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National-scale geodata describe widespread accelerated soil erosion

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ABSTRACT

Accelerated soil erosion can result in substantial declines in soil fertility and has devastating environmental impacts. Consequently, understanding if rates of soil erosion are acceptable is of local and global importance. Herein we use empirical soil erosion observations collated into an open access geodatabase to identify the extent to which existing data and methodological approaches can be used to develop an empirically-derived understanding of soil erosion in the UK (by way of an example). The findings indicate that whilst mean erosion rates in the UK are low, relative to the rest of Europe for example, 16% of observations on arable land were greater than the supposedly tolerable rate of 1 t ha^{-1} yr^{-1} and maximum erosion rates were as high as 91.7 t ha^{-1} yr^{-1}. However, the analysis highlights a skew in existing studies towards locations with a known erosion likelihood and methods that are biased towards single erosion pathways, rather than an all-inclusive study of erosion rates and processes. Accordingly, we suggest that future soil erosion research and policy must address these issues if an accurate assessment of soil erosion rates at the national-scale are to be established. The interactive geodatabase published alongside this paper offers a platform for the simultaneous development of soil erosion research, formulation of effective policy and better protection of soil resources.

1. Introduction

Inadequate and inappropriate management of land can result in accelerated soil erosion. Accelerated soil erosion and the subsequent decline in soil depth has damaging environmental, and consequently financial, impacts that have implications across all land cover and scales of land management (Montgomery, 2007). On arable land for example, the loss of top soil and nutrients bound to soil particles can result in a substantial decline in soil fertility. Losses in crop productivity can occur, necessitating the use of additional fertilizers, increasing management costs and raising the risk of further environmental degradation (Bakker et al., 2005; Lal, 1998; Pimentel, 2006). Furthermore, significant erosion and runoff events are particularly prevalent when crop cover is low, such as when crops are establishing, leading to further losses in annual crop yields (Evans, 1990a). In addition to on-site costs, off-site impacts such as decreased surface water quality (Brazier et al., 2007; Grand-Clement et al., 2014; Tilman et al., 2002), the sedimentation of receiving surface waters (Rickson, 2014; Vandaele and Poesen, 1995; Walling et al., 2002) and large muddy flooding events (Boardman, 1995; Holman et al., 2000), can also have significant cleanup and mitigation costs (Graves et al., 2015; Pimentel et al., 1995).

Historically, local and global requirements to intensify agricultural productivity to satisfy growing demand have been linked to increased rates of soil erosion (Montgomery, 2007). Consequently, with population growth projected to continue throughout the next century (Gerland et al., 2014), it is important that there is effective legislation in place to manage both food security/production and continued urban land

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expansion, in terms of long-term impacts on soil condition and fertile soil availability (Amundson et al., 2015; Barthel et al., 2019; Foley et al., 2005; Seto et al., 2011). Whilst unsustainable in the long-term, the impacts of declining soil fertility are unlikely to be felt within a single political cycle, unless a tipping point, e.g. a rapid food-price increase or widespread crop failure, is reached (Puma et al., 2015; Rocha et al., 2018). To this end, without quantitative and current evidence of soil erosion (and how rates change over space and time), little motivation exists for the development of legislation to manage soil resources better (Evans, 2010). Arguably, strong policy in this area is long overdue (Brazier et al., 2011).

Globally, there are very few examples of efforts to integrate empirical evidence of soil erosion at national, or other policy-relevant scales, with a few notable exceptions (García-Ruiz et al., 2015; Verheijen et al., 2009). However, this exercise has not yet led to legislation that delivers strategic management of soils to mitigate accelerated erosion and associated problems. Whilst Humphries and Brazier (2018), for example, call for national-scale soil condition monitoring, such environmentally progressive approaches have not been taken up, despite the long history of soil erosion research globally. We now set out to explore the potential for collating existing data to support policy and legislative needs, using the UK as an example.

In the UK, Evans' (1971) paper, through collating evidence of soil erosion and highlighting the need for soil conservation at a national-scale, presented an argument for quantitative assessments of soil erosion (Evans, 1971). As research methods evolved, *ad hoc* reports of soil erosion events paved the way for monitoring programs designed to develop a scientific understanding of the relationship between environmental conditions, land management and soil erosion rates (Evans, 1988; Harrod, 1998; Quine and Walling, 1991). As a result, the UK now has a rich dataset of soil erosion observations, which have been collected using a wide range of methodologies, across various spatial and temporal scales (Boardman, 2013, 2002; Brazier, 2004).

Ironically, although many of these soil erosion studies have illustrated that soil erosion can occur in the UK (Evans, 1988; Harrod, 1998; Walling, 2008), understanding whether or not the UK has a soil erosion problem still remains unclear. Furthermore, outside of the so-called 'tolerable' erosion rate of ≤ 1 t ha^{-1} yr^{-1} proposed by Verheijen et al. (2009), the definition of what would constitute an erosion problem in the UK remains problematic in the absence of a comprehensive national-scale, empirically-derived study and review of soil erosion rates (Reed, 1979). However, for the purpose of this paper, accelerated erosion is considered to be a problem if rates are in excess of 0.1 kg m^{-2} yr^{-1} or 1 t ha^{-1} yr^{-1}, as presented in the UK context by Morgan (1985). However, even at this level, 'tolerable' erosion rates will be significantly higher (often orders of magnitude higher) than soil formation rates (Evans et al., 2019).

Open access to geospatial, empirical measurements of soil erosion is imperative in light of the difficulties of accurately modelling soil erosion at national-scales, under high levels of uncertainty (Evans and Boardman, 2016a; Evans and Brazier, 2005). However, although numerous reviews of existing soil erosion studies have been carried out (e.g. Boardman, 2013, 2006, 2002; Boardman et al., 1990; Brazier, 2004; Evans, 2005, 1995, Evans et al., 2017, 2015), to the authors' knowledge, there is no single resource that brings all of this work together (in an open source, open access format) to facilitate better understanding of soil erosion at the national-scale. It could be argued, therefore, that a lack of data sharing, collation, and fair comparison has resulted in an ineffective national policy on soil erosion management in the UK to-date, and the same argument could well be levelled globally.

In the absence of a national-scale overview of soil erosion rates, collating and evaluating the existing state of knowledge on soil erosion at a national-scale can provide guidance for the heuristic development of both soil management policy and scientific research. The primary aim of this research was to identify the extent to which all available existing data and methodological approaches can be used to develop an empirically-derived understanding of soil erosion in the UK (by way of example), through:

1. Collating all available, empirically-derived soil erosion datasets into a spatially explicit and open access resource.
2. Developing an understanding of observed magnitudes of, and variability in, soil erosion rates.
3. Explore the significance of environmental controls on erosion rates, in a UK setting.
4. Evaluate the impact that monitoring approaches have had on the quantification of soil erosion rates.

Using this information, the paper then sets out recommendations for empirical soil erosion research moving forward, which is of local, national and global relevance to the soil erosion community, as well as environmental policy makers and regulators.

2. Materials and methods

Reflecting the dual purpose of our research i.e. to critically assess the value of existing data on soil erosion rates in the UK and to build an open access resource, before being transformed into a open access and global resource, geodata collation and analysis first focussed on UK-based data.

2.1. Data collation

Empirical, spatiotemporally explicit and on-site soil erosion observations were sourced from peer-reviewed literature, government-funded project reports and the personal datasets of members of the scientific soil erosion community who were willing to share their data. A critical, underpinning assumption of this work was that all data are of equal value to the erosion community and therefore should be included in the database, regardless of the technique used for erosion measurement. Whilst all techniques are open to criticism, for examples of commentary in this regard see Boardman (2006) and Parsons & Foster (2013), the approach was taken to ensure that, as far as possible, all data were included in an open access repository, so that end users could decide upon the value in future analyses of the data. However, we limited the records included in the database to spatially explicit observations of on-site erosion and thus records based on suspended sediment yields and bathymetry (i.e. off-site representations of soil erosion) were not included.

Where reported in the original material, the following information was recorded for each entry: location (including coordinates, country); details of the study site (including land cover, soil association or type, soil texture, topography, land use, precipitation); information on the study design (including spatial extent, monitoring technique and duration); and soil erosion observations (including rate or volume, erosion process and causes, if known). Preference was given to reporting and analysing erosion rates in the units of t ha^{-1} yr^{-1}, however, the observational unit used in the originally published material was also recorded and retained for each entry, to avoid future misrepresentation.

For compatiblity with UK-based data sources, Easting and Northing, based on the British National Grid (BNG) georeferencing system, were noted. Missing location data were extracted from the Ordnance Survey 1:50 000 Gazetteer, and the BNG reference for the nearest town or village was recorded. To allow the open access geodatabase to be visualised on a 1 km grid and to maintain sufficient anonymity with respect to specific erosion sites, the last three digits within each BNG coordinate were replaced with '500' e.g. TL123456 to 512,500 Easting, 245,500 Northing, before being converted to longitude/latitude, using WGS84.

To permit trends in erosion rates and descriptive statistics to be derived for the whole dataset, soil erosion units were standardised to t

$ha^{-1} yr^{-1}$. Volumetric observations were converted to $t\ ha^{-1}$ using the National Soil Inventory representative bulk density information for each individual Soil Series contained within the HORIZON Hydraulics database (© Cranfield University). Where bulk density information was missing for the Soil Association or the record lacked explicit bulk density information, the mean values for the habitat class reported by Emmett et al. (2007: Table 1.4) were used.

The spatial nature of the data permitted the extraction of further information from observed and modelled environmental datasets. For consistency soil texture information extracted from the NATMAP topsoil texture database (© Cranfield University) using ArcGIS Pro (version 2.1.3, Esri Inc. 2017) was used in our analysis. The NATMAP database categorises soil texture classes based on the percentage of sand, silt and clay particles. Where possible, the 1 km CEH Gridded Estimates of Areal Rainfall (CEH-GEAR) dataset (Tanguy et al., 2016) was used to extract the total precipitation for the 12-month period prior to the date of each study based on the location data if this information was not included in the original record. The mean total rainfall within the survey areal extent was used for overflight transect data (Evans, 1988) or where the centroid was not within a singular CEH-GEAR cell. The precipitation concentration index (Martin-Vide, 2004) was calculated using daily CEH-GEAR datasets, for the year prior to each study and the R package precintcon (Povoa and Nery, 2016). CEH-GEAR data were extracted using R version 3.3.0 (R Core Team, 2016) using RStudio version 0.99.902 (RStudio Team, 2015) and the following packages: ncdf4 (Pierce, 2017), lubridate (Grolemund and Wickham, 2011), rgdal (Bivand et al., 2018), raster (Hijmans, 2018), exactextractr (Baston, 2020), doParallel (Microsoft Corporation and Weston, 2019) and tidyverse (Wickham et al., 2019). Due to the licencing restrictions, these extracted data have not been included in the open access resource.

2.2. Data analysis

Statistical analyses and figure construction was carried out in R version 3.5.1 (R Core Team, 2016) using RStudio version 1.1.456 (RStudio Team, 2015) and the following packages: tidyverse (Wickham et al., 2019), ggplot2 (Wickham, 2016a), dunn.test (Dinno, 2017), scales (Wickham, 2016b), stringr (Wickham, 2016c) and cowplot (Wilke, 2017). Due to the skewed nature of the erosion rate data i.e. zero and low-rate inflated, the non-parametric Independent-Samples Kruskal-Wallis rank sum test was utilised for discrete data, followed by a pairwise Dunn's test to identify significant environmental or monitoring approach effects. Spearman's correlation was employed to assess the strength and direction of the relationship between continuous environmental variables (i.e. precipitation) and erosion rates. A critical value of $p \leq 0.05$ was used for statistical significance for the Kruskal-Wallis and Spearman's tests and a $p \leq \alpha/2$ (i.e. 0.025) for the post hoc Dunn's tests.

Historically the bulk of upland (defined as land over 300 m above sea level, typically moorland or mountainous areas) erosion studies in the UK have been based on sediment yield estimation (representing the off-site manifestation of soil erosion), which was outside of the remit of this paper, and therefore this data was not included in the geodatabase. The upland observations included in the geodatabase are based on the findings of McHugh et al. (2002) and represent a cumulative eroded volume at each location, rather than a rate of erosion. Due to the unknown time period the observations represent, a comparison between upland and the remaining land cover classes was not possible and has therefore been excluded from this study.

2.3. Web-based open access geodatabases

The geodatabase has been converted into an open access website using R version 3.5.1 (R Core Team, 2016) via the RStudio IDE, version 1.1.456 (RStudio Team, 2015) and the implementation of shiny (Chang et al., 2018) and leaflet (Cheng et al., 2018). The web interface includes a UK-focussed interactive web-map and open access geodatabase, an interactive global map, and a data collation and download tool for the collaborative development of a global resource.

The web-map allows users to explore the data collated for this paper and contribute further data for the creation of a global soil erosion resource. The web-map has been released as an open access resource alongside the publication of this paper, along with the source code used to build the web-map. The open access resources are available at https://piabenaud.shiny.io/SoilErosionMap, where members of the soil erosion community can also contribute data, and the source code and data is available at https://doi.org/10.5281/zenodo.3736496. Please see the supplementary information for further information on how to contribute.

3. Results

3.1. Soil erosion in the UK

Reporting units were not consistent across the data, reflecting the varied methodologies that have been utilised within soil erosion research. The units used across the different studies included: rate ($t\ ha^{-1} yr^{-1}$), volume ($m^3\ ha^{-1} yr^{-1}$), net rate (soil lost from field, $t\ ha^{-1} yr^{-1}$), gross rate (total mass of eroded sediment per field, $t\ ha^{-1} yr^{-1}$), areal extent (ha or number of fields). However, most of the records could be standardised to mean $t\ ha^{-1} yr^{-1}$, except for the upland values which have been excluded from analysis as a result.

The UK has a wealth of soil erosion observations: a total of 1566 individual records were compiled. Of the records held within the database (Fig. 1), 651 (41.5%) report a presence of soil erosion at the location, with rates ranging from $<0.01\ t\ ha^{-1} yr^{-1}$ to a maximum individual record of $143\ t\ ha^{-1} yr^{-1}$. The latter was reported by Chambers and Garwood (2000) for a single site in West Sussex, although the median for the 42 sites studied in the area was $0.08\ t\ ha^{-1} yr^{-1}$. The area experienced 91 mm of rainfall over 3 days and the main cause of erosion was thought to be related to landform rather than low crop cover (winter cereal, in this instance) (Chambers and Garwood, 2000).

Arable land had the highest recorded mean erosion values, followed by grassland and other(s), which included a woodland, hop-yard and golf course (Table 1). The mean erosion rate for arable land was $1.27\ t\ ha^{-1} yr^{-1}$, and the mean for grassland was $0.72\ t\ ha^{-1} yr^{-1}$. There was no significant difference between the grassland and arable erosion rates when comparing the distribution of all records ($p = 0.61$) and the distribution of the results with a presence of erosion ($p = 0.26$), as visible in Fig. 2A and Fig. 2B. The mean erosion rate for the arable records exceeds the suggested 'tolerable' erosion rate of $1\ t\ ha^{-1} yr^{-1}$, however, the median rate for all cover classes was $0\ t\ ha^{-1} yr^{-1}$. The skew towards low value erosion observations for both arable and upland data is clearly illustrated in Fig. 2C and Fig. 2D, which exclude the zero value observations for clarity, and Fig. 2A. 73.6% of all values were less than or equal to $1\ t\ ha^{-1}$, consisting of 65% of upland, 84% of arable and 80% of grassland records.

3.2. Environmental controls

3.2.1. Soil texture

The spatial distribution of soil erosion studies indicates that they have been carried out across a wide range of soil textures in England and Wales (Fig. 3). Of the data included, soil texture class was extracted for 381 arable, point-based records (ca. 55% of total). Grassland studies were not included in this analysis due to insufficient sample sizes. Clay loam was the most common soil texture in the selection, with 173 records, and fine sandy silt loam and loamy medium sand were the least represented, with a single record each. 146 of the records coincided with soil erosion, where loamy coarse sand soils had the highest incidence of soil erosion (100%, $n = 6$), while coarse sandy silt loam soils

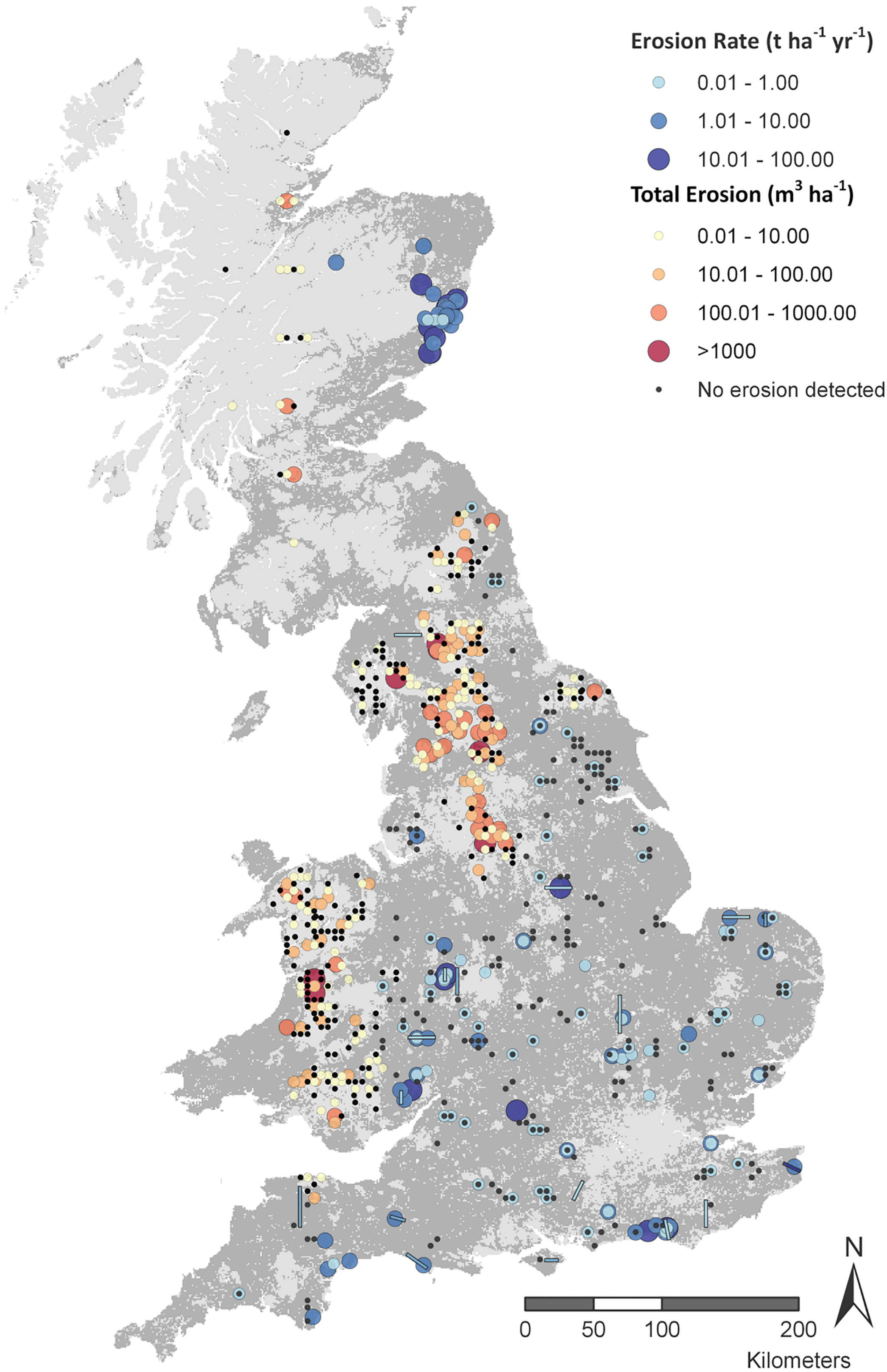

Fig. 1. The spatial distribution and magnitude of soil erosion records (t ha^{-1} yr^{-1} for arable and grassland classes, and total t ha^{-1} for upland classes). Rectangles are areas covered by Evans (1988) overflight transects. The darker shading indicates the distribution of arable or improved grassland (i.e. pasture) areas in the UK, based on LCM2000map.

had the lowest occurrence of erosion (17%, n = 6), excluding the textures with less than 3 records.

Fig. 4 shows the distribution of soil erosion rates within each soil texture class, based on the arable records with a presence of erosion. To test for significant difference in erosion rates between soil textures, pairwise analysis revealed that there was a significant difference

Table 1
Summary statistics for the records held within the database. *The values reported for uplands represent the presence of erosion at a single point in time (1999), rather than annual rates.

Land Cover	N	N (with erosion)	Mean (t ha^{-1} yr^{-1})	Median (t ha^{-1} yr^{-1})	Minimum (t ha^{-1} yr^{-1})	Maximum (t ha^{-1} yr^{-1})
Arable	698	260	1.27	0	0	91.7
Upland*	822	380	76.2*	0.06*	0	5409.1*
Grassland	31	9	0.72	0	0	5.3
Other(s)	15	2	0.44	0	0	6.5

between the distribution of the records for the silty clay loam class (n = 18) and the following texture classes: clay (p = 0.0038, n = 14), clay loam (p = 0.0077, n = 55) and sandy clay loam (p = 0.0143, n = 5). There was also a significant difference between the distributions of results within the loamy coarse sand texture class (n = 6) and both clay (p = 0.0163, n = 14) and sandy clay loam (p = 0.0239, n = 5). The highest erosion rate (91.7 t ha^{-1} yr^{-1}) was found on a medium sandy loam soil, demonstrating the excessively high soil loss rates that can occur on light soils under arable land use. Based on the erosion risk classification for the UK by Evans (1990b), where 1 = low risk and 5 = very high risk, this particular soil association has an erosion risk of 2 (although annotated as 'locally risk of erosion is greater'). Evans (1990b) used soil texture as well as land use, landform and erosion observations to classify UK soil associations into these risk categories. From the records with erosion, the loamy coarse sand group had the highest median erosion rate (4.35 t ha^{-1} yr^{-1}), where all soils were classified as a very high risk by Evans (1990b). Of the same subset, the sandy clay loam soils had the lowest median erosion rate (0.6 t ha^{-1} yr^{-1}), whilst all being soils classified as erosion risk 3 (moderate).

Fig. 5 further illustrates the relationship between erosion rate and erosion risk (according to Evans, 1990b classification), for all arable records (Fig. 5A) and arable records with a presence of erosion (Fig. 5B). Of the records held, the greatest number were for erosion risk 3 (moderate erosion risk) which had 130 records, followed by risk 2 (n = 112), risk 1 (n = 106), risk 5 (n = 23) and risk 4 (n = 8). For both datasets presented in Fig. 5, there was a significant difference in the distribution of erosion rates between erosion risk 2 and 3 (p = 0.0003 and p = 0.005, respectively) and 5 (p = 0.0100 and p = 0.0124, respectively). There was, however, no significant

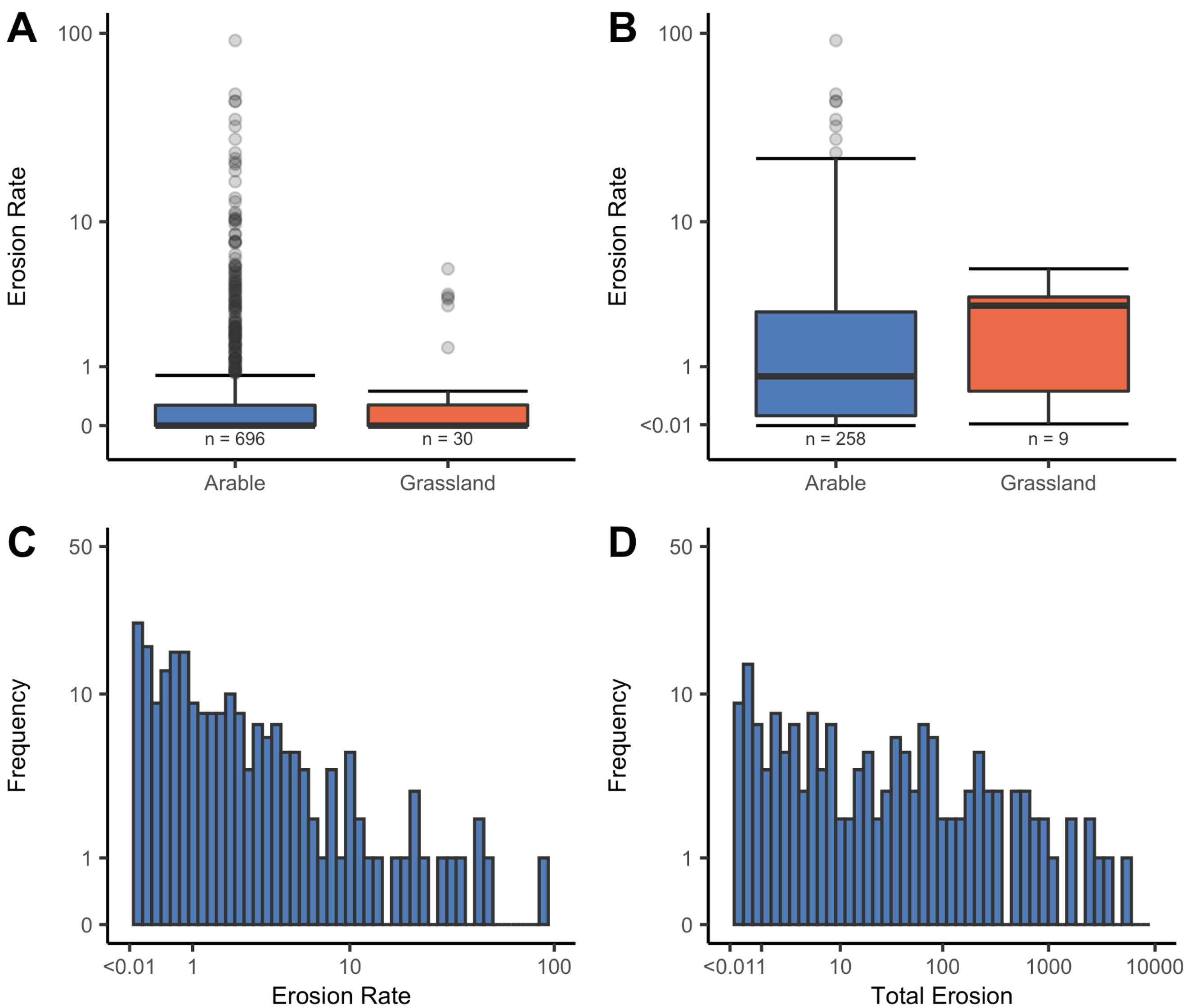

Fig. 2. Graphical representations of the distribution of erosion records: A) boxplots comparing all arable and grassland observations, B) boxplots comparing arable and grassland with erosion, C) histogram of the frequency of erosion rates (t ha^{-1} yr^{-1}) for arable land with erosion (n = 260), and D) histogram of the frequency of total erosion observed (t ha^{-1}), for upland observations collected in 1999 with erosion visible (n = 206) from McHugh et al. (2002).

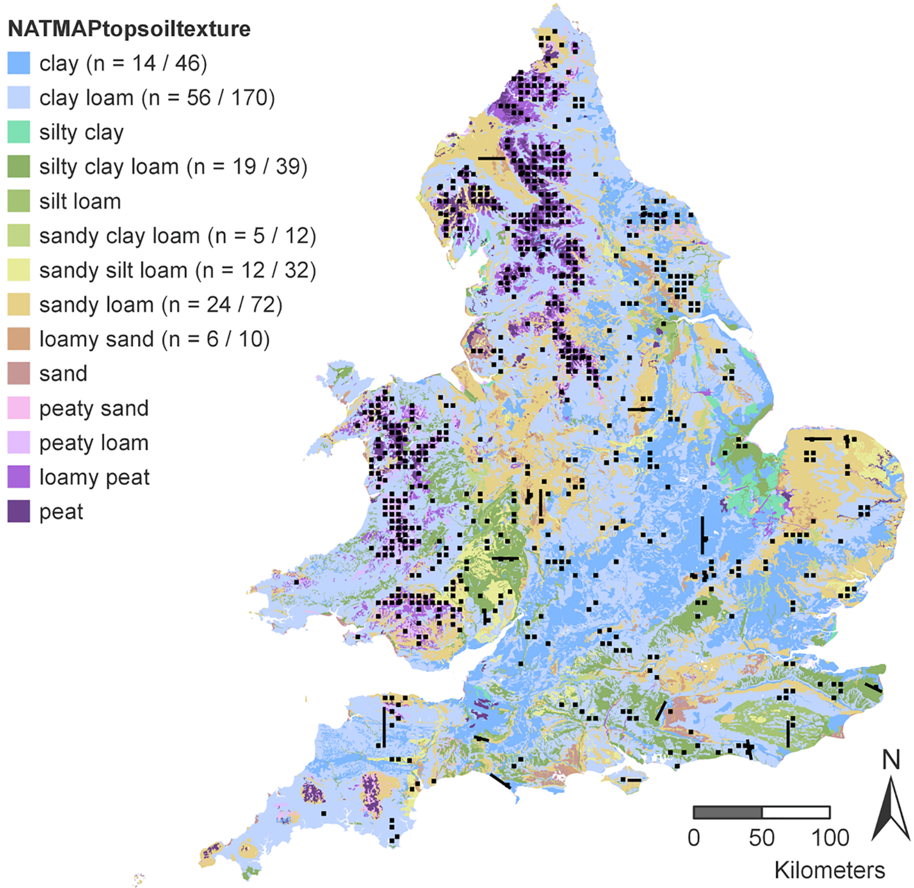

Fig. 3. The spatial distribution of soil textures in England and Wales, with the locations of soil erosion records held within the geodatabase shown in black, n values in parentheses are for arable records and indicate the number of sites with erosion detected (first value) and the total number of studies per soil texture. Rectangles are areas covered by Evans (1988) overflight transects. The soil texture information was sourced from the National Soil Map of England and Wales, NATMAP topsoil texture, 1:250,000 © Cranfield University.

difference between remaining pairs.

3.2.2. *The role of slope*

Fig. 6 illustrates the relationship between slope gradient (%) and erosion rate (t ha^{-1} yr^{-1}), based on n = 627 records. There is a significant moderate positive correlation between steeper slope gradients and erosion rates for arable ($\rho(595) = 0.39$, $p = <0.001$) and a non-significant weak positive correlation for grassland ($\rho(30) = 0.13$, $p = 0.48$). Furthermore, a wide range of erosion rates were associated with any slope gradient (Fig. 6). However, of the studies examined, 74.7% were carried out in localities with a mean slope of <10%, and only 4.5% were from areas with a mean slope greater than 20%.

3.2.3. *Precipitation*

Fig. 7A illustrates the distribution of results based on the annual precipitation data which could be extracted from CEH-GEAR and comparable data within a limited number of studies (i.e. annual totals), totalling 661 observations. For these records, total annual precipitation ranged between a minimum of 321.2 mm and a maximum of 1504 mm. There was a weak positive relationship between the total annual precipitation and soil erosion rates ($\rho(661) = 0.23$, $p = <0.001$). There was also a weak positive association between the 95th percentile of daily rainfall (over the 12 months prior to the study) and erosion rate ($\rho(622) = 0.21$, $p = <0.001$). There was no significant correlation between precipitation concentration index (PCI) and erosion rates when considering sites with and without erosion observed ($\rho(622)$ -0.04,

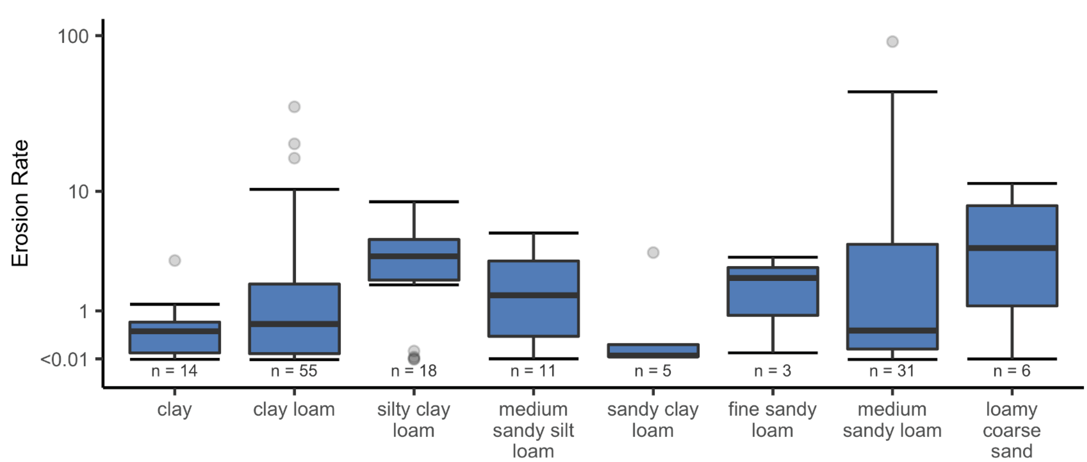

Fig. 4. The distribution of erosion rates (t ha^{-1} yr^{-1}), on a log scale, within each soil texture class, for arable records with a presence of erosion.

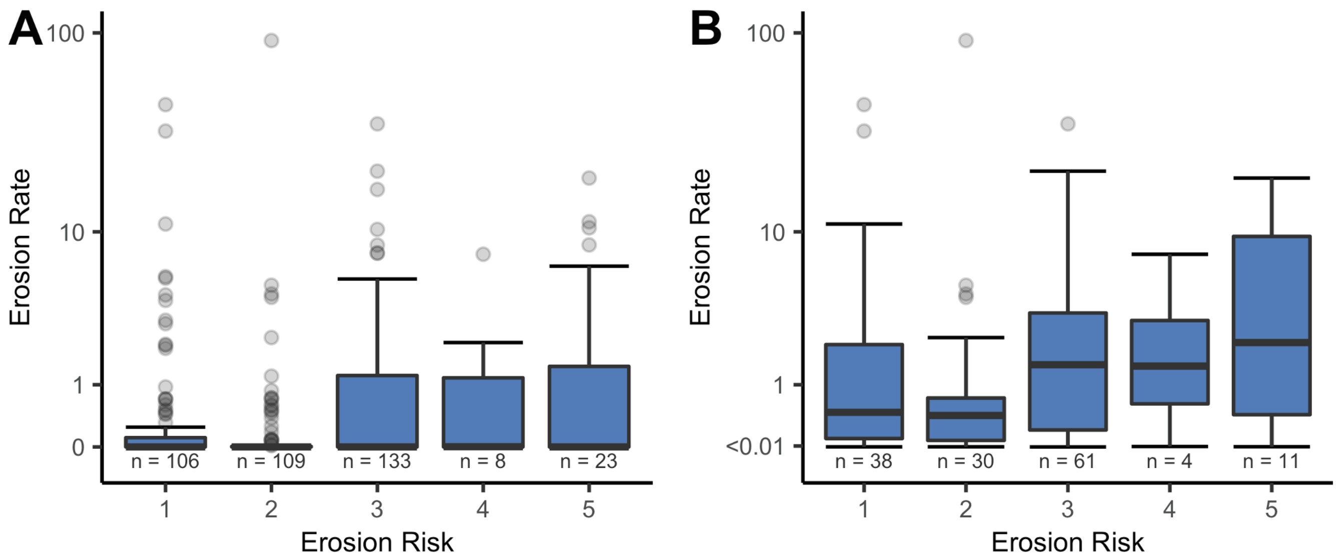

Fig. 5. The relationship between erosion risk (Evans, 1990b) and erosion rates (t ha^{-1} yr^{-1}) on a log scale for A) all values, and B) records with a presence of soil erosion.

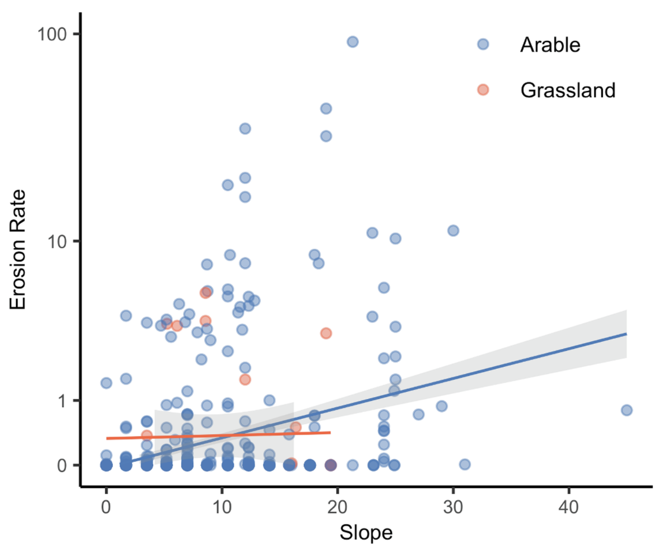

Fig. 6. The relationship between erosion rates (t ha^{-1} yr^{-1}) on a log scale and slope gradient (%) with 95% confidence intervals shaded.

p = 3) (Fig. 7B). A significant negative relationship between PCI and erosion rates for sites with erosion ($\rho(142) = -0.3$, $p = < 0.001$) was observed. There was no difference in the strength in the observed relationship for both annual precipitation and the daily 95th percentile when records without erosion were excluded from the analysis.

3.3. Monitoring approaches

3.3.1. Spatial scale

To analyse the importance of spatial scale, or more specifically the spatial extent of soil erosion studies, records were grouped into 5 classes: plot (for bounded areas), hillslope (for unbound areas), field (for areas naturally defined by constructed field perimeters, such as fences or hedgerows), catchments (for study perimeters defined by a hydrological catchment, extending greater than a single field) and regional (for studies covering multiple fields, over a large area i.e. overflight transects). Field-scale studies have been the most common method for conducting soil erosion assessments in the UK to-date (n = 575), and plot scale studies the least common (n = 20).

For the subset of data that included all arable records (Fig. 8A), there was a significant difference between the distributions of erosion rates collected at field scale and all categories ($p = < 0.001$). When considering only arable records with a presence of erosion (Fig. 8B), there was a significant difference between the field scale observations and plot studies ($p = 0.0193$), hillslope studies ($p = 0.0032$) and catchment studies ($p = 0.0011$). However, there was no significant difference between all other study scales, highlighting the complexity of scale relationships in soil erosion. Of the subset with erosion recorded, the field scale studies had the lowest median erosion rate (0.48 t ha^{-1} yr^{-1}), while the hillslope studies had the highest median erosion rate (2.95 t ha^{-1} yr^{-1}). As visible in Fig. 8, plot scale studies had the greatest interquartile variability in erosion rates. Interestingly, the field scale studies had the greatest range of rates observed, capturing the intersection between the *ad hoc* studies carried out following significant erosion events and the national-scale study carried out by the SSLRC.

3.3.2. Monitoring techniques

To assess the impact of different monitoring techniques on the soil erosion measurements, records were divided into three broad categories: volumetric surveys, runoff and sediment capture, and tracer experiments. Volumetric surveys were the most common, representing 92% of all records, followed by tracing experiments, which consisted solely of ^{137}Cs derived results. Runoff and sediment collection was the least common methodology, consisting of 3% of all records, reflecting the resource-intensive nature of this technique. Of the arable records in the subset of data with a presence of erosion (Fig. 9B), the ^{137}Cs surveys yielded the highest median erosion rates at 3.12 t ha^{-1} yr^{-1}, while the runoff and sediment collection studies had the lowest median erosion rate (0.25 t ha^{-1} yr^{-1}). Pairwise comparisons found a significant difference between the distributions of erosion rates measured by volumetric surveys and both other categories ($p = < 0.001$), for the all observations subset (Fig. 9A). However, when comparing the distribution of results for the records with a presence of erosion (Fig. 9B), there was a significant difference between the ^{137}Cs method and both volumetric surveys ($p = < 0.001$) and sediment and runoff collection ($p = < 0.001$).

3.3.3. Site selection

Records were grouped into the following categories based on the rationale for site selection: sites known or with a history of erosion, sites predicted to erode or perceived as ‘high-risk’ based on soil texture or land use, sites based on sampling grid design, sites selected on

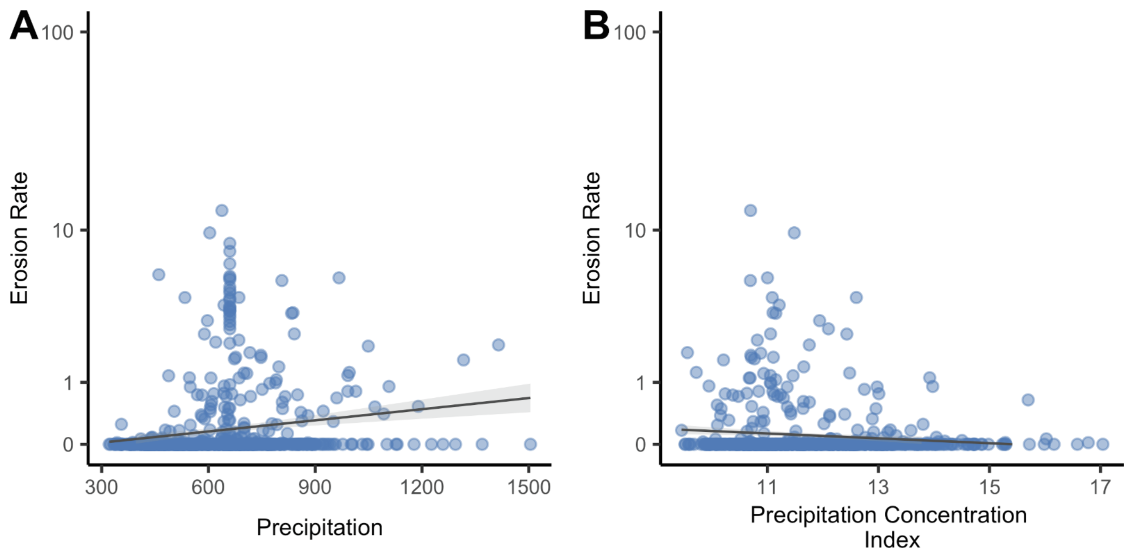

Fig. 7. The linear relationship between erosion rates (t ha^{-1} yr^{-1}) on a log scale and A) annual precipitation (mm) and B) precipitation concentration index, with 95% confidence intervals shaded.

statistically-based design, and others (which includes experimental farms). The distributions of the results within each category are presented in Fig. 10. In the UK, there were no sites selected on a statistically-based design. 14.1% of erosion records fell into the 'predicted to erode' group, 8.1% in the 'known to erode group' and 77.2% in the 'sampling grid' category. However, while only 17.7% of the sampling grid records report a presence of erosion, 99% of the sites selected based on a predicted likelihood of erosion and 96% of the known to erode sites report soil erosion (Fig. 10).

The distribution of erosion rates was significantly different between the sampling grid observations and both the sites known and predicted to erode ($p = < 0.001$), across the whole dataset (Fig. 10A). However, when considering only the records with erosion observed, there was a significant difference in the distribution of all groups (Fig. 10B). Whilst the number of the predicted to erode and sampling grid sites were similar, with 98 and 94 observations respectively, the range of erosion rates within the sampling grid group was much smaller, with a greater number of outliers (9.6%, compared with 3.1%). Furthermore, the median erosion rate for the 'known to erode' results was an order of magnitude higher than the 'sampling grid' median, with values of 1.1 and 0.08 t ha^{-1} yr^{-1} respectively. The impact that site selection can have on skewing the understanding of soil erosion is further demonstrated in Fig. 10A, which utilised all arable and grassland records. Unlike the median values for the known to erode and predicted to erode categories (2.97 and 1.06 t ha^{-1} yr^{-1} respectively), the median values for the sampling grid data set was 0 t ha^{-1} yr^{-1}, based on 552 records.

4. Discussion

4.1. The occurrence of soil erosion in the UK

Collating all readily available and empirically-derived soil erosion data from UK-based studies into a geodatabase has clearly identified that the UK has a rich history of soil erosion research that can be used to describe potential magnitudes of soil loss. The 1566 individual records equates to a density of 1 per 155 km^2, which far exceeds a density of 1 per 3986 km^2 found in the USA (García-Ruiz et al., 2015), for example. Whilst the median soil erosion rate for all land use groups was 0 t ha^{-1} yr^{-1}, soil erosion has occurred widely in the UK and 16% of arable records had soil loss in excess of the suggested 'tolerable'

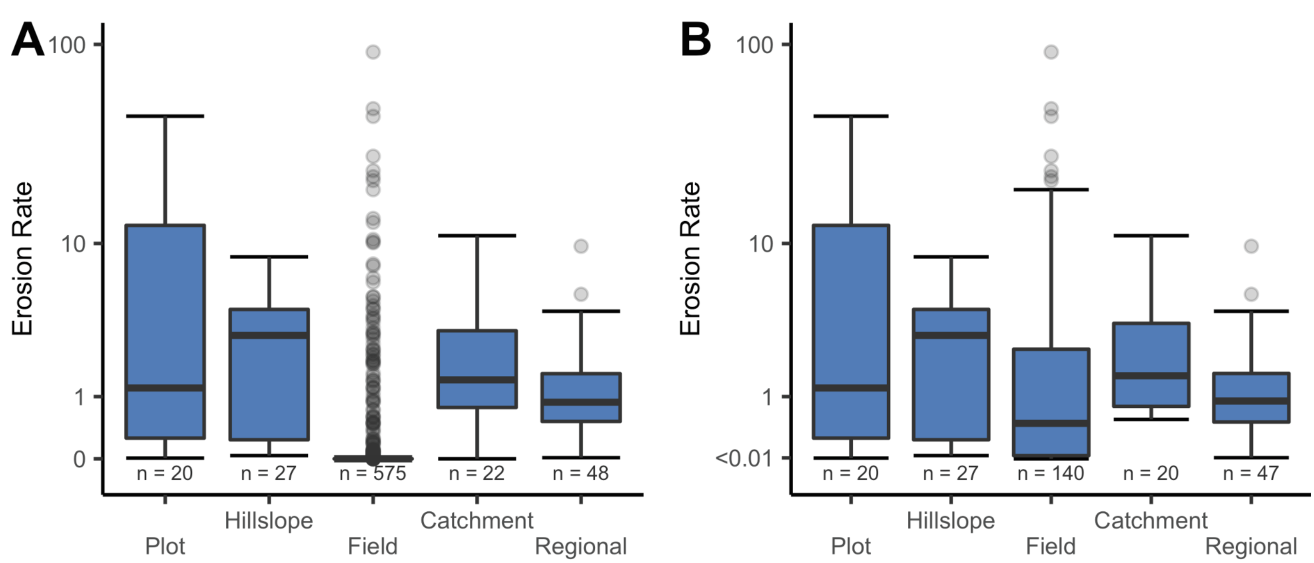

Fig. 8. The relationship between study spatial extent and soil erosion rates (t ha^{-1} yr^{-1}) on a log scale for A) all arable records, and B) arable records with a presence of erosion.

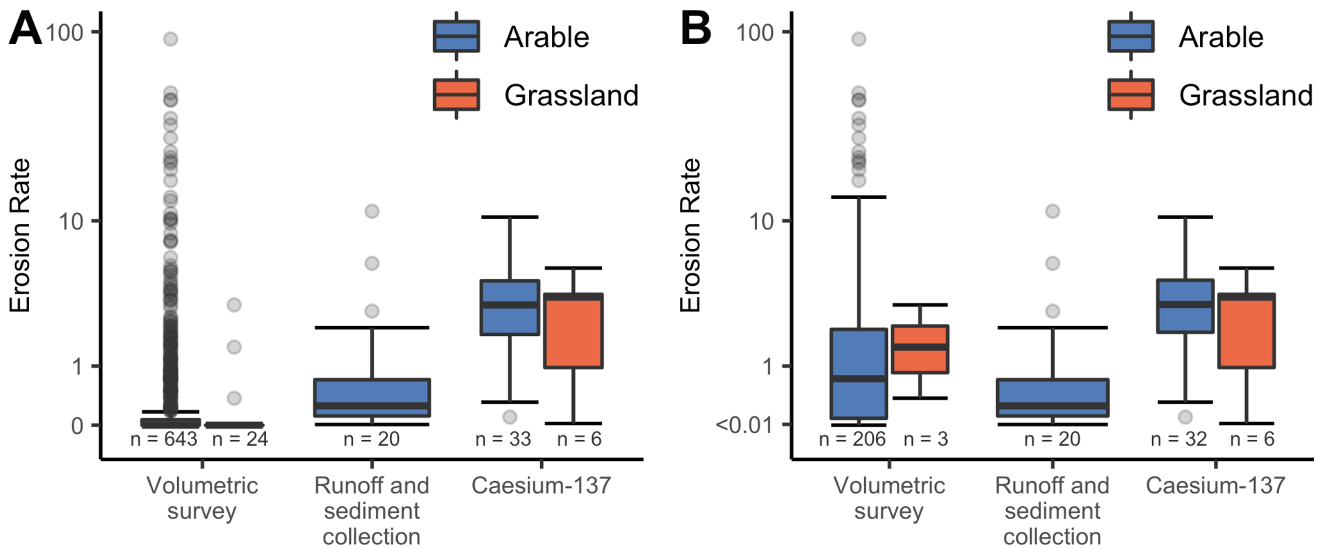

Fig. 9. The relationship between erosion rates (t ha^{-1} yr^{-1}) and the monitoring technique employed for the study, for A) all arable and grassland records, and B) arable and grassland records with a presence of erosion.

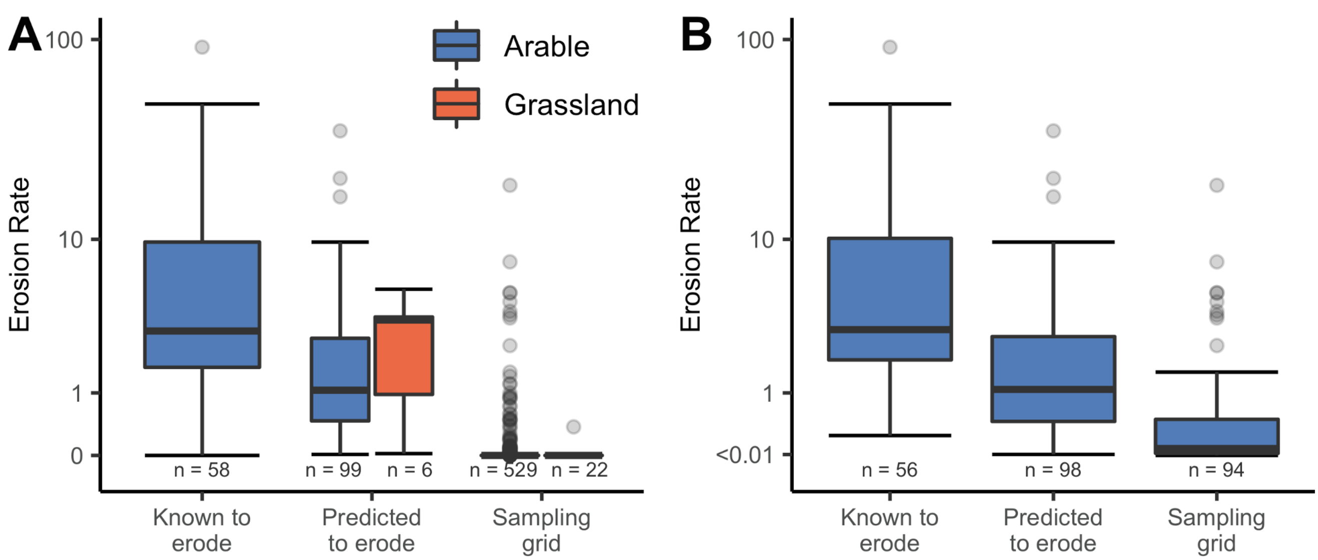

Fig. 10. The relationship between erosion rate (t ha^{-1} yr^{-1}) and the criteria used for selecting the study location, for sites with a presence of erosion, for A) all arable and grassland records, and B) arable and grassland records with a presence of erosion.

1 t ha^{-1} yr^{-1}. These findings are significantly lower than the median values of 5.2 t ha^{-1} yr^{-1} and 1.8 t ha^{-1} yr^{-1}, for arable and grassland respectively, found by a national ^{137}Cs derived survey conducted by Walling (2008), and lower than the range of median values reported by the Soil Survey of England and Wales (SSEW), which vary between 0.3 and 2.2 t ha^{-1} yr^{-1}, for arable land (Evans, 1988). However, this is most probably linked to the difference in monitoring approaches used by the two studies and the difference in soil erosion processes captured by the methods, as discussed further in Section 4.3.

Relative to the mean rates of soil erosion (4.5 to 38.8 t ha^{-1} yr^{-1}) reported across Europe by Verheijen et al. (2009), it could be argued that the UK does not have a severe on-site soil erosion problem. However, there *are* instances where soil erosion has exceeded 'tolerable' rates by one to two orders of magnitude and off-site impacts of soil erosion are substantial in the UK and lowland rivers do not have the capacity to transport the eroded sediment (Graves et al., 2015). Indeed, in ca. 20 instances, soil loss was greater than 10 t ha^{-1} yr^{-1}, with a maximum value of 143 t ha^{-1} yr^{-1}. These observations were typically associated with measurements following extreme rainfall events (e.g. Boardman, 1988; Chambers and Garwood, 2000; Evans and Morgan, 1974) and demonstrate the potential magnitude of soil loss in some areas of the UK. Such *ad hoc* measurements are also indicative of the episodic nature of soil erosion; however, they also have the potential to bias perceptions on the extent of soil erosion issues. This is best illustrated in the cluster of high erosion rates in eastern Scotland visible in Fig. 2, which result from a study over two winters, where only fields with significant erosion features were assessed (Watson and Evans, 1991). Consequently, developing a clear understanding of where the variability in UK soil erosion rates comes from is imperative, particularly given the increased likelihood of large scale rainfall/runoff events due to climate change (Favis-Mortlock and Mullan, 2011; Lee et al., 1999; Mullan, 2013; Nearing et al., 2004; Schaller et al., 2016).

It is important to note that an erosion rate of 1 t ha^{-1} yr^{-1} is most probably not tolerable, or actually sustainable, given the low soil formation rates typical of temperate soils (Evans et al., 2019) and the very shallow soils (< 200 mm) present in many locations (Boardman, 2003). Indeed Evans et al. (2019) report soil formation rates of 0.026–0.096 mm yr^{-1} on two UK soils currently supporting arable agriculture. Based on an assumed bulk density of 1.23 g cm^{-3} (Emmett et al., 2007) this is equivalent to soil formation of as little as 0.32 t ha^{-1} yr^{-1}, which is an order of magnitude lower than 1 t ha^{-1} yr^{-1}. Furthermore, as the records within the database do not present a statistically unbiased national-scale understanding of current and historic rates of soil erosion, it cannot be used to define a UK specific tolerable rate of erosion. As a result, we still do not understand what constitutes an acceptable level of soil erosion for the UK, though in the long-term, it is likely to be lower than the acclerated rates reported herein. It is therefore imperative that future studies are designed to allow the definition of tolerable rates of erosion to be constructed, while building on the present state of knowledge.

4.2. Understanding the impact of environmental conditions on rates of soil erosion

Analysis has revealed that the relationship between soil erosion rates and the environmental controls are variable. This study has identified that soil erosion in the UK is not limited to the 'high' or 'very high' risk soils (per Evans, 1990b), and highlights the need for the management of all soil classes/associations, rather than a risk-based approach. A representative sample of all risk classes is necessary if a true understanding of soil erosion is to be obtained. However, based on the 1 km^2 data, there has been an over representation of soils at moderate risk of soil erosion, relative to the other soil risk classes, to-date. Similarly, while increased slope gradient did have a weak positive relationship with erosion rates, consistent with the findings of García-Ruiz et al. (2015), the results revealed that rates can vary by an order of magnitude in fields of the same gradient. Therefore, it could be argued that there is a need to build an understanding of rates of erosion at higher slope angles in the UK, as the existing data appear to be skewed towards lower angled slopes, which may reflect the favourability of cultivating areas with limited slopes. However, the results demonstrate that even lower slopes can yield high erosion rates irrespective of land use.

Whilst accelerated soil erosion in the UK is frequently linked to significant rainfall events (Boardman, 1988; Boardman et al., 1996; Evans and Morgan, 1974), there was a dearth of information within the data on precipitation totals or intensities. Furthermore, there was little consistency in the reporting of precipitation across the soil erosion datasets collated herein, reducing the ability to compare meaningfully between results. Analysis using annual rainfall totals revealed a weak relationship, which was perhaps not surprising because it is more likely that erosion rates are driven by brief periods of intense rainfall. In the absence of reported data, the 95th percentile of daily rainfall totals and PCI were used to build an understanding of precipitation intensity in the 12 months prior to the studies. However, the results were inconsistent between PCI and the 95th percentile. Indeed a weak negative relationship between PCI and erosion rates was observed, suggesting that PCI alone cannot be used to predict soil erosion.

The weak correlation found between environmental factors and erosion rates is consistent with the difficulty of modelling soil erosion against these observed data (Evans and Boardman, 2016a; Evans and Brazier, 2005). Therefore, the present results provide a strong argument for the improved validation of soil erosion models, and demonstrate the usefulness of collating soil erosion observations into an open access geodatabase, both for this purpose and to guide soil erosion monitoring programs. The data collation exercise has also highlighted the need for standardised reporting of environmental controls within all publications in order to allow *post-hoc* analysis and integration with model predictions of erosion going forward.

4.3. The relationship between erosion observations and monitoring approaches

Whilst building an understanding of soil erosion primarily requires collecting information from an array of environmental and management conditions, the diverse methodologies used to quantify rates of soil erosion can reveal different information about the processes occurring. Historically, due to their replicability, plot scale studies capturing sediment and runoff leaving a bound area were used to build an empirical understanding of soil erosion under different land use and soil types, in an experimental setting (Nearing et al., 1999; Quinton and Catt, 2004). More recently, to minimise the bias created by bound runoff areas and short slope lengths, hillslope studies have been utilised for the same purpose (Deasy et al., 2009). However, both approaches are incredibly resource intensive and, particularly in the instance of bound experimental plots, typically only imitate natural conditions (Boix-Fayos et al., 2006). Furthermore, the spatial extent of empirical soil erosion studies, in particular plot-scale studies, has been cited as creating bias (Parsons et al., 2006). In the UK, there have been numerous efforts to quantify soil erosion *in situ*, with a particular focus on capturing erosion within the defined field, under 'natural' management conditions (Bilotta et al., 2008). For example, studies such as the regional overflight surveys carried out by the Soil Survey of England and Wales (SSEW) or the National Soil Inventory (NSI) locations based study by the Soil Survey and Land Research Centre (SSLRC) (Harrod, 1998), were carried out using this approach, and represent some of the most extensive attempts at quantifying soil erosion, on a national-scale, to date (Boardman, 2002; Evans, 2005, 1988; Harrod, 1998).

The key differences between the monitoring techniques are the erosion process(es) captured and the duration of the observation. For example, the very nature of runoff and sediment collection studies allows erosion observations to be derived for set periods of time or singular events, at the control of the researcher. Similarly, volumetric surveys are usually carried out under natural conditions, and can be used to calculate soil erosion rates for a defined period if starting conditions are known (typically one arable season is defined by the time the soil surface was prepared). However, traditional volumetric surveys are only useful for quantifying soil loss via convergent erosion processes, namely, rills and gullies. In contrast, the values reported by ^{137}Cs tracing may represent an average of all soil redistribution since the primary fallout, which is greater than 30 years for all observations in the database, calculated from concentrations within core samples collected at a single point in time. Consequently, it is argued that ^{137}Cs might be used to quantify all soil erosion processes (Quine and Walling, 1991), and runoff and sediment collection approaches can capture all soil leaving a known area.

If ^{137}Cs does capture all pathways of soil erosion (Chappell and Warren, 2003; Quine and Walling, 1991), it is not unreasonable that the ^{137}Cs results were an order of magnitude greater than other results, particularly in the context of reported rates of soil loss due to tillage erosion between 3 and 70 t ha^{-1} yr^{-1} (Van Oost et al., 2006). It is also important to consider that an annual mean surface lowering of 1 mm over an area of 1 ha, virtually impossible to quantify using volumetric transect surveys, would equate to a soil loss of 13 t in 1 year, based on a mean bulk density of 1.3 g cm^3, far exceeding tolerable rates of erosion. However, as differences resulting from monitoring technique are also evident in the comparison between erosion rates for locations that have been studied with different techniques (Brazier, 2004), these findings highlight the need for caution when drawing conclusions from erosion rates collected using varying methodologies, and the necessity of conducting soil erosion assessments using a unified approach.

Through primarily employing volumetric surveys for quantifying soil loss, the bulk of soil erosion research in the UK has focussed on visible erosion processes or pathways, namely, rilling and gullying. Although there is on-going discussion surrounding the merits of ^{137}Cs fallout for tracing soil redistribution patterns (Mabit et al., 2013; Parsons and Foster, 2013), the significant difference in distribution between the volumetric survey and ^{137}Cs results brings to light the importance of quantifying less-visible erosion processes, such as sheet wash. However, at present there is no clear and quantitative understanding of rates of soil loss derived from sheet wash in the UK, for example, leaving ambiguity and conflicting schools of thought amongst the research community (Evans and Boardman, 2016a,b; Panagos et al., 2016).

Through reporting on a 10 year study in the South Downs, Boardman (2003) illustrated that median erosion rates can vary by an order of magnitude and found that 89% of soil loss occured within 3 of the 10 years monitored. However, with the exception of the ^{137}Cs studies, most erosion rates were collected over periods of less than 4 years. Many of the existing studies are therefore unable to describe seasonal or inter-annual variability and the importance and frequency of extreme erosion events, such as the events described by Chambers and Garwood (2000) and Evans and Morgan (1974), in West Sussex and Cambridgeshire, respectively. Whilst ^{137}Cs has been used to determine long-term patterns and rates of soil erosion, without repeat measures to provide up-to-date inventories (Porto et al., 2014), it cannot be used to quantify contemporary responses to policy measures, such as the impact described by Evans (2010), or changes resulting from climate change. Volumetric surveys, rapidly deployed using the visual estimation approach of Evans (2017) have been suggested as an option by the same author for longitudinal monitoring studies, however, outside of specialist application, this is not a replicable approach and will not capture diffuse erosion pathways such as sheetwash. Novel approaches to volumetric surveys, such as the use of Terrestrial Laser Scanning or Structure-from-Motion Multi-view Stereo, could, however, present a replicable option for quantifying soil erosion rates with quantifiable uncertainties (Castillo et al., 2012; Fawcett et al., 2019; Glendell et al., 2017; Kaiser et al., 2014; Ouédraogo et al., 2014).

With the exception of the work carried out along the NSI sampling grid by the SSLRC (Harrod, 1998; McHugh et al., 2002), it was found that all other soil erosion research in the UK has been carried out in localities with a known propensity for soil erosion. This sampling bias, when compared to the distribution of soil erosion rates found by the sampling grid approach, had a significant impact on the distribution and magnitude of results. Perhaps unsurprisingly, the results highlighted that if erosion is monitored at sites known/predicted to erode, there is a greater likelihood of observing accelerated soil erosion, and hence points to the need to implement statistically unbiased erosion monitoring schemes in the future (Brazier et al., 2016).

Whilst the existing approaches have biased the current understanding of soil erosion in the UK towards localities known to erode, the intersection between these groups of studies illustrates the highly variable nature of soil erosion rates in the UK. Furthermore, the majority of soil erosion observations not only predate current policy, but are biased towards visible erosion pathways and therefore undermine confidence in any subsequent interpretations of changes in erosion rates affected by land use change. Consequently, although these studies do provide an insight into potential magnitudes of erosion in the UK, caution must be taken when drawing conclusions about national-scale erosion rates from the results.

4.4. Towards a national-scale understanding of soil erosion

The current findings suggest there is a need for a refined national-scale assessment of soil erosion in the UK and indeed more widely. Crucially, while previous reviews have argued that existing studies consistently quantified soil erosion in the UK (Evans et al., 2015), the findings herein have illustrated that there is much more complexity in existing understanding. The short-comings highlighted in this study can be overcome by applying the following criteria for future replicable, comparable and robust studies:

1. Unbiased statistical sampling design (as opposed to monitoring erosion where it is known to occur).
2. Including a representative range of environmental conditions from upland to lowland land use and farming practices.
3. Quantifying both visible (i.e convergent) and less-visible (i.e. diffuse) erosion pathways.
4. Capturing the seasonal and inter-annual variability of erosion rates.
5. Representing a selection of land use categories, including emerging land use under changing climates (for example, vineyards and fodder or biofuel crops such as maize/miscanthus).
6. Standardising erosion measurements (per unit space and time i.e. t ha^{-1} yr^{-1}) to ensure that results are comparable nationwide, as the same (and best) techniques are deployed.
7. Consistent reporting of environmental and management conditions (for example, precipitation totals and intensities, soil texture and tillage practices)

Soil erosion is an issue of global importance, and the problems discussed here in a UK-focussed case study are of relevance to many other locations worldwide. Through presenting an open source, open access platform for the collaborative development of a modern erosion observation database, it is anticipated that future research and globally integrated assessments of soil erosion rates can be brought together. Although caution must be employed when comparing rates of erosion across different scales for example (Parsons et al., 2006), using a standardised workflow, sharing data in an open access format and implementing a consistent unit of measure across the soil erosion community, will arguably create a robust platform from which a national-scale understanding of soil erosion can evolve in the UK and globally.

5. Conclusions

The analysis of national-scale geodata describing soil erosion has identified the potential magnitudes of soil erosion in the UK and has also illustrated that accelerated soil erosion can occur wherever agriculture is present, irrespective of slope gradients or expected erosion risk. Whilst the median rate of soil erosion on arable land was 0 t ha^{-1} yr^{-1}, 16% of arable records had soil loss in excess of the suggested 'tolerable' rate of 1 t ha^{-1} yr^{-1}, presenting evidence of unsustainable rates of soil erosion in the UK. Existing monitoring strategies have provided a useful insight into some of the relationships between environmental controls and rates of soil erosion. However, the current research has highlighted the costs associated with an *ad hoc* approach to soil erosion monitoring and the inconsistent reporting of the findings: their potential for describing national-scale erosion rates is limited. Existing monitoring approaches have been resource intensive in nature and/or biased towards a single erosion process, hindering their suitability for future, holistic national-scale monitoring programs. To this end, there is a real need to identify and develop unified monitoring techniques, supported by (and in turn informing) governmental policy, which are capable of meeting the above-listed requirements across changing spatial and temporal scales and different frequencies and magnitudes of erosion, to ensure a sustainable future for the UK's soils

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. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.geoderma.2020.114378 and https://doi.org/10.5281/zenodo.3736496.

References

Amundson, R., Berhe, A.A., Hopmans, J.W., Olson, C., Sztein, A.E., Sparks, D.L., 2015. Soil and human security in the 21st century. Science (80-.) 348.

Bakker, M.M., Govers, G., Kosmas, C., Vanacker, V., van Oost, K., Rounsevell, M., 2005. Soil erosion as a driver of land-use change. Agric. Ecosyst. Environ. 105, 467–481. https://doi.org/10.1016/j.agee.2004.07.009.

Barthel, S., Isendahl, C., Vis, B.N., Drescher, A., Evans, D.L., van Timmeren, A., 2019. Global urbanization and food production in direct competition for land: Leverage places to mitigate impacts on SDG2 and on the Earth System. Anthr. Rev. https://doi.org/10.1177/2053019619856672.

Baston, D., 2020. exactextractr: Fast Extraction from Raster Datasets using Polygons.

Bilotta, G.S., Brazier, R.E., Haygarth, P.M., Macleod, C.J.A., Butler, P., Granger, S., Krueger, T., Freer, J., Quinton, J.N., 2008. Rethinking the contribution of drained and undrained grasslands to sediment-related water quality problems. J. Environ. Qual. 37, 906–914. https://doi.org/10.2134/jeq2007.0457.

Bivand, R., Keitt, T., Rowlingson, B., 2018. rgdal: Bindings for the "Geospatial" Data Abstraction Library.

Boardman, J., 2013. Soil Erosion in Britain: Updating the Record. Agriculture 3, 418–442. https://doi.org/doi:10.3390/agriculture3030418.

Boardman, J., 2006. Soil erosion science: reflections on the limitations of current approaches. Catena 68, 73–86. https://doi.org/10.1016/j.catena.2006.03.007.

Boardman, J., 2003. Soil Erosion and Flooding on the Eastern South Downs, Southern England, 1976–2001. Trans. Inst. Br. Geogr. New Series 28, 176–196. https://doi.org/10.2307/3804444.

Boardman, J., 2002. The Need for Soil Conservation in Britain: Revisited. Area 34, 419–427. https://doi.org/10.2307/20004273.

Boardman, J., 1995. Damage to property by runoff from agricultural land, South Downs, Southern England, 1976–93. Geogr. J. 161, 177–191. https://doi.org/10.2307/3059974.

Boardman, J., 1988. Severe erosion on agricultural land in east Sussex, UK October 1987. Soil Technol. 1, 333–348. https://doi.org/10.1016/0933-3630(88)90013-X.

Boardman, J., Burt, T.P., Evans, R., Slattery, M.C., Shuttleworth, H., 1996. Soil erosion and flooding as a result of a summer thunderstorm in Oxfordshire and Berkshire, May 1993. Appl. Geogr. 16, 21–34. https://doi.org/10.1016/0143-6228(95)00023-2.

Boardman, J., Evans, R., Favis-Mortlock, D.T., Harris, T.M., 1990. Climate change and soil erosion on agricultural land in england and wales. L. Degrad. Dev. 2, 95–106. https://doi.org/10.1002/ldr.3400020204.

Boix-Fayos, C., Martínez-Mena, M., Arnau-Rosalén, E., Calvo-Cases, A., Castillo, V., Albaladejo, J., 2006. Measuring soil erosion by field plots: understanding the sources of variation. Earth-Sci. Rev. 78, 267–285. https://doi.org/10.1016/j.earscirev.2006.05.005.

Brazier, R.E., 2004. Quantifying soil erosion by water in the UK: a review of monitoring and modelling approaches. Prog. Phys. Geogr. 28, 340–365. https://doi.org/10.1191/0309133304 pp415ra.

Brazier, R.E., Anderson, K., Benaud, P., Evans, M., Farrow, L., Glendell, M., James, M.R., Lark, M., Quine, T.A., Quinton, J.N., Rawlins, B., Rickson, R.J., 2016. Developing a cost-effective framework to monitor soil erosion in England and Wales. Final report to Defra for project SP1311.

Brazier, R.E., Anderson, K., Quine, T.A., Quinton, J.N., Evans, M., Rickson, R.J., Bellamy, P.H., Rawlins, B., Ellis, M., 2011. Developing a cost-effective framework for monitoring soil erosion in England and Wales. Final report to Defra for project SP1303.

Brazier, R.E., Bilotta, G.S., Haygarth, P.M., 2007. A perspective on the role of lowland, agricultural grasslands in contributing to erosion and water quality problems in the UK. Earth Surf. Process. Landforms 32, 964–967. https://doi.org/10.1002/esp.1484.

Castillo, C., Pérez, R., James, M.R., Quinton, J.N., Taguas, E.V., Gómez, J.A., 2012. Comparing the accuracy of several field methods for measuring gully erosion. Soil Sci. Soc. Am. J. 76, 1319. https://doi.org/10.2136/sssaj2011.0390.

Chambers, B.J., Garwood, T.W.D., 2000. Monitoring of water erosion on arable farms in England and Wales. 1990–94. Soil Use Manage. 16, 93–99. https://doi.org/10.1111/j.1475-2743.2000.tb00181.x.

Chang, W., Cheng, J., Allaire, J., Xie, Y., McPherson, J., 2018. shiny: Web Application Framework for R.

Chappell, A., Warren, A., 2003. Spatial scales of 137Cs-derived soil flux by wind in a 25 km2 arable area of eastern England. Catena 52, 209–234. https://doi.org/10.1016/S0341-8162(03)00015-8.

Cheng, J., Karambelkar, B., Xie, Y., 2018. leaflet: Create Interactive Web Maps with the JavaScript "Leaflet" Library.

Deasy, C., Quinton, J.N., Silgram, M., Bailey, A.P., Jackson, B., Stevens, C.J., 2009. mitigation options for sediment and phosphorus loss from winter-sown arable crops. J. Environ. Qual. 38, 2121–2130. https://doi.org/10.2134/jeq2009.0028.

Dinno, A., 2017. dunn.test: Dunn's Test of Multiple Comparisons Using Rank Sums.

Emmett, B.A., Reynold, B., Chamberlain, P.M., Rowe, E., Spurgeon, D., Brittain, S.A., Frogbrook, Z., Hughes, S., Lawlor, A.J., Poskitt, J., Potter, E., Robinson, D.A., Scott, A., Wood, C., Woods, C., 2007. Countryside Survey: Soils Report from 2007. Wallingford, Oxfords.

Evans, D.L., Quinton, J.N., Tye, A.M., Rodés, Á., Davies, J.A.C., Mudd, S.M., Quine, T.A., 2019. Arable soil formation and erosion: a hillslope-based cosmogenic nuclide study in the United Kingdom. Soil 5, 253–263. https://doi.org/10.5194/soil-5-253-2019.

Evans, R., 2017. Factors controlling soil erosion and runoff and their impacts in the upper Wissey catchment, Norfolk, England: a ten year monitoring programme. Earth Surf. Process. Landforms 42, 2266–2279. https://doi.org/10.1002/esp.4182.

Evans, R., 2010. Runoff and soil erosion in arable Britain: changes in perception and policy since 1945. Environ. Sci. Policy 13, 141–149. https://doi.org/10.1016/j.envsci.2010.01.001.

Evans, R., 2005. Monitoring water erosion in lowland England and Wales—a personal view of its history and outcomes. Catena 64, 142–161. https://doi.org/10.1016/j.catena.2005.08.003.

Evans, R., 1995. Some methods of directly assessing water erosion of cultivated land-a comparison of measurements made on plots and in fields. Prog. Phys. Geogr. 19, 115–129.

Evans, R., 1990a. Water erosion in British farmers' fields-some causes, impacts, predictions. Prog. Phys. Geogr. 14, 199–219.

Evans, R., 1990b. Soils at risk of accelerated erosion in England and Wales. Soil Use Manage. 6, 125–131. https://doi.org/10.1111/j.1475-2743.1990.tb00821.x.

Evans, R., 1988. Water Erosion in England and Wales 1982–1984.

Evans, R., 1971. The need for soil conservation. Area 3, 20–23. https://doi.org/10.2307/20000505.

Evans, R., Boardman, J., 2016a. The new assessment of soil loss by water erosion in Europe. Panagos P. et al., 2015 Environmental Science & Policy 54, 438–447—a response. Environ. Sci. Policy 58, 11–15. https://doi.org/10.1016/j.envsci.2015.12.013.

Evans, R., Boardman, J., 2016b. A reply to panagos et al., 2016 (Environmental science & policy 59 (2016) 53–57, Environmental Science & Policy. https://doi.org/10.1016/j.envsci.2016.03.004.

Evans, R., Brazier, R.E., 2005. Evaluation of modelled spatially distributed predictions of soil erosion by water versus field-based assessments. Environ. Sci. Policy 8, 493–501. https://doi.org/10.1016/j.envsci.2005.04.009.

Evans, R., Collins, A.L., Foster, I.D.L., Rickson, R.J., Anthony, S.G., Brewer, T., Deeks, L.K., Newell-Price, J.P., Truckell, I.G., Zhang, Y., 2015. Extent, frequency and rate of water erosion of arable land in Britain – benefits and challenges for modelling. Soil Use Manage. https://doi.org/10.1111/sum.12210.

Evans, R., Collins, A.L., Zhang, Y., Foster, I.D.L., Boardman, J., Sint, H., Lee, M.R.F., Griffith, B.A., 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.

Evans, R., Morgan, R.P.C., 1974. Water Erosion of Arable Land. Area 6, 221–225.

Favis-Mortlock, D., Mullan, D., 2011. Soil erosion by water under future climate change, in: Soil Hydrology, Land Use and Agriculture: Measurement and Modelling. pp. 384–414.

Fawcett, D., Blanco-Sacristán, J., Benaud, P., 2019. Two decades of digital photogrammetry: Revisiting Chandler's 1999 paper on "Effective application of automated digital photogrammetry for geomorphological research" – a synthesis. Prog. Phys. Geogr. 43, 299–312. https://doi.org/10.1177/0309133319832863.

Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N., Snyder, P.K., 2005. Global consequences of land use. Science (80-) 309, 570–574. https://doi.org/10.1126/science.1111772.

García-Ruiz, J.M., Beguería, S., Nadal-Romero, E., González-Hidalgo, J.C., Lana-Renault, N., Sanjuán, Y., 2015. A meta-analysis of soil erosion rates across the world. Geomorphology 239, 160–173. https://doi.org/10.1016/j.geomorph.2015.03.008.

Gerland, P., Raftery, A.E., Ev Ikova, H., Li, N., Gu, D., Spoorenberg, T., Alkema, L., Fosdick, B.K., Chunn, J., Lalic, N., Bay, G., Buettner, T., Heilig, G.K., Wilmoth, J., 2014. World population stabilization unlikely this century. Science (80-) 346, 234–237. https://doi.org/10.1126/science.1257469.

Glendell, M., McShane, G., Farrow, L., James, M.R., Quinton, J., Anderson, K., Evans, M., Benaud, P., Rawlins, B., Morgan, D., Jones, L., Kirkham, M., DeBell, L., Quine, T.A., Lark, M., Rickson, J., Brazier, R.E., 2017. Testing the utility of structure-from-motion photogrammetry reconstructions using small unmanned aerial vehicles and ground photography to estimate the extent of upland soil erosion. Earth Surf. Process. Landforms 42, 1860–1871. https://doi.org/10.1002/esp.4142.

Grand-Clement, E., Luscombe, D.J., Anderson, K., Gatis, N., Benaud, P., Brazier, R.E., 2014. Antecedent conditions control carbon loss and downstream water quality from shallow, damaged peatlands. Sci. Total Environ. 493, 961–973. https://doi.org/10.1016/j.scitotenv.2014.06.091.

Graves, A.R., Morris, J., Deeks, L.K., Rickson, R.J., Kibblewhite, M.G., Harris, J.A., Farewell, T.S., Truckle, I., 2015. The total costs of soil degradation in England and Wales. Ecol. Econ. 119, 399–413. https://doi.org/10.1016/j.ecolecon.2015.07.026.

Grolemund, G., Wickham, H., 2011. Dates and times made easy with lubridate. J. Statiscial Softw. 40, 1–25.

Harrod, T., 1998. A systematic approach to national budgets of phosphorous loss through soil erosion and surface runoff and National Soil Inventory nodes. Final report to MAFF for project NT1014 (SSLRC project code: JF3181). North Wyke.

Hijmans, R.J., 2018. raster: Geographic Data Analysis and Modeling.

Holman, I.P., Hollis, J.M., Thompson, T.R.E., 2000. Impact of agricultural soil conditions on floods – autumn 2000. R&D Technical Report W5B-026/TR. Rio House Aztec West Almonds- bury Bristol BS32 4UD, UK.

Humphries, R.N., Brazier, R.E., 2018. Exploring the case for a national-scale soil conservation and soil condition framework for evaluating and reporting on environmental and land use policies. Soil Use Manage. 34, 134–146. https://doi.org/10.1111/sum.12400.

Kaiser, A., Neugirg, F., Rock, G., Müller, C., Haas, F., Ries, J., Schmidt, J., 2014. Small-

scale surface reconstruction and volume calculation of soil erosion in complex moroccan gully morphology using structure from motion. Remote Sens. 6, 7050–7080. https://doi.org/10.3390/rs6087050.
Lal, R., 1998. Soil erosion impact on agronomic productivity and environment quality. CRC. Crit. Rev. Plant Sci. 17, 319–464. https://doi.org/10.1080/07352689891304249.
Lee, J.J., Phillips, D.L., Benson, V.W., 1999. Soil erosion and climate change: assessing potential impacts and adaptation practices. J. Soil Water Conserv.
Mabit, L., Meusburger, K., Fulajtar, E., Alewell, C., 2013. The usefulness of 137Cs as a tracer for soil erosion assessment: a critical reply to Parsons and Foster (2011). Earth-Science Rev. 127, 300–307. https://doi.org/10.1016/j.earscirev.2013.05.008.
Martin-Vide, J., 2004. Spatial distribution of a daily precipitation concentration index in peninsular Spain. Int. J. Climatol. 24, 959–971. https://doi.org/10.1002/joc.1030.
McHugh, M., Harrod, T., Morgan, R.P.C., 2002. The extent of soil erosion in upland England and Wales. Earth Surf. Process. Landforms 27, 99–107. https://doi.org/10.1002/esp.308.
Microsoft Corporation, Weston, S., 2019. doParallel: Foreach Parallel Adaptor for the "parallel" Package.
Montgomery, D.R., 2007. Soil erosion and agricultural sustainability. Proc. Natl. Acad. Sci. U. S. A. 104, 13268–13272. https://doi.org/10.1073/pnas.0611508104.
Morgan, R.P.C., 1985. Soil erosion measurement and soil conservation research in cultivated areas of the UK. Geogr. J. 151, 11–20. https://doi.org/10.2307/633274.
Mullan, D., 2013. Soil erosion under the impacts of future climate change: assessing the statistical significance of future changes and the potential on-site and off-site problems. Catena 109, 234–246. https://doi.org/10.1016/J.CATENA.2013.03.007.
Nearing, M.A., Govers, G., Norton, L.D., 1999. Variability in soil erosion data from replicated plots. Soil Sci. Soc. Am. J. 63, 1829–1835. https://doi.org/10.2136/sssaj1999.6361829x.
Nearing, M.A., Pruski, F.F., O'Neal, M.R., 2004. Expected climate change impacts on soil erosion rates: a review. J. Soil Water Conserv.
Ouédraogo, M.M., Degré, A., Debouche, C., Lisein, J., 2014. The evaluation of unmanned aerial systems-based photogrammetry and terrestrial laser scanning to generate DEMs of agricultural watersheds. Geomorphology. https://doi.org/10.1016/j.geomorph.2014.02.016.
Panagos, P., Borrelli, P., Poesen, J., Meusburger, K., Ballabio, C., Lugato, E., Montanarella, L., Alewell, C., 2016. Reply to "The new assessment of soil loss by water erosion in Europe. Panagos P. et al., 2015 Environ. Sci. Policy 54, 438–447-A response" by Evans and Boardman [Environ. Sci. Policy 58, 11–15]. Environ. Sci. Policy 59, 53–57. https://doi.org/10.1016/j.envsci.2016.02.010.
Parsons, A.J., Brazier, R.E., Wainwright, J., Powell, D.M., 2006. Scale relationships in hillslope runoff and erosion. Earth Surf. Process. Landforms 31, 1384–1393. https://doi.org/10.1002/esp.1345.
Parsons, A.J., Foster, I.D.L., 2013. The assumptions of science. Earth-Sci. Rev. 127, 308–310. https://doi.org/10.1016/j.earscirev.2013.05.011.
Pimentel, D., 2006. Soil erosion: a food and environmental threat. Environ. Dev. Sustain. 8, 119–137. https://doi.org/10.1007/s10668-005-1262-8.
Pimentel, D., Harvey, C., Resosudarmo, P., Sinclair, K., Kurz, D., McNair, M., Crist, S., Shpritz, L., Fitton, L., Saffouri, R., Blair, R., 1995. Environmental and economic costs of soil erosion and conservation benefits. Science (80-) 267, 1117–1123. https://doi.org/10.1126/science.267.5201.1117.
Porto, P., Walling, D.E., Capra, A., 2014. Using 137Cs and 210Pbe measurements and conventional surveys to investigate the relative contributions of interrill/rill and gully erosion to soil loss from a small cultivated catchment in Sicily. Soil Tillage Res. 135, 18–27. https://doi.org/10.1016/j.still.2013.08.013.
Povoa, L. V, Nery, J.T., 2016. Precintcon: Precipitation Intensity, Concentration and Anomaly Analysis. R Package Version 2.3. 0. Comprehensive R Archive Network, Austria.
Puma, M.J., Bose, S., Chon, S.Y., Cook, B.I., 2015. Assessing the evolving fragility of the global food system. Environ. Res. Lett. 10, 024007. https://doi.org/10.1088/1748-9326/10/2/024007.
Quine, T.A., Walling, D.E., 1991. Rates of soil erosion on arable fields in Britain: quantitative data from caesium-137 measurements. Soil Use Manage. 7, 169–176. https://doi.org/10.1111/j.1475-2743.1991.tb00870.x.
Quinton, J.N., Catt, J.A., 2004. The effects of minimal tillage and contour cultivation on surface runoff, soil loss and crop yield in the long-term Woburn Erosion Reference Experiment on sandy soil at Woburn, England. Soil Use Manage. 20, 343–349. https://doi.org/10.1111/j.1475-2743.2004.tb00379.x.
R Core Team, 2016. R: A language and environment for statistical computing.
Reed, A.H., 1979. Accelerated erosion of arable soils in the United Kingdom by rainfall and run-off. Outlook Agric. 10, 41–48. https://doi.org/10.1177/003072707901000107.
Rickson, R.J., 2014. Can control of soil erosion mitigate water pollution by sediments? Sci. Total Environ. 468–469, 1187–1197 https://doi.org/http://dx.doi.org/10.1016/j.scitotenv.2013.05.057.
Rocha, J.C., Peterson, G., Bodin, Ö., Levin, S., 2018. Cascading regime shifts within and across scales. Science (80-) 362, 1379–1383. https://doi.org/10.1126/science.aat7850.
RStudio Team, 2015. RStudio: Integrated Development for R.
Schaller, N., Kay, A.L., Lamb, R., Massey, N.R., van Oldenborgh, G.J., Otto, F.E.L., Sparrow, S.N., Vautard, R., Yiou, P., Ashpole, I., Bowery, A., Crooks, S.M., Haustein, K., Huntingford, C., Ingram, W.J., Jones, R.G., Legg, T., Miller, J., Skeggs, J., Wallom, D., Weisheimer, A., Wilson, S., Stott, P.A., Allen, M.R., 2016. Human influence on climate in the 2014 southern England winter floods and their impacts. Nat. Clim. Change. 6, 627–634.
Seto, K.C., Fragkias, M., Güneralp, B., Reilly, M.K., Pidgeon, A., 2011. A meta-analysis of global urban land expansion. PLoS One 6, e23777. https://doi.org/10.1371/journal.pone.0023777.
Tanguy, M., Dixon, H., Prosdocimi, I., Morris, D.G., Keller, V.D.J., 2016. Gridded estimates of daily and monthly areal rainfall for the United Kingdom (1890-2015) [CEH-GEAR].
Tilman, D., Cassman, K.G., Matson, P.A., Naylor, R., Polasky, S., 2002. Agricultural sustainability and intensive production practices. Nature 418, 671–677. https://doi.org/10.1038/nature01014.
Van Oost, K., Govers, G., de Alba, S., Quine, T.A., 2006. Tillage erosion: a review of controlling factors and implications for soil quality. Prog. Phys. Geogr. 30, 443–466. https://doi.org/10.1191/0309133306 pp487ra.
Vandaele, K., Poesen, J., 1995. Spatial and temporal patterns of soil erosion rates in an agricultural catchment, central Belgium. Catena 25, 213–226. https://doi.org/10.1016/0341-8162(95)00011-G.
Verheijen, F.G.A., Jones, R.J.A., Rickson, R.J., Smith, C.J., 2009. Tolerable versus actual soil erosion rates in Europe. Earth-Sci. Rev. 94, 23–38. https://doi.org/10.1016/j.earscirev.2009.02.003.
Walling, D.E., 2008. Documenting soil erosion rates on agricultural land in England and Wales: Phase 2. Final report to Defra for project SP0413.
Walling, D.E., Russell, M.A., Hodgkinson, R.A., Zhang, Y., 2002. Establishing sediment budgets for two small lowland agricultural catchments in the UK. Catena 47, 323–353. https://doi.org/10.1016/S0341-8162(01)00187-4.
Watson, A., Evans, R., 1991. A comparison of estimates of soil erosion made in the field and from photographs. Soil Tillage Res. 19, 17–27. https://doi.org/10.1016/0167-1987(91)90106-8.
Wickham, H., 2016a. ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag, New York, USA.
Wickham, H., 2016b. scales: Scale Functions for Visualization.
Wickham, H., 2016c. stringr: Simple, Consistent Wrappers for Common String Operations.
Wickham, H., Averick, M., Bryan, J., Chang, W., McGowan, L., François, R., Grolemund, G., Hayes, A., Henry, L., Hester, J., 2019. Welcome to the Tidyverse. J. Open Source Softw. 4, 1686.
Wilke, C.O., 2017. cowplot: Streamlined Plot Theme and Plot Annotations for "ggplot2'.