



**ANALYSING THE EFFICACY OF LAND INTERVENTIONS
TO PREVENT AGRICULTURAL SOIL LOSS IN THE WEAR
CATCHMENT**

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[1] Introduction

Soil loss by water erosion is prolific and has called for in depth study in order to reduce the impacts of soil loss both on and off of agricultural land. On farms, soil erosion increases the need for fertiliser application due to removal of essential nutrients (Pimentel 2006), which has financial implications for land owners. Moreover, eroded material can cause contamination of local waterways, increasing flood risk and potentially endangering human and wildlife livelihood due to the presence of fertiliser chemicals (Pimentel 2006; Hillel 2003; Boardman 2010).

It is therefore necessary to understand how best to limit soil erosion from agricultural land to mitigate impacts and costs both on and off site. While there has been extensive study into erosion mitigation methods, there has not been any study of the Wear area specifically, including the sub catchment of Old Durham Beck. This investigation compares two land interventions which are available under the Sustainable Farming Incentive scheme; cover cropping and buffer strips to determine which is most suitable given the specific land conditions of the area.

[2] Review of Literature

2.1 Soil Erosion and Connectivity

Soil erosion is a key issue facing agricultural land today, creating a multitude of issues on and off site. Soil erosion and its subsequent impacts are a pressing area of study because the condition and prevalence of soils goes hand in hand with human and ecosystem health. As explored in extensive detail in Tate (2020), soil is an ecosystem in and of itself, providing a key environment for microorganisms as well as laying a foundation for other organism and processes such as agricultural plants. Soil also has large scale importance at the global level. Lal *et al.* (2021) identified soil as a key factor in eleven of the UN Sustainable Development Goals, including clean water, life on land, and zero hunger, that in conjunction with the idea of soil as a 'non-renewable resource' (Bazzoffi 2009) highlights the urgency with which soil erosion must be studied and counteracted.

There has been a great deal of research and evaluation of the impacts of soil erosion, with varying conclusions. On farms, the impact of soil erosion is generally discussed as the removal of valuable nutrients, hence reducing productivity and yield; and the cost associated with subsequent requirement for fertiliser application. Pimentel (2006) states that soil eroding from fields typically has three times the nutrients of the soil left behind, thus becoming a factor in impeded crop growth. However, other studies suggest that the effect soil erosion has on an individual farm and its yield is negligible (Boardman 1990), and that with the use of modern fertilisers, soil loss even at a rate of twenty five tons per hectare annually could be tolerated for more

than 200 years before crop growth would suffer significantly (Frost *et al.* 1990). This idea of soil-erosion tolerance has been debated quite extensively from its origins in the 1940s (Li *et al.* 2009), and indeed criticised quite harshly, notably by Johnson (1987) who called the idea of soil-erosion tolerance 'quasi-scientific folklore' (p.155). Johnson (1987) criticised the basis of many soil tolerance papers which could all be traced back to one paper which estimated soil-erosion and formation rates with rather dubious certainty. More recently, the idea of soil tolerance has been developed into a more holistic measure, taking into account not only the maximum soil loss which can be tolerated in terms of crop yield and soil formation rates, but also in terms of the impact on off-site environments and water quality (Li *et al.* 2009).

This relatively recent focus on off-site impacts of soil erosion is significant in drawing attention to an issue which results in extensive costs to individuals, companies and local governments. Eroded material from farms finds its way into streams and lakes, leading to contamination and in some cases flooding (Pimentel 2006). In central and western Europe especially, these transfers have caused significant damage, with 'muddy floods' (Boardman 2010) creating as much as €122M worth of damage to public infrastructure in Belgium alone. In England and Wales, the off-farm financial cost of soil erosion has been estimated to be more than one billion pounds per year (Hewett *et al.* 2018). Frost *et al.* (1990) highlight the difficulty of reconciling these off-site costs with on-site actions; whilst farmers may currently incur relatively little cost due to soil erosion, mitigation to prevent large off-site cost would have to occur on-site, increasing financial burden for farmers.

The physical process of soil erosion can be said to be a balance between erosivity and erodibility (Pimentel 2006). Hillel (2003) classifies erodibility as referring to the characteristics of the soil, such as texture, structure and aggregation, whereas erosivity refers to the aspects of the rainfall which drives erosion.

The erodibility of soil is a key consideration when studying erosion on agricultural land, as the land use according to yield and profitability is the key factor driving land decisions. The chemical characteristics of soil, such as soil organic carbon content can impact soil erodibility, increasing vulnerability. The study by Evrendilek *et al.* (2004) in Türkiye found a negative correlation between soil organic carbon and soil erodibility. This result is significant for agricultural land, as there is evidence to suggest that cropland has the lowest soil organic carbon content of other land types such as forest or grassland (Evrendilek *et al.* 2004; Masciandaro *et al.* 2018). There are also feedbacks at work further increasing agricultural soil erodibility, as erosion lowers soil organic matter content, leading to a chain of positive feedback of erosion and deterioration in soil organic carbon. Stavi and Lal (2011) also observed a feedback between erosion and erodibility, with erosion increasing soil erodibility, hence increasing vulnerability to subsequent erosion.

The concept of roughness also has a role in determining erodibility, with both surface roughness and topography playing a role in flow resistance (Smith 2014). At landscape scale, the topography will contribute to where flow patterns are created and the velocity at which runoff can flow, thus dampening the role of other roughness components such as vegetation (Pan *et al.* 2010). At the surface level, roughness can impact soil erodibility; uneven land allows the pooling of water which in turn dampens rainfall impact (Hillel 2003), thus reducing vulnerability to erosion. Conversely, roughness can also increase flow erosivity in some cases by channelising flow, creating features such as rills (Kiani-Harchegani *et al.* 2021), demonstrating the range of controls roughness has on landscapes. Vegetation can also play a role in increasing surface roughness therefore decreasing erodibility. Many studies have shown that surface runoff and consequently erosion is much greater for bare soil compared to vegetated areas (Heathwaite *et al.* 1990; Pimentel 2006; Hillel 2003; Frost *et al.* 1990).

On the other hand, erosivity refers to the ability of rainfall to detach soil particles and has strong connections to weather and climate. Erosivity can also encompass erosion by wind, however for this investigation, water erosion is of principle interest. Hudson (in Hillel 2003), breaks down the elements of rain most relevant to erosion: intensity, duration, and energy. How these factors interact has been studied by van Dijk *et al.* (2002) and Mineo *et al.* (2019), among others, to ascertain the best way to estimate erosion hazard. van Dijk *et al.* (2002) investigated rainfall kinetic energy as a key component of erosivity, though the duration and intensity of rainfall events must also be considered in order to gain an accurate picture of erosivity. Mineo *et al.* (2019) similarly highlight the importance of considering kinetic energy and intensity simultaneously. Both papers either explicitly or implicitly allude to the importance of the individual characteristics of a rainfall event, which ultimately controls the duration, intensity, and kinetic energy of rainfall. With an increase in extreme precipitation events due to climate change (Uber *et al.* 2024) it might be expected that erosivity, and thus soil erosion, will increase in the future due to anthropogenic climate change. The simulation of future erosivity by Uber *et al.* (2024) highlights the need to consider the representative concentration pathways in predicting soil-erosion, which arguably should also hold weight in land-use and soil-erosion related decision- and policy-making.

It has been elucidated here that soil erosion is dependent on both rainfall and soil characteristics. At the local and individual farm level, as is the focus of this investigation, the erodibility lends itself best to suggesting ways to limit soil erosion, as the land is the principal object over which farmers have control. Therefore, this review of literature will consequently focus on an examination of land interventions to reduce soil erosion, rather than ways to reduce anthropogenic related extreme weather events, though selected interventions should be tested against erosivity

parameters, reflecting the interconnectedness of erosivity and erodibility as discussed above.

Surface runoff, as briefly touched upon above, plays a significant role in the soil-erosion process, and there is a large volume of literature focusing on the hydrological aspects of erosion to support this idea. Increasingly, the idea of connectivity (Bracken & Croke 2007) has been used to understand hydrological systems holistically. Connectivity approaches have developed from distinguishing static and dynamic elements to the now more widely accepted distinction of structural and functional connectivity (Bracken & Croke 2007; Wainwright *et al.* 2011). The idea of connectivity stretches across multiple disciplines, with connectivity relating to soil erosion finding a middle ground between hydrological connectivity and sediment connectivity (Bracken *et al.* 2015). As Bracken *et al.* (2015) note, eroded material moves and is connected differently to the flow that moves it, though there are significant intersections, hence a case-by-case approach is likely to be the most successful.

Developments in connectivity theory come from gradual steps towards more representational theories and frameworks which seek to understand what is actually occurring in a system. This includes an appreciation of how structural connectivity: how landscape units are linked, is influenced by functional connectivity: how hydrological processes interact with the structural elements (Lexartza-Artza & Wainwright 2009). This consideration of dynamism is especially relevant for landscapes undergoing frequent change such as agricultural land.

2.2 Soil-Erosion Modelling

As soil-erosion often occurs at large scales, it is worthwhile to establish an effective representation of soil-erosion processes which can be manipulated to different scenarios to overcome the cost and practicality limitations of testing in the environment itself (Wainwright & Mulligan 2013). However, as Baird (2013) notes, this can be challenging due to the complexity of hillslope hydrology, and this challenge has been met with varying degrees of success in existing erosion models.

Soil-erosion models can generally be classified into four major types: empirical, conceptual, physical, and process oriented (Pandey *et al.* 2016). The empirical model method involved translating a process into measurable parameters and hence an algorithm which represents the relationship between these parameters (Alewell *et al.* 2019). This is the basis of the Universal Soil Loss Equation (USLE), upon which many subsequent erosion models have been based (Alewell *et al.* 2019; Pandey *et al.* 2016). However, the USLE has been criticised extensively, for instance for its consideration of gross rather than net erosion due to a lack of deposition in the model, which

results in model output data which does not match with measured field data (Alewell *et al.* 2019). Pandey *et al.* (2016) suggest that physically based erosion models overcome some of these issues by representing erosion and deposition processes more accurately, as well as allowing the application of the model to more complex conditions.

Models can also be categorised on the temporal scale, as event based or continuous (Pandey *et al.* 2016). Whilst Nearing (2006) highlights the importance of continuous simulation to effectively represent how erosional processes act and interact over time, there are also challenges to this approach. For instance, Batista *et al.* (2019) highlight the difficulty of establishing the initial conditions for a continuous simulation in order for it to be as representative as possible. Clearly, modelling erosion is a challenging task at all spatial and temporal scales, hence this area of research is constantly developing.

A step in this development has been the proposal of the event-based Model for Assessing Hillslope Landscape Erosion, Runoff and Nutrients (MAHLERAN) (Wainwright *et al.* 2008). This model was developed from a criticism of the reliance on suspension as the dominant mode of transport in hillslope erosion models, despite this form of transport occurring relatively rarely (Wainwright *et al.* 2008). Other specific elements were also included such as parameterisation of vegetation and classification of sediment to subsequently influence transport method (Wainwright *et al.* 2008), which as Pandey *et al.* (2016) discuss, the physical characteristics of a landscape are important considerations for erosion modelling.

2.3 Interventions

Owing to the importance of soil to human functioning and wellbeing, it is no surprise that a range of interventions to prevent soil erosion have been investigated and presented, though no one singular method or approach will be suitable for all scenarios and landowners. Considering a connectivity-based approach, land interventions which reduce the connectivity of the landscape by interrupting and slowing runoff pathways will be the focus of this section of the review.

One successful intervention to prevent the connectivity of sediment to outflow pathways is the application of Polyacrylamides (PAMs) to agricultural land. The majority of PAM studies originate in the USA in the 1990s and early 2000s, and are shown to be successful at reducing surface crusting and encouraging the aggregate stability of soil particles to reduce erosion (Green & Stott 2001; Entry *et al.* 2002). Despite some concerns over the safety of PAMs to human health, Ahn & Castle (2003) conducted a study which concluded there was little evidence to suggest any negative human health implications from PAM-contaminated water. This is a clear advantage to this intervention method, given that potential health implications of

pollution are a key concern for managing soil-erosion. Despite the apparent lack of limitations of PAMs as a soil-loss intervention method, there is little evidence of use in the UK other than an ongoing project studying the possibility of biopolymers as a more sustainable replacement to PAMs (Escudero Olabuenaga *et al.* 2024). This perhaps suggests that the environmental implications of the polymer aspect of PAMs have been a deterrent to use in the UK. Moreover, Sepaskhah and Bazrafshan-Jahromi (2006) identify that while PAMs are effective at reducing soil loss, they do little for reducing the underlying issue of runoff. Consequently, PAMs do not target a reduction in holistic connectivity, and allow runoff to continue despite reducing the volume sediment being transported with it.

Increasing vegetation cover and surface roughness is another approach to reducing soil erosion, which targets runoff and therefore overall landscape connectivity, rather than just the loss of soil itself. Many studies evidence that increased vegetation cover can reduce runoff velocity and detachment rates, though effectiveness can vary with factors such as slope gradient, percentage cover and particle size (Liu *et al.* 2008; Li & Pan 2018; Kaspar *et al.* 2001; Pan *et al.* 2015). Increasing vegetation cover on agricultural land can take different forms, including buffer strips and cover cropping, both of which will be discussed further below.

Vegetated buffer strips are a widely implemented and researched intervention to reduce soil erosion, focusing on reducing runoff (Lerch *et al.* 2017; Zhang *et al.* 2010; Li *et al.* 2009). Increased vegetation coverage has been linked to reduced raindrop energy and runoff volume and velocity (Pan *et al.* 2016; Lerch *et al.* 2017), which in turn enhances infiltration, allowing an increase in sediment deposition (Liu *et al.* 2008; Lerch *et al.* 2017). These effects are important for reducing off-site erosion impacts such as reduced water quality, negative impacts on aquatic ecology and increased on-site costs related to fertiliser application (Lerch *et al.* 2017; Pimentel 2006).

Despite these observed effective outcomes, Liu *et al.* (2008) emphasise that there is no universal best practise for buffer strips, as efficacy will vary due to site-specific complexities. However, in their review of a range of buffer studies (Liu *et al.* 2008), it was shown that buffer slope and width had the most bearing on buffer efficacy across the board. Other studies concur with this finding, with general trends suggesting that steeper slopes decreased the trapping efficiency (Pan *et al.* 2010), and strips of greater width trap the most sediment (Lerch *et al.* 2017; Zhang *et al.* 2010). The effectiveness of wider strips is to do with particle selectivity due to particle size, with higher percentage covers of grass reducing particle selectivity (Pan *et al.* 2016). The site-specific nuances of buffer strip efficacy can be understood in terms of source to buffer area ratio (Lerch *et al.* 2017), which takes into account the proximity of the buffer strip to the source of contaminant, or key sediment loss area, with regards to the width of the buffer. By taking this ratio into account, it should be possible to find

the optimum buffer width which is wide enough to be effective for fine sediment while leaving as much land as possible for production as not to negatively impact production profits for the farmer (Liu *et al.* 2008).

To reach a consensus with regards to the source to buffer area ratio, it is necessary to understand key sediment source areas of the highest risk, to target buffer interventions most effectively. The focus on 'critical source areas' by Reaney *et al.* (2019: 4) is a way of determining the best placement of mitigation measures, depending on the intersection of high source risk and high connectivity areas. A tool for understanding such intersection is the Sensitive Catchment Integrated Modelling and Analysis Platform (SCIMAP) (Reaney *et al.* 2019), which calculates spatial patterns of erosion potential and connectivity to understand erosion risk using publicly available data. SCIMAP as a desktop based platform is advantageous in that it requires no on the ground surveying to produce the risk maps, and can be deployed for a range of users and their needs (Reaney *et al.* 2019), which is essential for an issue which is a concern for multiple stakeholder groups such as farmers, landowners and local government.

Cover cropping is another intervention method related to increasing surface roughness and slowing runoff to reduce soil erosion. This intervention method has been in circulation for a significant length of time, from ancient farming methods in North America to the idea of crops for replenishing the soil used in the 18th century (Groff 2015). However this method was largely lost and is only gaining traction again in the last 40 to 50 years (Groff 2015), including a 50% cover crop use increase in the US between 2012 and 2017 (Wallander *et al.* 2021). Principally for this investigation, cover crops provide both above and below ground interventions in the generation of runoff and soil detachment (De Baets *et al.* 2011; Kaspar *et al.* 2001). Hydrologically, the velocity of surface runoff is important in determining the transport rate and volume of surface material eroded as it is related to the erosivity and detachment capability of the flow (Pan *et al.* 2015; Takken & Govers 2000). Therefore, the ability of cover crops to reduce the velocity of overland flow by increasing infiltration is key to understanding the effectiveness of cover cropping in reducing soil erosion (Kaspar *et al.* 2001).

There has also been research into which plants, and which plant components are most effective at reducing erosion. The study of De Baets *et al.* (2011) investigated six different cover crop species of different root structure to find that the species with higher root density with fine branching roots were more effective at reducing erosion than plants with thicker roots and lower root density due to improvements in soil stability. This below ground factor was found to be especially important during frost periods when above ground biomass was likely to be damaged (De Baets *et al.* 2011). This idea of a dense root system as beneficial is confirmed by Li and Pan (2018), who found that of the species tested, the plant with the thickest canopy and most dense

root was the most effective at reducing erosion. Nevertheless, all grasses tested significantly reduced overland flow rate from the no grass state (Li & Pan 2018), suggesting any grass cover is better than none. Aside from reducing erosion, cover cropping also has other benefits for soil chemistry and biology such as nitrogen fixation, carbon sequestration and maintenance of soil microorganisms (De Baets *et al.* 2011).

2.4 Initiative Schemes

Consideration of the incentives and support available to farmers to implement intervention methods is also important for assessing the feasibility and overall effectiveness of a particular method. The main scheme available for UK farmers is the Sustainable Farming Incentive (SFI), which offers payment to farmers for carrying out environmental land-management practises, under which reducing soil erosion fits. The most recent SFI report (Department for Environment, Food & Rural Affairs 2023) focuses on a flexible approach, allowing farmers to mix and match different management practises to suit their land. The actions related to soil erosion which are granted funding include: producing a soil management plan, multi species cover crop planting and buffer strips (Department for Environment, Food & Rural Affairs 2023). Buffer strips provide greater payment than cover crops: currently £515 per hectare compared to £129 per hectare for cover crops, though they must be between 5 and 12 metres wide (Department for Environment, Food & Rural Affairs 2023). The report also stipulates that buffer strips must buffer an existing feature, effectively delineating a field edge, for instance a hedgerow, wall, woodland, waterway or trackway (Department for Environment, Food & Rural Affairs 2023), which could dissuade in-field buffer strip placement due to lack of incentive.

Other incentive schemes include the Countryside Stewardship scheme, which offers grants for a range of actions, some of which overlap with the SFI. Notably, grants are available for buffer strips both at existing boundaries, and in-field buffer strips, with in-field strips offering £250 more per hectare than at field boundaries (GOV.uk 2024).

2.5 Relevance to this Investigation

This literature review has highlighted that the issue of soil erosion, especially on agricultural land is of great importance, due both the implications on farm profits associated with continued fertiliser application, as well as on off-site water quality and human and ecosystem health. The main interventions identified here have been PAM application, buffer strips and cover cropping. While there has been extensive research into these methods in a range of case studies, there is yet to be investigation into agricultural soil erosion in the Wear catchment, and more specifically Old Durham Beck. After some conversation with the Wear Rivers Trust, it seems that although farmers may be aware of soil erosion as an issue, they are unlikely to

change common practise if there is not a strong incentive to do so, such as monetary compensation through a fund such as the SFI or perhaps a specific local study of soil erosion.

This investigation will focus on buffer strips and cover crops as soil loss interventions as they are both mentioned in SFI incentive schemes, whereas PAMs are not mentioned, and from the literature do not appear to be used widely in the UK currently, which could provide another barrier to use.

[3] Methodology

The methodology for this research had a main component of running the MAHLERAN model, and a supplementary generation of a SCIMAP sediment risk map to understand the areas of interest in the Wear catchment.

Producing a sediment risk map for the Wear catchment was the first component for this project. In order to generate the final map, pre-processing was carried out in the QGIS software to produce the Digital Terrain Map (DTM), average rainfall, and landcover map which was delineated to the Wear catchment. These files were then used as the input for the SCIMAP programme (Reaney 2016) which runs within SAGA GIS. SCIMAP enabled the production of a final sediment risk map for the Wear catchment, assessing the connectivity of the catchment to identify key source risk areas within the catchment. With key high-risk areas identified, this was then used along with considerations such as data availability and site access to decide the location which would be used for running the MAHLERAN model. The area chosen was the field adjacent to Benthouse Lane in the Old Durham Beck catchment.

QGIS was used again to produce the input files running the model, which required spacial data such as vegetation cover, a Digital Terrain Model (DTM) as well as numeric data which included the rainfall data itself. This was selected from the large dataset of rainfall from Durham Observatory, with two storms of differing length and intensity chosen to simulate different conditions. Each model run took a significant number of hours to complete.

Once the model run had completed, output files could be read into the platform R to visualise the results.

[4] Results

4.1 SCIMAP

Figures 1 and 2 show the Wear catchment erosion risk map at two different scales created using SCIMAP (Reaney 2016). High risk areas appear in red, and low risk areas appear in green. From these figures it is clear that across the whole catchment, higher risk areas are found surrounding Durham city, particularly to the west. Figure 2, which shows Durham city in more detail, highlights some dark red high-risk areas either side of the A1 to the east of the city (including the field studied in this investigation), as well as to the northwest of the city.

4.2 Flow Patterns

Figure 3 shows the output map of peak flow velocity demonstrates key flow areas across the area studied. Three key flow lines dominate the figure, running approximately north to south across the field.

4.2 High intensity rainfall event

Figure 4 gives an overview of the sediment produced in each model run for the high intensity rainfall event. For every intervention scenario, 4.68kg (2d.p.) of sediment was lost in total. There were marginal differences between each intervention method, and overall volume of sediment produced appears to increase with the width of the buffer strips. Cover crops, represented using 0.25 and 0.5 vegetation cover, produced the most sediment output of 4.682kg (3d.p.).

Figures 5-9 show the sediment loss and discharge plotted against time for each vegetation scenario for the duration of the high intensity rainfall event. The event lasted 5 hours with a total rainfall of 18.2mm; peak discharge was reached after approximately 10,500 seconds (almost 3 hours). Sediment loss increased rapidly from just under two hours into the rainfall event and peaked at the point of peak discharge before reducing at an uneven rate back to 0kg s⁻¹ after approximately 4 hours.

4.3 Low intensity rainfall event

Figures 10-12 show the sediment loss and discharge plotted against time for each vegetation scenario for the duration of the low intensity rainfall event. The event lasted 17 hours with a total rainfall of 8mm; peak discharge was reached after approximately 5000 seconds (almost 1.5 hours). For the buffer strip scenarios, all sediment loss occurred in the first 15 000 seconds. Overall, the total sediment output for the low rainfall event was much lower than for the high rainfall scenario, at less than 0.001g output. Conversely to the shorter rainfall event, cover cropping produced 2x10⁻⁹kg less total sediment than the buffer strip scenarios, as shown in figure 13.

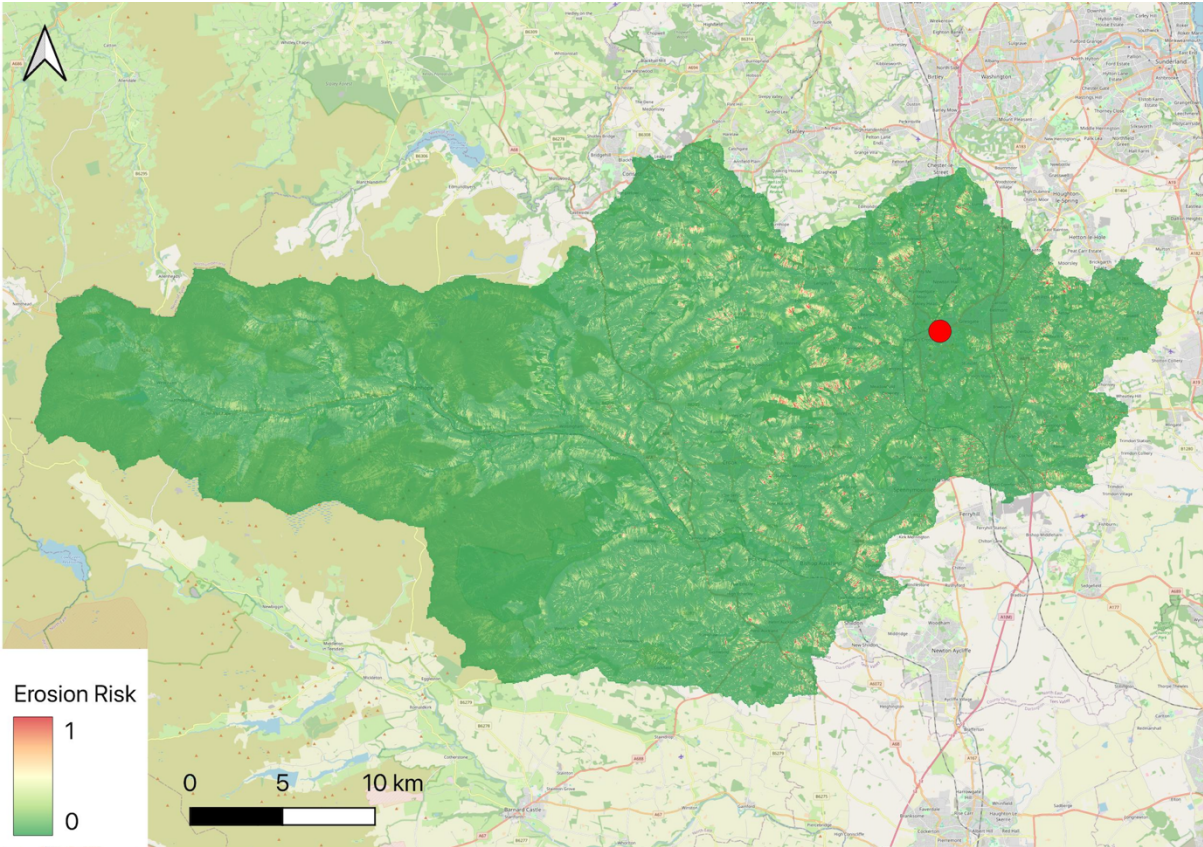


Figure 1 - SCIMAP risk map for the Wear catchment. Map created using QGIS 3.36

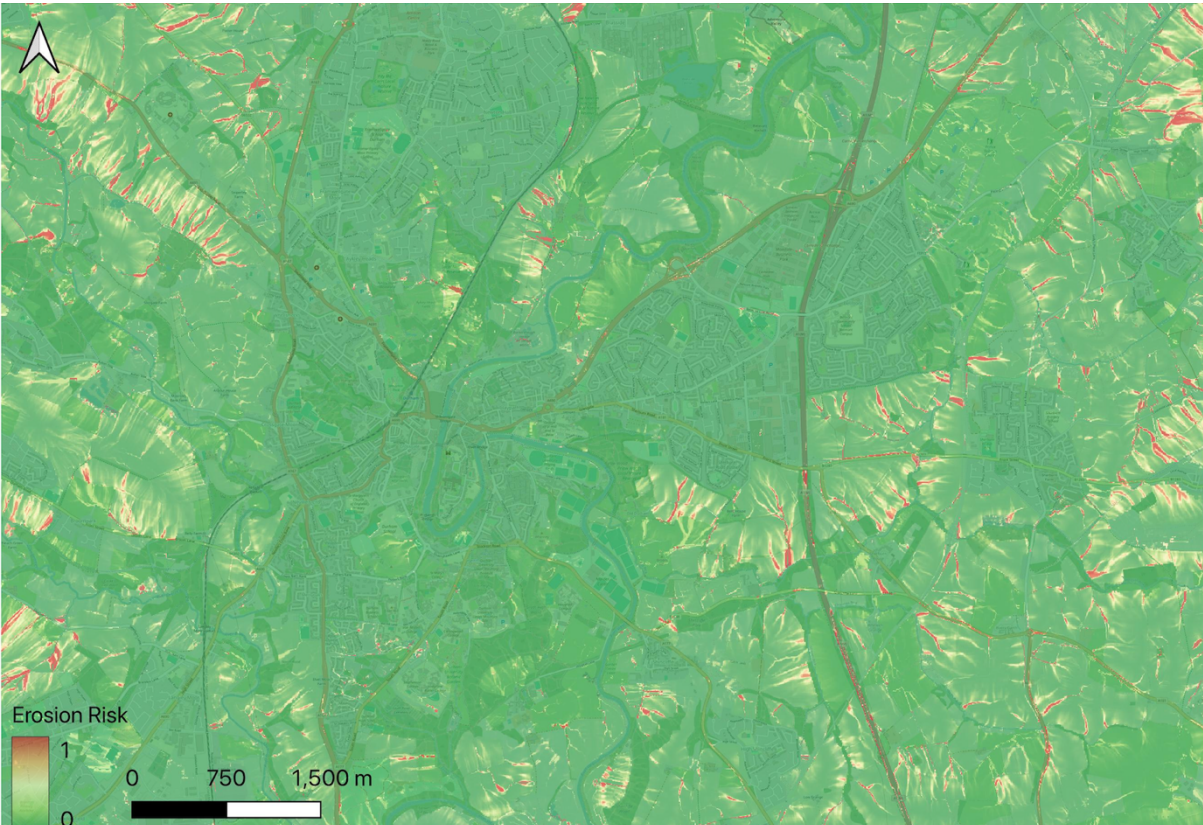


Figure 2 - SCIMAP risk map for Durham City. Map created using QGIS 3.36



Figure 3 - Map showing peak flow velocity output from MAHLERAN model
Created using QGIS 3.36

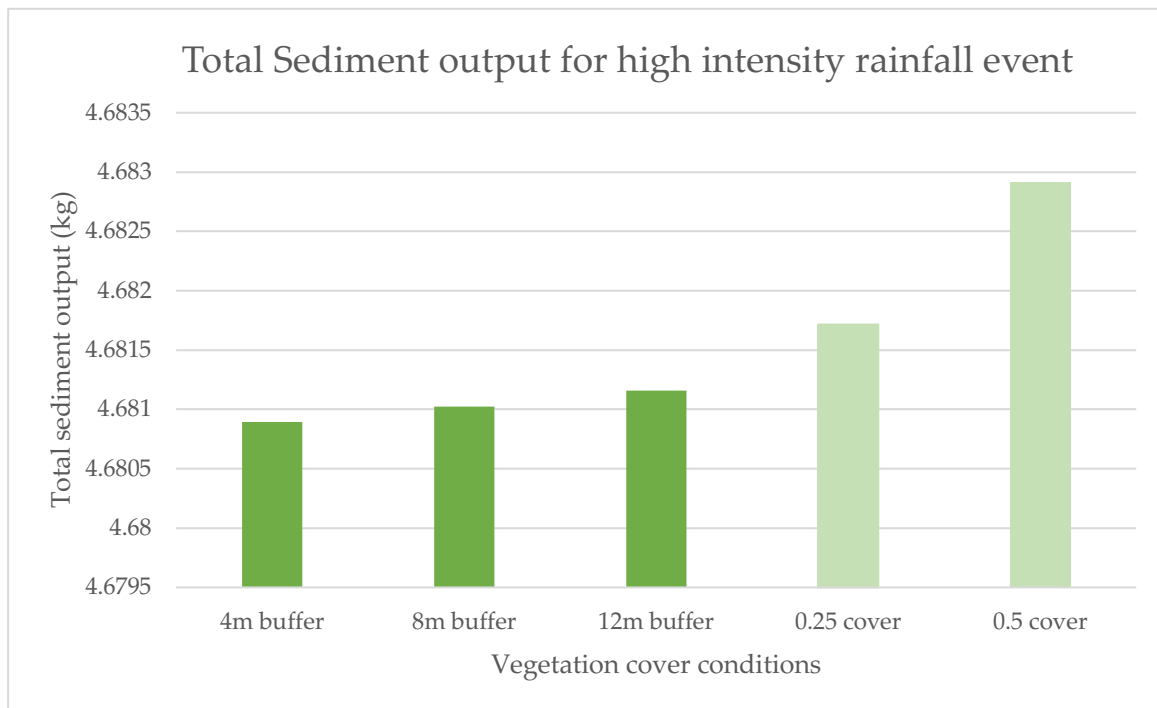


Figure 4 - Graph showing total sediment output for the high intensity rainfall event

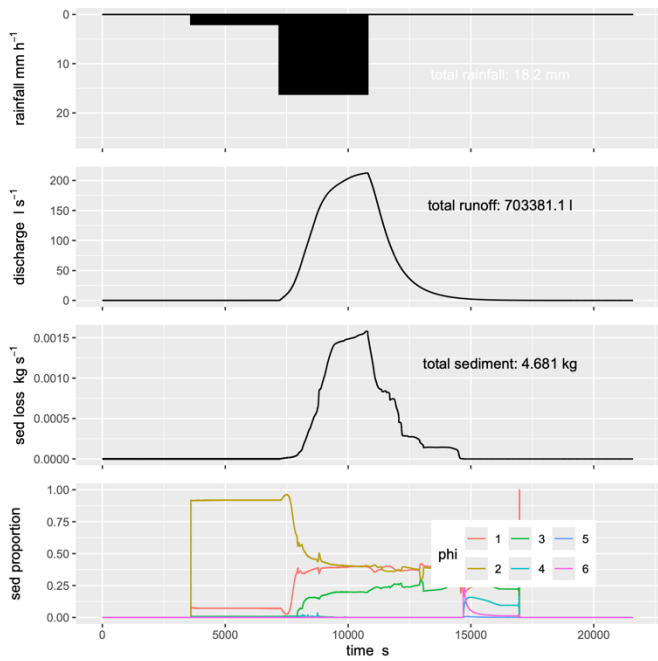


Figure 5 - Output graphs for 4m buffer scenario, high intensity rainfall event

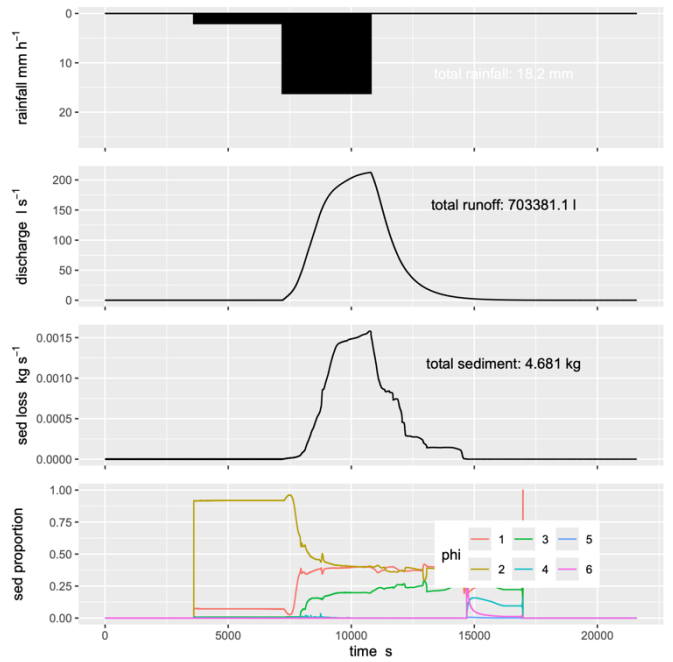


Figure 6 - Output graphs for 8m buffer scenario, high intensity rainfall event

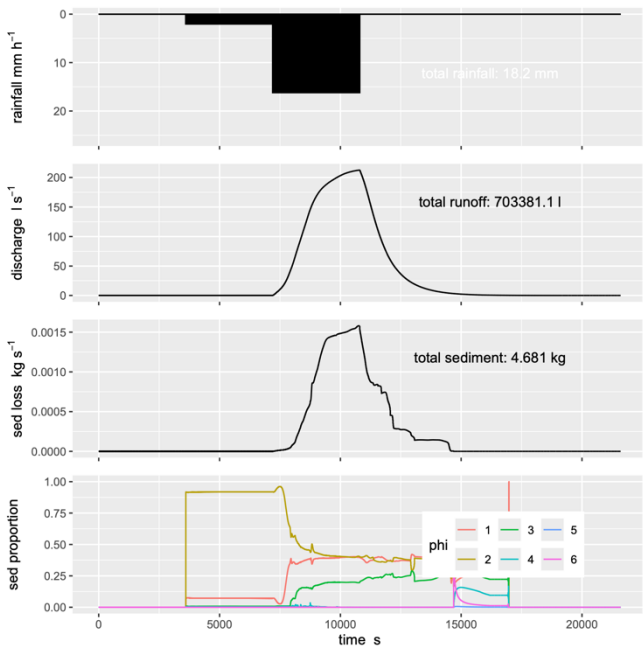


Figure 7 - Output graphs for 12m buffer scenario, high intensity rainfall event

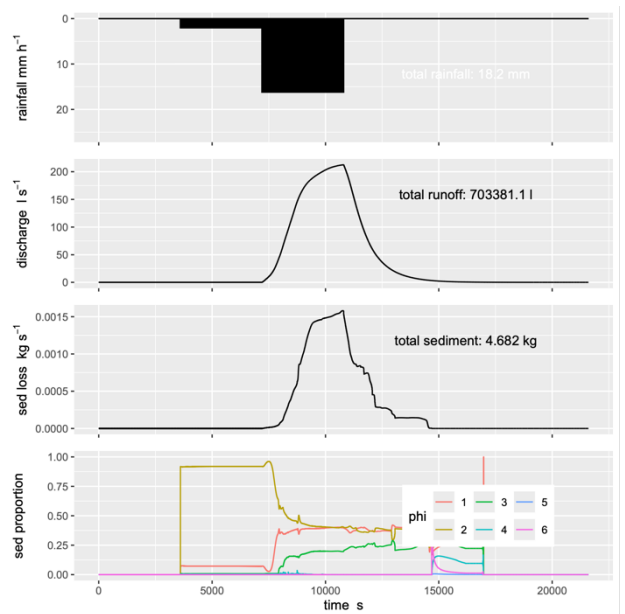


Figure 8 - Output graphs for 0.25 cover crop scenario, high intensity rainfall event

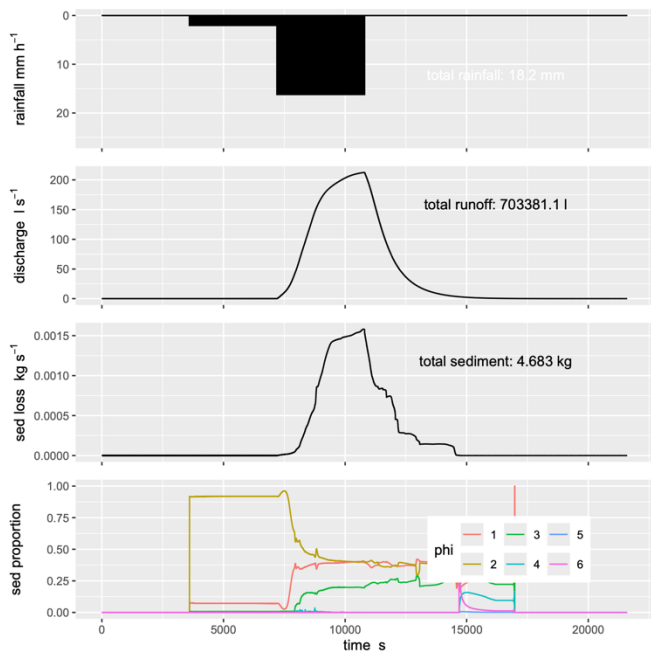


Figure 9 - Output graphs for 0.5 cover crop scenario, high intensity rainfall event

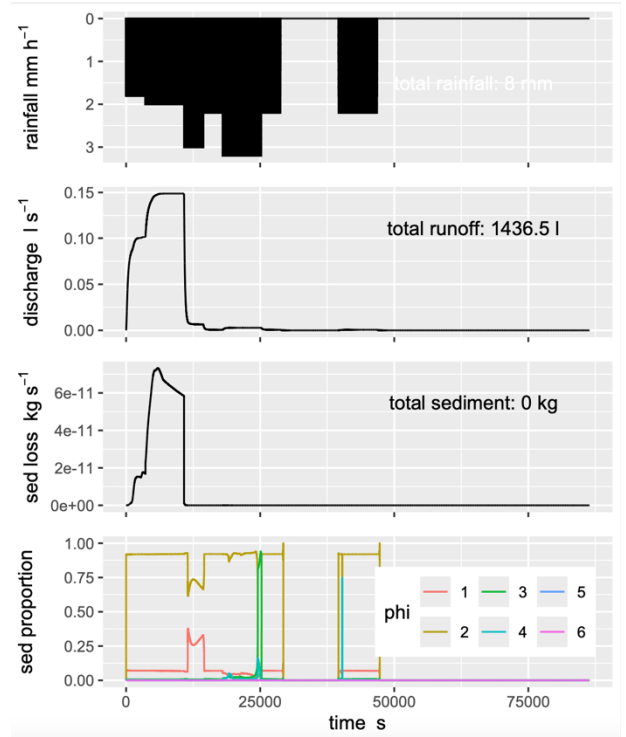


Figure 10 - Output graphs for 4m buffer scenario, low intensity rainfall event

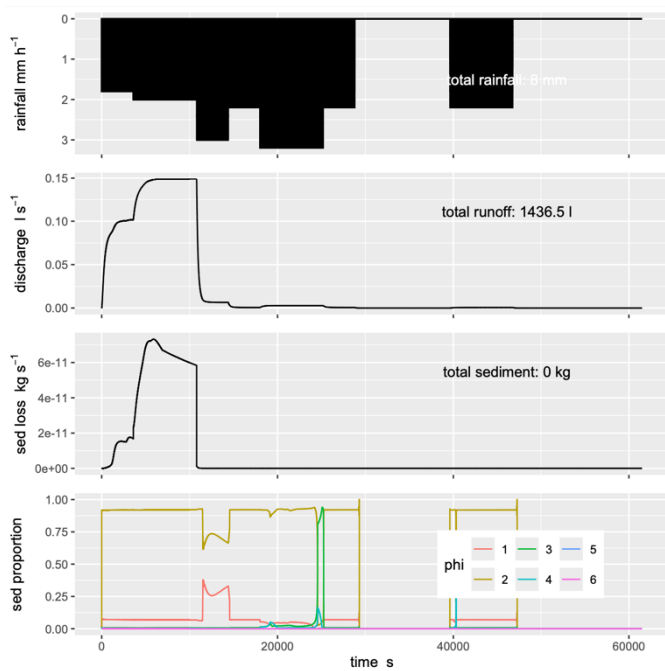


Figure 11 - Output graphs for 8m buffer scenario, low intensity rainfall event

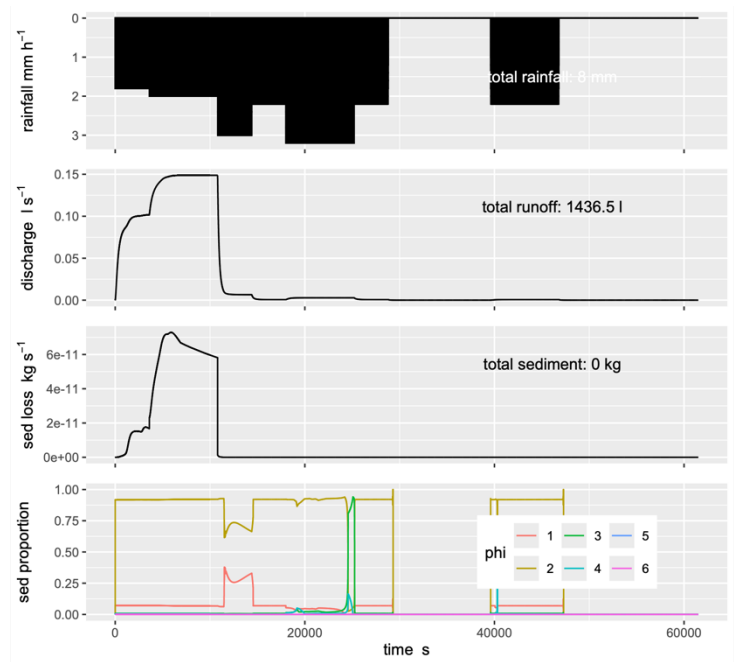


Figure 12 - Output graphs for 0.25 cover crop scenario, low intensity rainfall event

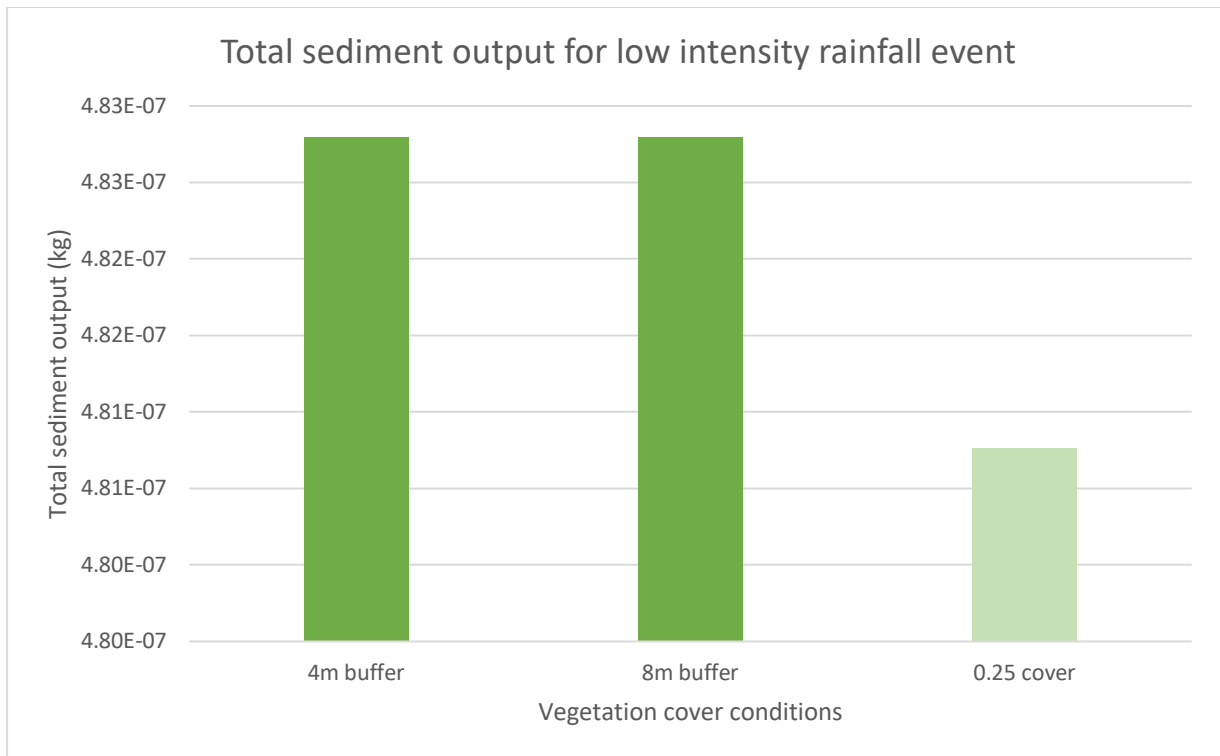


Figure 13 - Graph showing sediment output for low intensity rainfall event

[5] Discussion

The results in this report do not show an overwhelmingly clear pattern in order to confidently conclude which intervention method is most effective at reducing soil loss. However, for the high intensity rainfall event, the results suggest that buffer strips are slightly more effective than cover crops. For the low intensity event, cover crops appeared to be slightly more effective than buffer strips, though hardly any sediment was lost at all, indicating that perhaps for this particular field, the duration and intensity of the rainfall conditions is more important in determining soil loss than the intervention methods studied in this investigation.

5.1 Sediment loss for high intensity rainfall event

All runs for the high intensity rainfall event, regardless of intervention type, show a very similar pattern of runoff and sediment generation, with peak discharge and sediment loss being reached towards the very end of the rainfall event. This suggests a time lag between the rainfall and the generation of discharge and consequently sediment loss which demonstrates a similar principle to the delays between storm peak and peak channel discharge in river hydrology (Holden 2018). The decrease in sediment loss after the rainfall is more steady than the initial increase, and demonstrates a less linear relationship. The consistency between output patterns for each vegetation scenario perhaps suggests that for this rainfall event, the rainfall intensity overrode the intervention methods in place, allowing soil loss to occur at a magnitude of 4kg total loss.

Buffer strips

The pattern of increasing sediment loss with buffer strip width is quite surprising, and inconsistent with the literature which suggests that wider strips generally trap the most sediment (Lerch *et al.* 2017; Zhang *et al.* 2010). However, the differences in sediment loss between the three buffer strip outputs are less than 0.001 kg s^{-1} , or 1g, perhaps suggesting this difference is not stark enough to draw any clear conclusions about preferable buffer strip width in this instance. In further investigations, it may be useful to run the model again with buffer strips within the field as well as at the edges to see if this would yield results which differ more substantially. It may also be the case that for this specific location and rainfall event, all buffer strip widths tested were not wide enough to make any substantial difference to erosion, so had there been more time to complete further runs, larger buffer strips could also be considered.

Cover crops

Both cover crop scenarios yielded greater sediment loss overall than any of the buffer strip scenarios, suggesting cover cropping is a less effective method of reducing soil erosion for this high intensity rainfall event. This observation is

supported by the differing payment schemes awarded for the two techniques under the SFI; with buffer strips granting a larger payment per hectare than cover crops (Department for Environment, Food & Rural Affairs 2023), presumably due to the greater good for the soil and local environment of buffer strips. However, it must be noted that the difference between total sediment loss for the buffer strips and the cover crops is between 0.001 and 0.002 kg, which may be too small of a difference to be considered in decision making. Nonetheless, this rainfall event lasted only 5 hours, so future investigations should also run longer intense events to see whether differences between intervention methods become starker with prolonged events.

Interestingly, the greater percentage cover scenario yielded a greater total soil loss than the lower percentage cover scenario. This is inconsistent with the literature which suggests greater vegetation cover should reduce soil erosion (De Baets *et al.* 2011; Kaspar *et al.* 2001). However, this investigation did not include the adjustment of infiltration inputs of the MAHLERAN model; doing so may have allowed a more accurate representation of vegetation cover by taking root structure into account. Moreover, whilst the difference between soil loss values for the two different percentage cover crop scenarios was larger than for the buffer strip scenarios, it was still only 0.0012kg s⁻¹, suggesting that any difference between the two may be negligible, certainly to the human eye.

5.2 Sediment loss for low intensity rainfall event

The total sediment loss for the low intensity rainfall event was much lower than for the high intensity rainfall event, at only a negligible amount above zero for the buffer strip scenarios. This overall pattern is expected as many studies show that higher rainfall intensity yields higher runoff, therefore more erosion (van Dijk *et al.* 2002; Mineo *et al.* 2019). Other studies concerned with particle size distribution also demonstrate that high intensity rainfall often results in coarser sediment being eroded (Xiao *et al.* 2024), which would likely have a greater mass, leading to a higher total eroded sediment mass.

However, the very low total sediment loss perhaps suggests that this rainfall event was not the most suitable for understanding the most effective intervention methods as such a low number makes data presentation relatively difficult. For instance, figures 10 and 11 require such a small scale, which makes the total sediment loss change through time appear larger than in reality.

Buffer strips

Unfortunately, due to time constraints and the extended length of time it took to run the low intensity rainfall event, only the 4m and 8m buffer scenarios could be run for this event. Unlike for the high intensity event, there was no difference between sediment output for the different buffer strip scenarios. This could be either due to

the total runoff being too low to reach a threshold of extended erosion, or simply because the model does not have great enough resolution to present any miniscule differences in the two runs. This could alternatively suggest that at low rainfall intensities, buffer strips are more effective at limiting soil loss to a baseline level, though a full set of runs, including the 12m buffer strip scenario would be needed to support this conclusion.

Unlike for the high intensity rainfall event, the sediment erosion rate fell much more rapidly after peak erosion had been reached, and this decrease occurred during the period of rainfall, not after. Interestingly, the peak in sediment mass was reached before the time of peak rainfall, which does not display the same pattern of delay as for the high intensity rainfall event. This could be to do with the low intensity event having a longer period of rainfall before the peak is reached, which could have cause saturation excess overland flow (Holden 2018) to occur cumulatively over time as opposed to rapidly following heavy rainfall.

Cover crops

Unlike for the high intensity rainfall event, the cover crop scenario had a lower total sediment output than the buffer strips for this rainfall event, though only 2×10^{-9} kg less. Whilst this is a very low number, scaled up over a whole year and many prolonged rainfall events, this could have a significant impact on total annual sediment lost from the field. This result suggests that for this lower intensity, longer rainfall event, cover cropping was more effective at reducing sediment loss. However, as only one percentage cover (25%) was able to be tested for this rainfall event, it is not clear whether increasing cover would continue to reduce erosion. Due to the conflicting results between rainfall events, it would be helpful to conduct further investigation with a larger range of rainfall scenarios, to better understand the uncertainty of this investigation and ascertain whether there is a clear superior intervention method, or whether a combination of both should be adopted to account for a range of rainfall scenarios.

5.3 Final discussion

Despite not being able to be draw clear conclusions regarding an intervention scenario more effective than the rest, it is still interesting to note the total volume of sediment that can be lost in just one storm event. Over 4kg in 5 hours shows the potential impact a series of subsequent high intensity events could have on both the natural ecosystem, and the farmland itself. This mass of eroded material could easily make its way into local waterways, which are well connected in this area as found in the SCIMAP exercise (see fig. 2), and could consequently contribute to flooding in the area (Pimentel 2006; Boardman 2010). This is a key consideration for this area, given that there was high consistency between the high-risk areas identified using

SCIMAP, and the flow patterns in figure 3. Therefore, despite this investigation not finding a definitive conclusion about land intervention best practise, it is worth investigating further to reduce future risk to the local environment.

[6] Conclusion

Despite this investigation not finding a definitive conclusion about land intervention best practise to reduce soil loss, some interesting observations can be made. Running the MAHLERAN model demonstrated the flow patterns as predicted by the SCIMAP risk map, which indicates that the SCIMAP tool could be used across this sub catchment accurately; although other fields could be used in MAHLERAN to confirm this. This could be an invaluable tool for local farmers to understand where on their land is most at risk of erosion, or to understand how their landscape is connected to local waterways.

It is also clear that the high intensity rainfall event produced much more sediment than the low intensity event, giving an idea of just how much sediment could be entering into local waterways in the Old Durham Beck catchment – at least 4kg for one 5-hour storm. This information could not only prove interesting for local stakeholders but could also prompt behavioural changes in terms of adopting SFI management techniques.

And finally, the conflicting results regarding which intervention is most effective suggests that different rainfall scenarios could be best dealt with in different ways, which opens opportunity for further study using more rainfall examples. Although the numbers produced were only marginally different, the results of this investigation do suggest that buffer strips are more effective for high intensity rainfall, but cover cropping is more effective for low intensity rainfall. This information could also support campaigns for change in practise and adoption of a combination of new management techniques.

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