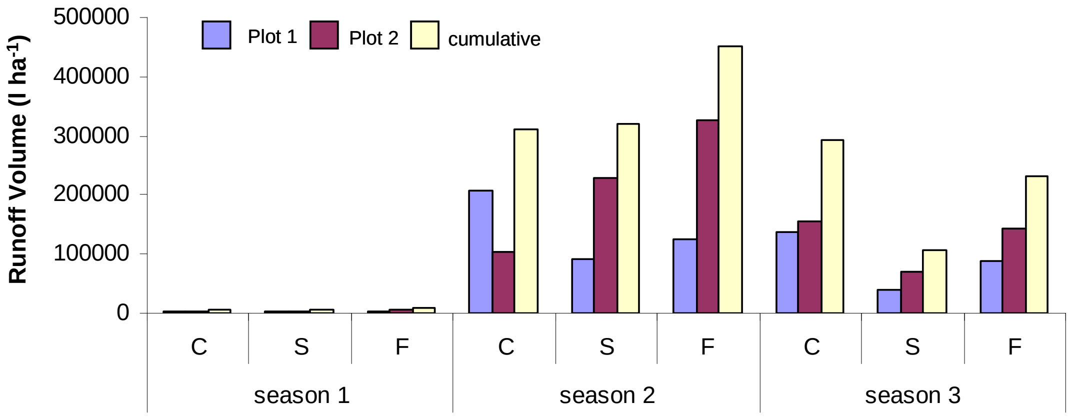


Figure 3.4-27 Tivington: runoff volumes for each season and overall. Letters indicate significant differences.



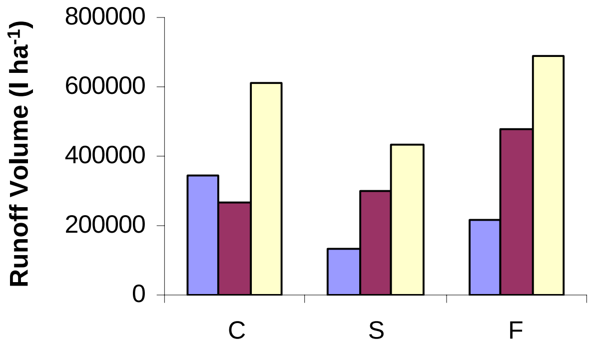


Figure 3.4-28 Tivington: runoff volume for each erosion plot and the combined cumulative results for each season (above) and overall (left)

Treatments:

C=conventional

S=SOWAP

F=Farmer's Preference.

The cumulative results for each treatment over the entire sampling period and for each season are presented in Figure 3.4-28. The graphs show the variation between plots of the same treatment. Variation was found for all treatments. The least volume of runoff was generated during the first season, accounting for only 1.3% of the total runoff generated. The majority of runoff was generated during season 2, accounting for over 62% of the total. This was particularly true for both conservation treatments, although the conventional treatment generated similar amounts for both the second and third season. Table 3.4-9 presents the relative percentages of runoff contributed by each treatment and in each season, compared to the total volume of runoff generated. The increase in generated runoff during season two is most likely due to the increased amount of rainfall received, as shown in Figure 3.4-29.

Table 3.4-9 Tivington: relative percentage of overall cumulative results from each treatment and for each season compared to the total runoff generated.

%	Conventional	SOWAP	Farmer's Preference	Total
Season 1	0.43	0.40	0.48	1.32
Season 2	17.87	18.49	25.95	62.31
Season 3	16.88	6.20	13.29	36.37
Total	35.19	25.09	39.73	100.00

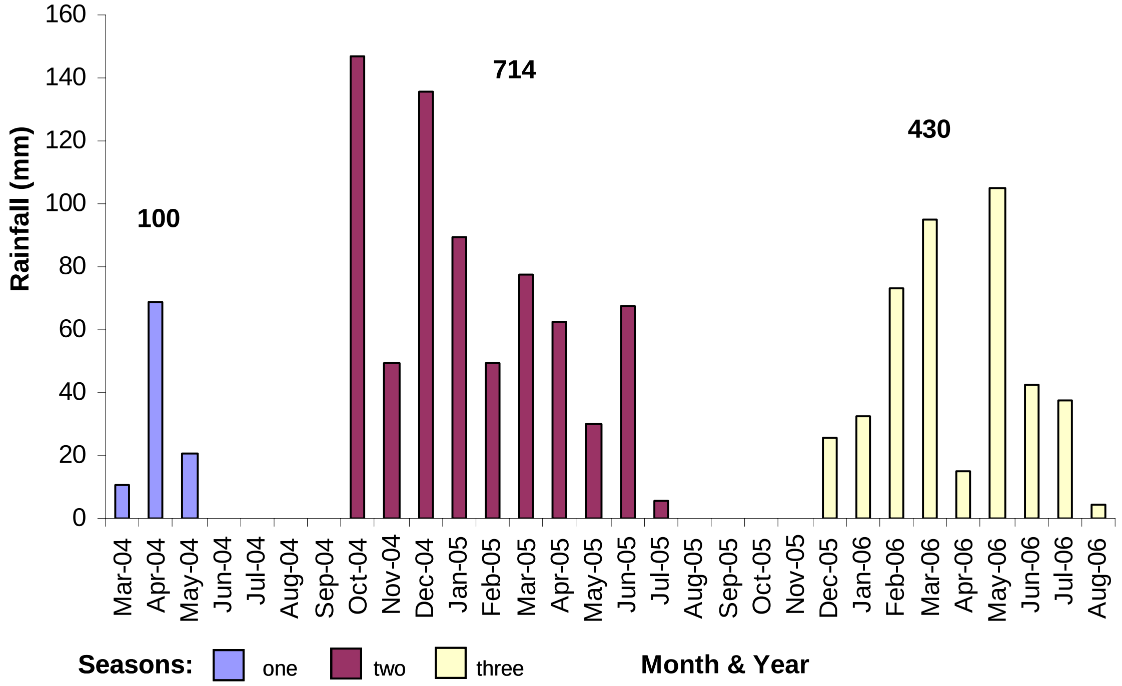


Figure 3.4-29 Tivington: precipitation data for each season. The data shown represents rainfall from installation to removal of the field erosion plots. Floating numbers above each season indicate total rainfall (mm) received for that period of time.

3.4.2.2.2 Nutrient Loss

Runoff generated from the field erosion plots was analysed for nutrient concentrations and total load. Loads were a function of the generated runoff volume. The nutrients analysed were nitrogen, phosphorus and potassium. Each nutrient will be addressed separately in the following text.

Nitrogen

Soluble nitrate was measured in the runoff to indicate nitrogen concentration and total load. Both the nitrogen concentration and load in the runoff were not normally distributed, so were normalised (Type I) to allow ANOVA to be carried out. Statistical analysis of runoff nitrogen concentration and load showed no significant differences between any of the treatments when compared at any of the 3 temporal scales - over the entire sampling period, seasonally and across all tank clearances. The mean nitrate concentrations ranged from 1.6 to 2.0 mg l^{-1}, and nitrate loads ranged from 6.7 to 14.0 g ha^{-1} for the three treatments (actual results can be found in Table 3.4-10).

Table 3.4-10 Tivington: overall mean nitrogen over the entire sampling period

Treatment	Concentration (mg l^{-1})	Loading (g ha^{-1})
Conventional	1.94	14.03
SOWAP	1.59	6.66
Farmer's Preference	1.97	12.75

Phosphorus

Collected runoff was analysed for phosphorus concentration and loading. The results of both were not normally distributed, so a Type I transformation was performed. Statistical analysis of both phosphorus concentration and load showed no significant differences between treatments means when compared over the entire sampling period, seasonally or across all tank clearances. The mean phosphorus concentrations and loadings for the treatments ranged from 0.11-0.28 mg l^{-1} and 0.74-2.60 g ha^{-1}. Actual results are presented in Table 3.4-11.

Table 3.4-11 Tivington: overall mean phosphorus over the entire sampling period

Treatment	Concentration (mg l^{-1})	Loading (g ha^{-1})
Conventional	0.11	0.74
SOWAP	0.13	0.82
Farmer's Preference	0.28	2.60

Potassium

Potassium concentrations and loads were measured in the runoff generated from the field erosion plots. The results were not normally distributed and so were transformed (Type II) to allow for statistical analysis. There were no statistically significant differences between treatment means for K concentrations or loads when compared over the 3 temporal scales - over the entire sampling period, on a seasonal basis or across all tank clearances. The mean concentrations of potassium overall were between 9 and 15 mg l^{-1} and between 40 and 89 g ha^{-1} for total loads in the runoff. Actual results are presented in Table 3.4-12.

Table 3.4-12 Tivington: overall mean soluble potassium over the entire sampling period

Treatment	Concentration (mg l^{-1})	Loading (g ha^{-1})
Conventional	9.05	62.98
SOWAP	12.79	39.56
Farmer's Preference	14.97	88.55

3.4.2.2.3 Carbon Loss

The runoff generated from the field erosion plots was analysed for concentrations and loads of total organic carbon for each treatment. The data were not normally distributed, so a Type IV transformation was carried out. The results were analysed at three different temporal scales – for the entire sampling period, on a seasonal basis and across all tank clearances. Concentrations of total organic carbon in the runoff were compared between treatments. The lowest concentration was from the SOWAP treatment (5.0 mg l^{-1}), and the highest from the conventional treatment (6.2 mg l^{-1}); Farmer's Preference has a TOC concentration of 5.8 mg l^{-1}. These were the results over the entire sampling period. Satistical analysis showed no significant differences at any of the temporal scales.

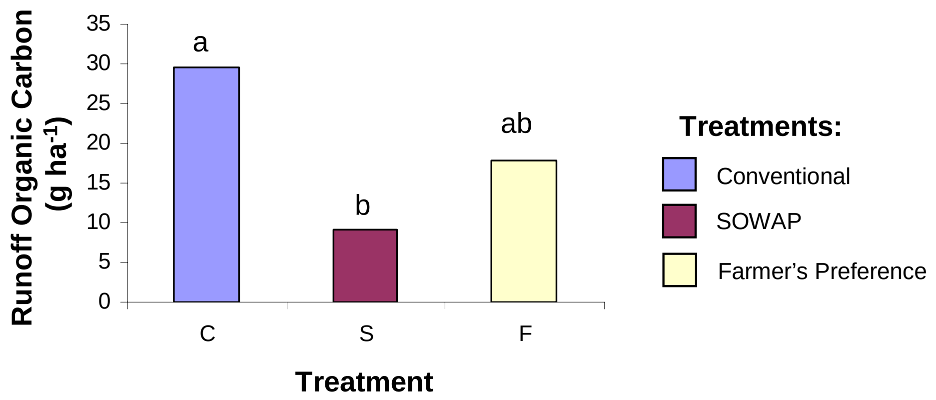


Figure 3.4-30 Tivington: Runoff Organic Carbon loading, over entire sampling period. Letters denote significant differences.

Statistical analysis of the total loads of organic carbon in the runoff (as calculated from the TOC concentration and runoff volume) did show significant differences between treatments over the entire sampling period (p=0.046). Organic carbon load in the runoff was significantly higher for the conventional treatment compared to the conservation treatment, SOWAP (Figure 3.4-30).

There were no significant differences between treatment means when compared on a seasonal basis, but this was not the case when analysed for each tank clearance (p=0.046). Graphical outputs of organic carbon load for all measured tank clearances can be found in Figure 3.4-31. The tank clearance on the 5th January 2006 has been highlighted here, because of the increased loads of organic carbon measured at this clearance.

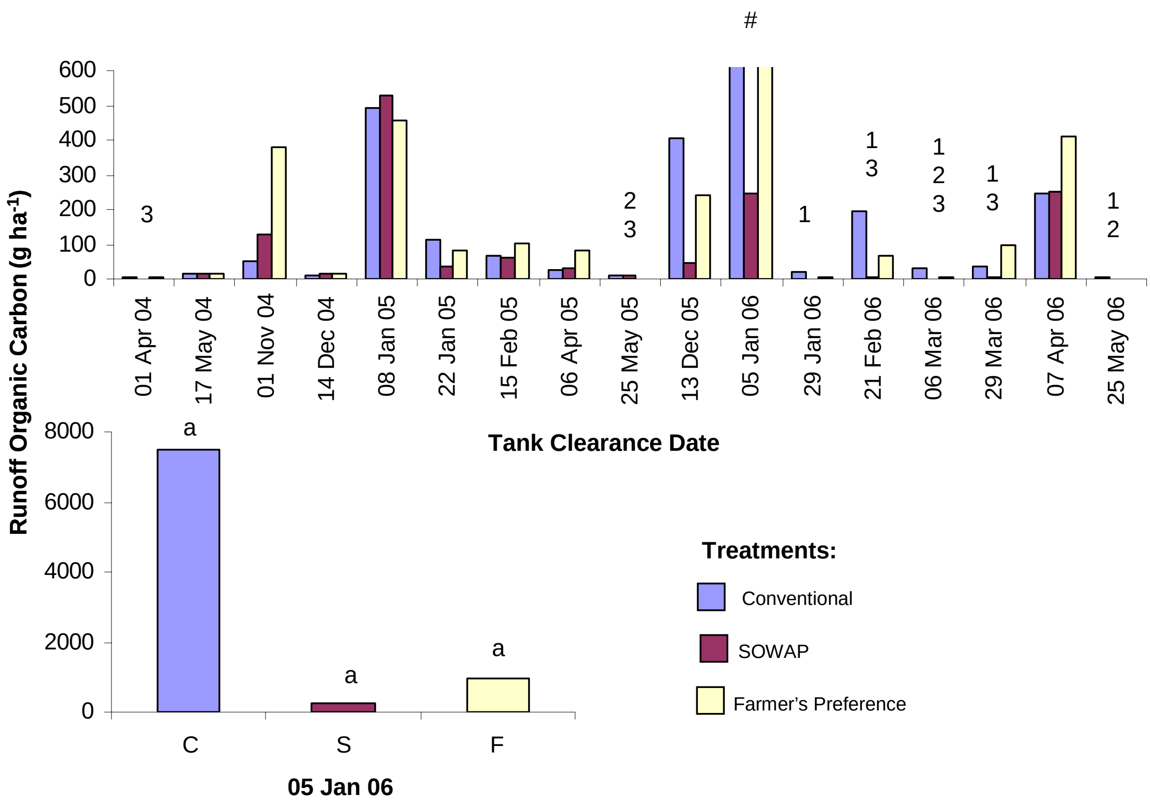


Figure 3.4-31 Tivington: Runoff Organic Carbon loading across all treatments (top) and the 5th January 2006 (bottom left). Floating letters and number symbolise significant differences between: 1= SOWAP and conventional, 2=Farmer's Preference and =SOWAP and Farmer's Preference

3.4.2.3 Soil Loss

3.4.2.3.1 Sediment

Soil losses at three temporal scales were investigated – over the entire sampling period, on a seasonal basis and for each tank clearance. The data were not normally distributed and so a Type I transformed was performed to allow statistical analysis.

This statistical analysis of the sediment data showed a significant difference between treatments ($p<0.001$) when analysed over the entire sampling period. Soil loss was greater from the conventional treatment, but in comparison to the SOWAP treatment only. This pattern was also found during the second and third seasons, although no significant differences were found. These results are presented in Figure 3.4-32.

Significant differences were found when treatment means were analysed across all tank clearances ($p<0.001$). Out of the 19 tank clearances, 10 of them showed significant treatment differences (Figure 3.4-33). In all cases bar one soil loss was highest from the conventional treatment compared to at least one of the conservation treatments. The one exception was during the third tank clearance, where soil loss was greatest from the Farmer's Preference treatment.

The cumulative soil loss results have been presented in Figure 3.4-34 for each season and over the entire sampling period. Losses are greater from the conventional treatment compared to both conservation treatments. The cumulative results also show the variation between the duplicate field plots of each treatment. This is the case for all treatments.

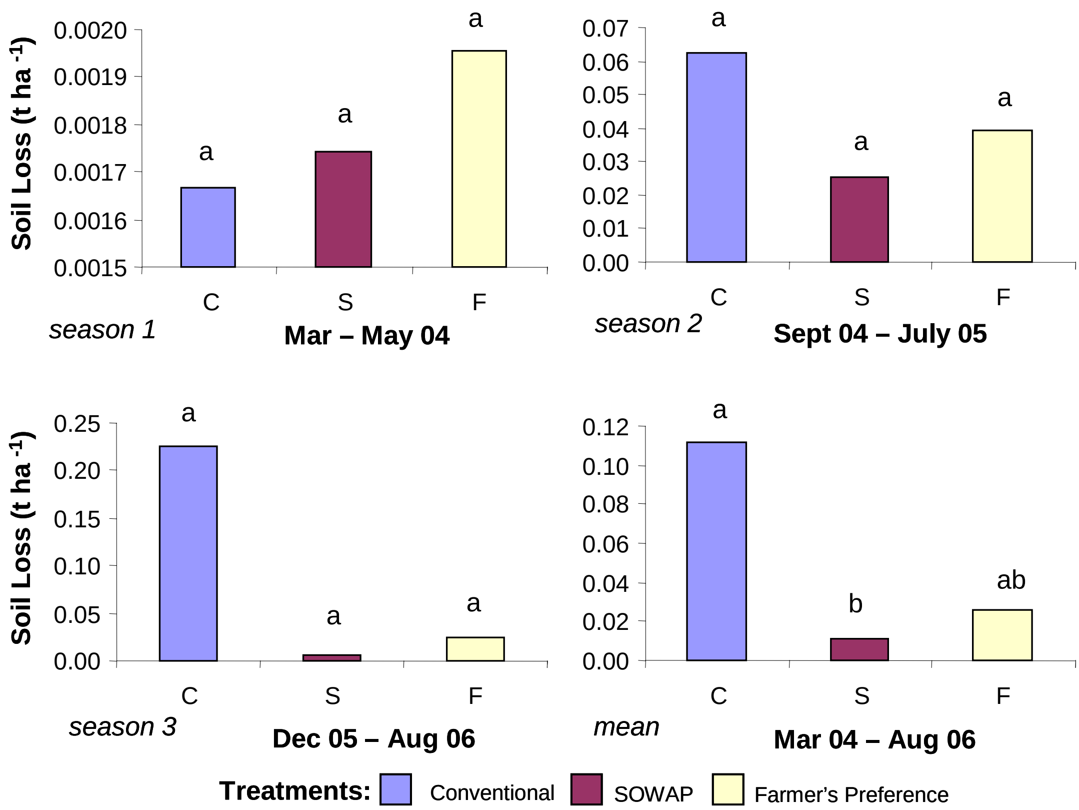


Figure 3.4-32 Tivington: mean soil loss for each season and overall. Letters denote significant differences.

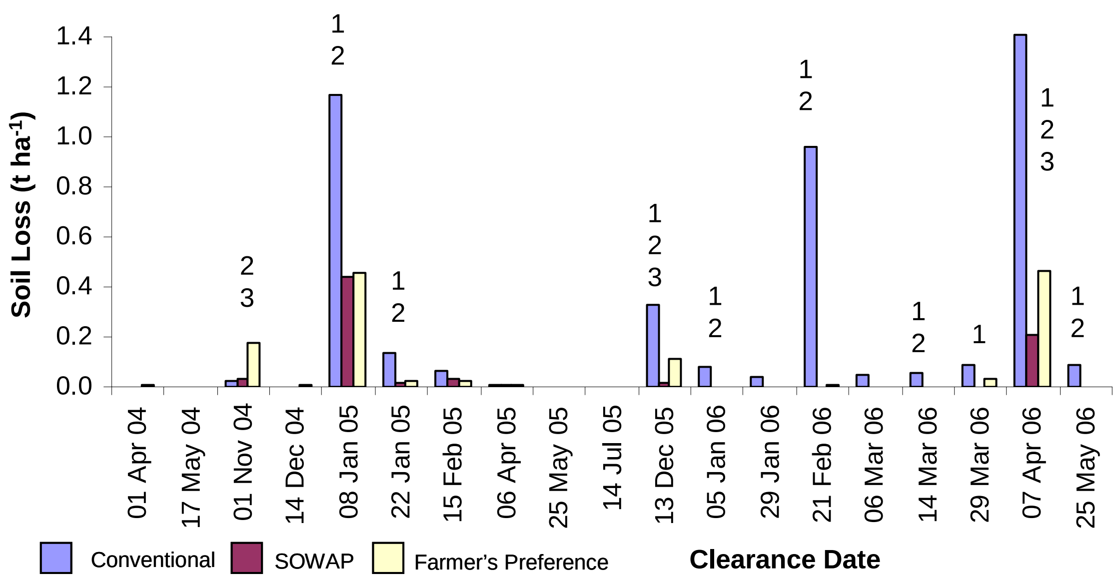
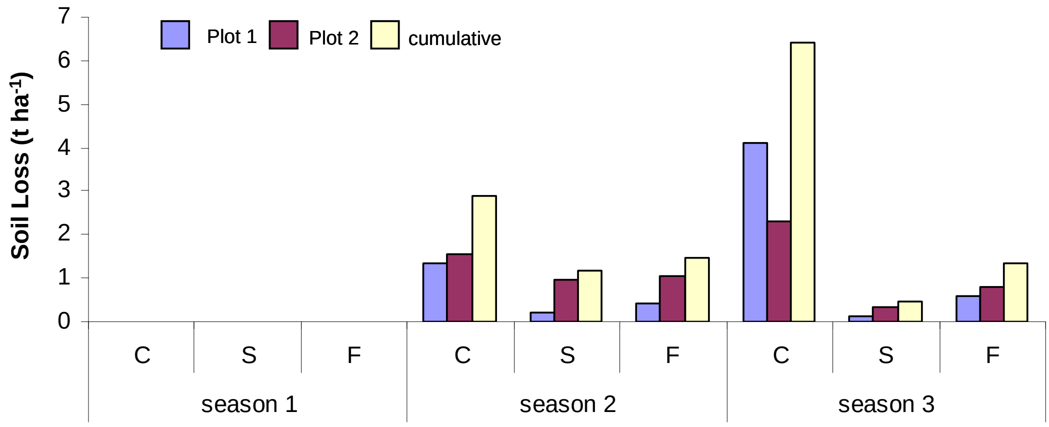


Figure 3.4-33 Tivington: soil loss across all tank clearances. Asterisks indicate where significant treatment differences occurred. Floating numbers symbolise significant differences between: 1= SOWAP and conventional, 2=Farmer's Preference and =SOWAP and Farmer's Preference



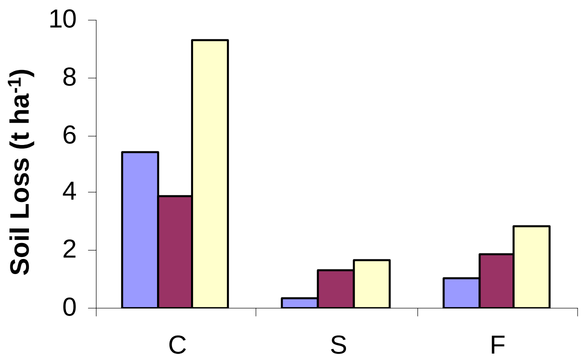


Figure 3.4-34 Tivington: soil loss (t ha^{-1}) for each erosion plot and the combined cumulative results for each season (above) and overall (left)

Treatments:

C=conventional

S=SOWAP

F=Farmer's Preference

The majority of soil loss from the conventional treatment occurred during the third season, during the second season for the SOWAP treatment and both the second and third season for the Farmer's Preference treatment. Table 3.4-13 presents a breakdown of soil loss for each treatment and each season as a relative percentage to the total soil lost over the entire sampling period

Table 3.4-13 Tivington: relative percentage of overall cumulative results from each treatment and for each season compared to the total soil loss.

%	Conventional	SOWAP	Farmer's Preference	Total
Season 1	0.06	0.05	0.07	0.18
Season 2	20.79	8.64	10.73	40.16
Season 3	46.42	3.39	9.85	59.66
Total	67.27	12.09	20.65	100.00

3.4.2.3.2 Nutrient Loss

Eroded soil was collected from the field erosion plots and analysed for nutrient concentration and load. The loads were a function of nutrient concentration and soil loss. The nutrients measured were total nitrogen, phosphorus and potassium. It was not always possible to analyse the sediment for nutrient content due to insufficient amounts collected during tank clearances, therefore only two temporal scales were investigated - over the entire sampling period and for each season.

Nitrogen

The concentration of nitrogen within the eroded soil was normally distributed and was statistically analysed for differences between treatment means. When treatments were compared over the entire sampling period concentrations of nitrogen were significantly lower ($p<0.001$) from the conventional treatment in relation to both the SOWAP and Farmer's Preference treatments (Figure 3.4-35). However, when nitrogen concentrations were analysed on a season basis, no significant treatment differences were found.

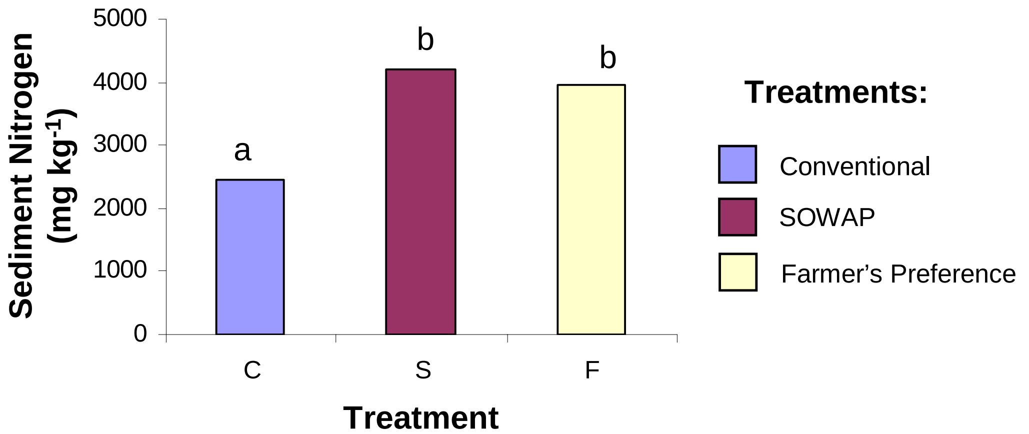


Figure 3.4-35 Tivington: Nitrogen concentration in sediment. Letters denote significant differences.

The total loading of nitrogen within the sediment was also statistically analysed (after being normalised via a Type I transformation) over the entire sampling period and for each season, but no significant treatment differences were found.

The mean nitrogen load in the sediment ranged 10 to 17 g ha^{-1}. Actual results are presented in Table 3.4-14.

Table 3.4-14 Tivington: mean nitrogen load over the entire sampling period

Treatment	Loading (g ha^{-1})
Conventional	292
SOWAP	105
Farmer's Preference	158

Phosphorus

Eroded soil was analysed for phosphorus concentration and total load. The phosphorus concentrations were normally distributed, whereas the total load data has to be normalised (Type I). Statistical analysis of phosphorus concentrations over the entire sampling period showed significant differences between treatment means (p<0.001), with the lowest P concentration found in the sediment from the conventional treatment, when compared to both conservation treatments (Figure 3.4-36). When analysed on a seasonal basis no significant treatment differences were found.

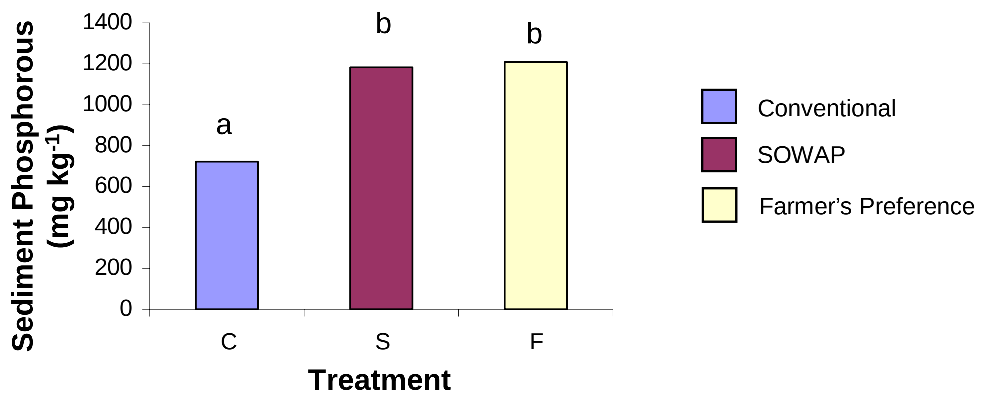


Figure 3.4-36 Tivington: Phosphorus concentration in sediment. Letters denote significant differences.

No statistically significant treatment differences were found when the total loads of phosphorus in the sediment were compared over the entire sampling period or for each season. The overall mean phosphorus loads in the sediment were 86 g ha^{-1} for the conventional treatment, and 32 and 49 g ha^{-1} for the SOWAP and Farmer's Preference treatments, respectively.

Potassium

Potassium concentrations and total loads were measured in the eroded soil collected from the field erosion plots. The results were statistically analysed for treatment differences over the entire sampling period and on a seasonal basis. The only significant treatment differences were found in sediment-associated potassium concentrations over the entire sampling period (p<0.01), where the concentrations from the conventional plots were significantly lower than from either of the conservation plots (Figure 3.4-37). The results of the mean potassium load within the sediment are presented in Table 3.4-15.

Table 3.4-15 Tivington: mean total potassium loads in sediment over the entire sampling period

Treatment	Loading (g ha^{-1})
Conventional	610
SOWAP	191
Farmer's Preference	302

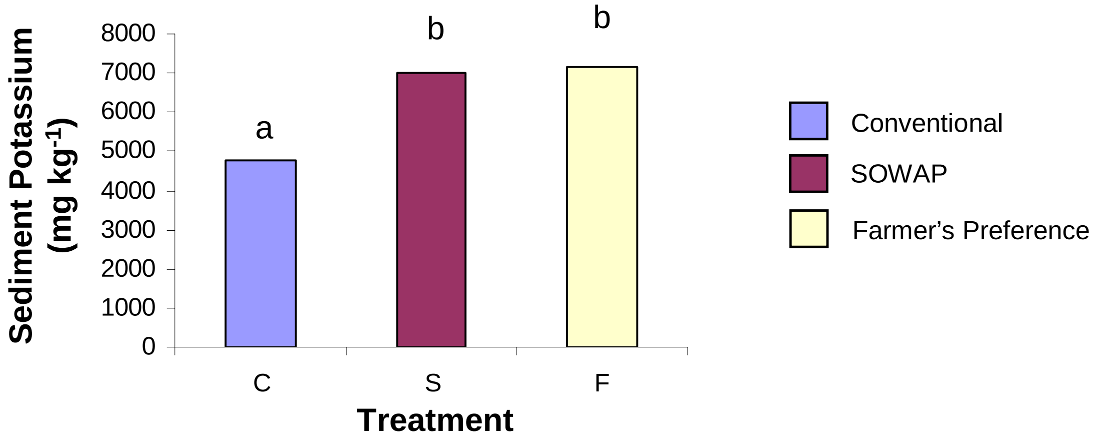


Figure 3.4-37 Tivington: Potassium concentrations in sediment. Letters denote significant differences.

3.4.2.3.3 Carbon Loss

Eroded soil was analysed for carbon concentration and total load. Data analysis for individual tank clearances could not be done due to insufficient sediment samples generated for analysis. Over the entire sampling period and for each season, organic carbon concentration data were normally distributed, whereas the load results had to be transformed (Type I). Statistical analysis showed no statistically significant differences between treatment means of organic carbon concentration or loading within the sediment at either temporal scale. The overall mean results of both concentration and loads are presented in Table 3.4-16.

Table 3.4-16 Tivington: mean carbon loss over the entire sampling period

Treatment	Concentration (mg kg^{-1})	Loading (g ha^{-1})
Conventional	24000	4497
SOWAP	26000	1924
Farmer's Preference	30000	1907

3.4.2.4 Additional analysis

Additional analysis was carried out on the sediment generated from the field erosion plots, including organic matter content and particle size. Correlations where carried out between these factors, including runoff volume and the amount of soil lost. Additional field soil properties and surface characteristics were also quantified with measurements which were taken adjacent to and within the field erosion plots to give supporting information for observed treatment differences. These have been referred to previously in section 3.3.2.

3.4.2.4.1 Correlations

The volume of runoff and mass of eroded soil were correlated against each other to see if there was a relationship. A simple correlation of normalised (both Type I) runoff and soil loss data showed a significant positive relationship (r=0.81). Therefore as runoff increased, so did the amount of soil lost (Figure 3.4-38).

There were so many significant correlations that they have been presented in a table rather than showing the graphical outputs (Table 3.4-17).

Variables	**Runoff**	**Soil Loss**	**Susp. Sed.**	**S-O.M. %**	**S-Sand %**	**S-Silt %**	**S-Clay %**
Runoff	n/a	0.81	0.95	0.29	-0.48	0.54	0.36
Soil Loss	0.81	n/a	0.87	x	-0.34	0.56	x
R-Nitrogen conc.	-0.22	-0.26	-0.29	-0.30	0.49	-0.32	-0.49
R-Phosphorus conc.	x	x	x	x	x	x	x
R-Potassium conc.	x	-0.22	x	0.49	x	x	x
R-Organic Carbon conc.	-0.22	-0.24	-0.20	-0.34	0.46	-0.49	-0.37
S-Nitrogen conc.	x	x	x	0.78	-0.69	0.33	0.76
S-Phosphorus conc.	x	x	x	0.77	-0.66	0.39	0.69
S-Potassium conc.	0.25	x	0.25	0.64	-0.79	0.49	0.81
S-Organic Carbon conc.	x	x	0.30	0.59	-0.63	0.42	0.66
S-O.M. %	0.29	x	0.26	n/a	-0.70	0.42	0.72
S-Sand %	-0.48	-0.34	-0.51	-0.70	n/a	n/a	n/a
S-Silt %	0.54	0.56	0.56	0.42	n/a	n/a	n/a
S-Clay %	0.36	x	0.39	0.72	n/a	n/a	n/a

Table 3.4-17 Tivington: table of significant correlations ($p<0.05$). S- before a variable denotes it is associated with the sediment e.g. S nitrogen is the nitrogen associated with the sediment. R- variable associated with the runoff. conc. = concentration. x = not significant. Suspended sediment is denoted as Susp. Sed.

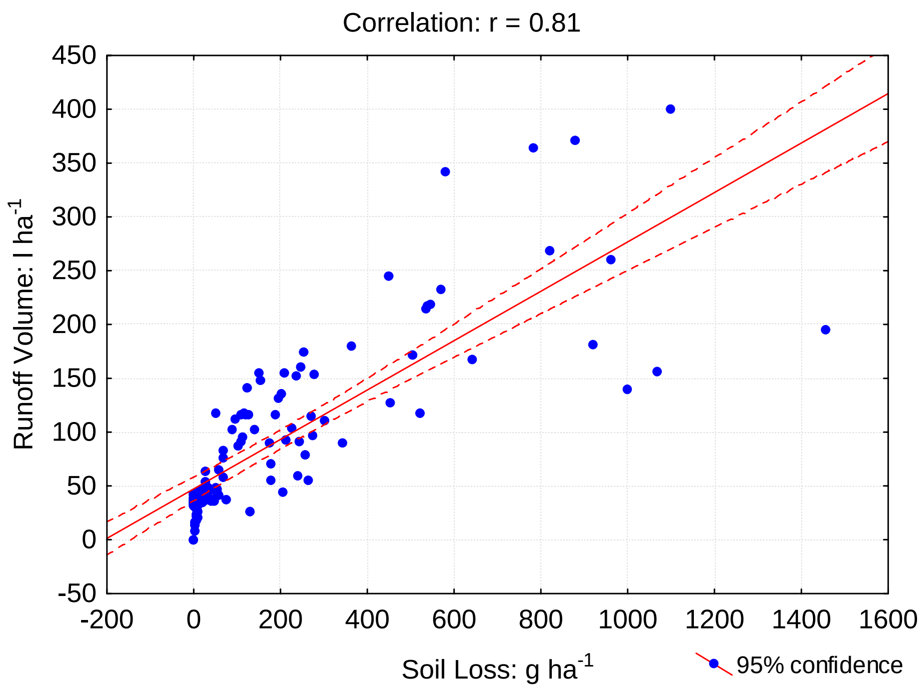


Figure 3.4-38 Tivington: correlation between runoff and soil loss (n=114, p<0.05)

3.4.2.4.2 Field Soil Properties

The soil properties which were measured were soil particle size (texture), organic matter content, moisture content (gravimetric and volumetric) and bulk density. These soil properties were measured twice a year, during spring and autumn. The properties have been measured to explain any observed differences found in runoff volume and soil, nutrient and carbon loss. More details as to why these soil properties were measured can be found in 1.2.5. Soil properties were measured in conjunction with the micro-plot field rainfall simulations. These did not start until the beginning of the second season, therefore soil property results will only be expressed for the second (Sept 04 – July 05) and third (Dec 05 – Aug 05) season and as an overall mean for the entire sampling period.

Textural Analysis

The soil clay content will influence erosion and associated nutrient loss due to the strong cohesive forces between clay particles. It is expected that as the clay content increases the risk of soil erosion and loss of nutrients will decline. As the silt content increases it is expected that the risk of seal formation will increase, contributing to greater erosion risk.

The results of percentage clay, silt and sand were all normally distributed with equal variance. Statistical analysis was carried out at two temporal scales; over the entire period of sampling and for each season. Significant differences between treatment means were found at both temporal scales for silt (p=0.01) and sand (p=0.03) content within the soil. Significant differences between means were only found over the entire sampling period for clay content within the soil (p=0.04), where the treatment differences were small. Graphical outputs can be found in Figure 3.4-39.

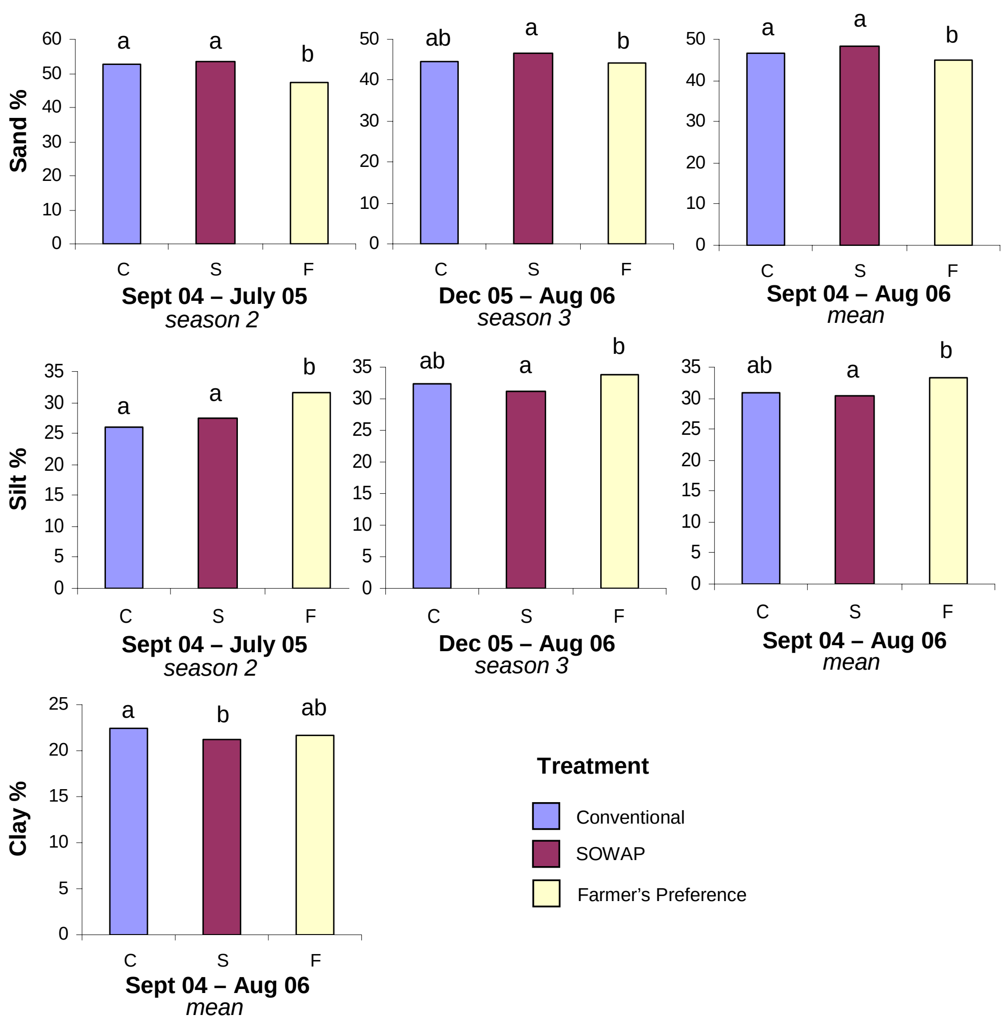


Figure 3.4-39 Tivington: Textural results from field soil samples. Letters denote significant differences.

Organic Matter

An increase of soil organic matter is expected to have a positive relationship with aggregate stability and a negative relationship with the formation of surface seals and crusts. Consequently, as SOM increases, erosion is expected to decrease.

The results of organic matter content in the soil were normally distributed with equal variance, allowing statistical analysis to be undertaken. No significant differences were found between treatment means over the entire sampling period or on a seasonal basis. Overall mean results are presented in Table 3.4-18.

Table 3.4-18 Tivington: overall mean SOM content

Treatment	SOM content (%)	SD	n
Conventional	1.83	0.35	12
SOWAP	2.00	0.36	12
Farmer's Preference	1.80	0.41	12

Moisture Content

It is expected that any differences in moisture content may contribute to explaining observed erosion results, as moisture content has been linked to the resistance of soil to raindrop impact, and having an influence on infiltration rates.

Gravimetric moisture content represents the wetness of a soil i.e. the mass of water per unit mass of soil. Statistical analysis showed that overall there was a significant difference (p=0.02) between the conventional and Farmer's Preference means (Figure 3.4-40) in gravimetric moisture content. On a seasonal basis there were no treatment differences.

The volumetric moisture content also represents soil water content, as a volume of water per volume of soil. Statistical analysis of the volumetric moisture content showed there to be no significant treatment differences when analysed over the entire sampling period. However, differences existed (p=0.04) between treatment means on a seasonal basis (Figure 3.4-41).

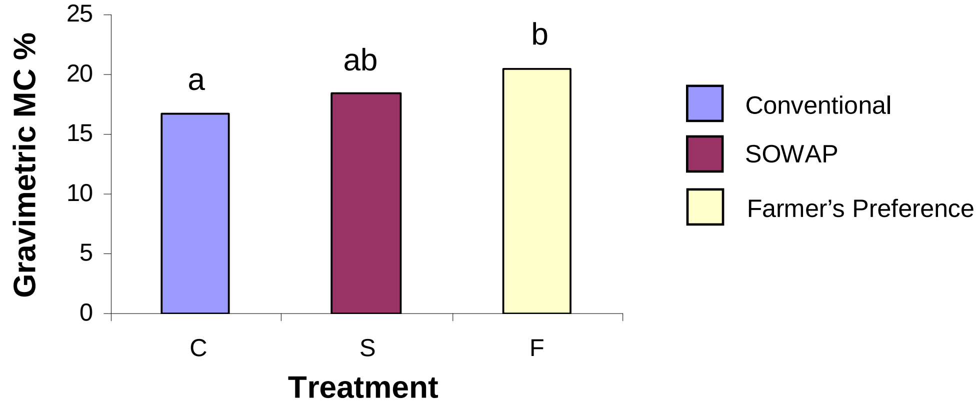


Figure 3.4-40 Tivington: Gravimetric moisture content over the entire sampling period. Letters denote significant differences.

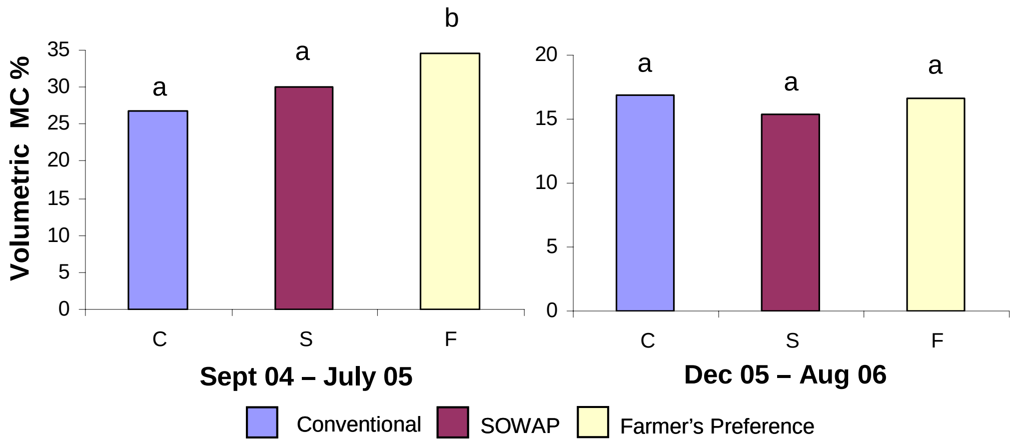


Figure 3.4-41 Tivington: seasonal difference in volumetric moisture content. Letters denote significant differences

Bulk Density

Bulk density has been found to have a negative association with infiltration rates and drainage. It is expected that as bulk densities increase, runoff and subsequent soil erosion will also increase.

The results of bulk density in the field soil were normally distributed, with equal variance, allowing statistical analysis to be undertaken. Bulk density results were compared between treatments over the entire sampling period and for each season. However, no significant differences were found at either temporal scale. Results are presented in Table 3.4-19.

Table 3.4-19 Tivington: overall mean bulk density

Treatment	Bulk density (g cm^{-3})	SD	n
Conventional	1.32	0.36	36
SOWAP	1.22	0.39	36
Farmer's Preference	1.22	0.39	36

Total Organic Carbon

Total organic carbon (TOC) has been shown to have positive relationships with aggregate stability and a reduction in erosion. The results of total organic carbon in the field soil were normally distributed with equal variance. When the results were analysed over the entire sampling period, no statistically significant differences were found between treatment means. When compared on a seasonal basis significant differences were found (p<0.01) between treatments, but only for the second season (September 2004 to July 2005), as shown in Figure 3.4-42. Mean results over the entire sampling period for each treatment were 0.94 % from the conventional treatment, 1.18 and 1.04 % for the SOWAP and Farmer's Preference conservation treatments.

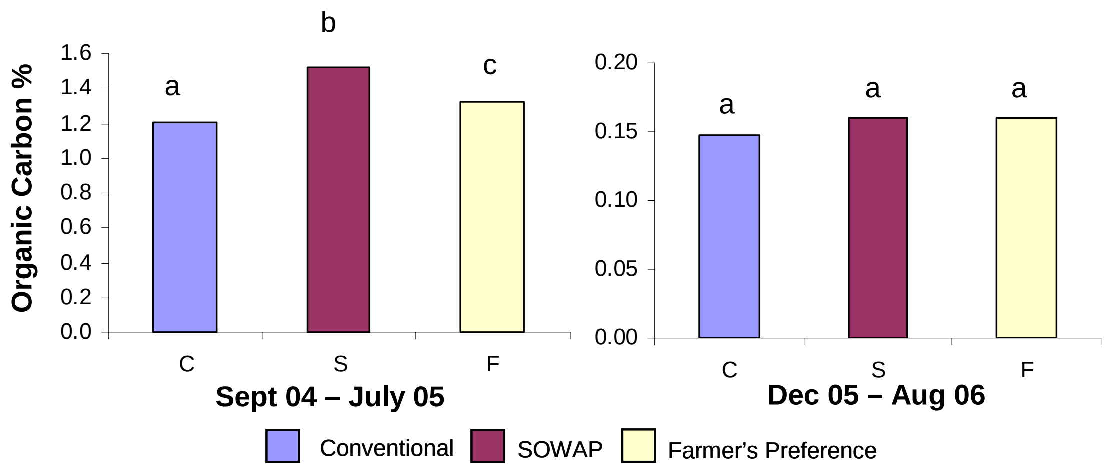


Figure 3.4-42 Tivington: seasonal comparison of organic carbon content in field soil. Letters denote significant differences

3.4.2.4.3 Field Soil Surface Characteristics

Soil Cover

Soil cover was assessed by measurements of percentage bare or exposed soil, the percentage of residues, weeds and stones. The presence of any surface cover physically protects the soil surface from direct rainfall impact, minimising soil detachment. It is therefore expected that an increase in cover will contribute to a reduction in erosion. The results are presented in the following text.

The percentage bare soil was statistically analysed over the entire sampling period and for each season separately. When analysed over the entire sampling period, the percentage of bare soil was significantly higher (p<0.01) for the conventional field erosion plots in comparison to both conservation treatments (Figure 3.4-43). However, when analysed on a seasonal basis, no differences were found.

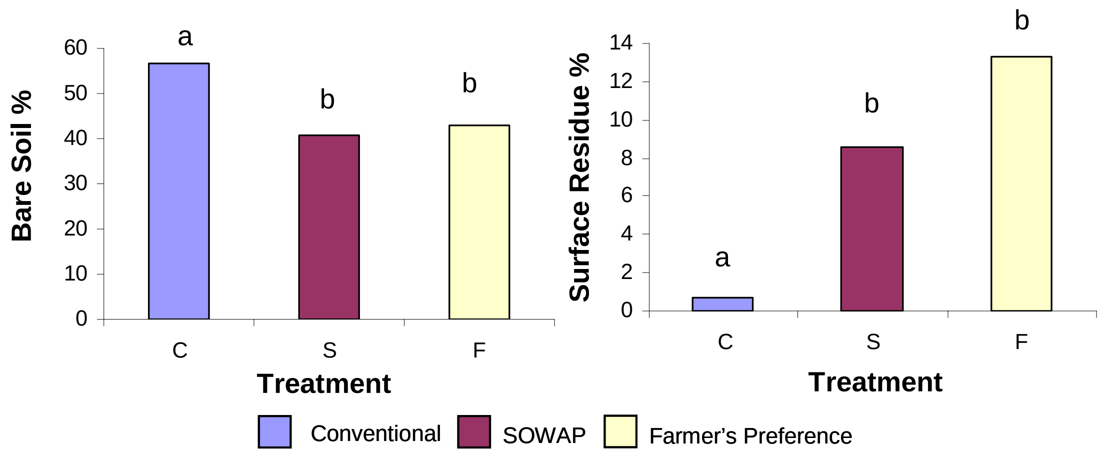


Figure 3.4-43 Tivington: percentage bare soil over the entire sampling period. Letters indicate significant differences.

The percentage of surface residues present on the conventional plots was statistically lower (p<0.001) than for both conservation treatments, when analysed over the entire sampling period (Figure 3.4-43), but no significant differences were found on a seasonal basis

The percentage weed cover was also analysed over the entire sampling period and for each season. At both scales significant differences ($p<0.001$) in percentage weed cover were found between treatments (Figure 3.4-44). The percentage of weed cover was significantly lower on the conventional plots compared to the SOWAP treatment except during season one. The mean (over the entire sampling period) results also showed the same treatment pattern.

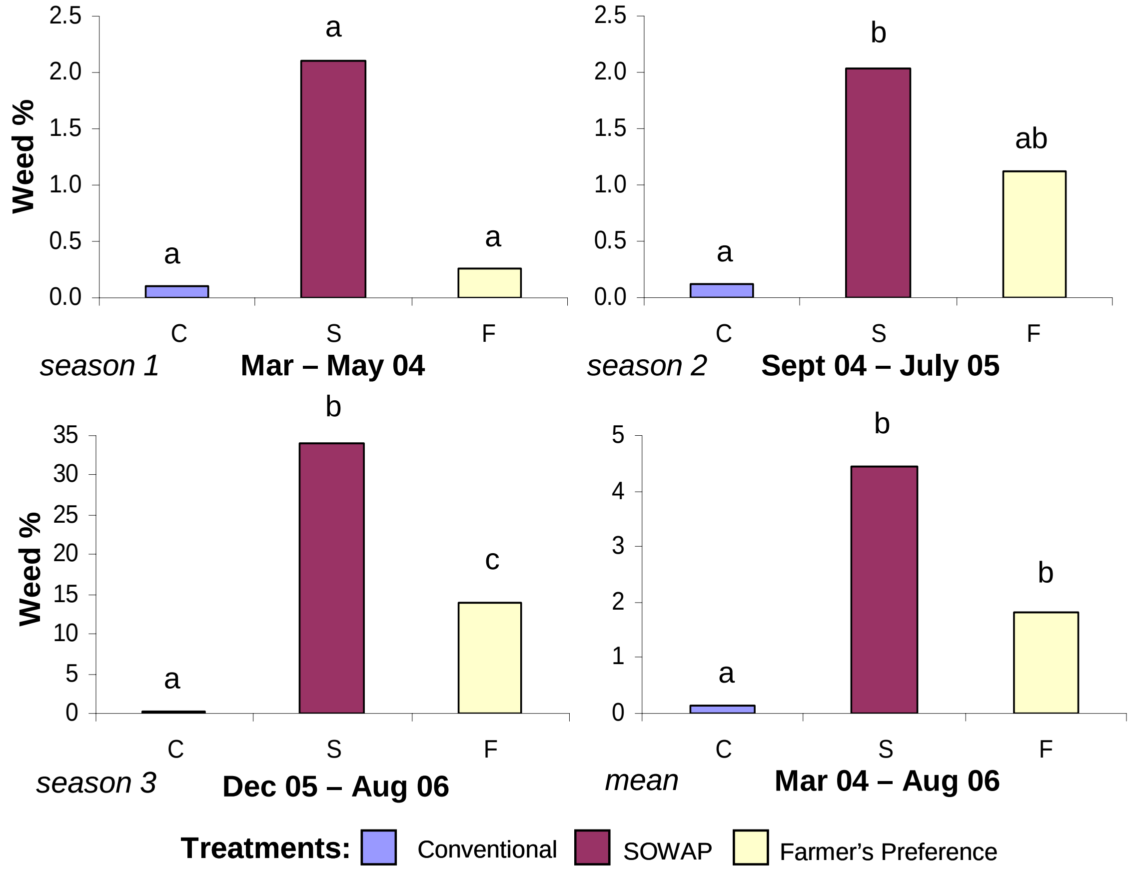


Figure 3.4-44 Tivington: Percentage soil cover from weeds. Letters indicate significant differences.

Statistical analysis of the percentage stone surface cover showed no significant treatment differences at either temporal scale. The stone cover for all three treatments ranged from 1.7 to 2.3%.

Surface Roughness

Increases in surface roughness have been linked to a reduction of overland flow (volume and velocity) and loss of sediment. The roughness of the surface soil on the field erosion plots was measured and statistically analysed for treatment differences over the entire sampling period and on a seasonal basis. Analysis showed that there were no significant differences between treatment means at either temporal scale. Mean results over the entire sampling period are presented in Table 3.4-20.

Table 3.4-20 Tivington: overall mean surface roughness

Treatment	Surface roughness (%)	SE	n
Conventional	5.06	0.38	51
SOWAP	6.72	0.51	50
Farmer's Preference	5.76	0.53	51

3.5 Discussion

3.5.1 Findings in relation to the original objectives

It is expected that slight inconsistencies may exist in the treatment results between site locations due to their inherent differences. These variations between sites should be addressed before the results are discussed in relation to the original objectives of this chapter. The Loddington site is located on clay (45%), whereas the Tivington site is a sandy clay loam (only 20% clay). As previously mentioned (1.2.5.2.1) clay has a positive effect on aggregate stability by reducing soil erodibility. The clay fraction is also associated with higher nutrient and carbon adsorption due to clay particles having a relatively high specific surface area. The soil organic matter content is 5.2% at the Loddington site and 0.8% at Tivington. Organic matter (1.2.5.2.2) also has a positive effect on aggregate stability and nutrient content. The field erosion plots at the Loddington site are on a more gentle slope of 3.5%, while at Tivington the slope is steeper at 7%. If all other factors were equal an increase in slope gradient would lead to an increase risk of erosion. These are important factors to consider during the discussion.

3.5.1.1 Objective one: to quantify runoff volume and soil loss under natural rainfall for three different soil management treatments

Soil is an important resource which is being lost through erosion. The benefits of minimising soil erosion are vast but include an increase in crop productivity, maintenance of soil health, conservation of the biological community and reduction in off-site pollution of water courses.

It was predicted in hypothesis one that runoff volume and soil loss would be significantly greater from the conventionally treated field erosion plots, compared to either or both of the conservation treatments - SOWAP and Farmer's Preference. Treatment means were analysed at three different temporal scales; over the entire sampling period, on a seasonal basis and across the individual tank clearances.

There were no significant treatment differences in runoff volume when analysed at all three temporal scales for both sites. Although not significant, noteworthy treatment differences were found at both sites. These differences were not consistent to site location. At Tivington, the SOWAP conservation treatment generated substantially lower volumes of runoff over the course of the project, contributing to only 25% to the total runoff generated from all treatments. Although this was expected, what was unexpected was that the other conservation tillage regime, Farmer's Preference, generated similar volumes of runoff to the conventional treatment. At Loddington there were also notable differences between treatments. The conventional treatment generated substantially lower volumes of runoff compared to both conservation treatments when mean and cumulative results were compared over the entire sampling period.

The results from both sites were unexpected as *a priori* reasoning would suggest that runoff should be greater from the conventional plots, due to the increased trafficking and thus compaction on this treatment, lower percentage cover to trap surface water leading to runoff generation, and greater breakdown of disturbed aggregates by raindrop impact, leading to the formation of a less permeable soil surface. However, there is some evidence in the literature to show that the long-term use of conservation tillage practices under some circumstances may actually enhance runoff generation (Moldenhauer et al. 1971; Lindstrom & Onstad 1984 and Mueller et al 1984). The low level of soil disturbance in the plough layer (unlike conventional tillage which disturbs this layer) can lead to soil consolidation over time, increasing bulk density and reducing infiltration, resulting in higher runoff generation. Such evidence, however, was not found in the present study. Over the entire sampling period, there was no significant change in bulk density at either site, for any treatment. Significant differences were observed on a seasonal basis, but only at Loddington. Where differences existed, bulk density was significantly higher on the conventional treatment compared to both conservation treatments. In any case, these differences in bulk density were not reflected in the volumes of runoff.

Treatment differences in generated volume of runoff could be related to changes in surface characteristics such as percentage cover by weeds, residues and stones. It was expected that the management applied to the conventional treatments would result in less soil cover, and a greater propensity for soil to generate runoff. This was found to be the case at Tivington, with the percentage bare soil and the surface roughness on the conventionally treated plots being either significantly higher or lower than one of the conservation treatments. Although this did not result in significant treatment differences in runoff volume, these changes in surface characteristics do explain the substantially higher runoff volumes generated from the conventional treatment. At Loddington the only significant difference in surface characteristics (over the entire sampling period) occurred in the roughness of the soil surface. Roughness was significantly higher on the Farmer Preference conservation treatment. This partially explains the runoff trend found, but not completely. Other processes must be operating.

The results have shown that significant differences in surface characteristics were found explaining in part the notable differences between treatments in terms of runoff volume. However, it is still unclear as to possible reasons why differences in runoff volumes between treatments were not significant. It is postulated that the lack of significant differences in runoff volume at either site was the result of a) data variability, and b) an extreme event (in the case of Loddington).

The majority of runoff volume generated from each treatment at any time interval was associated with high variability in the data. This was compounded by the fact that only 2 erosion plots were available for each treatment at each site – giving very high variability (Morgan 2005) and very low degrees of freedom when it comes to statistical testing. Significant differences may be seen if the experiments were to be run for a longer time period, and if more replication of treatments were possible. It should also be noted that man-made variability occurred during treatment cultivations, especially on the SOWAP plots at Tivington. This was through the presence of an additional tramline, which may have represented a preferential pathway for runoff and associated sediment. The consequence of this may have been an overestimation of erosion losses.

Plate 3.5-1 Additional tramline within a SOWAP plot at Tivington

Lack of treatment significance in runoff volume at the Loddington site specifically was also compounded by the unexpected runoff volumes from the conventional treatment. This was attributable to an extreme event on the 22nd

October 2004. The runoff volumes measured from this tank clearance (combined mean results from all treatments) represented 61.3% of the total runoff collected from all 11 clearances over the course of the entire project. The result from this clearance was therefore influential to the overall treatment differences (previously shown specific to the soil loss results). The percentage runoff volume generated from each treatment from this tank clearance was – conventional 19%, SOWAP 57% and Farmer's Preference 24%. The substantially lower runoff volumes from the conventional treatment were not expected (see previous reasoning) but can be explained by the uncharacteristic surface properties present at the time. Due to bad weather conditions at the time, ploughing on the conventional treatment was postponed until conditions improved. The consequence of this delayed field operation meant that the conventional treatment had significantly lower percentage area exposed as bare soil. The effect of this would be a lower risk of aggregate breakdown from raindrop impact, soil detachment and removal via runoff.

The unexpectedly high runoff volume from the SOWAP treatment was also in comparison to the other conservation treatment (Farmer's Preference). This can be explained by a substantially smoother soil surface and a significantly lower percentage of surface residues. Although both conservation treatments had surface residues during this season, the SOWAP treatment had been sown with a cover crop of mustard, which had only just started to emerge at the time of the rainfall event. The effect of both a reduction of surface roughness and residues would be a reduction in resistance to water flow and therefore an increase of overland flow.

Treatment differences in soil loss were site dependant. At Loddington a significant difference ($p<0.01$) was found during only one out of the eleven tank clearances but not at any of the other temporal scales (overall or by season). During this one tank clearance (22nd October 2004) soil loss was significantly higher from the SOWAP conservation treatment compared to the two other treatments. This reflected the results and relationships to soil and surface conditions found for runoff volume generation (as stated previously). The mass of soil collected from this tank clearance represented 87% of the total soil mass generated during the entire project (11 clearances in total). It was found that the results obtained from this one tank clearance were significantly influential to the overall results. During this clearance, soil loss from the SOWAP treatment was significantly greater than both the conventional and the Farmer's Preference treatment. The increase in soil loss from the SOWAP treatment could not be explained in terms of soil properties measured in autumn 2004. Instead the tank clearance results can be explained in terms of observed differences in surface characteristics measured at the same time of the tank clearance. These have already been described above in relation to runoff losses and are applicable to these soil losses.

At Tivington soil loss results were significantly different ($p<0.001$) between treatments when compared over the entire sampling period and across all tank clearances, but not on a seasonal basis. In total, 19 tank clearances were undertaken at Tivington, and out of these, 10 showed significant treatment differences. Of these, nine tank clearances showed soil losses to be significantly greater from the conventional treatment compared to at least one of the conservation treatments. This was also the case when mean results were compared over the entire sampling period. The increase in soil loss from the conventional treatment compared to at least one of the two conservation treatments at Tivington can be explained by surface characteristics and soil properties measured over the same time period. Significant differences were found overall in the percentage of bare soil, residue and weed cover on the conventional treatment compared with at least one conservation treatment. The percentage of surface cover by weeds was significantly less on the conventional plots during the second and third season, as well as overall compared to both conservation treatments. Although the presence of weeds is deemed unfavourable in terms of crop quality, it has a positive effect on soil conservation and erosion control. Weed growth is a form of soil cover and aids in the reduction of soil loss (Laflan & Colvin 1981). The percentage of soil surface cover from residues was also significantly less on the conventional treatment at Tivington compared to both conservation treatments. Unsurprisingly the percentage of exposed or bare soil was significantly higher on the conventional plots compared to both conservation treatments overall. This greater area of exposed soil on the conventional plots increases the exposure of soil to raindrop impact. The consequences of this can be an increase in aggregate breakdown, soil detachment, soil particle redistribution to form seals resulting in impeded drainage and an increase in runoff. An increase in runoff or overland flow will transport previously rainsplash-detached soil particles away from the system, as well as detaching more particles from the soil surface by surface flow, resulting in soil loss by erosion. Complementing the presence of weeds was the significantly higher content of organic carbon in the soil from the SOWAP treatment compared to the conventional treatment during the second season. The same treatment trend was found over the entire sampling period but the results were not statistically significant. An increase in organic carbon has been shown to reduce aggregate breakdown and soil detachment.

There was an anomaly to this pattern of higher soil losses from the conventional treatment. The third tank clearance on 1st November 2004 at Tivington was the only tank clearance where soil loss was greatest from the Farmer's Preference plots compared to both conventional and SOWAP treatments. To understand why this anomaly occurred, statistical analysis was carried out on the measurements taken as close to the November 2004 tank clearance as possible. Initially treatment differences in surface characteristics were investigated. No significant differences were found, indicating that other processes were taking place. This led to the analysis of soil samples taken in the autumn of 2004 for differences in soil properties between treatments. Analysis showed significantly higher contents of silt (p=0.002) for the Farmer's Preference treatment. The implication of this is increased risk to soil erosion due to aggregate breakdown, redistribution of silt on the surface leading to surface sealing. The consequence of this is the creation of an impermeable surface layer, limiting infiltration and leading to increased runoff. The increase in aggregate breakdown and runoff generation would result in the transportation of eroded material downslope. This would explain why soil losses were significantly higher from the Farmer's Preference treatment during the November 2004 clearance.

The application of conservation tillage at both sites (clay and sandy clay loam) did not significantly affect the volume of runoff generated, under different cropping regimes. However, the mass of soil eroded was affected by the tillage practice employed. At Loddington (clay), soil loss was increased with the use of the SOWAP conservation management practice. At Tivington (a sandy clay loam), the SOWAP based conservation tillage always minimised soil loss significantly. At both sites, results could be linked to treatment induced changes in surface characteristics including surface cover, type of cover and surface roughness. Soil loss results from Tivington could also be linked to soil properties, specifically to soil texture and organic carbon content.

It has been shown that tillage affects erosion in different ways depending on a site's risk to erosion. However, what has not been discussed is the context of these results in terms of sustainability. As said previously both sites have shown treatment differences in soil loss, it is important to relate these losses to yearly formation rates. Although losses were not calculated strictly on a yearly basis, losses were measured over known number of days. Therefore, a yearly figure could be calculated and compared to the rate of tolerable loss of 2 t ha^{-1} yr^{-1} (Morgan 2005). It was found at Loddington, rates of soil loss from all treatments were below tolerable levels, ranging between 0.008-0.120 t ha^{-1} yr^{-1}. At Tivington, it was found that soil loss from the conventional treatment was above tolerable levels, ranging between 0.2-3.5 t ha^{-1} yr^{-1}. The adoption of either conservation tillage treatment (SOWAP or Farmer's Preference) resulted in reductions in soil losses to below tolerable levels at both sites. This raises the question of whether a change in tillage regime is necessary to reduce soil erosion on sites which are not at risk of high erosion rates.

3.5.1.2 Objective two: to quantify nutrient and carbon losses associated with runoff and soil loss for three different soil treatments

Hypothesis two states that losses of carbon and nutrients (nitrogen, phosphorus and potassium) associated with the runoff and sediment would be greater from the conventional treatment compared to at least one of the conservation treatments. Nutrient and carbon results will be discussed separately in the following text.

3.5.1.2.1 Nutrient Loss

The loss of nutrients from the soil environment was considered important, with on-site and off-site consequences. Loss of nutrients from the soil environment can lead to changes in crop productivity, soil 'health' and the biological community affecting aggregate stability. The off-site implications include eutrophication of marine and freshwater bodies and contamination of drinking water.

Differences between treatments in terms of nutrient concentration and load associated with the runoff and soil loss were site specific. At Loddington significant treatment differences in nutrient loss were found in the runoff and sediment whereas at Tivington, significant treatment differences were only found associated with the sediment.

At Loddington, significant treatment differences in nutrient loss were only observed for nitrogen and phosphorus but not for potassium. Loss of soluble nitrogen (measured as nitrate) was significantly higher from the Farmer's Preference treatment compared to both conventional and SOWAP treatments. This only occurred during season one. This unexpected result could not be attributable to the surface characteristics or soil properties present at that time. This led to analysis of surface core samples taken at the time for content of total nitrogen. Although not significant the results of total N in the surface (0-5cm) soil samples show that N concentrations were substantially higher from the Farmer's Preference treatment in comparison to the other two treatments. This higher concentration of N in the field soil would lead to increased N concentration available to be lost via runoff.

Loss of soluble phosphorus (measured as phosphate) was significantly higher from the conventional treatments compared to both conservation treatments. This was for both mean (over the entire sampling period) P concentration and total loading (calculated as a function of runoff volume). This significant loss from the conventional treatment could be explained by the significantly lower levels of clay present within the field soil in comparison to the conservation treatments. As a result the overall adsorptive capacity and aggregate stability of soil from the conventional treatment would be substantially reduced. Therefore, less P is able to be retained within the soil and as a result available in solution.

As stated previously there was no significant treatment differences in nutrient concentrations or loadings associated with the runoff at Tivington. However, concentrations of all three nutrients were found to be significantly lower in sediment from the conventional treatment compared to both conservation treatments. Although not significant the mean (over the entire sampling period) calculated load (a function of soil loss) of nutrients lost in the eroded sediment were also substantially greater from the conventional treatment compared to both conservation treatments.

The significantly low concentrations and substantially higher loads of sediment associated N, P and K were related to both soil properties and surface conditions. Firstly, the clay content of soil from the conventional treatment was significantly less than for the conservation treatments. Therefore it would be expected that higher concentrations of nutrients would be retained due to a) the increase adoptive capacity, and b) increase in aggregate stability, both associated with clay particles. However, the latter would suggest that the total loading would also be lower due to decreased erodibility. This was not reflected in the loading results, but can be explained by the surface characteristics present on the conventional treatment. It was found that the percentage of bare soil on the conventional treatment was significantly higher and surface roughness lower on the conventional treatment. This suggests that the risk to erosion could be higher on this treatment. This was supported by the soil loss results.

Although the concentration of sediment associated nutrients lost was reduced by the use of conventional tillage, the total mass of sediment entrained nutrients lost increased by this form of tillage. The latter has on-site implications for the farmer, forcing increased amounts of fertilisers to be added to the field to maintain crop productivity and yield.

It is important to put these nutrient losses in context. Threshold values of nitrogen as nitrate, in drinking water are 50mg l^{-1} for the UK and as low as 25 mg l^{-1} for Finland (Journeaux 2003). The results from this current research showed that concentrations of nitrate did not reach either of these threshold limits, although it should be noted that a substantial amount of nitrate is lost from agriculture via leaching due to the high solubility of nitrate. Therefore even though treatment differences were found (at Loddington only) in runoff associated nitrogen, the levels found do not pose a threat to drinking water. However, the results from Loddington (not Tivington) from all treatments do exceed the environmental threshold limit set by New Zealand of 3.2 mg l^{-1} of nitrate (Journeaux 2003).

Loss of soluble phosphorus also has implications for drinking water quality. Guidelines state that concentrations of P that are allowed in drinking water are 2.2 mg l^{-1} (DWI 2003). Results obtained from this current research showed that loss of soluble P did not exceed this guideline level at either site, indicating that although treatment differences did exist in soluble nutrient loss, the adoption of different soil managements in the mitigation against this loss in terms of drinking water quality is not necessary. However, work by Sharpley & Smith (1990) has shown that concentrations of soluble P of 10 ug l^{-1} are sufficient to cause accelerated eutrophication. The soluble P losses from both sites for all treatments exceeded this level, making adoption of conservation tillage a useful tool in the mitigation against eutrophication from agricultural runoff.

Although not presented in the main text, enrichment ratios were calculated for each treatment, using the mean sediment concentration of each nutrient within the eroded sediment against the concentration found within the field soil (means were calculated over the entire sampling period). It was found that all enrichment ratios from all treatments were above 1, indicating that a higher concentration of nutrients were lost than were present within the field. This occurred for phosphorus and potassium loss at both sites, and nitrogen loss at Loddington only. The results also showed that at Loddington enrichment ratios where higher for all nutrients in sediment from the conventional treatment compared to both conservation treatments. However, at Tivington this was only the case for nitrogen while enrichment ratios for both phosphorus and potassium were lowest from the conventional treatment. This implies that the way in which nutrients are lost from the soil system vary not only with different soil management regimes but with site location as well due to differences in soil properties including texture. Results indicate that the adoption of conservation treatments can reduce loss of sediment associated nutrients on sites less at risk to erosion but may enhance loss on sites with a higher risk to erosion. This has on-site implications for farmers needing to apply additional fertilisers to the field to maintain crop productivity and yield and off-site effects contributing to water pollution including eutrophication.

3.5.1.2.2 Carbon Loss

The loss of carbon from the soil environment was deemed important due to the on-site implications of such a loss. Loss of carbon can lead to a decrease in aggregate stability, increase in surface capping and sealing and a general decline in crop productivity, soil structure and substrate for microbial communities.

There were no significant differences between treatments in the concentration or loads of organic carbon associated with eroded sediment, at either site. No significant differences were found in carbon concentration associated with the runoff at either site location. However, significant treatment differences were found in the loading of organic carbon associated with the runoff at Tivington, but not at Loddington. Mean carbon load in the runoff (over the entire sampling period) was significantly greater from the conventional treatment compared to the SOWAP conservation treatment.

The observed results in carbon losses could not be related to the inherent soil properties, but were related to the surface characteristics, specifically the percentage bare soil. The lack of cover on the soil surface on the conventional treatment would have resulted in increased risk to soil detachment from rain drop impact. As a consequence, the soil surface would be more prone to surface sealing, so infiltration would be reduced. The overall outcome would be an increase in runoff generation. This was the case, as previously shown in relation to treatment difference of runoff volume. Consequentially, higher amounts of soluble carbon could be easily transported via runoff. The same pattern of loss was also found with sediment associated carbon. Although not significant, a substantially greater mass of sediment associated carbon was also lost from the conventional treatment at Tivington.

The ability of soil to sequest carbon is important for global issues of climate change and for a farmer to maintain crop productivity (Charman & Murphy 2000) and yield. Smith (2004) estimates that rates of carbon sequestration range between 0.3-0.8 t C ha^{-1} yr^{-1}. Although not presented in this current research it was observed that the total cumulative mass of carbon lost (combined mass of C loss from runoff and sediment) was below estimated sequestration rates. This was the case for all treatments at both sites. Site differences in carbon lost were apparent. Tivington lost substantially (an order of magnitude) greater amounts of C than Loddington. Possibly due to the increased risk to erosion at Tivington and a clay content 25% lower than that of Loddington.

The adoption of conservation tillage has shown that at Loddington (clay) this soil management treatment does not significantly affect the loss of runoff- or sediment-associated organic carbon. However, at Tivington, (a sandy clay loam) conservation soil management practices can minimise the loss of runoff-associated organic carbon, not by reducing carbon concentration, but by lowering volume of runoff generated. This supports the findings found by Owens et al. (2002) in relation to soil loss.

3.5.2 Implication of this study

This work has shown the effects of different soil management treatments on water, soil, nutrient and carbon losses. It also highlights that these effects are site specific, due to differences in risk to erosion, as a result of changes in soil texture, organic matter and slope gradient. Consequently, universal conclusions regarding the impact of conservation tillage on soil, water, nutrient and carbon losses are not possible.

This study has reaffirmed the significance of soil properties and surface characteristics on the loss of water, soil and associated nutrients, but that the relative importance of soil properties and surface characteristics can be site specific, depending on soil texture, cropping regimes, topography and meteorological conditions. This implies that management systems affecting these properties and characteristics must be designed to be site specific.

The present work has also shown the difficulties associated with data generated from field scale erosion plots. Natural variability over small spatial scales in terms of rainfall received, soil properties and surface characteristics, plus the man-made variability due to treatment application during cultivation operations (e.g. presence of tramlines within field erosion plots) result in highly variable data in terms of runoff and soil loss. Limited replication adds to the margin of error associated with such high levels of variability. The observations obtained were determined by a set of highly complex hydrological and soil variables, which have been taken into account in this study. This research has highlighted this complexity, although an attempt has been made to use data to explain the results generated. The outcome is a unique data set of soil, water and nutrient losses in the UK under different soil management practices.

3.5.3 Future research recommendations

Increasing the frequency of sampling is recommended, so that data can be collected on an event basis rather than as soon as logistically possible after an event or a series of events. Sampling during an event would allow better assessment of the loss of water, sediment, nutrients and carbon over time, whilst simultaneously taking into account the intensity, kinetic energy and volume of rainfall. This resolution of monitoring requires considerable expenditure on instrumentation (e.g. automated flumes and sediment samplers), which was not available to the present study.

3.6 Conclusion

The main findings of this study were that the application of conservation tillage did not result in significant differences in runoff volume compared to conventional practices at either site. Differences in soil loss between treatments were site specific; at Loddington soil loss was highest from the SOWAP treatment and at Tivington in the majority of cases, soil loss was greatest from the conventional treatment, compared to at least one of the conservation treatments. Where significant differences existed these could be explained by inherent surface characteristics (and soil properties at Tivington). The presence of soil cover was of specific importance in the reduction of soil erosion at both sites.

Losses of soluble nutrients can be minimised through the application of conservation tillage due to the increase in clay and associated adsorptive capacity. However, based on the results from this study, success may only be possible on clay soils, as different tillage treatments on a sandy clay loam soil appear to have little impact on runoff associated nutrient losses. Sediment-associated nutrients can also be reduced through the introduction of conservation based soil management due to changes in soil properties, specifically clay content. However, this only applied to clay soils. The adoption of conservation tillage on the sandy clay loam soil used in this study resulted in an increase in sediment-associated nutrient loss. This was also attributable to the changes in soil texture.

At Loddington, the loss of organic carbon associated with runoff and soil loss was not significantly altered by the adoption of conservation tillage practice compared to a conventional treatment. At Tivington, however, the implementation of conservation based management (specifically SOWAP) did reduce runoff-associated loss of organic carbon. This was related to a reduction of runoff generated from this treatment as a result of a lower percentage bare soil.

In conclusion, the application of conservation soil management at a field scale under natural conditions does not always reduce soil and water losses nor the associated losses of nutrients or carbon. The results observed appear to be related to site specific conditions, which vary in space and time. The results from this study also indicate that surface characteristics are the most influential factors in the initiation of erosion on soils with a high clay content (>40%), higher organic matter and gentler gradient. The combination of surface characteristics and soil properties determine the erosion rates on a soil with a lower clay content, lower organic matter and greater gradient. Irrespective of soil type nutrient losses are controlled by changes in soil properties, specifically the change in adsorptive particles specifically from clay content. Carbon loss does not appear to be influenced by soil properties or surface characteristics directly, but instead by the volume of runoff generated from any given treatment.

4 Micro-plot scale erosion assessment – rainfall simulations

4.1 Introduction

Monitoring and measurement of soil erosion is important in the assessment of the effects agriculture has on the environment. Runoff plots are just one of the many devices to generate data on erosion rates. There are many permutations of runoff/erosion plots, related to plot size, plot boundaries (absence/presence and material used) and the use of natural or artificial rainfall. Large erosion plots (>100m^2) with collection systems installed to retain eroded material and runoff are the most direct way of assessing field erosion, and probably give the most reliable results on soil loss per unit area (Hudson 1995; chapter 3). They are widely used in soil erosion research. Runoff plots are generally expensive, time consuming and reliant on natural rainfall events which are inevitably variable and unrepeatable in space and time. Information and results relating to field scale erosion plots used in the present research can be found in chapter 3. The identified constraints and limitations of field scale erosion plots have lead to the use of micro-erosion plots (≈1m^2) in conjunction with a rainfall simulator, allowing controlled experiments to be replicated quickly and inexpensively.

Micro-erosion plots in conjunction with rainfall simulators allow the researcher to create a storm event of a specific duration and intensity. Simulations can be repeated numerous times, shortening the time span required to carry out replicated data collection. Advances in the type of rainfall simulators has led to better control and understanding of rainfall erosivity characteristics, such as raindrop diameter, size, velocity, fall height, kinetic energy and intensity (Morgan 2005). Pioneer research in this area was led by Laws (1941) who began to measure the fall velocity of raindrops. Despite a great deal of research into rainfall erosivity, simulated rainfall will never be able to replicate natural rainfall due to the latter's inherently variable character. However, control of rainfall erosivity is fundamental when wishing to isolate and thus assess the effect of a specific soil or surface property on erosion.

Rainfall simulators can be used under laboratory conditions or in the field in conjunction with micro-erosion plots of a specified area. The use of rainfall simulators in laboratory based erosion research is covered in chapter 5. Their use in the field is discussed in this chapter. Laboratory based rainfall simulators have advantages over field based ones. Experimental conditions at the start of the simulated rainfall event can be controlled, in particular soil moisture and wind velocity. These factors cannot be completely controlled in the field and inevitably this can lead to variation in erosion data generated. Despite this, field based rainfall simulations are still an important tool for soil erosion research due to their flexibility (i.e. ability to modify rain events, and use on different terrains and field treatments).

Spatial scale implications must be considered when estimating field scale soil erosion from data generated from micro-plot rainfall simulations. It is important to consider whether the erosion processes operating over only a few square metres (micro-plot scale) compare with those operating over $500m^2$ (field scale). For example, at the micro-plot scale, the process of rain splash will be the dominant erosion process compared to rill formation at the field scale (Morgan 2005; Rickson 2006). Results obtained from micro-erosion plots may not reflect the considerable differences in soil conditions occurring across a field. So, why are they still used? Rainfall simulators are still the best method for controlling rainfall, allowing controllable experimental conditions in which to study specific treatment responses to changes in soil or surface properties. By using micro-erosion plots, researchers can give indications of treatment responses to erosion, i.e. can state whether one treatment is better than another in minimising erosion (Andraski et al. 1985). Hudson (1995) advised caution in the use of micro-erosion plots when comparing similar treatments, due to the variability in erosion rates at different spatial scales, so that direct extrapolation of results was not possible. More details on spatial scale extrapolation and integration will be addressed in chapter 6.

4.2 Aim, Objectives and Hypotheses

4.2.1 Aim

The aim of this chapter is to investigate soil erosion and runoff generated from different soil management treatments, using simulated rainfall on micro-erosion plots in the field. Parameters to be measured and analysed are soil loss, runoff volume and runoff rate.

4.2.2 Objectives

I To identify if significant differences in runoff volume and soil loss occur between three different soil management treatments; conventional tillage (C) and two forms of conservation tillage (SOWAP - S and Farmer's Preference -F). Any differences found will be explained in terms of supporting field evidence.

II To critically evaluate whether there are any differences in the rates of runoff generation for the three different soil management treatments.

4.2.3 Hypotheses

4.2.3.1 Hypothesis One

It is expected that runoff volume from the conventional treatment will be significantly higher than for both conservation treatments – SOWAP and Farmer's Preference. Consequently, the application of conventional soil management will also increase the risk of soil erosion by water. Before considering the impact of different soil management practices on runoff generation, it is necessary to consider the processes leading to runoff generation and the factors affecting these processes (Table 4.2-1). With this in mind, the ways in which soil management can affect these factors can be investigated.

Table 4.2-1 Processes leading to runoff generation and factors affecting these processes

Runoff generation		
Process	**Factor**	**Description**
Infiltration rate	Seal formation	A surface seal consists of two parts, a) an area of redistributed eroded particles and b) a 'washed-in' zone where clay is dispersed and clogs soil pores just below the surface (Lado et al. 2004). Both contribute to seal formation thereby impeding infiltration. The creation of surface seals has been linked to a high clay and silt content (Slattery & Bryan 1994) and low organic matter content.
	Bulk density - BD	Consolidation of the soil matrix will lead to a decline in infiltration rates, of which BD is an indicator. A high BD represents increased amount of compaction or consolidation.
	Moisture content -MC	Soil moisture content affects runoff generation in two ways. The gravitational pull of water down through the matrix is exacerbated by the suction from dry aggregates (Charman & Murphy 2000). Infiltration rates therefore increase with a decline in moisture. However, this decrease in MC also increases the strength of pre-formed crusts, reducing infiltration rates (Bennett et al 1964).
	Biological community	The presence of an active soil biological community has been linked to the maintaining of soil structural porosity (Lavelle et al. 2006). Preferential pathways of flow within the soil profile are also created via the burrow action of plant roots and earthworms. This biological action maintains infiltration rate.
	Surface characteristics	The presence of a rough surface has been linked to an increase of infiltration rates. Surface stones and rock fragments and the stem base and root zone of vegetation (Morgan & Rickson 1988), have all be associated with infiltration rate increase.
Flow resistance	Surface cover	Surface residues, rock fragments and vegetation stems all reduce flow resistance. The effect of the latter, changes over time as the vegetation grows and stems widen.
	Surface roughness	An increase in surface pitting i.e. roughness will create a higher resistance to flow, disrupting and dissipating the energy flow of water thereby lowering its velocity (Takken et al. 2001) energy available for sediment transport.

In terms of erosion processes at the micro-plot scale, and how soil management practices may affect these, it is expected that surface cover will be a primary factor determining soil detachment, by controlling the area of soil exposed to rainfall. The effect of rain drop impact on exposed aggregates will then be determined by the inherent soil properties, which determine the erodibility of soil aggregates and detachment rates. The dominant transport mechanisms of eroded soil particles will be through splash-effects and entrainment within overland flow/runoff. The latter being affected by infiltration rates and resistance to flow, controlled by the combination of soil properties and surface characteristics (see Table 4.2-2). Details of the effect of soil properties and surface characteristics on soil erosion can be found in section 1.2.5. An overview of how soil properties and surface characteristics affect soil detachment and transport at the micro-erosion plot scale are presented in Table 4.2-1 and Table 4.2-2.

Table 4.2-2 Factors which affect the resistance of aggregates to breakdown

Factor	Process
Clay	Strong cohesive forces exist between clay particles and as a result have been shown to have a positive relationship on aggregate stability (Le Bissonnais et al. 2002). A reduction in clay will increase the susceptibility of soil aggregates to rain drop impact and splash erosion
Organic matter and carbon	Organic matter and carbon are intrinsically linked and have a strong positive relationship with aggregate stability by increasing cohesion between soil particles, therefore a reduction in OM and SOC will decrease aggregates' stability and increased soil erodibility
Moisture content	The moisture content affects particle cohesion within an aggregate. If an aggregate is too dry or too wet then cohesive strength is reduced. Dry aggregates are more at risk of slaking. This is where air is forced out of an aggregate as water enters the aggregate. During this process soil particles are forced apart.
Soil biota	The soil biota is an important factor in soil erodibility. For example, plant roots and fungal hyphae produce organic cements and connectors which increase inter-particulate cohesion, and thus aggregate stability.

As seen in Table 4.2-1 and Table 4.2-2 soil properties and surface characteristics are drivers of runoff generation and soil erosion. These factors can be affected by the application of different soil management practices. Such treatment effects on each factor are discussed in the following text.

It is expected that the percentage of surface cover will be greater on the conservation treatment compared to the conventional plots. The percentage cover from crops should be similar, but weed growth has been shown to be higher on conservation treatments. Although the latter is deemed as a problem regarding a decline in crop productivity, it can help reduce erosion (Table 4.2-2). A cover crop (e.g. mustard) may be planted to protect the soil surface against rain drop impact before the main crop is established. However, the most well known conservation practice is the application of residues. Not only do residues protect the soil surface from rain drop impact, but they have been associated with an increase in organic matter and carbon contents, reductions in the extremes of diurnal soil temperatures and conservation of soil moisture. Surface protection of the soil is also given by stone cover. Differences are expected in stone cover for the different soil management treatments, as the action of ploughing and soil inversion on the conventional treatment will redistribute stones within the soil profile.

The action of ploughing and increase in mechanical manipulation of the soil on the conventional treatment is expected to reduce surface roughness as aggregates are broken down to form a 'fluffy' seed bed. Both conventional and conservation based treatments will experience a decline in surface roughness over time, caused by soil consolidation by raindrop impact and subsequent "settling" or re-packing of the disturbed aggregates (Morgan 2005). Even so, it is expected that surface roughness will be higher on the conservation plots as they have experienced fewer cultivations which can potentially lead to aggregate breakdown.

An increase in bulk density is associated with poor structure and soil matrix consolidation. The latter resulting from compaction primarily caused by field operations. As conventional tillage involves more field operations, the risk of compaction is expected to be greater than that experienced on the conservation treatments. However, the lack of inversion tillage associated with the long term application of conservation tillage, has been shown to increase bulk density, but the practice of sub-soiling can mitigate it. It is not expected that this problem will occur during the relatively short time period involved in the project. As well as experiencing less mechanical disturbance, it is also assumed that soil structure will be better on the conservation plots, due to the less disturbed biological community associated with non-inversion tillage systems (such as increased population of earthworms). Overall it is expected that bulk density will be higher on the conventional treatment, compared to the conservation treatments.

As stated previously the application of surface residues helps maintain soil moisture content and increases organic matter and carbon content. The lack of soil inversion and reduction of soil disturbance also helps to minimise organic matter mineralisation (Turley et al. 2003; Roldán et al. 2005) i.e. the biological decomposition of organic material into simple organic or inorganic products (Brady & Weil 2002) via soil microorganisms. Therefore the soil under conservation management is expected to have higher moisture contents and levels of organic matter and carbon compared to conventionally treated soil.

It has been found that clay is lost through erosion events, preferentially during small erosion events (Quinton et al. 2001). Erosion is estimated as being greater on conventional treated soil (stated previously) inferring clay loss would also be higher. Therefore a more rapid decline of clay is expected with time from the conventional treatment. An implication of clay reduction is a decrease in aggregate stability leading to increased risk to erosion and therefore forms a self perpetuating process.

The combination of reduced surface cover, lower organic matter content, reduced aggregate stability and increased bulk density associated with the conventional treatment results in an increased risk of runoff generation and erosion, compared with that from the conservation treated soil.

4.2.3.2 Hypothesis Two

In hypothesis one it was stated that there will be differences in the runoff volumes and soil eroded between the two types of soil management. Hypothesis two deals with the difference in rates at which runoff is generated. When the rate of runoff remains the same, equilibrium is reached. It is predicted that the equilibrium of runoff rate will be reached at some point in time for both treatment types, but it is expected that this will occur within a shorter period of time for the conventional treatment.

It is predicted that the soil surface on the conventional plots will become sealed at a faster rate, as decreased aggregate stability leads to more splash erosion and sealing. The consequence of this is a reduction in infiltration rates, depression storage and increased overland flow. These erosion processes are expected to be the result of treatment induced changes in surface cover and roughness, bulk density and organic matter as discussed in hypothesis one. Sealing rates are also expected to be higher for the conventional treatment due to the expected lower surface roughness for this treatment. Hence the potential for depression storage is less and ponds will quickly form, but will also break faster thereby 'releasing' the retained water leading to the generation of runoff (Morgan 2005). It is this relationship of storage and release which will differ between treatments. A hypothetical representation of this can be seen in Figure 4.2-1. The storage time will be of a longer duration on the conservation plots, but water release will occur quickly and equilibrium will be reached shortly after. Conversely, storage time on the conventional plots will be of a relatively short duration, and release will occur over a much longer period of time. It should be noted that where the simulated storm used in this research lies on the storage/release curve is unknown.

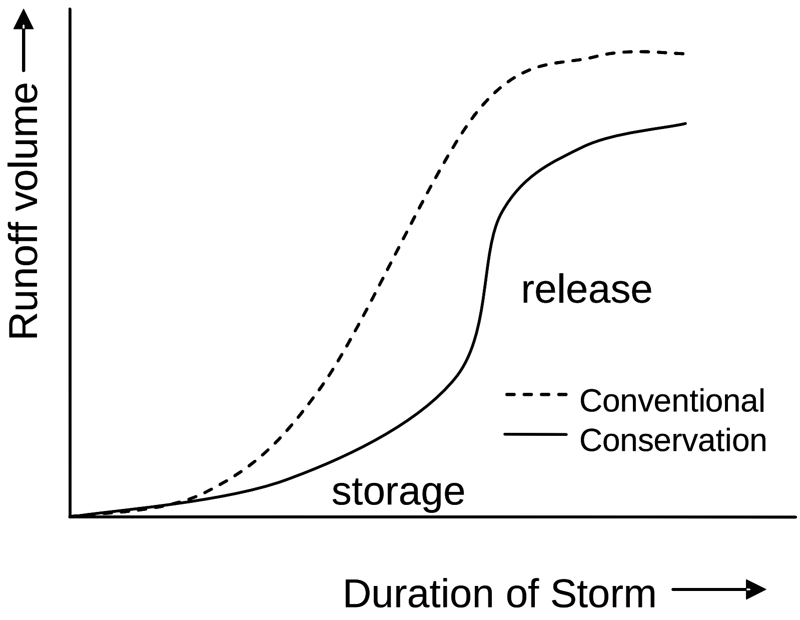


Figure 4.2-1 Hypothetical storage-release curves of runoff generation

Runoff rates will be compared between treatments over a standard 30 minute storm. This differs from hypothesis one where storm length varied.

4.3 Methodology

Two sites located in the UK were used in this investigation, the first at Loddington, Leicestershire and the second at Tivington, in Somerset. Three soil management treatments were adopted at each site. The conventional (C) tillage treatment involved the use of a mouldboard plough to invert the soil. Neither of the two conservation tillage treatments inverted the soil. The two conservation treatments were used - SOWAP (S), and Farmer's Preference (F). In advance of this present study a demonstration field was identified and the treatments applied. A detailed description of each site and soil managements practices adopted can be found in chapter 2.

Details of the date, season and crop present at each micro-erosion plot trial can be found in Table 4.3-1. The April 2004 simulations at Loddington were carried out as a pilot run, however the data are valid and have been incorporated into the dataset.

Table 4.3-1 Micro-erosion plot trial descriptions

Site	Season	Crop	Trial	Date
Loddington	1	Winter Wheat	1	April 2004
	2	No crop sown *	2	September 2004
	3	Spring Beans	3	March 2005
	4	Winter Wheat	4	November 2005
			5	April 2006
Tivington	2	Winter Wheat	1	September 2004
			2	March 2005
	3	Winter Beans	3	November 2005
			4	April 2006

*** treatments were: C – stubble from previous crop until late ploughing; S – tilled and cover crop sown; and F – ploughed and residue left**

4.3.1 Micro-erosion plot location

The area required per rainfall simulation was 3m (across slope) by 4m (up-down slope). This allowed an effective work area without risk of damage to neighbouring plots. Three rainfall simulations were carried out for each treatment, twice a year in the spring and autumn. At Loddington simulations ran from spring 2004 to spring 2006, making a total of 15 simulations. At Tivington simulations ran from autumn 2004 to spring 2006, making a total of 12 simulations. At Loddington two areas of land were set aside above each of the 3 treatments. These areas (9m by 12m) were assigned for rainfall simulations and associated sampling (Figure 2.1-4). This area was divided into blocks of 3m by 4m and randomly assigned a simulation date; an example of this can be seen in Figure 4.3-1. Each area of land was situated at the top of each erosion plot.

At Tivington, the area of available land for sampling and simulations was adjacent to the erosion plots (Figure 2.1-5) and covered an area 25m by 9m. The 25m sampling strip was sectioned off into 15 blocks of 3m by 5m. The 12 simulations to be carried out per treatment were randomly assigned a sampling block; an example of this can be seen in Figure 4.3-2.

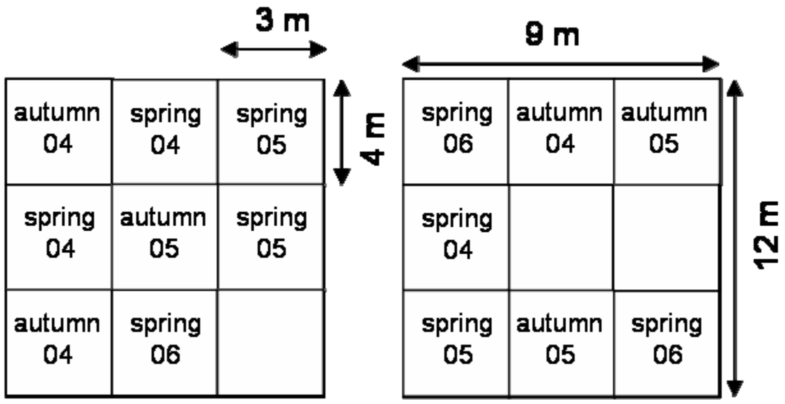


Figure 4.3-1 Rainfall simulation sampling design: Loddington

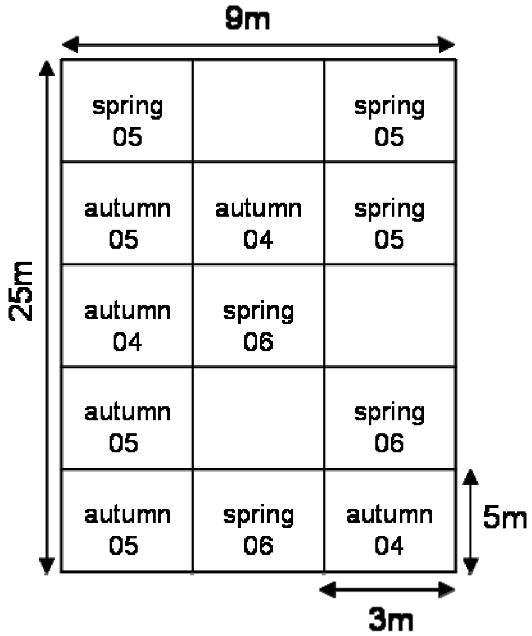


Figure 4.3-2 Rainfall simulation sampling design: Tivington

Measurements taken during the rainfall simulations included time to runoff, runoff rate, runoff volume, sediment concentration, calculated sediment loading, and plot surface characteristics – extent of ponding and seal formation, surface roughness and percentage cover

4.3.2 Rainfall simulation equipment

The micro-erosion plot area was rectangular in shape, measuring 1.5m^2 (1.5m up/downslope and 1m across slope). Metal sheeting was set into the ground to a depth of at least 10cm to delineate the plot area, leaving the downslope side open for a collection system. The collection system (Figure 4.4-3) consisted of a metal or plastic edging which fed into a trough where runoff and sediment were channelled into a beaker and measured. A protective covering was used (Figure 4.3-3) to prevent rainfall entering the collection system, which would have lead to an overestimation of runoff generation.


Figure 4.3-3 Micro-plot collection system (left) and protective covering (right)

There are two main designs of rainfall simulator, ones that produce raindrops from nozzles under pressure or under a low static head of water. Irrespective of the design used, it is important to maintain pressure to produce consistent fall velocity and rainfall distribution. An in-depth review of rainfall simulators can be found in Hudson (1995) and Rickson (2006). The rainfall simulator used in this research produced simulated rainfall 3m above the ground through a nozzle (Nozzle Lechler 460.788, Figure 4.3-4) at 0.39 bar, with a mean rainfall intensity of 35mm h^{-1}. Over a 30 minute storm the D_{50} drop size was 1.04mm and the total kinetic energy was 0.16 J m^{-2} s^{-1} details of which can be found in the Appendix C.

The intensity of rainfall used in past research has been varied, reflecting conditions of the test area. Past research has used intensities of less than 15 mm h^{-1} (Olayemi & Yadav 1983; Myers & Wagger 1996), 25 mm h^{-1} (Karunatilake 2002), 35 ±3 mm h^{-1} (Slattery & Bryan 1994; Wan & EL-Swaify 1999;

Somaratne & Smettem 1993), 45-65 mm h^{-1} (Gómez & Nearing, 2005; Wan & EL-Swaify 1999; Myers & Wagger 1996) and over 70 mm h^{-1} (Bedaiwy & Rolston 1993; Ndiaye et al, 2005; Bedaiwy & Rolston 1993; Andraski et al. 1985). A rainfall intensity of 35mm h^{-1} was universally adopted across all countries involved in the parent demonstration project, SOWAP.

Figure 4.3-4 Lechler nozzle 460.788

The simulated storm duration was originally run until runoff reached equilibrium (i.e. when runoff rate became constant). This was estimated to be at least 40 minutes, based on work from the University of Leuven, Belgium on loess soils using the same equipment and the same rainfall intensity (personal communication; Govers, G. 2004). After the first trial simulations, the time taken to reach equilibrium ranged from 25min to 1hr 20min on the UK soils. As previously mentioned the time needed to reach equilibrium is influenced by many soil and surface factors. For example a study on the effects of antecedent moisture content on runoff generation compared a pre-wetted and dry soil, both receiving simulated rainfall for 25 minutes. It was found that the dry soil reached a steady state of runoff after 25 minutes, but the pre-wetted soil did not (Le Bissonnais & Singer 1992). As differences in soil properties and surface characteristics was expected between treatments, a standard time allowing equilibrium to be reached for all treatments could not be defined. Instead a set simulation time was chosen, based on operation logistics, taking into account limitations of time, water supply and man power. A simulation running time of 30 minutes was deemed sufficient to generate runoff, and possibly reach equilibrium for some treatments. This running time has been used in other studies (Schiettecatte et al. 2005; Ndiaye et al. 2005; Wan & El-Swaify 1999; Myers & Wagger 1996).

Wind velocity during rainfall simulation was an issue at both sites. The mean wind velocities were 1.26m s^{-1} and 2.19m s^{-1} at Tivington and Loddington respectively. On the Beaufort scale these velocities are defined as "just perceptible" to "a gentle breeze" (Met Office 2006). Despite these relatively low mean wind velocities, it was found that the velocity was not constant, with gusts of winds occurring sporadically. The maximum velocities received during the cropping seasons were 6.12m s^{-1} at Tivington and 9.44m s^{-1}at Loddington, classed as a "fresh breeze" to "fresh gale" (Met Office 2006). These unexpected gusts of wind caused variability in the distribution of the simulated rainfall – sometimes enough for the rainfall to miss the micro-erosion plots. To solve this problem a wind break was constructed and set up adjacent to the micro-erosion plots during simulation (Plate 4.3-1).

Plate 4.3-1 Wind breaks used during rainfall simulations

A small two-stroke petrol engine was used to pump water through the rainfall simulator and onto the micro-erosion plots. The water used at both sites was obtained from the local mains supply and was carried in containers onto the field for each simulation. Previous work has shown that water from a local mains supply is similar in erosivity to natural rainfall (Barton 1994).

Consideration was given as to the initial conditions of the plots prior to the application of rainfall. Results from previous studies have shown that runoff and soil loss results are affected by the initial soil moisture content. This has led to some researchers pre-wetting plots before testing (Hudson 1995). Others have carried out successive simulations on the same plot area; an initial dry run and subsequent wet runs 24 hours later (Wan & El-Swaify 1999). Pre-wetting has been shown to reduce seal development (Le Bissonnais & Singer 1992). In the case of this study, pre-wetting was not used. Soil moisture content can differ with changes in tillage treatments due to the presence of surface residues for example. If plots were pre-wetted any possible treatment effects would then be lost. It was felt that for this study, pre-wetting was not advisable and that simulations should be carried out 'dry' or rather at the field condition present at the time.

Once the micro-erosion plot was installed, plot and soil measurements were carried out before, during and after the simulation. These will be discussed in the following text; a summary of these measurements can be found in Table 4.3-2.

4.3.2.1 Soil and micro-erosion plot properties before and after rainfall simulation

Before a simulation was carried out, the slope gradient of the micro-erosion plot was measured. Slope is an important factor in determining runoff generation and velocity. Steeper slopes generate more runoff than gentler slopes; therefore it is important that the slope gradient should be constant between micro-erosion plots. Soil surface roughness was also measured using a small link ball chain, 1m in length. As previously stated, surface roughness can affect the rate at which runoff occurs. The ball chain was draped along the soil surface and measured; if the surface was completely smooth (i.e. flat) then the length would be 1m: less than 1m would indicate some degree of surface roughness. Surface roughness was measured before and after simulation in the same place, vertically down slope. A visual assessment of the surface cover of the micro-erosion plot before simulation was also carried out. The percentage total cover was measured which included the crop, weeds, residues and stones; from this the percentage bare soil was calculated. Surface cover from crop, weeds, residue and stones is important in physically protecting the soil surface from rainfall impact, also the presence of residue aids in preventing the soil surface drying out. The percentage plot area covered by surface seals[3] and ponds[4] was also visually assessed before and after simulation. Three soil surface soil samples (0-5cm depth) were randomly taken outside of the plot area before simulation, and inside the plot area after simulation. These samples were then used to determine gravimetric and volumetric moisture content and bulk density before and after the application of simulated rainfall. Differences in moisture content and bulk density can lead to changes in infiltration rates which affect runoff rates. Once analysis was complete the samples taken outside the plot (representing the "before" simulation condition) were used to determine total organic carbon content. Before and after the simulation took place a photograph was taken of the micro-erosion plot, so that the plot could be revisited to help resolve any unanswered questions

4.3.2.2 Measurements during rainfall simulation

Three rainfall gauges were placed in proximity to the micro-erosion plot to collect any natural rainfall that may fall during the simulation. Ideally rainfall simulation should not be carried out when it is raining. During simulations rain

---

3 Seals or sealing represents the area of soil surface where reconstruction of aggregates has occurred to form an impermeable layer.

4 Ponds or ponding refers to an area of surface soil where water has been unable to infiltrate or drain away causing water to collect to form a pool.

may start to fall making it important to collect any possible natural rain that may fall and factor it in to any calculations made. Four additional rainfall gauges were placed adjacent to the micro-erosion plot to collect simulated rainfall. From these four gauges the total volume of rainfall can be measured, and the rainfall intensity (mm hr^{-1}) over a given time can be calculated. Other measurements which were taken during the simulation were; the start time of the simulation, the time after rainfall commenced to the start of runoff and the time taken to collect every 100ml of runoff to give an indication of runoff rates. Also during the simulation a visual assessment was made of the spatial extent (percentage plot area) of surface seals and ponds. This assessment was carried out every 5-10 minutes, which also allowed an assessment of the rates of seal and pond formation. Notes were also taken of any experimental variability, for example problems of wind gusting or change in direction, and the presence of any erosion features or preferential pathways of flow within the plot area. Once the simulation had finished the end time was noted and all runoff samples taken for that specific run were amalgamated, and agitated. Three sub-samples of total runoff were taken to calculate sediment loss.

Table 4.3-2 Micro-erosion plot and rainfall simulation measurements overview

Measurement	Method	Location	Time
slope of MP plot (%)	clinometer	Within the MP plot	before the rainfall experiment
area of MP plot (m^2)	Measure width (m) x length (m)	MP plot	before the rainfall experiment
bulk density (g cm^{-3})	use copecki-rings (density rings) at surface (0-5cm) 3 replications	outside the MP plot inside the MP plot	before the rainfall experiment after the rainfall experiment
volumetric and gravimetric moisture content (%)	use soil samples from copecki-rings (see above), 3 replications	outside the MP plot	before the rainfall experiment

		inside the MP plot	after the rainfall experiment
Surface cover (%) of - crops, residues, weeds and stones	use $1m^2$ quadrats of the MP plot estimate the cover (%)	MP plot	before the rainfall experiment
rainfall intensity (mm/h)	place four rainfall gauges evenly around the MP plot and measure rainfall volume	put rainfall gauges next to the MP plot	during the rainfall experiment
assessment of ponding (%)	visual assessment	MP plot	every 5-10 min during the experiment
assessment of sealing (%)	visual assessment	MP plot	every 5-10 min during the experiment
time of runoff start (min:sec)	visual assessment		during the rainfall experiment
time to runoff (min:sec)	a) measure time of every 100 ml increments until 1.0 l is collected b) after that, measure time to every 500 ml until the end of the experiment		during the rainfall experiment
total runoff volume (l)	Amalgmate all the runoff water into a large container and measure the total runoff volume at the end of the experiment.		after the rainfall experiment
sediment concentration ($g\ l^{-1}$)	a) agitate total runoff and sediment collected (after rinsing out the collecting tube) b) take three sub-samples of 100ml each c) dry out the samples d) weigh sediment and express as sediment mass (kg) and concentration ($g\ l^{-1}$)		after the rainfall experiment
total sediment loss (g)	multiply the calculated sediment concentration by the total runoff volume		after the rainfall experiment

MP = micro-erosion plot

4.3.3 Rainfall intensity calibration

The rainfall intensity generated by the simulator was calibrated by placing a $1.5m^2$ grid under the rainfall target area, with catch cups of known diameter placed uniformly on the grid. The catch cups collected rainfall during a set period of time, and the amount of rainfall retained was measured. The mean rainfall intensity was measured as 35mm hr^{-1} (the results can be found in the Appendix D).

After the rainfall simulator had been calibrated it was still important to confirm that the simulator was consistently generating a rainfall intensity of 35mm hr^{-1} for all experimental runs. Differences in rainfall intensity will have an effect on the volume and rate of runoff. Rainfall intensities were analysed for both sites and there were no significant differences in rainfall applied between individual experimental runs.

4.3.4 Rain drop size distribution calculation

The rain drop size distribution of the simulated rainfall was measured to determine the median drop size (D_{50}) and kinetic energy of the 30 minute rainfall event. The method used in the present study was modified from the flour pellet method presented by Hudson (1964). Household flour was sieved through a 2mm mesh, and scooped into a tin (10cm diameter by 5cm deep) to a depth of at least 2cm (representing the minimum depth needed to cushion the impact of raindrops). The surface of the flour in the tins was also roughened to aid absorption of droplet impact.

Each prepared tin was placed under the simulated rainfall for 1-2 seconds, which is sufficient time to capture individual raindrops before they coalesce - a few trial runs were carried out in order to identify the appropriate time of exposure. A tray was placed under the rainfall target area and split into sections and a prepared tin was placed in each section. This allowed multiple tins to be exposed to rainfall at once. Fresh tins of flour were exposed to rainfall for 1-2 seconds every 5 minutes over a 30 minute period.

The exposed tins containing the captured raindrops were then placed in an oven at 105°C for a minimum of 12 hours. Once removed from the oven they were placed into desiccators to avoid re-absorption of moisture in the air. The flour pellets formed on drop impact became hardened and were sieved to determine pellet size classes. The nest of sieves used in this calibration was 700µm, 1, 1.4, 1.7, 2, 2.3 and 2.8mm. The ratio of pellet mass to raindrop mass is known for the flour used, and this ratio was applied to determine raindrop sizes. From the distribution of raindrop sizes, the median drop size (D_{50}) can be calculated. The D_{50} calculated in this experiment was 1.04mm, which is realistic for temperate conditions, where the D_{50} of rainfall is estimated to be between 1-2mm (personal communication; R.J. Rickson, 2005). The maximum drop size recorded was 3.76mm (for more detailed see Appendix C)

4.4 Results

This results section will show all data relating to the micro-erosion plots and rainfall simulations in relation to the previously set out objectives. The data from each site will be considered separately in turn, due to the differences in site characteristics. For example, slope gradient varied between sites. At Loddington the mean slope percentage was statistically the same for all treatments at 3.6%. At Tivington the mean slope over all treatments was much steeper at 7.0%.

Although a protocol was set so that rainfall intensity, simulation duration and plot area were constant between experimental runs, this was not always possible due to equipment variability. The data were therefore standardised to allow comparison between simulation trials. The runoff volume and soil loss results for each simulation were converted to litres or grams per hour, per unit of rainfall (mm) per square metre. The statistical analysis undertaken is described in section 2.2.

4.4.1 Hypothesis one

It is expected that the mean runoff and soil loss will be significantly higher from the conventionally treated plots compared to the conservation treatments (SOWAP and Farmer's Preference). The adoption of conventional tillage will lead to an increase in runoff generation and soil erosion.

The results of runoff volume and soil loss are presented for each site separately. For each site, the runoff volume and soil loss data are analysed for treatment differences at 3 temporal scales:

- The mean loss for the entire sampling period (the mean of each treatment over all micro-erosion plot simulations)
- The mean loss for each cropping season (the mean of each treatment over all micro-erosion plot simulations within each season)
- The mean loss for each simulation trial date (the mean loss of each treatment from 3 replicate micro-erosion plot simulations)

Where results were not normally distributed, and had unequal variance, the data were transformed to allow ANOVA to be performed. Details of the statistical analysis undertaken in this chapter can be found in section 2.2. Where statistically significant differences between treatments have been found, these have been highlighted.

The breakdown of which simulation trials occurred during which cropping season at each site can be found in Table 4.4-1.

Table 4.4-1 Dates of micro-erosion plot simulation trials in relation to cropping season

Site	Cropping Season	Simulation Trial	
		Date	Number
Loddington	One: March 2004 – June 2004	April 2004	1
	Two: September 2004 – March 2005	September 2004	2
	Three: March 2005 to August 2005	March 2005	3
	Four: September 2005 – July 2006	November 2005	4
		April 2006	5
Tivington	One: March 2004 – August 2004	No simulations were performed	
	Two: September 2004 – July 2005	September 2004	1
		March 2005	2
	Three: November 2005 – August 2006	November 2005	3
		April 2006	4

4.4.1.1 Loddington

The runoff volume and soil loss data were not normally distributed and so were transformed (Type I) to satisfy the assumptions of ANOVA. The treatment means were compared over the 3 previously mentioned temporal scales – over the entire sampling period, on a season basis, and across each simulation trial date.

Treatment comparisons of runoff volume over the entire sampling period showed no significant differences (Figure 4.4-1). Although not statistically significant, the mean runoff volume from the conventional treatment was substantially higher compared to the two conservation treatments. When treatment means were compared on a seasonal basis and for all simulation trials no statistically significant differences were found. Although not significant it would appear that during season 3 the runoff volume generated by the Farmer's Preference treatment was the highest of all treatments, but in season 4, the conventional treatment generated the highest volume (Figure 4.4-2).

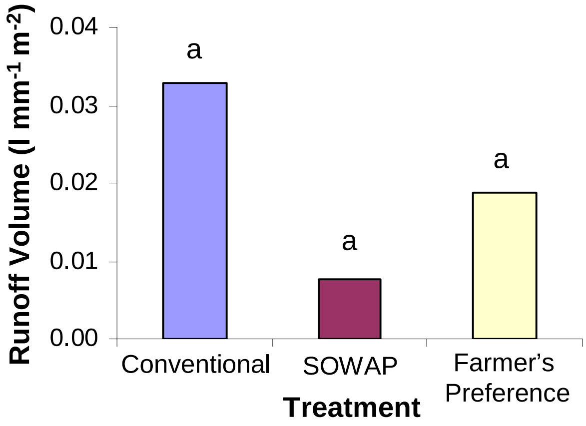


Figure 4.4-1 Loddington: mean runoff volume over the entire sampling period. Letters denote significant differences

These treatment trends on a seasonal basis were confirmed when data were compared across individual simulation trials, where significant differences were found (p=0.003), but only for the third and fourth simulation (season 2 and 3 respectively), where runoff volumes were highest from the Farmer's Preference and conventional treatment respectively (Figure 4.4-3).

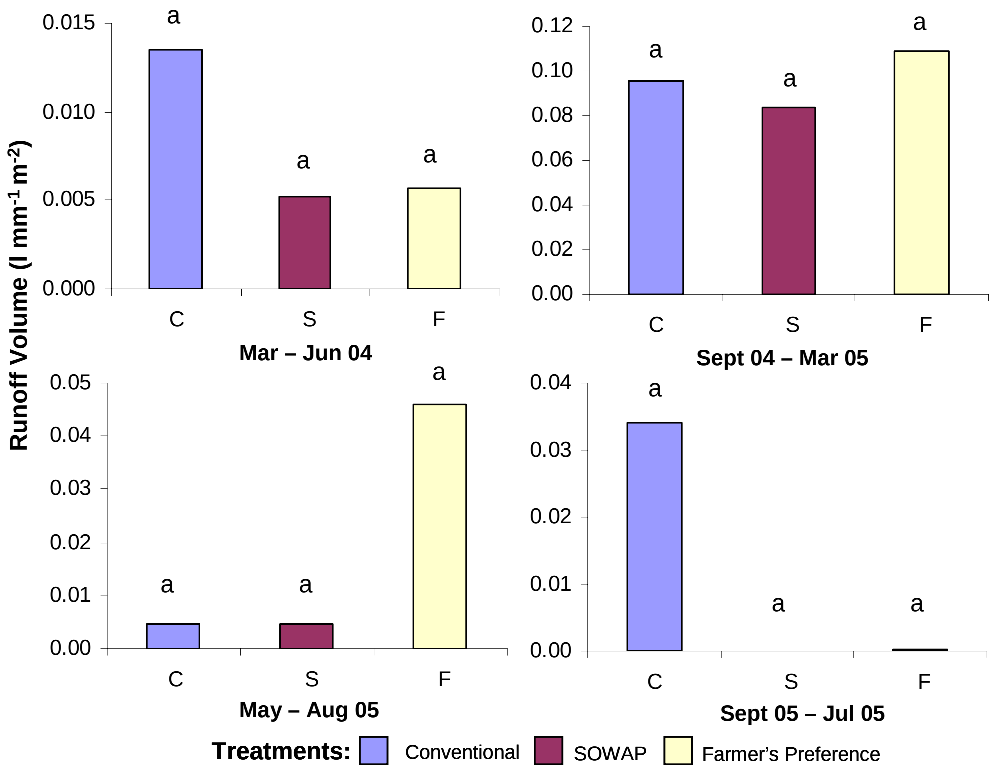


Figure 4.4-2 Loddington: mean runoff volume on a seasonal basis. Letters indicate significant differences.

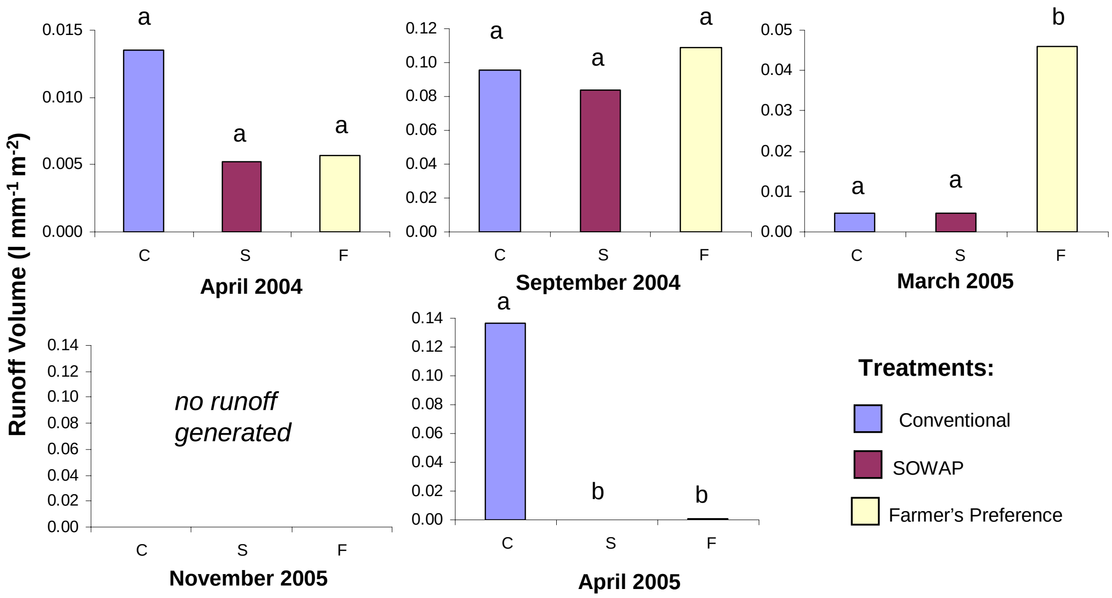


Figure 4.4-3 Loddington: mean runoff volume across all simulations. Letters indicate significant differences.

When mean soil loss was compared between treatments over the entire sampling period, no statistically significant differences were found (Figure 4.4-4). Although not statistically significant, the soil loss from the conventional treatment was notably higher than that from the SOWAP treatment. Comparison of treatment means on a seasonal basis also showed no significant differences in soil losses (Figure 4.4-5).

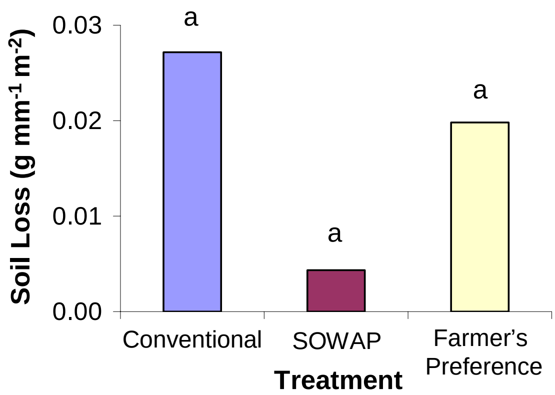


Figure 4.4-4 Loddington: mean soil loss over the entire sampling period. Letters denote significant differences

When data were analysed across each simulation there were significant differences (p=0.01). The results are presented in Figure 4.4-6, and show that there does not seem to be a regular pattern in treatment means, i.e. one treatment does not consistently generate more soil loss than another. The only statistical treatment differences occurred during the third and fifth simulations (season 3 and 4 respectively). In the first case the soil losses were highest from the Farmer's Preference treatment, but only compared to the SOWAP treatment. In the second case, soil loss from the conventional treatment was higher than both conservation treatments.

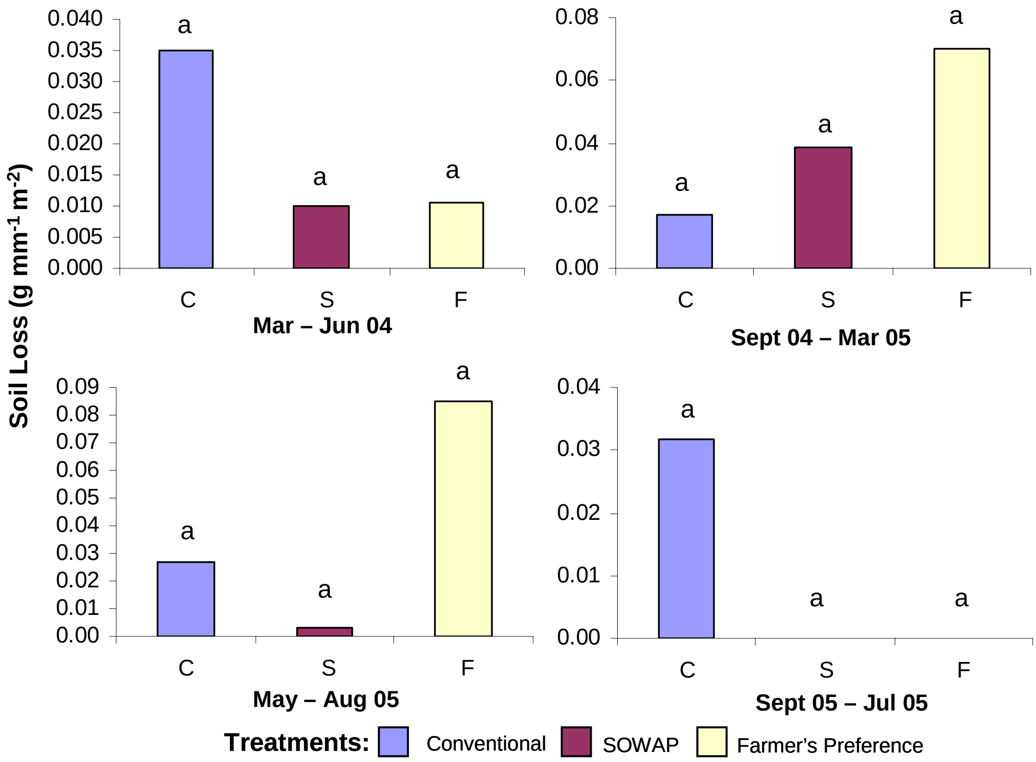


Figure 4.4-5 Loddington: mean soil loss on a seasonal basis. Letters indicate significant differences.

When comparing treatment differences at any of the temporal scales, it should be noted that the variability of the data was sometimes high. However, soil losses do appear to follow similar trends to the runoff results. A simple correlation was done to see if the amount of soil loss could be linked with the volume of runoff generated. This was confirmed by a positive correlation ($r=0.87^*$, $n=39$) between the runoff volume and soil loss, as shown in Figure 4.4-7.

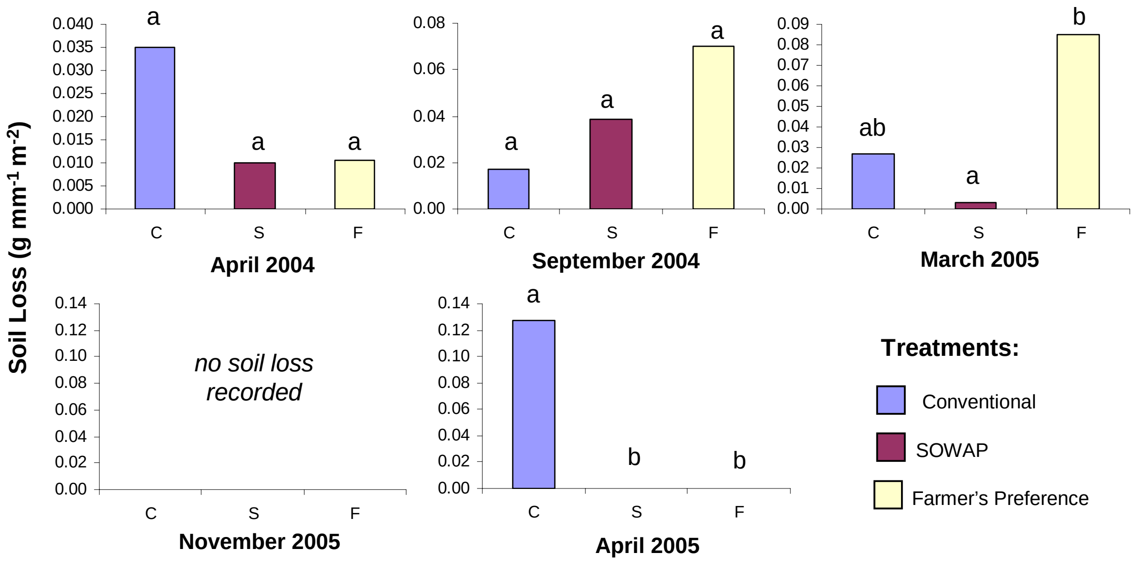


Figure 4.4-6 Loddington: mean soil loss for each simulation trial. Letters indicate significant differences

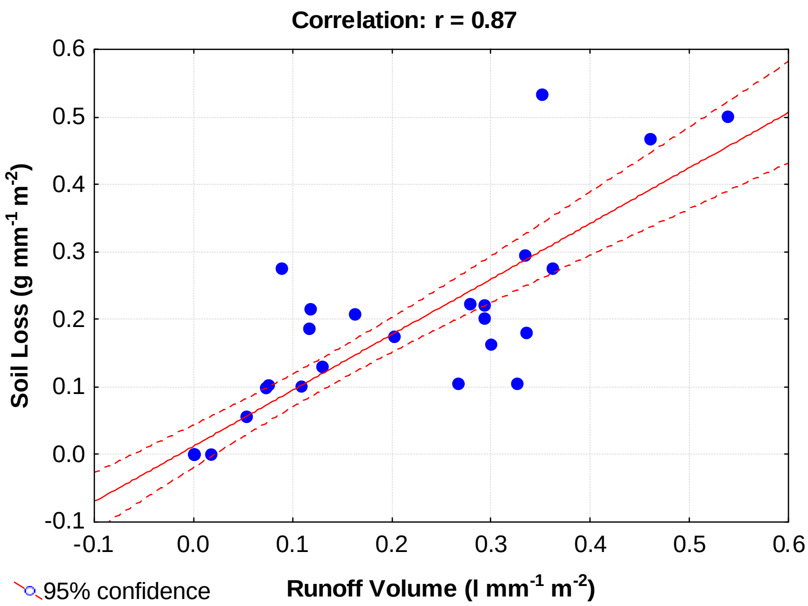


Figure 4.4-7 Correlation between soil loss and runoff volume data from the Loddington micro-erosion plot simulations ($p<0.05$, n=39)

Other correlations were carried out between the soil loss / runoff volume data and surface characteristics / soil properties data as measured at the time of each micro-erosion plot trial. The latter properties include percentage cover (incorporating different types of cover), soil texture, organic matter, organic carbon, moisture content and bulk density. Only the significant correlations have been shown, these have been presented in Table 4.4-2. Two unexpected correlations were found - between runoff and soil loss and organic carbon, and between runoff volume and bare soil/total cover. Both gave the opposite results than what would have been expected.

Table 4.4-2 Loddington: significant correlations between runoff volume, soil loss and soil and surface properties

Factor One	Factor Two	Correlation
Runoff volume	Volumetric moisture content	+0.32
Runoff volume	Bulk density	+0.45
Runoff volume	Soil organic carbon (0-5cm)	+0.65
Runoff volume	Bare soil	-0.42
Runoff volume	Organic matter	-0.53
Soil Loss	Bulk density	+0.41
Soil Loss	Stone cover	+0.43
Soil Loss	Organic carbon	+0.63

4.4.1.2 Tivington

To satisfy the criteria of using ANOVA, the data were tested for normality. The runoff results were normally distributed, but the soil loss results had to be normalised (Type I). Both runoff volume and soil loss were statistically analysed for treatment differences between means at 3 temporal scales – over the entire sampling period (all simulations), by season (all simulations within a given season) and by each individual simulation trial

Statistical analysis of treatment means over the entire sampling period showed the runoff volume from the conventional treatment to be significantly higher than both conservation treatments (p=0.003). Seasonal comparisons showed there were no significant treatment differences, even though in the third season (Nov 05-Jul 06) runoff appears to be much higher than for the two conservation treatments (Figure 4.4-8). No statistically significant differences were found when treatments were compared across the simulations either. Despite the lack of significance it does appear that runoff volume from the conventional treatment was notably higher than for either of the conservation treatments for the first, third and fourth simulation trial (Figure 4.4-9).

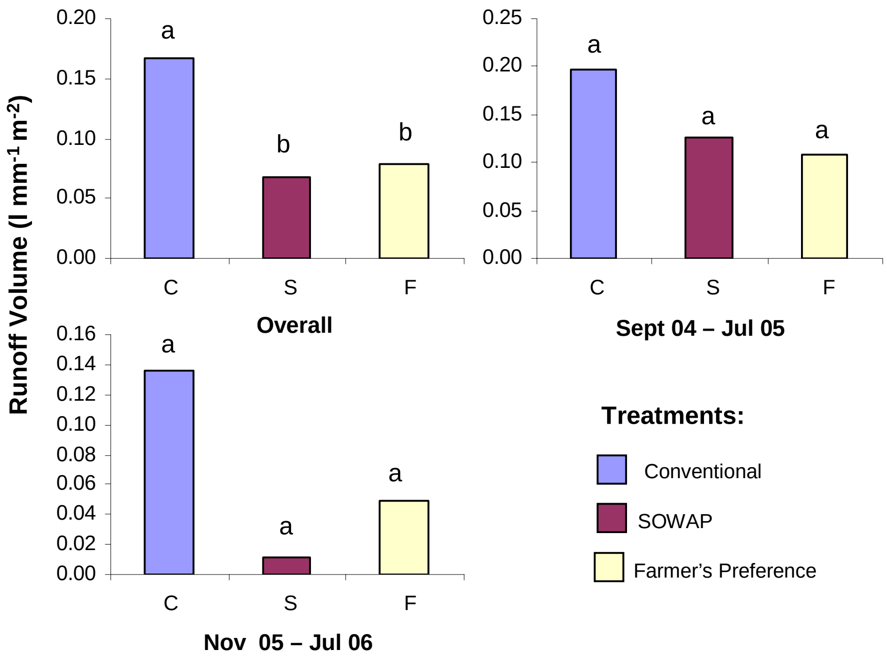


Figure 4.4-8 Tivington: runoff volume overall and for each season. Letters indicate significant differences.

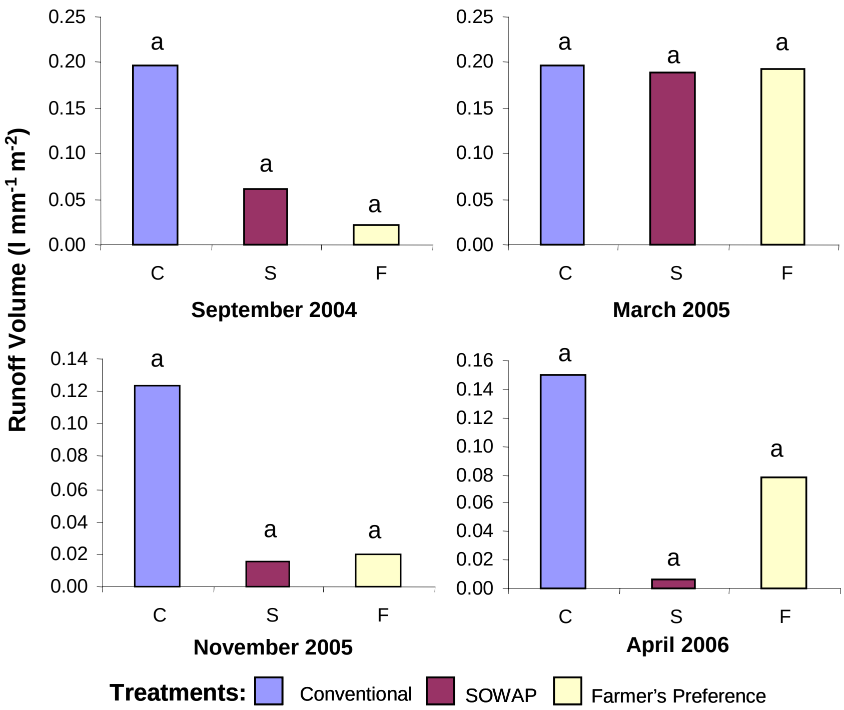


Figure 4.4-9 Tivington: mean runoff volumes for each simulation trial. Letters indicate significant differences.

Statistical analysis of mean soil loss between treatments was carried out for the 3 temporal scales, as previously mentioned. Graphical representation of the results can be found in Figure 4.4-10 and Figure 4.4-11. Statistically significant treatment differences were not found at any of the temporal scales investigated. Although not statistically significant, soil loss from the conventional treatment appears to be higher than at least one of the conservation treatments. This was the case at all 3 temporal scales.

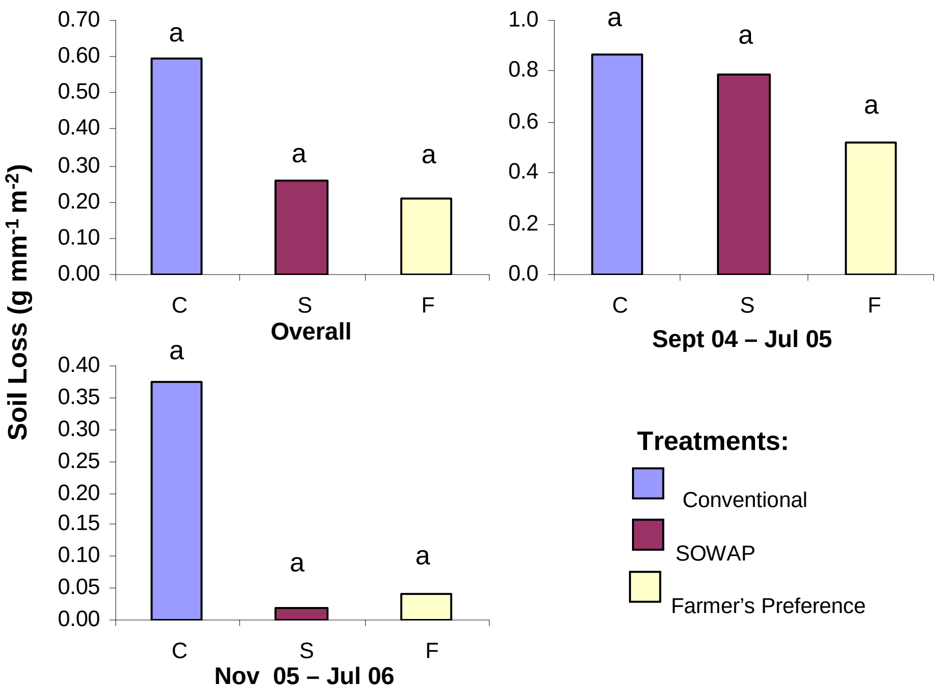


Figure 4.4-10 Tivington: mean soil loss over the entire sampling period and on a seasonal basis. Letters denote significant differences.

As with the results from Loddington, there appeared to be a similarity between the results of runoff volume and soil loss. This is confirmed by a significant positive correlation (r=+0.85*, n=36) between runoff and the transformed soil loss results (Figure 4.4-12). Soil and surface properties were correlated to runoff and soil losses and significant associations have been presented in Figure 4.4-11.

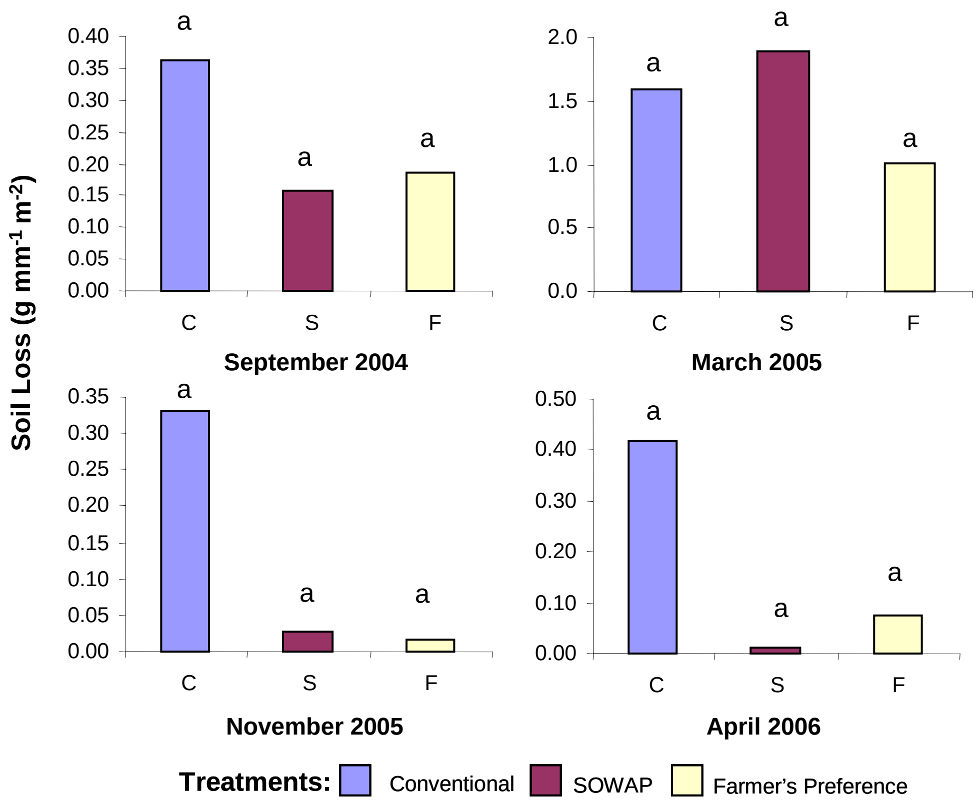


Figure 4.4-11 Tivington: mean soil loss for each treatment for every simulation trial. Letters denote significant differences.

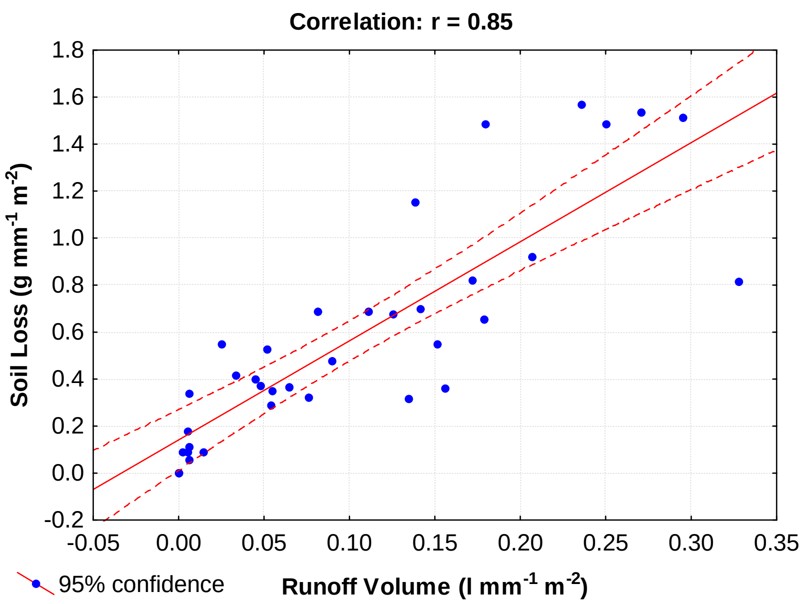


Figure 4.4-12 Correlation between runoff volume and transformed soil loss data from the Tivington micro-erosion plot trials ($p<0.05$, $n=36$)

Unexpected significant correlations were found between soil loss and organic carbon, and the effect of crop cover on runoff volume and soil loss.

Table 4.4-3 Tivington: significant correlations between runoff volume, soil loss and soil and surface properties

Factor One	Factor Two	Correlation
Runoff volume	Bulk density	+0.53
Runoff volume	Crop cover	+0.52
Runoff volume	Residue cover	-0.54
Runoff volume	Surface roughness	-0.36
Soil Loss	Organic carbon	+0.40
Soil Loss	Volumetric moisture content	+0.58
Soil Loss	Bulk density	+0.62
Soil Loss	Crop cover	+0.74
Soil Loss	Residue cover	-0.50
Soil Loss	Surface roughness	-0.38

4.4.2 Hypothesis two

It is expected that the rate of runoff will be faster from the conventionally treated plots compared to both conservation treatments. This is due to expected treatment induced changes in surface cover and roughness, bulk density and organic matter, resulting in increased splash erosion and sealing, and lower infiltration rates compared with the conservation treatments.

Treatment differences in the rate of runoff generation were compared by plotting the cumulative runoff (l) against time for each simulation run. The results for Loddington and Tivington are presented in Figure 4.4-13 and Figure 4.4-14 respectively. Trial 1 at Loddington was part of the pilot simulation, where only 1 replicate was carried out for each treatment. The results have been included as the results are still valid. It should be noted that trial 4 is missing from Figure 4.4-13, because no runoff was generated for those simulations. The remaining 3 trials all consisted of 3 replicates per treatment, although not all simulations generated runoff.

The runoff trends at Loddington all appear to show a linear relationship between cumulative runoff over time. During trial 3 one replicate from the Farmer's Preference treatment generated runoff at a substantially faster rate. However, where runoff was generated there seems to very little difference in the runoff trends from the different treatments overall.

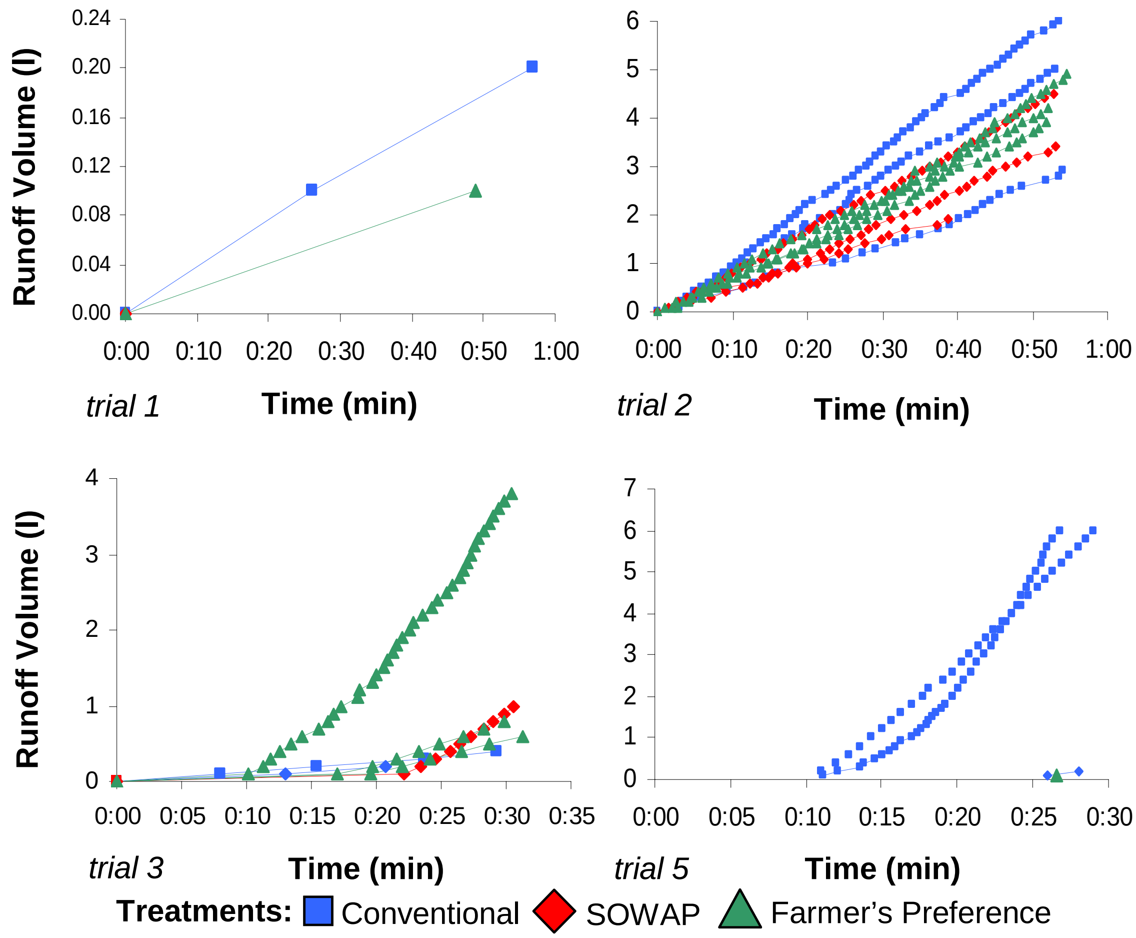


Figure 4.4-13 Loddington: micro-erosion plot cumulative runoff. Dates for the trials are as follows: trial 1 – April 2004; trial 2 – September 2004; trial 3 – March 2005; and trial 5 – April 2006. Treatments were: C = conventional, S = SOWAP and F = Farmer's Preference. Trial 4 generated no runoff.

The results from the Tivington site show that the cumulative runoff from the conventional treatment appears to be markedly different from the conservation treatments, especially during trial 1. The conventional runoff trends during trial 1 indicate that runoff occurs at a much faster rate, with more runoff being generated over a shorter space of time. Trials 3 and 4 also show that runoff is being generated more quickly from the conventional plots compared to the conservation treatments but this is not as pronounced as trial 1.

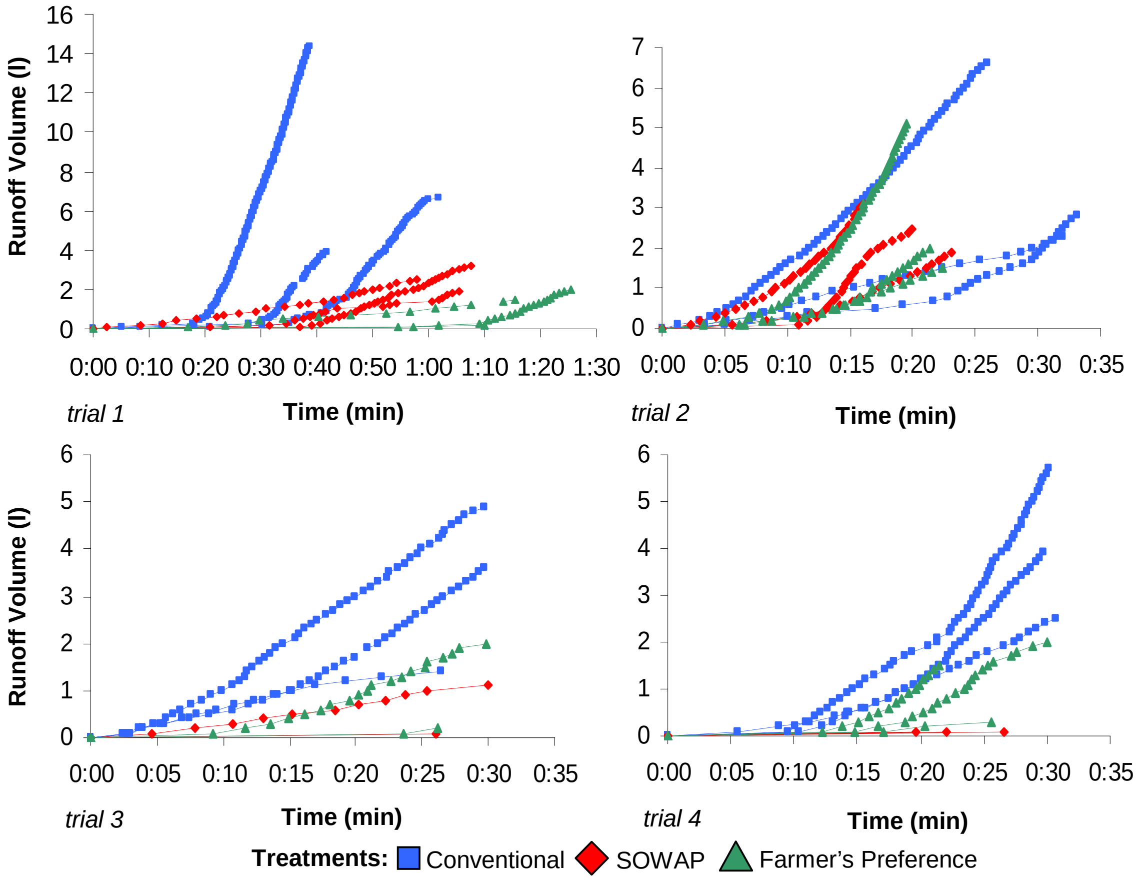


Figure 4.4-14 Tivington: micro-erosion plot cumulative runoff. Dates for the trials are as follows: trial 1 – September 2004; trial 2 – March 2005; trial 3 – November 2005; and trial 4 – April 2006. All trials n=9. Treatments were: C = conventional, S = SOWAP and F = Farmer's Preference

Plotting cumulative runoff against time only allows a qualitative assessment of treatment differences in runoff rate. To allow a more robust comparison, the number of 100ml increments of runoff collected over a 5 minute period was compared between treatments for a total of 30 minutes.

Statistical analysis of the Loddington results show that over a 30 minute period there were no significant differences between treatments in the number of 100ml increments generated on a 5 minute basis. Analysis of the Tivington results also shows there to be no statistical difference between treatments. This confirms the qualitative assessment given by the cumulative runoff shown in Figure 4.4-13 at

Loddington, but not at Tivington. The runoff generated within a 5 minute period over 30 minutes has been presented in Figure 4.4-15. Despite the lack of statistical significance it appears that as time passes the difference in the volume of runoff generated within a 5 minute between the conventional and conservation treatments increases. It is suggested that if simulations were to be carried out for longer than 30 minutes the differences between conventional and conservation treatments would increase in time. This implies that runoff rates between treatments are different; specifically that they are comparatively higher from the conventional treatment towards the end of a 30 minute storm, rather than at the start.

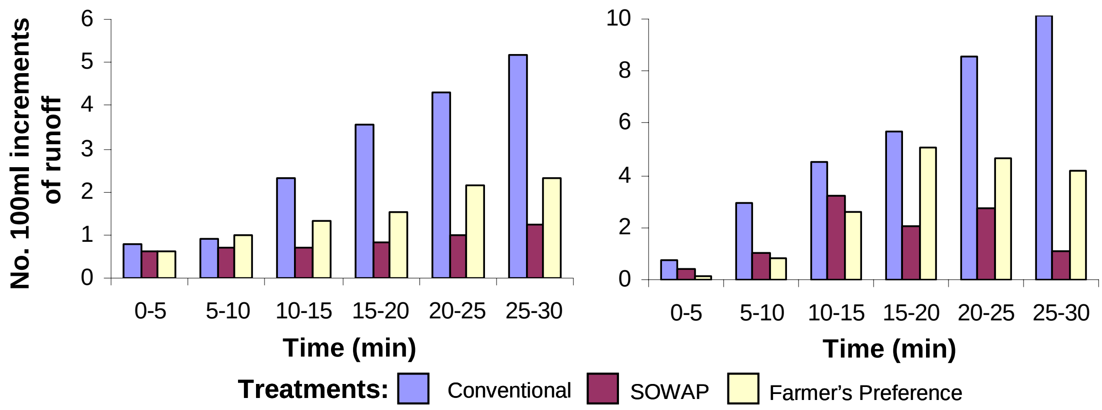


Figure 4.4-15 Overall mean runoff generation within 5 minute increments over a 30 minute period at Loddington (left) and Tivington (right).

4.4.3 Additional correlations

Additional correlations show that at both sites the percentage of seals present before simulations were carried out significantly influenced the percentage of ponds that were present after the simulations. This was an expected response. The percentage area covered with surface seals at the end of the rainfall event showed a significant positive relationship with the mass of soil loss. This was an expected relationship but only occurred at Tivington. The percentage area covered by seals was also significantly correlated with runoff volume, however, this was a negative relationship and was not expected. As seal formation increased runoff was shown to decrease. Looking at the previous set of results, the percentage of seals was the highest on the conservation plots. It was predicted that the conservation treatments would have a higher surface roughness, giving more surface storage capacity within depressions. This was confirmed by a significant negative correlation between surface roughness and runoff volume at Loddington (as stated in hypothesis one). In addition to this, measurements taken of surface roughness over the entire sampling period at Loddington (see field data shown in chapter 3), show the results from the Farmer's Preference treatment were significantly higher than for the conventional treatment. It is therefore unlikely that the percentage area covered by seals reduces runoff, but it is in fact related to degree of surface roughness instead. No relationship between runoff and surface sealing or pond formation was found at Tivington.

Table 4.4-4 Significant correlations between runoff volume and soil loss with the presence of seal and ponds

Site	Factor	Seal %: time zero	Seal %: time end	Pond %: time end
Loddington	Total runoff (l)	x	-0.61	x
Loddington	Total soil (g)	x	x	x
Tivington	Total runoff (l)	x	x	x
Tivington	Total soil (g)	x	0.55	x

4.5 Discussion

It was predicted that the runoff volume and soil lost from the conventional treatment would be significantly higher than that generated from the two conservation treatments. In addition to this, runoff rates were also expected to be significantly higher from the conventional treatments. Anticipated responses were due to changes in soil properties (organic matter, carbon and clay content and bulk density) and surface characteristics (surface roughness and percentage cover) as discussed previously. Changes in these properties were expected to lead to differences in infiltration rates, aggregate stability, and resistance to overland flow. Therefore these properties were measured including spatial extent of surface seals and ponds to gain an understanding of the observed results.

This discussion will be split into two sections. The first will discuss the actual loss of runoff volume and soil mass. The second will be concentrating on rates of runoff generation. Both these sections will be specific to differences between soil management treatments.

4.5.1 Runoff volume and soil loss

Predicted treatment responses of runoff and soil loss were supported by the current data set. At both sites, mean (over the entire sampling period) runoff volume and soil loss was substantially greater from the conventional treatment in comparison to at least one of the conservation treatments. However, only treatment differences in runoff volume at Tivington were statistically significant.

At Loddington the substantially higher volumes of runoff and mass of soil lost from the conventional treatment (over the entire sampling period) could be explained in terms of observed treatment induced changes in soil / surface characteristics. It was discovered that levels of organic matter, organic carbon and clay content were significantly lower on the conventional treatment. The effect of this would be a reduction in cohesive strength between soil particles leading to lower aggregate stability and resultant lower resistance to rain drop impact and splash erosion. The effect of organic matter on runoff was affirmed by a significant negative correlation (r=-0.53*, n=36). In addition, the risk of surface sealing would be greater and risk to runoff generation increased. Increased formation of sealing on the conventional treatment was found not to be the case. However, the percentage area of ponding was substantially greater, indicating an inability rainfall to infiltrate. It was also found that the conventional treatment had overall a significantly lower surface roughness. As a result the resistance against overland flow would have been substantially lower, allowing runoff and entrained sediment to flow downslope more easily. The connection between runoff generation and soil loss was supported by a significant positive correlation (r=+0.87*, n=39).

At Tivington, runoff generation was significantly and soil loss substantially higher from the conventional treatment in comparison to both conservation regimes. This observed treatment difference could be related to present soil and surface properties. It was observed that the clay content of soil from the conventional treatment was significantly lower than at least one of the conservation treatments. As stated previously the implication of this would be a reduction in cohesive strength between soil particles and a lower aggregate stability. Erodibility would therefore be reduced. Surface cover was found to be significantly lower on the conventional treatment to both conservation treatments. This was shown in percentage cover from weeds and surface residues. Although the former is deemed as unfavourable, reducing crop quality, the presence of weeds physically protects the soil from rain drop impact (Laflan & Colvin 1981). The presence of residues also physically protects the soil surface from rainfall and splash erosion. It also has an additional benefit of increasing the microtopography, although a significant treatment difference in surface roughness was not observed. Significant negative correlations affirmed the effect of surface residue on runoff volume ($r=-0.55^*$, $n=34$) and soil loss ($r=-0.50^*$, $n=34$). The overall higher percentage area of exposed soil on the conventional treatment increases risk of soil detachment, surface sealing, pond formation and runoff generation.

Where treatment differences were found the losses of soil and runoff from the conventional treatment were notably higher than the conservation treatments; occurring at both site locations. This pattern was not only found over the entire sampling period (as previously mentioned) but also on a seasonal and micro-erosion plot trial basis. There was however, an exception, which occurred at Loddington during the third simulation trial, March 2005 (season 3). Soil loss was significantly greater from the Farmer's Preference treatment. Runoff volume (although not statistically significant) was also notably higher from the Farmer's Preference treatment. This result could not be explained by measured soil and surface properties, therefore other processes must have been operating. This led to further investigation of field notes and photographs taken during this trial. Visible differences in the plot surfaces were apparent. The Farmer's Preference plots appeared to have pronounced downslope lines, which related to where the crop had just been drilled. These were also found on the SOWAP treatment but to a lesser extent. Expectedly no dill lines were present on the conventional treatment. Images of these plots can be found in Plate 4.5-1.

Conventional

SOWAP

Farmer's Preference

Plate 4.5-1 Loddington: surface conditions during micro-erosion plot simulation trial 3 (March 2005).

It would have been expected that the presence of drill lines would have increased infiltration rates. However, as it can be seen there was little soil cover which would have led to increased risk to soil detachment and sealing. If the drill lines had become sealed quickly, any accumulated water on the surface could be easily channelled downslope via the drill lines into the collection system. This is actually what occurred as shown by Plate 4.5-2.

Plate 4.5-2 Loddington: Farmer's Preference (March 2005) drill line feeding into the collection system.

Overall it was found that the predicted treatment response to erosion was supported by the data from this study. These results could be explained through observed soil properties and surface characteristics. It was found that changes in runoff volume and soil loss on clay soils with a gentle slope gradient were effected predominantly by changes in soil properties. However, differences in surface characteristics seem more influential on sandy clay loam soils with a greater slope gradient.

4.5.2 Rates of runoff generation

It was expected that runoff generation would be faster on the conventional treatments due to *a priori* reasoning of reduction in infiltration and resistance to surface flow. The treatments were compared over a 30 minute storm measuring the volume (in 100ml increments) generated every 5 minutes. No significant treatment differences were found at either site in relation to runoff generation and time. Despite the lack of significance, noteworthy trends emerged.

It was found that during the early part of the storm treatment differences were not apparent, however, as time progressed runoff generation from the conventional treatment increased compared to the conservation treatments. This was an expected response.

During the first part of a storm, rain falls, and is infiltrated unless already capped (e.g. from previous erosion). As time progresses, the soil surface starts to deteriorate due to the impact from rain drops leading to seal formation. From this point treatment differences become apparent. On the conservation treatments surface cover was found to be substantially greater than the conventional treatment. This means that a lower percentage of surface soil is exposed to rainfall, lowering the risk of seal formation. Aggregate stability was also expected to be higher on the conservation treatments, due to the observed higher soil content of organic matter, carbon and clay. Therefore soil erodibility would also be lower. These changes in soil and surface properties would have led to reduction in infiltration, which was visibly seen as surface ponding.

During the latter stages of the simulated storm, i.e. towards 30 minutes, the difference in percentage area covered with surface ponds between treatments reduced. Yet the distinction between conventional and conservation treatments in terms of runoff generation was highly pronounced. At this point in time, the presence of a rough soil surface (as found on the conservation treatments) would have been important in providing increased depressional storage and resistance to overland flow, therefore a reduction in runoff generation.

Plotted time lines of pond formation for each micro-erosion plot trial confirmed that the rate of ponding was greater (in the majority of cases) on the conventional treatment; most visible at Tivington. Although not presented in the text these time lines can be found in the Appendix E.

The results from this current study have important implications to the potential benefit of adopting conservation tillage in the minimisation of runoff generation. The results imply that at the start of a storm and/or for short storms (<10minutes), the form of tillage employed appears to have little or no difference on runoff generation. However, towards the end of a storm (of around 30 minutes) the adoption of conservation tillage can reduce runoff generation.

These findings show that the benefit in reducing runoff generation through the adoption of conservation tillage is dependent on rainfall patterns (i.e. short or long storm events). This has implications for policy makers when utilising conservation tillage as a flood management tool.

4.5.3 Implication of this study

The implication of this study to current research is the reaffirmation that tillage induced changes in soil properties and surface characteristics are intrinsically linked to the risk of erosion. The retention of organic matter, organic carbon and clay are important soil properties, reducing soil erodibility. Surface characteristics including surface cover (from residues and weeds) and roughness are crucial in providing surface depressions for water storage and increasing resistance to overland flow.

Another implication of this study is the effect storm duration has on identifying treatment differences in rate of runoff generation. Treatment differences in runoff rate only appear as storm duration increases. This has implications for future experimental designs and relating results from micro-erosion plots to field scale losses, taking into account natural rainfall patterns.

4.5.4 Future research recommendations

Due to constraints in water supply, time and manpower, the present data were generated before hydrological equilibrium was achieved for any of the treatments. Further studies should run the simulations until equilibrium is reached for each treatment. This will allow observations as to a) when equilibrium is achieved for each treatment and b) the treatment differences (if any) at equilibrium. Such a study will also allow further comparison of runoff and soil losses over time, as well as the changes in soil and surface properties during a rainfall event.

4.6 Conclusion

This present study has shown that the adoption of conservation tillage can be effectively used to minimise runoff generation and soil loss as a result of treatment induced changes in soil properties and surface characteristics. These include increases in organic matter, organic carbon and clay content, higher surface roughness and surface cover. The application of conservation tillage also reduced the rate at which runoff was generated, however this benefit was not found during the early stages of a storm, only as storm duration increased (data not measured over 30 minutes).

The results have shown that changes in soil properties and surface characteristics influence the volume and rate to which erosion occurs. However, it was found that the specific influence of soil / surface properties was site specific. Clay soils with a higher organic matter content and gentle slope gradient were more affected by changes in soil properties. In contrast to this, sandy clay loam soils with low organic matter content and a steeper slope gradient were primarily affected by fluctuations in surface characteristics.

5 Small scale erosion assessment – aggregate stability

5.1 Introduction

The susceptibility of a soil to erosion (i.e. its resistance to detachment and transportation by erosive agents) is related to field characteristics (slope gradient and length, and aspect), meteorological conditions (rainfall and wind intensity, duration and erosivity), anthropogenic influences (tillage operations, choice of land use) and inherent soil characteristics (aggregate stability, texture, organic matter content, porosity and soil biota).

The stability of soil aggregates is often used as a predictive tool to assess soil erodibility (Bryan 1968; Six et al. 2000). The more stable aggregates present within a soil, the less likely the soil is to erode. Aggregate stability is related to many different soil parameters including organic matter content, texture, moisture content, calcium content and microbiology.

There is a danger that this method of assessing soil erodibility is overused to indicate field scale erosion risk, because this approach is relatively easy and inexpensive, compared to the installation of field scale erosion plots. There are several different methods of measuring aggregate stability such as Emerson's dispersion method, slaking tests, wet sieving of aggregates, and exposure of aggregates to simulated rainfall and single dropper tests. It is vital that the appropriate method is representative of the field or design condition in question (e.g. rainfall intensity). This work will test three methods of assessing soil aggregate stability. Each method will assess the differences in aggregate stability as affected by three different tillage practices. Comparisons will be drawn as to the reliability and effectiveness of the different techniques in assessing soil aggregate stability.

The term aggregate stability is the capacity of an aggregate to resist degradation from external forces. In the field these forces include breakdown by machinery and tillage practices, and by water and wind. It is important that during sample collection, transportation, handling and testing care is taken to minimise disruptive forces. The stability of soil aggregates has been used in many investigations as an indicator of soil erodibility, soil processes and soil 'health', as it is a property that changes with changes in soil properties. Examples include work on seal formation (Lado et al. 2004), organic matter (Chenu et al. 2000), infiltration (Abu-Hamdeh et al. 2006), soil microbiology (Kandeler & Murer 1993a; Kiem & Kandeler 1997), rangeland health (Herrick et al. 2001) and runoff generation and erosion processes (Barthès et al. 2000). Despite aggregate stability being commonly used as an indicator of erodibility, there is no standard method to assess this critical soil characteristic. Therefore it is important that methods of testing aggregate stability (often involving the artificial simulation of aggregate breakdown) are consistent, allowing comparisons between results of other research studies or datasets.

5.1.1 Soil and Aggregate Formation

Soil consists of aggregates formed through aggregation of primary soil particles which arrange into different sized structural units. Subunits are bound together to form aggregates through biological and physical-chemical processes. Small aggregate formation (<1mm) is largely controlled by physical-chemical processes associated mainly with clay particles. Larger aggregate formation or formation in sandy soils (with little clay content) relies substantially on biological processes. Primary soil particles are bound together to form aggregates through stabilising agents. These can be biotic, abiotic or environmental (Brady &Weil 2002; Stuttard 1985).

Biotic aggregation can result from organic cements, physical binding agents and through physical movement i.e. by burrowing organisms and plant roots. Organic cements/exudates bind soil particles together and are produced from root hairs, bacteria, fungi and other microbes. Plant roots (especially root hairs) and fungal hyphae bind soil particles together through the formation of sticky networks of organic compounds. This process of physically binding soil particles together leads to the formation of macro-aggregates (≈0.3mm) (Brady & Weil 2002; Rowell 1994). Aggregates are also formed as soil organisms (worms, certain arthropods like termites) and plant roots burrow through the soil, pushing soil particles together. This has been previously mentioned in chapter 1, section 1.2.5.2.5.

Abiotic aggregate formation occurs as a result of cohesive forces between clay and water, flocculation of clay platelets into clay domains or floccules, and from inorganic cements like calcium, iron and aluminium sesquioxides. Flocculated clays are cemented by adsorbing cations (Brady & Weil 2002).

Environmental effects of temperature and water cycles in particular are important in aggregation processes, especially on smectite dominated soils, such as vertisols which are montmorillionite-rich clays with shrink-swell charactersitics. The shrinking and swelling action causes a soil to crack and reform, breaking down aggregates and forming new ones. This soil re-formation and re-organisation can help reduce impacts of soil compaction.

5.2 Aim, Objectives and Hypotheses

5.2.1 Aim

The aim of this chapter is to investigate the effect of different soil management treatments on soil erodibility, as measured by 3 different methods of aggregate stability testing. Any changes in results due to method choice will be highlighted.

5.2.2 Objectives

I To evaluate whether soil management practice alters surface soil aggregate size distribution.

II To quantify and explain the effect of soil management practice on surface soil aggregate stability.

III To make critical comparisons between the results of 3 different methods of aggregate stability testing.

5.2.3 Hypotheses

5.2.3.1 Hypothesis One

It is expected that the surface soil horizon (top 10cm) of the conventionally (C) managed plots would have a different aggregate size distribution with lower mean weight diameter (MWD) compared to soil from either of the conservation treatments (SOWAP, S and Farmer's Preference, F). Mean weight diameter is defined as the summation of the mean diameter in proportion to the total sample weight within each size fraction studied (Kemper & Rosenau 1986) It is also expected that the relative ranking of both dry and wet mean weight diameters (MWD) for all treatments would remain the same, but that results obtained using the wet method would be lower than that of the dry. The wet method of MWD is more destructive as it involves the quick wetting of the soil aggregates, which can lead to the process of slaking (Le Bissonnais 1996b).

Conventional soil tillage involves primary and secondary cultivations. Primary cultivation involves the inversion of the topsoil, burying of previous crop material along with any surface nutrients or residues, including organic matter. Secondary cultivation breaks down the surface aggregates further to create a seed bed of small sized aggregates, which give maximum soil/seed contact. Conservation tillage in contrast to conventional practices, involves a reduction in the mechanical manipulation of the soil as there is only one cultivation applied, and no inversion of the soil takes place. Therefore differences in aggregate size distribution and mean weight diameter (MWD) would be expected. Thus the MWD of soil from the conventional treatment is expected to be lower than the two conservation treatments used in this study, reflecting the assumption that the majority of aggregates within the conventionally tilled soil are composed of smaller sized aggregates.

The aggregate size distribution is important within the soil as this is linked to the processes of soil erosion. As shown by Hjulström's curve (Hjulström 1935; Figure 1.2-2) for a given energy input (e.g. rainfall, runoff etc), particle size affects whether processes of detachment, transport or deposition will take place. This is also reflected by work by Poesen (1992). The curve suggests that both small (<0.2mm) and large aggregates (>10mm) are less erodible, due to cohesive effects in the small aggregates, and mass of particle for the larger aggregates. The implication is that if soil management does affect aggregate size, this will indirectly affect the erosion processes operating.

5.2.3.2 Hypothesis Two

It is expected that soil surface aggregates on conventionally treated plots will be more erodible compared to aggregates on conservation treated plots.

Aggregate stability is expected to be less under conventional tillage because of increased mechanical manipulation of the soil, decreasing levels of organic matter primarily because of the burial of relatively highly organic top soil during ploughing. Organic matter and carbon have been found to have a positive relationship to aggregate stability (Robinson & Philips 2001) and has been previously discussed in chapter 1, section 1.2.5.2.2. As well as organic matter being buried during inversion, surface soil biota which help bind aggregates together are also disturbed by conventional ploughing. Networks of plant roots and fungal hyphae are damaged during ploughing. Conservation soil management aims to reduce the amount of mechanical manipulation of the soil, and generally does not involve topsoil inversion. Conservation management encourages the retention of surface residues or the use of a cover crop during periods when the conventional soil surface would be bare. Residues and cover crop add organic matter (Robinson & Blackman 1989) and provide a food source for soil biota, which in turn can increase the stability of aggregates as discussed above. Specific detail on how soil properties affect soil erodibility can be found in section 1.2.5.2.

5.2.3.3 Hypothesis Three

The degree of aggregate breakdown will vary between different methods of aggregate stability assessment. However, the relative treatment ranks of aggregate stability will remain constant.

Absolute results are expected to differ between the different methods employed to test aggregate stability. This is because each test applies different destructive forces to the aggregates. The immersion based methods of wet sieving and the field test kit are expected to be more destructive when compared to the water droplet method (using a gravity fed rain tower). The immersion based methods simulate flooding conditions, subjecting the aggregates to quick, but total wetting which results in slaking. The raindrop impact method simulates natural rainfall which is destructive but not as destructive as the immersion methods, because water is applied intermittently (as individual raindrops). It is therefore expected that the immersion based methods will yield fewer stable aggregates than the raindrop impact method. Despite the differences between methods, the relative ranking of aggregate stability between tillage treatments is expected to remain the same, irrespective of method employed.

5.3 Methodology

Two sites located in the UK were used in this investigation, at Loddington, Leicestershire and Tivington, Somerset. Three soil management treatments were tested; conventional tillage (C) and two forms of conservation tillage - SOWAP (S) and Farmer's Preference (F). For more detailed site and treatment descriptions see chapter 2). Samples of the soil aggregates were obtained adjacent to the micro-erosion plots (section 4.3.1) to allow comparison of results at the two spatial scales. Sampling occurred at Loddington in April 2004, September 2004 and March 2005 which corresponded to 3 different cropping seasons; winter wheat, cover crop/stubble and spring beans (for more details see section 2.1.4). At Tivington sampling dates were September 2004 and March

2005 which corresponded to only 1 cropping season; winter wheat. The number of sampling dates was constrained by the time available for the tests to be carried out, and subsequent analysis. Also, the key aim of this study was to compare differences between treatments at any given point in time, rather than changes in aggregate stability over time. It was expected that such differences may not become apparent in the time scale of this project (<3 years).

5.3.1 Sampling

During sample collection of soil aggregates, disturbance is inevitable as soil is removed from the ground when using a spade, trowel or auger. Handling and transportation also causes damage. As soon as soil is removed from the field it is instantly modified; some have argued that this makes tests for stability unrealistic of the field situation (Kemper & Rosenau 1986). This may be true, but if a protocol is devised and kept to, relative comparisons between different soil management treatments are possible.

To minimise disturbance great care was taken during each stage of sampling the aggregates. Once the sample had been removed it was stored and prepared for testing as soon as possible after sampling. It has been shown that stability can increase with storage time (Kemper & Rosenau 1986). This relates to soil particles dry out causing neighbouring particles to bond together through cohesion and the concentration of carbonates and organics.

Three random areas of undisturbed soil adjacent to the micro-erosion plots (section 4.3.1) were used for sample generation in this investigation. Soil samples were delicately removed from the top 5 cm of soil using a hand trowel. This soil was then placed into large plastic bags and cushioned during transport to avoid disturbance. Each sample bag was gently emptied and laid out one aggregate layer thick across a drying tray (or across several depending on the amount of soil). All aggregates were left to air dry at room temperature away from direct sunlight at a constant humidity for 3 days.

5.3.2 Sieving for Aggregate Size Distribution

The air dried aggregates were manually sieved on flat bed sieves, through a wide range of sizes to ascertain the aggregate size distribution of each sample. Seventeen sieve sizes were used in the determination, ranging from 12.5mm to 0.075mm. Manual sieving causes inevitable breakdown of aggregates as they are passed over each sieve, so to minimise this, three separate stacks of sieves were used at a time. The first stack comprised of sieve sizes 12.5, 9.5, and 5mm, the second used sizes 4, 3.35, 2.8, 2.35, 2, 1.7 and 1.4mm and the third used sizes 1, 0.71, 0.5, 0.425, 0.3, 0.15 and 0.075mm. The large number of sieve sizes was chosen to detect any possible treatment differences in aggregate size distribution. Increasing the number further would have made the work impractical and time consuming. The sizes used gave a detailed distribution of aggregate sizes within a soil sample, unlike expressions such as mean weight diameter (MWD), which is used in the majority of research on soil aggregate sizes. The latter only gives one figure to represent the distribution of aggregate sizes within a soil. The method employed here gave much more detailed treatment comparisons.

The air dried aggregates were placed on the top of each stack of sieves and manually shaken horizontally a total of 10 times and then gently tapped down 3 times to make sure all loose particles fell through. This technique was derived from previous pilot runs where visual assessments were made on the number of oscillations it took for the majority of soil to pass through the sieves. After 10 oscillations, soil appeared to breakdown as a result from the action of the mechanical sieving itself. As sieving (manual or mechanical) causes aggregates to breakdown, due to abrasion, it was important to standardise the method used.

Both Chepil (1962) and Lyles et al. (1970) highlighted the problems of breakdown of aggregates through sieving. It has been suggested that re-sieving a soil several times and tracking the rates of breakdown between each re-sieve allows back extrapolation as to what the original soil distribution would have been. Caution must always be taken when extrapolating results. For this investigation it was not required, as treatment differences were being compared and any errors associated with breakdown due to the sieving process itself were assumed to be equal for all treatments, and thus were effectively cancelled out.

The mass of aggregates retained by each sieve was measured, and the aggregates were bagged and labelled for later testing. The mass of aggregates on each sieve were calculated as a percentage of the total mass of aggregates, and plotted to produce an aggregate size distribution graph. The mean weight diameter (MWD), as pioneered by Van Bavel (1949), was also calculated as follows:-

$$\sum_{i=1}^{n} x_i w_i \qquad \textbf{(5.3-1)}$$

Where x_i is the mean diameter of each size class (mm) and w_i is the proportion of total weight of that class size in relation to the whole. It was expected that the conventional treatment would consist of more small sized aggregates, as represented by a smaller MWD in comparison with the conservation treatments.

5.3.3 Tests for Aggregate Stability

Testing for aggregate stability can be achieved using wet or dry methods. Dry methods involving dry sieving such as using a rotary sieve which is used to ascertain soil susceptibility to wind erosion (Chepil 1962). Wet methods simulate rainfall or flooding conditions, as used in this investigation. Three wet based methods of aggregate stability testing were used - a raindrop impact method (using a gravity fed rain tower to simulate a natural rainfall event), a wet sieving method and a field test kit method (both simulating flooding conditions).

5.3.3.1 Pre-treatment

In both dry and wet methods of aggregate stability, it was important that the starting conditions of the soil samples being tested are the same, if comparisons are to be made. Aggregates with different moisture contents will react to disturbances in different ways. Even if testing with air-dry aggregates, moisture will still be present unless the soil is oven dried, but this will change the inherent soil properties under investigation. Therefore pre-wetting aggregates to a uniform moisture content was chosen to ensure identical starting conditions. The method used to achieve uniform moisture content is also important, as different techniques lead to differing degrees of destruction. Three common ways to pre-wet an aggregate are through quick wetting, suction or the use of an atomiser.

The first technique of quick wetting involves the complete immersion of soil aggregates into water and allowed to stand for a set period of time, to ensure complete wetting (Grieve 1979). This method is fast and can be done in any laboratory, but it is the most destructive of the three techniques, causing the most amount of aggregate breakdown. When water is applied to an air dried aggregate, trapped air is forced out (Emerson 1954), sometimes with an explosive force (Lyles et al. 1974) – a process known as "slaking". It is therefore important to immerse the air dried aggregates as slowly as possible. This is difficult in itself, but the water must also be applied at a constant rate for every aggregate.

The second technique of suction or wetting via capillarity involves placing air dried aggregates on to a permeable cloth placed on a sand table, where the aggregates are wetted up from beneath. Different rates of suction can be applied by the use of vacuums to achieve set moisture contents such as field capacity. This method was adopted in work by Low (1954). Unfortunately this technique can take up to 2 days for each sample to reach complete saturation, making it impractical if testing a large number of samples. Also, once saturated, the aggregates are extremely difficult to transport (e.g. to the rainfall simulator) without breakdown.

The third technique uses an atomiser. Aggregates are placed within a chamber and a fine mist is sprayed by an atomiser for several hours or until saturation is reached. This equipment needs to be calibrated before use to identify the time needed for saturation to be achieved. Aggregates are saturated slowly, reducing the amount of damage caused to the aggregates. Unfortunately this equipment is very expensive and was unavailable to this present study.

Due to equipment and time constraints, the first technique of fast wetting had to be adopted. The moisture content therefore used was saturation. It was also felt to be the most appropriate as other known moisture contents such as field capacity change with site and season. To obtain a uniform moisture content, 10g [± 0.5] of the air-dry aggregates were placed onto a concave glass dish where water was introduced slowly at a constant rate until all aggregates were completely immersed. The aggregates were allowed to stand for 30 minutes to ensure complete saturation. This was sufficient time (Grieve 1979) for even the larger aggregates to become saturated. It was imperative that water was introduced on very slowly to minimise destruction of the test aggregates.

5.3.3.2 Aggregate sizes tested

Before stability tests were carried out it was important to assess the relationship of aggregate stability and size on soil from each site location. Previous research indicates that relationships exist between particle size and stability (Hjulström 1935; Poesen 1992).

Aggregates sized 0.5mm to 9mm were tested for stability in a preliminary experiment to assess the effect of aggregate size on aggregate stability. Five size classes were tested, these were 9.95-5, 5-3.35, 3.35-2, 2-1 and 1-0.5mm (the range of aggregate sizes tested is shown by the two dotted lines in Figure 5.3-1). The smallest size was determined by the aperture of the sieve used in the wet sieving experiment (see section 5.3.3.3.1). The results of this preliminary test can be found in Appendix F. This experiment found that there were no significant differences in stability of the different aggregate classes. Therefore the size range 3.35-5mm was used in aggregate stability tests. Similar size ranges have been found in other research (Gimeno-García et al. in preparation; Legout et al. 2005; Low 1954; Barzegar et al. 2003). This range of aggregates was used for all stability methods employed in this chapter. In this method, any aggregates retained on the 0.5mm sieve after the test is carried out are assumed to be "stable". The aggregates to be tested for stability were taken from those produced by the aggregate distribution sieving (see section 5.3.2).

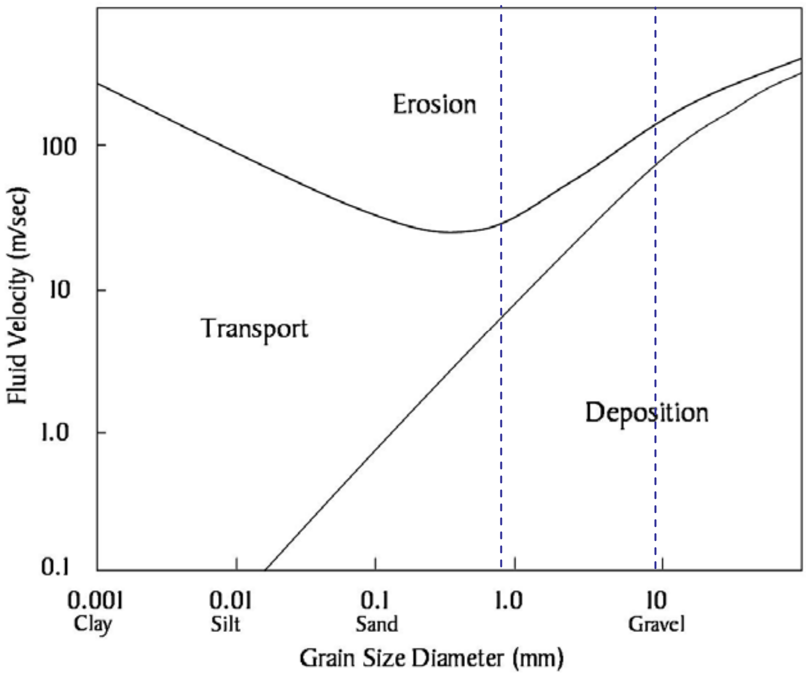


Figure 5.3-1 Hjulström's curve – as modified from Hjulström (1935)

Herrick et al. (2001) recommends aggregates from 5-9mm in size are used in the field test kit (FTK), however, in the present study aggregates between 3.35-5mm were also tested to allow a fair comparison with the results from the wet sieving method. Only results from the aggregates sized 3.3.5-5mm are shown for the FTK because the stability results gained from both sizes (5-9 and 3.35-5mm) were statistically the same. In the FTK method, any aggregates retained on the 1.6mm sieve after the test is carried out are assumed to be "stable".

Aggregates between 3.35-5mm were also tested via rain drop impact under the gravity fed rain tower, again to allow comparisons to be made between methods. In this method, as with the wet sieving method, any aggregates retained on the 0.5mm sieve after the test is carried out are assumed to be "stable".

5.3.3.3 Immersion based methods of aggregate stability

Testing the stability of aggregates in water began with work based on Stokes Law, looking at the settling velocity of aggregates in a viscous fluid of known density and viscosity. The method is similar in principle to the determination of particle size distribution but with larger sized particles. Notable work was done by Middleton (1930) who devised the dispersion ratio, and by Davidson & Evan (1960).

Yoder (1936) modified work originally by Tiulin (1928) to devise the now well used method of wet sieving aggregates to test for stability and size distribution. Wet sieving involves soil aggregates being placed on the top sieve of a nest of sieves, which are then submerged in a container of water. The sieves are then oscillated for a set time period at a set rate of oscillations per minute. Aggregates remaining on the sieve surfaces are collected, weighed, oven dried and re-weighed. The mass of aggregate remaining above a defined sieve size (as a percentage of the initial mass) gives an indication of stability. The size and number of sieves used are specific to the work being undertaken. The mean weight diameter (MWD) can also be calculated Van Bavel (1949). This is the proportion of each aggregate size in relation to the whole, multiplied by the mean diameter of all sizes measured (as shown in Equation (5.3-1). This results in a single number to represent the mean aggregate size (mm) of that soil. The general assumption is that the greater the MWD, the more stable the soil (Yoder 1936; Le Bissonnais 1996). Wet sieving and the use of MWD appear to be common place in research into wet aggregate stability (Kandeler & Murer 1993; Legout et al. 2005; Six et al 2000).

Other notable tests of wet aggregate stability are the Emerson Aggregate Test (Emerson 1967) which generates a stability index for 3-5mm diameter aggregates when immersed in water; Child's quick wetting test (Childs 1940), which simulates flooding and sheet erosion (Morgan & Rickson 1988). This was later modified by Collis-George & Figueroa (1984). More recently in 2001, Herrick et al. devised a field test kit that is quick and inexpensive to use, which combined the Emerson Aggregate Test (Emerson 1967) and elements of the wet sieving test, using a 1.65mm sieve.

The two immersion based methods of wet aggregate stability testing used in this investigation were wet sieving and the field test kit

5.3.3.3.1 Wet Sieving

There are no standard procedures (i.e. number of oscillations, aggregate initial moisture content, sieve sizes or duration of experiment) for determining water stable aggregates in soils. This study has chosen a method based on work by Low (1954) and Van Bavel (1952), which has been widely adopted in several countries. Air dried aggregates were selected and pre-treated using the method mentioned in section 5.3.1. A sub-sample of pre-treated aggregates (3.35-5mm) was taken from each sample to determine initial moisture content before wet sieving. Soil aggregates, 3.35-5mm (10g, ±0.5) were saturated and placed onto a nest of sieves in a randomised design. Each sieve stack consisted of 5 different sieve sizes - 2, 1, 0.5, 0.125, 0.1mm aperture. The nest of sieves were bolted together to avoid soil being lost during the experiment, immersed slowly at an angle into a large water container making sure there were no air locks. Six sieve stacks could be tested at once. The immersed sieves holding the aggregates were moved up and down through the water at a rate of 30 strokes per minute. Previous runs (Low 1954) using this method have shown that no further appreciable breakdown of aggregates occurs after 500 strokes, therefore a running time of 17 minutes (510 strokes) was deemed to be appropriate and concurred with previous research using the same equipment (Plate 5.3-1).

During the 17 minutes of agitation the water level within the container was kept constant. After the test, the nest of sieves was removed slowly from the water and allowed to drain. The sieves were then unbolted and separated from each other. Any soil material retained on the surface of each sieve was transferred to a pre-weighed tin, oven dried for 48 hours at 105°C and reweighed (step a).

Plate 5.3-1 wet sieving apparatus (top left); a single stack of sieves (top right); retained aggregates after sieving (left)

To correct for the presence of coarse primary particles, the oven dry aggregates (from the previous step) greater than 0.5mm were submerged in water and all soil particles washed through a 0.5mm sieve. The remaining material (coarse primary particles) on the sieve surface were oven dried for 24 hours at 105°C and reweighed. The oven dry weight of the coarse primary particles was then subtracted from the air dry weight of material >0.5mm as presented in step a. This figure represents the mass of stable aggregates above 0.5mm (step b). This mass of stable aggregates was then represented as a percentage of the original 10 g of air dry soil which had been standardised for its initial moisture content. The MWD was also calculated (see Equation (5.3-1) for more details).

5.3.3.3.2 Field Test Kit

The field test kit is a visual assessment of aggregate breakdown and was based on the design by Herrick et al. (2001), who designed a simple kit that could be easily and inexpensively used in the field to determine the stability of a soil. Within 10 minutes it is possible to test up to 18 soil aggregates for soil stability. This method is based on immersion and sieving of soil fragments in water. The kit consists of a segmented box. Within each section there is a small 1.65 mm sieve. Specific detail on how to recreate the equipment (Plate 5.3-2) can be found in Herrick et al. (2001).

Plate 5.3-2 Field test kit (FTK) apparatus. Test box dimensions 21 x 10.5 x 3.5 cm. Sieve size 2cm diameter and 1.65cm aperture.

Air-dry soil aggregates, 3.35-5mm (10g ±0.5) (as described in section 5.3.1) were placed onto each sieve outside the box and left for 1 hour to ensure uniform moisture content. The segmented box was then filled to a depth of 2cm with deionised water. Each sieve holding the aggregates was carefully and slowly immersed into a segment within the box and left for 5 minutes. Each sieve was then completely lifted out of the water and re-immersed slowly over 2 seconds. The lifting and re-immersion process was carried out a total of 5 times. Each sieve and any remaining soil material were completely removed from the water and placed on a work bench. A visual assessment of the percentage soil remaining on the sieve was used to assign a stability class to each aggregate surface (Table 5.3-1).

Table 5.3-1 Field test kit stability classes. Modified from Herrick et al. (2001)

Stability Class	Criteria for assignment to stability class
5	<10% soil remaining on sieve
4	10-25% soil remaining on sieve
3	25-50% soil remaining on sieve
2	50-75% soil remaining on sieve
1	75-100% soil remaining on sieve

As this is a visual assessment of percentage soil aggregate remaining it is important to be consistent and to use the same person in the assessment each time.

5.3.3.4 Water droplet impact method of aggregate stability

Aggregate stability testing via water drop impact is carried out either by exposing the aggregates to simulated rainfall, as described by Lovell & Rose (1988) and Loch (1994), or by allowing single water drops to fall onto individual aggregates (Farres & Cousen, 1985). The single dropper test originated with work by McCalla (1944). The method has been used and developed by Smith & Cernuda (1951), Low (1954), Bruce-Okine & Lal (1975), Bergsma & Vaienzuela (1981) and Farres & Cousen (1985). The premise of this method is to place an aggregate over an aperture of known size and allow individual water droplets to fall systematically onto the aggregate. When the aggregate falls through the aperture it is considered eroded or destroyed. The number of water droplets it takes to destroy the aggregate is noted, and through this comparisons can be made between aggregates (Rahim 1990).

When using drop forming devices it is important to obtain a consistent fall rate, with uniform sized droplets. The problem of drift of the falling drops is associated with water drop devises and is exacerbated at higher drop heights. Work by Gunn & Kinzer (1949) and Mutchler (1965) attempted to reduce drift in flight and limit droplet spin, to increase the number of droplets hitting an aggregate. They achieved this by grinding off the points of the drop forming needles. Drift and droplet spin are also important factors when using a rain tower. Drift is especially important as rain towers are usually higher than single dropper apparatus and so the vertical raindrop fall zone should be enclosed to limit the effect of wind on drift.

Raindrop size is also important and is dependent on the type of drop-former used (glass burette, capillary tubing or hypodermic needle). It is best to obtain a drop size that is as close to natural rainfall (Laws & Parsons 1943). The median drop size (D_{50}) is usually used, but this varies with rainfall intensity. A typical rainfall event in the UK would have a D_{50} of between 1-2 mm (personal communication; R.J. Rickson, 2005). Drop height is another important factor, as this affects the fall velocity and thus energy of the drop to disrupt the target aggregate. To simulate natural rainfall, drops should fall as close to terminal velocity as possible. Hudson (1964) highlighting work by Laws & Parsons (1943) showed that a drop height of 8m is sufficient for a 4mm diameter droplet to reach 95% of its terminal velocity.

A rain tower uses multiple needles to simulate a rainfall event of known intensity, which is controlled by the number of needles used and the head of water above them. The head of water must be constant to maintain pressure and rate of droplet formation. In both rain tower and single dropper tests it is important that the chemistry and temperature of the water used to simulate rainfall are constant. The water should be clean and treated to remove any salts, dirt and certain chemicals like calcium, to prevent needle blockage. The water used in this experiment was tap water which had gone through a reverse osmosis process, which removes all salts. Work by Barton (1994) showed that water treated in this way did not affect aggregate stability results and gave similar results to those using natural rain water.

In this investigation a gravity fed rain tower was used to determine aggregate stability via water drop impact. Aggregates were subjected to artificial rainfall generated from a 9m high tower. The height of the tower ensured that a high percentage of droplets would reach terminal velocity (Low 1954). At the top of the tower a tray housing 860 hypodermic needles is filled with water at a constant rate. The hypodermic needles were gauge size 24 (0.559 mm (Jensen Global 2006), producing water droplets large enough to overcome surface tension. A fine mesh was placed 1m below the needles to split the falling drops into randomised, smaller droplets. The rainfall intensity was altered by the number of needles which were left open (for details on the layout of the needles left open see Appendix G, and the height of water above the needles, which was set at 16mm by means of a weir at the exit point of the tray. Before the aggregates were tested the rainfall intensity and drop size distribution were calibrated.

5.3.3.4.1 Rainfall intensity calibration

The rainfall intensity of the rainfall tower was calibrated by placing a $0.5m^2$ grid under the rainfall target area, with catch cups of known diameter placed uniformly on the grid. The catch cups collected rainfall during a known period of time, and the amount of rainfall retained was measured. By calculating rainfall intensity across the grid, the spatial distribution of the rainfall was identified. The mean rainfall intensity was measured as 35mm hr^{-1} (the results can be found in the Appendix G).

5.3.3.4.2 Rain drop size distribution calculation

The rain drop size distribution was also measured to determine the D50 of droplet size distribution and thus kinetic energy of the rainfall event. The method used was modified from the flour pellet method by Hudson (1964). This has been previously used in chapter 4 and so detailed descriptions of this method can be found in section 4.3.4. The D_{50} calculated in this experiment was 0.86mm, which is slightly low for temperate conditions, where the D_{50} of rainfall is estimated to be between 1-2mm (personal communication; R.J. Rickson, 2005). The maximum drop size recorded was 4.53mm (for more detailed see Appendix H).

5.3.3.4.3 Aggregate stability testing

A sub-sample of pre-treated aggregates was taken before testing from each sample to determine initial moisture content. This was achieved by taking 10g [±0.5] of air dried aggregates, 3.35-5mm (that had been pre-treated as per the method in section 0) and saturating them in 20ml of deionised water for 30 minutes, making sure each aggregate was fully immersed. This was sufficient time for even large aggregates to become saturated.

The saturated aggregates were placed in the centre of a 0.5mm sieve with a receiver. This sieve size was used to allow comparison with the results gained from the wet sieving method. Four sieves (Plate 5.3-3) were placed under the rainfall tower at any one time, and a sample of aggregates was assigned to each sieve in a randomised design. Saturated aggregates were then rained on for 17 minutes (allowing comparison with the wet sieving method) at an intensity of 35mm hr^{-1} which represents a 1-6 year storm (NERC 1975). This intensity was also chosen to allow comparisons to work undertaken in chapter 4. After rainfall application, soil material remaining on top of the sieve was washed off into a weighed tin. The contents of the tin were then oven dried at 105°C for 48 hours and re-weighed. Soil and water collected in the receiver were also collected into a weighed tin and oven dried in the same way. Once the tins were re-weighed the soil from the receiver was discarded, but the soil collected on the sieve was flushed through a 0.5mm sieve. This was done to separate aggregates from any coarse primary particles. The latter was held on the sieve and collected into a tin, oven dried at 105°C for 24 hours and re-weighed. This weight was then subtracted from the total weight of material held on the sieve following exposure to rainfall, to calculate the percentage of aggregates retained on the 0.5mm sieve compared to the initial mass of aggregates (taking into account initial moisture content).

Plate 5.3-3 Rain tower apparatus; needle tray (top left), rainfall catch area (top right), and four sieves with saturated aggregates ready for testing (left)

5.4 Results

This section will show all data relating to aggregate stability in relation to the previously set out objectives. The data will be separated out into method and site location. A description of the statistical analysis carried out can be found in section 2.2.

5.4.1 Hypothesis one

It was expected that soil from the conventionally managed plots would have a different aggregate size distribution, with lower mean weight diameter (MWD) compared to soil from the two conservation treatments (SOWAP or Farmer's Preference). It was also expected that there would be no difference in relative treatment ranking for both dry and wet mean weight diameter (MWD) results.

5.4.1.1 Dry aggregate size distribution

At Loddington the percentage of dry aggregates retained by the different sieve sizes was statistically ($p<0.001$) different between treatments (Figure 5.4-1). At Tivington the results also showed a significant difference ($p<0.001$) between treatments (Figure 5.4-2). At both site locations significant treatment differences were observed for both the entire sampling period and for individual seasons.

It was expected that aggregates from the conventionally treated plots (C) would have a lower percentage of large sized aggregates in comparison to the other two conservation treatments; SOWAP (S) and Farmer's Preference (F). This was the case at Loddington where there was a tendency for there to be more aggregates sized 0.15mm to 1mm compared to the conservation treatments. For both individual and combined seasons most significant treatment differences at Loddington occurred in the proportion of aggregates sized 4mm and above. However, at Tivington this was not the case, and there was a tendency for the proportion of aggregates between 0.15 and 4mm to be lower from the conventional treatment compared to the SOWAP conservation treatment. Also the proportion of aggregates of 5mm and above from the conventional treatment was much higher than for the SOWAP treatment, which was the opposite of what was expected.

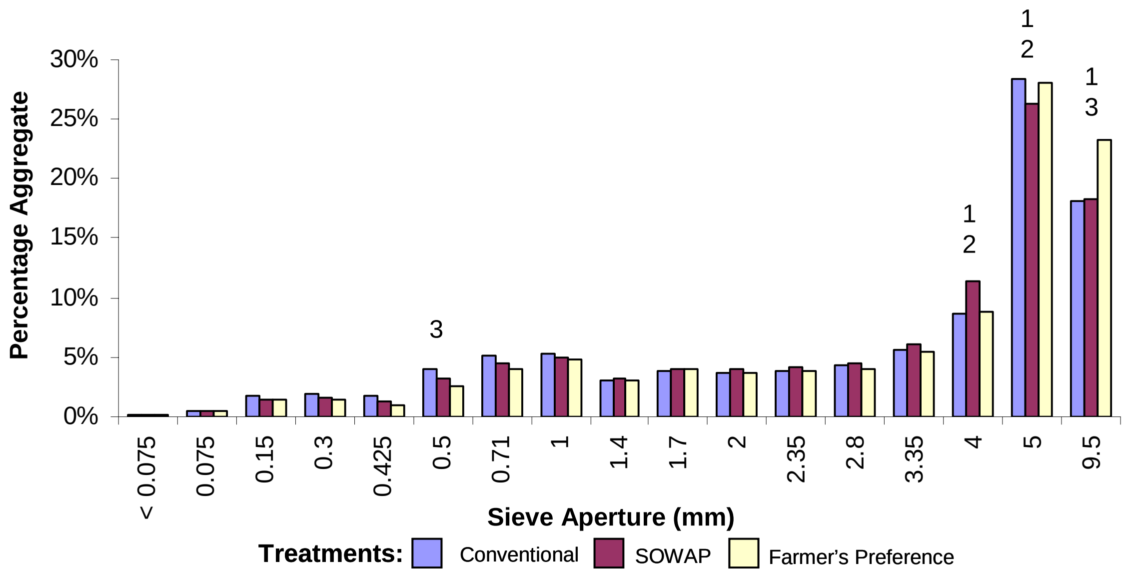


Figure 5.4-1 Loddington: aggregate size distribution for the entire sampling period - March 04 to August 05. Floating numbers denote significant differences between: 1=SOWAP and Farmer's Preference, 2=SOWAP and conventional and 3=Farmer's Preference and conventional.

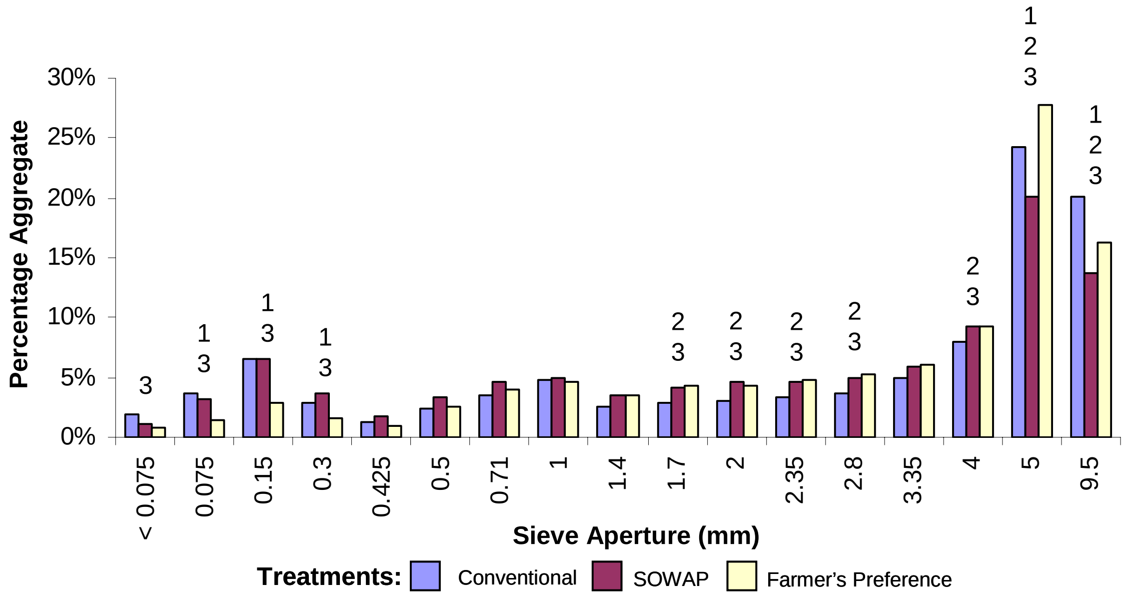


Figure 5.4-2 Tivington: aggregate size distribution for the entire sampling period - March 04 to August 05. Floating numbers denote significant differences between: 1=SOWAP and Farmer's Preference, 2=SOWAP and conventional and 3=Farmer's Preference and conventional.

5.4.1.2 Dry mean weight diameter

The dry mean weight diameter was calculated from the dry aggregate size distribution data. It was expected that the mean weight diameter of aggregates from the conventional plots would have a lower MWD compared to the two conservation treatments; SOWAP and Farmer's Preference. At Loddington over the entire sampling period, treatment differences in dry MWD were found (*p*=0.017). The only sampling date where the conventional treatment dry MWD was not significantly lower than either or both of the conservation treatments was in March 05 (under spring beans). At Tivington treatment differences were also found (*p*<0.001), but the opposite trend to Loddington was found. The dry MWD of aggregates from the conventional treatment was higher than for the SOWAP conservation treatment over the entire sampling period and at both sampling dates.

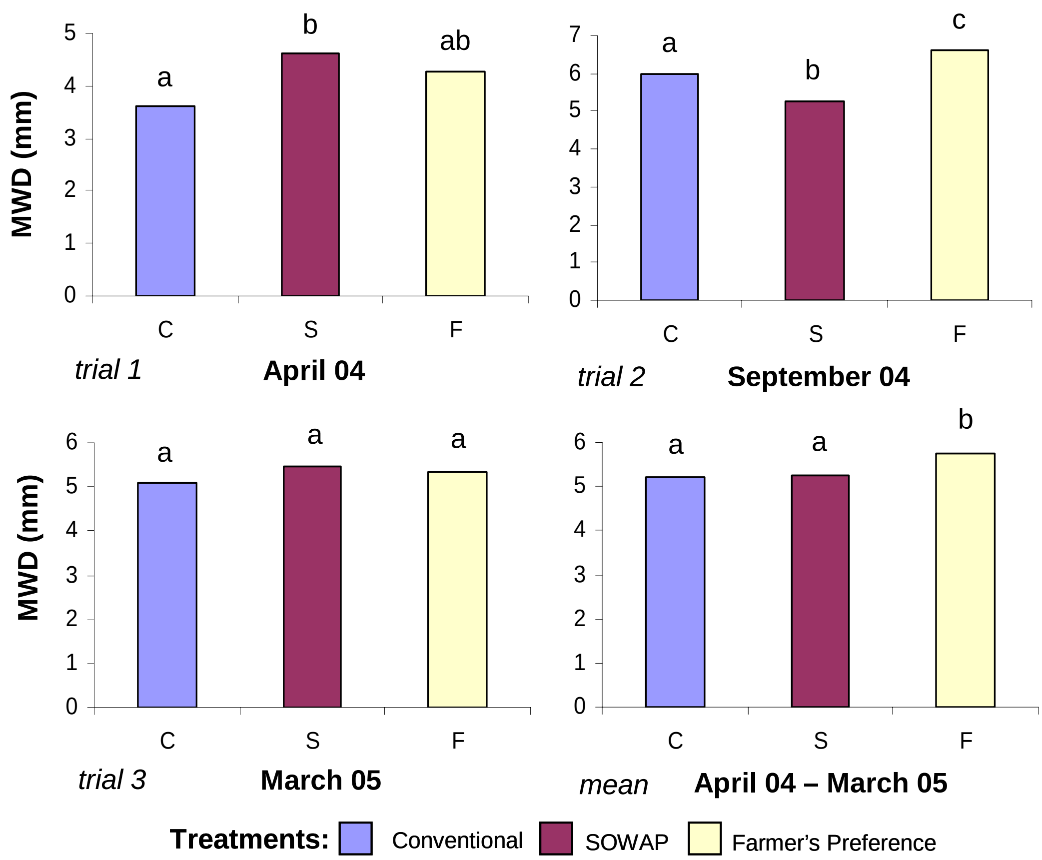


Figure 5.4-3 Loddington: Dry MWD. Letters denote significant differences.

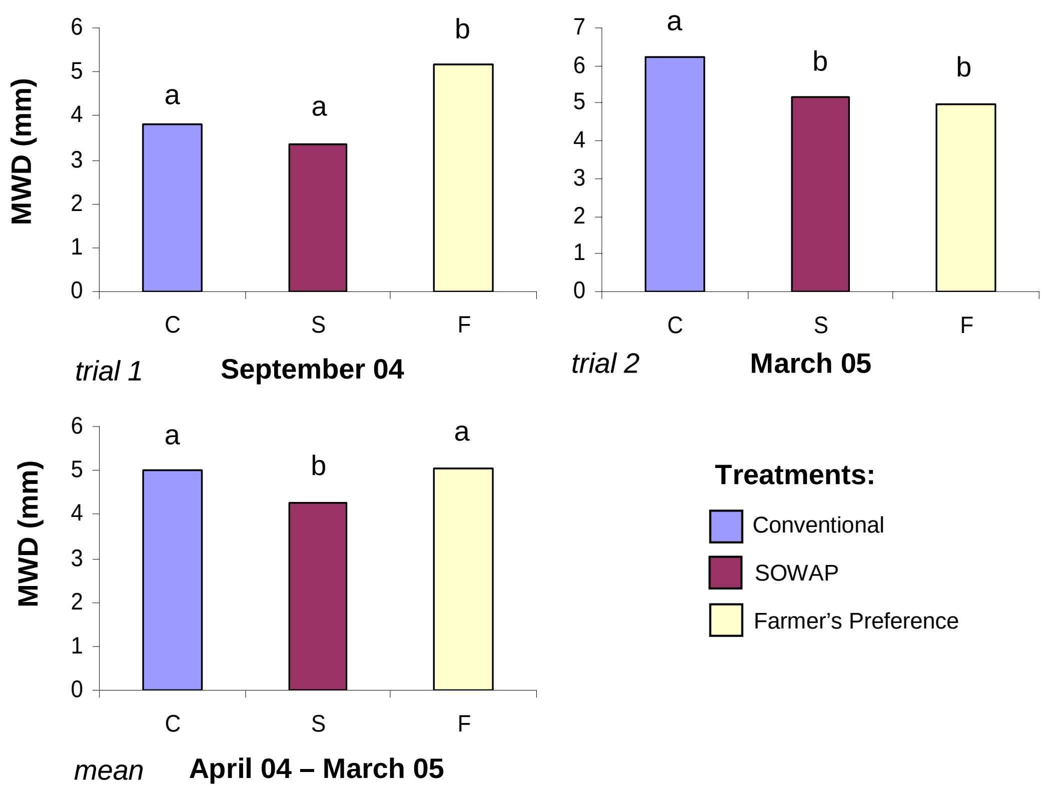


Figure 5.4-4 Tivington: Dry MWD. Letters denote significant differences.

5.4.1.3 Wet Mean Weight Diameter

The wet mean weight diameter was calculated from the aggregate stability wet sieving data. It was expected that the mean weight diameter of aggregates from the conventional plots would have a lower MWD compared to the SOWAP and Farmer's Preference treatments. This was the case at Loddington, with all three treatments having significantly different wet MWD during the September 04 sampling. The conventional treatment had the lowest mean wet MWD of all results (p<0.001). This was also the trend in April 04, although differences were not significant. Results from Tivington showed significant differences between all treatments for the soil sampled in September 04, and when considering the mean over the entire sampling period (p<0.001). The wet MWD from the conventional plots was lower than the SOWAP conservation treatment, but not the Farmer's Preference treatment.

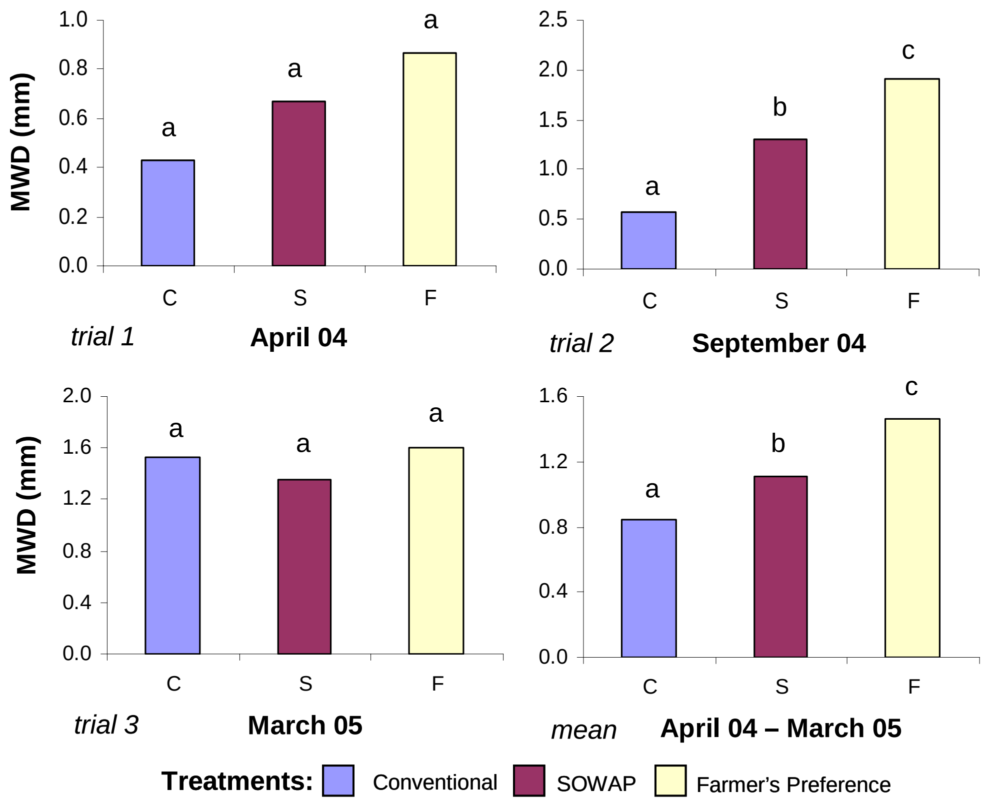


Figure 5.4-5 Loddington: wet MWD. Letters denote significant differences.

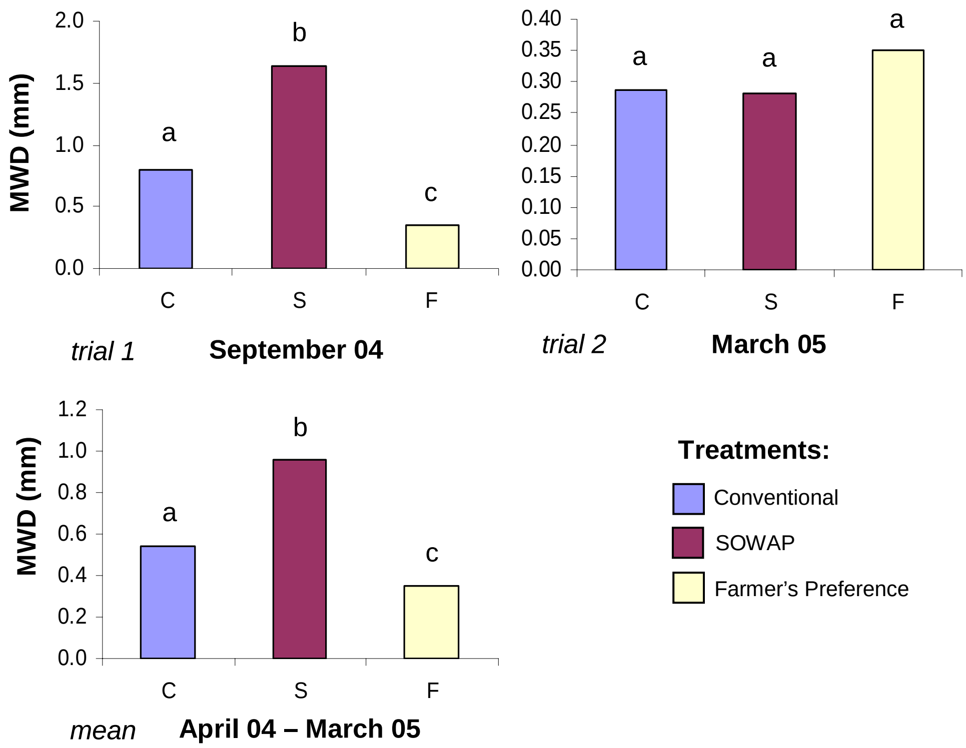


Figure 5.4-6 Tivington: wet MWD. Letters denote significant differences.

5.4.1.4 MWD method comparison

It was expected that the ranking of mean weight diameter (MWD) for all treatments would be the same, irrespective of method used (i.e. dry or wet sieving). MWD for each treatment as derived from the dry and wet methods were significantly (p<0.001) different at each site (Figure 5.4-7). The treatment ranking of MWD was similar for both methods at Loddington (Figure 5.4-8), with the highest MWD from soil derived from the Farmer's Preference treatment. However, at Tivington (Figure 5.4-8) there was no consistency between methods. The wet method of MWD showed the Farmer's Preference had the lowest MWD and the SOWAP treatment had the highest MWD. The opposite was true using the dry MWD results.

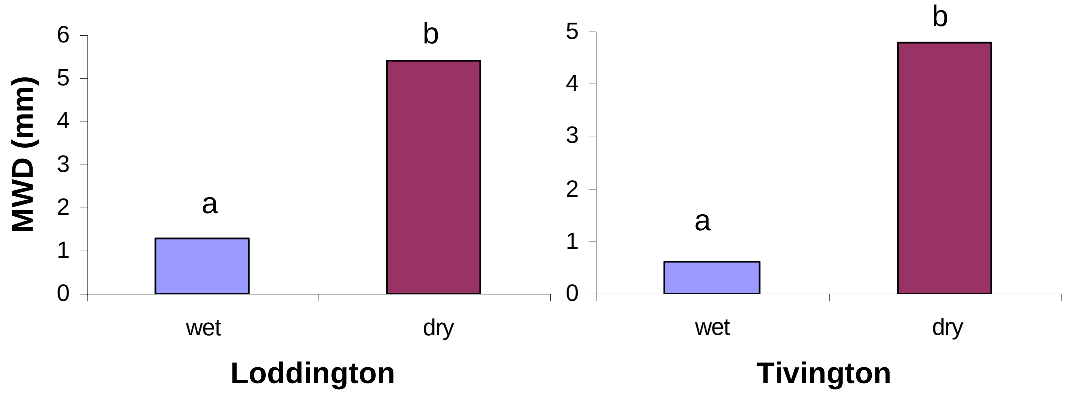


Figure 5.4-7 wet and dry method comparison of MWD over the entire sampling period

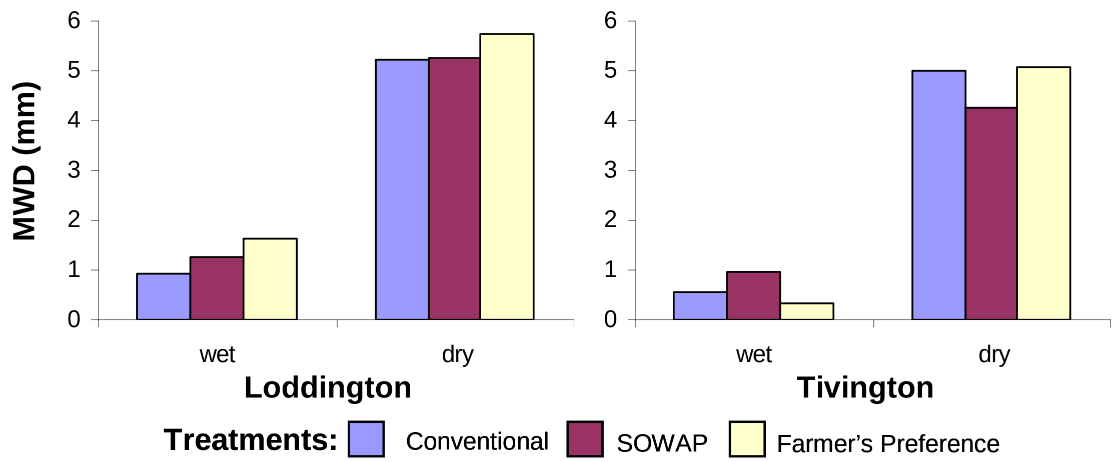


Figure 5.4-8 MWD method comparison (wet and dry) – mean relative treatment ranks, over the entire sampling period

5.4.2 Hypothesis two

It was expected that soil surface aggregates from conventionally treated plots would be more erodible (having fewer stable aggregates) in comparison to soil from conservation treated plots. This hypothesis was tested through the results from three different methods of aggregate stability testing; raindrop impact, wet sieving and a field test kit. To allow fair comparisons to be made, the size of the aggregates tested were between 3.35 and 5 mm diameter (section 4.3).

5.4.2.1 Gravity Fed Rain Tower

At Loddington (Figure 5.4-9) there were no treatment effects on the percentage of soil retained on a 0.5mm sieve, implying there are no differences in the stability of aggregates from the three different treatments. Results from April 2004 shows a trend of fewer stable aggregates being present from the conventional plots in comparison to both of the conservation treatments, which supports the stated hypothesis. At Tivington, again no statistically significant treatment differences were found (Figure 5.4-10). During March 2004 a trend was visible with the conventional treatment tended to have a higher percentage of stable aggregates compared to the conservation treatments, in particular when compared with the Farmer's Preference treatment. This is opposite to Loddington. However, it must be noted that neither trend found at either site was prominent.

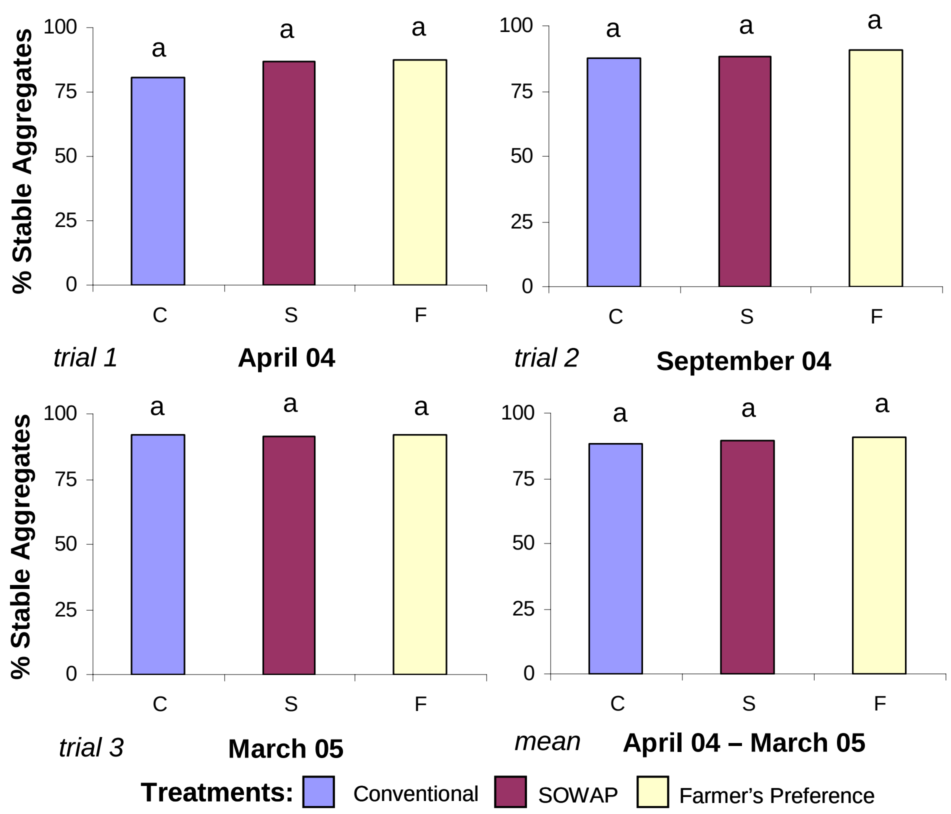


Figure 5.4-9 Loddington - percentage stable aggregates following raindrop impact

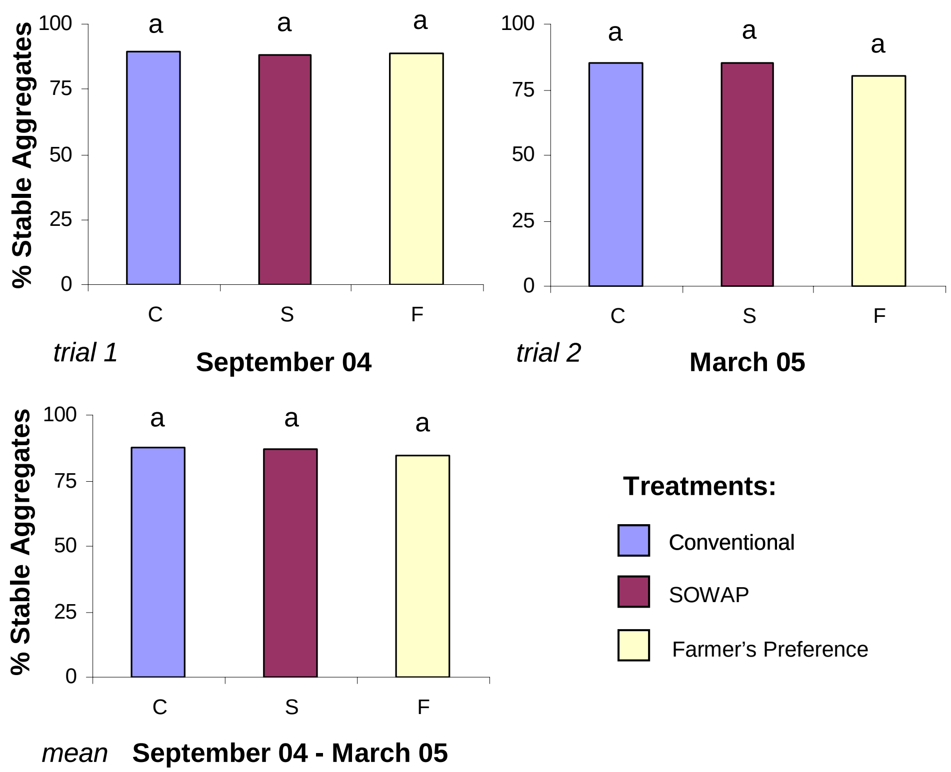


Figure 5.4-10 Tivington - percentage stable aggregates following raindrop impact

5.4.2.2 Wet Sieving

At Loddington (Figure 5.4-11) there were significant (p<0.001) treatment differences in the percentage stable aggregates when results were compared for each sampling date and the overall mean (i.e. over the entire sampling period). The lowest percentage of stable aggregates after wet sieving came from the conventionally treated plots in comparison to at least one conservation treatment. At Tivington treatment significant effects on aggregate stability were also found (p<0.001). Overall (mean over the entire sampling period) and from September 2004 sampling, aggregates from the conventional plots had fewer stable aggregates than the SOWAP treatment (Figure 5.4-12), but had more stable aggregates compared to the Farmer's Preference treatment. The results from both sites support the stated hypothesis that fewer stable aggregates would be found on the conventional plots. However, this was not always in comparison to both conservation treatments.

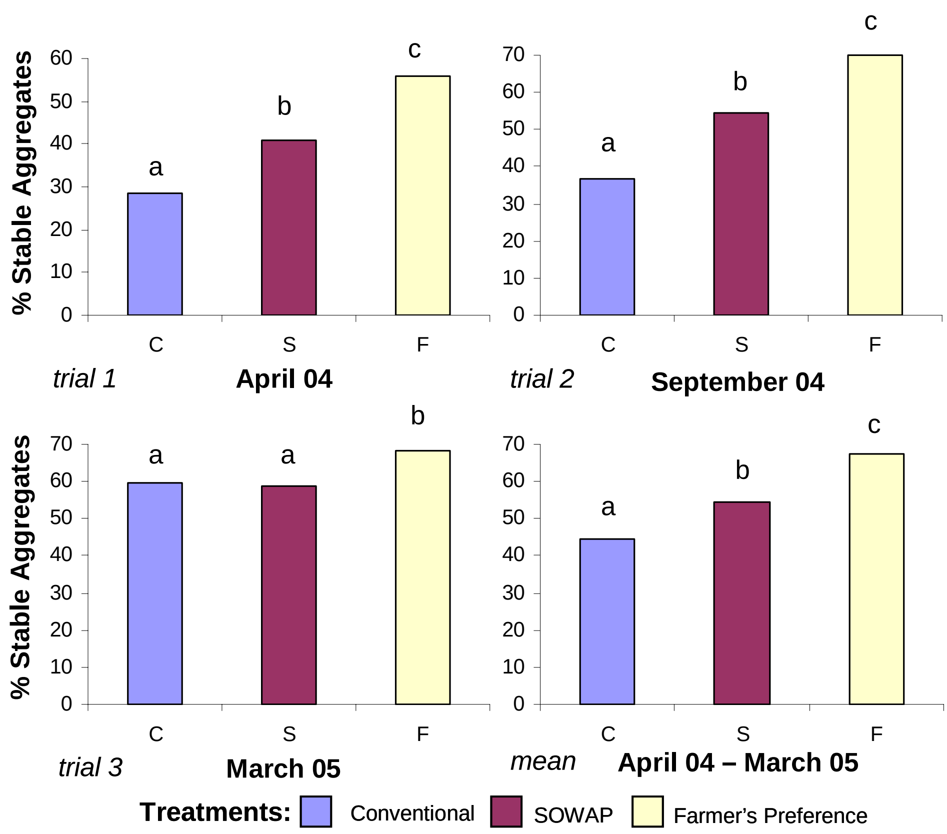


Figure 5.4-11 Loddington - percentage stable aggregates after wet sieving

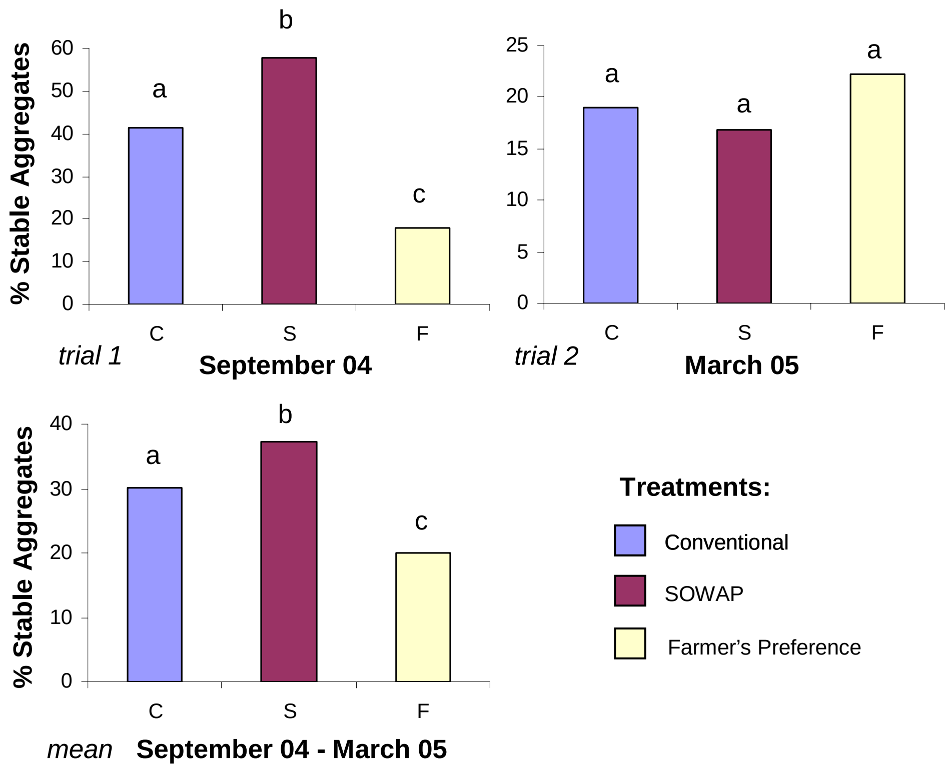


Figure 5.4-12 Tivington - percentage stable aggregates after wet sieving

5.4.2.3 Field Test Kit

At Loddington treatment effects on the percentage stable aggregates were found in April and September 04, and for the mean over the entire sampling period (p<0.001). In all these case there were fewer stable aggregates from the conventional treatment compared to both of the conservation treatments (Figure 5.4-13). At Tivington (Figure 5.4-14) treatment effects on the percentage of stable aggregates were found in September 04 (p=0.005) and for the mean over the entire sampling period (p<0.001). In both cases the percentage stable aggregates from the Farmer's Preference was significantly lower than the conventional and the SOWAP treatment. This finding does not support the hypothesis that more stable aggregates would be found on the conventional treatments when compared with the conservation treatments.

In conclusion, the results from Loddington (only) support the stated hypothesis that fewer stable aggregates will be found from the conventional treatment, compared to the conservation treatments.

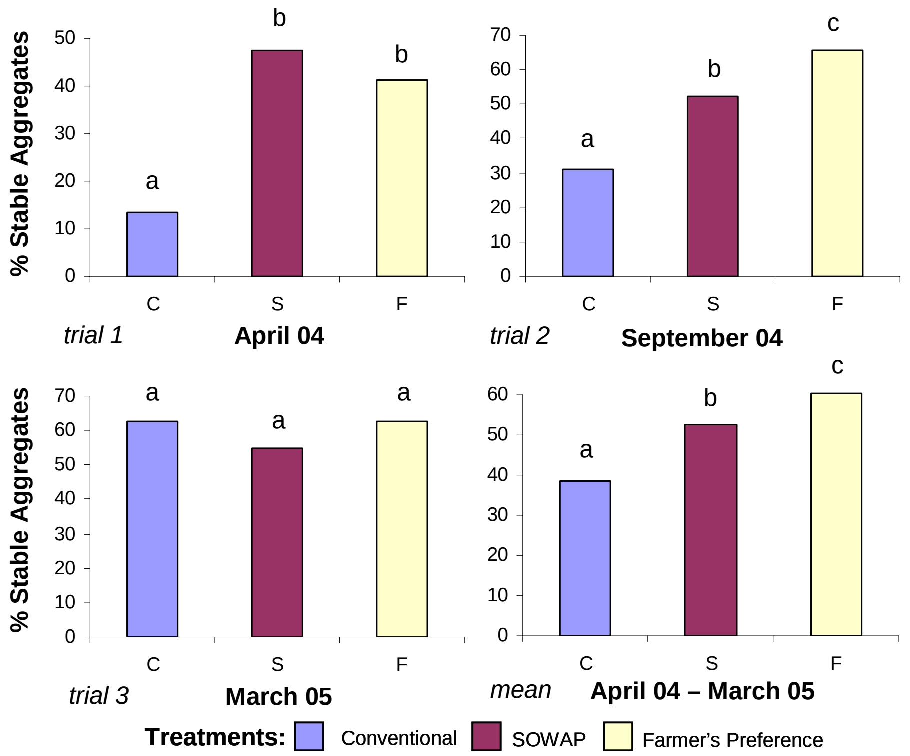


Figure 5.4-13 Loddington - percentage stable aggregates after FTK

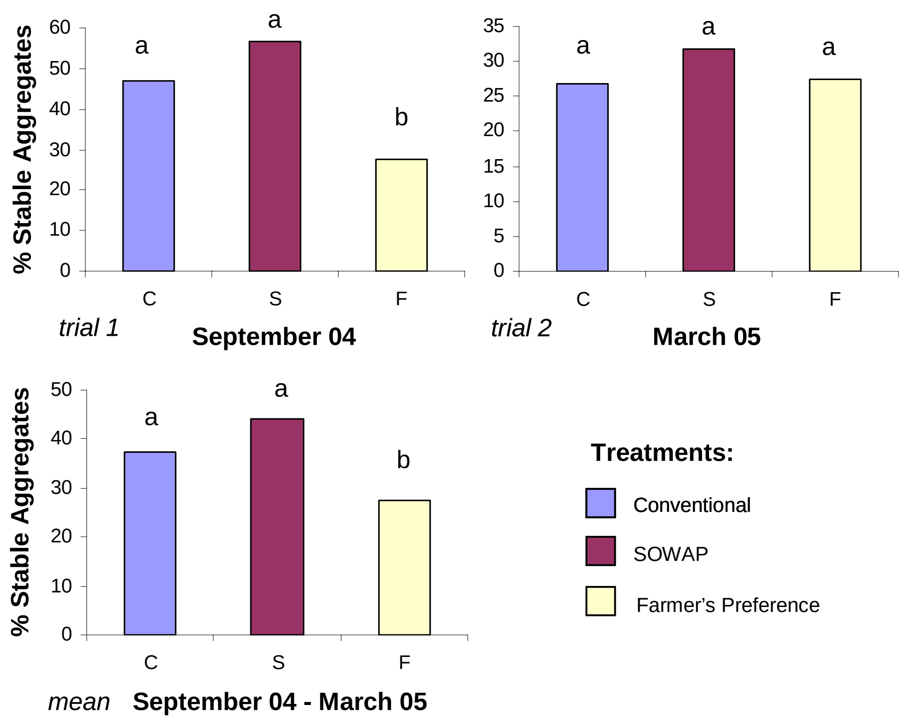


Figure 5.4-14 Tivington: - percentage stable aggregates after FTK

5.4.2.4 Overview of methods of assessing aggregate stability

At Loddington all three methods of assessing soil aggregate stability support the hypothesis that soil surface aggregates obtained from conventionally treated plots will be more erodible in comparison to aggregates from conservation treated plots. At Tivington only two out of the three methods (the two immersion based methods) support the hypothesis, that the conventional treatment has fewer stable aggregates than the conservation treatments, although this was only when compared with the SOWAP treatment. The hypothesis was not supported in terms of proportion of stable aggregates when comparing the conventional and Farmer Preference treatments.

5.4.3 Hypothesis Three

Substantial differences in the absolute results of aggregate breakdown between aggregate stability methods were expected. The immersion based methods were expected to result in lower percentages of stable aggregates compared to the rain drop impact method, due to the relative severity of disruptive forces operating. However, the relative ranking of aggregate stability between treatments was expected to be consistent.

At Loddington (Figure 5.4-15) there were large absolute differences in the percentage stable aggregates using the different methods, especially when comparing the immersion and rain drop impact methods. The two immersion methods (wet sieving and field test kit, FTK) were very similar in treatment effect on aggregate breakdown. All three methods showed the same trend in treatment ranking of aggregate stability, with the lowest percentage stable aggregates from the conventional treatment and the highest percentage stable aggregates from the Farmer's Preference treatment, as shown in Table 5.4-1.

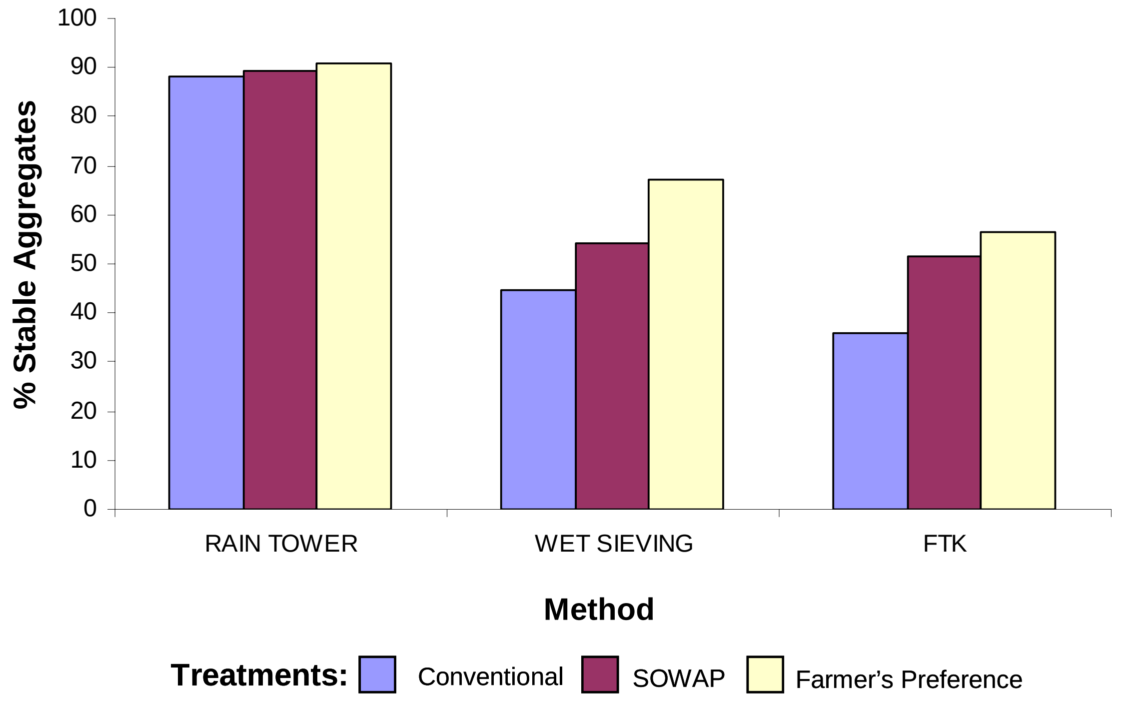


Figure 5.4-15 Loddington method comparison

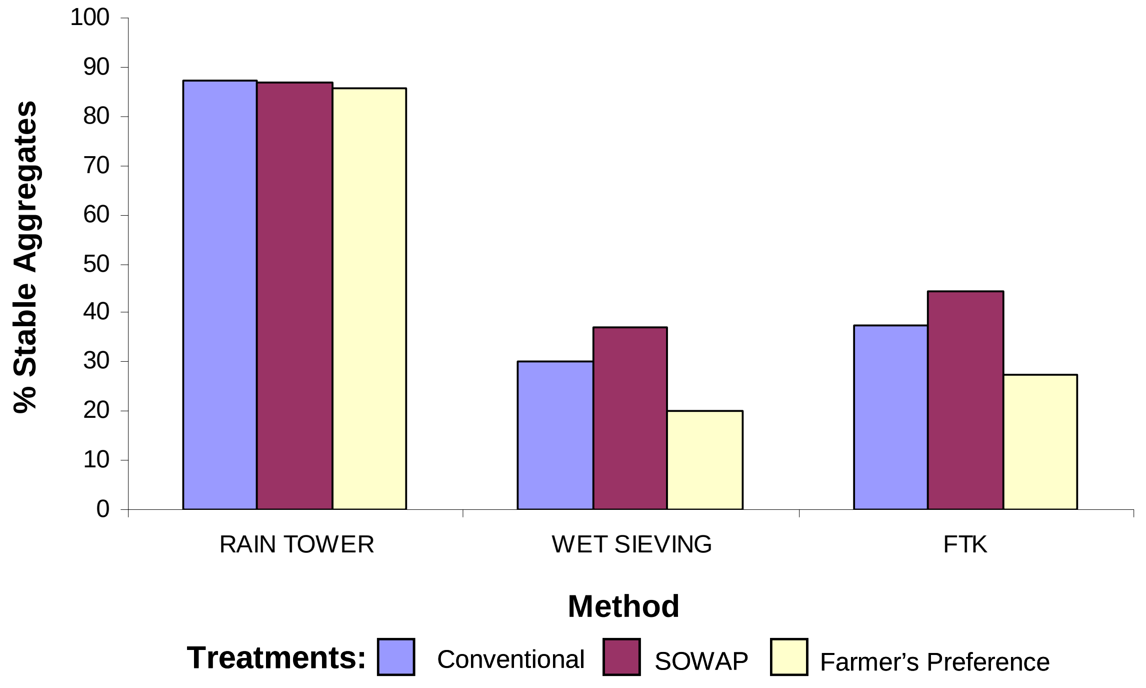


Figure 5.4-16 Tivington method comparison

At Tivington (Figure 5.4-16) there were also substantial differences in absolute results between the immersion and rain drop impact methods, with percentage stable aggregates from the rain drop impact (rain tower) method being the highest. Similar to Loddington, the two immersion based methods (wet sieving and FTK) showed the same treatment effect on aggregate stability, with the lowest percentage stable aggregates coming from the Farmer's Preference treatment and the highest percentage from the SOWAP conservation treatment. Results from the rain tower method also show a trend of the Farmer's Preference treatment having the lowest percentage of stable aggregates, but the highest percentage was from the conventional treatment, rather than the SOWAP conservation treatment as might have been expected (Table 5.4-1)

To summarise the results, the relative ranks of the different treatments can be found in Table 5.4-1.

Table 5.4-1 Treatment ranking under different methods of stability analysis (1 = highest % stable aggregates; 3 = lowest % stable aggregates)

Site	Date	Method	Ranking		
			Conventional	**SOWAP Conservation**	**Farmer's Preference**
Loddington	Mean (n = 3)	Rain Tower	3	2	1
Loddington	Mean (n =3)	Wet Sieving	3	2	1
Loddington	Mean (n=3)	Field Test Kit	3	2	1
Tivington	Mean (n=2)	Rain Tower	1	2	3
Tivington	Mean (n=2)	Wet Sieving	2	1	3
Tivington	Mean (n=2)	Field Test Kit	2	1	3

5.4.4 Additional information

Correlations were carried out between aggregate stability and MWD results with soil properties and surface characteristics to help explain observed results. The results are presented in Table 5.4-2 for Loddington and Table 5.4-3 for Tivington.

Table 5.4-2 Loddington: Correlations of aggregate stability and MWD with soil and surface properties ($p<0.05$)

Factor	Wet sieving % stable	RT % stable	FTK % stable	Wet MWD	Dry MWD
Dry bulk density	x	x	x	x	x
Volumetric MC	x	x	x	x	x
Gravimetric MC	x	x	x	x	x
Clay	0.61	x	x	0.45	x
Bare soil	x	x	x	x	x
Residue	x	x	x	x	x
Surface roughness	x	x	x	x	x
Organic matter	x	0.52	x	x	x
Organic carbon	0.51	x	0.48	x	x

x denotes non-significant correlations

Table 5.4-3 Tivington: Correlations of aggregate stability and MWD with soil and surface properties ($p<0.05$)

Factor	Wet sieving % stable	RT % stable	FTK % stable	Wet MWD	Dry MWD
Dry bulk density	x	x	x	x	x
Volumetric MC	-0.5	x	-0.64	x	x
Gravimetric MC	-0.55	x	-0.58	x	x
Clay	x	x	-0.52	x	x
Bare soil	0.55	x	x	x	x
Surface roughness	x	x	-0.52	x	x
Organic matter	x	x	x	x	x
Organic carbon	x	x	0.66	x	x

x denotes non-significant correlations

5.5 Discussion

5.5.1 Findings in relation to the original objectives

Results of MWD and stability will be related to field soil and surface properties measured at the time of sampling. These results have not been presented in the main text but are shown in the Appendix I. The soil properties and surface characteristics results presented in Appendix I are those where significant treatment differences existed. These results are from season 1-3 at Loddington and season 2 at Tivington, and therefore differ to the soil and surface properties presented in chapter 3. As such the results present in Appendix I and the erodibility measurements presented in this current chapter were carried out over the same temporal scale.

5.5.1.1 Objective one: soil management treatments affect surface soil aggregate size (and) distribution

It was predicted in hypothesis one that soil from the conventionally managed plots would have a different aggregate size distribution to those from the two conservation treatments. It was expected that the soil from the conventional treatments would have a higher percentage of smaller sized aggregates as a result of greater soil breakdown by the increased number of cultivation practices. This statement is supported by the data from Loddington. At Tivington this statement is also supported, but only when the conventional and Farmer's Preference conservation treatments are compared.

This relative increase in smaller sized aggregates would imply that the mean weight diameter (MWD) from the conventional treatment would be lower compared to the conservation treatments. In addition this lower MWD would occur under both dry and wet methods of determination.

At Loddington where significant differences existed, MWD was lower than at least one of the conservation treatments for dry MWD and both conservation treatments for wet MWD. The lower MWD for both methods relates to two factors, a) increase in mechanical manipulation of the soil surface from tillage and b) treatment induced changes on soil properties. Soil conventionally treated was subjected to increase tillage practices comprising of primary and secondary cultivation, part of which included soil inversion. The resultant effect of this would have been a mechanical breakdown of soil clods and larger soil aggregates into smaller aggregates to increase the seed / soil contact ratio. This effect of tillage was operating at both site locations. This mechanical manipulation also would have led to changes in soil and surface properties.

It was found that soil from both conservation treatments had greater contents of organic matter and clay. Both of these properties are known to increase aggregate stability by increasing cohesive strength between soil particles (Le Bissonnais et al. 2002; Levy & Mamedov 2002; Robinson & Phillips 2001). Organic matter also has the affect of decreasing aggregate wettability by the creation of a hydrophobic organic film (Ellerbrock et al. 2005). The latter is important for the wet method of MWD, by reducing aggregate wettability, the risk of slaking is reduced. Changes in these soil properties would have been the result of different tillage regimes. The conventional treatment had increased number of cultivations including soil inversion which would have buried fertile organic top soil. These changes in soil properties influence stability and therefore relate to MWD. The increase in stability of soil from one treatment over the other means the resistance to breakdown is higher. As a result of less aggregate breakdown, a relatively higher percentage of large aggregate sizes would be present. There are large differences in destructive energy being applied between the two methods (wet and dry) which raises the question of which MWD method better represents the distribution of aggregate sizes of field soil? This is discussed later in the discussion.

As expected the absolute values from the wet method of MWD were lower than the dry method. The wet method is more destructive on soil aggregates, causing more breakdown. The treatment ranking found for both wet and dry methods were the same when the means over the entire sampling period were compared.

At Tivington the results from the wet method of MWD were again expectedly lower compared to the dry method. However, what was unexpected were very different treatment ranking of stability between the dry and wet methods. Results from the dry method showed that the mean (over the entire sampling period)

MWD was significantly less from the SOWAP treatment compared to the two remaining treatments (C and F). This results was opposite from the wet method, where MWD was significantly greater from the SOWAP treatment. These conflicting results can be explained by soil properties. The clay content of the three treatments reflects the pattern seen under the dry method, and the organic matter content follows a similar pattern to the wet method. Both organic matter and clay content are associated with an increase in aggregate stability. It would appear that clay content had a greater effect on aggregate stability under dry conditions compared to organic matter under wet. Organic matter reduces the risk of slaking by decreasing wettability rate by the presence of a hydrophobic film.

The results from Tivington highlight that different processes aggregate breakdown are occurring, that of abrasion under the dry MWD method and slaking during the wet method. This raises the question as to which method (wet or dry) better represents the MWD of field soil. The answer to this is dependent upon what is being investigated; before or after erosion. The dry method better represents the MWD of dry soil conditions which have been subjected to light rainfall or wind erosion. The wet method of MWD simulates breakdown of aggregates after a heavy storm leading to saturated or flooded conditions causing maximum amounts of erosion. It is believed that they dry method of MWD better represents the affect of cultivation as best soil management encourages farmer's to cultivate during dry periods, when the ground is hard.

5.5.1.2 Objective two: soil management treatment affects surface aggregate stability

Hypothesis two proposed that surface soil aggregates from conventionally treated plots would be more erodible, with proportionately fewer stable aggregates, in comparison to conservation treated plots. This statement was supported by the results from Loddington when aggregates were tested using the immersion based methods, but not for the raindrop impact method. This method resulted in no statistically significant treatment differences. At Tivington, the hypothesis could be supported when overall treatment means generated from the immersion based methods were compared, but only when the conventional treatment was compared with the SOWAP conservation treatment (and not the Farmer's Preference treatment). The raindrop impact method showed no significant treatment differences.

The differences in stability between treatments could be explained by measured soil and surface properties. The conventional treatment was shown to be the least stable at Loddington. This was supported by significantly lower soil contents of organic carbon, matter and clay (p<0.001). All of these soil properties have been linked to aggregate stability and wettability, as mentioned in hypothesis one.

The treatment ranks of aggregate stability at Tivington differed to those at Loddington. At Tivington over the entire sampling period the Farmer's Preference treatment had the fewest stable aggregates, above this was the conventional treatment and the most stable aggregates was found in soil from the SOWAP treatment. Relating soil properties to these results was less clear cut. Significant differences were found in soil content of organic carbon, organic matter, silt and clay. However, the differences in aggregate stability between treatments were not reflected in the difference from any one soil properties but a combination of all of them. A larger percentage of stable aggregates from the SOWAP treatment in comparison to the conventional treatment could be explained by increases in particle cohesion brought about by greater soil contents of organic carbon. However, surprisingly soil from the SOWAP treatment also had significantly lower clay and organic matter contents to the conventional treatment, both of which would indicate a lower stability but this was not the case. This also highlights that organic carbon content is not always directly linked to organic matter content. The percentage of stable aggregates was significantly higher from the conventional treatment compared to the other conservation treatment – Farmer's Preference. This relationship can be supported by higher organic matter contents and lower silt content; both implying increased stability through particle cohesion.

The results from both sites indicate that the relationship between soil properties and aggregate stability cannot be generalised. Seasonal fluctuations in these soil properties were also investigate but did not clarify the found treatment difference in aggregate stability. This implies other factors and processes are operating. The presence of an active microbial community has been shown to increase soil organic matter content, soil aggregation and aggregate stability. Investigations were carried out on soil from both site locations in research carried out by Allton (2006). It was found that the microbial community size was significantly larger on the conservation treatment compared to the conventional treatment. However, this was only found at Loddington and not Tivington. What is not known is the changes in the micro-biological community from when the samples were removed from the field, through to when the samples were air dried and then during storage before sample were tested for stability. Although soil from both sites underwent the same processes, the inherently different soil properties of each site (e.g. organic matter content), may have resulted in a different microbial response.

5.5.1.3 Objective three: implications of different methods to measure aggregate stability

It was expected that there would be substantial differences in the absolute results of percentage stable aggregates between the test methods used, but that the relative treatment rankings would be consistent. This was the case at Loddington where percentage stable aggregates were much higher under the raindrop impact (rain tower) method compared to the two immersion based methods. Relative treatment ranking was the same for all three methods, although the treatment differences were not significant under the rain tower. At Tivington there were also large absolute differences between the percentage stable aggregates measured using the immersion and the rain drop impact methods. The relative treatment ranks were the same for the two immersion based methods, but these ranks were different to those from the raindrop impact test (although treatment differences were not significant under the rain tower). In summary, the statement that relative treatment ranks are consistent, irrespective of test method cannot be supported by the results from Tivington, but the Loddington results do reflect the expected relationships.

The substantial difference between immersion and rain drop impact methods of testing aggregate stability was expected due to the very different forces being applied to the aggregates. The rain drop impact method simulates more closely natural rainfall including processes of detachment and splash erosion. Splash erosion is highly destructive compared to other erosion processes such as overland flow (Morgan 2005). However, the processes of fully immersing aggregates in water is more destructive due to the explosive affects of fast wetting as water is forced into air filled pores (Lyles et al. 1974; Emerson 1954). The rate at which this occurs is affected by soil properties, as shown previously in the discussion. Clay and organic matter content will influence the cohesive strength between soil particles and the wetting rate of soil aggregates; although the affect of these properties will be common for both methods.

The relationship between tillage treatment and aggregate stability varies between the methods used. This raises the important question of which method is the most appropriate to assess treatment differences in terms of aggregate stability. The field test kit (FTK) and the wet sieving methods produced similar results. This is not surprising as the FTK method is a simplified version of wet sieving; and both are immersion based methods. Therefore the consideration to make is between immersion and rain impact methods. This decision should be made in relation to the hydrological processes operating in the field.

Immersion based methods simulate flooding conditions. Therefore, results gained represent an extreme rainfall event, when the soil is totally immersed in water and saturated. This technique however does not simulate the impact of rainfall on the soil surface. The other disadvantage of using immersion methods is that each aggregate is tested individually, and as such are spatially isolated from other aggregates. Again, this does not represent field conditions and processes (Plate 5.5-1). Individual aggregates are also used during the rain impact method (rain tower), but when the saturated aggregates are rained on they break down and form a surface seal (Plate 5.5-2). This is representative of processes that actually occur during rainfall on an *in situ* field soil.

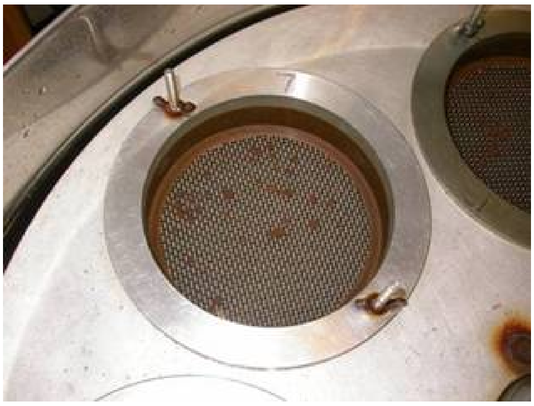


Plate 5.5-1 after wet sieving

Plate 5.5-2 after rain fall under the rain tower

This study showed that the results generated between the two immersion based methods (wet sieving and field test kit) were very similar. Therefore, the field test kit could be a useful indicator of erodibility, which is quick and easy, and can be used in the field. This has important future implications which are discussed later on.

5.5.2 Discussion beyond the set objectives

As part of the design process of the methodology, the relationship between aggregate size and stability was raised. Work by Elliot (1986), Kay (1990) and Bearden & Petersen (2000) show changes in soil properties (organic carbon, porosity and soil biota including plant roots, respectively) in relation to aggregate size. This would therefore have implications on aggregate stability. Six et al. (2000) highlights the fact that measuring aggregate stability on a specific aggregate size does not represent the soil as a whole. This was an important point to consider. Initially in the method design the use of a whole soil could not be used as this investigation was specifically compared differences between treatments. A soil sample representing the "whole soil" could not be defined and therefore could not be replicated. Therefore, an aggregate range was chosen based on work by others. As stated in the section 5.3, a test was carried out to compare aggregates size to stability, to identify if any relationship did exist. The results (Appendix J) of this test contradicted the general theory by showing that there was no significant difference between aggregate size and stability. However, it should be noted that this test was carried out under rain drop impact and not the commonly used wet sieving method. This again links to previous discussions on the use of different methods of stability testing in relation to the set out objectives. This test was carried out using the rain drop impact method because this better simulates natural rainfall to gain a better understanding of the erosion processes occurring at the field scale, which is investigated in chapter 3 and 4.

It could still be argued that although this additional work looked at the relationship of aggregates size and stability this still does not represent a whole soil (Six et al. 2000). This point raised by Six et al. (2000) was tested in this current research through a pilot run to try and address this question. Soil samples obtained in September 2004 at both sites were used in this pilot run, representing the first sampling data common to both sites. The method of wet sieving (as described previously in section 5.3.3.3.1, was used as is commonly used in aggregate stability research and is a relatively quick (in comparison to the gravity fed rain tower) test. Aggregates sized 3.35-5mm and a whole soil sample were compared. The whole soil sample was taken from the pre-air dried samples as used in section 5.3.3, before sieving for classes was done. The whole soil and 3.35-5mm samples were tested using an experimental random design incorporating soil from all three soil management treatments. A summary of the results can be found in the Appendix J. It was found that at Loddington that there was no statistically significant difference between the percentage of stable aggregates retained from aggregates sized 3.35-5mm or a whole soil sample. This was case when the two aggregate classes were compared under all four statistical analyses, define in Table 5.5-1. At Tivington, there significant differences between aggregate sizes were found when compared under analysis a) and d) as shown in Table 5.5-1. This was due to a significantly higher percentage of stable aggregate retained on the 2mm aperture sieve from whole soil sample from the SOWAP treatment.

These exceptionally high results (in comparison to the other treatments and aperture size) can be related to the significantly higher organic carbon content on the SOWAP treatment. Six et al. (2004) observed that as aggregate size increases the concentration of carbon rose. This implies that organic carbon is more important in stability in the whole sample which contains aggregates larger than 5mm i.e. larger that than found in aggregates sized 3.35-5mm. It should also be noted that this work was carried out on soil taken from only one sampling date and does not take into account temporal variation.

Table 5.5-1 Statistical analysis of difference in percentage stable aggregates from aggregates sized 3.35-5mm and a whole soil sample.

a) For differences between aggregate classes of percentage stable aggregates >0.5mm, irrespective of treatment difference i.e. when the mean was taken from the results of all treatments;
b) For differences between aggregate classes of percentage stable aggregates >0.5mm irrespective of treatment at different sieve aperture sizes i.e. the mean result from all treatments at each sieve aperture; 0.5, 1.0 and 2.0mm.
c) For differences between aggregate classes of percentage stable aggregates >0.5mm in relation to treatment effects.
d) For differences between aggregate classes of percentage stable aggregates in relation to treatment effects at different sieve aperture sizes i.e. the mean result for each treatment at each sieve aperture; 0.5, 1.0 and 2.0mm.

This work highlights that initial aggregate size (i.e. whole soil versus 3.35-5mm sub-sample) could have an effect on the stability results obtained, but that this was only the case when dealing with soil with high organic carbon content. This study does reinforce the need to run pilot studies on test soil to identify if stability will be affected by aggregate size.

5.5.3 Implication of this study

The implication of this study is that the adoption of conservation tillage can reduce soil erodibility and therefore has potential to reduce the risk of water erosion. The research also highlighted the influence of different aggregate stability methods on the results obtained, in terms of relative treatment ranks and absolute differences. The use of either immersion or rain drop impact based methods should be chosen in relation to the hydrological processes being investigated, i.e. a flooding events or rainfall.

This research has also highlighted that established theories on the relationship between aggregate size and stability do not always apply. It is important not to assume that this theory applies to all soils. It is vital that before tests of aggregate stability are undertaken, pilot investigations should be carried out to identify what relationships exist.

A field test kit can be used as a quick and cheap alternative to wet sieving. Both methods are based on immersion of aggregates in water to assess stability. The field test kit yields very similar results to those of the wet sieving method in terms of relative treatment differences. The former method is easily accessible to all and has potential to be used in important on-site assessment of soil stability. One example of this is greater quantification of erosion risk as part of a Soil Management Plan required in environmental stewardship schemes.

5.5.4 Future research recommendations

It was found that the rain drop impact method generated no treatment differences unlike immersion based methods. Further research is needed to understand the lack of significant results. It is believed that this is due to the saturated starting conditions of the test aggregates. This was done to allow comparison with the immersion based methods. However, in the field, aggregates are not in a permanent state of saturation. Therefore it would be advantageous to carry out further experiments using a variety of starting conditions of test aggregate in conjunction with the gravity fed rain tower.

It would be advantageous to increase the number of samples used in the aggregate size and stability methodology to include more seasons. This would gain a better understanding of the influence of cropping regimes and timings have on soil properties and therefore on aggregate stability. This work should be expanded upon to include further tests on the relationship between aggregate size and stability in relation to tillage induced changes in soil properties.

Previous work by Kemper & Rosemau (1986) looked at the effect of sampling (equipment used, transportation and storage) in relation to aggregate stability. However, more research is required into the effect of sampling procedure on stability results in relation to different tillage treatments. For example, a treatment which promotes fungal growth would be expected to increase aggregate stability, but when samples are removed from the ground, stored and treated this benefit might be lost, as the biological connectors are broken down by handling. Changes in moisture content are very important to soil erodibility (Morgan 2005). Different tillage treatments can lead to changes in soil moisture, for example by the application of surface residues. These are beneficial for preventing drought conditions during summer months. However, the moisture content will influence the amount of damage cause by the action of removing the soil sample from the ground. In addition further work is required to investigate the relationship between soil properties (including soil biology) at the time of sampling and the changes over time during storage until the samples are tested for stability.

5.6 Conclusion

It was discovered that the adoption of conservation tillage instead of conventional cultivation affected the mean weight diameter (MWD) and aggregate stability. The effect of conservation tillage on MWD and aggregate stability was found to fluctuate between site locations due to differences in inherent soil properties. However overall findings from both sites can be made and are stated in the following. Soils which had been cultivated under conventional tillage had lower MWDs caused from the action of primary, secondary and inversion tillage. Conventional tillage also led to changes in soil properties, specifically a reduction (in the majority of cases) in levels of including organic matter, carbon and clay content. *A priori* reasoning would suggest that this would have led to increases in erodibility. This was found to be true as measured by the percentage of stable aggregates after stability tests were performed. These results were only found when immersion based methods were employed. The use of rain drop impact methods resulted in no significant differences in stability between treatments. It was concluded that the two types of aggregate stability method represent different hydrological conditions – immersion based methods characterise soils under flooded conditions or under heavy rainfall, while the gravity fed rain tower simulated more closely the impact from natural rainfall. The latter of which is important for the comparison to field base processes at the two field sites used in this study.

6 Spatial Integration

6.1 Introduction and Background

The challenge of comparing erosion rates and erosion risk at different spatial scales of investigation is a high priority in erosion research today. Previous work has shown that comparisons between scales may be possible, but this is rarely attempted, and caution should always be exercised. The majority of spatial scale comparison research into soil erosion has been based on field scale and above, and tends to be model based, utilising data collected from field scale research and inputting this into regional, national and in some cases, cross-national scales.

Extrapolations are often applied in erosion modelling (Smith & Quinton 2000), despite the known errors and variability associated with data from different spatial scales. Some researchers have attempted to extrapolate results from one scale to another with mixed success. For example, work by Amore et al. (2004) investigated two erosion models; first an empirical model, the Universal Soil Loss Equation (USLE) and the second a physically based model from the Water Erosion Prediction Project (WEPP). This work showed that the models were not sensitive to the spatial resolution under investigation, however, it should be noted that the work only modelled areas from field size and above. Another example of spatial scale comparison comes from an investigation by Cerdan et al. (2004) into the extrapolation of runoff coefficients between spatial scales. This work utilised another model, the STREAM model, which looked into infiltration and overland flow parameters at the plot scale. This work also looked at runoff coefficients across three different scales; field plot of $500m^2$ and two catchment areas of 90 and 1100 ha. This investigation concluded that runoff coefficients decreased as the area increased. It was shown that catchment scale results cannot be used to represent the sum of the individual fields found within that area. This was attributable to a decline in connectivity, which was associated with a decrease of arable land within the increased of study area i.e. as the study area increased the percentage of arable land within that area declined therefore the connectivity between arable fields reduced.

This confirms that there are no rules as to the scaling up or down of erosion data. It should be noted that models are rarely employed below a field scale, so they may be better equipped at coping with field scale erosion processes rather than small scale ones. This present study hopes to add to this research field by directly comparing actual losses per unit area of soil and runoff, and runoff coefficients between field and micro- plot scales. As stated previously, the majority of research that has studied erosion at different spatial scales has in the main been model based, using data from plots at the field scale and above. Direct extrapolation of results from one scale to another is fraught with error and uncertainty. When drawing comparisons between field erosion plots and micro-erosion plots, there are important issues to be considered. These are outlined below.

a) Differences in erosion processes operating at different spatial scales

There are underlying differences in the erosion processes which are taking place at different spatial scales (Imeson & Lavee 1998; Rickson 2006). These differences in erosion processes are summarised in Table 6.1-1.

Raindrop impact is expected to be the dominant erosion process at the micro-plot scale, while overland flow and rill erosion will operate at the field scale. Even early gully formation is possible. The amount of erosion is related to the efficiency of these erosion processes, which has been succinctly presented by Morgan (2005) and is summarised here in Table 6.1-2.

Despite splash erosion being more efficient at soil detachment, the distance sediment is transported is very limited, due to the length of splash trajectory and hence it transports the least amount of sediment. At the field plot scale overland flow is more dominant, and sheet flow is more likely compared with small plots, as well as formation of rills. From this it could be assumed that erosion rates would be greater on field plots compared to micro-plots, as more efficient processes of sediment transport (channelled flow) are more likely to take place. However, deposition fans (Plate 6.1-1) are likely to be present at the field scale because of changes in slope gradient and the greater incidence of surface irregularities caused by stones, crop residues, and surface micro-topography. Such variations are proportionately less on the 1m^2 micro-erosion plots At this plot size, opportunities for deposition are less, and eroded soil will flow straight into the collection system. The difference in deposition between spatial scales indicates that sediment production would be greater from the micro-erosion plots.

Table 6.1-1 Scale scales of soil erosion research (modified from Rickson 2006)

Area	Scale*	Dominant processes operating	Research Technique	Reference (examples)
mm^2 - cm^2	Nano	Rain splash dominant; overland flow / deposition limited. No gullies, stream bank erosion or mass movements.	Aggregate stability, splash cups and laboratory trays	Ellison 1944; Morgan et al.1998; Yoder 1936
m^2	Micro	Rain splash and overland flow; some deposition. Some to no gullying or mass movements. No stream bank erosion	Runoff rig and Field plot	Kamalu 1993; Wischmeier & Smith 1978
ha - km^2	Meso	Rain splash, overland flow and deposition. Gullying and mass movements possible. Some to no stream bank erosion.	Field and Sub-catchment	Evans & Boardman 1994; Hudson, 1981
km^2	Macro	Rain splash, overland flow and deposition. Some gullying and mass movement possible. Stream bank erosion.	Catchment / landscape	Dickinson & Collins, 1998

***dimension descriptors as described by Wickenkamp et al. (2000)**

Table 6.1-2 Efficiency of forms of water erosion (summarised from Morgan 2005)

Form	Mass*	Typical velocity (m s^{-1})	Kinetic energy	Energy for erosion	Observed sediment transport (g cm^{-1})
Raindrops	R	6.0	18R	0.036R	20
Overland flow	0.5R	0.01	2.5 x 10-5R	7.5 x 10-7R	400
Rill flow	0.5R	4	4R	0.12R	19,000

***Assumes rainfall mass of *R* of which 50% contributes to runoff.**

Plate 6.1-1 Deposition fan in Somerset. Source www.sowap.org

In summary, the size of the plot under investigation will influence the erosion process operating, and hence the amount of soil loss taking place.

b) Factors affecting erosion vary at different spatial scales

At the smallest of spatial scales (nano), emphasis is placed on understanding the mechanics involved in soil erosion. Early work by Ellison (1944) showed the relationship between raindrop impact and soil erodibility by developing the use of small, 7 cm diameter splash cups, which isolated rainsplash effects on soil detachment and transport. Here, soil properties alone determined the amount of erosion measured, independent of any other of the factors affecting erosion. The effects of slope length for example on erosion will vary at different spatial scales. Micro-erosion plots (in this study 1.5m^2) will have a shorter slope length to generate surface flow. Hence as the spatial scale increases, the erosion process of overland flow will begin to dominate (Rickson 2006). There will also be changes in the type of erosion feature between spatial scales as previously shown in Table 6.1-1

c) Methods of generating soil loss vary at different spatial scales

Comparison between micro-erosion plots and field erosion plots is also problematic due to the different methods employed to generate soil loss and runoff at the different scales. Rainfall simulators are commonly used for small scale erosion assessment. This can be laboratory based using artificially packed soil with known physical properties or field samples removed and used under laboratory conditions. In either case soil is subjected to artificial rainfall of known intensity and duration. Rainfall simulators are also frequently used on micro-erosion plots set up within the field, therefore testing natural field conditions and processes. Micro-erosion plots (cm^2-m^2) may also be used utilising natural rainfall but this is less common due to the unpredictability of natural rainfall events. The next stage up from micro-erosion plots is the use of field erosion plots (<ha, Stoosnijder 2005), which in the main rely on natural rainfall events. All of these techniques allow direct measurement of erosion via the collection of eroded sediment and runoff. However, there has been little work comparing the results from these different spatial scales.

Rainfall simulators in the majority of cases apply a more intensive storm compared to natural rainfall events. This is because of the difficulty in simulating realistic drop size distributions, kinetic energy and rainfall intensity simultaneously, especially at low intensities. This is compounded by the challenge of simulating uniform rainfall distributions in space and time. The consequence of the higher simulated intensity is an increase in soil bulk density and sealing due to rainfall impact on the soil surface, which results in a decrease in infiltration, causing higher erosion rates and runoff losses (Bedaiwy & Rolston 1993). This is supported by previous research, which has shown proportionately greater soil losses from the smaller plots in comparison to field plots (Evans 1993; Boardman & Favis-Martlock 1993). Work by Andraski et al. (1985), carried out investigations of soil erosion using different tillage methods (1 plough and 3 conservation based) using field erosion plots. They compared the response of each treatment to natural and simulated rainfall. The smallest artificial storm was 72±2 mm hr^{-1} and the soil used was a silt loam, part of the Griswold series. It should be noted that only 1 erosion plot replicate per treatment was used, and as highlighted by various authors (Morgan 2005) at least 2 replicates need to be used to quantify variation. However, despite these experimental limitations, it was concluded that the treatment ranking was similar for both the artificial and simulated rainfall.

Given these limitations, it is not surprising that few studies have compared erosion rates at a number of spatial scales. However, the present study will attempt to integrate the results of soil loss from 3 spatial scales – involving soil aggregates (mm^2), micro-erosion plots (<$2m^2$) and field erosion plots (>$500m^2$). Direct comparison of absolute results between these scales is not possible, as the tests on aggregates measure the <u>erodibility</u> of soil rather than actual soil loss. Despite this, Barthes et al (2000), successfully correlated aggregate stability with runoff from micro-erosion plots. Although this approach is promising, the present study will compare the relative rankings of 3 tillage treatments in terms of soil erosion/ aggregate breakdown at the different spatial scales as a basis for comparison.

If the rankings remain consistent for the 3 spatial scales, it could be inferred that it might not always be necessary to quantify erosion losses - comparison of relative treatment ranks may be sufficient. For example, the application of this current study could be used to help farmers decide which soil management treatment is <u>relatively</u> the most effective at reducing soil loss and runoff on their farms. With many farmers joining government run schemes designed to manage their farms more environmentally e.g. the Entry Level Stewardship, they have to show that the soil management schemes adopted are working. Cheaper, more replicable small scale assessments of erosion through aggregate stability or rainfall simulations could be used to indicate (but not predict in absolute terms) field scale erosion response through treatment ranks rather than quantification of actual losses.

6.2 Objectives and Hypotheses

6.2.1 Objectives

The first objective of this chapter is to assess whether direct extrapolations can be made from the micro-plot scale to the field scale in terms of a) actual losses of soil and water; and b) runoff coefficients (i.e. the ratio of rainfall received to that which runs off as overland flow).

Second, the study will evaluate whether relative treatment effectiveness remains consistent at each spatial scale of investigation, in terms of erosion / erodibility.

6.2.2 Hypotheses

6.2.2.1 Hypothesis One

It is expected that extrapolations from the micro-plot scale to the field scale of a) actual soil and water losses per unit area, and b) runoff coefficients are not reliable, due to expected differences in erosion processes occurring at each scale.

The micro-erosion plots used in this study were 1.5m in length and 1m wide compared to the erosion plots which were approximately 0.05ha. As stated previously the large difference between plot sizes means that different erosion processes will be occurring. It is expected that losses of runoff and soil (per unit area) and calculated runoff coefficients will be greater from the micro-erosion plot in comparison to the field erosion plots and relates to the following.

a) Erosion processes will differ between spatial scales, with splash erosion dominating at the micro-plot scale and overland flow and rill formation at the field scale. It would therefore be expected that soil and water losses would be greater (per unit area) from the field erosion plots, however, deposition fans are more likely to form at the field scale due to fluctuations in slope gradient and micro-topography. In addition to deposition fans, sediment and runoff reaching the collection system as the base of the field erosion plots will be lower than the micro-erosion plots due to contour tillage in front of the collection funnels. Contouring will not be present on the micro-erosion plots.

b) There will be differences between scales in the type of rainfall (intensity, duration, drop size and kinetic energy). The rainfall intensity used at the micro-plot scale (35 mm h^{-1}) simulates a 1-6 year storm (NERC 1975). It is predicted that this will be a higher intensity than that generated naturally on the field erosion plots in the lifetime of the research project. This will result in the rainfall on the micro-erosion plots being more erosive. However, an attempt to minimise this difference was made by expressing the runoff and soil loss results per unit of rainfall received at both spatial scales.

c) Soil erodibility and risk of erosion will vary due changes in soil properties and surface characteristics. Fluctuations in soil and surface properties will occur spatially and temporally due to natural variation and tillage induced change. Temporal variations in both soil and surface properties will not be represented at the micro-plot scale. The field erosion plots measured soil and water loss during the majority of the year, from when a crop has been sown until harvest. The micro-erosion plots measured soil and water loss at a specific point in time. Therefore the results from the micro-erosion plots may be more sensitive to the specific soil conditions at that point in time, whereas the erosion plot results reflect a variety of soil conditions over a longer period of time. It may be possible to relate the time specific results of the micro erosion plots to a corresponding tank clearance from the field plots, so that land management conditions (such as crop cover) are comparable.

d) Actual losses of soil and water would be expected to differ between the two spatial scales due to the very different land areas of which they cover. Therefore the losses must be standardise from the two spatial scales, so that soil and water losses measured are presented as losses per unit area (hectare).

In summation, it is expected that differences in soil and water losses between the two spatial scales are the result of different processes or factors affecting erosion operating, which in turn will vary over time. In theory, some factors or processes suggest that erosion would be greater on the field erosion plots, while other factors may suggest higher losses from the micro-erosion plots. On balance, it is expected that losses (runoff, sediment) and runoff coefficients per unit area will be greater from the micro-plot scale compared to the field scale, due to key factors or processes. These are:

I higher rainfall erosivity from artificial rainfall on the micro-plot

II simulations carried out during periods where risk to soil by water erosion is at its highest, specifically before crop establishment

III less opportunity for infiltration and storage of runoff, and less opportunity for deposition at the micro-plot scale, leading to higher runoff and soil losses

The expected discrepancies in soil and water losses at the two spatial scales have led to the formulation of hypothesis two.

6.2.2.2 Hypothesis Two

As previously stated in section 6.2.2 differences in actual soil and water losses are expected between spatial scales, due to the presence of different processes and factors dominating at each of those scales. However, the relative effectiveness of each tillage treatment should be the same. For example, if soil loss was greatest from the conventional treatment compared to the SOWAP treatment at the field plot scale, it would be expected that this relationship would also occur at micro-plot scale. The expected differences between soil management treatments can be found in sections 3.4 and 4.4, which suggest that runoff and soil losses and runoff coefficients will be greater from the conventional treatment compared to both conservation treatments (SOWAP and Farmer's Preference). This is because of the increase in mechanical manipulation of the soil by tillage implements, and decreases in organic matter, surface roughness, residue cover and infiltration rates associated with the conventional treatment. The relative difference between conventional and the conservation treatments is expected to be consistent at both spatial scales; field (≈0.05ha) and micro-plot scale (<2m^2).

6.3 Methodology and Analysis

The results used in this chapter have been obtained from the methods previously laid out in chapters 3, 4 and 5. In brief, each chapter comprised of:

Chapter three: two field erosion plots were installed approximately 0.05 ha for each treatment. Runoff and sediment were collected in tanks at the base of each plot and were emptied when necessary. The erosion plots were installed after crop drilling and were removed just before harvest. During this time the plots were subjected to natural rainfall events and monitored for erosion generation. Data were obtained for all seasons at both sites; Loddington (1-4) and Tivington (1-3).

Chapter four: rainfall simulations were carried out on micro-erosion plots, 1m wide by 1.5m long; three replicates per treatment. Runoff and sediment generated from these plots were collected and measured. Micro-erosion plot rainfall simulations were carried out biannually in spring and autumn. Each simulation was run for around 30 minutes at 35 mm hr^{-1}. Results were measured from all seasons for Loddington (1-4) but only during season two and three at Tivington.

Chapter five: aggregate stability was measured from soil collected from each soil management treatment. Samples were obtained biannually adjacent to the micro-erosion plot experiments. Three replicate soil samples were taken adjacent to each micro-erosion plot rainfall simulation trial. Soil aggregates were tested for stability by means of three different methods; one rain drop impact method (gravity fed rain tower) and two immersion based methods (wet sieving apparatus and a field test kit). Results were obtained from seasons one, two and three at Loddington and only season two at Tivington.

Runoff and soil loss results were standardised for rainfall input and plot size by presenting results as losses per unit of rainfall received (mm) per unit area (hectares).

6.4 Results

To test hypothesis one, results obtained from chapter three and four only will be used (field scale compared with micro-plot scale) because the data from chapter five (aggregate stability) measured soil erodibility rather than actual soil losses. Results were compared for the entire sampling period, and on a seasonal basis at each site location. Micro-erosion plot simulations were only carried out in season 2-3 at Tivington therefore to allow comparison between the two spatial scales only data from season 2 and 3 were used from Tivington. At Loddington all seasons (1-4) were included.

To test hypothesis two, the results from all three chapters (3-5) were used but to allow comparison, only results from season 1, 2 and 3 were used from Loddington and only season 2 from Tivington. These seasons represent the time periods within which soil samples were taken for aggregate stability testing. Comparisons were made of data over the entire sampling period and on a seasonal basis.

Statistical analysis was not undertaken because of the inherent variability and error associated with data collected at each spatial scale. None of the data sets were normally distributed and had unequal variance, even after multiple transformations were carried out. Non-parametric methods of analysis were also tried, including Mann-Whitney U Test and Kruskal-Wallis ANOVA. These tests were not used as they were not robust enough to deal with the bias associated with extreme events (importance of which was highlighted in section 3.4). Where references are made to statistical differences they are based on statistical analysis carried out on data from previous chapters.

6.4.1 Hypothesis one

It is expected that runoff and soil losses will be greater per unit area from the micro-erosion plots compared to the field erosion plots, as will also be the case for runoff coefficients. The results used in this section will be runoff and soil loss represented as losses (litres for runoff and grams for soil loss) per unit area (hectares) per unit of rainfall received (mm). Runoff coefficients will also be compared. Using both actual losses and runoff coefficients allows direct comparison between spatial scales.

The runoff coefficient (*RC*) represents the ratio between the amount of rainfall received and the amount of runoff generated (Hudson 1995). To calculate the runoff coefficient Equation 6.1 was used:

$$\mathrm{RC} = \frac{\text{output (volume of runoff (ml) generated from the plot area)}}{\text{input (volume of rainfall (ml) received over the plot area)}} \qquad \textbf{(6.4-1)}$$

6.4.1.1 Loddington

The results of runoff, soil loss and calculated runoff coefficients from both the micro- and field- erosion plots can be seen in Equation (6.4-1). The results show that when data are compared between spatial scales overall (combined results of all seasons and for all treatments) the results from the micro-erosion plots are always greater per unit area per unit rainfall than those from the erosion field plots; this is the case for runoff, soil loss and *RC* values. This was also the case when results were compared between spatial scales for each season. An overview table of the results including relative percentages can be found in Appendix K.

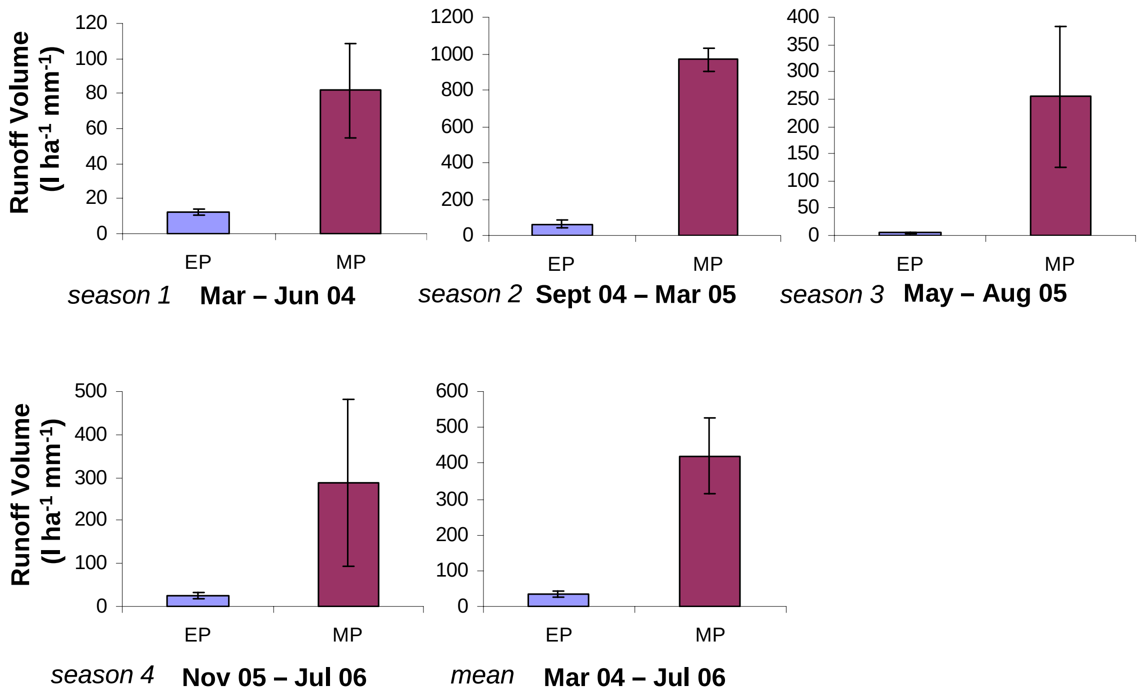


Figure 6.4-1 Loddington spatial comparison of actual runoff losses. Field erosion plots (EP) and micro-plots (MP). Error bars denote standard error.

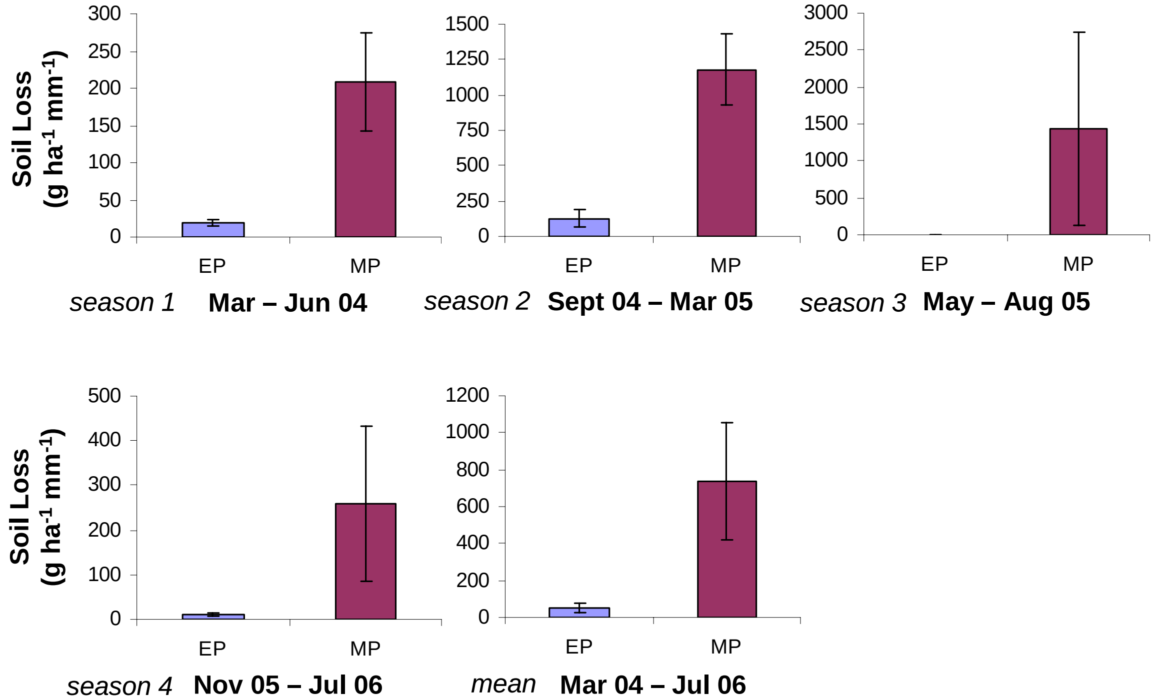


Figure 6.4-2 Loddington spatial comparison of actual soil losses. Field erosion plots (EP) and micro-plots (MP). Error bars denote standard error

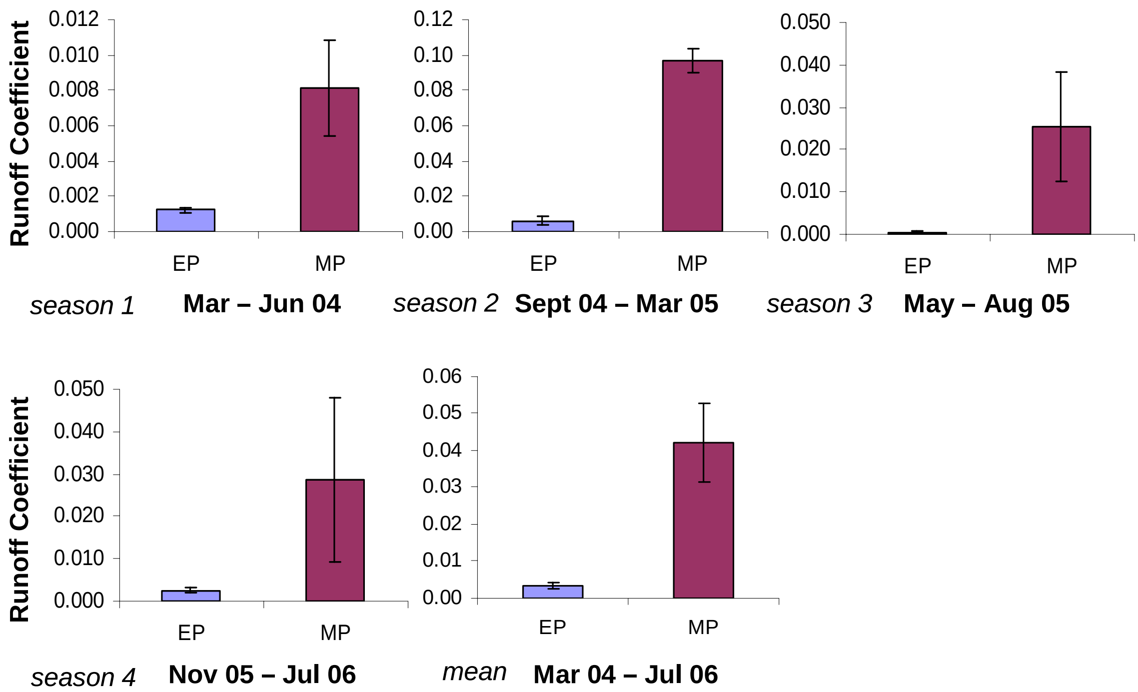


Figure 6.4-3 Loddington spatial comparison of runoff coefficients. Field erosion plots (EP) and micro-plots (MP). Error bars denote standard error

Although no statistical analysis has been carried out it can be seen that there are notable differences in variability of results between spatial scales. Also there are considerable differences in results obtained at these two spatial scales with higher runoff, soil losses and *RC* values from the small scale (micro-erosion plots) compared to the larger field erosion plots.

6.4.1.2 Tivington

The mean result for each season of runoff, soil loss and runoff coefficients from both the field erosion plots and the micro-erosion plots are presented in the Appendix K. The results represent combined data from all treatments. In all cases bar one, the results from the field plots are lower than the micro-plots. The exception was the soil loss results during season 3 where the opposite occurred. The results from each measured parameter were compared between spatial scales by calculating the percentage loss or coefficient from the erosion plots relative to the micro-plot results. These results are shown in Appendix K.

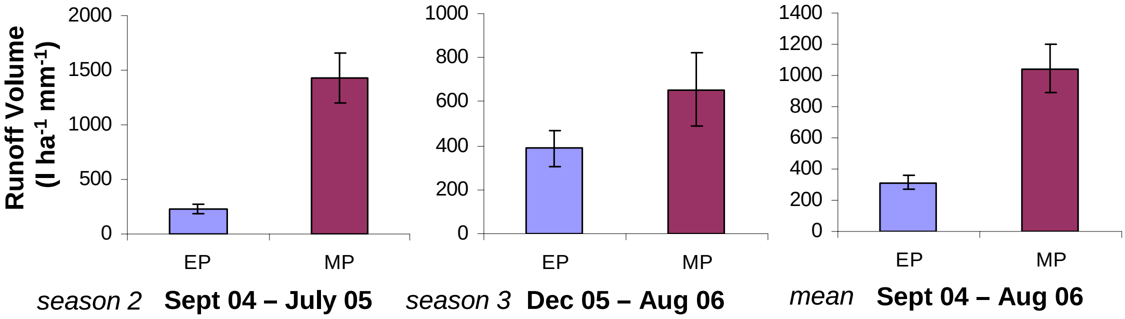


Figure 6.4-4 Tivington spatial scale comparison of actual runoff losses. Field erosion plots (EP) and micro-plots (MP). Error bars denote standard error.

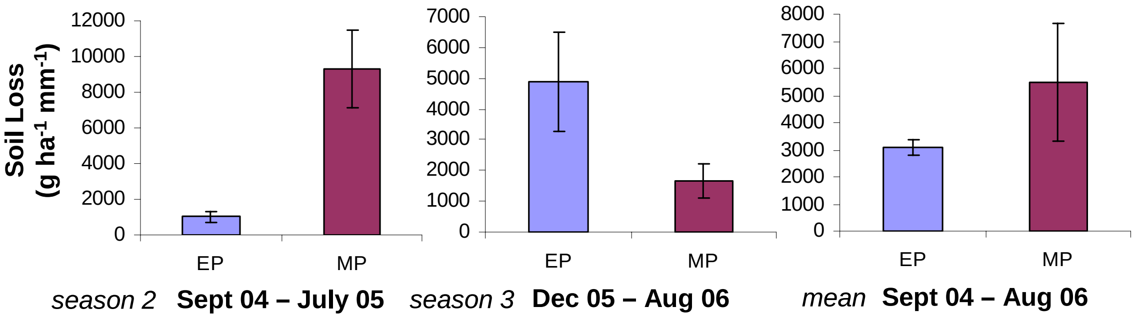


Figure 6.4-5 Tivington spatial scale comparison of actual soil losses. Field erosion plots (EP) and micro-plots (MP). Error bars denote standard error.

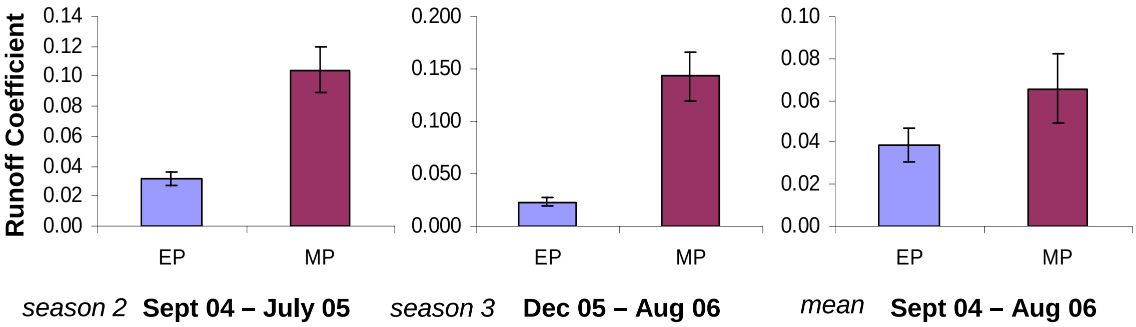


Figure 6.4-6 Tivington spatial scale comparison of runoff coefficients. Field erosion plots (EP) and micro-plots (MP). Error bars denote standard error.

Again no statistical analysis was undertaken but it can be seen that the variation between replicates tends to be greater from the micro-plot results compared to that from the erosion plots. Variability within the micro-erosion plot results appears smaller at the Tivington site compared to Loddington. This is likely to be a function of higher losses from the Tivington site (Nearing et al. 1999). The relative difference between micro- and field- erosion plots also appears reduced at Tivington compared to Loddington, although this could be related to a reduction in error as previously mentioned.

6.4.2 Hypothesis two

It was expected that runoff, soil loss and soil erodibility would be highest from conventionally managed soil in comparison to the conservation treatments. It was also expected that the ranking of soil erosion/erodibility results between treatments would remain constant, irrespective of the spatial scale from which the data were generated.

To allow a more direct comparison between spatial scales in terms of treatment ranking, only seasons where data was generated at all scales were initially used. The limiting spatial scale was the small scale assessment of aggregate stability; these tests were only carried out at Loddington during the first three seasons and at Tivington during the second season only.

The means results over these data sets for all spatial scales from each site location can be found in Appendix K. The results show the relative rank of each mean for the three treatments, the site location, and the scale from which the data were collected and the methods used to obtain these results. Overall it was found that the relative ranks of the three treatments differed between spatial scales. It was important to investigate if this discrepancy between spatial scales existed on a seasonal basis. As stated previously, to allow comparison only season 2 was used from Tivington, therefore this site could not be used to show seasonal changes in data. In light of this, only seasonal data from Loddington was investigated, the results of which are presented in Appendix K.

To allow an easier comparison, results from the two conservation treatments (SOWAP and Farmer's Preference) were converted to percentages, relative to that of the conventional treatment. This was done for each site location and for each spatial scale and method employed. These results were plotted for both conservation treatments at each site location. The relative results of the conservation treatments to those of the conventional treatment have also been presented on a seasonal basis. Again, as stated previously, only data from Loddington could be used to highlight seasonal changes (Figure 6.4-9). Overall spatial comparisons have been presented in Figure 6.4-7 for Loddington and Figure 6.4-8 for Tivington.

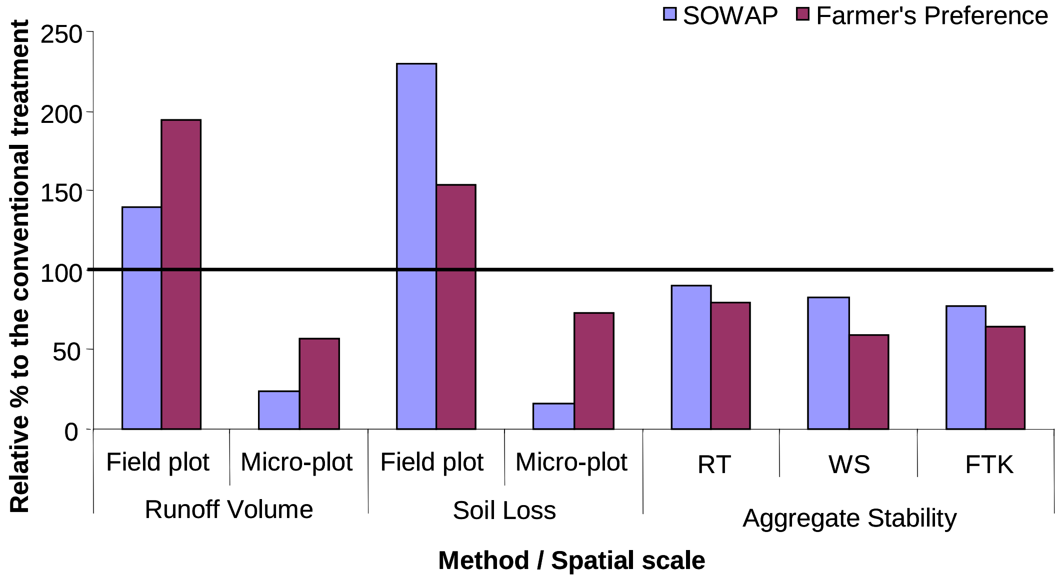


Figure 6.4-7 Loddington: relative percentage losses of runoff, soil loss and aggregate stability in comparison to the conventional treatment. Aggregate stability methods are: RT = gravity fed rain tower, WS= wet sieving and FTK = field test kit.

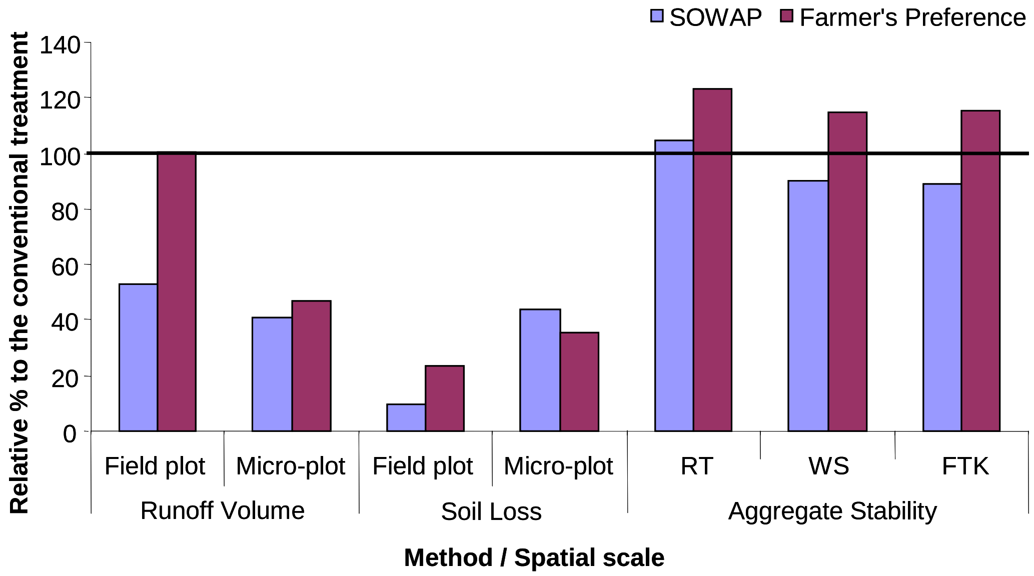


Figure 6.4-8 Tivington: relative percentage losses of runoff, soil loss and aggregate stability in comparison to the conventional treatment. Aggregate stability methods are: RT = gravity fed rain tower, WS= wet sieving and FTK = field test kit.

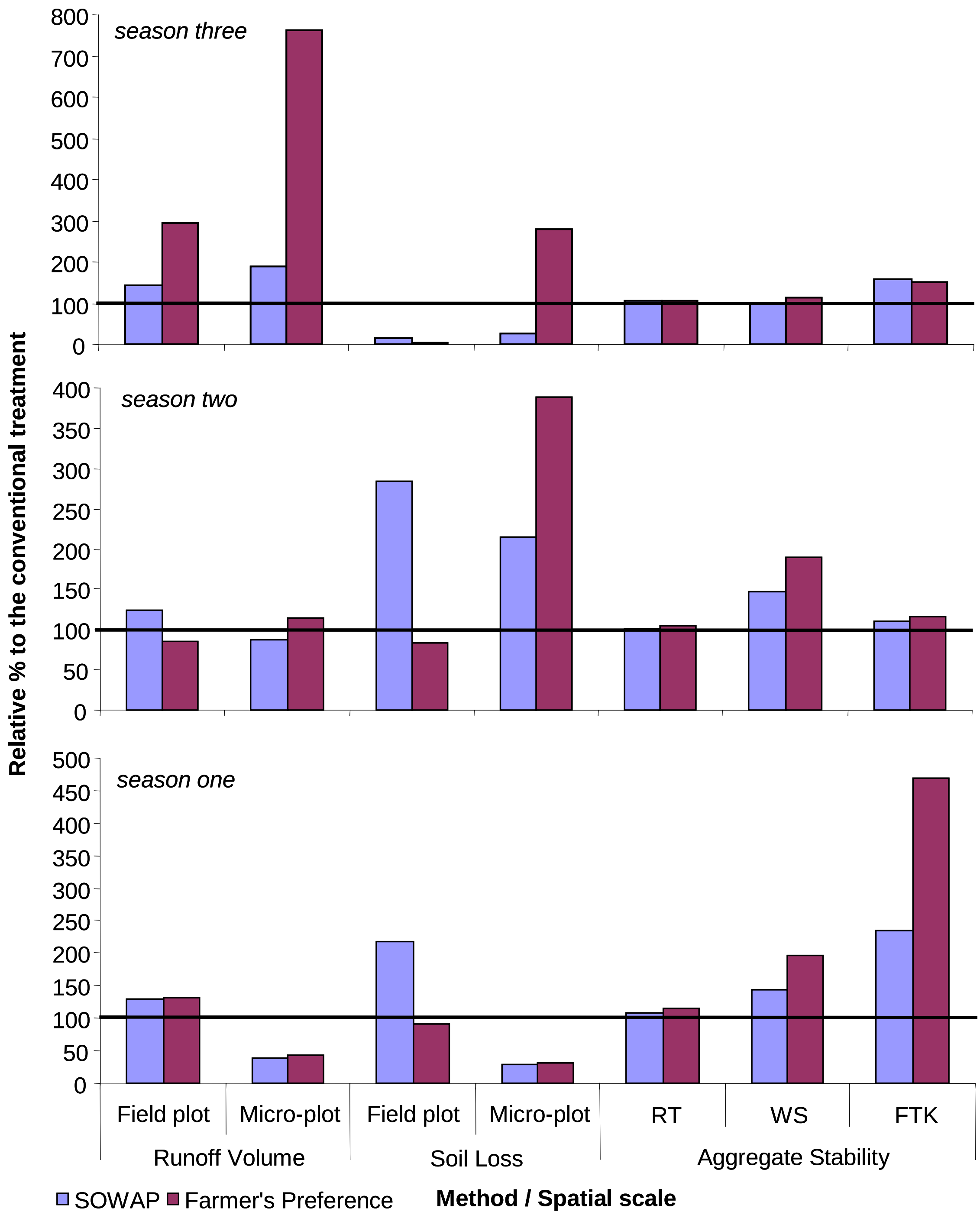


Figure 6.4-9 Loddington: relative percentage losses of runoff, soil loss and aggregate stability in comparison to the conventional treatment for seasons one to three. Aggregate stability methods are: RT = gravity fed rain tower, WS= wet sieving and FTK = field test kit

6.4.3 Additional information

To gain a better understanding of the soil loss results in particular, the kinetic energy for the rainfall received at both the micro- and field- plot scales was calculated. Background and the overall findings of this analysis can be found in the Appendix L.

6.5 Discussion

6.5.1 Extrapolation between spatial scales in terms of actual losses

As stated in hypothesis one, it was expected that reliable extrapolation of runoff and soil loss from the micro-plot scale to the field scale could not be done. Instead it would be possible to use relative treatment differences at small spatial scales to indicate field scale differences. The small scales include micro-erosion plot generation of runoff and soil loss; and aggregate stability results of erodibility.

The difference between scales was expected to exist because of the different erosion processes and factors involved (Imerson & Lavee 1998). For example it is expected that at the micro-plot scale erosion would be primarily be controlled by rain splash processes, while at the field scale overland flow and rill formation would dominate. However, overall losses per unit area, per unit of rainfall received were expected to be greatest from the micro-plot scale when compared to the field plots.

It was found the mean runoff volumes and runoff coefficients were higher from the micro-erosion plots in comparison to the field plots. This was also the case for the mean (all seasons used in analysis) soil loss. The only exception occurred at Tivington during season 3. To gain a better understanding as to the processes which had occurred during this season, photographs were revisited before and after the micro-plot simulation were carried out and then compared to previous seasons. Photographs clearly showed the importance of splash erosion at the micro-plot scale, with visible deposition of detached particles on the plot boundaries. However, this was not apparent in the photographs take during the simulations of season 3 (example of splash erosion during season 2 is shown in

Plate 6.5-2), implying that during season 3 soil aggregates were less susceptible to impact of rain drops. In addition to this, it was also found that bulk density and moisture contents were substantially lower during this season. These changes in these soil properties would have increased infiltration rate and capacity, reducing the amount of water on the surface and thus reducing splash erosion. The beneficial effect of depth of water reducing rain drop impact only occurs at a critical depth, before this depth is reach, increase of water depth will exacerbate rain drop impact (Palmer 1964).

Apart from the exception at Tivington (which has been explained above) losses (calculated per unit area) were substantially greater from the micro-erosion plot compared to field plots. It has already been stated that splash erosion was an important factor in explaining differences between scales, but this was not the only reason.

One of the previously highlighted factors affecting spatial scales comparison (section 6.2.2.1) is the difference between natural and artificial rainfall. The simulated rainfall was set to and measured an intensity of 35 mm h^{-1}, representing a 1-6 year storm event (NERC 1975). The mean rainfall intensity of the natural rainfall received at the field scale was around 3 mm h^{-1} for both site locations. Results out which can be found in Appendix L. The I_5 (the maximum rainfall intensity received over a 5 minute period) was also calculated for both artificial (micro-plot scale) and natural rainfall (field scale). It was found that the I_5 was not significantly different between artificial and natural rainfall. However, it is believed that this does not represent the duration of the storms received at each scale. Although the maximum intensity over a 5 minute period was the same between scales, at the field scale it was uncommon for rain to fall for more than 5 minutes at a time. The micro-erosion plots received rainfall at a high intensity (35 mm h^{-1}) for around 30 minutes at a time. This vastly different rainfall pattern received at each spatial scale would explain the large differences in losses found.

Another reason for erosion results being much greater from the micro-plot scale was due to an overestimation caused by the difference in collection systems at each scale. Whether at the micro or field plot scale, inherent error is associated with installing a collection system for sediment and water at the soil interface. The collection systems have to be placed partially beneath the soil to minimise the effect of undercutting by overland flow as a result of rainfall. At the soil/collection system interface, soil stability is expected to be less as aggregates are partially fragmented from the main soil body due to disturbance during installation. When rain falls, these fragmented particles are more easily eroded compared to the undisturbed soil on the plot. Once eroded, particles are easily collected directly due to the proximity of the collection system. Although this error is associated with both micro and field plots, the ratio of plot area and width of the collection system was felt to be important. As the collection width to plot area ratio increases a greater proportion of eroded soil would be collected. Also the collection system at the field plot scale has undergone a settlement process. Even though the metal boundary is removed for each field operation the collection system remains in place. Soil adjacent to the collection system therefore has time to settle and consolidate over time, unlike the more temporary micro plot collection system which is only present for a matter of hours. The collection systems at each spatial scale are shown in Plate 6.5-1.

Plate 6.5-1 Micro-plot (left) width of 1m compared to field plot (right) plot width 9m (collection funnel 1.5m)

6.5.1.1 The impact of site location on spatial scale comparison

It appears that relative differences in measured parameters between the micro-plot and field plot scale are less at Tivington than at Loddington. This was unexpected as the risk of splash erosion would have been substantially greater due to inherent differences between site location (discussed below). Therefore greater runoff and soil loss would have been expected on the micro-erosion plots compared to the field plots at Tivington. Interestingly the impact of splash erosion was (as expected) less noticeable at Loddington, with most of the erosion being caused by overland flow detachment and transport (Table 6.5-1). So there must be another reason as to why relative losses between spatial scales were smaller at Tivington.

Plate 6.5-2 Splash erosion at Tivington (left) and overland flow at Loddington (right) during micro-plot trials

Firstly at Tivington a higher magnitude of runoff volume and mass of soil lost was generated, which helped to reduce the error at the micro-plot scale. Secondly, absolute differences in runoff and soil loss could also be attributable to inherent soil properties and site characteristics. The Tivington site has a higher gradient of 7% compared to Loddington at only 3.5%. The soil at Tivington is classed as a sandy clay loam, part of the Worcester series which has been identified as having an minimal risk to water erosion, whereas, the Loddington soil is a clay, (>44%) part of the Hanslope and Denchworth clay series, and has no risk to water erosion mentioned (further detail can be found in chapter 2).

Loddington soil has 20% more clay content that of Tivington and 4.4% more organic matter, therefore it would be considered that the risk of water erosion at the Loddington site would be less.

Finally, there were site differences in wind velocity. The effect of wind was a great problem during the micro-plot trials at both sites. The mean wind velocities were 1.26 m s^{-1} at Tivington and at Loddington 2.19 m s^{-1} but the maximum velocities recorded were over 9 m s^{-1} and 6 m s^{-1} at Loddington and Tivington, respectively. A more detailed breakdown of the wind velocities received at both sites can be found in the Appendix M. Even gentle breezes can affect rainfall distribution in terms of volume and erosivity of rainfall received on a plot (Helming 2001). Observations during experimental runs noted considerable variation in rainfall received by the micro-plots especially. This may also lead to an overestimation of runoff at the micro-plot scale, as simulated rain is also blown into the collection systems. Strong winds can lead to a net movement of eroded soil particles down slope (Warburton 2005), and into the collection systems at the base of the micro and field plots. Wind breaks were used during micro-plot trials but did not solve the problem completely.

It has been shown that the results obtained from rainfall simulations carried out on micro-plots cannot be used to predict erosion losses at the field scale – the discrepancies between spatial scales are too substantial. The next step was to see if the results from the three soil management treatments when ranked could be compared across the different spatial scales. The spatial scales used in this comparison were field plots and micro-plots where runoff volumes and soil losses were measured and small scale assessment of soil erodibility where aggregate stability was measured.

6.5.2 Spatial scale comparison of relative treatment ranks

The hypothesis was whether relative treatment ranks or percentages, in terms of runoff and soil losses generated from the field plots, can be predicted by the micro-erosion plot and aggregate stability results.

The results varied between site locations and parameters measured (runoff or soil loss). Table 6.5-1 shows a summary of these results. At Loddington, neither the micro-erosion plot nor aggregate scale results showed the same treatment ranking as that found at the field scale for either runoff or soil loss. However, at Tivington soil erodibility as measured by use of the gravity fed rain tower did result in the same treatment ranking as found at the field scale; this was only in terms of runoff. This finding concurs with the work done by Barthés et al. (2000), who found positive relationships between aggregate stability and runoff volumes. The most number of ticks (showing the same treatment rank as the field scale) came from the gravity fed rain tower method to measure aggregate stability. This is encouraging, as it reflects the fact that this method of stability assessment is able to simulate field erosion processes (i.e. aggregate breakdown by raindrop impact). There is concern that the other methods of aggregate stability testing which involve total immersion of aggregates (wet sieving and field test kit) do not simulate actual erosion processes experienced in the field.

This data only compared treatment ranks for the entire sampling period, but did not take season into account. In order to allow a more direct comparison between scales, a breakdown of results by season could not happen at the Tivington site as previously discussed previously, therefore seasonal results will only be shown from Loddington. As with the results for the entire sampling period, the treatment ranks found at each spatial scale were not consistent. The results have been summarised in Table 6.5-2 to gain a clearer understanding of which spatial scale and method matched the treatment ranks produced from the field scale.

Table 6.5-1 Relative treatment ranking of runoff and soil loss from both site locations. Ranking order: I – highest to III – lowest values at the field scale. ✓ represents agreement in treatment ranking for different assessment methods

Site	Factor	Treatment	Rank	Micro-plot	Aggregate		
					RT	WS	FTK
Loddington	Runoff	Conventional	III				
		SOWAP	I				
		Farmer's Preference	II				
Loddington	Soil Loss	Conventional	II	✓			
		SOWAP	I				
		Farmer's Preference	III		✓	✓	✓
Tivington	Runoff	Conventional	III		✓		
		SOWAP	II	✓	✓		
		Farmer's Preference	I		✓	✓	✓
Tivington	Soil Loss	Conventional	I	✓			
		SOWAP	III			✓	✓
		Farmer's Preference	II				

The ranked results for runoff generation from the micro-plot scale gave the best agreement (most ticks) with the field plots scale, with this being the case for all treatments during season three. However, in relation to soil loss, the micro-plot scale results did not match a single treatment rank from the field scale. The most number of matched results came from the small scale assessment of soil erodibility using the gravity fed rain tower method. This included all treatments from season three.

It was found that at both sites the best agreement (i.e. most number of ticks) with results from the field plot scale came from the gravity fed rain tower of assessing aggregate stability. If this method could be used to indicate field scale response of erosion the implications would be a faster, more replicable and cheaper method compared to setting up and running of micro-erosion plots in conjunction with a rainfall simulator. However, the results are not completely conclusive and further work would be necessary.

Table 6.5-2 Relative treatment ranking of runoff and soil loss from Loddington across three seasons. I – highest to III – lowest values at the field scale. ✓ represents agreement in treatment ranking for different assessment methods

Factor	Season	Treatment	Rank	Micro-plot	Aggregate		
					RT	WS	FTK
Runoff	1	Conventional	III				
	1	SOWAP	II		✓	✓	✓
	1	Farmer's Preference	I				
Runoff	2	Conventional	II	✓			
	2	SOWAP	I				
	2	Farmer's Preference	III		✓	✓	✓
Runoff	3	Conventional	III	✓			
	3	SOWAP	II	✓	✓		
	3	Farmer's Preference	I	✓			
Soil Loss	1	Conventional	II				
	1	SOWAP	I				
	1	Farmer's Preference	III		✓	✓	✓
Soil Loss	2	Conventional	II				
	2	SOWAP	I				
	2	Farmer's Preference	III		✓	✓	✓
Soil Loss	3	Conventional	I		✓	✓	✓
	3	SOWAP	II		✓		
	3	Farmer's Preference	III		✓	✓	

Although ranking has given an indication of the difference or similarities between spatial scales it may not be the most effective comparison, as it gave no indication of how relatively close the results from the different treatments were. To try and resolve this, the relative percentage losses were calculated for the conservation treatments (SOWAP and Farmer's Preference) compared to the conventional treatment which had a percentage of one hundred. The results showed that even where the treatment ranks where the same between spatial scales, the variation in relative results was sometimes high.

To solve the problem of data variation, statistical differences based on the results from previous chapters, were used to try and compare the data in a statistical fashion. Direct statistical comparison between spatial scales was not possible because of the nature and inherent variability associated with them. The results showed that in the majority of cases no significant differences were found between tillage treatments in terms of runoff or soil loss at the field or micro-plot scale. Significant treatment differences were found at the aggregate scale when immersion base methods were employed but not when the rain drop impact method was used. The latter better representing natural rainfall. The lack of significant treatment differences at each spatial scale failed to highlight interesting and important differences that existed. It is therefore felt that the sole use of statistics is insufficient when attempting to make comparisons of erosion between tillage treatments across different spatial scales.

6.5.3 Implications of this study

The implication of this study is that small scale assessments of erosion and erodibility cannot be used to predict field scale erosion losses, but that scale comparisons improve on sites which are more susceptible to erosion. There is the potential to use small scale assessment of erodibility through use of a rain tower to indicate relative effectiveness of different soil management treatments at the field scale; but caution should be used. Different erosion processes and factors affecting these have been identified at small and larger spatial scales, but further research is required to understand the limitations, regarding factors such as crop cover and soil surface management.

6.5.4 Future recommendations

This study has shown that there is a great deal of variation associated with erosion data which makes spatial comparison extremely difficult. Small scale assessment of erodibility has the potential to be a powerful tool in assessing field scale erosion, as it represents the resistance of soil aggregates to raindrop impact, which is the primary process of erosion. It is well known that soil management practices affect soil stability and in turn soil erosion, however, getting that direct link has proved difficult.

This study took aggregates from different management treatments and subjected them to erodibility tests. The results were then compared with erosion data collected in the field. The results were compared on a seasonal basis and for the entire sampling period. This study would have benefited from direct comparisons of soil erodibility assessments in the laboratory with erosion results obtained from the field scale on an event basis. This was achieved already by comparing results from the micro-erosion plots to aggregate stability results. However, the agreement between results from the micro-plot scale and the field erosion scale was not consistent. Therefore, an inferred relationship between the field scale and aggregate stability results could not be made. In addition to the micro-erosion plot experiments did not simulate natural rainfall intensities or storm duration.

More work is needed to understand the why small scale assessments of aggregate stability were more efficient at indicating field scale erosion on a lighter, more erodible soil (Tivington) than one with a higher clay content (Loddington). The study has also highlighted the fact that measurements of runoff, erosion and erodibility are affected by the methodology used. This is important to remember when reviewing previous research and data concerned with soil erosion.

It is predicted that better data comparison can be achieved between aggregate stability results obtained via rain drop impact methods and field erosion results if the starting condition of the test aggregates are dry and not saturated. Further work is needed (as already highlighted in chapter 5) of the impact starting conditions of test aggregates have on the stability results.

6.6 Conclusion

Mean runoff, soil loss and erodibility associated with 3 different soil management treatments have been compared at 3 spatial scales: field erosion plots, micro-erosion plots and individual aggregates. The results have been expressed in three ways, through treatment ranking, relative percentages to the conventional treatment and use of statistical analysis within each scale. The results from this study have not been able to give a conclusive answer as to whether small scale assessment of erodibility (aggregate scale) or erosion (micro-plot) can be used to predict the relative ranking of treatment runoff or soil loss from the field plot scale. Better agreement is obtained on a seasonal basis, and on sites with a higher risk of erosion, but there is still a great deal of uncertainty.

Small scale assessment of erodibility using a gravity fed rain tower gave similar ranking results as obtained from the field scale in more cases than observed for the micro-plot scale. This study has been able to conclude that from these results, micro-plot scale experiments should not be used to extrapolate actual results of runoff volume, soil losses and runoff coefficients to the field plot scale. The erosion processes involved at each scale and associated error with the collection systems installed at each scale are too great to allow confident extrapolation of results. This is an important outcome of this study for many researchers who still extrapolate erosion from one scale to another especially through the use of erosion prediction models.

7 Conclusions and Recommendations

This chapter puts the present study into the context of current research and legislation relating to soil erosion. This chapter addresses two main questions: a) how does this present study contribute to soil and water protection research? and b) what are the implications and applications of the findings of the current study?

7.1 Uniqueness of this research

This study has made a number of original contributions to present day research related to losses of soil, water, nutrients and carbon. The infrastructure developed to monitor soil and water losses in this project is rarely found in erosion studies. Fully instrumented field erosion plots were installed at two localities within the UK. Each site was subjected to intensive monitoring programs utilising an extensive protocol. Comprehensive instrumentation was installed at each site location, measuring a broad spectrum of meteorological and environmental parameters, and monitoring depth of runoff and eroded material at each field collection system. A unique data set was thus created, encompassing information on soil, water, nutrient and carbon lost through erosion, in conjunction with supporting field evidence.

Sites were simultaneously operational, seldom found in erosion research. Ordinarily only one geographical location is used (Quinton 2005, Quinton et al. 2006, Rickson 1994 and Govers & Poesen 1988). The sites were operational for over two years, enabling data analysis over four temporal scales - individual events, series of events, by cropping seasons and over the entire project duration. Such breadth of temporal scales is infrequently found in erosion research.

The results over these different temporal scales were not just confined to one plot size, but simultaneously investigated over three different spatial scales. The use of one spatial scale is commonly found, two is more infrequent and three exceptionally scarce, especially in non-model, field based research. This unique study allowed the following question to be asked. Is it possible to extrapolate soil and water losses from one spatial scale to another? Results indicate that direct extrapolation of actual soil and water losses from micro-erosion plots to the field scale is unreliable and prone to error. More success was found when the relative (ranked) differences between tested treatments were compared for each spatial scale. Although more successful than direct extrapolation, results were still inconsistent. The conclusion drawn is that spatial extrapolations of erosion data are associated with a substantial degree of error which requires further investigation, explanation and understanding. This finding calls into question the validity of previous research, much of which has attempted to extrapolate between spatial scales, without acknowledging the limitations of this approach.

Using soil aggregate stability as an indicator of soil erodibility is not a novel concept. Previous research has focused on the influence of different experimental protocols (e.g. antecedent moisture content, procedures of pre-wetting aggregates) on results gained. This present study has directly compared three recognised methods of aggregate stability testing (2 based on total aggregate immersion and one based on raindrop impact).

When comparing the two immersion based methods of aggregate stability (wet sieving and a field test kit, designed by Herrick et al. (2001) the present study indicates that the field test kit can replace the more laborious laboratory methods, when comparing relative (ranking) differences between management treatments. The potential benefits of this are extensive, enabling a wide range of people to perform aggregate stability tests in the field quickly, with minimal expenditure and nominal training. However, it should be noted that both immersion based methods are highly destructive, simulating flooded conditions. A distinction must therefore be made between results gained from immersion based methods and those generated from raindrop impact. The latter represent more closely the process of natural rainfall.

Thus, the results gained highlight the sensitivity and validity of using different assessments of aggregate stability as an accurate indicator of soil erodibility. The differences between results for the different methods raise questions as to the rigour of previous research. In particular, this study has shown the influence of method on results gained - comparison of results from different methods may be unsound, and yet this is rarely acknowledged in the literature.

It emerged from this research that the effectiveness of any soil management regime will be site specific. This outcome implies that there are no universal panaceas with regard to the use of tillage treatment on mitigating soil and water losses, despite the many claims being made to this effect (ECAF 2004). This could have significant implications in the formation and application of policy. This point shall be expanded upon in section 7.3.

7.2 Overall observations

Data generated in the field are associated with a high degree of variability, due to the considerable number of variables operating. Examples include natural topographic variations and human induced inconsistencies during implementation of field operations. Each field erosion plot was approximately 0.05 of a hectare and duplicated for each treatment. Although stated as the minimal number of plots recommended (Morgan 2005), the number of field erosion plots per treatment was considered to be a limiting factor in generating statistically significant relationships between soil/water losses and site conditions (see section 7.4). However, this limitation must be considered in light of the fact that this project was part of a larger demonstration project (SOWAP), and it was imperative that the plots used were of sufficient size that farmers were convinced that the findings were realistic and representative of the field scale. Irrespective to these project limitations, some overall conclusions can be made.

The effectiveness of conservation soil management in reducing soil and water loss was spatially variable. Field scale investigations revealed that the conservation tillage regimes were unable conserve water by reducing runoff volumes, compared to the conventional tillage treatment. This has implications for future flood risk and control policy formation. Conservation tillage, therefore, should not be employed as a management practice for flood management and alleviation.

The success of soil management in the reduction of soil loss via water erosion was site specific, confirming that stated by Holland (2004). This can be explained by the inherent difference in soil erosion risk. There were distinctive differences between the two UK sites. The soil from the Tivington site was more erodible than at Loddington. This was not surprising due to the fact soil at Tivington was a sandy clay loam compared to the heavy clay of Loddington. Also soil from Tivington had lower contents of organic matter and the site had a relatively higher slope gradient, increasing the risk of erosion.

It was observed that the presence of surface residues, a rougher soil surface and increases in clay content had a more profound effect on sediment generation on more erodible soils. This explained why the adoption of conservation tillage (field scale) was only successful in minimising soil loss in comparison to conventional tillage at Tivington. The finding that conservation tillage is more effective on erodible soils is supported by data from the Belgian sites. Here, conservation tillage is highly effective at controlling soil erosion on highly erodible loess soils. Interestingly it was found at Loddington, soil loss was on some occasions exacerbated by use of conservation tillage. This was attributable to uncharacteristic surface conditions on the conventional treatment and a high amount of experiment error. The latter relating to very small amounts of sediment generated through the study period. It was concluded that further research is needed to understand the role of soil texture on the effectiveness of conservation tillage.

The effect of tillage treatment on losses of soluble and sediment-associated nutrients and carbon was highly variable. The effectiveness of conservation tillage in reducing nutrient loss was only found on the heavy clay soil of Loddington. At Tivington (sandy clay loam), the adoption of conservation tillage did not control loss of nutrients. Also at Tivington, conservation soil management led to increased losses of organic carbon compared with the conventional treatment, but only during small rainfall events. This might be linked to the fact that only the small fractions are eroded in such events (Quinton et al. 2006), and yet these are the fractions associated with higher adsorption of nutrients and carbon. At Loddington carbon losses did not differ significantly between tillage treatments. Such variation between site locations meant no generic conclusions could be made for the effectiveness of conservation tillage on nutrient or carbon losses.

Important findings from this research were that the adoption of conservation tillage as a ‘best management practice’ can reduce soil loss on a sandy clay loam soil by the improvement of soil properties and surface conditions. However, these benefits might be counterbalanced by an increased risk of carbon and nutrient loss. The effectiveness of conservation tillage in comparison to conventional practices in controlling soil, water, nutrient and carbon losses is site specific, and is affected by changes in soil type, slope gradient and rainfall patterns. Therefore, no generic statements as favoured by policy makers can be made. Individual site assessments are needed before ‘best management practices’ such as conservation soil management can be devised and implemented.

7.3 Applications of this study’s findings

The findings of this study have two applications: to existing soil erosion research, and to policy formulation on a national and EU level. Each area of application will be considered individually.

7.3.1 Existing research

The present study has addressed some of the research gaps as identified in chapter 1 in particular – generating and analysing data on the effectiveness of conservation soil management practices on losses of soil, water, nutrients and carbon.

Data obtained during this study, represent losses from typical arable systems under a UK temperate climate. This unique dataset could form the baseline for future assessments of changes in soil and environmental conditions as a result of changes in land management, land use and (of particular relevance today) climate change. The latter is an important and ever increasing research topic (EA 2004b). This extensive dataset could be important for future model building and validation, encompassing two UK locations consisting of different soil types, topography and rainfall regimes.

Extensive research has been undertaken on the effect of soil properties and surface characteristics on soil erosion, however, changes in these factors as a result of different tillage regimes are less common. The findings of this present study can contribute to a number of on-going debates within the discipline of soil erosion.

Published, peer-reviewed literature indicates the presence of organic matter and carbon within the soil reduces soil erosion, due to improvements in soil structure and aggregate stability (King et al. 2005, Robinson & Phillips 2001; Le Bissonnais et al. 2002). Soil with a higher organic matter and carbon content is therefore expected to reduce the propensity to erosion. Observed results from this study confirm this relationship. The Tivington site with a measured baseline organic matter content of 0.84% generated substantially higher volumes and mass of runoff and sediment in comparison to the Loddington site at 5.2%. At both sites, the adoption of conservation tillage led to an expected accumulation of organic matter and carbon, due to the lack of soil disturbance by inversion tillage and higher levels of biological material in the form of surface residues and/or cover crop. However, these increases in organic matter / carbon did not always result in reductions in runoff or soil loss. This was due to the influence from other soil properties and surface characteristics present at the time. This illustrates the complexity of interactions between factors affecting the processes of runoff generation and soil erosion.

There is evidence in the literature showing that the long-term use of conservation tillage under some circumstances may actually enhance runoff generation (Holland 2004). This may be caused by a lower level of soil disturbance in the plough layer (relative to conventional tillage) leading to soil consolidation over time. As a result bulk density is increased and infiltration reduced, causing higher runoff generation. In this research where significant differences in bulk density exist (Loddington), it is the conventional treatment which shows higher bulk density compared to at least one of the conservation treatments. Therefore, the results from this research contradict the general assumption that the adoption of conservation tillage increases bulk density over time as shown in work by Cavalaris & Gemtos (2002). This might be related to the limited duration of the current study. Observations of increased bulk density under conservation tillage tend to occur after 5 years. This illustrates the need for longer term monitoring of different soil management practices to quantify the true benefits or limitations of different treatments.

At both site locations surface characteristics were found to be influential in the generation of soil and water losses. These were reduced by the presence of surface cover, and increased undulation of the microtopography associated with surface residues and tillage induced roughness. The influence of surface roughness was expected to reduce over time, as a result of consolidation from rainfall impact, and this was supported by the observed data (especially at Tivington). However, the effect of the reduction in surface roughness in minimising soil and water generation could not be isolated from other soil and surface properties. The presence of a surface cover in the protection of soil against erosion have been studied or expressed by numerous researches (Morgan 2005; Mandal et al. 2005; Hudson 1995; Robinson & Phillips 2001; Fullen & Catt 2004; Wischmeier & Smith 1978). Observations from this current research generally support this relationship, being of particular importance during periods of heavier rainfall.

7.3.2 Policy

Currently there is active EU legislation in place giving farmers incentives to follow 'best management practices', as initiated under programmes such as the reformed Common Agricultural Policy or CAP. Recently, agricultural policy has moved away from issues of production and markets, towards incentive-based legislation which requires the integration of wider environmental benefits to existing farming practices. Advantages include a reduction of soil and water loss, minimising nutrient and carbon losses and the improvement of local biodiversity. The adoption of conservation based soil management practices is currently being advocated as appropriate farming methods to achieve this goal (DEFRA 2005; ECAF 2004)

The observed results from this current study indicate that the application of conservation soil practices is not always associated with environmental benefits. EA (2004b) identified that policy on the importance of soil on carbon dynamics (sinks and sources) must be strengthened. The observed results indicate that the adoption of conservation soil management on sandy clay loams with low organic matter content, may be detrimental to the capacity of soils to retain carbon, releasing it in runoff and eroded sediment. As previously stated conservation soil management did not reduce runoff generation, and in some cases increased soil loss. Despite this the adoption of conservation tillage resulted in the increase in organic matter and carbon content, a reduction in bulk density and maintenance of soil moisture. These improvements are beneficial in terms of soil erodibility. Also, as a result of the changes in soil properties, the soil 'health' is improved affecting the soil biota (Allton 2006). In this sense, this study suggests that the goal of improving local biodiversity with conservation tillage can be achieved. This is explicitly mentioned in the EU Habitats Directive (EUROPA 2006d).

7.4 Research limitations

A high degree of variation was observed for the field erosion plots due to natural system variability. Two plots per treatment were perhaps insufficient to overcome error associated with random features including natural changes in soil and underlying geology, drainage pathways, gradient fluctuations, patchy crop growth, animal damage and human induced features such as tramlines and turning marks. However, an increase in plot replication would have led to a decrease in plot area, which would be unrepresentative of the field scale.

Quantification of soil erodibility via aggregate stability tests was found to be challenging. Aggregates are highly sensitive to outside destructive forces and begin to breakdown from their natural state during removal from the soil environment, transportation and preparation before testing. Despite these inevitable concerns, comparisons were still possible, as it can be argued that such variability is consistent for all treatment. To allow comparison between aggregate stability methods, aggregates had the same starting conditions. Due to time constraints, a well used, saturation method of fast wetting was employed. Unfortunately this method was highly destructive and caused partial breakdown of aggregates before stability tests could be carried out. Therefore realistic representation of field conditions may not be possible and was thought to be the reason why expected treatment differences did not occur when aggregates were tested using the gravity fed rain tower.

Also, assessing soil erodibility by using individual aggregates of a specific size has been stated as being unrepresentative of the whole soil (Six et al. 2000). Tests on whole soil samples, and aggregates of different ranges of sizes showed that aggregate size did not significantly affect stability results. This relationship is seldom acknowledged in research relating to aggregate stability.

7.5 Recommendations

Soil erosion and erodibility measurements should be measured on an event basis rather than over a series of events i.e. a cropping season. The latter is too coarse a temporal resolution, so that variations within a season cannot be isolated and explained. At the field scale, an improvement in data collection of rainfall erosivity is required to allow better comparison with smaller spatial scales. Erosion data collection following a rainfall event from the field scale should be repeated soon afterwards at the micro-plot scale with a rainfall simulation using the same rainfall intensities and duration as the natural event. In conjunction with this, soil samples adjacent to the micro-plots should be taken for aggregate stability testing at the same time. Aggregates should be tested using raindrop impact methods using the same rainfall intensity and duration as that found at the field and micro-plot scale. A better comparison of field scale erosion can be made by covering the micro-plot test area during the field scale rainfall event. By covering the plot it ensures that the antecedent soil conditions between spatial scales are similar, i.e. moisture content and surface conditions (sealing, aggregate breakdown and presence of erosion features); allowing more robust spatial scale data comparison.

Nutrients and carbon losses should also be measured on an event basis and in addition to field studies, losses should also be carried out at the micro-plot scale. Surface soil samples should also be taken just before and after rainfall to assess in field changes in nutrient and carbon content. Surface residues have been shown to minimise soil erosion, however they have also been associated with increases in nutrient loss. The application of surface residues is an integral part of conservation soil management. More work is therefore needed on the effect of surface residues on nutrient and carbon loss. This research should also be performed at multiple spatial scales. This should also be repeated with differences in soil type as a result of the inconsistencies observed between site locations.

During this study the effect of treatment on subsurface flow (through-flow) and infiltration capacity and rates was not investigated. These are very important hydrological processes, which must be considered to gain a full understanding of tillage effects on soil, water, nutrient and carbon loss. It has been reported that nutrient and carbon losses in sub-surface flow can be significant (Zhao et al. 2001).

The way in which aggregates were prewetted was an important influence in the results gained, due to the destructive nature of the method. Therefore, different starting conditions should be tested. Whilst this is not novel research, comparison of results gained from different levels of pre-wetting with results of erosion from other spatial scales is original. It is expected that the use of different starting conditions would result in treatment differences occurring when using the raindrop impact method of aggregate stability, due to the fact that the effect of fast wetting causes slaking, and therefore aggregate breakdown before the aggregate stability tests are carried out. The relative effect from rain drop impact after slaking is thought to be minimal, simulating the effect of natural rainfall on soil under flooded conditions. By using different starting conditions, i.e. field capacity or air dry, the effect of tillage treatment on soil aggregates under different field conditions can be simulated. It is expected that as a result better comparison between spatial scales would be possible.

As stated previously, Six et al. (2000) expressed concern over the study of individual aggregates of a specified size to represent soil erodibility rather than tests on whole soil samples. Although work was carried out to address this concern, tests were carried out on aggregates from a whole soil rather than a soil sample in its entirety i.e. an undisturbed sample. Therefore, comparative tests should be carried out between aggregate stability methods on individual aggregates with splash cups using undisturbed soil samples. Samples for both should be tested using rain drop impact rather than immersion based methods to represent field conditions better.

It is known that the soil microbiological community are intrinsically linked to aggregate stability. Different microbial organisms inhabit specific niches determined in part, by pore and aggregate size (Killam 1994). It would be expected therefore, that the microbial community would differ with changes in aggregate size, and erodibility would also vary as a result. The observed results from this present study showed no significant differences in erodibility with changes in aggregate size. The use of aggregate stability in the assessment of erodibility will continue to be well used in erosion research. Therefore, more research is essential on the influence of soil management on the soil microbial community, and thus on the soil susceptibility to erosion.

All of the above should lead to a better understanding of the discrepancies observed between spatial scales. If field scale losses of soil, water, nutrient and carbon could be predicted, even on a relative basis, from small scale investigations the benefits would be vast. This is especially true at a time when policy makers and land managers are seeking effective ways to protect the environment whilst maintaining food, fibre and fuel security. This study has contributed to this debate, not least in demonstrating why conservation tillage is not a universal panacea to mitigate soil, water, nutrient and carbon losses.

7.6 Summary of Conclusions

The effectiveness of using tillage treatments in the mitigation of runoff generation and soil loss was found to be site specific. Results of runoff loss did not substantially differ between site locations; the opposite was true of soil loss. At Tivington, soil was found to be more erodible than at Loddington, primarily as a result of increased clay and organic matter content, and a gentler slope. It was noted that field data was associated with a high degree of variability due to natural topographic variation, limited replication and human error during field operations. Variability at the field scale was expected, two other spatial scales were therefore investigated. Results indicated that aggregate stability tests (<5mm) may be more reliable in assessing treatment effect on soil erodibility than field rainfall simulators trials ($1.5m^2$). It should be noted that the choice of method was crucial in the determination of aggregate stability and therefore treatment effects.

In summation, treatment effect on soil erodibility was site specific and dependant on the spatial scale being investigated. Caution should be used when using smaller scale assessment of soil erodibility to indicate field scale erosion as extrapolation from one scale to another can be fraught with error; an important consideration to modellers. The results suggest that there can be no universal method to mitigate against soil and water loss. Management should be tailored to be site specific and that this concept should be integrated into future policy.

8 References

Abrahams, A.D. & Parsons, A.J., 1991. Resistance to overland flow on desert pavement and its implications for sediment transport modelling. *Water Resource Research*, 27, 1827–1836.

Abu-Hamdeh, N.H., Abo-Quadais, S.A. & Othman, A.M., 2006. Effect of soil aggregate size on infiltration and erosion characteristics. *European Journal of Soil Science,* 57(5), 609-616.

Allton, K. E., 2006. Interactions between soil microbial communities, erodibility and tillage practices. Unpublished PhD thesis, Cranfield University, UK.

Amore, E., Modica, C., Nearing, M.A. & Santoro, V.C., 2004. Scale effect in USLE and WEPP application for soil erosion computation from three Sicilian basins. *Journal of Hydrology*, 293, 100-114.

Andraski, B.J., Daniel, T.C., Lowery, B. & Mueller, D.H., 1985. Runoff results from natural and simulated rainfall for tillage systems. *Transaction of the ASAE*, 28(4), 1219-1225.

Barthés B., Azontonde, A., Boli, B.Z., Prat, C. & Roose, E., 2000. Field-scale run-off and erosion in relation to topsoil aggregates stability in three tropical regions (Benin, Cameroon, Mexico). *European Journal of Soil Science*, 51, 485-495.

Barton, A. P., 1994. *Water chemistry as a factor influencing the erosivity of simulated rain.* Unpublished MSc. thesis, Cranfield University, UK.

Barton, A.P., Fullen, M.A., Mitchell, D.J., Hocking, T.J., Liu, L., Bo, Z.W., Zheng, Y. & Xia, Z.Y., 2004. Effects of soil conservation measures on erosion rates and crop productivity on subtropical Ultisols in Yunnan Province, China. *Agriculture, Ecosystems and Environment*, 104, 343-357.

Barzegar, A.E., Asoodar, M.A., Khadish, A., Hashemi, A.M. & Herbert, S.J., 2003. Soil physical characteristics and chickpea yield responses to tillage treatments. *Soil & Tillage Research*, 71, 49-57.

Basic, F., Kisic, I., Mesic, M., Nestroy, O. & Butorac, A., 2004. Tillage and crop management effects on soil erosion in central Croatia. *Soil & Tillage Research*, 78, 197-206.

Bedaiway, M.N. & Rolston, D.E., 1993. Soil surface densification under simulated high intensity rainfall. *SoilTechnology*, 6, 365-376.

Bearden, B.N. & Petersen L., 2000. Influence of arbuscular mycorrhizal fungi on soil structure and aggregate stability of a vertisol. *Plant and Soil,* 218, 173–183.

Bellamy, P., Bradley, I., Kirk, G., Lark, M. & Loveland, P., 2005. Carbon loses from soils across England and Wales 1978 – 2003. *Nature*, 437, 245-248.

Benites, J. & Vaneph, S., 2001. *1 World Congress on Conservation Agriculture: A Worldwide Challenge* [online]. Madrid (Spain), Conservation Agriculture: for a better environment. Available from: http://www.ecaf.org/Congress/Report.htm [Accessed 6 September 2006].

Bennett, O.L., Ashley, D.A. & Doss, B.D., 1964. Methods of reducing soil crusting to increase cotton seedling emergence. *Agronomy Journal*, 56, 162-165.

Bergsma, E. & Valenzuela, C.R., 1981. Drop testing aggregate stability of some soils near Merida, Spain. *Earth Surface Processes and Landforms*, 6, 309-18.

Boardman, J. & Favis-Mortlock, D.T., 1993. Simple methods of characterizing erosive rainfall with reference to the South Downs, southern England. *In*: S. Wicherek, ed. *Farm land erosion: in temperate plains environment and hills.* Elsevier Science Publishers B.V: Amsterdam.

Boix-Fayos. C., Martínez-Mena, M., Arnau-Rosalén, E., Calvo-Cases, A., Catillo, V. & Albaladejo, J., 2006. Measuring soil erosion by field plots: Understanding the sources of variation. *Earth-Science Reviews*, 78, 267-285.

Boix-Fayos, C., Martinez-Mena, M., Calvo-Cases, A., Arnau-Rosalén, E., Albaldajeo, J. & Castillo, V., in press. Sources and processes of measurement variability of experimental soil erosion field plots in semiarid Mediterranean conditions. *Earth Surface Processes and Landforms*, doi:10.1002/esp.1382.

Brady, N.C. & Weil, R.R., 2002. *The nature and properties of soils*. 13th ed. New Jersey: Prentice Hall.

Bruce-Okine, E. & Lal, R., 1975. Soil erodibility as determined by the raindrop technique. *Soil Science*, 119, 149-157.

Bryan, R.B., 1968. The development, use and efficiency of indices of soil erodibility. *Geoderma*, 2, 5-25.

Burney, J.R. & Edwards, L.M., 1994. Facilities for continuous monitoring of soil erosion in warm and cool season in Prince Edward Island, Canada. II. Erosion plots. *Catena*, 341-355.

Callebaut, F., Gabriels, D. & De Boodt, M., 1985. *Assessment of soil surface sealing and crusting*. Flanders Research Centre for Soil Erosion and Soil Conservation, Ghent, Belgium.

Cavalaris, C.K. & Gemtos, T.A., 2002. Evaluation of Four Conservation Tillage Methods in the Sugar Beet Crop. Agricultural Engineering International: the CIGR Journal of *Scientific Research and Development*, manuscript LW 01 008, vol 5, 1-24.

Cerdá, A., 2001. Effects of rock fragment cover on soil infiltration, interrill runoff and erosion. *European Journal of Soil Science*, 52, 59-68.

Cerdan, O., Le Bissonnais, Y., Govers, G., Lecomte, V., Van Oost, K., Couturier, A., King, C. & Dubreuil, N., 2004. Scale effect on runoff from experimental plots to catchments in agricultural areas in Normandy. *Journal of Hydrology*, 299, 4-14.

Chan, K.Y., 2001. An overview of some tillage impacts on earthworm population abundance and diversity – implications for functioning in soils. *Soil & Tillage Research,* 57, 179-191.

Charman, P. & Murphy. B., 2000. *Soils: their properties and management*. 2nd ed. Oxford: Oxford University Press.

Chenu, C., Le Bissonnais, Y. & Arrouays, D., 2000. Organic matter influence on clay wettability and soil aggregate stability. *Soil Science Society of America Journal,* 64, 1479-1486.

Chepil, W.S., 1962. A compact rotary sieve and the importance of dry sieving in physical soil analysis. *Soil Science Society of American Proceedings,* 26, 4-6.

Childs, E.C., 1940. The use of soil moisture characteristics in soil studies. *Soil Science*, 50, 239-252.

Collis-George, N. & Figueroa, B.S., 1984. 'The use of high energy moisture characteristics to assess soil stability'. *Australian Journal of Soil Research,* 22, 349-356.

Correchel, V., Bacchi, O.O.S., De Maria, I.C., Dechen, S.C.F. & Reichardt, K., 2006. Erosion rates evaluated by the ^{137}Cs technique and direct measurements on long-term runoff plots under tropical conditions. *Soil & Tillage Research*, 86, 199-208.

Darboux, F., Davy, Ph., Gascuel-Odoux, C. & Huang, C., 2001. Evolution of soil surface roughness and flowpath connectivity in overland flow experiments. *Catena*, 45, 125-139.

Davidson, J.M. & Evans, D.D., 1960. Turbidimeter technique for measuring the stability of soil aggregates in a water-glycerol mixture. *Soil Science Society of America Proceedings,* 24, 75-79.

DEFRA., 2003. *NITRATES – Reducing Water Pollution from Agriculture* [online]. London, Department for Environment, Food and Rural Affairs. Available from: http://www.defra.gov.uk/ENVIRONMENT/water/quality/nitrate/nitrogen.htm [Accessed 12 September 2006].

DEFRA., 2004a. *The first soil action plan for England: 2004-2006.* London: Department for Environment Food and Rural Affairs, (PB9411).

DEFRA., 2004b. *Nitrates – Reducing water pollution from agriculture. Nitrate Vulnerable Zones in England* [online]. London, Department for Environment, Food and Rural Affairs. Available from: http://www.defra.gov.uk/environment/water/quality/nitrate/nvz.htm [Accessed 11 September 2006].

DEFRA., 2005. *The development of national guidelines for sustainable soil management through improved tillage practises (SP0513)* [online]. London, Department for Environment, Food and Rural Affairs. Available from: http://www2.defra.gov.uk/research/Project_Data/More.asp?I=SP0513&M=KWS&V=ICE [Accessed 11 September 2006].

DEFRA., 2006. *Environmental Stewardship – Latest news* [online]. London, Department for Environment, Food and Rural Affairs. Available from: http://www.defra.gov.uk/erdp/schemes/es/default.htm [Accessed 10 September 2006].

Diaz-Zorita, M., Grove, J.H. & Perfect, E., 2002. Aggregation, fragmentation and structural stability measurements. *In:* Lal, R., ed. *Encyclopaedia of Soil Science*. UK: Taylor & Frances Ltd, 56-59.

Dickinson, A. & Collins, R., 1998. Predicting erosion and sediment yield at the catchment scale. *In:* Penning de Vries, F.W.T., Agus, F. & Kerr, J. (eds) *Soil Erosion at Multiple Scales: Principles and Methods for Assessing Causes and Impact*. Wallingford, UK: CAB International, 317–342.

DWI., 2003. *Information leaflet: Tap water – where does it come from and how is it made safe to drink?* [online]. London, Drinking Water Inspectorate.

Available from: http://www.dwi.gov.uk/consumer/faq/tapwater.htm [Accessed 21 July 2006].

EA., 2004a. *The State of Soils in England and Wales*. Bristol: Environmental Agency.

EA., 2004b. Soil, the hidden resource: Towards an Environment Agency strategy for soil protection, management and restoration: A consultation document. Environment Agency.

ECAF., 2004. *Fund of Friends of Conservation Agriculture* [online]. Brussels (Belgium), European Conservation Agriculture Federation. Available from: http://www.ecaf.org/FOCA.htm [Accessed 16 June 2006].

Einstein, H.A. & Barbarossa, N.L., 1951. River channel roughness. *Trans. Am. Soc. Civ. Eng*, 117, 1121– 1132.

Ellerbrock, R.H., Gerke, H.H., Bachmann, J. & Goebel, M.O., 2005. Composition of organic matter fractions for explaining wettability of three forest soils. *Soil Sci. Soc. Am. J.,* 69, 57-66.

Elliott, E.T., 1986. Aggregate structure and carbon, nitrogen, and phosphorus in native and cultivated soils. *Soil Sci. Soc. Am. J.* 50, 627–633.

Ellison, W.D., 1944. Studies of raindrop erosion. *Agricultural Engineering,* 25, 131-136, 181-182.

Emerson, W.W., 1954. The determination of the stability of soil crumbs. *Journal of Soil Science*, 5, 233-250.

Emerson, W.W., 1967. A classification of soil aggregates based on their coherence in water. *Australian Journal of Soil Research*, 5, 47-57.

EC., 2001. *Environment 2010: Our Future, Our Choice. The Sixth Environment Action Programme 2001-10*. Luxembourg: Office for Official Publications of the European Communities, (L-2985).

EUROPA., 2003. *CAP reform – a long-term perspective for sustainable agriculture* [online]. Brussels (Belgium), European Commission - Agriculture and Rural Development. Available from http://ec.europa.eu/agriculture/contact/index_en.htm [Accessed 12 September 2006].

EUROPA., 2006a. *Seven Environmental Thematic Strategies* [online]. Brussels (Belgium), European Commission on the Environment. Available from: http://ec.europa.eu/environment/newprg/strategies_en.htm [Accessed 4 November 2006].

EUROPA., 2006b. *A strategy to keep Europe's soils robust and healthy* [online]. Brussels (Belgium), European Commission on the Environment. Available from: http://ec.europa.eu/environment/soil/index.htm [Accessed 15 October 2006].

EUROPA., 2006c. Directive of the European Parliament and of the Council – establishing a framework for the protection of soil and amending Directive 200/35/EC [online]. Brussels, Commission of the European Communities. Available from: http://ec.europa.eu/environment/soil/pdf/com_2006_0232_en.pdf [Accessed 22 August 2006].

EUROPA., 2006d. *Nature & Biodiversity - Habitats Directive* [online]. Brussels (Belgium), European Commission on the Environment. Available from: http://ec.europa.eu/environment/nature/nature_conservation/eu_nature_legislation/habitats_directive/index_en.htm [Accessed 30 August 2006].

Evans, R., 1993. Extent, frequency and rates of rilling of arable land in localities in England and Wales. *In*: Wicherek, S. (ed) *Farm land erosion: in temperate plains environment and hills*. Amsterdam: Elsevier Science Publishers B.V., 177-190.

Evans, R.,1996. *Soil erosion and its impacts in England and Wales*. Friends of the Earth: London

Evans, R. & Boardman, J., 1994. Assessment of water erosion in farmers' fields in the UK. In: Rickson, R.J. (ed.) *Conserving Soil Resources: European Perspectives*. Wallingford, UK: CAB International, 13–24.

Eynard, A., Schumaucher, T.E., Kohl, R.A. & Malo, D.D., 2006. *Soil wettability relationships with soil organic carbon and aggregate stability* [online]. USA, South Dakota State University. Available from: http://crops.confex.com/crops/wc2006/techprogram/P16051.HTM [Accessed 15 August 2006].

Farres, P.J. & Cousen, S.M., 1985. An improved method of aggregate stability measurement. *Earth Surface Processes and Landforms*, 10, 21-329.

Ferry, D.M. & Olsen, R.A., 1975. Orientation of clay particles as it relates to crusting of soils. *Soil Science*, 120, 367-375.

Fiener, P., Auerswald, K. & Weigand, S., 2005. Managing erosion and water quality in agricultural watersheds by small detention ponds. *Agriculture, Ecosystems and Environment*, 110, 132-142.

Foster, G.R., 1982. Modelling the erosion process. *In*: Haan, C.T., Johnson, H.D. & Brakensiek, D.L., ed. *Hydrologic modelling of small watersheds*, ASAE Monogr., vol. 5. *Am. Soc. Agric. Eng*., St. Joseph, Michigan, 295– 380.

Fournier, F., 1972. Soil conservation. *Nature & Environment Series, no. 5,* Council of Europe.

Fullen, M.A. & Catt, J.A., 2004. Soil *management: problems and solutions*. Arnold: London.

Gimeno-Garcia, G., Rickson, R.J. and Rubio, J-L. (in preparation). The effect of simulated fire on soil aggregate stability. Paper submitted to the *European Journal of Soil Science*, 2006.

Gobin, A., Jones, R., Kirkby, M., Campling, P., Govers, G., Kosmas, C. & Gentile, A.R., 2004. Indicators for pan-European assessment and monitoring of soil erosion by water. *Environmental Science & Policy*, 7(1), 25-38.

Gómez, J.A., Nearing, M.A., Giráldez, J.V. & Alberts, E.E., 2001. Analysis of sources of variability of runoff volume in a 40 plot experiment using a numerical model. *Journal of Hydrology*, 248, 183-197.

Gómez, J.A. & Nearing, M.A., 2005. Runoff and sediment losses from rough and smooth soil surfaces in a laboratory experiment. *Catena*, 59, 253-266.

Govers, G., 2004. Personal communication. Laboratory for Experimental Geomorphology, K.U Leuven.

Govers, G. & Poesen, J., 1988. Assessment of the interrill and rill contributions to total soil loss from an upland field plot. *Geomorphology*, 1, 343-354.

Greenland, D.J., Rimmer, D. & Payne, D., 1975. Determination of the structural stability class of English and Welsh soils, using a water-coherance test. *Journal of Soil Science*, 26, 303.

Grieve, I.C., 1979. Soil aggregate stability tests for the geomorphologist. *British Geomorphological Research Group (BGRG) Technical Bulletin* No. 25. Geoabstracts, Norwich.

Gunn, R. & Kinzer, G., 1949. The terminal velocity of fall for water droplets in stagnant air. *Journal of Meteorology,* 6, 243-248.

Hartanto, H., Prabhu, R., Widayat, S.E. & Asdak, C., 2003. Factors affecting runoff and soil erosion: plot-level soil loss monitoring for assessing sustainability of forest management. *Forest Ecology and Management*, 180, 361-374.

Helming, K., 2001. Wind speed effects on rain erosivity. In: Stott, D.E., Mohtar, R.H. & Steinhardt, G.C. ed. Sustaining the Global Farm from the 10th International Soil Conservation Organization Meeting, 24-29 May 1999, Purdue University USA. Indiana: National Soil Erosion Research Laboratory, 771-776.

Herrick, J.E., Whitford, W.G., de Soyza, A.G., Van Zee, J.W., Havstad, K.M., Seybold, C.A. & Walton, M., 2001. Field soil aggregate stability kit for soil quality and rangeland health evaluations. *Catena*, 44, 27-35.

Hill, P. & Mannering, J., 1995. *Conservation Tillage and Water Quality* [online]. Indiana (USA), Purdue University (School of Agriculture). Available from: https://secure.agriculture.purdue.edu/store/item.asp?itemID=15797&ListType=&subcatID=88&catID=30 [Accessed 1 September 2006].

Hjulström F. (1935) Studies on the morphological activity of rivers as illustrated by the River Fyris. *Bulletin, Geological Institute of Upsala,* volume 25, 221-527.

Hodgson, J.M. (ed.), 1997. *Soil survey field handbook: describing and sampling soil profiles,* 3rd ed. Technical monograph No 5. Soil Survey and Land Research Centre, UK

Holland, J.M., 2004. The environmental consequences of adopting conservation tillage in Europe: reviewing the evidence. *Agriculture, Ecosystems and Environment,* 103, 1-25.

Hudson, N.W., 1957. The design of field experiments on soil erosion. *Jour. Agric. Eng. Research*, 2, 56-65. *In*: Morgan, R.P.C., 1986. *Soil erosion and its control*. New York: Van Nostrand Reinhold Company.

Hudson, N.W., 1964. The flour pellet method for measuring the size of raindrops. Research Bulletin, No.4, Dept. Conservation and Extension, Salisbury, Zimbabwe.

Hudson, N.W., 1965. *The influence of rainfall on the mechanics of soil erosion with particular reference to Southern Rhodesia*. MSc Thesis, University of Cape Town.

Hudson, N.W., 1981. *Soil conservation*. 2nd ed. Batsford Limited: London.

Hudson, N.W., 1993. Field measurement of soil erosion and runoff. *FAO Soils Bulletin*, 68. Silsoe Associates, Amptill, Bedford, UK.

Hudson, N.W., 1995. *Soil conservation*. 3rd ed. Batsford Limited: London.

Imeson, A.C. and H. Lavee. 1998.Soil erosion and climate change: the transect approach & the influence of scale. *Geomorphology,* 23, 219-227.

Isensee, A.R. & A.M. Sadeghi., 1999. Quantification of runoff in laboratory-scale chambers. *Chemosphere,* 38, 1733–1744.

Jastrow, J.D. & Miller, R.M., 1991. Assessment of the effects of the biota on soil structure: methods for assessing the effects of biota on soil structure. *Agriculture, Ecosystems and Environment*, 34, 279-303.

Jensen Global, 2006. *Needles : Teflon* [online]. Santa Barbara (USA), Jensen Global – Dispending Solutions. Available from: http://jensenglobal.com/cgi-bin/jgd1/contact.html [Accessed 5 January 2006].

Journeaux, P., 2003. Overview of the linkages between agricultural activities, water pollution, and water use. *In: OECD expert meeting on agricultural water quality and water use indicator, 7-10 October 2003 Gyeongju (Korea).* New Zealand: Policy Division, Ministry of Agriculture and Forestry, 1-11.

Kamalu, C., 1993. *Soil erosion on road shoulders*. Thesis (PhD). Cranfield University.

Kandeler, E. & Murer, E., 1993. Aggregate stability and soil microbial processes in a soil with different cultivation. *Geoderma*, 56, 503-513.

Karunatilake, U.P & van Es, H.M., 2002. Rainfall and tillage effect on soil structure after alfalfa conversion to maize on a clay loam soil in New York. *Soil & Tillage Research*, 67, 135-146.

Kay, B.D., 1990. Rates of change of soil structure under different cropping systems. *Adv. Soil Sci*., 12, 1–52.

Kemper, W.D. & Rosenau, R.C., 1984. Soil cohesion as affected by time and water content. *Soil Sci, Soc, Am, J*, 48, 1001-1006.

Kemper, W.D. & Rosenau, R.C., 1986. Aggregate stability and size distribution. *In:* Klute, A. *Methods of soil analysis: part 1. Physical and mineralogical methods*, 2nd, ed. Madison, WI: ASA and SSSA.

Kiem, R. & Kandeler, E., 1997. Stabilization of aggregates by the microbial biomass as affected by soil texture and type. *Applied Soil Ecology*, 5, 221-230.

Killham, K., 1994. *Soil Ecology*. UK: Cambridge University Press.

King, J.A., Bradley, R.I. & Harrison, R., 2005. Current trends of soil organic carbon in English arable soil. *Soil Use & Management,* 21(2), 189-195.

Kladivko, E.J., 2001. Tillage systems and soil ecology. *Soil & Tillage Research*, 61, 61-76.

Lado, N., Ben-Hur, M. & Shainberg, I., 2004. Soil wetting and texture effects on aggregate stability, seal formation and erosion. *Soil Sci. Soc. Am. J.*, 68, 1992-1999.

Laflan, J.M. & Colvin, T.M., 1981. Effect of crop residue on soil loss from continuous row cropping. *Transaction of the American Society of Agricultural Engineers,* 24, 605-609.

Lal, R., 1976. Soil erosion problems on an Alfisol in western Nigeria and their control. IITA Monograph No.1. Cranfield University, Silsoe, UK

Larink, O., Werner, D., Langmaack, M. & Schrader, S., 2001. Regeneration of compacted soil aggregates by earthworm activity. *Biol. Fertil. Soils*, 33, 395–401.

Lavelle, P., Decaens, T., Aubert, M., Barot, S., Blouin, M., Bureau, F., Margerie, P., Mora, P., & Rossie, J.P., 2006. Soil invertebrates and ecosystem services. *European Journal of Soil Biology*, 1-13.

Laws J.O., 1941. Measurement of the fall velocity of water drop and rain drops. *Trans. Am. Geophys. Union*, 34, 452-459.

Laws, J.O. & Parsons, D.A., 1943. The relation of raindrop size to intensity. *Trans. Am. Geophys. Un*., 24, 452-60.

Le Bissonnais, Y., 1996a. Soil characteristics and aggregate stability. *In:* Agassi, M., ed. *Soil erosion, conservation and rehabilitation*. New York: Marcel Dekker, 41-60.

Le Bissonnais, Y., 1996b. Aggregate stability and assessment of soil crustability and erodibility. 1. Theory and methodology. *Eur. J. Soil Sci, 47, 425-43.*

Le Bissonnais, Y. & Singer, M.J., 1992. Crusting, runoff, and erosion response to soil water content and successive rainfalls. *Soil Science Society of American Bulletin,* 56(6), 1898-1903.

Le Bissonnais, Y. & Arrouays, D., 1997. Aggregate stability and assessment of soil crustability and erodibility: II. Application to humic loamy soils with various organic carbon contents. *European Journal of Soil Science*, 48, 39-48.

Le Bissonnais, Y., Cros-Caypt, S., & Gascuel-Odoux, C., 2002. Topographic dependence of aggregate stability, overland flow and sediment transport, *Agronomie*, 22, 489-501.

Leake, A., 2005. *Allerton Project* [online]. Loddington, Allerton Research & Educational Trust. Available from: http://www.allertontrust.org.uk/frame.html [Accessed 4 September 2006].

Legout, C., Leguédois, S. & Le Bissonnais, Y., 2005. Aggregate breakdown dynamics under rainfall compared with aggregate stability measurements. *European Journal of Soil Science*, 56(2), 225-238.

Levy, G.J. & Mamedov, A.I., 2002. High-energy-moisture-characteristic aggregate stability as a predictor for seal formation. *Soil Sci. Soc. Am. J.*, 66, 1603-1609.

Linden, D.R., D.M. Van Doren, Jr., & R.R. Allmaras., 1988. A model of the effects of tillage-induced soil surface roughness on erosion. *In: Tillage and traffic in crop production*, Proceedings of the 11th ISTRO Conference, 11-15th July 1988, Edinburgh, Scotland, 373-378.

Lindstrom, M. J. & Onstad, C. A., 1984. Influence of tillage systems on soil physical parameters and infiltration after planting. *Journal of Soil & Water Conservation*, 39, 149-152.

Llewellyn, C., 2006. Practical soil protection and stabilization in Mediterranean viticulture. Unpublished MPhil Thesis. Cranfield University.

Loch, R.J., 1994. Structure breakdown on wetting. *In*: So, H.B., Smith, G.D., Schafer, B.M., & Loch, R.J. (eds) Sealing crusting and hardsetting soils: productivity and conservation. *Australian Soc. Soil Sci*. Brisbane, Australia: Old Branch, 113–132.

Lovell, C.J. & Rose, C.W., 1988. Measurement of soil aggregate settling velocities, II. Sensitivity to sample moisture content and implications for studies of structural stability. *Australian Journal of Soil Research*, 26(1), 73-85.

Low, A.J., 1954. The study of soil structure in the field and the laboratory. *J. Soil Sci*., 5, 57-74.

Lyles, L., Dickerson, J.D., & Disrud, A.D., 1970. Modified rotary sieve for improved accuracy. *Soil science,* 109, 207-210.

Lyles, L., Dickerson, J.D., & Schmeidler, N.F., 1974. Soil detachment from clods by rainfall: effects of wind, mulch cover and essential soil moisture. *Trans. Am. Soc. Agric. Engin.,* 17 (4), 697-700.

Mandal, U.K., Rao, K.V., Mishra, P.K., Vittal, K.P.R., Sharma, K.L., Narsimlu, B. & Venkanna, K., 2005. Soil infiltration, runoff and sediment yield from a shallow soil with varied stone cover and intensity of rain. *European Journal of Soil Science*, 56(4), 435-444.

McCalla, T.M., 1944. Water drop method of determining stability of soil structure. *Soil Sci*, 58, 117-21.

McLaughlin, A. & Mineau, P., 1995. The impact of agricultural practices on biodiversity. *Agriculture, Ecosystems & Environment*, 55, 201-212.

Met Office. (2006) *The Beaufort Scale* [online]. Devon, Met Office. Available from: http://www.metoffice.com/education/secondary/students/beaufort.html [Accessed 26 September 2006].

Mueller, D. H., Wendt, D. H. & Daniel, T.C., 1984. Soil and water losses as affected by tillage and manure application. *Soil Science Society of America*, 48, 896-900.

Meyer, L.D. & Wischmeier, W.M., 1969. Mathematical simulation of the process of soil erosion by water. *Transactions of the American Society of Agricultural Engineers*, 12, 754-8, 762.

Middleton, H. E., 1930. Properties of Soils Which Influence Erosion. *U.S. Dep. Agric., Tech. Bull.*, 178, 16.

Miller, N., 2004. *Phosphorus mobilisation from European soils by overland flow.* Unpublished PhD thesis, Cranfield University, UK.

Moldenhauer, W. C., Lovely, W. G., Swanson, N. P. & Currence, H. D., 1971. Effect of row grades and tillage systems on soil and water losses. *Journal of Soil & Water Conservation*, 26 (5), 193-195.

Morgan, R.P.C., Quinton, J.N., Smith, R.E., Govers, G., Poesen, J.W.A., Auerswald, K., Chisci, G., Torri, D. & Styczen, M.E., 1998. The European soil erosion model (EUROSEM): a dynamic approach for predicting sediment transport from fields and small catchments. *Earth Surface Processes and Landforms*, 23, 527-544.

Morgan, R.P.C.,1980. Field studies of sediment transport by overland flow. *Earth Surface Processes*, 3, 307-16.

Morgan, R.P.C. & Rickson, R.J., 1988. *Approached to modelling the effects of vegetation on soil erosion by water*. *In:* Morgan, R.P.C. and Rickson, R.J., ed. *Erosion Assessment and Modelling*. Proceedings of a workshop held in Brussels, Belgium 2-3 December 1986, sponsored by the Commission of the European Communities, Directorate-General for Agriculture, Coordination of Agricultural Research. EUR 10860 EN, 73-91.

Morgan, R.P.C., 1995. *Soil erosion and conservation*. 2rd ed. Longman Publishing Group.

Morgan, R.P.C., 2005. *Soil erosion and conservation*. 3rd ed. Oxford: Blackwell Publishing.

Mutchler, C.K. & Hermsmeier, L.F., 1965. A review of rainfall simulators. *ASAE Trans.* 8, 63-65.

Mwendera, E.J. & Feyen, J., 1994. Effects of tillage and rainfall on soil surface roughness and properties. *Soil Technology*, 7(1), 93-103.

Myers, J.L & Wagger, M.G. (1996) Runoff and sediment loss from three tillage systems under simulated rainfall. *Soil & Tillage Research*, 39, 115-129.

NCDC, 2006. *Boscastle and North Cornwall Floods* [online]. Wadebridge, North Cornwall District Council. Available from: http://www.ncdc.gov.uk/index.cfm?articleid=9675 [Accessed 4 September 2006].

Ndiaye, B., Esteves, M., Vandervaere, J.P., Lapetite, J.M. & Vauclin, M., 2005. Effect of rainfall and tillage direction on the evolution of surface crusts, soil hydraulic properties and runoff generation for a sandy loam soil. *Journal of Hydrology*, 307, 294-311.

Nearing, M.A., Govers, G. & Norton, L.D., 1999. Variability in soil erosion data from replicated plots. *Soil Science Society of America Journal*, 63, 1829-1835.

Nelson, P., 2006. *An assessment of conservation tillage in the EU*. Unpublished M.Sc. thesis, Cranfield University.

NERC (Natural Environment Research Council)., 1975. *Flood Studies Report,* Vol 1-5 and Flood Studies Supplementary Reports. London: Whitefriars Press Ltd.

Ngatunga, E.L.N., Lal, R. & Uriyo, A.P., 1984. Effects of surface management on runoff and soil erosion from some plots at Mlingano, Tanzania. *Geoderma*, 33(1), 1-12.

Olayemi, F.F & Yadav, R.C., 1983. Rainfall simulator for tillage research in the tropics. *Soil & Tillage Research*, 3, 397-405.

Oldeman, L.R., Hakkeling, R.T.A. & Sombroek, W.G.,1991. *World map of the status of human-induced soil degradation: an explanatory note*. Second revised edition. ISRIC/UNEP.

Ollesch, G. & Vacca, A., 2002. Influence of time on measurement results of erosion plot studies. *Soil & Tillage Research*, 67, 23-29.

Owens, L.B., Malone, R.W., Hothem, D.L., Starr, G.C. & Lal, R., 2002. Sediment carbon concentration and transport from small watersheds under various conservation tillage practices. *Soil & Tillage Research,* 67(1), 65-73.

Owens, P.N., Rickson, R.J., Clarke, M.A., Dresser, M., Deeks, L.K., Jones, R.J.A., Woods, G.A., Van Oost, K. and Quine, T.A. 2006. Review of the existing knowledge base on magnitude, extent, causes and implications of soil loss due to wind, tillage and co-extraction with root vegetables in England and Wales, and recommendations for research priorities. NSRI Report to DEFRA, Project SP08007, Cranfield University, UK.

Oygarden, L., 2003. Rill and gully development during an extreme winter runoff event in Norway. *CATENA*, 50(2-4), 217-242.

Palmer, R.S, 1964. The influence of a thin water layer on waterdrop impact forces. *International Association of Scientific Hydrology Publication,* 65, 141-148.

Palmer, W.C., 1965. *Meteorological drought*. Research Paper No. 45, U.S. Department of Commerce Weather Bureau, Washington, D.C.

Paustian, K., Andren, O., Janzen, H.H., Lal, R., Smith, P., Tian, G., Tissen, H., Van Noordwijk, M. & Woomer, P.L., 1997. Agricultural soils as a sink to mitigate CO_2 emissions. *Soil Use & Management,* 13(4), 229-304.

Poesen, J., Ingelmo-SaÂnchez, F. & MuÈcher, H., 1990. The hydrological response of soil surfaces to rainfall as affected by cover and position of rock fragments in the top layer. Earth *Surface Processes and Landforms*, 15, 653-671.

Poesen, J.W.A., 1992. Mechanisms of overland-flow generation and sediment production on loamy and sandy soils with and without rock fragments. *In*: Parsons, A.J. & Abrahams, A.D., ed. *Overland flow: hydraulics and erosion mechanics*. London: UCL Press, 275-305.

Poesen, J. & Lavee, H., 1994. Rock fragments in top soils: significance and processes. *Catena*, 23, 1-28.

Pulleman, M.M., Siz, J., van Breemen, N. & Jongmas, A.G., 2005. Soil organic matter distribution and microaggreate characteristics as affected by agricultural management and earthworm activity. *European Journal of Soil Science*, 56, 453-467.

Quinton, J., 2005. *Field testing of mitigation options - PEO206 MOPS (page 33) in Co-ordination Project Descriptions* [online]. Aberystwyth (Wales), Institute of Grassland and Environmental Research. Available from: http://www.iger.bbsrc.ac.uk/DEFRA_Phosphorus/documents/Projectswithabstractdesc_003.doc [Accessed 3 September 2006].

Quinton, J.N., Catt, J.A. & HESS, T.M., 2001. The selective removal of phosphorus from soil: is event size important? *Journal of Environmental Quality,* 30, 538-545.

Quinton, J.N., Catt, J.A., Wood, G.A. & Steer, J., 2006. Soil carbon losses by water erosion: Experimentation and modelling at field and national scales in the UK. *Agriculture, Ecosystems and Environment*, 112, 87-102.

Rahim, S.E., 1990. *Spatial and seasonal variations of soil erodibility*. Unpublished PhD thesis, Cranfield University, UK.

Reed, A.H., 1979. Accelerated erosion of arable soils in the United Kingdom by rainfall and runoff. *Outlook on Agriculture*, 10, 41–48.

Rickson, R.J., 1994. *Conserving soil resources: European perspectives*. Page 94-104. Oxon: CAB International.

Rickson, R.J., 2005. Personal communication. School of Applied Sciences, Cranfield University.

Rickson, R.J. (2006) Management of sediment production and prevention in river catchments: A matter of scale? *In:* Collins, A.J. & Owens, P., ed. *Soil erosion and sediment redistribution in river catchments: measurement, modelling and management in the 21st century*. CAB International, 228-238.

Robinson, D.A. & Blackman, J.D., 1989. Soil erosion, soil conservation and agricultural policy for arable land in the U.K. *Geoforum*, 20(1), 83-92.

Robinson, D.A. & Phillips, C.P., 2001. Crust development in relation to vegetation and agricultural practice on erosion susceptible, dispersive clay soils from central and southern Italy. *Soil & Tillage Research*, 60(1-2), 1-9.

Roldán, A., Salinas-García, J.R., Alguacil, M.M., & Caravaca, F., 2005. Changes in soil enzyme activity, fertility, aggregation and C sequestration mediated by conservation tillage practices and water regime in a maize field. *Applied Soil Ecology*, 30, 11-20.

Romero-Díaz, A., Cammeraat, L.H., Vacca, A. & Kosmoas, C., 1999. Soil erosion at three experimental sites in the Mediterranean. *Earth Surface Processes and Landforms*, 24, 1243-1256.

Rowell, D.L., 1994. Soil science: methods and applications. London: Prentice Hall

SCC, 2006. *The Parrett Catchment Project – 'A Future when it rains'* [online]. Taunton, Somerset County Council. Available from: http://www.somerset.gov.uk/somerset/ete/pcp/ [Accessed 4 September 2006].

Sharpley, A.N. & Smith, S.J., 1990. Phosphorus transport in agricultural runoff: the role of soil erosion. *In*: Boardman, J., Foster, I.D.L. & Dearing, J.A., ed. *Soil erosion on agricultural land*. Chichester: John Wiley & Sons Ltd, 351-66.

Schiettecatte, W., Jin, K., Yao, Y., Cornelis, W.M., Lu, J., Wu, H., Verbist, K., Cai, D., Gabriels, D., & Hartmann, R., 2005. Influence of simulated rainfall on physical properties of a conventionally tilled loess soil. *Catena*, 64, 209-221.

Simmons, R., 1998. *The effect of polyacrilamide based soil conditioners on structural sealing at a sub-process level*. Unpublished PhD thesis, Canterbury Christ Church University, Kent.

Six, J., Elliott, E., & Paustian, K., 2000. Soil Structure and Soil Organic Matter: II. A Normalized Stability Index and the Effect of Mineralogy. *Soil Science Society of America Journal*, 64, 1042-1049.

Six, J., Bossuyt, H., Degryze, S. & Denef, K., 2004. A history of research on the link between (micro) aggregates, soil biota, and soil organic matter dynamics. *Soil & Tillage Research*, 79, 7-31.

Slattery, M.C & Bryan, R.B., 1994. Surface seal development under simulated rainfall on an actively eroding surface. *Catena,* 22, 17-34.

SMI., 2005. Case Study 1 – Reduced cultivation with disc harrows or direct drill [online]. Chester, Soil Management Initiative. Available from: http://www.smi.org.uk/casestudies/study01.html [Accessed 6 June 2006].

SMI., 2005b. *Case Study 4 – Progressive adoption of reduced cultivation* [online]. Chester, Soil Management Initiative. Available from: http://www.smi.org.uk/casestudies/study04.html [Accessed 6 June 2006].

Smith, D.L.O., 1987. Measurement, interpretation and modelling of soil compaction. *Soil Use and Management*, 3, 87-93. *In:* Rowell, D.L., 1994. *Soil science: methods and applications.* London: Prentice Hall, 73-76.

Smith, P., 2004. Soils as carbon sinks: the global context. *Soil Use and Management*, 20, 212-218.

Smith, R.M. & Cernuda, C.F, 1951. Some applications of water drop stability testing of tropical soils of Puerto Rico. *Soil Sci*, 71, 337-45.

Smith, P. & Conen, F., 2004. Impacts of land management on fluxes of trace greenhouse gases. *Soil Use and Management*, 20, 225-263.

Smith, R. & Quinton, J.N., 2000. Dynamics and scale in simulating erosion by water. *In:* Schmidt, J., ed. *Soil Erosion: Application of Physically Based Models*. Berlin, Germany: Springer, 282–294.

Somaratne, N.M. & Smettem, K.R.J., 1993. Effect of cultivation and raindrop impact on the surface hydraulic properties of an Alfisol under wheat. *Soil & Tillage Research,* 26, 115-125.

Stocking, M.A. & Elwell, H.A.,1973. Prediction of sub-tropical storm soil losses from field plot studies. *Agricultural Meteorology*, 12, 193-201

Stoosnijder, L., 2005. Measurement of erosion. Is it possible? *Catena*, 64, 162-173.

Stuttard, M.J., 1985. *The water drop test of soil erodibility*. Pamphlet. Silsoe: Cranfield University, Silsoe, UK

Takken, I., Govers, G., Jetten, V., Nachtergaele, J., Steegen, A. & Poesen, J., 2001. Effects of tillage on runoff and erosion patterns. *Soil & Tillage Research*, 61 (1-2), 55-60.

Teixeira, P.C. & Misra, R.K., 1997. Erosion and sediment characteristics of cultivated forest soils as affected by the mechanical stability of aggregates. *Catena,* 30 (2-3), 119-134.

Tisdall, J.M. & Oades, J.M., 1982. Organic matter and water-stable aggregates in soils. *Journal of Soil Science*, 33, 141-163.

Tiulin, A.F., 1928. Questions on soil structure: II. Aggregate analysis as a method for determining soil structure. Perm. Agr. Exp. Sta. Div. Agr. Chem. Report 2: 77-122. *In*: Low A.J., 1954. The study of soil structure in the field and the laboratory. *Journal of Soil Science,* 5(1), 57-74.

Troeh, F.R. & Thompson, L.M., 1993. *Soils and Soil Fertility*. New York: Oxford University Press.

Turley, D.B., Phillips, M.C., Johnson, P., Jones, A.E., & Chambers, B.J., 2003. Long-term straw management effects on yields of sequential wheat (*Triticum aestivum* L.) crops in clay and silty clay loam soils in England. *Soil & Tillage Research*, 71, 59-69.

Uri, N.D.,1999. Factors affecting the use of conservation tillage in the United States. *Water, Air and Soil Pollution*, 116, 621-638.

Uri, N.D., Atwood, J.D. & Sanabria, J., 1998. The environmental benefits and costs of conservation tillage. *Science of the Total Environment*, 216, 13-32.

Usón, A. & Ramos, M.C., 2001. An improved rainfall erosivity index obtained from experimental interrill soil losses in soils with a Mediterranean climate. *Catena*, 34, 293-305.

Van Bavel, C.H.M., 1949. Mean weight diameter of soil aggregates as a statistical index of aggregation. *Soil Sci Soc Am Proc.,* 14, 20-23.

Van Bavel, C.H.M., 1952. Compact wet sieving apparatus for soil aggregate analysis. *Agronomy J.,* 44, 97-102.

Van-Camp. L., Bujarrabal, B., Gentile, A-R., Jones, R.J.A., Montanarella, L., Olazabal, C. & Selvaradjou, S.K., 2004a. Reports of the Technical Working Groups Established under the Thematic Strategy for Soil Protection. EUR 21319 EN/1, 872 pp. Office for Official Publications of the European Communities, Luxembourg. Volume 1

Van-Camp. L., Bujarrabal, B., Gentile, A-R., Jones, R.J.A., Montanarella, L., Olazabal, C. & Selvaradjou, S.K., 2004b. Reports of the Technical Working Groups Established under the Thematic Strategy for Soil Protection. EUR 21319 EN/2, 872 pp. Office for Official Publications of the European Communities, Luxembourg. Volume 2

Vere, D., 2005. Research into conservation tillage for dryland cropping in Australia and China. Impact assessments areas number 33, Australian Centre for International Agricultural Research, Canberra.

Verheijen, F.G.A., Bellamy, P.H., Kibblewhite, M.G. & Gaunt, J.L., 2005. Organic carbon ranges in arable soils of England and Wales. *Soil Use and Management*, 21, 2-9.

Wan, Y. & El-Swaify, S.A., 1999. Runoff and soil erosion as affected by plastic mulch in a Hawaiian pineapple field. *Soil & Tillage Research*, 52, 29-35.

Warburton, J. 2005. Wind-splash erosion of bare peat on UK upland moorlands. *Catena,* 52, 191–207.

Wendt, R.C., Alberts, E.E. & Hjelmfelt, Jr. A.T., 1986. Variability of runoff and soil loss from fallow experimental plots. *Soil Science Society of America Journal*, 50, 730-736.

WEPP., 2006. *WEPP Software – Water Erosion Prediction Project* [online]. Indiana (USA), National Soil Erosion Research Lab (Agricultural Research Service, United States Department of Agriculture). Available from: http://topsoil.nserl.purdue.edu/nserlweb/weppmain/ [Accessed 10 September 2006].

Wickenkamp, V., Duttmann, R. & Mosimann, T., 2000. A multi-scale approach to predicting soil erosion on cropland using empirical and physically based soil erosion models in a Geographic Information System. *In:* Schmidt, J., ed. *Soil Erosion: Application of Physically Based Models*. Berlin, Germany: Springer, 109–133.

Williams, J.D., 2004. Effects of long-term winter wheat, summer fallow residue and nutrient management on field hydrology for a silt loam in north-central Oregon. *Soil & Tillage Research*, 75, 109-119.

Wischmeier, W.H. & Smith, D.D., 1978. *Predicting rainfall erosion losses: a guide to conservation planning*. Agriculture Handbook No. 537, Washington, DC: Science and Education Administration, U.S. Dept. of Agriculture.

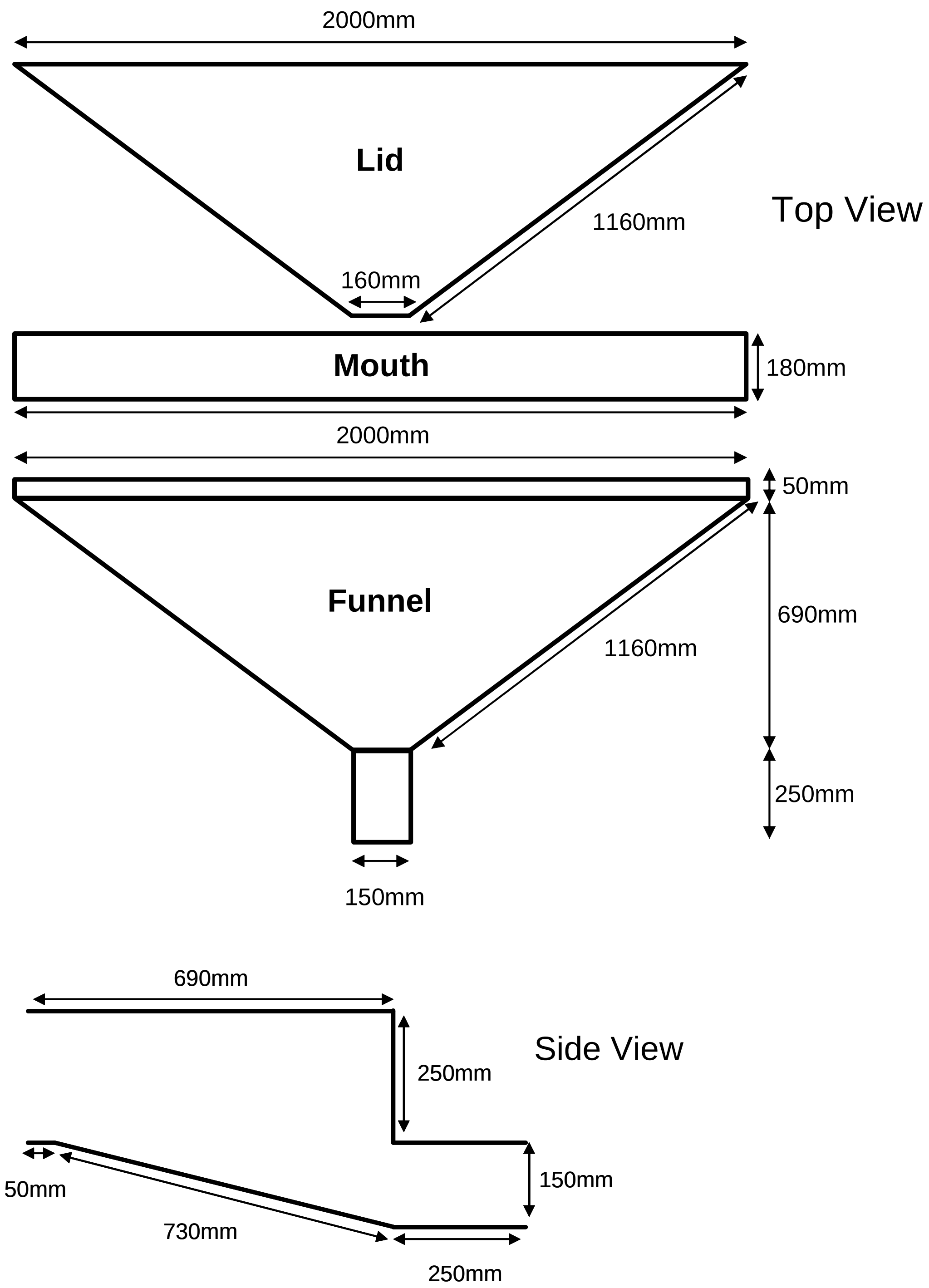
Yoder, R.E., 1936. A direct method of aggregate analysis of soil and a study of the physical nature of erosion losses. *Journal of Am. Soc. Agron*, 28, 337-351.

Zhao, S.L., Gupta, S.C., Huggins, D.R. & Moncrief, J.F., 2001. Tillage and nutrient source effects on surface and subsurface water quality at corn planting, *Journal of Environmental Quality*, 30, 998-1008.

Zheng, F., He, X., Gae, X., Zhang, C. & Tang, K., 2005. Effects of erosion patterns on nutrient loss following deforestation on the Loess Plateau of China. *Agriculture, Ecosystems and Environment*, 108, 85-97.

Zobisch, M.A., Klingspor, P. & Oduor, A.R., 1996. The accuracy of manual runoff and sediment sampling from erosion plots. *Journal of Soil and Water Conservation*, 51(3), 231.

Appendix A Erosion plot funnel layout (based on a drawing from Ceri Llewellyn, NSRI)



Appendix B Methods of runoff, sediment and soil analysis

A variety of analyses were carried out on runoff, sediment and soil, Table G.1 gives an overview of these methods.

Table B.1 Methods of runoff, sediment and soil analysis

Determinand	Method
Runoff	
Soluble nitrate & Soluble phosphorus	Atomic absorption spectrometry
Soluble potassium	Corning 4000 flame photometer
*Total dissolved organic carbon	High temperature combustion and Non-Dispersive Infared (NDIR) detection
*Total suspended sediment	Oven dry evaporation of an aliquot sample
Soil & Sediment	
Bulk density	Density rings
Soil moisture content	Gravimetric and volumetric
*Particle size distribution	Pipette sedimentation method
*Organic matter content	Wet oxidation / Walkley Black
*Total nitrogen	Dumas
Total carbon	Vario EL determination (acid digest
*Total Potassium & Total phosphorus	Acid digestion and determination through Inductively Coupled Plasma Emission Spectroscopy (ICP-OES).

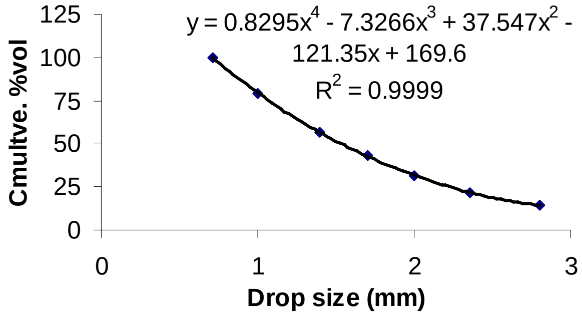
***indicate methods performed by Natural Resource Management Ltd, Bracknell, UK**

Appendix C Drop size calibration of field rainfall simulator (adapted from Simmons 1998)

Sieve size	No. of pellets	Mass of pellets (g)	Mass of single pellet (g)	Mass of single drop (g)	Size Drop (mm)	Volume of single drop (mm^3)	Total vol. of drops (mm^3)	Cumulative vol.	Cumulative % vol.	% of total sample drop volume	Vol. rainfall in size class (mm^3)	No. of drops of size class in rain event	Estimated drop velocity ($m\ s^{-1}$)	Estimated K.E of single drop (J)	K.E derived from drop size class in event ($J\ m^{-2}\ s^{-1}$)	K.E of drops in event ($J\ m^{-2}\ min^{-1}$)	% contribution of class to total K.E.
2.8	51	1.19	0.023	0.021	3.28	20.74	1058	1058	14	14.28	1385	67	9.64	9.6E-04	0.064	3.86	40.22
2.36	71	0.66	0.009	0.008	2.39	8.07	573	1631	22	7.74	751	93	7.26	2.1E-04	0.020	1.19	12.37
2	134	0.78	0.006	0.005	2.05	5.04	676	2307	31	9.13	885	175	6.33	1.0E-04	0.018	1.06	11.07
1.7	294	1.04	0.004	0.003	1.73	3.03	891	3198	43	12.04	1167	385	5.47	4.5E-05	0.017	1.05	10.90
1.4	530	1.14	0.002	0.002	1.46	1.84	974	4172	56	13.16	1276	694	4.75	2.1E-05	0.014	0.86	9.01
1	1901	2.00	0.001	0.001	1.15	0.89	1690	5862	79	22.83	2213	2489	3.91	6.8E-06	0.017	1.01	10.57
0.71	4607	1.86	0.000	0.000	0.83	0.33	1541	7403	100	20.82	2018	6033	3.05	1.6E-06	0.009	0.56	5.87
				D50=	**1.04**	**Total**	**7403**			**Total K.E of rainfall ($J\ m^{-2}\ s^{-1}$)**					**0.160**	**9.595**	

Power Function Mass of Drop y=0.9431 x1.0162 (TESCO flour);
Drop velocity = 0.822 + 2.69 (size drop)

Constants	
Rain Intensity (mm/hr)	34.9
vol of rain falling over 1 m^2 in 1 sec (mm^3)	9694.44
vol of rain falling over 1 m^2 in 1 sec (m^3)	9.7E-06
vol of rain falling over 1 m^2 in 1 sec (l)	0.00969



Appendix D Field rainfall simulator – RI calibration

The field rainfall simulator was calibrated by carrying out successive runs at 0.39 bar which had previously been tested at generating a rainfall intensity of approximately 35 mm hr^{-1}. The plot area used for calibration was $1.5m^{-2}$ within which 15 catch cup were evenly spaced (3 across by 5 down). These were numbered 1 to 15. The simulator was run for approximately 30 minutes, after which the water retained in each catch cup is measured and the rainfall intensity, *RI*, is calculated using Equation D.1. Mean rainfall intensity was calculated at 34.9 mm hr^{-1}, descriptive statistics of these data are presented in Table D.2.

$$RI = \left(\frac{R}{\pi r^2} \right) \times \left(\frac{60}{t} \right) \times 10 \qquad \text{(D.1)}$$

Where:

R, is the amount of rainfall received (ml)
r, is the catch cup radius (cm)
t, is the simulation running time (min)

Table D.1 Calculated rainfall intensity of each simulation run

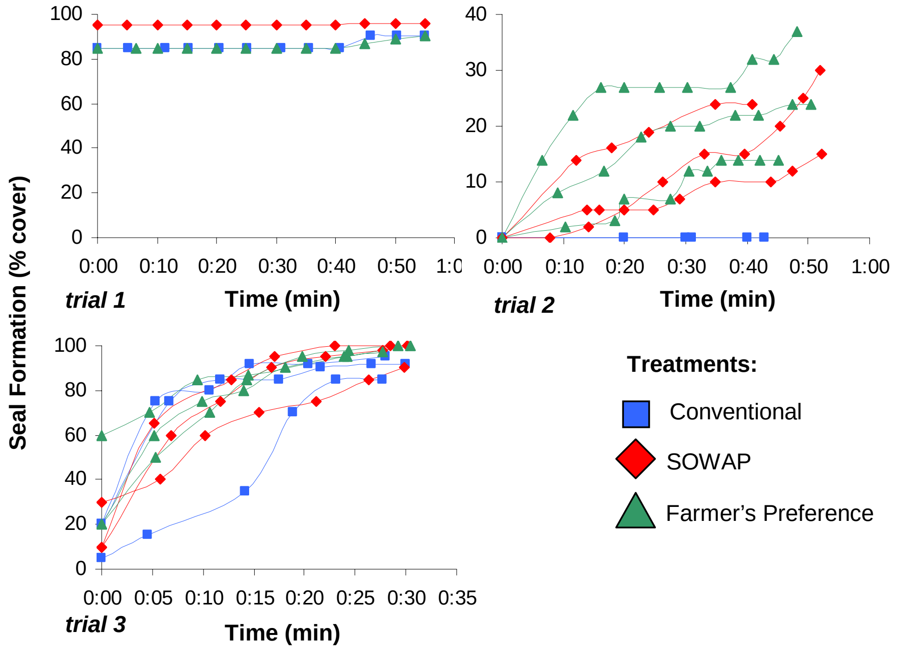
cup #	1	2	3	4	5	6	7	8	9	10	11
1	11.54	53.36	46.99	43.04	24.45	12.23	29.05	13.43	59.47	62.98	28.78
2	26.66	22.03	38.67	36.19	24.45	8.31	32.20	16.63	86.33	86.67	33.25
3	17.86	5.38	16.15	48.42	54.77	4.40	46.09	27.50	99.12	87.92	50.52
4	7.97	59.23	44.55	42.06	15.16	13.20	30.31	15.35	60.75	57.99	24.30
5	9.07	49.44	26.43	22.98	16.63	31.30	31.57	15.99	67.15	68.59	26.86
6	14.57	18.11	30.84	43.52	37.66	8.80	46.73	22.38	131.10	99.14	41.57
7	6.05	36.71	39.65	26.41	14.18	7.82	31.57	15.99	57.56	56.12	23.66
8	7.97	30.35	16.15	20.05	15.65	8.80	32.20	16.63	57.56	58.61	24.30
9	12.37	40.14	23.01	38.63	26.90	7.82	44.20	20.46	99.12	74.82	35.17
10	6.60	40.14	25.94	30.81	15.16	10.76	41.04	21.10	54.36	52.38	27.50
11	8.52	67.55	15.66	22.01	16.14	8.31	35.36	20.46	57.56	56.74	27.50
12	13.74	70.49	27.90	36.19	33.25	5.87	46.73	23.66	76.74	63.60	35.17
13	12.92	21.05	35.25	55.75	29.34	15.65	54.30	30.70	47.96	49.26	27.50
14	9.89	28.39	43.57	40.59	30.81	11.25	46.73	25.58	57.56	56.12	26.22
15	16.49	30.84	32.31	39.12	28.36	7.34	54.93	31.34	63.95	54.87	35.17
mean	**12.15**	**38.22**	**30.87**	**36.38**	**25.53**	**10.79**	**40.20**	**21.15**	**71.75**	**65.72**	**31.17**

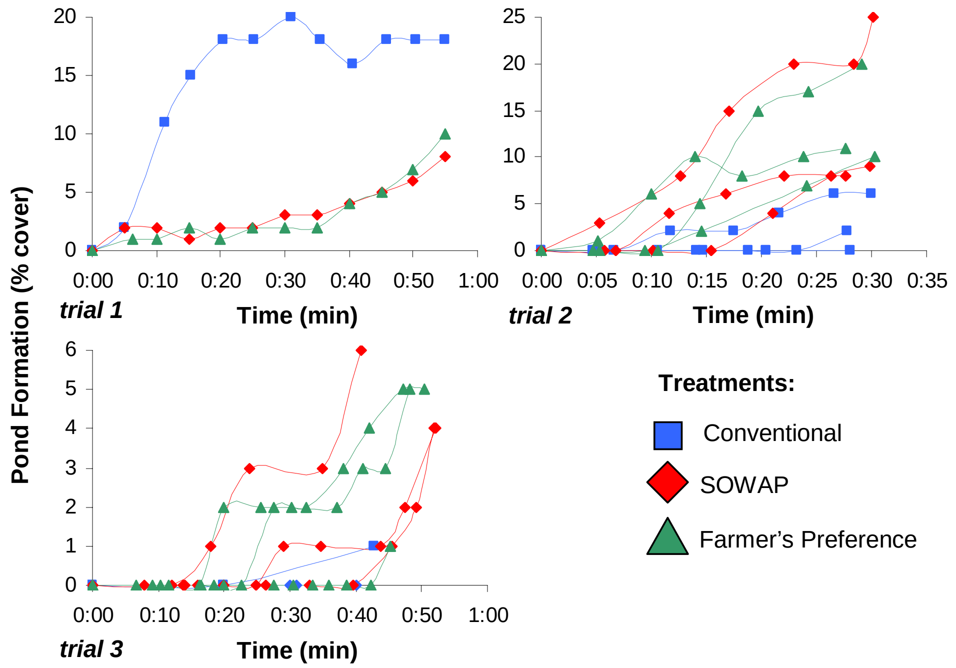
Table D.2 Descriptive statistics of rainfall intensity

	N	Mean	Std.Dev.	Std.Err	-95.00%	95.00%
Total	11	34.90	19.38	5.84	21.88	47.92

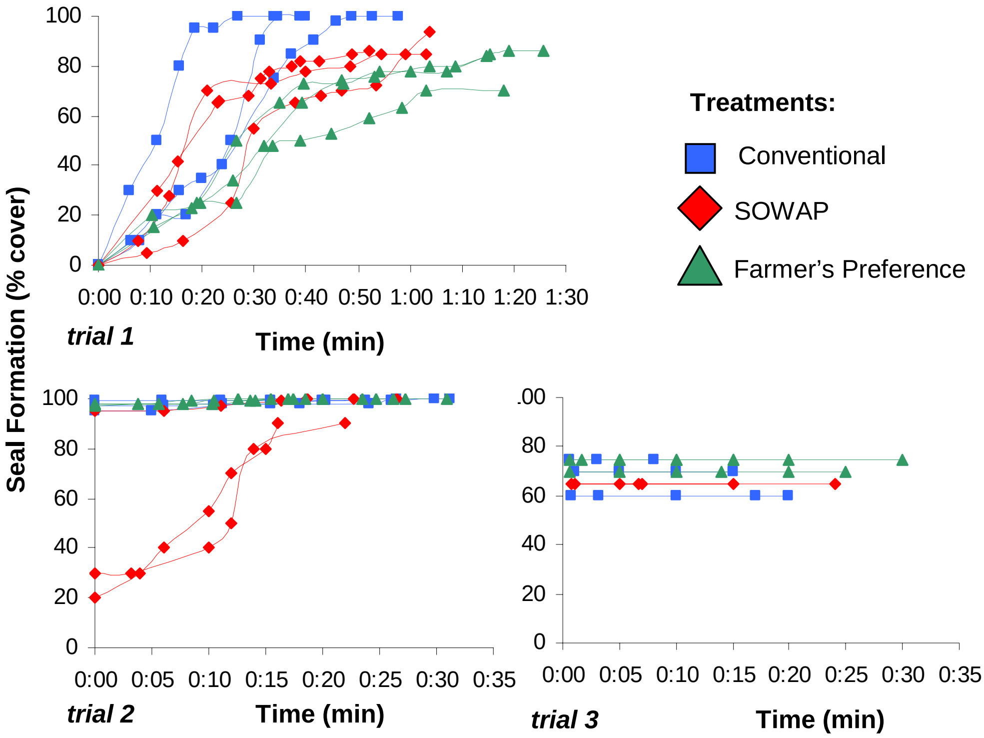
Appendix E Seal and pond formation on the micro-erosion plots

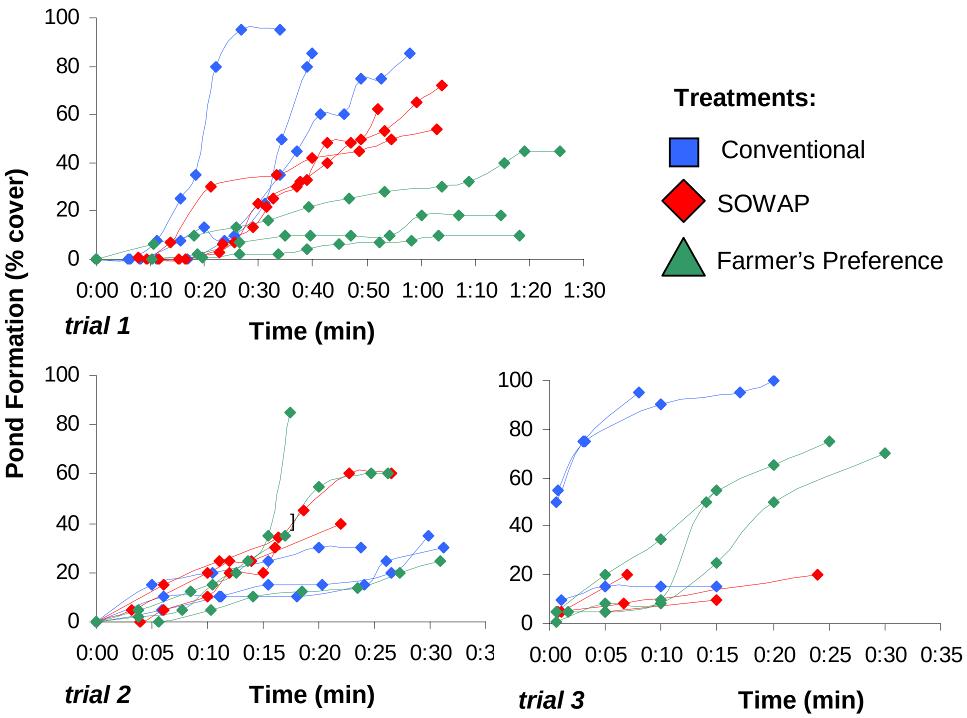
Loddington results





Tivington results





Appendix F Aggregate stability and size relationship

This section was to assess if a relationship existed between soil aggregate size and stability. It was expected that a relationship would exist based on work by Hjulström (1935) and Poesen (1992). It is predicted that there will be a positive relationship between aggregate size and stability, using aggregates sized between 0.5 and 9mm. This is based on Hjulstrom's curve which shows the energy requirements to detach, transport and deposit soil particles in water (Hjulström 1935). If a relationship does exist between aggregate size and stability, this would have implications on the 3.35 – 5mm aggregate size used for stability tests. To test this aggregates were tested for stability under the gravity fed rain tower as layout in chapter 5.

At Loddington there were no significant treatment differences in relation to aggregate size and stability. The outputs have been presented in Figure F.1 and Figure F.2. Again there is no significant difference between different aggregate sizes and the percentage stable aggregates present. There is however a positive trend. As aggregate size increases so does the stability of aggregates, except using aggregates sized 3.35-5mm.

Data from Tivington showed no significant differences in aggregate size and percentage stable aggregates in relation to treatment. The outputs have been presented in Figure F.3 and Figure F.4. There was no significant effect of aggregate size on the stability of aggregates. There is a trend towards their being a negative relationship between aggregate size and percentage stable aggregates.

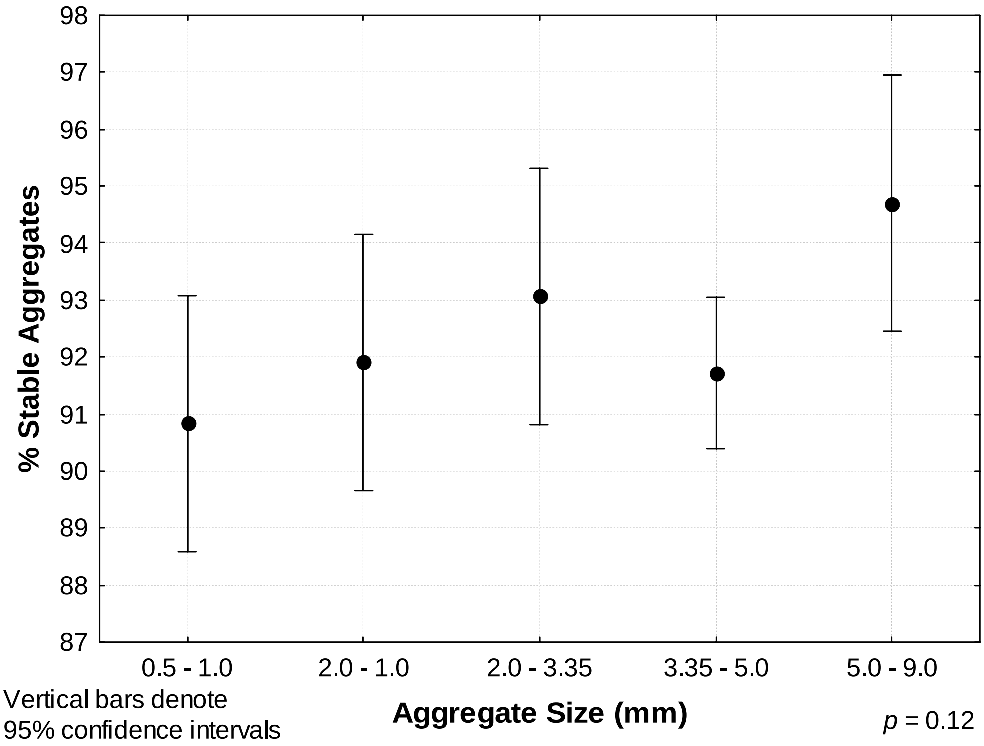


Figure F.1 Loddington: relationship between aggregate size and stability, n = 47

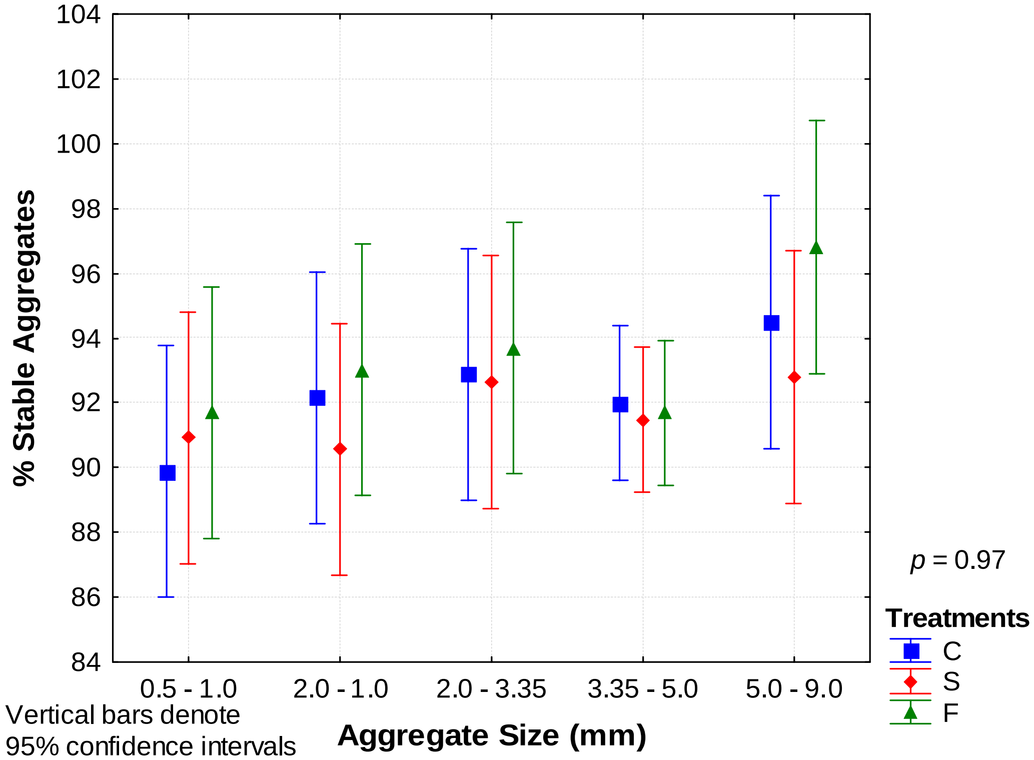


Figure F.2 Loddington: relationship between aggregate size and stability in relation to different tillage treatments. C=conventional, S=SOWAP and F=Farmer's Preference, n = 47

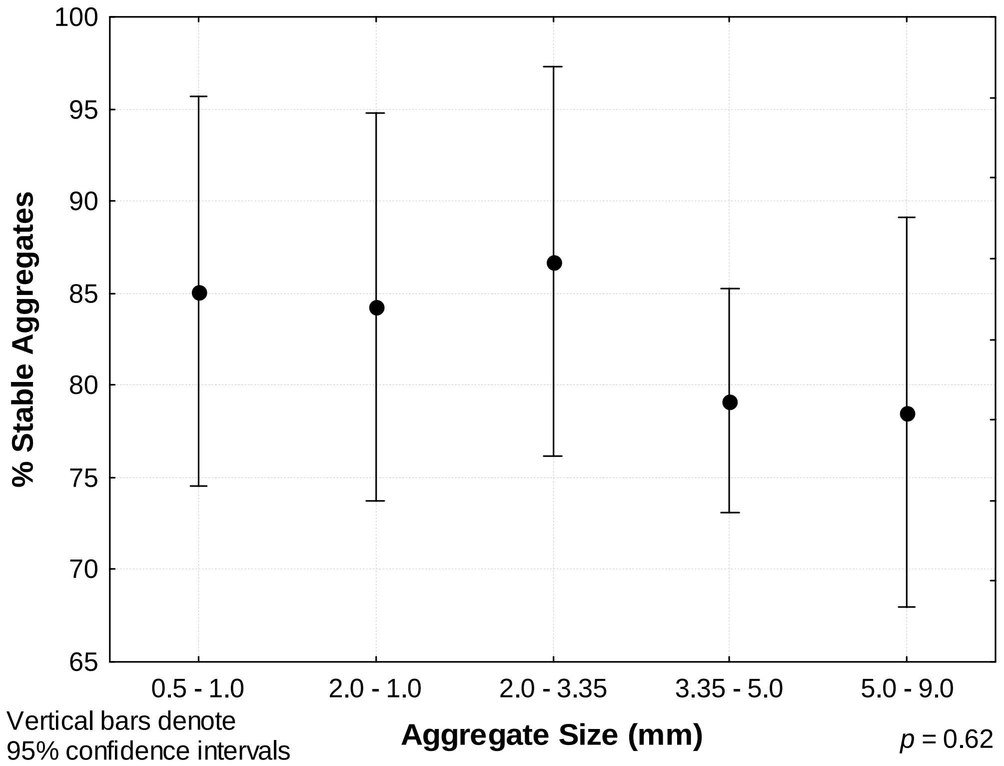


Figure F.3 Tivington: relationship between aggregate size and stability, n = 48

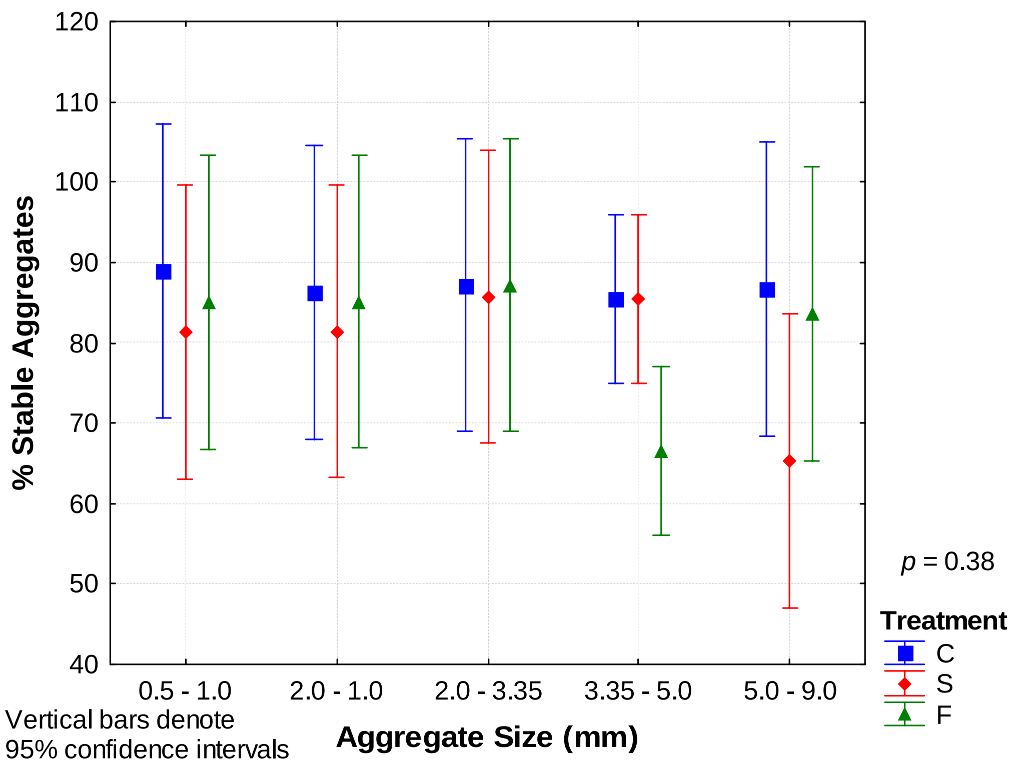


Figure F.4 Tivington: relationship between aggregate size and stability in relation to different tillage treatments. C=conventional, S=SOWAP and F=Farmer's Preference, n =48

It was expected that a positive relationship would exist between aggregate size and stability. Statistically this was not the case at either site, but at Loddington there was a positive trend with aggregate stability increasing from aggregate sizes 0.5 to 9mm. There was a reduction in stability for aggregate sizes 3.35-5mm. At Tivington there was no visible trend / relationship between aggregate size and stability. Therefore using this current data set the hypothesis cannot be supported.

The lack of relationship was not a function of treatment but could be related to the method used to obtain the results. In order to test the hypothesis linking aggregate size to stability, aggregates were only tested under the rain tower. The results from hypothesis two show that the wet sieving - immersion based method is more destructive, producing lower percentages of stable aggregates compared to those from the rain impact method (rain tower). The wet sieving method could not have been used as it does not allow testing of aggregates down to a size of 0.5mm. The smallest size that could have been tested using this method would have been 2mm. The results from the rain impact method will only pick up very large differences and trends.

Appendix G Laboratory rain tower – rainfall intensity calibration

The laboratory rain tower was calibrated for rainfall intensity by carrying out successive runs with 75 needles open and a weir of 16mm. Numerous combinations of needles open and weir depth were tried before the current set up was kept (Figure G.1).The rainfall target area was $0.5m^{-1}$ under which catch cups of a known diameter were placed uniformly.

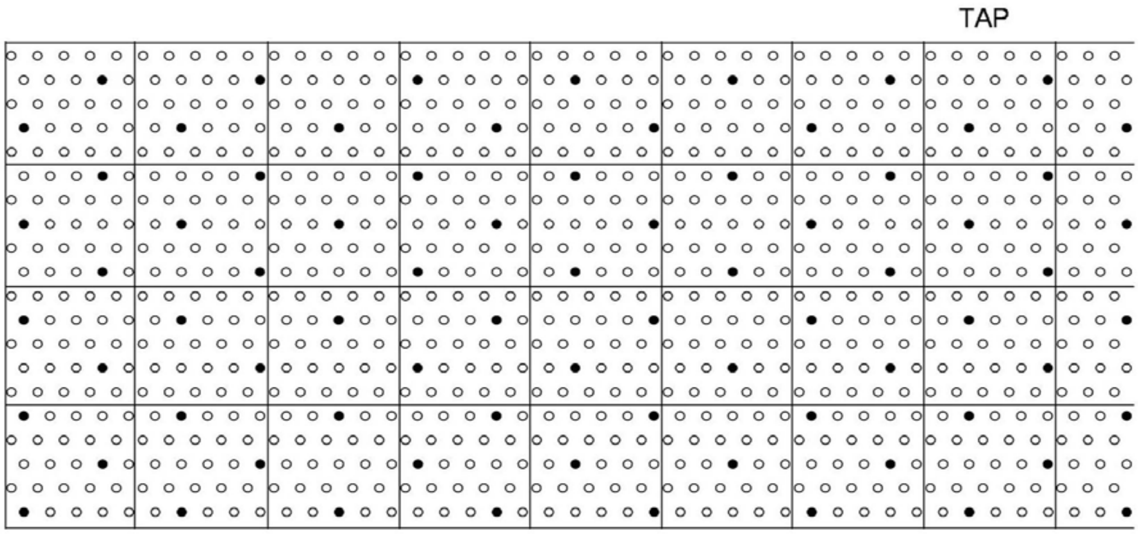


Figure G.1 Needle tray layout, plan view. Dots denotes open needles

The rainfall simulations were run for 17 minutes, after which the water retained in each catch cup was measured and the rainfall intensity, *RI*, is calculated using Equation G.1. Where *R*, is the amount of rainfall received (ml), *r*, is the catch cup radius (cm) and *t*, is the simulation running time (min). The mean rainfall intensity after 19 simulations was calculated at 35.48 mm hr^{-1}, descriptive statistics of this data are presented in Table G.1 and Table G.2.

$$RI = \left(\frac{R}{\pi r^2}\right) \times \left(\frac{60}{t}\right) \times 10 \qquad \text{(G.1)}$$

Where:
R, is the amount of rainfall received (ml)
r, is the catch cup radius (cm)
t, is the simulation running time (min)

Table G.1 Descriptive statistics of rainfall intensity

	N	Mean	Std.Dev.	Std.Err	-95.00%	95.00%
Total	19	35.48245	2.271077	0.521021	34.38782	36.57707

Table G.2 Calculated rainfall intensity of each simulation run

cup #	1	2	3	4	5	6	7	8	9
1	39.44	36.68	34.85	33.93	33.02	32.10	29.81	29.81	33.93
2	39.44	38.06	36.68	36.23	34.85	32.56	33.02	32.10	36.68
3	38.52	35.77	35.77	33.93	36.68	32.56	32.10	33.02	36.68
4	34.85	32.10	32.10	31.18	34.39	31.18	30.26	30.72	33.93
5	31.18	27.05	28.43	27.51	30.72	27.05	27.05	27.51	28.43
6	40.35	36.68	35.31	36.68	36.23	32.10	30.72	30.26	32.56
7	43.10	38.98	38.52	39.44	39.44	36.23	34.85	33.93	40.35
8	41.27	38.52	38.52	40.35	39.89	37.14	36.68	35.31	40.35
9	38.52	33.93	36.23	37.14	37.14	33.93	34.39	33.02	37.60
10	33.02	28.89	31.18	31.18	33.02	29.35	30.26	31.64	33.02
11	42.19	38.06	36.23	37.60	35.77	32.10	32.10	31.18	38.52
12	43.56	39.44	38.06	40.35	39.44	36.23	35.77	34.85	41.27
13	42.65	38.52	38.52	40.35	40.35	35.77	36.68	36.68	40.35
14	40.35	34.85	36.68	38.52	38.52	35.77	36.23	35.31	39.44
15	36.23	30.26	32.10	32.10	33.93	31.18	31.18	30.72	33.47
16	43.10	38.52	36.68	37.60	34.85	32.10	32.10	31.18	38.52
17	44.02	40.35	39.44	40.35	38.98	36.23	34.85	35.77	42.19
18	44.02	38.52	39.44	40.35	40.35	36.68	37.14	37.14	41.27
19	41.27	34.85	36.68	38.52	39.44	35.77	36.68	35.77	38.52
20	36.68	30.26	32.10	32.10	34.85	31.18	32.10	30.26	33.02
21	43.10	38.52	36.68	36.68	33.93	33.02	30.72	34.85	38.52
22	44.94	40.35	39.44	40.35	38.06	36.23	34.85	36.23	41.27
23	44.94	39.44	39.44	40.35	40.81	36.23	36.68	34.85	41.27
24	41.27	35.77	36.68	38.52	39.44	34.85	35.31		38.52
25	36.68	30.26	32.10	35.31	33.93	30.26	31.18	29.35	33.02
26	41.27	38.52	37.60	33.02	33.02	31.18	30.26	29.35	38.52
27	44.94	39.44	39.44	39.44	37.60	34.85	33.02	33.93	41.27
28	44.02	38.52	39.44	39.44	38.98	34.85	35.31	34.85	40.35
29	42.19	36.23	36.68	36.68	37.14	32.56	34.85	33.93	38.52
30	36.23	29.81	31.18	30.72	32.56	28.43	28.89	29.35	31.64
mean	**40.44**	**35.90**	**36.07**	**36.53**	**36.58**	**33.32**	**33.17**	**32.86**	**37.43**

cup #	11	12	13	14	15	16	17	18	19
1	28.43	33.02	35.77	36.68	34.85	34.85	32.10	31.18	34.65
2	30.26	36.68	38.52	38.52	35.77	37.14	34.85	33.93	36.96
3	30.72	34.85	36.68	37.60	34.85	35.77	33.93	32.10	36.19
4	31.18	33.02	33.93	33.93	32.56	32.10	29.35	29.81	32.34
5	30.26	27.51	28.43	27.51	27.51	26.60	23.84	24.76	26.18
6	30.26	37.60	38.52	39.44	36.68	37.14	34.85	33.93	37.73

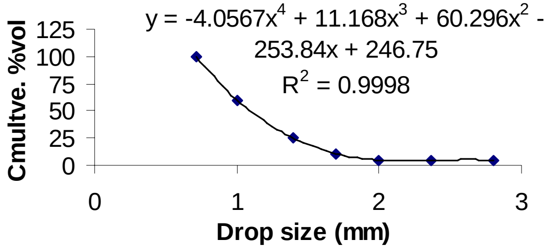
7	33.02	39.44	41.73	42.19	39.44	41.27	37.60	36.68	40.04
8	33.93	39.44	40.35	41.27	39.44	39.44	36.68	36.68	40.04
9	33.02	35.77	36.23	36.68	35.77	34.85	33.02	32.10	36.19
10	30.72	29.35	31.18	30.72	29.35	28.43	26.60	25.68	30.03
11	30.26	37.60	39.44	39.44	37.60	37.60	35.77	33.93	39.27
12	33.02	39.44	43.10	41.73	41.27	40.35	37.60	35.77	40.81
13	34.85	38.98	42.65	42.19	41.27	40.35	36.23	36.68	40.04
14	33.02	35.77	38.98	38.52	37.60	36.68	33.02	33.02	37.34
15	30.72	30.26	33.93	32.10	31.18	30.26	27.51	27.51	32.34
16	30.72	37.60	39.89	38.52	38.52	37.14	36.68	34.85	39.27
17	33.02	39.44	41.27	42.65	41.27	40.35	38.52	36.68	42.35
18	33.93	38.52	41.73	42.19	40.81	40.35	36.68	35.77	41.96
19	33.93	35.77	41.27	38.52	37.60	36.68	33.47	33.02	37.73
20	31.18	30.26	33.47	33.02	32.10	30.72	27.51	28.89	32.34
21	31.18	36.68	39.44	39.89	36.68	37.60	35.77	33.93	38.50
22	33.93	39.44	42.19	43.10	40.35	40.35	37.60	35.77	41.58
23	34.39	39.89	41.27	42.19	40.35	39.44	36.68	35.77	41.58
24	33.93	35.77	38.06	38.52	38.52	35.77	33.02	33.02	36.96
25	30.26	30.26	32.56	32.10	32.10	35.77	27.51	27.51	32.34
26	30.72	35.77	38.52	37.60	35.77	30.26	33.93	32.10	37.73
27	33.93	37.60	40.81	39.44	39.44	38.52	35.77	33.93	40.04
28	34.39	38.06	40.35	39.89	39.44	37.60	35.77	33.93	39.27
29	32.56	34.85	36.68	36.68	36.68	33.93	32.10	31.18	36.57
30	28.89	28.43	30.72	30.72	32.10	28.43	26.60	26.60	30.41
mean	**32.02**	**35.57**	**37.92**	**37.78**	**36.56**	**35.86**	**33.35**	**32.56**	**36.96**

Appendix H Drop size calibration of laboratory rain tower, (adapted from Simmons 1998)

Sieve size	No. of pellets	Mass of pellets (g)	Mass of single pellet (g)	Mass of single drop (g)	Size Drop (mm)	Volume of single drop (mm^3)	Total vol. of drops (mm^3)	Cumulative vol.	Cumulative % vol.	% of total sample drop volume	Vol. rainfall in size class (mm^3)	No. of drops of size class in rain event	Estimated drop velocity ($m\ s^{-1}$)	Estimated K.E of single drop (J)	K.E derived from drop size class in event ($J\ m^{-2}\ s^{-1}$)	K.E of drops in event ($J\ m^{-2}\ min^{-1}$)	% contribution of class to total K.E.
2.8	2	0.12	0.061	0.055	4.5	54.9	109.8	111	5	4.5	444	8.09	13.02	4.7E-03	0.038	2.26	38.66
2.36	0	0.00	0.000	0.000	0.0	0.0	0.0	111	5	0.0	0	0.00	0.82	0.0E+00	0.000	0.00	0.00
2	2	0.01	0.006	0.005	2.0	4.8	9.5	119	5	0.4	39	8.09	6.22	9.2E-05	0.001	0.04	0.77
1.7	43	0.14	0.003	0.003	1.7	2.7	117.9	237	10	4.8	477	174.02	5.31	3.9E-05	0.007	0.40	6.92
1.4	216	0.45	0.002	0.002	1.4	1.8	381.4	619	25	15.7	1543	874.13	4.70	2.0E-05	0.017	1.02	17.51
1	1918	1.01	0.001	0.000	0.9	0.4	840.9	1460	60	34.5	3403	7761.97	3.26	2.3E-06	0.018	1.08	18.57
0.71	3376	1.18	0.000	0.000	0.8	0.3	975.8	2435	100	40.1	3949	13662.36	2.94	1.3E-06	0.017	1.03	17.57
				D50=	**0.86**	**Total**	**2435**			**Total K.E of rainfall (J/m2/s)**					**0.097**	**5.842**	

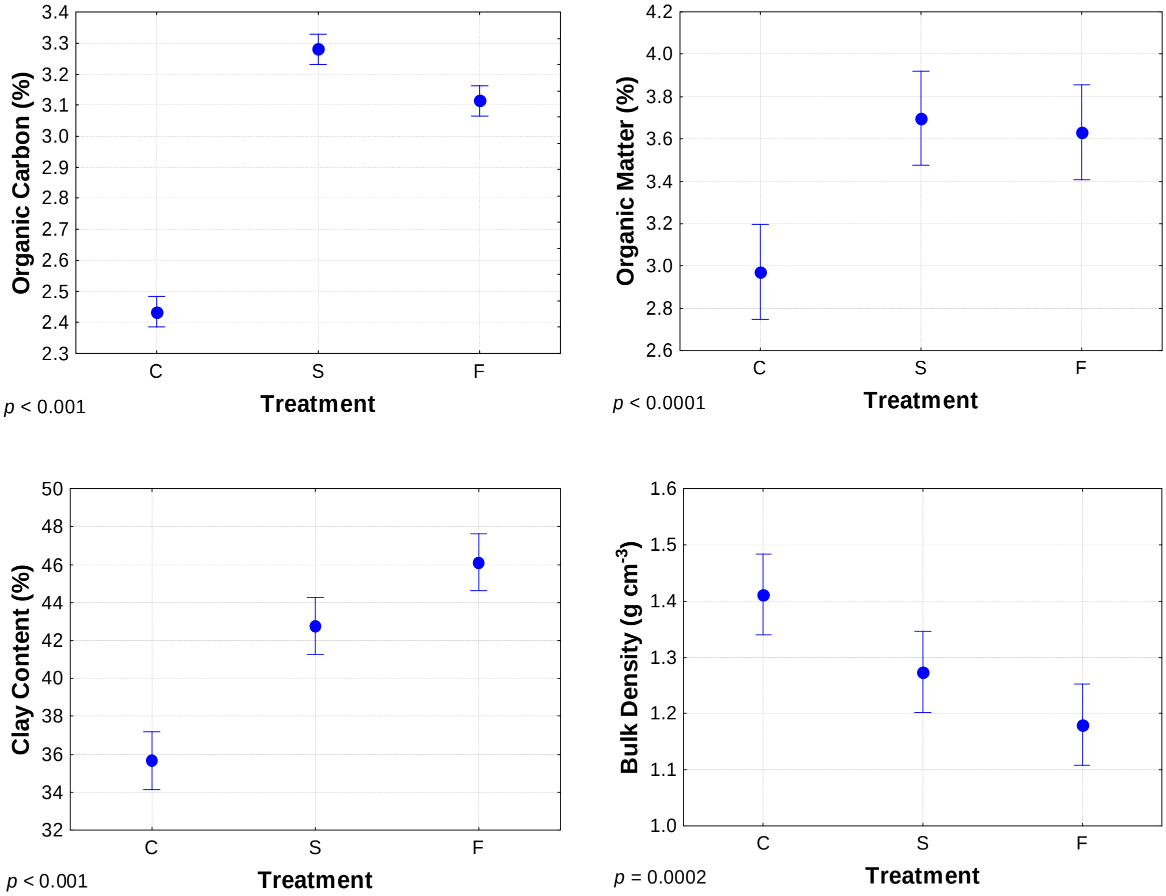
Power Function Mass of Drop y=0.9431 x1.0162 (TESCO flour); Drop velocity = 0.822 + 2.69 (size drop)

Constants	
Rain Intensity (mm/hr)	35.48
vol of rain falling over 1 m^2 in 1 sec (mm^3)	9855.56
vol of rain falling over 1 m^2 in 1 sec (m^3)	9.86E-06
vol of rain falling over 1 m^2 in 1 sec (l)	0.00987



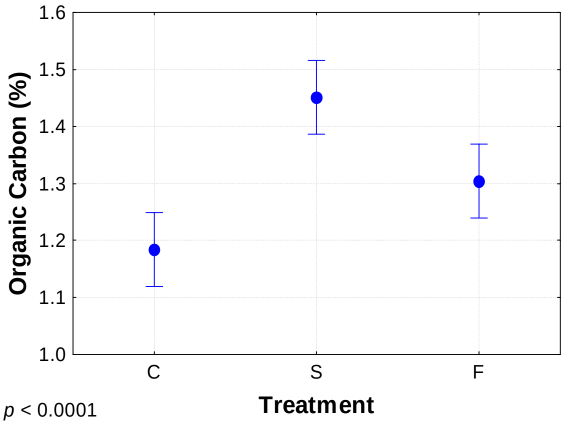
Appendix I Relationship between soil properties and soil aggregates

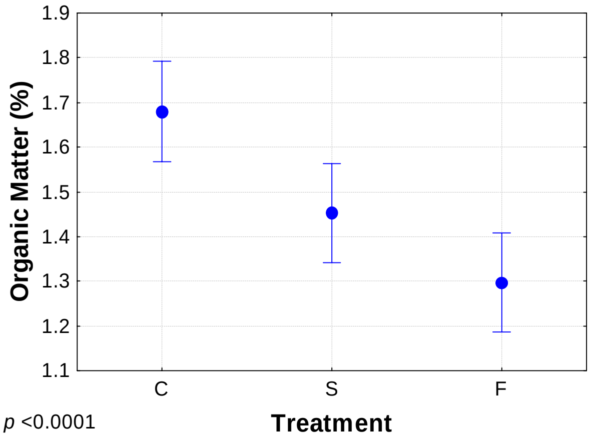
Loddington season 1 to 3

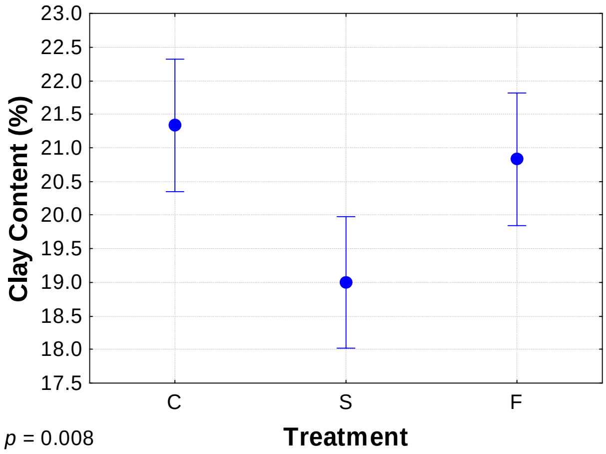


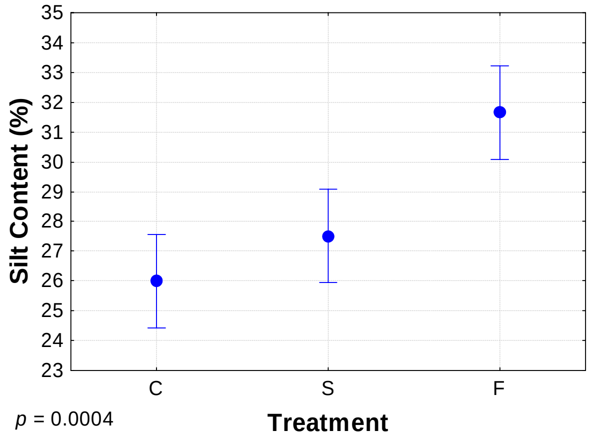
Error bars denote confidence intervals at 95% confidence; n = 99 (organic carbon); 72 (organic matter); 18 (clay content); and 54 (bulk density).

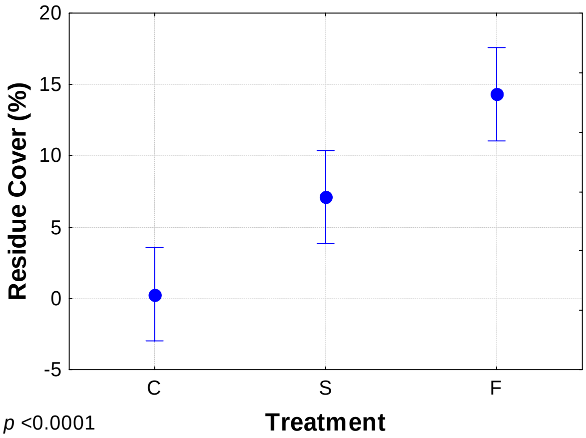
Tivington season 2











Error bars denote confidence intervals at 95% confidence; n = 75 (organic carbon); 72 (organic matter); 12 (clay content); 12 (silt content) and 69 (residue cover).

Appendix J Aggregate stability: whole soil versus 3.35-5mm

This section was to assess if there was a significant difference in stability when aggregates were tested using sizes 3.35-5mm against a whole soil sample. This study was carried to address the concern of whether aggregates of a specified size reflect the stability of a whole soil sample (Six et al. 2000). This had been discussed in chapter 5; in this section only the graphical outputs will be presented.

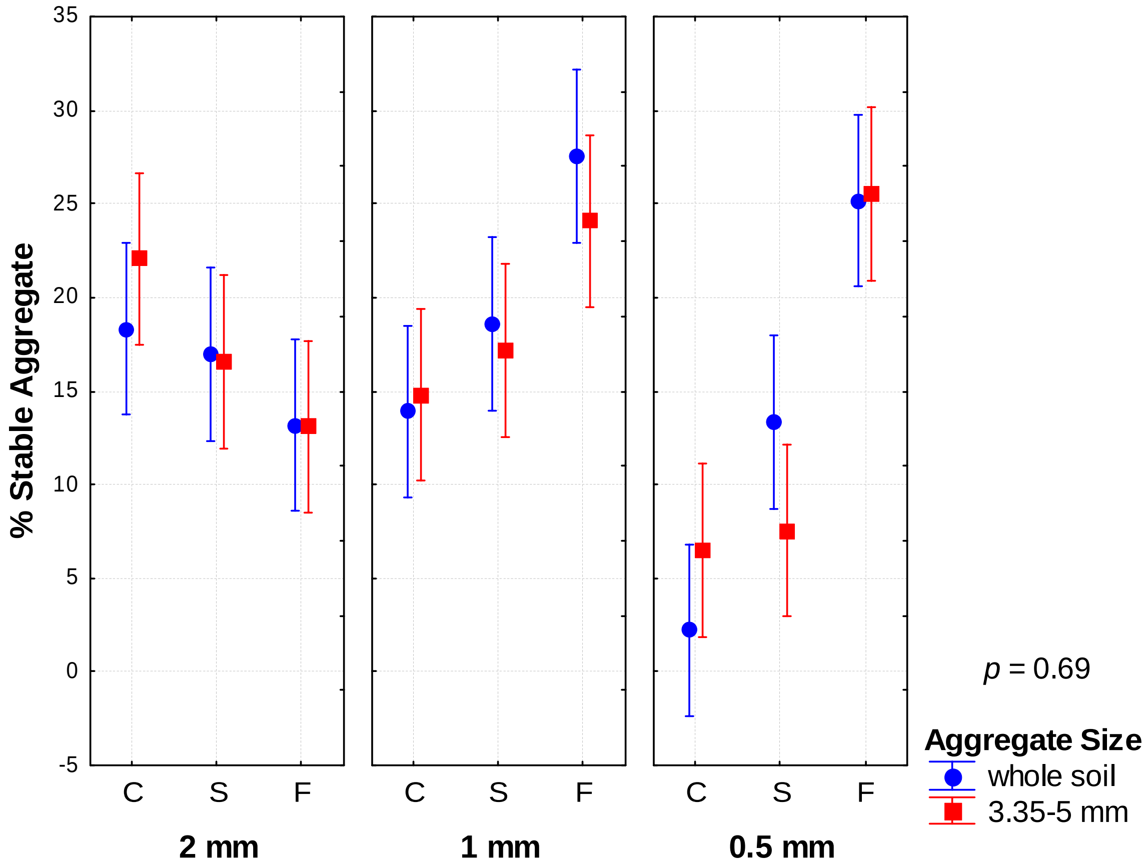


Figure J.1 Loddington: aggregate stability at different sieve sizes, n=36

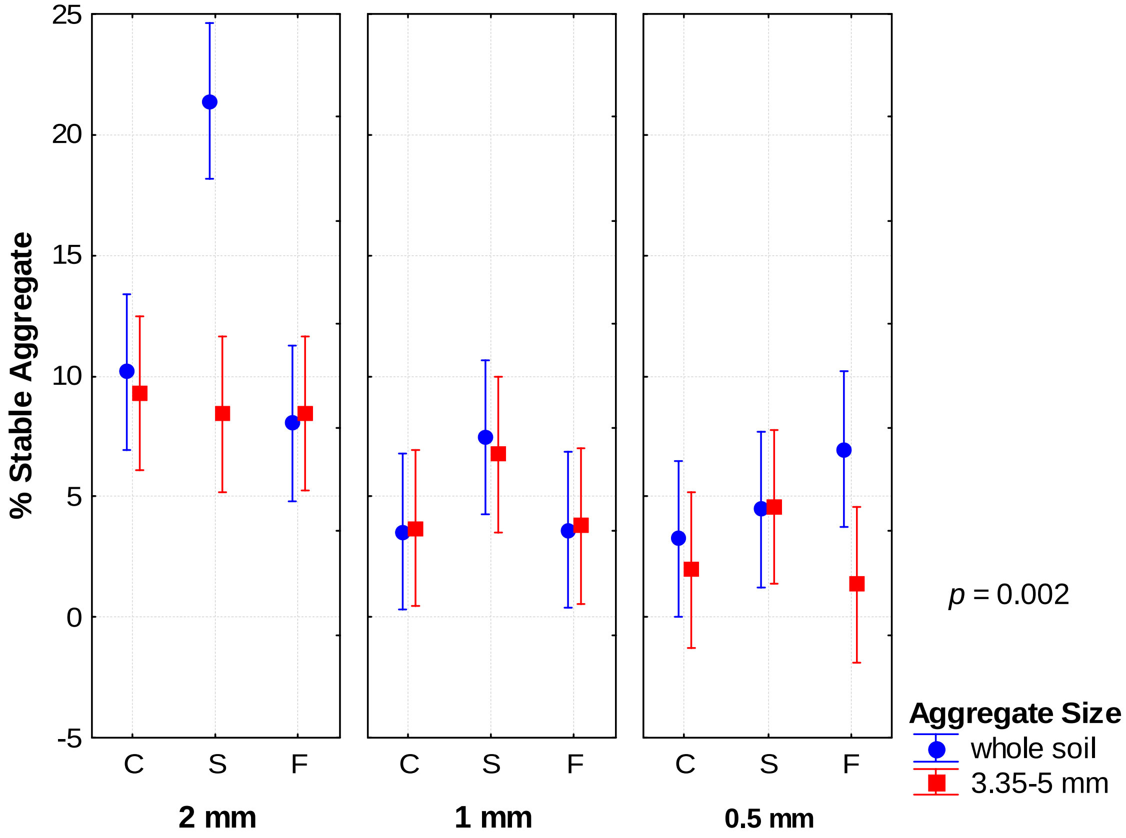


Figure J.2 Tivington: aggregate stability at different sieve sizes, n = 36

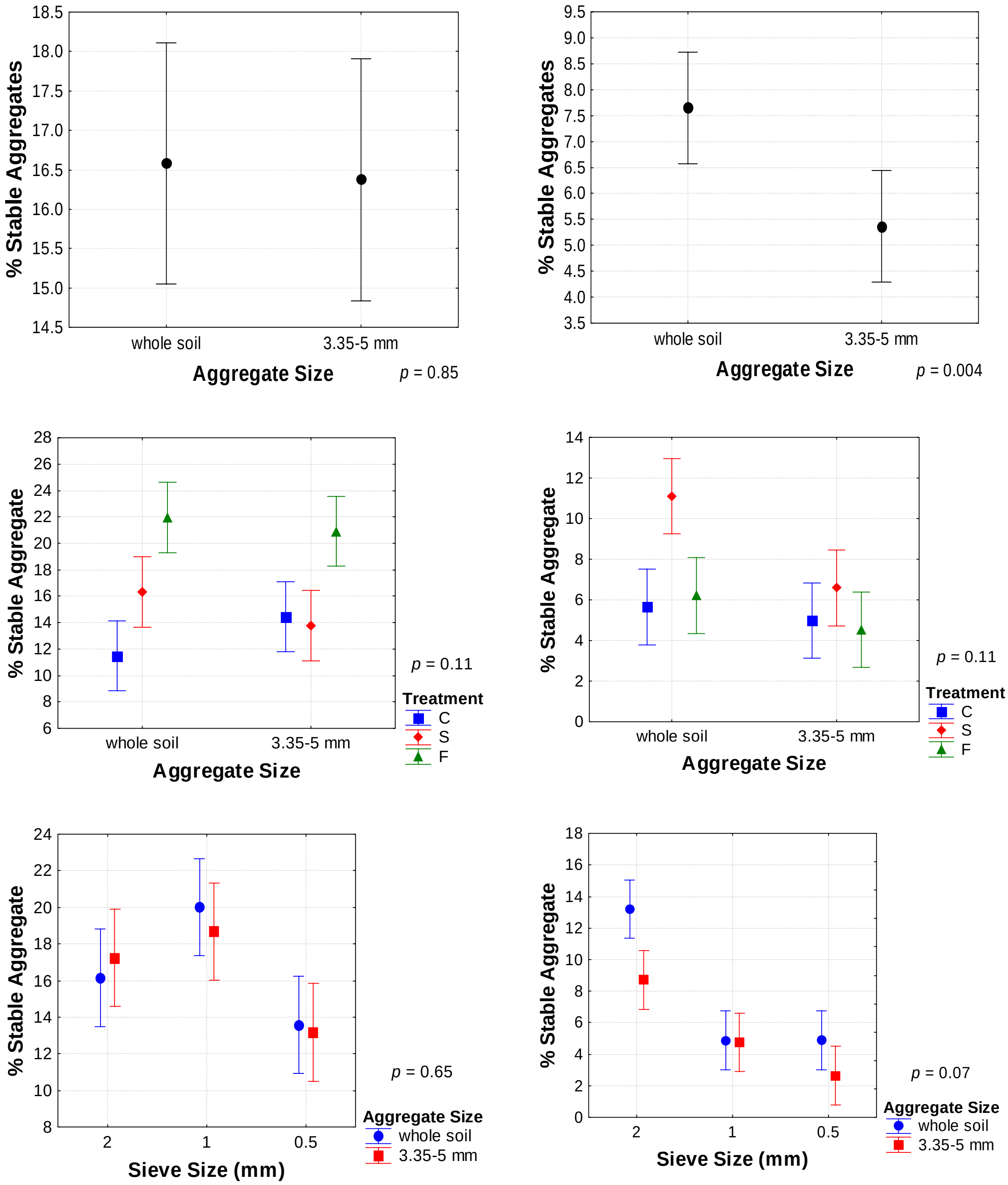


Figure J.3 Loddington (left) and Tivington (right): 3 sets of graphs to show the relationship between aggregates sized 3.35-5mm and a whole soil sample in terms of aggregate stability, n = 36

Appendix K Spatial scale comparisons

Table K.1 Loddington: comparison of mean results of runoff, soil loss and runoff coefficients (R*C*) between two spatial scales; field erosion plots (EP) and micro-plots (MP). S = season O = overall

Time		Runoff ($l\ ha^{-1}\ mm^{-1}$)			Soil Loss ($g\ ha^{-1}\ mm^{-1}$)			*C* value		
		EP	MP	%	EP	MP	%	EP	MP	%
O	mean	52.5	737.4	8.0	33.8	420.7	7.1	0.0034	0.042	8.0
	S.E.	737.4	315.4		8.7	105.8		0.0009	0.011	
S1	mean	12.3	81.6	15.1	19.5	209.3	9.3	0.0012	0.008	15.1
	S.E.	1.6	27.0		3.8	66.1		0.0002	0.003	
S2	mean	61.4	966.8	6.4	127.3	1179.5	10.8	0.0061	0.097	6.4
	S.E.	21.4	65.2		60.1	250.0		0.0021	0.007	
S3	mean	5.2	254.1	2.1	0.0	1429.8	0.0	0.0005	0.025	2.1
	S.E.	1.5	130.2		0.0	1305.0		0.0002	0.013	
S4	mean	25.7	287.4	8.9	10.5	258.3	4.1	0.0026	0.029	8.9
	S.E.	7.2	193.5		4.5	174.2		0.0007	0.019	

Table K.2 Tivington: comparison of mean results of runoff, soil loss and runoff coefficients (R*C*) between two spatial scales; field erosion plots (EP) and micro-plots (MP). S = season O = overall

Time		Runoff ($l\ ha^{-1}\ mm^{-1}$)			Soil Loss ($g\ ha^{-1}\ mm^{-1}$)			*C* value		
		EP	MP	%	EP	MP	%	EP	MP	%
O	mean	313.7	1042.1	30.1	3072.8	5479.8	56.1	0.031	0.104	30.1
	S.E.	46.3	155.7		889.4	1277.1		0.005	0.016	
S2	mean	231.7	1429.4	16.2	1026.8	9297.2	11.0	0.023	0.143	16.2
	S.E.	41.6	233.1		288.4	2172.8		0.004	0.023	
S3	mean	386.5	654.8	59.0	4891.5	1662.3	294.3	0.039	0.065	59.0
	S.E.	78.3	166.7		1627.6	529.8		0.008	0.017	

Table K.3 Direct comparison of mean results and treatment ranks of treatments at each spatial scale

Site	Scale / Method		Units	Parameter	Conventional		SOWAP		Farmer		*p*
					mean	rank	mean	rank	mean	rank	
L	f	EP	$l\ ha^{-1}$	runoff	1508	III	1890	I	1541	II	NS
L	m	RS	$l\ m^{-2}\ mm^{-1}$	runoff	0.05	II	0.04	III	0.07	I	NS
L	f	EP	$g\ ha^{-1}$	sediment	1210	II	3268	I	1031	III	NS
L	m	RS	$g\ m^{-2}\ mm^{-1}$	sediment	0.03	II	0.02	III	0.08	I	NS
L	a	RT	% stable	soil	88.3	I	89.4	II	90.8	III	NS
L	a	WS	% stable	soil	44.7	I	54.3	II	67.3	III	***
L	a	FTK	% stable	soil	38.7	I	52.5	II	60.5	III	***
T	f	EP	$l\ ha^{-1}$	runoff	11289	III	11965	II	18187	I	NS
T	m	RS	$l\ m^{-2}\ mm^{-1}$	runoff	0.20	I	0.13	II	0.11	III	NS
T	f	EP	$g\ ha^{-1}$	sediment	62521	I	25168	III	39237	II	NS
T	m	RS	$g\ m^{-2}\ mm^{-1}$	sediment	1.08	I	1.04	II	0.67	III	NS
T	a	RT	% stable	soil	87.4	III	86.8	II	84.5	I	NS
T	a	WS	% stable	soil	30.2	II	37.2	III	20.1	I	**
T	a	FTK	% stable	soil	37.3	II	44.2	III	27.5	I	***

Site locations: Loddington (L) and Tivington (T). Spatial scales: field level ≈0.05ha (f), micro-plots 1.5m^2 (m), aggregate scale mm (a). Methods used: erosion plots (EP), rainfall simulations (RS), rain tower (RT), wet sieving (WS) and field test kit (FTK). NS = not significant; * = *p*<0.05; ** = *p*<0.01; * = *p*<0.001. Ranking order: most erodible or highest erosion (I) to least erodible or lowest erosion (III).**

Table K.4 Tivington: direct comparison of mean result and treatment ranks of treatments at each spatial scale across three seasons

S	Scale / Method		Units	Parameter	Conventional mean	Conventional rank	SOWAP mean	SOWAP rank	Farmer mean	Farmer rank	*p*
1	f	EP	$l\ ha^{-1}$	runoff	1496	III	1933	II	1953	I	NS
1	m	RS	$l\ m^{-2}\ mm^{-1}$	runoff	0.014	I	0.005	III	0.006	II	NS
1	f	EP	$g\ ha^{-1}$	sediment	2115	II	4619	I	1927	III	NS
1	m	RS	$g\ m^{-2}\ mm^{-1}$	sediment	0.035	I	0.010	III	0.011	II	NS
1	a	RT	% stable	soil	81	I	88	II	92	III	NS
1	a	WS	% stable	soil	28	I	41	II	56	III	***
1	a	FTK	% stable	soil	13	I	31	II	63	III	***
2	f	EP	$l\ ha^{-1}$	runoff	2853	II	3509	I	2431	III	NS
2	m	RS	$l\ m^{-2}\ mm^{-1}$	runoff	0.097	II	0.084	III	0.109	I	NS
2	f	EP	$g\ ha^{-1}$	sediment	2435	II	6896	I	2044	III	NS
2	m	RS	$g\ m^{-2}\ mm^{-1}$	sediment	0.018	III	0.039	II	0.071	I	NS
2	a	RT	% stable	soil	87	I	88	II	91	III	NS
2	a	WS	% stable	soil	37	I	54	II	70	III	***
2	a	FTK	% stable	soil	48	I	52	II	55	III	***
3	f	EP	$l\ ha^{-1}$	runoff	95	III	136	II	278	I	NS
3	m	RS	$l\ m^{-2}\ mm^{-1}$	runoff	0.007	III	0.014	II	0.055	I	NS
3	f	EP	$g\ ha^{-1}$	sediment	0.0041	I	0.0006	II	0.0001	III	NS
3	m	RS	$g\ m^{-2}\ mm^{-1}$	sediment	0.041	II	0.010	III	0.115	I	NS
3	a	RT	% stable	soil	88	I	91	II	92	III	NS
3	a	WS	% stable	soil	60	I	59	I	68	III	***
3	a	FTK	% stable	soil	41	I	66	III	63	II	NS

Spatial scales: field level ≈0.05ha (f), micro-plots 1.5m^2 (m), aggregate scale mm (a). Methods used: erosion plots (EP), rainfall simulations (RS), rain tower (RT), wet sieving (WS) and field test kit (FTK). NS = not significant; * = *p*<0.05; ** = *p*<0.01; * = *p*<0.001. Ranking order: most erodible or highest erosion (I) to least erodible or lowest erosion (III).**

Appendix L Kinetic energy of rainfall at both micro-plot and field plot scales

Rainfall data was collected at each scale to gain an understanding of the results found. At the micro-plot scale rainfall intensities measured over a 30 minute period (mean) and results were recalculated to mm h^{-1}. Rainfall at the field plot scale was measured using an automated system at the field scale. Rainfall was automatically measured every 5 minutes and the information was electronically stored. Calculating a form of intensity was more complex. The time periods involved at each spatial scale were vastly different; hourly at the micro-plot scale, and daily to monthly at the field scale.

The total amount of rainfall received was considered to show differences in rainfall between spatial scales. Although total rainfall would show possible spatial scale difference it gave no comprehension as to the severity, i.e. how much rainfall was received over a specific period of time. Rainfall intensity has been shown to be closely linked to soil loss (Morgan 2005), therefore the use of total rainfall was abandoned as an explanation tool. However, these results were incorporated into the calculation of runoff coefficients and runoff and soil losses per unit rainfall received.

To gain a better appreciation of the intensity of rainfall being received at the field plot scale, calculations were made using a form of intensity known as Ro, an indice used in the Morgan-Morgan-Finney model (Morgan 2005). It is calculated by dividing the annual or mean annual rainfall (mm) by the number of rain days within that year. Although appropriate to show the level of intensity received at this scale, the results could not be easily linked to that at the micro-plot scale and so were set aside.

The basis of erosivity indices based on kinetic energy of rainfall as used in erosion models were then trialled. Erosivity of rainfall is calculated on intensity and duration of the rain storm and also the mass, diameter and velocity of the raindrops within that storm. The information required for the latter part of this calculation was not measured during this present study, but previous research allows the kinetic energy of a rain storm to be calculated from rainfall intensity alone, through equations applicable to particular countries. The way in which rainfall intensity is used is a point for discussion and will be mentioned here in brief. The rainfall intensity used in the calculation of kinetic energy has differed in past research. Rainfall intensities (I) calculated from 30 (Wischmeier & Smith 1978), 15 (Stocking & Elwell 1973 in Morgan 2005) and 5 (Usón & Ramos 2001 in Morgan 2005) minute periods have all been used. Calculating rainfall intensity over a shorter period of time was felt to be the most appropriate measurement of intensity at the field scale.

The maximum rainfall intensity over 30, 15 and 5 minute periods were all tried but it was found that the use of 5 minute periods was the most accessible. This was because the automated rainfall collector measured rainfall every 5 minutes of every day during the experiment. The experiment ran for over 2 years which resulted in a very extensive database from which proved difficult to extract specific information. In analysing the results, it was found that for the majority of time at both sites very little rain fell. On days when rain fell it was sporadic often with no more than 5-10 minutes of rain fall at any particular time. Rainfall intensities measured over a 5 minute period (I_5) where the most suitable as to not miss any large rainfall events, which might be lost using I_{15} or I_{30} indexes.

As previously stated rainfall was not always continuous and so the kinetic energy of a rain storm could not be calculated. The kinetic energy (KE) was calculated for the maximum 5 minute intensity. The equation used to calculate the KE was that used by Hudson (1965) from work carried out in Zimbabwe and was applied to intensities above 10 mm h^{-1}, which is more applicable to temperate latitudes (Morgan 2005). The calculation of kinetic energy can be found in Equation L.1 where, KE represents the kinetic energy of rain (MJ ha^{-1} mm^{-1}) and I is the rainfall intensity (mm h^{-1}).

$$KE = 0.298\left(1 - \frac{4.29}{I}\right) \quad \textbf{(L.1)}$$

The results of rainfall intensity and kinetic energy can be seen in Table L.1. The maximum rainfall intensity received was in all but one case greater on the micro-plots than the field plots at both site locations. The one exception was during season four at Loddington where the opposite was true. As all maximum rainfall intensities measured were over 10 mm h^{-1} the maximum kinetic energy could be calculated. The relationship between spatial scales followed that of the maximum I_5 results; the kinetic energy was greater on the micro-plots except during season four at Loddington where *KE* was greater on the erosion field plots. The mean I_5 was also calculated, the results were greatly lower on the field erosion plots compared to the micro-plots. The rainfall intensities were below 10 and so the kinetic energy could not be calculated for average rainfall on the field plots.

Table L.1 Calculation of erosivity and rainfall intensity at both sites

Spatial scale	Season	Max I_5 (mm)	Max I (mm h^{-1})	KE > 10 (MJ ha^{-1} mm^{-1})	Mean I_5 (mm)	Mean I (mm h^{-1})
Site location: Loddington						
EP	1	1.40	16.80	0.022	0.25	2.94
MP	1		34.41	0.261		32.18
EP	2	2.20	26.40	0.250	0.24	2.90
MP	2		42.12	0.268		32.79
EP	3	1.60	19.20	0.231	0.27	3.24
MP	3		45.64	0.270		37.33
EP	4	4.80	57.60	0.276	0.26	3.09
MP	4		53.44	0.274		40.62
EP	mean	2.5	30.00	0.255	0.25	3.04
MP	mean		43.90	0.269		35.73
Site location: Tivington						
EP	2	3.00	36.00	0.262	0.26	3.09
MP	2		48.88	0.272		31.60
EP	3	3.20	38.40	0.265	0.24	2.88
MP	3		49.02	0.272		37.11
EP	mean	3.10	37.20	0.264	0.25	2.99
MP	mean		48.95	0.272		34.36
Site location: Both (gravity-fed rain tower)						
Aggregate	n/a		40.44	0.266		35.48

Appendix M Wind velocities

Site	Season/Mean	Mean velocity ($m\ s^{-1}$)	Maximum velocity ($m\ s^{-1}$)	Beaufort scale*
Loddington	1	1.86	4.93	
	2	2.63	9.44	
	3	1.96	5.48	
	4	2.31	7.36	
	mean	2.19		Gentle breeze
Tivington	1	0.84	3.24	
	2	1.47	6.12	
	3	1.34	4.66	
	mean	1.26		Just perceptible

*** scale obtained from www.metoffice.com**