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Agricultural land lacks resistance to water erosion during the wettest winters of the past decade

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A B S T R A C T

With changing climate and increased frequency of wet weather extremes, increased attention is being directed towards understanding the resilience of agroecosystems and the goods and services they deliver. The world's most instrumented and monitored farm (the North Wyke Fam Platform - a UK National Bioscience Research Infrastructure) has been used to explore the resilience of sediment loss regulation delivered by lowland grazing livestock and arable systems under conventional best management. The robustness of water quality regulation was explored using exceedance of modern background (i.e. pre-World War II) net soil loss rates (i.e., sediment delivery) during both typical (2012–13, 2015–16) and the most extreme (2013–14, 2019–20, 2023–24) winters (December - February, inclusive), in terms of seasonal rainfall totals, over the past ~decade. Exceedances of maximum modern background sediment loss rates from pasture were as high as 2.4X when scheduled ploughing and reseeding for sward improvement occurred immediately prior to the winters in question. Exceedances of maximum modern background sediment loss rates in the arable system (winter wheat and spring oats) were as high as 21.7X. Over the five monitored winters, the environmental damage costs for cumulative sediment loss from the permanent pasture system ranged from £ 163–203 and £ 197–245 ha^{-1} to £ 321–421 and £ 386–507 ha^{-1}. Over the same five winters, environmental damage costs for cumulative sediment loss from catchments subjected to reseeding and, more latterly, arable conversion, ranged between £ 382–584 and £ 461–703 ha^{-1} to £ 1978–2334 and £ 2384–2812 ha^{-1}. Our data provide valuable quantitative insight into the impacts of winter rainfall and land use on the resilience of sediment loss regulation.

1. Introduction

Agroecosystems are facing both abiotic and biotic stresses including climate change (Meuwissen et al., 2020). The capacity of these systems to cope with such challenges can be broadly conceptualized as 'resilience'. Here, three system capacities have been proposed to understand agroecosystem resilience to stresses; 'robustness', 'adaptability' and 'transformability' (Meuwissen et al., 2019). Robustness represents the system ability to withstand stresses. Water regulatory services, including soil and sediment loss regulation, are important features and indeed expectations of our agroecosystems, yet such services are threatened by land use intensification and changing climate including extreme wet weather (Zhang et al., 2022; Upadhayay et al., 2024).

More specifically, soil erosion on agricultural land is a global problem for land management (Parsons, 2019; Pandey et al., 2021) and degrades soil ecosystem services for food production, carbon storage, biodiversity and water filtration (Lal, 2013; Pimentel and Burgess, 2013). Global estimates of soil erosion by water range between 23.7 and 120 billion t yr^{-1} (Doetterl et al., 2012). Erosion is caused by different agents including water and, in the specific case of the UK, concerns over water erosion have been expressed for over 50 years (Evans, 1996). Land managers and policy teams require reliable soil erosion data at the management scale, meaning that field scale data are preferable. Much soil erosion data, however, has been measured on bounded or more rarely on unbounded plots (Mutchler, 1963; Imeson et al., 1998; Stevens et al., 2009). The former have ranged in size from 0.24 m^2 to 20,000 m^2 (Mutchler, 1963; Imeson et al., 1998; Stevens et al., 2009). Here, a range of problems for generating reliable soil erosion data have been discussed including the use of rainfall simulators and concomitant challenges in representing natural rainfall and the impacts of plot boundaries on natural water and sediment movement. Small plots are therefore best placed for providing relative values of soil loss (Parsons, 2019). Since soil erosion data are scale-dependent, meaning that soil loss does not scale linearly with increasing upslope area (Parsons et al., 2010), plot

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scale data do not provide a reliable means of understanding field scale erosion rates, but the latter are preferable given that fields are the principal management unit on farms.

As an alternative to plot scale approaches, soil erosion has also been estimated using traditional field surveys of rill and gully volumes and applying an assumed rate of soil wash (Evans, 2002; Boardman, 2003; Evans et al., 2016). But, assigning a reliable estimate of sheetwash in the absence of direct measurements remains uncertain, meaning that traditional surveys run the risk of underestimating soil loss at field scale (Govers and Poesen, 1988; Parsons, 2019). With changing wet weather patterns, it is evident from field walkover surveys that the occurrence of sheetwash is increasing in both spatial extent and frequency (Evans et al., 2016). Given challenges and uncertainties associated with traditional plot or field scale approaches, some researchers have used natural or fallout radionuclides to estimate time-averaged soil erosion and sediment redistribution rates, including at field or slope scale, but the validity of the underlying assumptions has been challenged (Parsons and Foster, 2011; Mabit et al., 2014).

Long-term field scale studies are considered most appropriate for robust estimates of soil loss (Evans and Boardman, 2016; Steinhoff--Knopp and Burkhard, 2018). Intensive farming has been reported to elevate soil loss in many parts of the world (Montgomery, 2007; Zhao et al., 2013; Borrelli et al., 2018), including the UK (Collins and Zhang, 2016), but an evaluation of the severity of the soil erosion problem requires a management threshold. Here, one approach to threshold setting concerns the concept of tolerable soil loss. But the use of this threshold has faced criticism given the inherent spatial and temporal variability of the tolerable limit (Mandal et al., 2006; Verheijen et al., 2009) and the lack of a robust harmonized approach to its calculation (Carollo et al., 2023). The proposed range of the tolerable limit for European soils, for example, ranges between 0.3 – 2 t ha^{-1} yr^{-1} (Di Stefano et al., 2023), with a general threshold of either 1 or 2 t ha^{-1} yr^{-1} applied in the absence of localised estimates (Panagos et al., 2020).

In many countries, including the UK, increasing concern has been sounded over the off-site consequences of accelerated soil erosion and sediment delivery from agricultural fields to rivers, especially since such unintended impacts can frequently outweigh those on-site. These consequences include muddy floods and property damage (Evans and Boardman, 2003), increased drinking water treatment costs (OFWAT, 2011) and harm to all trophic levels of aquatic ecology (Kemp et al., 2011; Jones et al., 2012, 2014). In the context of the criticism of the tolerable erosion limits, an alternative approach to setting a management target for the erosion problem on agricultural land proposed in the UK, relies on estimated 'modern background' sediment loss (Collins et al., 2021). This approach pays due regard to the fact that net (i.e., the rate of loss from the field unit) rather than gross soil loss is most important, since soil remaining within a field can be redistributed by annual tillage in arable fields or by less frequent scheduled ploughing and reseeding in grassland settings.

Examination of the robustness (i.e., resistance) of water regulating services delivered by agricultural land to changing climate and especially wet weather, is warranted given widespread concern over this increasingly pervasive management challenge from multiple stakeholders. The work reported in this paper therefore used net soil loss data measured on a well-established replicated field scale facility in the southwest of England to examine the robustness of sediment loss regulation during recent winters. The specific objectives were to: (i) estimate, compare and contrast net erosion rates during the most recent wettest winters with those in two more 'typical' winters in the context of long-term average rainfall volume; (ii) use those estimates together with the appropriate range of 'modern background' sediment loss to compare and contrast the robustness of grass and arable land to the water erosion problem in the context of changing climate and prolonged wet winter weather experienced over the past 12 years, and; (iii) to monetise the 'excess' soil loss on the basis of environmental damage to illustrate societal costs.

2. Material and methods

2.1. The approach for assessing robustness of sediment loss regulation

Assessment of the robustness of sediment loss regulation necessitated selection of a realistic basis for exploring the resistance of agricultural land to wet weather. Given the increasing focus on the off-site consequences of soil erosion and sediment delivery in many parts of the world, concentration-based thresholds have been reported as management targets (Collins et al., 2011). These have been proposed to identify definite negative ecological impacts of erosion and sediment delivery in freshwater receptors with, for example, the thresholds for fish ranging from > 80 mg L^{-1} (Alabaster and Lloyd, 1980) to > 100 mg L^{-1} (DFO, 1983). However, in-depth reviews of sediment-dose response data have underscored the over-simplification embodied in a concentration-based approach to sediment thresholds (Berry et al., 2003). Here, for instance, in the case of various freshwater fish species, published dose-response relationships range from 4 – 330,000 mg L^{-1}. Such a broad range undoubtedly reflects complex interactions between various factors including, for example, species life stage and history, duration of exposure and sediment quality, pointing to the need for site-specific data and thresholds and the unrealistic resource demands therein.

Given the above challenges, an alternative approach is to identify a management target based on the sediment regime (i.e., suspended sediment yield). In the specific case of the UK, the typical coefficient of variation of catchment annual suspended sediment yields has been estimated to be 50–60 %, with Walling et al. (2008) using standard error statistics to recommend that 25 years of data are required to measure robust suspended sediment yields to within ± 20 % of the actual value. This clearly poses challenges given the resource demands inherent in conventional monitoring strategies. Additionally, another important consideration is that some soil erosion and sediment delivery are required for healthy off-site aquatic habitats, pointing to the need to build 'natural' sediment regimes into such management targets (Poff et al., 1997; Collins et al., 2011).

In recognition of both the potential of regime-based approaches, albeit with the need for longer-term data, and the desire to incorporate the 'natural' sediment regime, work in the UK has used palaeo-environmental reconstruction on dated lake cores to estimate so-called 'modern background' sediment loss from agricultural land (Foster et al., 2011). This concept addresses the need for any target to be based on longer-term data by spanning sediment regime responses over ~100–150 years and is based on the sediment regime pre-dating post WWII intensification which resulted in the most recent substantial elevation of sediment loss (Rose et al., 2011). Furthermore, the concept of 'modern background' sediment loss pays due regard to the fact that we need food production from some agricultural land, implying that adoption of a truly 'intrinsic' estimate of sediment loss regulation is less meaningful and, in fact, impractical.

In matching dated lake sediment profiles to dominant land covers, Foster et al. (2011) estimated uncertainty ranges for modern background sediment loss (Table 1). Given the small scale of the lake catchments used in the work, these values represent sediment delivery

Table 1
Uncertainty estimates of modern background net sediment loss for lowland agricultural land across England (Foster et al., 2011).

Land cover	Minimum (t ha^{-1} yr^{-1})	Maximum (t ha^{-1} yr^{-1})
Lowland agriculture (A)[a]	< 0.1	0.15
Lowland agriculture (B)[b]	< 0.2	0.35

[a] Group A has soils at very low or low-moderate risk of water erosion using the classification of Evans (1990) and/or are less heavily dissected (<3°).

[b] Group B has soils at high to very high risk of water erosion using the classification of Evans (1990) and/or are more heavily dissected (slopes >3 °).

from agricultural land to freshwater receptors rather than realistic targets for larger scale catchments. Accordingly, the values in Table 1 can be compared directly with net sediment loss measured from fields neighbouring a river channel system as a means of exploring the robustness of sediment loss regulation to current wet weather. Since the environmental setting of our study site bridges categories A and B in Table 1 (e.g., slopes range from 3 to 12°), sediment loss regulation was deemed 'robust' if both the lower and upper 95 % confidence limits of measured net soil loss (t ha^{-1} yr^{-1}) fell within the corresponding uncertainty range (i.e., 0.15 and 0.35 t ha^{-1} yr^{-1}; Table 1) from Foster et al. (2011). Exceedance of this uncertainty range was used as evidence of a lack of robustness to water erosion. Exceedance of the minimum modern background rate only, was used as evidence of a partial lack of robustness.

2.2. Study site

The North Wyke Farm Platform (NWFP; 50°46′10′′N, 3°54′05′′W; Orr et al., 2016) was constructed in 2010 for detailed systems analysis of the sustainability and resilience of UK farming systems. This National Bioscience Research Infrastructure combines three farmlets (21 ha each), each comprising five catchments (1.62–8.08 ha), with an overall total of 21 fields (Fig. 1). Historical tile drains are no longer functioning due to the lack of maintenance of the permeable backfill and drain outlet freeboards. The individual catchments are now hydrologically-isolated using the combined effect of a 9.2 km network of French drains (trenches 800 mm deep with a perforated drainage pipe backfilled to the surface with 20–50 mm clean stone chips), together with the local topography and impermeable clayey soils. Hallsworth (Dystric Gleysol) and Halstow (Gleyic Cambisol) soils (Avery, 1980) are present and these comprise a slightly stony clay loam topsoil (~36 % clay) over mottled stony clay (approximately 60 % clay). The underlying geology is Carboniferous Culm rocks (Harrod and Hogan, 2008). More specifically, the geology is formed by the Carboniferous Crackington Formation, consisting of clay shales (known locally as 'shillot') and subsidiary thin sandstone bands. When waterlogged, the shales break down to form clay with the clay minerals being predominantly illitic. The sandstone bands comprise about a quarter of the sequence but are rarely thicker than 30–40 cm. In its entirety, the Carboniferous dips to the north and is affected by folds with east-west axes. The restricted number of local exposures, mainly in the bed of the local River Taw, reveal steep, near vertical dips, with some overturning. The clay shale geology drives seasonal waterlogging of soils resulting in excess water moving by surface and sub-surface lateral flow to the French drains bordering the fields.

The outfalls (see example in Fig. 2) of the catchments each have H-flumes (TRACOM Inc., Georgia, USA; 1 in 50-year storm event capacity) and pressure transducers (OTT hydromet, Loveland, CO., USA) for monitoring stage and discharge, multi-parameter sondes (originally YSI 6600V2 and latterly YSI EXO 2, Xylem Inc Rye Brook, New York, U.S; including turbidity for estimating net soil loss) and an automatic water sampler (ISCO 3700, Teledyne ISCO). All sondes are installed in a by-pass flow cell due to discontinuous field runoff driven by ambient soil moisture conditions and the concomitant technical problems associated with frequent drying out of sensors.

Given the location of the NWFP catchments immediately adjacent to the River Taw, the experimental platform provided an ideal means of comparing winter sediment loss with the modern background values in Table 1. Since our work examined the robustness of agricultural land to winter weather over the past ~decade, we focussed on the red and green farmlets on the NWFP (Fig. 1). It was anticipated that these two farmlets would illustrate contrasting levels of robustness to wet winter weather. The long-term permanent pasture (green farmlet; Fig. 1) in catchments 4, 5, 6, 12, 13 was maintained throughout the study period. In contrast, the original long-term grass cover in catchments 1, 2, 3, 10 and 15 on the red farmlet was subjected to scheduled ploughing and reseeding for sward improvement in summers 2013–15 (see details in Table S1). Arable crop production was subsequently introduced in these catchments in autumn 2019 with the sowing of winter (Crusoe) wheat. The arable rotation also included spring (Merlin) oats (see details in

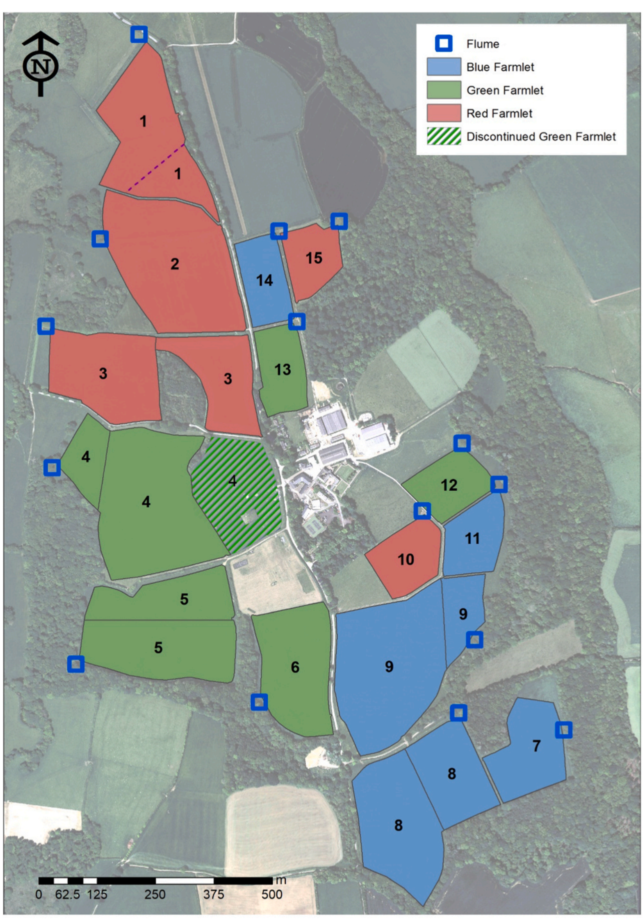

Fig. 1. Layout of the NWFP showing individual fields and flume locations.

Fig. 2. Example turbid flume runoff on the NWFP.

Table S2).

2.3. Rainfall characteristics during the recent wettest winters

Rainfall was measured using tipping bucket rain gauges (range = 0 – 100 mm hr^{-1}; resolution = 0.2 mm). Rainfall data was extracted from the NWFP data portal (https://nwfp.rothamsted.ac.uk/). The characteristics of rainfall during the wettest winters over the past ~decade were mainly summarised using indicators proposed by Dunn et al. (2020). These were: total rainfall for the winter months; maximum 1 day rainfall (R1x); number of days with rainfall > 1 mm (R1D); > 1 mm wet day rainfall amount (R1A); > 10 mm rainfall (R1A); simple rainfall density index (SDII; mm/day); maximum consecutive dry days with rainfall < 1 mm (CDD1); maximum consecutive dry days with rainfall < 10 mm (CDD10); maximum 5-day rainfall (Max5A); maximum consecutive wet days > 1 mm (CWD), and; total rainfall in CWD (CWD-A). Seasonal EI_{30} (Wischmeier and Smith, 1978), representing the sum of monthly rainfall erosivity generated by the maximum 30-minute rainfall intensity for the months comprising each winter, was also calculated (Cardoso et al., 2024). For comparative purposes, these rainfall-based indicators were also estimated for the winters 2012–13 and 2015–16 since those manifested the rainfall totals closest to the long-term average for the study site.

2.4. Hydro-sedimentological data collection and analysis

Hydro-sedimentological data are recorded at 15-minute intervals. Three-monthly calibrations of the turbidity sensors are undertaken. Two sets of sensors enable continuous replacement at the time of re-calibration. Flow-proportional automatic water samples collected at the flumes are filtered gravimetrically (using known sample volumes) for suspended sediment concentrations, using pre-weighed, dried glass fibre filter papers (1.2 µm pore size, Whatman GFC) oven dried at 105 °C for 1 h and re-weighed to determine suspended sediment mass. A total of four calibration relationships were developed and applied for converting turbidity to SSC (Fig. 3). Each calibration was constructed with 95 % confidence limits to estimate the uncertainty ranges for winter sediment loss. Table S3 summarises the use of the calibrations over the study period. Two calibrations ('grass cover original' for YSI 6600V2 sondes, 'grass cover updated' for YSI EXO 2 sondes) were applied to winters where well-established grass cover was present. The 'freshly reseeded grass cover' calibration (for YSI 6600V2 sondes) was used for winters immediately post scheduled ploughing and reseeding of grass cover on the green farmlet between 2013 and 2015. The 'freshly converted to arable cover' calibration (for YSI EXO 2 sondes) was applied to the red farmlet for the winters immediately post tillage and sowing for arable crops in 2019–20 and 2023–24. The data used in this paper are deposited in https://doi.org/10.5281/zenodo.14163304.

2.5. Environmental damage costs

Societal damage costs arising from elevated winter sediment loss were calculated using 2021 unit prices of £ 0.39–0.47 kg^{-1} of sediment (UK Department for Environment, Food and Rural Affairs pers. comm.). These costs encompass those for water treatment or clean up and damage to aquatic ecology and were applied across all winters for consistency. Defra policy teams typically generate these damage costs on the basis of an ecosystems damage cost for the status quo for sediment pressure on the aquatic environment. The costs are then divided by the annual agricultural pollutant loss to derive a typical unit cost which, in

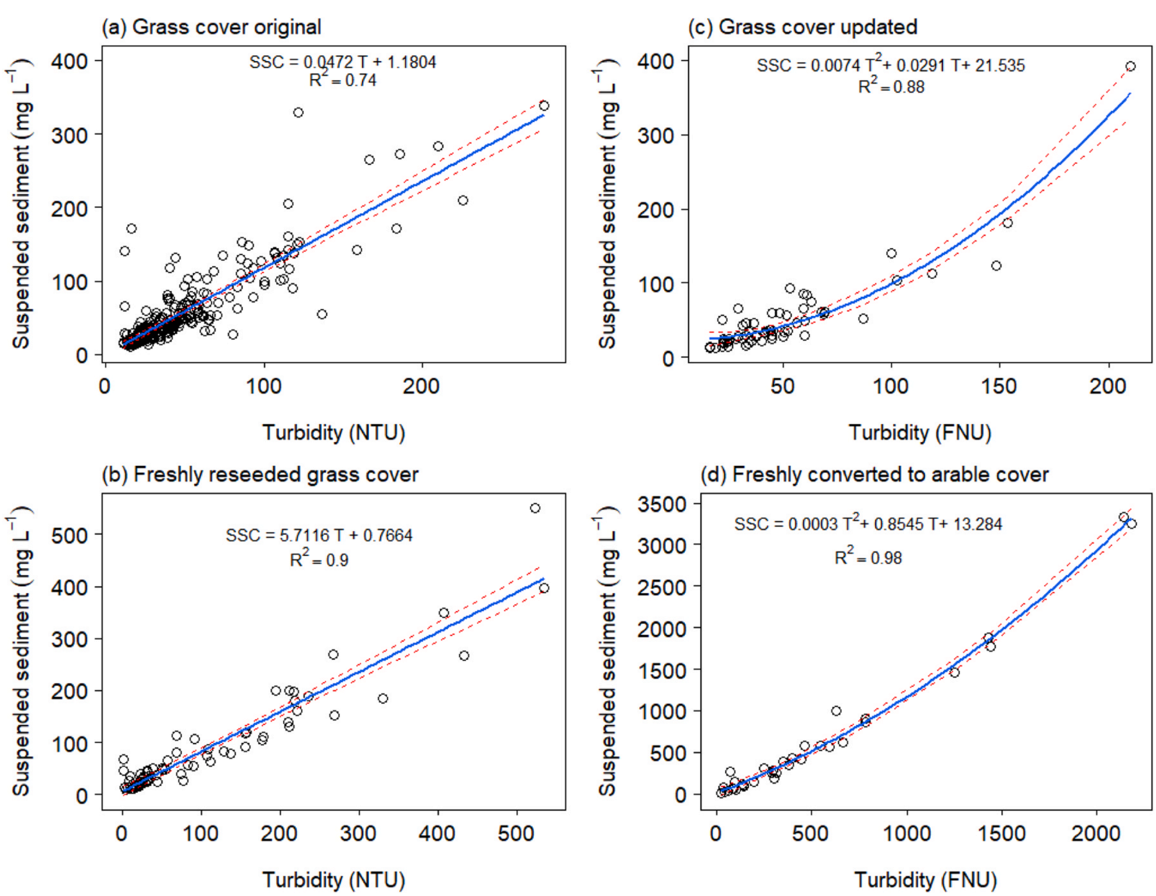

Fig. 3. Calibration relationships for converting the turbidity time series into suspended sediment concentrations. NTU – nephelometric turbidity units. FTU – formazine turtbidity units.

turn, can be multiplied by the sediment loss per winter to estimate the seasonal damage costs reported herein. Since environmental damage resulting from a lack of robustness for sediment loss regulation can vary in space and time, policy teams implement Monte Carlo analysis to generate a distribution for the unit damage costs. This approach assumes flat priors for the estimates and uses 1000 random draws from the distribution to build a probability density function for the overall unit damage costs and associated uncertainty range. The costs assume symmetry between costs to the environment and the benefits of pollution mitigation where mitigation is being assessed explicitly.

3. Results

3.1. Rainfall characteristics of recent winters

Table 2 summarises the rainfall characteristics for the study winters. Winters 2012–13 (344.6 mm) and 2015–16 (333.7 mm) were closest to the long-term average winter rainfall (340.3 mm) for the study site. Winters 2013–14 (514.2 mm) and 2019–20 (408.6 mm) were the two wettest winters, experiencing the highest daily rainfall totals of 45.8 mm and 33.9 mm, respectively. Those same two winters also experienced the highest (60 and 56, respectively) numbers of wet days (>1 mm), > 1 mm wet day rainfall totals (508.6 mm and 404.9 mm), and > 10 mm wet day rainfall totals (259.8 mm and 208.9 mm). The maximum 5-day rainfall totals were, however, highest in winters 2015–16 (76.9 mm) and 2019–20 (78.3 mm). EI_{30} was highest (530.1 MJ ha^{-1} mm^{-1}) in winter 2013–14, with the second highest (369.2 MJ ha^{-1} mm^{-1}) in winter 2019–20. Whilst the CV for winter rainfall totals was only 20 %, maximum daily rainfall (63.8 %), consecutive dry days < 1 mm (62.5 %) and seasonal EI_{30} (57.7 %) varied more between the study winters.

3.2. Exceedance of modern background sediment loss

Table 3 summarises the estimates of winter net sediment loss and corresponding exceedance of the modern background rates. During winter 2012–13, net sediment loss from the field catchments which were all under long-term permanent pasture, ranged from 0.04 to 0.06 t ha^{-1} (catchment 6) to 0.15–0.18 t ha^{-1} (catchment 15). There was no exceedance of modern background rates during this winter. During winter 2013–14, catchments 2 (0.29–0.37 t ha^{-1}) and 15 (0.36 – 0.45 t ha^{-1}), which were ploughed and reseeded during the preceding summer (Table S1), both generated net sediment loss exceeding both the minimum (1.95X and 2.43X, respectively) and maximum (1.05X and 1.30X, respectively) modern background rates (Table 3). Similarly, during winter 2015–16, catchments 1 and 10, which experienced scheduled plough and reseed in the previous summer (Table S1), generated net sediment loss rates (0.27 and 0.19 t ha^{-1}, respectively) greater than the minimum modern background rate. Catchment 3, which was not subject to sward improvement in the preceding summer, also generated net sediment loss (0.35–0.39 t ha^{-1}) exceeding minimum (2.31X) and maximum (1.13X) modern background rates. For winter 2019–20, catchments 1, 2, 3, 10 and 15 had all been converted from long-term pasture to arable (Table S2) and the minimum and maximum net sediment loss (0.67–3.26 t ha^{-1} and 0.92–3.76 t ha^{-1}, respectively) exceeded both the minimum and maximum rates by between 4.49X-21.74X and 2.63X-10.74X (Table 3). For pasture, only catchment 5 generated net sediment loss (0.25–0.35 t ha^{-1}) exceeding (1.66X) or matching modern background rates. No other grass catchments exhibited exceedance during winter 2019–20. Finally, for winter 2023–24, when the arable catchments were all under over-winter stubble, catchments 2 (1.12–1.36 t ha^{-1}), 3 (1.83–2.19 t ha^{-1}) and 15 (0.67–0.97 t ha^{-1}) exceeded modern background rates of net sediment loss (4.45X-12.22X minimum and 2.76X- 6.26X maximum background). No grass catchments generated net sediment loss in exceedance of modern background rates during this winter (Table 3).

Across the five study winters, cumulative net sediment loss (Fig. 4) from permanent pasture ranged between 0.78 and 0.98 t in catchment 12–5.53–7.30 t in catchment 5. For the grass catchments subjected to scheduled sward improvement and arable conversion over the five study winters, cumulative sediment losses ranged from to 1.90–2.90 t in catchment 10–34.70–40.93 t in catchment 3. Over the two most recent study winters (2019–20 and 2023–24) for the catchments converted from pasture to arable, cumulative losses ranged between 1.30 and 2.20 t in catchment 10–29.90–35.40 t in catchment 3. For comparison, the cumulative losses from permanent pasture in the same two most recent winters (2019–20 and 2023–24) ranged between 0.22 and 0.32 t and 2.02–3.10 t. The cumulative net sediment losses from arable land over the two most recent winters were therefore 5.9–11.4 times those from long-term permanent pasture.

3.3. Environmental damage costs

During winter 2012–13, when all catchments were under long-term permanent pasture, environmental damage costs associated with winter net sediment loss ranged between £ 17–25 and £ 20–30 ha^{-1} in catchment 6 to £ 58–71 and £ 70–86 ha^{-1} in catchment 15 (Table 4). For winter 2013–14, the corresponding costs for catchment 15, which had been ploughed and reseeded in the previous summer increased to between £ 142–177 and £ 172–213 ha^{-1}. Winter 2013–14 experienced the highest overall total rainfall, maximum daily rainfall, wet days > 1 mm, wet day rainfall, > 10 mm rainfall total, simple rainfall density index and seasonal EI_{30} (Table 2). For winter 2015–16, damage costs for sediment loss from the permanent pasture catchments were between £ 39–48 and £ 47–58 ha^{-1} for catchment 4 and £ 113–130 and £ 136–156 ha^{-1} for catchment 6 (Table 4). The corresponding highest

Table 2
Winter (December to February, inclusive) rainfall characteristics.

	Winter					
Indicator	2012–2013	2013–2014	2015–2016	2019–2020	2023–2024	CV (%)*
Total (mm)	344.6	514.2	335.7	408.6	382.8	20.0
R1X (mm)	211.3	45.8	32.1	33.9	20.8	63.8
R1D (mm)	44	60	50	56	50	22.5
R1A (mm)	337.4	508.6	327.7	404.9	377.8	21.2
R10A (mm)	146.8	259.8	168.4	208.9	184.9	27.9
SDII (mm/day)	7.7	8.5	6.6	7.2	7.6	12.3
CDD1 (days)	18	12	7	8	14	62.5
CDD10 (days)	33	24	23	23	22	8.2
Max5A (mm)	76.5	71.8	76.9	78.3	58.4	6.2
CWD (days)	19	11	10	23	15	50.0
CWD-A (mm)	166.3	112.8	100.4	177.8	120.7	22.3
Seasonal EI_{30} (MJ ha^{-1} mm^{-1})	190.9	530.1	262.7	369.2	264.5	57.7

* calculated as (Qn/median) * 100

Table 3
Winter net sediment loss and corresponding exceedance of modern background rates.

Catchment	Lower and upper estimates of net sediment loss (kg ha^{-1})									
Grass 2012–2016; arable 2019–2024	Winter 2012–2013	Ex	Winter 2013–2014	Ex	Winter 2015–2016	Ex	Winter 2019–2020	Ex	Winter 2023–2024	Ex
1	0.05	0	0.07	0	0.27	1.80	0.96	6.38	0.01	0
	0.07	0	0.12	0	0.32	0	1.27	3.62	0.08	0
2	0.08	0	0.29	1.95	0.19	1.24	3.26	21.74	1.12	7.48
	0.10	0	0.37	1.05	0.21	0	3.76	10.74	1.36	3.79
3	0.12	0	0.24	1.60	0.35	2.31	2.65	17.65	1.83	12.22
	0.15	0	0.29	0	0.39	1.13	3.11	8.87	2.19	6.26
10	0.07	0	0.04	0	0.19	1.30	0.67	4.49	0.01	0
	0.08	0	0.06	0	0.23	0	0.92	2.63	0.20	0
15	0.15	0	0.36	2.43	0.32	2.15	1.65	11.03	0.67	4.45
	0.18	0	0.45	1.30	0.37	1.05	2.10	6.01	0.97	2.76
Permanent grass										
4	0.08	0	0.24	1.63	0.10	0	0.10	0	0.05	0
	0.11	0	0.30	0	0.12	0	0.17	0	0.13	0
5	0.11	0	0.16	1.08	0.25	1.65	0.25	1.66	0.05	0
	0.13	0	0.20	0	0.29	0	0.35	0	0.11	0
6	0.04	0	0.04	0	0.29	1.94	0.14	0	0.02	0
	0.06	0	0.07	0	0.33	0	0.21	0	0.07	0
12	0.07	0	0.05	0	0.19	1.23	0.11	0	0.01	0
	0.08	0	0.06	0	0.21	0	0.15	0	0.03	0
13	0.08	0	0.12	0	0.11	0	0.13	0	0.02	0
	0.09	0	0.14	0	0.13	0	0.19	0	0.05	0

Ex – exceedance of corresponding modern background rates (Table 1) expressed as a multiplier

costs of between £ 132–150 and £ 159–181 ha^{-1} were estimated for catchment 3 which was reseeded during the summer of 2014 (Table S1). For winter 2019–20, the damage costs for net sediment loss from the catchments converted to arable in autumn 2019 ranged between £ 262–359 and £ 316–432 ha^{-1} (catchment 10) and £ 1272–1466 and £ 1533–1767 ha^{-1} (catchment 2). The corresponding ranges of costs for the permanent pasture catchments were £ 38–66 and £ 45–79 ha^{-1} and £ 97–136 and £ 117–164 ha^{-1} (Table 4). For winter 2023–24, the highest damage costs for the arable catchments with over-winter stubble were £ 698–834 and £ 842–1006 ha^{-1} for catchment 3, whilst those for the permanent pasture catchments were £ 19–49 and £ 22–59 ha^{-1} for catchment 4 (Table 4).

Across the five study winters, cumulative environmental damage costs from permanent pasture ranged between £ 163–203 and £ 197–245 ha^{-1} in catchment 12 to £ 321–421 and £ 386–507 ha^{-1} in catchment 5. For the grass catchments subjected to scheduled sward improvement and arable conversion over the five study winters, cumulative damage costs ranged from £ 382–584 and £ 461–703 ha^{-1} in catchment 10 to £ 1978–2334 and £ 2384–2812 ha^{-1} in catchment 3. Over the two most recent study winters (2019–20 and 2023–24) for the catchments converted from pasture to arable, cumulative damage costs ranged between £ 514–849 and £ 620–1023 ha^{-1} in catchment 10 to £ 11680–13804 and £ 14075–16635 ha^{-1} in catchment 3. For comparison, the cumulative damage costs for sediment loss from permanent pasture in the same two recent winters ranged between £ 86–126 and £ 104–152 ha^{-1} for catchment 12 and £ 786–1208 and £ 948–1456 ha^{-1} for catchment 5. The cumulative damage costs for arable land were therefore 9.1–10.3 times those for permanent pasture over the same time period.

4. Discussion

Soil erosion by water is acknowledged as a global threat to soil and water resources (Xiong and Leng, 2024). Robust land management and policy requires reliable field scale data over several years, with an explicit assessment of uncertainty bounds (Parsons, 2019). The results herein provide valuable quantitative insight, inclusive of uncertainty, into the contrasting robustness of sediment loss regulation provided by permanent or reseeded grassland and arable land in response to winter rainfall over the past ~decade. Table 2 illustrates the contrasting nature of rainfall among winters at the study site. Such variability among the rainfall indices points to the fact that to a degree, soil erosion is sporadic in nature, meaning it is challenging to identify consistent controls across years.

Drawing direct comparisons with previous soil erosion work in the UK is challenged by the fact that those studies tend to report annual, as opposed to winter losses *per se*, but it is reasonable to assume that the winter season will account for a large proportion of annual sediment losses. On that basis, the estimates in Table 3 for permanent pasture fall within the ranges reported by previous studies for grassland on clay soils (Brazier, 2004; Walling er al., 2002; Bilotta et al., 2010; Evans et al., 2017). The presence of a mature grass sward intercepts raindrop impact, improves aggregate stability and water infiltration, reduces runoff slope length and imparts resistance to the erosive energy of overland flow (Morgan and Rickson, 1995). As a result, even during the two winters (2013–14 and 2019–20) with the two highest overall rainfall totals, two highest maximum 1-day rainfalls and two highest EI_{30}, a lack of robust sediment loss regulation (Table 3) was only observed in catchment 5 during winter 2019–20 which, at the time, was the 5th wettest winter recorded in the UK since 1862 (e.g., https://www.metoffice.gov.uk/about-us/pressoffice/news/weatherandclimate/2020/2020-winter-february-stats). Lack of robustness, as evidenced by exceedance of both minimum and maximum modern background rates of net sediment loss, was not observed for the permanent grassland in any other winters. A partial lack of robust sediment loss regulation, signified by exceedance of minimum modern background net sediment loss only, was observed for catchments 4 and 5 in winter 2013–14 and catchments 5, 6 and 12 in winter 2015–16.

Winters 2013–14 (catchments 2, 15) and 2015–16 (catchments 1,10) were both preceded by scheduled ploughing and reseeding for sward improvement (Table S1). Such management activities are recognized as the most destructive interventions in grazing systems (Skinner and Chambers, 1996; MacDonald et al., 2011), running the risk of elevated erosion and water pollution before soil aggregates recover and the sward is fully matured. Winter 2013–14 was characterised by the highest overall rainfall total, highest EI_{30}, highest number of wet days with > 1 mm of rainfall, highest wet day rainfall total and highest > 10 mm rainfall total (Table 2). Catchments 2 and 15, both recently reseeded, manifested a lack of robust sediment loss regulation during this winter (Table 3). Winter 2015–16 experienced the second highest 5-day maximum rainfall but was less stand-out for the other rainfall metrics (Table 2). In response to the rainfall characteristics of this specific

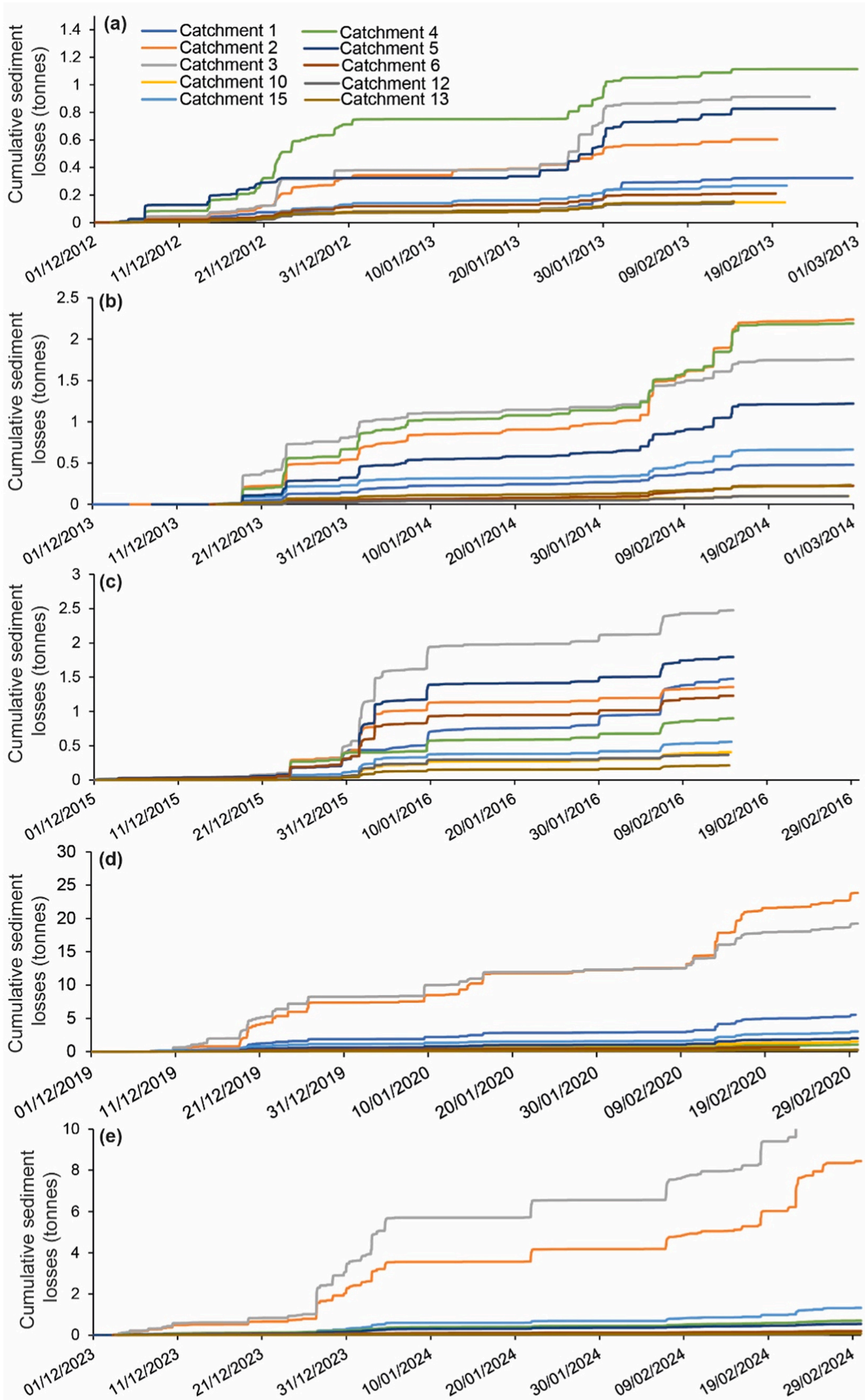

Fig. 4. Cumulative sediment loss over the five winters.

Table 4
Environmental damage costs associated with winter net sediment loss.

Catchment	Lower* and upper# estimates of environmental damage costs (£ ha^{-1})									
Grass 2012–2016; arable 2019–2024	Winter 2012–2013 lower	Winter 2012–2013 upper	Winter 2013–2014 lower	Winter 2013–2014 upper	Winter 2015–2016 lower	Winter 2015–2016 upper	Winter 2019–2020 lower	Winter 2019–2020 upper	Winter 2023–2024 lower	Winter 2023–2024 upper
1	21–29	26–35	29–46	35–55	106–125	127–151	373–494	450–595	1–31	1–37
2	30–39	37–47	114–143	138–172	72–84	87–101	1272–1466	1533–1767	438–532	528–641
3	47–57	57–68	91–109	110–131	132–150	159–181	1009–1184	1216–1426	698–834	842–1006
10	26–32	32–39	15–25	19–30	76–89	91–107	262–359	316–432	3–79	3–95
15	58–71	70–86	142–177	172–213	126–143	152–173	646–821	778–989	260–377	314–455
Permanent grass										
4	46–61	56–74	95–116	115–140	39–48	47–58	38–66	45–79	19–49	22–59
5	44–52	53–62	63–78	76–94	96–112	116–134	97–136	117–164	20–44	24–52
6	17–25	20–30	16–28	20–34	113–130	136–156	53–81	64–97	8–29	10–35
12	27–31	32–38	18–23	22–28	72–82	87–99	42–57	50–69	4–11	5–13
13	31–35	37–42	44–55	55–66	44–49	53–59	52–72	62–87	9–21	11–25

* calculated by combining the lower unit price for environmental damage with the lower and upper values for net sediment loss; # calculated by combining the upper unit price for environmental damage with the lower and upper values for net sediment loss

winter, recently reseeded catchments 1 and 10 generated net sediment loss in exceedance of minimum modern background only (Table 3), signifying a partial lack of robustness. Interestingly, catchment 3, which had been reseeded in the summer of 2014 (Table S1), also exhibited a lack of robust sediment loss regulation for this winter (Table 3) pointing to the prolonged impact of reseeding. Collectively, these results are significant, since on average, 11.8 % (range 4.2–50 %) of farm area in the UK is reseeded annually for grass (Agriculture and Horticulture Development Board, 2018).

The net sediment loss estimates for arable land in Table 3 exceed some of the values reported by previous work on clay soils in the UK (Morgan, 1980; Evans, 2002; Walling et al., 2002; Deasy et al., 2009). All catchments converted to arable in the autumn of 2019 lacked robust sediment loss regulation in the severe wet winter of 2019–20. Conventional tillage practices such as those applied at the study site (Table S2) are well known to modify soil structure, increase losses of macro-aggregate occluded soil organic matter, reduce the water stability of soil aggregates and expose bared tilled soils to the erosive energy of winter rainfall and surface runoff. As a result, rilling was observed on the arable fields in winter 2019–20 (Fig. 5). Despite over-winter stubbles, robust sediment loss regulation was not evident for three of the five arable catchments (catchments 2, 3 and 15) during winter 2023–24 (Table 3). The latter experienced only the third highest rainfall total, > 10 mm rainfall total and EI_{30}, and the lowest five-day rainfall total and maximum daily rainfall (Table 2).

Fig. 5. Winter rilling on an arable field on the NWFP in winter 2019–20.

Up to December 2023, farmers in the UK, for so-called Cross Compliance Pillar I direct payments, were expected to comply with Good Agricultural and Environmental Condition (GAEC) rules 4 – providing minimum soil cover and 5 - minimising soil erosion. Between 2024 and 2027 Cross Compliance payments are being phased out with financial incentives now provided by the new Environmental Land Management Scheme (ELM). However, farmers in England are expected to comply with The Reduction and Prevention of Agricultural Diffuse Pollution (England) Regulations 2018, also known as the New Farming Rules for Water. More specifically, Rule 6 - Reasonable precautions to prevent soil erosion - asks that farmers should take all reasonable precautions to prevent significant soil erosion and runoff from the application of organic manure and manufactured fertiliser, as well as land management and cultivation practices (such as seedbeds, tramlines, rows, beds, stubbles). Where farmers are not planning to establish green cover by October 15th, justifications must be provided for leaving land bare over winter. These can include agronomic or environmental reasons, such as delaying drilling for activities to control persistent weeds, such as blackgrass, or leaving medium or heavy soils to weather before a spring root crop. The data in Table 3 for 2023–24 clearly point to the need to include preventative measures beyond over-winter stubbles to deliver robust sediment loss regulation for heavy arable land in winter. Here, in the new ELM scheme, farmers are now incentivized in the Sustainable Farming Incentive (SFI) to sow a multi-species cover crop by action CSAM2.

Elevated soil erosion and sediment delivery constitute a so-called negative externality, wherein the social cost of intensive farming and the resultant erosion and sediment loss is higher than the private cost. The latter is especially true in the UK where the low erosion rates rarely impact crop productivity. Although the field of environmental economics has resulted in the widespread use of environmental damage costs, based on willingness to pay, this approach to monetization can be critiqued for being simplistic (Centemeri, 2009). One simplification is that the costs are not redistributed to different individuals or between current and future generations (Vallee, 2002). A second is that valuing environmental pollution implies a full understanding of cause and effect and assumes a linear and constant relationship. In reality, agricultural catchments are complex and tipping points may exist wherein further increased pollution exceeds the environmental limit of absorbance and irreversible change results (Sagoff, 1981). Environmental damage costs also assume immediate societal burden and fail to consider time lags inherent in pollution transfer through fluvial systems. Readers should acknowledge that the assumptions used to estimate environmental damage costs can vary, resulting in different values being published in the international literature (e.g., Eory et al., 2013). For our work herein, we used the latest environmental damage costs released by the UK Department for Environment, Food and Rural Affairs. Whilst this study used the grass and arable crop covers implemented on the NWFP over the past decade or so, readers are reminded that soil erosion rates are typically higher for various other so-called high-risk crops, such as maize, potatoes, spring cereals and salad crops (Evans et al., 2016). Climate change might feasibly see a change in cropping patterns in the SW of England (Ritchie et al., 2019). Such a scenario would see a further challenge to the robustness of sediment loss regulation including during not only winter in tandem with projected changes in the extent of cereal cultivation in the SW of England, but also other seasons such as spring and summer if other high-risk crops become more extensive. Our work, however, focussed solely on winter soil loss and winter weather over the past ~ decade. Equally, there is, albeit spatially-variable, evidence that rainfall intensity is changing across the UK under climate change (Burt et al., 2015) and this would also clearly impact the robustness of sediment loss regulation under either current or projected alternative cropping patterns in the future. These limitations for our work herein should therefore be borne in mind.

5. Conclusion

Changing climate is driving a desire among farmers, land managers and environmental agencies to understand, in a quantitative sense, the impacts of wetter winters on the resilience of agroecosystems. Heavily instrumented and data intensive platforms assembling longer-term time series provide a valuable opportunity to respond to this information need. The results herein clearly point to the more pronounced lack of resistance to elevated winter rainfall in arable systems, but also underscore the risks to the resilience of sediment loss regulation in pasture systems due to disturbances associated with scheduled ploughing and reseeding where such sward management activities are followed by a wetter than average winter. The use of paleolimnological evidence to frame the assessment of robustness, as a component of resilience, warrants wider application and therein, improved estimates of modern background sediment loss rates could be generated by expanding available dated lake core databases.

CRediT authorship contribution statement

A.L. Collins: Writing – review & editing, Writing – original draft, Supervision, Project administration, Methodology, Funding acquisition, Formal analysis, Conceptualization. **H.R. Upadhayay:** Writing – review & editing, Formal analysis. **Y. Zhang:** Writing – review & editing, Formal analysis. **L. Olde:** Writing – review & editing, Data curation. **H. Sint:** Data curation.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agee.2025.109713.

Data Availability

I have shared in data in: https://doi.org/10.5281/zenodo.14163304

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