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Pulley, S., Morten, C., Morgan, S., Cardenas, L. M. and Collins, A. L. 2021. Sediment detachment by raindrop impact on grassland and arable fields an investigation of controls . *Journal of Soils and Sediments.* https://doi.org/10.1007/s11368-021-03098-4

Sediment detachment by raindrop impact on grassland and arable fields: an investigation of controls

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Abstract

Purpose: Modern agricultural practices have increased the losses of fine sediment and associated pollutants to watercourses with associated off-site degradation of water quality and aquatic biodiversity. Despite significant investment into agri-environment initiatives which aim to reduce these losses, limited empirical mechanistic evidence exists for the efficacy of many on-farm interventions. As the most likely mechanism of particle detachment in many landscapes, understanding the controls for rainsplash erosion is key to generating this mechanistic understanding.

Methods: Soil properties were compared to rainsplash erosion rates on a grassland and an arable field in Southwest England. Soil cores were retrieved and measured for dry density, aggregate stability, vegetation cover and loss on ignition. A rainsplash erosion trap consisting of a plastic funnel containing filter paper was placed into the hole left by each core and the mass of sediment trapped over a one month and two-week period was recorded.

Results: Soil compaction on the grassland field was limited to a very small proportion of the total field area at its margins and close to water troughs and was not often associated with a reduction in aggregate stability. Neither soil dry density nor aggregate stability were associated with an increase in rainsplash erosion rate. On arable land, aggregate stability was significantly lower than on the grassland and rainsplash erosion rates were higher. However, vegetation cover, rather than aggregate stability, was the major controlling factor.

Conclusion: The hypothesis that livestock are not causing an increase in erosion by raindrop impact on grassland fields but the same soils, when cultivated, experience a significant increase in erosion rate can be accepted based on the results generated. In lowland grassland landscapes with recommended best grazing management practices currently implemented further reductions in

sediment losses may therefore not be achievable. However, a reduction in elevated soil loss from arable fields can be achieved by ensuring vegetation cover is present during wet winter months.

Keywords: splash erosion, agriculture, mitigation, ruminants, cereals

1. Introduction

Soil degradation and its on-site and off-site impacts have been recognised as a global problem (Lal and Moldenhauer 2008; Borrelli et al. 2017). The onset of modern intensive farming has been associated with a significant increase in soil losses to water, especially in the case of arable systems (Rose et al. 2010; Foster et al. 2011; Collins and Zhang 2016; Cerdan et al. 2020). Importantly, however, approximately 20% of the world's pastureland is estimated to be degraded in some form (Steinfeld et al. 2006). It is estimated that 0.83 million km^2 of pasture worldwide is degraded through overgrazing causing the defoliation and trampling of soils (Oldeman 1994).

On a well-vegetated temperate grassland, interrill erosion caused by raindrop impact particle detachment is the most likely mechanism for soil losses (Borst and Woodburn 1942; Young and Wiersma 1973; Parsons et al. 1994). Raindrop impact can exert a pressure of up to 10^5 Pa on the surface of the soil (Ghadiri and Payne 1981), which disrupts soil aggregates and causes the ejection of detached discrete particles and smaller aggregates from its surface (Auerswald 1998; Le Bissonnais 1996). Soil aggregate stability, antecedent moisture, shear strength, slope, vegetation cover and raindrop properties all influence soil susceptibility to particle detachment (Bubenzer and Jones 1971; Al-Durrah and Bradford 1981; Riezebos and Epema 1985; Moore and Singer 1990; Slattery and Bryan 1992). The dispersion of soil by raindrop impact alone is, however, not sufficient to cause sediment loss and overland flows able to transport the detached particles out of the field are also required (Parsons et al. 1994; Hardy et al. 2017; Pulley and Collins 2019). As a result, the compaction of soils can also increase runoff and consequently soil loss (Bilotta et al. 2007).

Raindrop properties and slope represent factors which cannot be controlled through farm management. However, other factors such as improved aggregate stability, adequate vegetation cover and reduced soil compaction form part of good land management (Newell Price et al. 2011). Compaction is caused by the pressure exerted by livestock on the soil surface, which in the case of cattle, can be a static stress of up to 200 kPa (Di et al. 2001; Bilotta et al. 2007). The introduction of larger farm machinery in both grassland and arable systems has also been associated with increased soil compaction (Keller et al. 2019). Compaction reduces the volume of soil macropores, leading to an increase in bulk density, impeded vegetation growth, reduced water infiltration and storage capacity and elevated surface runoff (Hamza and Anderson 2005).

Wet aggregate stability represents soil resistance to disintegration and dispersion in water (Arshad and Coen 1992; Chan et al. 2001). Soil tillage has been associated with a significant reduction in aggregate stability (Elliott 1986; Kay 1990; Lal et al. 1994) which contributes to an increased risk of erosion (Montgomery 2007; Evans et al. 2017; Benaud et al. 2020; Cerdan et al. 2020). Trampling by livestock may also disrupt soil aggregates (Johnston 1962; Warren et al. 1986b; Proffitt et al. 1995; Broersma et al. 2000; Ferrero and Lipiec 2000; Shah et al. 2017); as such, negative correlations have been reported between aggregate stability and soil bulk density (Cattle and Southorn 2010; Stavi et al. 2011; Cox and Amador 2018). High grass sward cover and soil organic matter contents are typically associated with increased aggregate stability (Oades 1984; Cox and Amador 2018).

Vegetation cover is also heavily impacted by grazing and trampling by livestock. A loss of vegetation increases the proportion of a fields surface exposed to raindrop impact increasing erosion risk (Busby and Gifford 1981; Sanjari et al. 2009). Piospheres are a widely observed phenomena where grazing pressure around a watering point creates a gradient of pressure on vegetation and soils which increases with proximity to its centre (Thrash and Derry 1999; Hart et al. 1993).

Reducing soil compaction and increasing sward cover through measures such as changing the length of the livestock grazing season, reducing stocking rates, periodically moving drinking troughs and feeders and alternative livestock grazing methods such as rotational grazing are recommended to improve soil structure (Holechek et al. 1999; Newell Price et al. 2011). Additionally, where soils are identified as being heavily compacted, loosening the affected layers has been reported to have moderate to high uptake rates by farmers (Newell Price et al. 2011; Natural England 2019). Maintaining and enhancing soil organic matter levels to maintain good soil structure, fertility and aggregate stability is also commonly recommended as good practice on both arable and grassland farming systems (Newell Price et al. 2011; Natural England 2019). These measures are, however, associated with significant capital and labour costs to farm businesses (Gooday et al. 2014).

Whilst some studies have shown significant increases in soil loss associated with livestock grazing (Alderfer and Robinson 1947; Knoll and Hopkins 1959; Rauzi and Hanson 1966; Warren et al. 1986a; Mwendera and Saleem 1997), limited data exist on the relationships between the impact of livestock on soil properties and quantified rates of soil erosion in most landscapes (Bilotta et al. 2007). In contrast to the above studies, Gill et al. (1998) observed no clear trend in sediment losses with increased grazing intensity in grasslands in Alberta Canada. As a result, the potential cost-benefit of interventions aimed at mitigating erosion and sediment loss are uncertain. Pulley and Collins (2021) compared sediment yields from 15 grassland fields to cattle and sheep stocking rates with good livestock management and low erodibility soils in Southwest England and found no significant relationships between the two factors, suggesting that significant increases in soil loss are not likely being caused by livestock in this case. It was therefore concluded that targeting additional management interventions above and beyond recommended business-as-usual would deliver limited value to the public purse. In contrast, significant benefits for mitigating substantial excess soil loss were found to be possible in conjunction with the proper timing of ploughing operations in the same fields when they were ploughed and reseeded as part of routine sward management (Pulley and Collins 2020). Importantly, insufficient data were available with which to determine the impact of livestock grazing management and ploughing on soil properties and how this may have impacted erosion rates and, concomitantly, the potential benefits that could be delivered by additional management measures. Earlier work on the same grassland fields identified that detachment of particles by raindrop impact was likely the dominant erosion mechanism rather than detachment by concentrated overland flows (Pulley and Collins 2019). The objective of this study was therefore to investigate the soil and vegetation factors that influence rainsplash erosion in a grazed pasture and wheat field with partial vegetation cover on the same soil types studied by Pulley and Collins (2019, 2020, 2021). It was hypothesised that livestock are not causing an increase in erosion by raindrop impact on grassland fields but the same soils, when cultivated, experience a significant increase in erosion rate. As a result, mitigation measures targeted at recently cultivated land may deliver significantly greater cost-benefit than those aimed at improving grassland.

2. Study site

The study was conducted at Rowden and on the North Wyke Farm Platform (NWFP), Rothamsted Research, Devon, UK (Fig. 1; 50°46'39.48"N 3°55'10.68"W). The grassland field at Rowden has been sub-divided into three pairs of two plots aimed at comparing rotational grazing (Earl and Jones 1996) to set stocking cattle management. It has a total area of 11.8 ha with the three plots managed as rotational grazing, with each 1 ha in size. Each rotational grazing plot is positioned next to a larger set stocking plot of 1.8 ha (SS1, SS2 and SS3) where a fixed stocking rate is used throughout the grazing season. Plots SS2 and SS3 were examined in this study. The remaining field area is grazed by cattle but is not part of a controlled experiment. The field is positioned on the south side of a hill with a mean slope of 5.20° and receives a mean annual rainfall of 1053 mm. Soils within the field are mostly Halstow association with Hallsworth association also present in the Southwest. The Halstow association (sand 22%, silt 47%, clay 31% at 0-21 cm depth) is a slowly permeable clayey non-calcareous pelosol which is mottled below 400 mm and has low water storage capacity and the Hallsworth association is similarly a slowly permeable clayey soil (sand 31%, silt 43%, clay 26% at 50-100 mm depth) (Harrod and Hogan 2008). Fences dividing the experimental plots are mostly temporary electric wires although a permanent wire stock fence runs along the eastern edge of plot SS2. The permanent stock fence along the edge of SS2 has been in place since 2013, prior to which the whole field was ploughed to a depth of approximately 200 mm and reseeded with a predominantly native Perennial Rye-grass (*Lolium perenne*) seed mix with a small portion of white clover. The water trough in plot SS2 has been in the same position since ploughing in 2013 and the water trough in plot SS3 has been in position since 2018. The two set-stocking plots (SS2 and SS3) examined in this study are managed using a fixed stocking rate of approximately 1,500 kg of liveweight ha^{-1}, which equates to between 1.7

and 2.7 LU ha^{-1} depending on whether the cattle are in their first or second grazing season. The plots are continuously grazed through the grazing season with no active management of sward heights. All cattle are stabiliser beef cattle and are housed during the wet winter months of October – April to avoid soil damage and follow good-practice guidelines for the UK (Defra 2009).

The arable field studied is on the NWFP and forms the upper portion of Flume catchment 3 (50°46'16.32"N 3°54'15.71"W). The NWFP was established to compare the sustainability of beef and sheep grazing systems although recent work on catchment 3 is also testing the impacts of conversion of grassland to arable land (Orr et al. 2016; Takahashi et al. 2018). This field was studied as conversion of grassland to arable is forecast to become more common in this region of the UK and may not be environmentally sustainable (Ritchie et al. 2019). Ploughing disintegrates soil aggregates and bare earth is common in wet winter months as the crop establishes. Therefore, two major risk factors exist which could potentially cause increased rainsplash erosionand which could be targeted by mitigation measures. As with the grassland, however, the potential value different mitigation options might deliver is uncertain. The upper part of Flume catchment 3 is 2.8 ha in area and composed of 82% Halstow 14% Denbigh and 4% Hallsworth association soils, with a mean slope of 8.80°. The Denbigh association (sand 18%, silt 50%, clay 32% at 0-110 mm depth) is a free draining permeable soil unlike the Halstow and Hallsworth associations (Harrod and Hogan 2008). The field was converted from grassland to winter wheat production in 2019 leaving a ~5 m grass buffer between ploughing and the bottom field margin. At the time of sampling in February 2021, the second annual crop of wheat plants was approximately 100 mm high and small grass plants had re-established as a volunteer weed in some locations. The field had undergone rill erosion during the winter of 2020-2021 and there were up to ~20 mm deep deposits of unconsolidated sediment where runoff breached the grass buffer at the lowermost field margin and up to ~50 mm deep rills present along two lines of flow accumulation.

3. Methods

Traps as described in Section 3.1 were used to determine rates of rainsplash erosion across the grassland SS2 and SS3 plots at Rowden and the arable field at North Wyke. These data were compared to soil aggregate stability, vegetation cover, organic matter content and dry bulk density measured in cores taken from the trap locations to identify the major controls on particle detachment rates. In the grassland field, it was determined if there was a significant difference between rainsplash erosion rates and soil properties in visibly trampled areas of the field and the remaining field area. In the arable field, detachment rates were compared in areas with bare soil, partial cover by small wheat and grass plants and undisturbed grass buffers which contained recent deposits of eroded sediment.

3.1. Sample collection

Sampling of rainsplash erosion rates focused on set stocking plots 2 and 3 (SS2; SS3) as areas of compacted and heavily trampled soil were not clearly observed in the cell grazing plots. As this process can only act on the soil surface, cores were collected to a depth of 100 mm and cut to a depth of 30 mm. 73 cores were collected using a stratified randomised sampling design across the two plot areas and 27 additional points targeting field margins and visibly trampled areas of soil were also sampled. A rainsplash erosion trap based upon the design of Jordán et al. (2016) and tested by Fernández-Raga et al. (2019), with a diameter of 120 mm, was inserted into the hole left by each core at each sampling point. The traps consisted of a funnel containing a 100 mm diameter qualitative filter paper placed inside of a paper coffee filter. The coffee filter was used to prevent any soil from entering the filter paper from below in association with rising soil water. A second funnel with the spout removed was placed on top of the filter paper to prevent the sample from being washed or blown out of the trap (Fig. 2). The traps were installed into the core holes with 10 mm of the rim protruding above the soil surface to prevent runoff entering the traps, ensuring that only rainsplash mobilised particles were collected. The traps were deployed from the 28th January to the 25th February 2021. During this time 147 mm of rainfall fell in total distributed over 23 of the 29 days of monitoring; on five of these days over 10 mm of rainfall was recorded.

Within the arable field, 60 cores were retrieved and cut to 30 mm depth and the same number of rainsplash erosion traps were installed in the resultant holes following the same methods described above. The cores were sampled in three transects across the field with the coring and trap installation conducted in pairs at points spread along each transect. Each pair consisted of one sample specifically positioned on bare earth with no vegetation cover and one trap positioned at the nearest point with greatest vegetation cover by small wheat plants and volunteer grass weeds. In addition, two samples were taken in grass buffers which were undisturbed by ploughing and eleven were taken within the grass buffer at the lower field margin where eroded sediments at the base of rills had formed deposits up to 20 mm deep within the grass cover. These additional traps were aimed at determining rainsplash erosion rates in undisturbed grassland, as well as rates where eroded sediment was present in thick vegetation cover. The traps were deployed from the 9th to the 26th of March 2021when a total of 46.6 mm of rainfall was recorded. More than 10 mm of rainfall only occurred on two days during this period; on the 3rd March and 26th March. Sediment was only found in the traps after the latter event. During this period, warmer weather had resulted in lower soil moisture than when the grassland field was investigated.

3.2. Sample analysis

The cores were initially weighed when wet before a subsample (~20% of the cores mass) was sliced from the edge of each and dried at 105°C to record its percentage water content. The remainder of the core used for aggregate stability measurement was partially dried at room temperature and gently broken up into smaller pieces by hand, taking care not to disrupt soil aggregates. The 6-2 mm fraction of the sample was isolated through sieving and then fully dried at 40°C. After drying, the <2 mm fraction was then removed to account for particles that had shrunk or broken apart during the process.

Aggregate stability analysis was performed based upon the methods of van Bavel (1950) and Kemper and Rosenau (1986). A 10 g subsample of the isolated and dried 6-2 mm fraction was placed onto a 50 µm cloth mesh supported by a plastic frame and submerged in deionised water for 10 minutes. The sample was then rapidly submerged into the water five times before being gently shaken. A rapid movement was used as it was viewed as reflecting raindrop impact which is the identified erosion mechanism on the study fields (Pulley and Collins, 2019). The material remaining on the mesh was then oven dried at 40°C before being weighed. The dried sample was passed through a sieve stack of 2, 1, 0.5, 0.25 and 0.125 mm diameter meshes and the mass in each size fraction was recorded. Water-stable aggregate stability was expressed as mean weight diameter (MWD) calculated by equation 1:

$$\mathrm{MWD} = \sum_{i=1}^{n} w_i \overline{X}_i \qquad (1)$$

where: x□ is the arithmetic mean diameter of each size fraction (mm) and w_i the proportion of the total water-stable aggregates in the corresponding size fraction.

All parts of the core were then recombined and dried at 105°C so that soil dry bulk density could be recorded. The sample was then disaggregated using a pestle and mortar and any stones with a diameter >2 mm were removed and their mass and volume recorded to correct the bulk density measurement. Approximately 5 g of the <2 mm fraction of the soil was used for loss on ignition (LOI) at 450°C for four hours to estimate the organic matter content of the samples (Grimshaw et al. 1989). The remainder of the core was placed into a 2 l beaker of water and further disaggregated by hand to separate vegetation from the soil particles. The mass of vegetation was recorded for each core after drying at 105°C.

For the cores on the cultivated field, vegetation cover was highly spatially variable so the mass of vegetation on the cores surface did not accurately reflect vegetation on the surrounding soil. The samples were, instead, categorised for analysis into vegetated, unvegetated and grassland buffer categories. As the arable soils were recently cultivated, bulk density was not recorded as any surface compaction will have been reduced during ploughing and cultivation.

To record the mass of sediment retrieved in the rainsplash erosion traps, the coffee filter containing the filter paper was removed. Because of the presence of insects and vegetation in some traps, the sediment was washed from the filter paper using deionised water and passed through a 250 µm mesh. Material passing through the mesh was dried at 105°C and weighed. All sediment particles were observed to pass through this diameter mesh.

3.3. Data analysis

In the grassland field, soil dry bulk density and MWD were initially mapped to determine how soil properties varied across the plot areas. It was then tested if there was a significant difference between the aggregate stability (MWD), organic matter content (LOI) and bulk density of visibly trampled edge of field areas and the centre of the field which showed little indication of soil damage by livestock. Soil aggregate stability was then compared to the other soil properties and vegetation cover to identify any correlations. Correlations between the mass of sediment retrieved from the rainsplash erosion traps, soil properties and vegetation cover were then examined. The relative importance of the areas of soil with low rainsplash erosion rates and those with high rates on overall sediment losses from the grassland field were then compared. The area of the piesospheres around troughs and fences found to have a high rate of rainsplash erosion was multiplied by the the mean mass of sediment deposited per unit area of the traps within it. Similarly the area of the rest of the field was multiplied by the mean mass of sediment retrieved from the traps within it and a percentage of total rainsplash eroded material coming from each part of the field of was calculated. These two rainsplash erosion rates were combined and compared to the average sediment yield moving through the flumes of the unploughed fields on the NWFP between 1/10/2012–31/05/2014 to gain an indication as to if they were comparable to the overall sediment yield expected from the field.

The aggregate stability of the soils in the cultivated field was compared to that in the grassland field to determine the impact of ploughing. Soil LOI and moisture content were compared to aggregate stability to identify significant correlations. A scatter plot between the mass of sediment retrieved in the rainsplash erosion traps and soil MWD with individual points colour coded by whether they were vegetated, unvegetated or from the grass buffers, was used to identify if soil aggregate stability or vegetation cover was the more important control on rainsplash erosion rate. Tests for significant differences were conducted using a Chi-squared test and tests for correlations used a Spearman Rank analysis.

4. Results

4.1. Impacts of soil properties on soil rainsplash erosion on grassland

The mean bulk density of the cores retrieved, excluding the targeted visibly trampled and field margin samples, was 0.62 g cm^{-3} with a standard deviation of 0.14 g cm^{-3} (Fig. 3a). The mean bulk density of the edge of field samples was significantly higher ($p = 0.001$) to those in the centre of the field with a mean of 0.82 g cm^{-3} and standard deviation of 0.33 g cm^{-3}. The heaviest compaction was along the permanent fence at the edge of the western plot (SS2) where the water trough is positioned. Bulk density close to water troughs was lower along the temporary fencing at the edge of the eastern plot (SS3) where the trough has been in place for fewer years (2018 – present) compared to the water trough in SS2 (2013 – present). The area directly next to the water troughs in both plots is similarly heavily compacted.

There was no significant difference ($p=0.32$) between the aggregate stability (MWD) (Fig. 3b) of the samples retrieved from the visibly damaged soils and edge of fields (mean 2.97 mm; standard deviation 0.61 mm) and samples from the rest of the field area (mean 3.15 mm; standard deviation 0.28 mm). There were, however, five samples from visibly damaged areas with low MWDs of 1.85, 1.12, 1.31, 2.12 and 2.19 mm. Three of these were in piospheres next to water troughs on the most disturbed areas of soil, and two were in bare patches along the permanent fence of SS2. Vegetation cover was, however, not correlated ($p=0.19$) with a low MWD. Seven samples retrieved from soils with little grass cover (<0.027 g cm^{-2}) had a higher MWD than most samples which had significant vegetation cover (mean 0.1 g cm^{-2}; standard deviation 0.072 g cm^{-2} for the entire dataset). There was also no significant correlation ($p=0.82$) between organic matter content (LOI) and aggregate stability, with the samples exhibiting low aggregate stability having LOI values of 16.2-28.3% compared to a mean of 22.8% (standard deviation 3.8%) for the entire dataset. Similarly, whilst four of the five cores with low aggregate stability had a high bulk density (1.19, 1.21, 1.41 and 1.45 g cm^{-3}) compared to the mean of 0.70g cm^{-3} (standard deviation 0.25 cm^{-3}) for the entire dataset, four samples with a low aggregate stability had comparable bulk densities. One sample with a low aggregate stability also had a low bulk density of 0.62 g cm^{-3}. Therefore, soil bulk density, vegetation cover and organic matter content are poor predictors of aggregate stability.

There was no significant correlation ($r^2=0.13$; $p>0.05$) between the mass of sediment retrieved from the rainsplash erosion traps shown in Fig. 3 and aggregate stability, bulk density, vegetation cover or organic matter content. Whilst all four traps which trapped more than 0.5 g of sediment had a high bulk density (>1 g cm^{-3}), many samples with a high bulk density had low rates of rainsplash erosion. Similarly, whilst three of the four sampling points with high rainsplash erosion rates had a low aggregate stability, one had a high MWD of 3.76 mm. A low vegetation cover also characterised all the samples with high amounts of sediment trapped; however, many areas of bare earth also had low masses of sediment trapped. The highest masses found in the traps were all close to troughs or in heavily eroded depressions.

Converting the mass of sediment retrieved from the traps into a yield for the entire catchment is challenging as the precise area of soil contributing detached particles to the traps is unclear.

However, the relative importance of the areas of soil with low rainsplash erosion rates and those with high rates on a field can be compared. Multiplying the mean mass of sediment deposited per unit area of the traps (13.68 g m^{-2}; excluding the four values with more than 0.5 g trapped) by the area of the plots examined equates to 132 kg of rainsplash detached sediment in the two plots over the February monitoring period. Multiplying the mean (357 g m^{-2}) which was trapped around the two troughs and the patch of bare earth along the fence of SS2 with high rainsplash erosion rates by the 43m^{-2} of damaged soil in these areas (measured in ArcMap 10.5 using an airal photograph taken in September 2019) equates to 15.3 kg of soil detached by raindrop impact. Therefore, on a field scale, the areas of heavily damaged soils are likely to have contributed only ~10% of the total rainsplash eroded particles during the February period. When combining the two values and multiplying by the area of the plots and 16.1% equating to the proportion of sediment which is, on average, lost from the NWFP field scale catchments in the month of February (Supplimentary Fig. 1), a yield of 0.26 t ha yr^{-1} is estimated which is comparable to the mean sediment yield of 0.27 t ha yr^{-1} moving through the flumes of the unploughed fields on the NWFP between 1/10/2012–31/05/2014 (Pulley and Collins 2019).

4.2. Impacts of soil properties on soil rainsplash erosion on a recently cultivated field

The aggregate stability of the cores retrieved from the cultivated field (mean MWD 1.16 mm; standard deviation 0.34 mm.) was significantly lower (p=0.00) than in the cores retrieved from the Rowden grassland field (mean 3.15 mm; standard deviation 0.28 mm). Only one sample retrieved in the grassland field had a MWD lower than the mean value for the cultivated field. The aggregate stability of the samples retrieved in the grassland buffer surrounding the cultivated area (mean MWD 1.91 mm; standard deviation 0.64 mm), which contained recently deposited sediment, was also substantially lower than in the grassland field. There was a significant positive relationship between soil organic matter content and aggregate stability (r^2=0.29; p=0.00), and soil water percentage and organic matter content (r^2=0.63; p=0.00).

Despite low soil aggregate stability on the cultivated field, an absence of vegetation is far more strongly associated with increased rainsplash erosion rate (Fig. 4). Whilst both the bare and vegetated parts of the cultivated field had comparable aggregate stability, the mean mass of sediment retrieved from the traps on the bare soil was 0.10 g (standard deviation 0.11 g) compared to only 0.03 g (standard deviation 0.02 g) on the partially vegetated soil. The mass of sediment retrieved in the grassland buffers where sediment deposits were present was comparable to that in the vegetated cultivated land at all but one sampling point where a high mass of sediment was retrieved.

5. Discussion

The hypothesis that livestock are not causing an increase in rainsplash erosion rate on grassland fields but the same soils, when cultivated, experience a significant increase in erosion rate can be accepted based on the results generated. Soil bulk density and aggregate stability were not found to be good predictors of rates of rainsplash erosion rate on the heavy soils at Rowden and on the NWFP on either grassland or arable fields. This contrasts with the findings of some previous work which has reported the controlling effect of aggregate stability on rainsplash erosion rates, including by Farres (1978) on a calcareous silty soil with loess material additions, and Mermut et al. (1997) on loess soils. The response of soils on the NWFP to trampling also contrasts with those studied by Cattle and Southorn (2010), Stavi et al. (2011) and Cox and Amador (2018) on clay loamy and coarse-loamy soils wherein a reduction in soil aggregate stability was associated with increasing soil bulk density. The lack of an impact of grazing livestock on monitored field-scale sediment loss on the NWFP reported recently by Pulley and Collins (2021) can therefore now be explained by two factors. First, under recommended best practice grazing management, including no outdoor grazing over the winter drainage period, soil compaction is mostly limited to the field margins which are preferentially used by cattle (Pulley and Collins 2021). Within this damaged area, the magnitude of soil compaction is variable and is most severe in small areas closest to water troughs which cover an extremely small proportion (<0.3%) of the total plot area in a piosphere effect (Hart et al. 1993; Thrash and Derry 1999). Second, in these compacted areas, there is no evidence for a consistent reduction in soil aggregate stability or an associated increase in rainsplash erosion rate.

There was no indication that soil organic matter content (LOI) was associated with lower rainsplash erosion rates in the grassland field. However, LOI was lower in the arable field with its higher rainsplash erosion rates compared to the grassland, and a weak relationship was found between LOI and aggregate stability in the arable field. However, soil tillage rather than organic matter content appears to be the major control on reduced aggregate stability at our study site as relationships between the two factors were weak. Previous work has reported the destructive effect of soil tillage on aggregate stability (Angers 1992; Oades 1995; Hajabbasi and Hemmat 2000; Weidhuner et al. 2021). On the NWFP, Pulley and Collins (2020) estimated sediment yields in fields which were recently ploughed and reseeded of 0.33, 0.64, 0.66 and 0.71 t ha yr^{-1} between October 2012– and June 2014; compared to a mean of 0.27 t ha yr^{-1} for fields which had not undergone ploughing operations. After ploughing during wet soil conditions sediment yields could, however, be far higher.

Our new evidence points very clearly to the limited impacts on sediment loss which could be expected to accrue from some currently recommended and highly spatially-targeted on-farm measures for controlling sediment loss from lowland grazed landscapes on heavy soils, including moving troughs and feeder rings at regular intervals and constructing troughs with a concrete base (Collins et al. 2021). An additional consideration for when attempting to reduce sediment loss is that the low masses of sediment detached per unit area of the grass field (13.68 g m^{-2} during the February period monitored) leave limited scope for further improvement. This is especially so when considering that, if extrapolated into an annual sediment yield, the resulting estimate of 0.26 t ha yr^{-1} rainsplash erosion rate represents a considerable proportion of the typical yield for a UK catchment (Walling et al. 2008). Evans et al. (2017), using extrapolated suspended sediment yield data from Cooper et al. (2008), sediment source fingerprinting estimates and assumed sediment delivery ratios, estimated lowest mean grassland erosion rates of 0.18, 0.21, 0.25 and 0.47 t ha yr^{-1} for light (<18% clay), medium, heavy (>35% clay) and chalk/limestone soils, respectively, and corresponding upper rates of 0.46, 0.48, 0.62 and 1.18 t ha yr^{-1}. Therefore, accepting these extrapolations and assumptions the rates of grassland erosion occurring at North Wyke are likely to be typical of many grassland landscapes across England. Therefore, it is suggested that many recommended mitigation options (Newell Price et al. 2011; Natural England 2019) for reducing sediment loss would be unlikely to deliver additional benefit, above and beyond those reductions associated with best practice grazing management, if used on the NWFP and in similar landscapes elsewhere. For cultivated fields, ensuring adequate vegetation cover during winter months is likely to deliver far greater benefits than options to increase soil aggregate stability such as increasing its organic matter content. Previous work has suggested that vegetation and ground cover can reduce rainfall erosivity by ~50% (Wainwright et al. 1999).

6. Conclusions

The results of this study suggest that in lowland grazing livestock landscapes with heavy soils, such as those represented by the NWFP, where recommended best management grazing practices are already implemented, further substantial reductions in elevated sediment loss from grassland fields may be unlikely. Some elevation of sediment loss from grassland, and the associated detrimental off-site impacts for water quality, water treatment and aquatic ecology, due to lowland beef and sheep farming with recommended grazing management, is therefore an inevitable trade-off for society. This trade-off, nonetheless, is clearly reduced in magnitude by the uptake of recommended best grazing management, but there is limited scope for closing the sediment pollution gap any further. To the contrary, there is, however, greater scope for reducing elevated soil loss from arable land on the soils studied in our work by ensuring maximum vegetation cover during wet months and optimising the timing of ploughing and sowing operations to when soils are dry (Pulley and Collins 2020). Mitigation measures aimed at increasing soil aggregate stability on either grassland or arable soils studied in our work herein are unlikely to produce a significant reduction in rainsplash erosion rates, which represent the first mechanistic stage in soil erosion.

Declarations

Funding The work reported in this paper was funded by the UKRI-BBSRC (UK Research and Innovation - Biotechnology and Biological Sciences Research Council) funded institute strategic programme Soil to Nutrition via grants BBS/E/C/000I0320 and BBS/E/C/000I0330. The Cell Grazing project funded by the Agri-tech Cornwall Programme (05R16P00366), which is part-funded by the European Regional Development Fund, is gratefully acknowledged. The authors also gratefully acknowledge the funding provided for The North Wyke Farm Platform UK National Capability (UKRI-BBSRC grant BBS/E/C/000J0100).

Conflicts of interest/Competing interests: The authors declare that they have no conflicts of interest

Availability of data and material: Data is available from the North Wyke Farm Platform Data Portal at https://nwfp.rothamsted.ac.uk/

Code availability: Not applicable

7. References

Alderfer RB, Robinson RR (1947) Runoff from pastures in relation to grazing intensity and soil compaction. J Amer Sot Agron 39948: 958. https://doi.org/10.2134/agronj1947.00021962003900110002x

Al-Durrah MM, Bradford JM (1981) New methods of studying soil detachment due to waterdrop impact. Soil Sci Soc Am J 45: 949-953. https://doi.org/10.2136/sssaj1981.03615995004500050026x

Angers DA (1992) Changes in soil aggregation and organic carbon under corn and alfalfa. Soil Sci Soc Am J 56: 1244-1249. https://doi.org/10.2136/sssaj1992.03615995005600040039x

Arshad MA, Coen GM (1992) Characterization of soil quality: Physical and chemical criteria. Am. J Altern Agric 7: 25-32. https://doi.org/10.1017/S0889189300004410

Auerswald K. (1998) Bodenerosion durch Wasser. In: Richter, G. (Ed.), Bodenerosion Analyse und Bilanz eines Umweltproblems. Wiss. Buchges, Darmstadt, pp. 33–42.

Benaud P, Anderson K, Evans M, Farrow L, Glendell M, James MR, Quine TA, Quinton JA, Rawlins, Rickson RJ, Brazier RE (2020) National-scale geodata describe widespread accelerated soil erosion. Geoderma 371: 114378. https://doi.org/10.1016/j.geoderma.2020.114378

Bilotta GS, Brazier RE, Haygarth PM (2007) The impacts of grazing animals on the quality of soils, vegetation, and surface waters in intensively managed grasslands. Adv Agron 94: 237-280. https://doi.org//10.1016/S0065-2113(06)94006-1

Borrelli P, Robinson DA, Fleischer LR, Lugato E, Ballabio C, Alewell C, Meusburger K, Modugno S, Schütt B, Ferro V, Bagarello V, Van Oost K, Montanarella L, Panagos P (2017) An assessment of the global impact of 21st century land use change on soil erosion. Nat Commun 8: 2013. https://doi.org/10.1038/s41467-017-02142-7

Borst HL, Woodburn R (1942) The effect of mulching and methods of cultivation on run-off and erosion from Muskingum silt loam. Agric Eng 23: 19-22.

Broersma K, Krzic M, Thompson DJ, Bomke AA (2000) Soil and vegetation of ungrazed crested wheatgrass and native rangelands. Can J Soil Sci 80: 411–417. https://doi.org/10.4141/S99-082

Bubenzer GD, Jones BA (1971) Drop size and impact velocity effects on the detachment of soils under simulated rainfall. Trans ASAE 14: 625-628. https://doi.org/10.13031/2013.38353

Busby FE, Gifford GF (1981) Effects of livestock grazing on infiltration anderosion rates measured on chained and unchained pinyon-juniper sites in Southern Utah. J Range Manage 34: 400–405. https://doi.org/10.2307/3897914

Cattle SR, Southorn NJ (2010) Macroporosity of pasture topsoils after three years of set-stocked and rotational grazing by sheep. Aus J Soil Res 48: 43–57. https://doi.org/10.1071/SR09004

Cerdan O, Govers G, Le Bissonnais Y, Van Oost K, Poesen J, Saby N, Gobin A, Vacca A, Quinton J, Auerswald K, Klik A, Kwaad FJPM, Raclot D, Ionita I, Rejman J, Rousseva S, Muxart T, Roxo MJ, Dostal T (2010) Rates and spatial variations of soil erosion in Europe: a study based on erosion plot data. Geomorphology 122(1–2): 167–177. https://doi.org/10.1016/j.geomorph.2010.06.011

Chan KY, Bowman A, Oates A (2001) Oxidizible organic carbon fractions and soil quality changes in an oxic paleustalf under different pasture leys. Soil Science 166(1): 61-67. https://dx.doi.org/10.1097/00010694-200101000-00009

Collins AL, Zhang Y (2016) Exceedance of modern ‘background’ fine-grained sediment delivery to rivers due to current agricultural land use and uptake of water pollution mitigation options across England and Wales. Environ Sci Pol 61: 61-73. https://doi.org/10.1016/j.envsci.2016.03.017

Collins AL, Zhang Y, Upadhayay HR, Pulley S, Granger SJ, Harris P, Sint H, Griffith B (2021) Current advisory interventions for grazing ruminant farming cannot close exceedance of modern background sediment loss – Assessment using an instrumented farm platform and modelled scaling out. Environ Sci Policy 116: 114-127. https://doi.org/10.1016/j.envsci.2020.11.004

Cooper DM, Naden P, Old G, Laize C (2008) Development of Guideline Sediment Targets to Support Management of Sediment Inputs Into Aquatic Systems. Natural England Research Report NERR008 Natural England, Sheffield.

Cox AH, Amador JA (2018) How grazing a□ects soil quality of soils formed in the glaciated northeastern United States. Environ Monit Assess 190: 159. https://doi.org/10.1007/s10661-018-6550-5

DEFRA (2009) Protecting our water, soil and air: A code of good agricultural practice for farmers, growers and land managers. TSO, Norwich, UK.

Di HJ, Cameron KC, Milne J, Drewry JJ, Smith NP, Hendry T, Moore S, Reijnen BA (2001) Mechanical hoof for simulating animal treading under controlled conditions. NZ J Agric Res 44: 111–116. https://doi.org/10.1080/00288233.2001.9513465

Earl JM, Jones CE (1996) The Need for a New Approach to Grazing Management - Is Cell Grazing the Answer? Rangel J 18(2): 327-350. https://doi.org/10.1071/RJ9960327

Elliott ET (1986) Aggregate structure and carbon, nitrogen, andphosphorus in native and cultivated soils. Soil Sci Soc Am J 50: 627–633. https://doi.org/10.2136/sssaj1986.03615995005000030017x

Evans R, Collins AL, Zhang Y. Foster IDL, Boardman J, Sint H, Lee MRF, Griffith BA (2017) A comparison of conventional and ^{137}Cs-based estimates of soil erosion rates on arable and grassland across lowland England and Wales. Earth Sci Rev 173: 49-64. https://doi.org/10.1016/j.earscirev.2017.08.005

Farres PJ (1978) Role of aggregate size and time in the soil crusting process. Earth Surf Process 3: 243-254. https://doi.org/10.1002/esp.3290030304

Fernández-Raga M, Campo J, Rodrigo-Comino J, Keesstra SD (2019) Comparative Analysis of Splash Erosion Devices for Rainfall Simulation Experiments: A Laboratory Study. Water 11(6): 1228. https://doi.org/10.3390/w11061228

Ferrero A, Lipiec J (2000) Determining the e□ect of trampling on soils in hillslope-woodlands. Int Agrophys 14: 9-16.

Foster IDL, Collins AL, Naden PS, Sear DA, Jones JI, Zhang Y (2011) The potential for palaeolimnology to determine historic sediment delivery to rivers. J Palaeolim 45: 287–306. https://doi.org/10.1007/s10933-011-9498-9

Ghadiri H, Payne D (1981) Raindrop impact stress. Eur J Soil Sci 32(1): 41-49. https://doi.org/10.1111/j.1365-2389.1981.tb01684.x

Gill SI, Naeth MA, Chanasyk DS, Baron VS (1998) Runoff and sediment yield from snowmelt and rainfall as influenced by forage type and grazing intensity. Can J Soil Sci 78: 699706. https://doi.org/10.4141/S97-067

Gooday RD, Anthony SG, Durrant C, Harris D, Lee D, Metcalfe P, Newell-Price P, Turner A (2015) Farmscoper Extension. Report for Defra Project SCF0104.

Grimshaw HM, Allen SE, Parkinson JA (1989) Nutrient elements. In: Allen SE (ed.) Chemical analysis of ecological materials. Blackwell, Oxford, UK. pp. 368.

Hamza MA, Anderson WK (2005) Soil compaction in cropping systems: A review of the nature, causes and possible solutions. Soil Till Res 82(2): 121-145. https://doi.org/10.1016/j.still.2004.08.009

Hardy RA, James MR, Pates JM, Quinton JN (2017) Using real time particle tracking to understand soil particle movements during rainfall events. Catena 150: 32-38. https://doi.org/10.1016/j.catena.2016.11.005.

Harrod TR, Hogan DV (2008) The Soils of North Wyke and Rowden. URL www.rothamsted.ac.uk/ sites/default/files/SoilsNWRowden.pdf. Accessed 6th January 2021.

Hart RH, Bissio J, Samuel MJ, Waggoner JW (1993) Grazing systems, pasture size, and cattle grazing behaviour, distribution and gains. J Range Manage 46: 81-87. https://doi.org/10.2307/4002452

Holechek JL, Gomes H, Molinar F, Galt D, Valdez R (2000) Short-duration grazing: the facts in 1999. Rangelands 22(1): 18-22. https://doi.org/10.2458/azu_rangelands_v22i1_holechek

Johnston A (1962) Effect of grazing intensity and cover on water intake rates of fescue rangeland. J Range Manage 15: 79-82. https://doi.org/10.2307/3894865

Jordán A, Zavala LM, Granged AJ, Gordillo-Rivero ÁJ, García-Moreno J, Pereira P, Bárcenas-Moreno G, de Celis R, Jiménez-Compán E, Alanís N (2016) Wettability of ash conditions splash erosion and runoff rates in the post-fire. Sci Total Environ 572: 1261–1268. https://doi.org/10.1016/j.scitotenv.2015.09.140

Kay BD (1990) Rates of change of soil structure under different cropping systems. In Stewart BA. (ed.). Advances in Soil Science 12, New York: Springer, 1–44. https://doi.org/10.1007/978-1-4612-3316-9_1

Keller T, Sandin M, Colombi T, Horn R, Or D (2019) Historical increase in agricultural machinery weights enhanced soil stress levels and adversely affected soil functioning, Soil Till Res 194: 104293. https://doi.org/10.1016/j.still.2019.104293.

Kemper WD, Rosenau RC (1986) Aggregate stability and size distribution. In: Klute A (ed) Methods of soil analysis. American Society of Agronomy□Soil Science Society of America, Wisconsin, USA.

Knoll G, Hopkins HH (1959) The effects of grazing and trampling upon certain soil properties. Trans Kansas Acad Sci 62: 221-231. https://doi.org/10.2307/3626373

Lal R, Moldenhauer WC (1987) Effects of soil erosion on crop productivity. Crit Rev Plant Sci 5(4): 303-367. https://doi.org/10.1080/07352688709382244

Lal R. Mahboubi AA, Fausey NR (1994) Long□Term Tillage and Rotation Effects on Properties of a Central Ohio Soil Soil Sci Soc Am J 58: 517-522. https://doi.org/10.2136/sssaj1994.03615995005800020038x

Le Bissonnais Y (1996) Aggregate stability and assessment of soil crustability and erodibility: I. Theory and methodology. Eur J Soil Sci 47: 425-43. https://doi.org/10.1111/j.1365-2389.1996.tb01843.x

Li T, Xiao HL, Li XR (2001) Modelling the effects of crust on rain infiltration in vegetated sand dunes in arid desert. Arid Land Res Manage 15(1): 41–48. https://doi.org/10.1080/15324980119192

Mermut AR, Luk SH, Römkens MJM, Poesen JWA (1997) Soil loss by splash and wash during rainfall from two loess soils. Geoderma 75(3-4): 203-214. https://doi.org/10.1016/S0016-7061(96)00091-2.

Montgomery DR (2007) Soil erosion and agricultural sustainability. Proc Natl Acad Sci 104: 13268–13272. https://doi.org/10.1073/pnas.0611508104

Moore DC, Singer MJ (1990) Crust formation effect on soil erosion processes. Soil Sci Soc Am J 54: 1117-1123. https://doi.org/10.2136/sssaj1990.03615995005400040033x

Mwendera EJ, Mohamed Saleem MA (1997) Hydrologic response to cattle grazing in the Ethiopian highlands Agric Ecosyst Environ 64(15): 33-41. https://doi.org/10.1016/S0167-8809(96)01127-9

Natural England (2019) Catchment Sensitive Farming evaluation report – water quality phases 1 to 4 (2006-2018). http://publications.naturalengland.org.uk/publication/4538826523672576. Accessed 06 January 2020.

Newell Price JP, Harris D, Taylor M, Williams JR, Anthony SG, Duethmann D, Gooday RD, Lord EI, Chambers BJ, Chadwick DR, Misselbrook TH (2011) User manual — ‘ALL’. An inventory of methods and their effects on diffuse water pollution, greenhouse gas emissions and ammonia emissions from agriculture. Final report for project WQ0106(5), Defra, UK.

Oades JM (1984) Soil organic matter and structural stability: mechanisms and implications for management. Plant Soil 76: 319–337. https://doi.org/10.1007/BF02205590

Oldeman LR (1992) The global extent of soil degradation. In Soil Resilience and Sustainable Land Use; Greenland DJ, Szabolcs I. Eds. CAB International: Wallingford, UK, pp. 99–118.

Orr RJ, Murray PJ, Eyles CJ, Blackwell MSA, Cardenas LM, Collins AL, Dungait JAJ, Goulding KWT, Griffith BA, Gurr SJ, Harris P, Hawkins JMB, Misselbrook TH, Rawlings C, Shepherd A, Sint H, Takahashi T, Tozer KN, Whitmore AP, Wu L, Lee MRF (2016) The North Wyke Farm Platform: effect of temperate grassland farming systems on soil moisture contents, runoff and associated water quality dynamics. Eur J Soil Sci 67: 374-385. https://doi.org//10.1111/ejss.12350

Parsons AJ, Abrahams AD, Wainwright J (1994) Rainsplash and erosion rates in an interrill area on semi-arid grassland, Southern Arizona. Catena 22: 215-226. https://doi.org/10.1016/0341-8162(94)90003-5

Proffitt APB, Jarvis RJ, Bendotti S (1995) The impact of sheep trampling and stocking rate on the physical-properties of a Red Duplex soil with two initially different structures. Aust J Agric Res 46: 733–747. https://doi.org//10.1071/AR9950733

Pulley S, Collins AL (2021) Does cattle and sheep grazing under best management significantly elevate sediment losses? Evidence from the North Wyke Farm Platform, UK. J Soils Sediments 21: 1875–1889. https://doi.org/10.1007/s11368-021-02909-y

Pulley S, Collins AL (2019) Field-scale determination of controls on runoff and fine sediment generation from lowland livestock grazing land. J Environ Manage 249: 109365. https://doi.org//j.jenvman.2019.109365

Pulley S, Collins AL (2020) Sediment loss in response to scheduled pasture ploughing and reseeding for lowland ruminant farming: the importance of soil moisture content in controlling risk. Soil Till Res 204: 104746. https://doi.org//10.1016/j.still.2020.104746

Rauzi F, Hanson L (1966) Water intake and runoff as affected by intensity of grazing. J Range Manage 19: 351-356. https://doi.org/10.2307/3895570

Riezebos H Th, Epema GF (1985) Drop shape and erosivity, Part II: Splash detachment, transport and erosivity indices. Earth Surf Process Landf 10: 69-74. https://doi.org/10.1002/esp.3290100109

Ritchie PDL, Harper AB, Smith GS, Kahana R, Kendon EJ, Lewis H, Fezzi C, Halleck-Vega S, Boulton CA, Bateman IJ, Lenton TN (2019) Large changes in Great Britain’s vegetation and agricultural land-use predicted under unmitigated climate change. Environ Res Lett 14: 114012. https://doi.org/10.1088/1748-9326/ab492b

Rose NL, Morley D, Appleby PG, Battarbee RW, Alliksaar T (2010) Sediment accumulation rates in European lakes since AD 1850: trends, reference conditions and exceedance. J Paleolimnol 45: 447–468. https://doi.org/10.1007/s10933-010-9424-6

Sanjari G, Yu B, Ghadiri H, Ciesiolka CAA, Rose CW (2009) Effects of time-controlled grazing on runoff and sediment loss. Aust J Soil Res 47: 796–808. https://doi.org//10.1071/SR09032

Shah AN, Tanveer M, Shahzad B, Yang G, Fahad S, Ali S, Bukhari MA, Tung SA, Hafeez A, Souliyanonh B (2017) Soil compaction e□ects on soil health and crop productivity: An overview. Environ Sci Pollut Res 24: 10056–10067. https://doi.org/10.1007/s11356-017-8421-y

Slattery MC, Bryan RB (1992) Hydraulic conditions for rill incision under simulated rainfall: a laboratory experiment. Earth Surf Process Landf 17: 127-146. https://doi.org/10.1002/esp.3290170203

Stavi I, Ungar ED, Lavee H, Sarah P (2011) Soil aggregate fraction 1-5 mm: An indicator for soil quality in rangelands. J Arid Environ 75: 1050-1055. https://doi.org/10.1016/j.jaridenv.2011.04.035

Steinfeld H, Gerber P, Wassenaar T, Castel V, Rosales M, de Haan C (2006) Livestock's Long Shadow. Environmental Issues and Options; Food and Agriculture Organisation of the United Nations: Rome, Italy.

Takahashi T, Harris P, Blackwell MSA, Cardenas LM, Collins AL, Dungait JAJ, Hawkins JMB, Misselbrook TH, McAuliffe GA, McFadzean JN, Murray PJ, Orr RJ, Rivero MJ, Wu L, Lee MRF (2018) Roles of instrumented farm-scale trials in trade-off assessments of pasture-based ruminant production systems. Animal 12(8): 1766-1776. https://doi.org//10.1017/S1751731118000502

Thrash I, Derry JF (1999) The nature and modelling of piospheres: a review. 1999. Koedoe 42: 73-94. https://doi.org/10.4102/koedoe.v42i2.234

Van Bavel CHM (1950) Soil aggregate stability as affected by sterilization with ethylene oxide and heat. Plant Soil 2: 395–404. https://doi.org/10.1007/BF01343359

Walling DE, Webb BW, Shanahan J (2008) Investigations into the use of critical sediment yields for assessing and managing fine sediment inputs into freshwater ecosystems. Natural England Research Report NERR007. Sheffield, UK.

Warren SD, Nevill MB, Blackburn WH, Garza NE (1986a) Soil response to trampling under intensive rotation grazing. Soil Sci Soc Am J 50: 1336–1341. https://doi.org//10.2136/sssaj1986.03615995005000050050x

Warren SD, Thurow TL, Blackburn WH, Garza NE (1986b) The influence of livestock trampling under intensive rotation grazing on soil hydrologic characteristics. J Range Manage 39: 491-495. https://doi.org/10.2307/3898755

Weidhuner A, Hanauer A, Krausz R, Crittenden SJ, Gage K, Sadeghpour A (2021) Tillage impacts on soil aggregation and aggregate-associated carbon and nitrogen after 49 years. Soil Till Res 104878. https://doi.org/10.1016/j.still.2020.104878.

Young RA, Wiersma JL (1973) The role of rainfall impact in soil detachment and transport. Water Resour Res 9: 1629-1636. https://doi.org/10.1029/WR009i006p01629

Figure Captions

Fig. 1: The Rowden experimental plot field and Upper NWFP Flume 3 catchment with sampling locations.

Fig. 2: Rainplash erosion traps positioned on bare and vegetated parts of the cultivated field after a rainfall event.

Fig. 3: Mapped soil dry bulk density, MWD and masses of sediment retrieved in the rainsplash erosion traps in 0-3 cm deep cores in plots SS2 and SS3.

Fig. 4: The mass of sediment retrieved in the rainsplash erosion traps compared to soil aggregate stability.

Figures

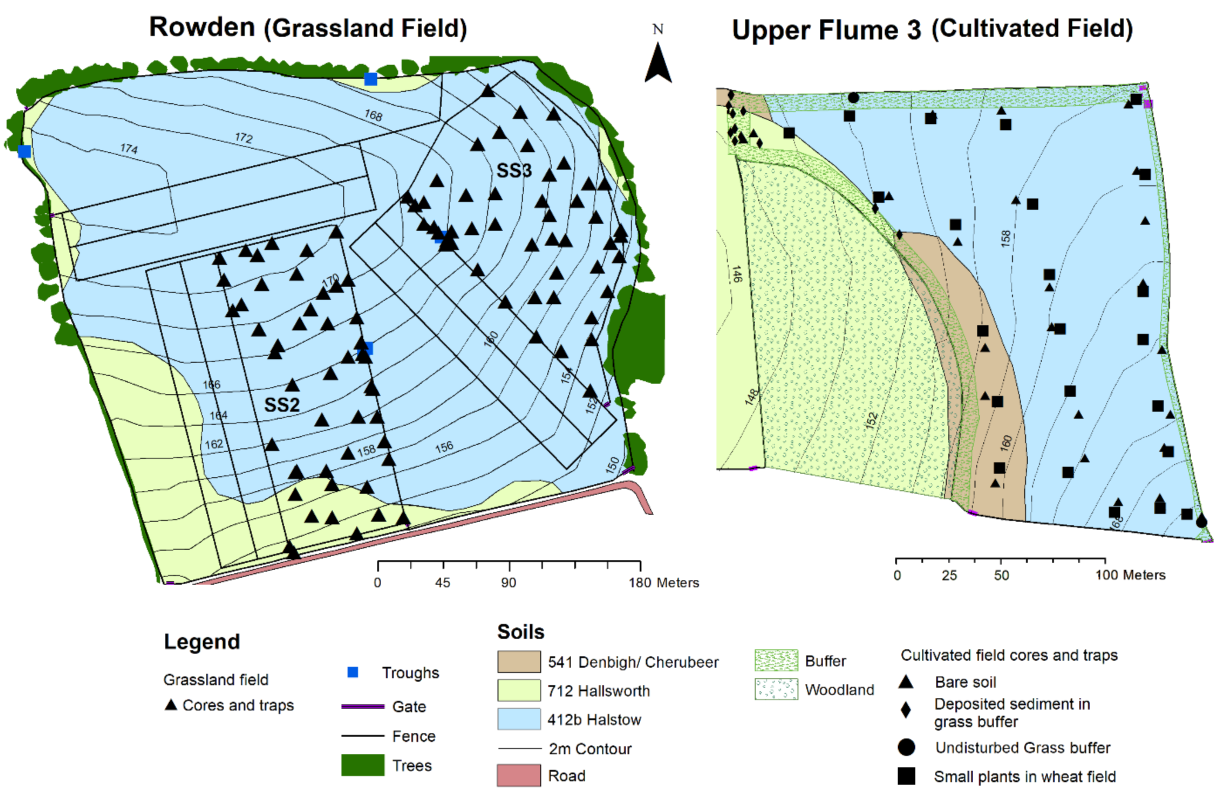


Fig. 1

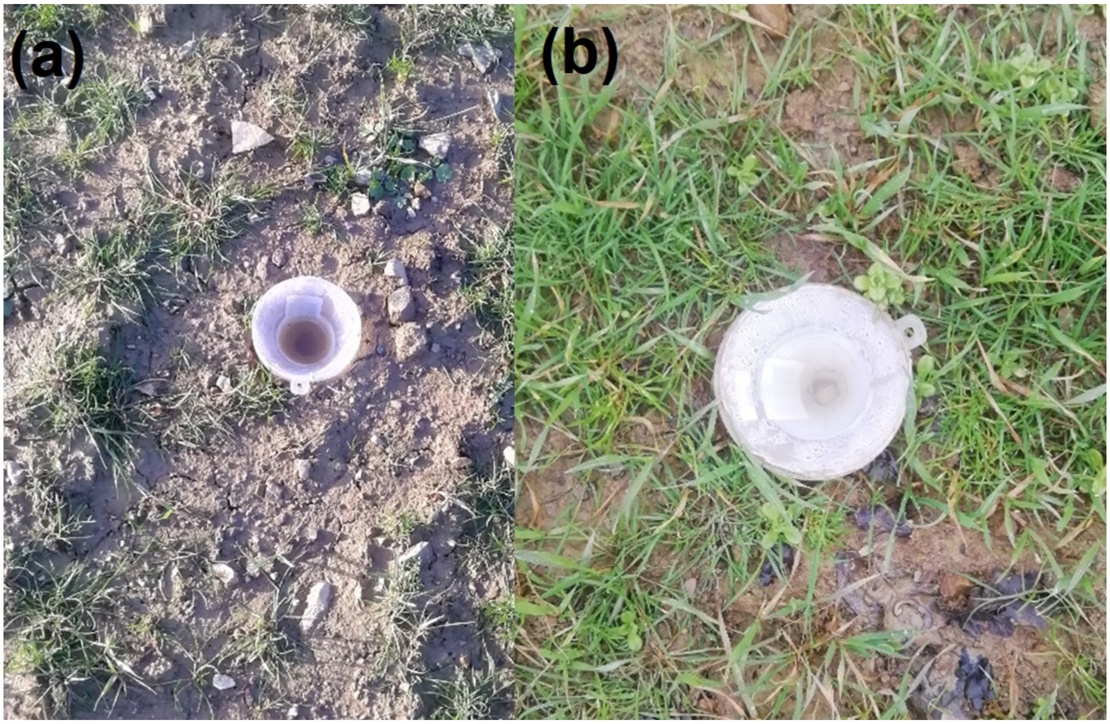


Fig. 2

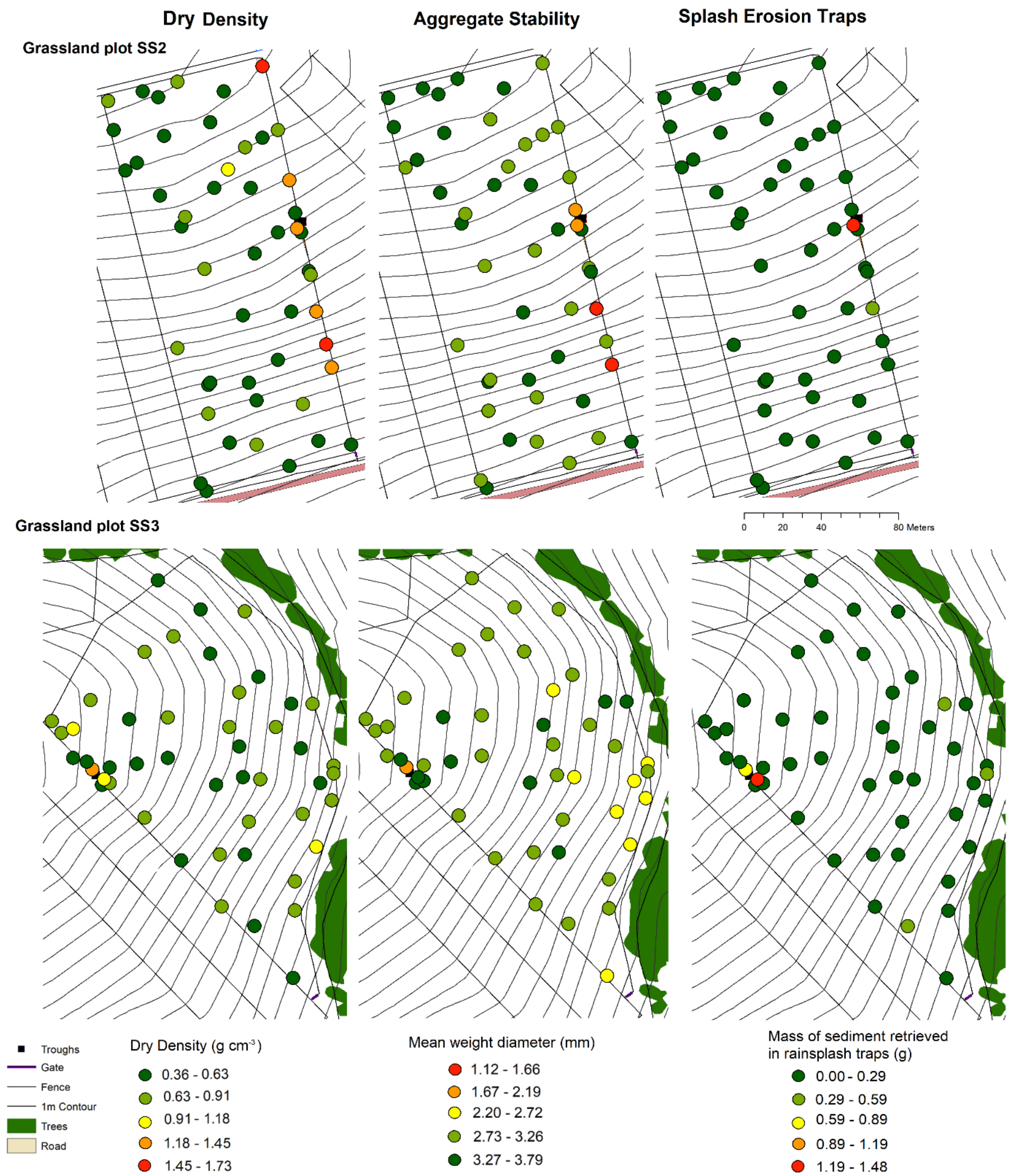

Dry Density
Aggregate Stability
Splash Erosion Traps
Grassland plot SS2
Grassland plot SS3
0
20
40
80 Meters
Troughs
Gate
Fence
1m Contour
Trees
Road
Dry Density (g cm-3)
0.36 - 0.63
0.63 - 0.91
0.91 - 1.18
1.18 - 1.45
1.45 - 1.73
Mean weight diameter (mm)
1.12 - 1.66
1.67 - 2.19
2.20 - 2.72
2.73 - 3.26
3.27 - 3.79
Mass of sediment retrieved in rainsplash traps (g)
0.00 - 0.29
0.29 - 0.59
0.59 - 0.89
0.89 - 1.19
1.19 - 1.48

Fig. 3

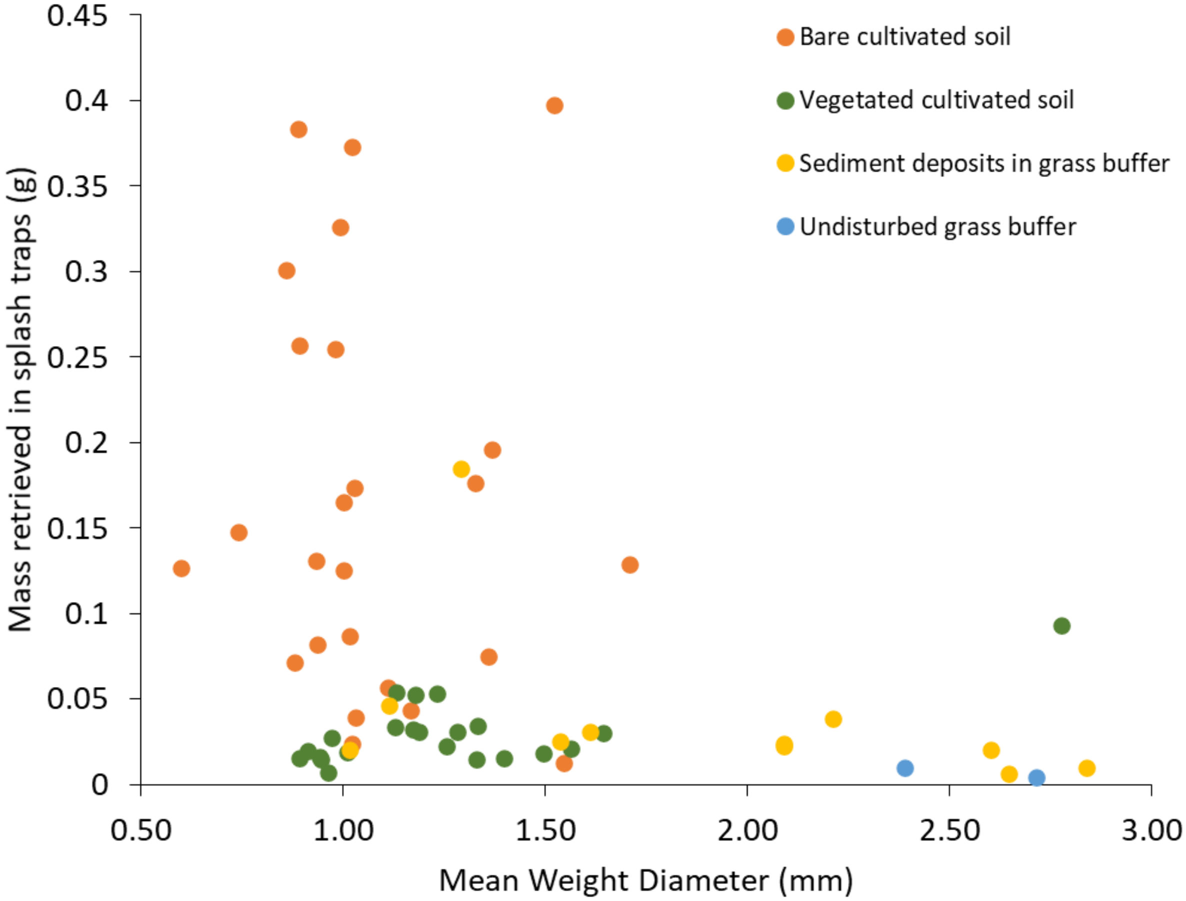
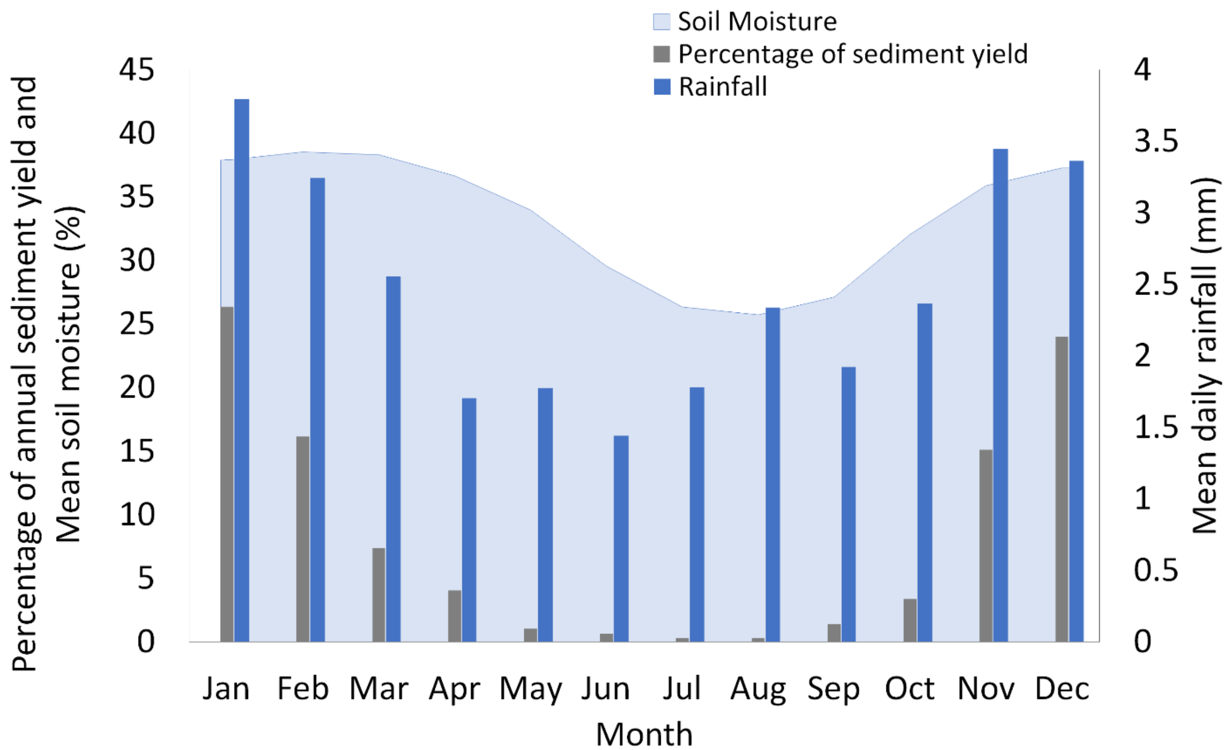

Bare cultivated soil
Vegetated cultivated soil
Sediment deposits in grass buffer
Undisturbed grass buffer
Mass retrieved in splash traps (g)
Mean Weight Diameter (mm)
0.45
0.4
0.35
0.3
0.25
0.2
0.15
0.1
0.05
0
0.50
1.00
1.50
2.00
2.50
3.00

Fig. 4

Supplementary Figures



Supplementary Fig. 1: Monthly percentage contribution to annal sediment yield, mean rainfall and mean soil moisture.