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Invited review

A comparison of conventional and ^{137}Cs-based estimates of soil erosion rates on arable and grassland across lowland England and Wales

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ABSTRACT

Soils deliver a range of ecosystem services and underpin conventional global food production which must increase to feed the projected growth in human population. Although soil erosion by water and subsequent sediment delivery to rivers are natural processes, anthropogenic pressures, including modern farming practices and management, have accelerated soil erosion rates on both arable and grassland. A range of approaches can be used to assess and document soil erosion rates and, in the case of the UK, these mainly comprise the ^{137}Cs-based approach, conventional surveys using volumetric measurements, integration of information on suspended sediment flux, fine sediment source apportionment and landscape sediment retention and traditional bounded hydrological monitoring at edge-of-field using experimental platforms. We compare the erosion rates for arable and grassland in lowland England assessed by these different techniques. Rates assessed by volumetric measurements are similar to those generated by integrating information on suspended sediment flux, sources and landscape retention, but are much less than those estimated by the ^{137}Cs-based approach; of the order of one magnitude less for arable land. The ^{137}Cs approach assumes an initial distribution of ^{137}Cs uniformly spread across the landscape and relates the sampled distribution to erosion, but other (transport) processes are also involved and their representation in the calibration procedures remains problematic. We suggest that the ^{137}Cs technique needs to be validated more rigorously and conversion models re-calibrated. As things stand, rates of erosion based on the distribution of ^{137}Cs may well overstate the severity of the problem in lowland Britain and, therefore, are not a reliable indicator of water erosion rates.

1. Introduction

Soils are essential for delivering a range of ecosystem services including food production, carbon storage, water filtration, climate regulation and biodiversity (Lal, 2013). Reliable estimates of soil erosion are needed not only to assess impacts and costs in terms of the degradation of ecosystem functioning and services but also to assess the loss of what is best considered a non-renewable resource that provides most of the world's food. According to Pimental and Burgess (2013, pp. 443) - '…humans worldwide obtain more than 99.7 % of their food (calories) from the land and less than 0.3 % from the oceans and aquatic ecosystems, preserving cropland and maintaining soil fertility should be of the highest importance to human welfare'. The central thread of the arguments presented by Pimental and Burgess (2013), in part supported by the rapidly growing global human population (UN, 2011), is that the maintenance and augmentation of world food supply basically depends on the productivity and quality of all agricultural soils. Loss of soil is therefore unacceptable and every effort should be made to prevent this invaluable resource escaping from within the field boundary.

Against this global context, there has been concern about soil erosion in the UK, especially on arable land, for at least 40 years (Evans, 1971; Boardman, 2013a) with particular concern expressed over water, rather than, wind erosion (Evans, 1996, p31; Boardman, 2013a). For arable land, the exposure of bare worked soil in conjunction with the annual calendar of seeding and subsequent harvest poses the most significant risk of soil erosion as a result of rainfall and runoff. But,

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increasingly, the risk of erosion on grassland resulting from high stocking densities in lowland areas, grazing practices including the growing popularity of outdoor wintering of stock and associated impacts including soil compaction, poaching and pugging which promote runoff and the exposure of surface soil to mobilisation by water, has been highlighted by the erosion community (Foster and Walling, 1994; Collins et al., 1997; Evans, 1998a; Bilotta et al., 2007). In this review, we compare different methods of estimating soil erosion rates by water in lowland England and Wales. Early attempts to assess water erosion were to discern if it was a problem – where did it occur, how often and at what rate? Monitoring of cultivated fields and taking simple volumetric measurements of erosion channels showed that erosion (Evans, 1996, 2005) was a problem in some locations (e.g. sandy soil associations where a wide range of crops was grown) and a better understanding gained of erosion, its causes (physical, e.g. soil, land management, and economic factors, e.g. subsidies), and its impacts (e.g. muddy floods). Based on soil, land use and relief factors, erosion risk was assessed and predicted (Evans, 1990a). In other words, based on the findings of the water erosion monitoring scheme of 1982–86 (Evans, 1988) risk of erosion was assigned to one of five categories (Evans, 1990a) for each lowland soil association portrayed on the National Soil Map of England and Wales. Risk was defined by percent area of land covered by eroded fields which, in turn, was dependant on the dominant soil series within the association, relief as portrayed on topographic maps and land use as described in the National Map legend. It is noteworthy that soil was not treated as an individual factor as it is, for example, in erosion models, but as part of an integrated whole based on mapped soil associations with common soil, land use and relief characteristics. Later, there was a suggestion that the distribution across landscapes (including both arable and grass fields) of ^{137}Cs inventories could indicate where erosion might have occurred and with conversion and modelling of the reference and erosion site inventories could indicate rates of erosion (Walling and Quine, 1990). In this context, we therefore compare methods used for estimating rates of soil erosion in lowland England and Wales, their drawbacks and the rates of erosion they generate. It is only in Britain that data has been collected by survey-based approaches in sufficient quantity to permit strategic comparison with the estimates generated using other techniques.

Initial work in England and Wales during the 1970s focused on on-site issues including the impacts of soil erosion on crop yields (Evans, 2005), but during the 1980s and 1990s and ever since, there has been a widening scope of interest towards off-site issues such as sedimentation of river channels or reservoirs (Collins and Walling, 2007; Butcher et al., 1993), muddy floods and associated property damage (Boardman, 1988a, 2001, 2003, 2010; Boardman et al., 1996, 2006, 2009; Evans, 1996; Evans and Boardman, 1994, 2003), drinking water treatability (Environment Agency, 2007; OFWAT, 2011) as well as the detrimental impacts of excess soil erosion and fine-grained sediment delivery on aquatic ecology (Wood and Armitage, 1997; Greig et al., 2005; Yarnell et al., 2006; Collins and Anthony, 2008a, b; Collins et al., 2009, 2011; Kemp et al., 2011; Jones et al., 2012, 2014). Such on-site and off-site problems reflect the increasing sensitivity of agricultural land to soil erosion in the UK, which has been driven by a range of factors and practices including, among others, arable and root/fodder cropping, declining organic matter content, soil compaction, hedgerow and boundary removal, de-stoning and changing weather patterns (Boardman, 2015).

A range of methods can be used to assess and monitor, or to estimate soil erosion, including experimental plots (Fullen and Read, 1986; Nearing et al., 1999), field measurements of erosional forms (Evans and Boardman, 1994; Herweg, 1996; Stocking and Murnaghan, 2001; Prasuhn, 2011, 2012) tracer techniques (Quine and Walling, 1991; Walling and Quine, 1990, 1991), historical documents, river suspended sediment yields (Walling and Webb, 1987; Evans, 2006), lake and reservoir sedimentation (Butcher et al., 1993; Rowan et al., 1995; Foster, 2006, 2010; Foster et al., 2011), aerial photography (Evans, 1988) and modelling (Evans and Brazier, 2005). Models to estimate runoff and soil erosion rates range in complexity from empirical erosion calculators based on simple rules governing soil loss (such as the Universal Soil Loss Equation [USLE] or Revised Universal Soil Loss Equation [RUSLE]) (Lane et al., 1988; Prasuhn et al., 2007) to deterministic frameworks. The latter include various well-known models including SWAT (Ghaffari et al., 2009), TOPMODEL (Beven, 1997, WEPP (Pandey et al., 2009), CHILD (Tucker and Bras, 1998), CAESAR (Coulthard et al., 2007), PSYCHIC (Davison et al., 2008; Stromqvist et al., 2008), PESERA (Licciardello et al., 2009) and, MORGAN-MORGAN-FINNEY (Morgan and Duzant, 2008). Estimates of erosion rates made using these different techniques cover a wide range of values ($\mathrm{t\,ha^{-1}\,yr^{-1}}$), of the order of magnitudes.

Not all methods to monitor, assess and estimate erosion are appropriate at all scales. Modelled rates of water erosion based on plots cannot be extrapolated linearly across the landscape (Evans and Boardman, 2016a, b; Evans et al., 2016). ^{137}Cs can be used to make estimates at a field scale and by using Geographical Information Systems and statistical techniques, rates can be extrapolated across landscapes and nations (Chappell et al., 2011; Walling and Zhang, 2010). Survey-based assessments can be made for within-field, field and soil association (Evans, 2002). Assessments based on sediment loads can be applied at multiple scales but are mostly, but not exclusively, used at catchment scale. With respect to the latter, there is a need to apportion the sources of sediment since sediment loads integrate soil loss from multiple sources and the number of sources frequently increases with scale.

Despite the range of approaches available, estimating, or modelling erosion rates, has proved problematic for many years. Monitoring or modelling studies have been undertaken at a variety of scales (e.g. small/large hillslope plot, field scale, grid square, small to large catchment) all of which will produce differences in estimated or calculated erosion rates. One key factor here concerns the area over which erosion rates are measured and averaged and the general tendency, in many environments, for net erosion rates per unit area to decrease with an increase in catchment size due to greater opportunities for sediment storage (i.e. a decreasing sediment delivery ratio with increased catchment area). Another control concerns the likelihood of different factors influencing rates of soil erosion at different scales Similarly, estimates of sediment yield derived from river monitoring or lake sediment-based reconstruction will underestimate hillslope erosion rates largely because not all eroded soil is delivered to rivers and streams (e.g. Walling et al., 2002, 2006; Parsons et al., 2004; Walling and Collins, 2008; Foster, 2010; Parsons, 2012). The percentage of soil delivered from any one part of a catchment to a river or stream is not simply a function of travel distance (Parsons et al., 2006). Landscape connectivity via surface (e.g. Boardman, 2013b; Collins et al., 2013; Rickson, 2014) or sub-surface pathways (e.g. Chapman et al., 2003, 2005; Foster et al., 2003; Zhang et al., 2016) may provide a high propensity for sediment delivery whereas some fields in close proximity to a river may be poorly connected to it in terms of sediment and water delivery (Collins and Walling, 2004). Assessments based on the USLE, or RUSLE, as for example proposed for Europe (Panagos et al., 2015, 2016), when compared with estimates of rates of erosion from traditional survey-based methods in Britain, predict erosion rates that are far too high (Evans and Boardman, 2016a, b). It seems likely, from our experience of field evaluation of erosion that this criticism holds for other assessments made using USLE or other plot-based models.

There are alternative ways of estimating erosion rates at the field scale using either surveys of rill and gully volumes (a method favoured, for example, by Boardman et al., 2009; Evans, 1990b, 2002, 2005) or by using fallout radionuclides (FRN; see Walling and Quine, 1991, 1993, Zapata, 2002; IAEA, 2014). While initially developed using ^{137}Cs, conversion models have also been developed for other FRNs such as unsupported Pb-210 (Mabit et al., 2014) and Be-7 (Walling et al., 1999; Schuller et al., 2010). The conceptualisation of the ^{137}Cs technique is elegant and often persuasive and a large international literature exists in relation to this method of erosion estimation (see debate by Parsons and Foster, 2011, 2013; Mabit et al., 2013). It is also possible to combine estimates of suspended sediment yield at catchment outlets with sediment source apportionment information to estimate net soil loss to catchment outlets (e.g. Collins et al., 2010). In turn, those estimates of net export from a landscape can be processed further by taking account of the most important component of long-term landscape sediment storage associated with floodplain deposition, since that storage term plus the source-weighted yield at the monitored outlet provides a more robust estimate of the total amount of soil released to the channel network from any given land use or source. Sediment stored other than on floodplains is not accounted for in such an estimate, but empirical evidence suggests that floodplains represent the most significant long-term sediment stores in lowland England and Wales (Walling et al., 1998).

Given that the traditional field and FRN conversion methods often give rather different results (see Brazier, 2004 for example) we re-visit this problem and compare rates of erosion estimated for lowland England and Wales using ^{137}Cs-based conversion models with those obtained for arable land from survey-based measurements/estimates of volumes eroded and deposited in farmers' fields in selected localities. For grassland, extra data were assembled using either sediment source fingerprinting combined with estimates of suspended sediment yield and long-term landscape retention factors, or hydrologically-isolated field scale experimental platforms. This comparison of soil erosion estimates generated using different approaches is timely for a number of reasons. There has been widespread recognition of the need for a nationwide soil erosion policy in England and Wales (Boardman, 1988b). Due to the unsustainable use of soils compromising soil quality, biodiversity and water quality, the European Commission adopted a Soil Thematic Strategy in September 2006 with the specific objective of soil protection. More recently, a draft Soil Framework Directive was tabled as a means of implementing a legal requirement for policies to tackle soil degradation across Europe although this was withdrawn in May 2014 as part of a bureaucracy reduction programme. Nonetheless, the EU 7th Environment Action Programme adopted in November 2013 requested that Member States commit to a ‘land degradation neutral world’. Furthermore, and despite the decision of the UK to instigate its departure from the European Community in 2017, the 2011 UK government Natural Environment White Paper (The Natural Choice (Defra, 2011)) expressed the goal of managing soils sustainably to tackle soil degradation successfully by 2030. In response, a new pilot national monitoring programme was recently trialled in England and Wales to compare techniques and the data generated. At the same time, another study, funding the work reported in this paper, was commissioned to investigate the level of soil erosion protection required at landscape level to meet specific reduction targets and part of that programme also involved a comparison of soil erosion data generated by different methods.

2. Sources of information

2.1. The datasets used to compare soil erosion estimates for agricultural land across England and Wales

The datasets used to compare the estimates of soil erosion in lowland England and Wales are listed in Table 1. Methods used to assemble these datasets inevitably involved contrasting procedures and assumptions. Key limitations and uncertainties for each dataset are described in the Discussion section.

2.2. Survey-based assessments of soil erosion on arable land

Between 1982 and 1986, a scheme was administered by the then Soil Survey of England and Wales (SSEW) to assess if soil erosion was a

Table 1
Datasets used to compare soil erosion rates on lowland agricultural land across England and Wales.

Dataset	Period data collected	Location	Method	No. fields or catchments	Reference	Comment
SSEW monitoring project	1982–87	17 localities in England Wales covering on average 708 km^2 yr^{-1}	Survey-based	1702 fields	Evans, 2005	Volumetric estimates of rill erosion, no assessment of sheet erosion
Boardman	1982–1991	36 km^2 South Downs, Sussex, England	Survey-based	114 fields	Boardman, 2003	Volumetric estimates of rill erosion, no assessment of sheet erosion
University of Exeter	2005–2008	Through-out England and Wales	^{137}Cs, estimate for last 50 years	248 fields	Walling and Zhang, 2010	Sum of all processes affecting ^{137}Cs redistribution
Grassland surveys	Various (mean annual)	River sub-catchments in England and Wales	Integration of sediment source apportionment, sediment yield ranges and landscape retention	71 sub-catchments	Collins et al., 2010, 2012; Cooper et al., 2008	Source fingerprinting reflects only the < 63 µm fraction
North Wyke Farm Platform	2012–2013	Devon, SW England	Turbidity monitoring in flow from hydrologically-isolated fields	63 ha divided into 15 sub-catchments	Griffith et al., 2013; Orr et al., 2016	Conversion of turbidity to sediment mass dependent on ratings from gravimetric filtration of water samples

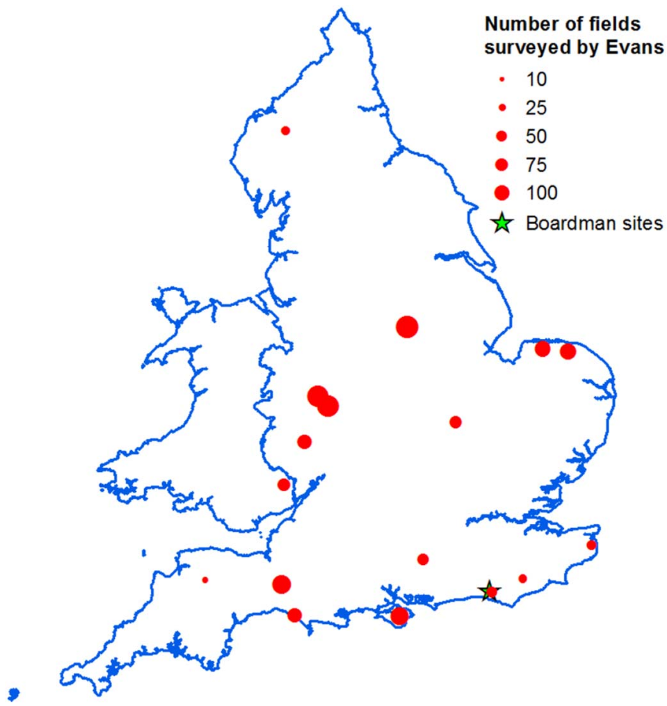

Fig. 1. Locations of SSEW erosion transects and number of monitored fields surveyed each transect; and locality monitored by Boardman.

problem. Eroded fields were identified on aerial photographs, and was followed by field measurements where volumes of soil eroded and deposited were estimated. An evaluation was also made as to why erosion did not occur in all fields with similar physical characteristics. Fieldwork was carried out in 17 localities across England and in one locality in Wales (Fig. 1). The results have been widely reported elsewhere (Evans, 1990b, 1993, 1998b, 2005, 2013).

Independently of the SSEW monitoring programme, Boardman monitored erosion on the hilly chalk land of the South Downs of southern England (Fig. 1) for a 10-year period between 1982 and 1991. The monitored area partly coincided with the SSEW transect. Again, the results have been widely reported (e.g. Boardman, 1990, 2003). Both Evans and Boardman measured lengths of rills and ephemeral gullies and their cross-sectional areas to produce estimates of the volumes of soil eroded.

Evans (1990a) described actual, that is occurring, erosion, and potential erosion risk for the soil associations portrayed on the National Soil Map (SSEW, 1983) based on extent and rates of erosion as found in the SSEW erosion project (Fig. 2) and land use and relief of the soil associations (soil landscape) as described in the map legend and accompanying six Bulletins (SSEW, 1984). That information has stood the test of time with respect to providing a reliable dataset on soil erosion rates on lowland arable land (Evans et al., 2016). For these field-based surveys, gross erosion was the total volume of soil eroded in the field. Not all soil was retained in the field since some was transported out of the boundary (Evans, 2006).

Evans' classification of actual and potential erosion are related, as noted above, to soil associations portrayed on the Soil Association Map of England and Wales. The information (Fig. 2) was not originally published in map form because of copyright problems, but a raster version of it was published by Friends of the Earth (Evans, 1996) and was later updated to give a better quality map (Evans et al., 2016). Five classes of risk were defined based on the estimated percent of the area of the soil association covered by eroded fields (rare or not at all, small < 1%, moderate 1–5%, high 5–10%, very high > 10%) and not on rates of erosion because mean and median rates of eroded fields varied less between localities but were generally higher where erosion was more extensive (Evans, 1998b). However, this was not always the case, for example, soils with high fine sand or coarse silt content could have high rates of erosion but often were not in localities that were predominantly arable, or had low relative relief. Erosion was most extensive where soils were sandy and where a wide range of crops were grown so that land was vulnerable to erosion for more of the year. In other words, Evans' map was essentially based on soils and land use and what was actually happening on the ground in the 1980s.

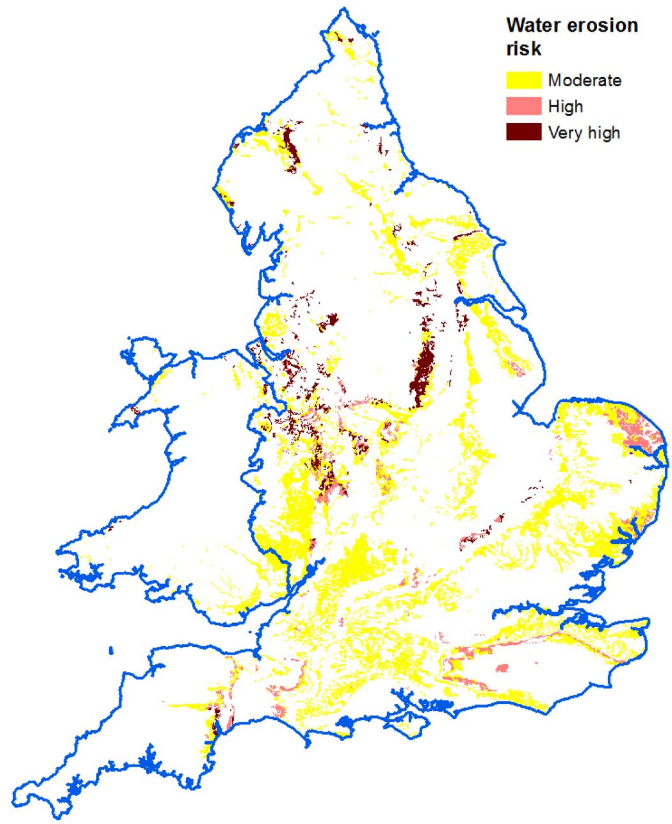

Fig. 2. Risk of soil erosion by water (after Evans, 1990a).

2.3. The England and Wales ^{137}Cs-based erosion survey for arable and grassland

As part of a national assessment of soil erosion rates in England and Wales (University of Exeter, 2008; Walling and Zhang, 2010) funded by the UK Department for Environment Food and Rural Affairs (Defra), ^{137}Cs measurements were taken from 248 fields distributed throughout lowland England and Wales (Fig. 3), but with a bias to south western England, and well-established conversion models were used to estimate gross and net erosion rates. The previous methodological work on developing the application of the ^{137}Cs-based approach to soil erosion assessment in the UK has been widely reported elsewhere (e.g. Quine and Walling, 1991; Walling et al., 2002). The annual average rates of erosion are those pertaining over approximately the past 50 years and, where necessary, are corrected for any Chernobyl nuclear accident inputs. Critically, however, the rates generated reflect all processes governing soil redistribution, rather than water erosion alone, because of the conversion model used.

2.4. Estimates of grassland soil erosion based on a combination of sediment source apportionment, suspended sediment yield and long-term landscape retention

The estimates of gross erosion for grassland are derived from

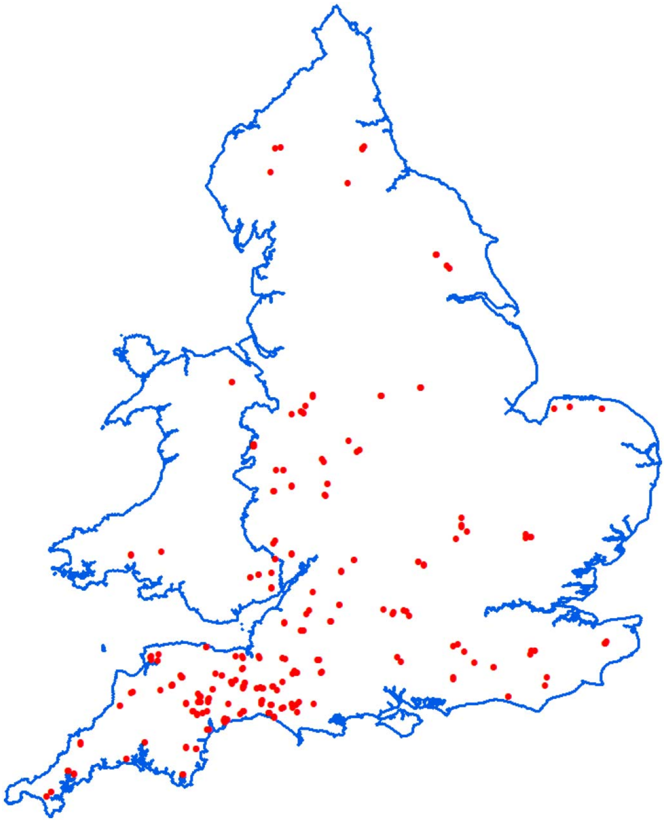

Fig. 3. The locations of the fields used in the national ^{137}Cs-based soil erosion survey by Walling and Zhang (2010).

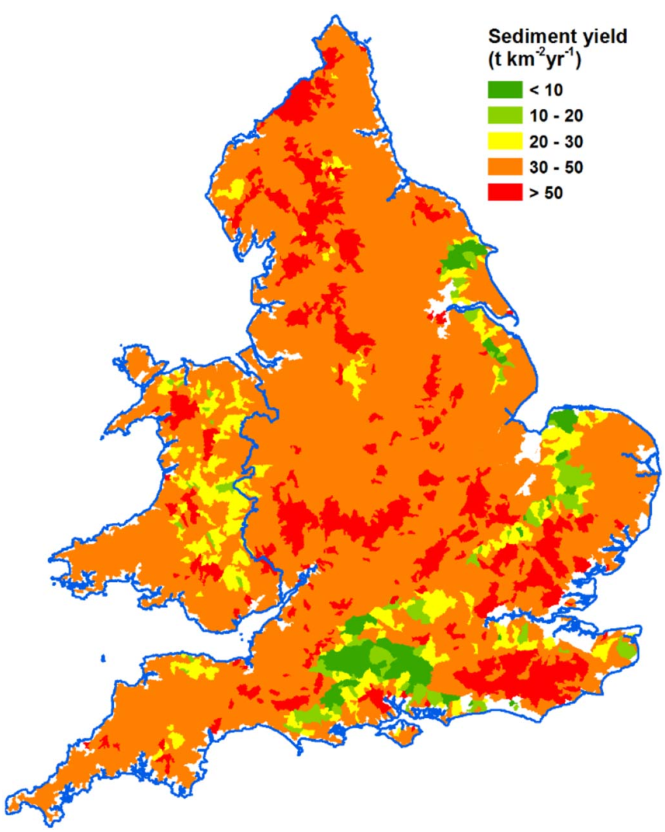

Fig. 4. Ranges in suspended sediment yields for river catchments across England and Wales (after Cooper et al., 2008).

integrating sediment source fingerprinting data on the proportions of fine-grained (< 63 μm) sediment derived from grassland sources (topsoils – upper 0/2 cm depth; e.g. Collins et al., 2010, 2012), the proportion of each sub-catchment represented by grass based on the ADAS land use database (cf. Comber et al., 2008) and the Centre for Ecology and Hydrology (CEH) national map of contemporary suspended sediment yields (Cooper et al., 2008) for England and Wales (Fig. 4). Integration of these datasets provides estimates of sediment loss to watercourses from grass fields, but in the context of catchment outlet suspended sediment yields being net of long-term upstream landscape sediment storage e.g. on floodplains. Multipliers (see Discussion section) were therefore used to derive estimates of total net soil loss from grassland to the river channel network. Two multipliers were used: 1) 1.25, for a sediment delivery ratio of ~0.8 (i.e. 20% landscape sediment storage), and 2) 1.45, for a sediment delivery ratio of ~0.7 (i.e. 30% landscape sediment storage). Estimates of total net soil loss from grassland were assumed to represent gross erosion given that, in the grassland landscapes used for this work, the grassed/vegetated areas in riparian zones which amount to overbank deposition areas, represent the most important deposition zones for sediment released by soil erosion processes on grass fields upslope. The ranges for gross soil erosion in grassland environments in England and Wales were derived using the lower and upper sediment yields for each location (Cooper et al., 2008) in conjunction with the estimated average mean/median contribution from grassland topsoil sediment sources generated using the Monte Carlo mass balance modelling from sediment source fingerprinting (e.g. Collins et al., 2010) plus/minus the corresponding 95% confidence limits about the average mean/median grassland contribution. Since the average mean/median grass contributions estimated by sediment source tracing are the product of Monte Carlo uncertainty modelling, they can be taken to reflect central tendency in grass contributions by taking account of the entire range of outputs provided by the uncertainty analysis. The results for grassland soil erosion were derived on this basis for 71 sub-catchments across England and Wales (Fig. 5).

2.5. *The North Wyke Farm Platform*

The North Wyke Farm Platform (NWFP) was established in South West England (50°46′10″ N, 3°54′05″ W) as a UK National Capability for collaborative research, training and knowledge exchange in agro-environmental sciences with a remit to research agricultural productivity and ecosystem responses to different management practices for beef and sheep production in lowland grasslands (Orr et al., 2016). A system based on permanent pasture has been implemented on three 21-ha farmlets to obtain baseline data on responses including hydrology and soil loss from April 2011. The three farmlets were further divided into 5 sub-catchments (15 in total) which were hydrologically-isolated through a combination of topography and a network of French drains (800 mm deep trenches containing a perforated drainage pipe back-filled to the surface with stone) which were constructed at the edges of the sub-catchments (Fig. 6). Each of the 15 measurement sites incorporates an H-flume, a pre-chamber for the collection of samples, a power and IT cabinet terminating mains power and fibre optic cables, and a flume laboratory housing pumping equipment, a by-pass cell, telemetry devices and turbidity sensors (Griffith et al., 2013).

3. Comparing erosion rates on arable land using ^{137}Cs and traditional field survey methods

3.1. *Soil landscape scale*

The estimate of gross mean annual erosion across the SSEW monitored air photo transects (Table 2) is based on the area of eroded fields recorded over the monitored period multiplied by the mean rate of erosion in those fields divided by the total area of the monitored

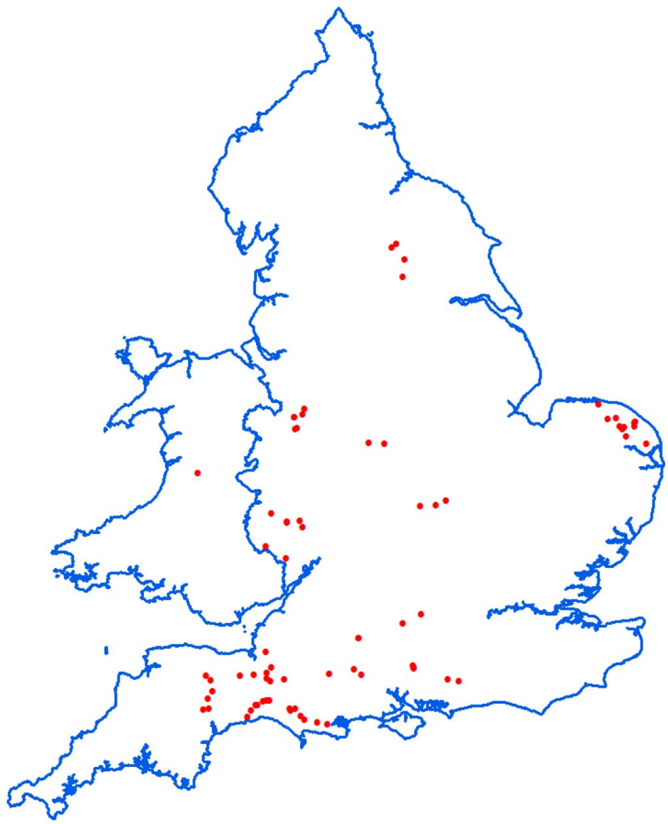

Fig. 5. The locations of the sub-catchments for which estimates of grassland soil erosion were generated by combining information on sediment source apportionment, suspended sediment yield and landscape retention.

landscape. To estimate mean annual gross erosion using the ^{137}Cs technique, estimates were made from the outputs of Walling and Zhang (2010) (Section 3.3 below, and Fig. 7) by locating the kilometre squares that correspond to those on the SSEW monitored transect and assuming the mid-point values of erosion rate classes, i.e. 50 t km^{-1} yr^{-1}, 125 t km^{-2} yr^{-1}, 175 t km^{-2} yr^{-1}, 300 t km^{-2} yr^{-1} and 658 t km^{-2} yr^{-1} and using these values to derive the erosion rate for that transect.

^{137}Cs-based estimates of gross soil erosion in the SSEW soil landscapes are consistently greater than field-based estimates (Table 2). Estimates are less than one order of magnitude greater where relief is low (e.g. Hampshire erodible soils, Isle of Wight erodible soils, Norfolk East and West, Nottinghamshire sandy soils, Shropshire, and Staffordshire sandy soils) and ^{137}Cs estimates are low, except for erodible soils in Somerset where relief is greater. In all these instances, gross erosion assessed in the field using the survey-based approach was extensive and rates were high. ^{137}Cs-based estimates of gross erosion are 1.5 (Cumbria, Nottinghamshire silty and Sussex East) to two (Devon, Hampshire less erodible soils, Nottinghamshire clayey and Somerset less erodible soils) orders of magnitude greater where erosion assessed using the survey-based approach was uncommon and corresponding rates were low.

3.2. Comparison of the ^{137}Cs-based estimates of erosion and Boardman's findings on the South Downs, East Sussex, England

Boardman (1990, 2003) monitored soil erosion on the South Downs chalk (Fig. 1) land between Brighton and Lewis in Sussex for 10 years between 1982 and 1991. Almost all significant erosion was related to concentrated flow; there was little evidence of sheet flow and it was regarded as contributing < 0.3 m^3 ha^{-1} yr^{-1} to soil loss (Evans, 1990b). In the 10-year period, one instance of wind erosion was observed on one field. Tillage erosion is not significant as chisel ploughs are generally used on the light shallow soils of the South Downs. The area monitored covered c. 3600 ha (36 km^2). Over the period 1982–1991, rills were found in 124 fields. Erosion was recorded in one field in nine of the 10 years but most fields eroded only once or twice. However, there were many fields that did not erode in any monitored year. The area of land affected by erosion each year varied greatly (a highly positively skewed distribution) but, on average, 372.8 ha suffered erosion each year, c. 10.3% of the area monitored. Gross erosion for the monitored area of c. 3600 ha (36 km^2) was estimated at 1836.0 t y^{-1}. The mean rate for the eroded fields was 4.9 t ha^{-1} yr^{-1} but is much less, 0.51 t ha^{-1} yr^{-1} for the whole of the monitored area.

By comparison, gross soil erosion for the same area estimated from the mid-point values of the erosion rates provided by Walling and Zhang (2010) was estimated at 11051 t yr^{-1}, i.e. 6 times greater than the corresponding traditional survey-based estimate. The net survey-based erosion rate of these silty soils was estimated to be 420.4 t yr^{-1}, compared to 9477 t yr^{-1} indicated by the ^{137}Cs-based study by Walling and Zhang (2010), i.e. the ^{137}Cs-based estimate is 22.5 times higher. The mean extent and mean rate of soil erosion in this South Downs location, part of which was monitored in the SSEW scheme, were found by Boardman to be much greater than those estimated from the earlier SSEW survey. This difference in mean extent and rate of erosion is primarily explained by the severe erosion which occurred during a rare large storm in autumn 1987 the year following the end of the SSEW project (Boardman, 1988b). The period of monitoring (1982–1991) by Boardman almost certainly represents the maximum extent of winter cereals and therefore maximum risk of soil erosion in this area in modern times. The rates recorded are therefore not representative of the post Second World War period. Many of the local fields with high average rates for 1982–91, especially those around Rottingdean, have not been cultivated since 1987 but have been under grass; rates for those fields can therefore be adjusted (Table 3).

One of the three fields chosen by Walling and Quine (1991) to demonstrate the potential of the ^{137}Cs-based method for assessment of rates of soil erosion was in the South Downs monitored area (Boardman, pers. comm.). About 6.25 ha of the field was assessed. The total area of the field is 30.9 ha and therefore direct comparisons between results in Walling and Quine (1991) and the current work is not straightforward. The 10-year average for the 30.9 ha field estimated by Boardman is 0.4 m^3 ha^{-1} yr^{-1} (0.5 t ha^{-1} yr$^{-1)}$ compared with a gross rate of 4.3 t ha^{-1} yr^{-1} in Walling and Quine (1991). Similarly, in Nottinghamshire, direct comparisons between results in Walling and Quine (1991) and the current work is not straightforward. Fig. 3 in Quine and Walling (1991) shows the area in a field in Nottinghamshire sampled for ^{137}Cs. Again, as on the South Downs, this is only part of a field of c.8.8 ha in area. The gross rate of erosion for that part of the field is estimated to be 12.2 t ha^{-1} y r^{-1}. Field monitoring not only suggests that in one year, rill erosion took place only in the valley floor, dominantly a zone of deposition according to the ^{137}Cs-based assessment, but that when erosion was averaged across the whole field in three years (out of 5), rates varied from 0.1–0.7 t ha^{-1}. In two years, no erosion or deposition in the field was recorded. Whatever the difficulties of exact comparison, the difference between the two methods of assessment appears to be at least an order of magnitude and therefore consistent with the comparisons between the SSEW transect rates and ^{137}Cs-based predictions described above.

3.3. National scale

To map spatial patterns of ^{137}Cs erosion rates from arable and grassland across lowland agricultural areas, spatial datasets were integrated to estimate the relative distribution of unique combinations of land use category (arable or grassland), slope gradient category (< 3, 3–7, 7–11 and > 11°) and soil texture categories (> 10 categories based on the SSEW classification) at 1 km spatial resolutions. The

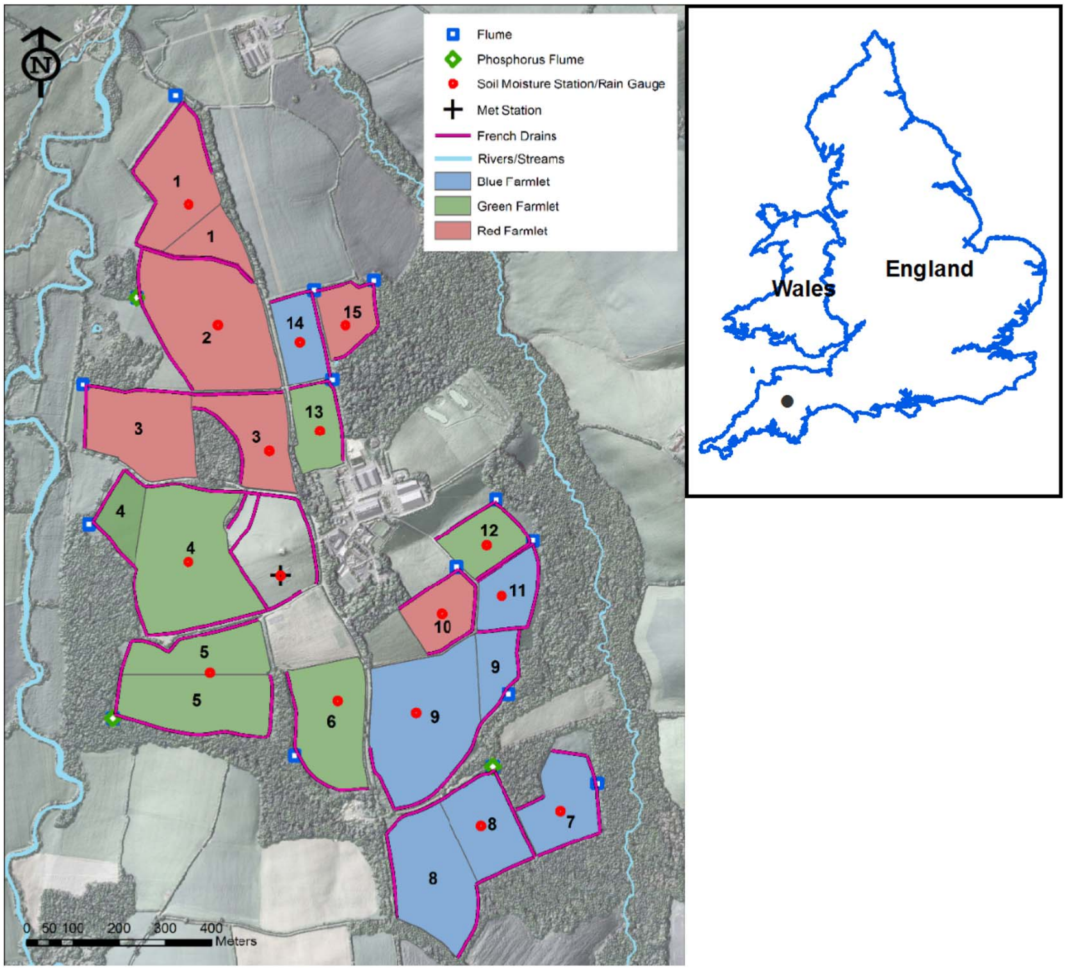

Fig. 6. The North Wyke Farm Platform, and its location in SW England.

relevant spatial datasets used included: the 1991 CEH land cover map where the arable category includes 'Tilled' only and grassland which includes grass heath, mown/grazed turf, meadow/verge/semi-natural, moorland grass; a 50 m Digital Elevation Model (DEM) which was used to calculate slope gradient, and National Soil Inventory (NSI) texture data for top soils which were used to map soil texture based on the SSEW classification system. The limited data available precluded the estimation of erosion rates for all unique combinations of land use, slope and soil texture. A compromise was therefore adopted. Firstly, average erosion rates for different combinations of land use and slope gradients (R_{ls}) and different combinations of land use and soil texture (R_{lt}) were estimated based on the erosion rates generated by the ^{137}Cs-based approach for the fields sampled (Fig. 3). Erosion rates (R) for each unique combination of land use, slope and soil texture was subsequently estimated using the following formula: $R = \sqrt{R_{ls} * R_{lt}}$. For any 1 km cell, the average erosion rate was calculated as the areal-weighted erosion rates for the different combinations of land use (arable or grass only), slope and soil textures present.

The ^{137}Cs-based gross erosion rate (Fig. 7) represents the total erosion from the eroding portions within the zone of the field sampled, divided by the total area of that zone. The net erosion (Fig. 8) represents the sum of the erosion from the eroding portions minus the sum of the deposition occurring within the non-eroding portions, divided by the area of the zone considered. The ^{137}Cs-based maps use ^{137}Cs inventories converted into rates of soil erosion using a mass-balance model for cultivated land and a diffuse and migration model for uncultivated (pasture) land. Erosion classes of 0–100, 100–150, 150–200, 200–400 and 400–917 (gross erosion) or 400–841 (net erosion) t km^{-2} yr^{-1} are mapped (Fig. 7).

The range of survey-based assessments of erosion rates estimated within individual fields is similar to (see below) the range estimated for individual km grid squares (^{137}Cs-based estimates provided by Walling and Zhang, 2010, Fig. 7) but the survey-based assessments reported herein show that not all fields within a kilometre grid square erode; indeed very few fields erode in any one year (Table 4), and those that do, generally cover 5% or (much) less of the landscape in question (Evans et al., 2016). To compare estimates of gross amounts of soil eroded across England and Wales, i.e. at a national scale, using the two different approaches, assumptions necessarily have to be made. In the case of traditional survey-based assessments, it is estimated that for soil associations or landscapes at very small risk of erosion, 0.1% of the fields within the landscape erode in any one year, for small risk 0.5%, moderate risk 2.5%, high risk 7.5% and very high risk 12.5%. Erosion rates are assigned on the basis of the nearest equivalent soil type and land form to that in the 1982–86 SSEW monitoring scheme, for example: 0.7 t ha^{-1} (= 70 t km^{-2}) for clay landscapes with low relief (e.g. Bedfordshire, England) and 2.2 t ha^{-1} (220 t km^{-2}) for clay landscapes with higher relief and greater rainfall (e.g. Dorset, England); 3.0 t ha^{-1} (300 t km^{-2}) for sandy soils with low relief (West and East Midlands of England) and 6.6 t ha^{-1} (660 t km^{-2}) for coarse loamy and sandy soils with stronger relief (Isle of Wight, England), and; 5.5–5.7 t ha^{-1} (550–570 t km^{-2}) for silty soils with low relief (Kent and Somerset, England). For ^{137}Cs-based assessments (Figs. 7 and 8), a conservative (low) estimate of gross and net erosion of 50 t km^{-2} yr^{-1} (0.5 t ha^{-1} yr^{-1}) is made for England and Wales.

The lowland soil landscapes of England and Wales cover an area of

Table 2
^{137}Cs and field-based estimates of gross erosion for soil landscapes covered by the SSEW aerial photograph transects. Data for transects from the original SSEW project were derived from estimated rates of erosion, area of eroded fields and frequency of erosion.

Soil landscape	Estimated gross annual erosion rate		Area	Total		Ratio
	t ha^{-1}		Ha	Tonnes		
	^{137}Cs	Field		^{137}Cs	Field	^{137}Cs:field
Bedfordshire	0.5	0.0133	10,500	5250	140	37.5
Cumbria	0.5	0.0077	4593	2296	35	65.6
Devon	1.755	0.0111	7655	13,396	85	157.6
Dorset	1.5	0.0597	4832	7248	288	25.2
Gwent	0.5	0.0121	3673	1836	44	41.7
Hampshire						
Erodible	0.5	0.3483	1819	909	634	1.4
Less erodible	0.5	0.0008	1984	992	1.6	625.0
Hereford	0.5	0.0401	5593	2796	224	12.5
Isle of Wight						
Erodible	0.5	0.2873	1397	954	549	1.7
Less erodible	1.75	0.0473	1908	2445	66	21.2
Kent	1.0	0.0275	3594	3594	99	36.3
Norfolk East	0.5	0.1142	3313	1656	378	4.4
Norfolk West	0.5	0.0672	4362	2181	293	7.4
Nottinghamshire						
Sandy	0.5	0.4057	3705	1852	1503	1.2
Silty	0.5	0.0255	639	319	16	93.9
Clayey	0.5	0.0041	1709	539	4.4	122.5
Shropshire	0.5	0.1732	3109	1554	538	2.9
Somerset						
Erodible	1.25	0.2878	3560	4450	1025	4.3
Less erodible	1.25	0.0117	1034	1292	12	107.7
Staffordshire						
Sandy	0.5	0.3191	3608	1804	1151	1.6
Less erodible	0.5	0.0162	1951	975	32	30.5
Sussex East	1.0	0.0119	4912	4912	58	84.7
Sussex West						
Erodible	3.0	0.0197	2837	8511	311	27.4
Less erodible	0.5	0.0273	853	431	23	18.8

106,281 km^2 and it is estimated from data derived from the SSEW monitored transects that 287,623 t of soil erodes in a typical year, equivalent to a mean of 2.71 t km^{-2} yr^{-1} (0.027 t ha^{-1} yr^{-1}), 18.4 times less than the conservative ^{137}Cs-based estimate taken from Fig. 7. While it is acknowledged that some of the difference relates to the fact that the ^{137}Cs-based method does not directly distinguish the type of erosion, but accounts for the sum of erosion by water, tillage and wind (c.f. Brazier, 2004) it is highly unlikely that the nearly 20-fold difference in average erosion rate predicted by these two methods is a function of the extra soil loss associated with wind and tillage erosion. In addition, even though the traditional field surveys focused on volumetric erosion features (rills and gullies) and did not measure sheet wash, existing literature suggests that sheet wash only typically accounts for a soil loss of 0.1–0.3 t ha^{-1} yr^{-1} (Evans, 2006; Evans et al., 2016). This specific soil loss process is therefore also not able to explain the disparity between the traditional survey-based and ^{137}Cs-based estimates. Here, it is also noteworthy that Parsons and Foster (2011) underscored that subsurface losses of ^{137}Cs (and soil) were also generally ignored by ^{137}Cs conversion models, but there is growing evidence (reported above) that subsurface pathways represent an important national scale route (e.g. Zhang et al., 2016 for recent information) for sediment (and thereby ^{137}Cs) loss to rivers in England and Wales. Direct measurement of sediment issuing from land drains in lowland England has shown it to have consistently high ^{137}Cs activities and almost always measurable activities of the short-lived nuclide ^{7}Be, testifying to the dominant topsoil origin of this sediment (Chapman et al., 2005). In part, the discrepancy between lower average topsoil and higher drain sediment ^{137}Cs activities is explained by the very fine nature of the latter which is usually ≪32 μm in diameter (Chapman et al., 2005), since particle size exerts a strong control on ^{137}Cs adsorption by soil and sediment (He and Walling, 1996).

As noted above, Evans and Brazier (2005) found little relation between relief and rainfall and extent and severity of erosion. It was a surprise therefore to see that the ^{137}Cs maps (Figs. 7 and 8) portray erosion rates that relate well to relief, not to soil landscapes or land use (Fig. 2). Areas of more resistant rocks with their greater relief are depicted by the ^{137}Cs-based approach to have higher rates of erosion, for example, the Jurassic limestone and Cretaceous chalk outcrops and the fringes of the uplands. It was this apparent discrepancy that caused some of us to question the ^{137}Cs technique, following earlier concerns, and which resulted in the new quantitative comparisons described herein.

3.4. Comparison of grassland soil erosion rates estimated using ^{137}Cs and the integration of sediment source, landscape retention and yield estimates

A comparison between the grassland soil erosion rates using the integration of traditional data and the ^{137}Cs-based approach is provided in Table 5. For the purpose of this exercise the grassland erosion rates were compared on the basis of the Cross Compliance soil categories used by agricultural advisors in England and Wales (see textural definitions in the footnote of Table 5). Accordingly, the grassland erosion rate data provided by the two different approaches were grouped into national populations by soil category since the ^{137}Cs-based survey included 115 fields under grass and these did not always correspond to the landscapes for which the data based on the more traditional data streams exist. Table 5 shows that the estimates of grassland soil erosion rates for light textured soils ranged from 0.17–0.48 t ha^{-1} yr^{-1} assuming a landscape retention multiplier of 1.25 and 0.20–0.55 t ha^{-1} yr^{-1} with a multiplier of 1.45, compared with a range of 1.31–4.07 t ha^{-1} yr^{-1} using the ^{137}Cs-based approach. The comparisons between the two datasets for the remaining Cross Compliance soil categories give a similar picture. For example, the grassland soil erosion rates for heavy soils using the traditional data ranged from 0.24–0.63 t ha^{-1} yr^{-1} and 0.28–0.73 t ha^{-1} yr^{-1}, compared with a corresponding range of 1.31–4.07 t ha^{-1} yr^{-1} using the ^{137}Cs-based approach (Table 5). Soil erosion estimates generated using the ^{137}Cs-based conversion model are therefore consistently higher.

4. Discussion

Direct comparisons of erosion estimates generated using different methods face several challenges, not least, the contrasting temporal and spatial coverage and process representation of different techniques (Brazier, 2004). Accordingly, it is instructive to recognise and compare some of the principal assumptions and limitations of individual methods (Table 6). Since the methods used by Boardman and Evans are based on the measurement of a volume of soil loss produced by a single water-related process (rill/gully erosion) it is unsurprising that the ^{137}Cs-based method would produce higher estimated rates as it will include soil erosion by tillage, surface wash, wind erosion, soil redistribution in conjunction with harvesting equipment (e.g. for root crops), and subsurface sediment transfer, although these processes, other than tillage erosion, are not included in current FRN conversion models. In reporting the results of the National Survey of soil erosion using ^{137}Cs (University of Exeter, 2008), Walling and Zhang (2010) did not make such a comparison with independent measures of erosion but, instead, reported comparisons with ranges obtained using FRN in other studies. Unsurprisingly, results appeared comparable and therefore seem to fully justify the use of the FRN methodology to evaluate soil erosion nationally (e.g. Walling et al., 2003; Walling and Zhang, 2010). Comparisons of erosion estimates discussed in this paper, albeit not without limitations of their own, strongly suggest that further evaluation of the data generated using the ^{137}Cs-based approach is urgently required. This is particularly so, if the approach is used as a basis to estimate soil

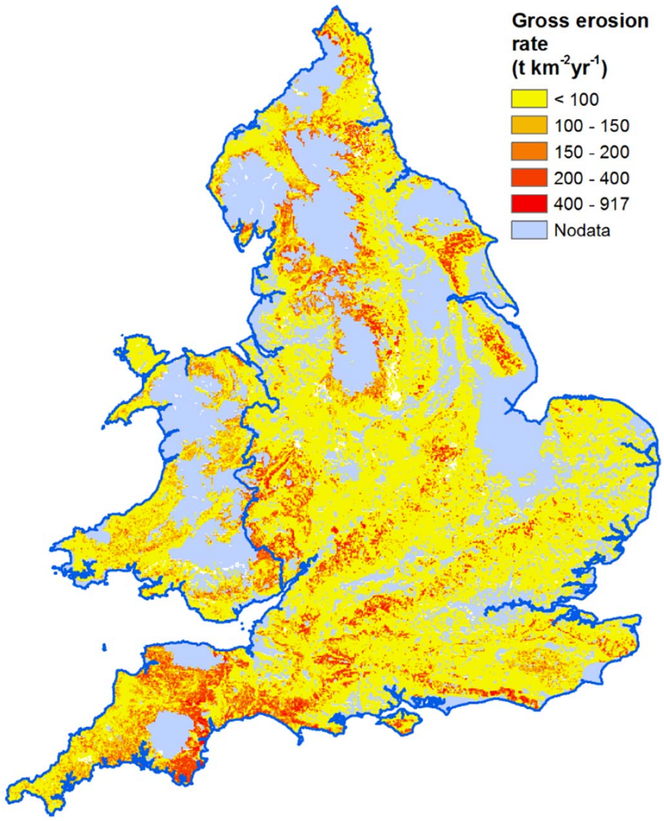

Fig. 7. Gross soil erosion rates estimated from the ^{137}Cs-based approach (University of Exeter, 2008; Walling and Zhang, 2010).

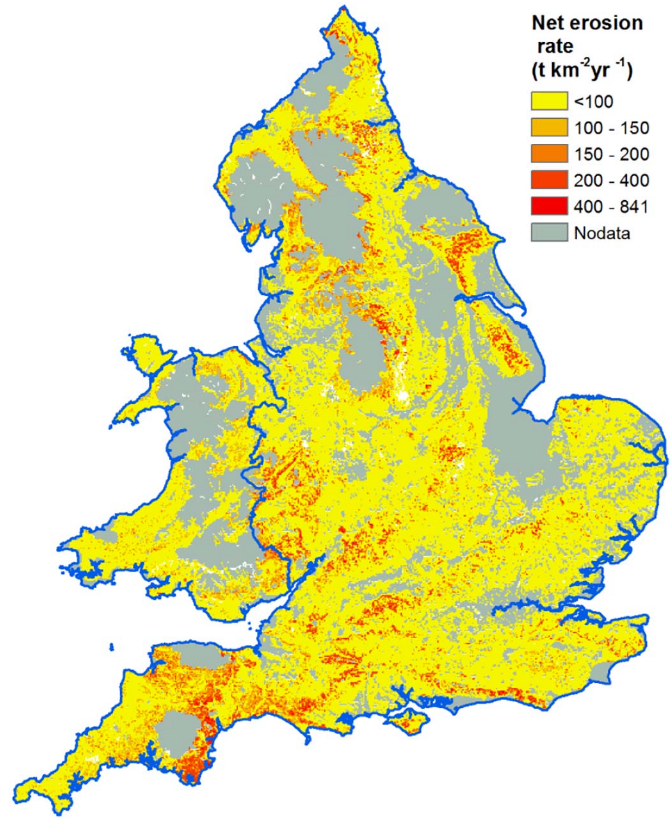

Fig. 8. Net soil erosion rates estimated from the ^{137}Cs-based approach (University of Exeter, 2008; Walling and Zhang, 2010).

Table 3
Erosion rates (t ha^{-1} yr^{-1})[a] in seven fields that have not been cultivated since 1987 at Rottingdean, South Downs, for different lengths of time.

Field number	1987–88	10-years average	Average 1982–20,012
72	81.9	8.2	2.6
77	300.3	30.3	9.7
78	76.7	7.7	2.5
79	97.5	9.7	3.1
80	26	2.6	0.8
81	48.1	4.8	1.5
82	68.9	6.9	2.2

[a] Volume converted to mass, assuming soil bulk density of 1.3 t m^3.

erosion by water. There are currently very few evaluations of ^{137}Cs derived erosion rates at field scale because of the lack of comparable empirical data at appropriate temporal scales. Available comparisons against measured data tend to be indirect, e.g. against USLE or RUSLE (Turnage et al., 1997), or at micro-catchment (< 2 ha, Porto et al., 2003) or plot scale (cf. Ritchie and McHenry, 1990). We note that Porto et al. (2003) highlighted the ongoing urgent need for independent validation of erosion estimates generated using the ^{137}Cs technique, including estimates for cultivated soils, and those generated using different conversion models and procedures. This need has also been underscored by Golosov et al. (2017). There are some evaluation studies in the international literature (Porto et al., 2001, 2003, 2011, 2013, 2014, 2016). A key point here though, concerns the small scales adopted in these evaluation exercises. The comparisons herein clearly underscore the issues associated with linear extrapolation of point-based ^{137}Cs estimates collected at field or micro-scale to larger landscape units. It is therefore not surprising that existing validation exercises suggest that ^{137}Cs is generating erosion estimates in the right order of magnitude since the most recent ones are all at micro-

Table 4
Area of eroded (rilled) fields as a percentage of the agricultural land in the 17 SSEW localities monitored from 1982 to 1986.

Locality	Mean %	Range %
Bedfordshire	2.0	0.03–3.1
Cumbria[a]	1.1	0.01–2.9
Devon[a]	0.5	0.2–1.0
Dorset	2.7	0.9–4.5
Gwent	2.4	0.6–4.0
Hampshire		
Erodible	6.4	1.5–19.8
Less erodible	0.5	0–2.7
Hereford	1.9	0.2–4.9
Isle of Wight		
Erodible	10.2	2.4–19.7
Less erodible	2.0	0–4.7
Kent	1.9	1.0–3.0
Norfolk East	8.3	2.5–14.1
Norfolk West	5.2	1.4–8.2
Nottinghamshire		
Sandy	13.9	1.5–24.0
Silty	3.9	0–7.5
Clayey	1.6	0–4.0
Shropshire	9.2	4.6–14.8
Somerset		
Erodible	5.1	2.0–9.0
Less erodible	0.3	0–0.5
Staffordshire		
Sandy	9.1	6.0–15.7
Less erodible	1.9	0–5.1
Sussex East[a]	1.1	0–2.3
Sussex West		
Erodible[a]	7.7	3.6–12.6
Less erodible[b]	3.5	1.5–7.3

[a] 4 years data.
[b] 3 years data.

Table 5
Comparison of grassland soil erosion rates generated using a combination of sediment source apportionment, suspended sediment yield and landscape retention factors with those based on the ^{137}Cs approach.

Conventional methods	Assuming a landscape retention multiplier of 1.25 (i.e. long-term sediment storage of ~20% = a sediment delivery ratio of 0.8)			
Cross compliance soil category[a]	Mean/median erosion rate $t\ ha^{-1}\ yr^{-1}$[b]	95% confidence limits $t\ ha^{-1}\ yr^{-1}$[b]	Mean/median erosion rate $t\ ha^{-1}\ yr^{-1}$[c]	95% confidence limits $t\ ha^{-1}\ yr^{-1}$[c]
Light (sandy and light silty)	0.18	0.17–0.18	0.46	0.45–0.50
Medium	0.21	0.21–0.22	0.48	0.47–0.50
Heavy	0.25	0.24–0.25	0.62	0.61–0.63
Chalk/limestone	0.47	0.46–0.48	1.18	1.15–1.20
Conventional methods	Assuming a landscape retention multiplier of 1.45 (i.e. long-term sediment storage of ~30% = a sediment delivery ratio of 0.7)			
Light (sandy and light silty)	0.20	0.20–0.21	0.54	0.52–0.55
Medium	0.24	0.247–0.25	0.56	0.54–0.58
Heavy	0.29	0.28–0.29	0.72	0.71–0.73
Chalk/limestone	0.55	0.54–0.558	1.37	1.34–1.39

^{137}Cs-based	Mean/median erosion rate $t\ ha^{-1}\ yr^{-1}$	95% confidence limits $t\ ha^{-1}\ yr^{-1}$
Light (sandy and light silty)	2.69	1.31–4.10
Medium	2.28	1.80–2.77
Heavy	3.19	2.00–4.40
Chalk/limestone	1.72	0.86–2.58

[a] Light: < 18% clay; 0–40 cm depth.
Medium: 18–35% clay; 0–40 cm depth.
Heavy: > 35% clay; 0–40 cm depth.
[b] Using the lower suspended sediment yield range in Cooper et al. (2008).
[c] Using the higher suspended sediment yield range in Cooper et al. (2008).

catchment scale. To address the research gap surrounding evaluation of ^{137}Cs-based erosion estimates underscored by some researchers using the ^{137}Cs approach, this contribution has described comparisons at field, landscape and national scales in lowland England and Wales where independent traditional survey-based and monitoring data from both arable and grass fields are available.

The discrepancy between ^{137}Cs-based soil erosion estimates for arable land and those generated using more traditional field survey methods has been noted previously (Brazier, 2004; Boardman, 2013a). Quine and Walling, 1991 compared rates of erosion assessed by ^{137}Cs in the fields they sampled with rates assessed by field surveying, and concluded "the erosion rates calculated from the ^{137}Cs data are in-line with other published measurements" (pp. 176), although ^{137}Cs estimates are on the high side. However, the survey-based assessments, as do the ^{137}Cs-based assessments, relate to individual fields, not to soil- or landscape units. In later work (University of Exeter, 2008; Walling and Zhang, 2010) ^{137}Cs-based assessments were scaled up to landscapes and to national scale using a GIS extrapolation procedure. The mapped ^{137}Cs-based soil erosion rates at national scale (Walling and Zhang, 2010; Figs. 7 and 8) introduce additional uncertainties, including those associated with the land cover data used, and assumptions about the combinations of land use, slope and texture. The CEH 1991 land cover map (LCM) was used to identify cultivated land (tilled) and grassland (grass heath, mown/grazed turf, meadow/verge/semi-natural, moorland grass). Only these identified cultivated and grassland land cover categories were included for the GIS upscaling exercise of the ^{137}Cs-based estimates, i.e. other land uses were not included. On this basis, strictly speaking, the estimated erosion rates presented in Figs. 7 and 8 correspond to those selected land cover types only, and not for the whole 1 km^2 cell. The upscaling of the ^{137}Cs-based field scale data involved the estimation of the median erosion value for different combinations of land use and soil texture and different combinations of land use and slope, since the population of sampling sites used in the national ^{137}Cs-based soil erosion survey (Fig. 3) did not provide sufficient data to generate estimated erosion rates for all unique combinations of these factors. Some data manipulations were undertaken to integrate the different spatial resolution datasets for slope (50 m DEM from CEH), land cover (1991 1 km × 1 km data from CEH) and soil texture (5 km × 5 km data from NSRI, Cranfield). Assumptions had to be made about the distribution of different land uses. For example, it was assumed that arable land will occupy more gentle slopes within a landscape. On the basis of the above, slope/relief was used for upscaling from the ^{137}Cs survey sites in Fig. 3, but is not included in the ^{137}Cs conversion models for estimating erosion rates from ^{137}Cs inventories. Regardless of scale, however, the comparisons undertaken herein, clearly demonstrate that, in the case of arable land, ^{137}Cs-based estimates of both gross and net soil erosion are consistently higher than those assembled using traditional field surveys based on volumetric measurements.

The variable but consistent (in terms of direction of bias) differences between the ^{137}Cs-based and more traditional estimates of gross and net soil erosion across landscapes within England and Wales can be explained, in part, by the fact that erosion does not occur every year, everywhere within a landscape, (Evans et al., 2016). By way of example, in the SSEW survey, the greatest extent of erosion was recorded on the sandy soils of Nottinghamshire; on average 13.9% of the landscape eroded in a single year (range 1.5–24.0%). Elsewhere, except for the Isle of Wight, eroded fields on average covered < 10% of the monitored area (Table 4). In the Sompting catchment in southern England, also on the South Downs but 18 km to the west of the area monitored by Boardman, prior to the mid-1980s mixed farming predominated but after that until 2006 winter cereal cropping was dominant. In 2006 the way in which farmers were funded changed and ownership of the largest farm in the catchment also changed, land use and mixed farming was brought back and dominated the landscape (Evans, 2010). Over 16 years (1991–2006) on average, though varying greatly from year to year, fields with rills (often affecting very small parts of fields) covered 26.1% of the c.10 km^2 catchment. In the following ten years, rilled fields covered on average only 7.3% of the catchment, a long term mean (1991–2016) of 18.9%. In one crop year

Table 6
Comparison of the key elements of the erosion data collection methods discussed in this paper.

	Volumetric survey	^{137}Cs technique	Integration of source fingerprinting, suspended sediment yields, landscape retention factors	Bounded field scale hydrological monitoring
Temporal scale	Single or multiple events (days)	From 1954 to sample collection year	Dependent on duration of sourcing study (typically up to 2 years), extrapolated suspended sediment yields are annual estimates	Dependent on duration of experiment. Summaries commonly produced by hydrological year
Processes represented	Rill and gully erosion by water	All soil redistribution processes, mainly including water erosion, wind erosion, tillage translocation, loss associated with the harvesting of root crops, etc.	Soil redistribution by water	Soil redistribution by water
Prior information required	Catchment areas for the rill networks or field area Bulk density of eroded soil	Local reference inventory Land use categories: permanent grassland or cultivated land	Mapping of grassland releasing mobilised sediment to river channel network for targeting source sampling, suspended sediment yield estimates, retention multiplier	
Fieldwork	Identify rills and then measure cross section areas along rill lengths to estimate rill volumes or map deposition areas and depths to estimate proxy of erosion volumes.	Coring soil generally up to 70 cm depth For grassland sites, sectioned cores from a reference site to establish relaxation depth For cultivated sites, sectioned cores from eroding part to estimate tillage depth, bulk density, etc.	Source material and target sediment sampling	Collection of suspended sediment concentration data using either turbidity sensors with conversion ratings or a combination of regular and storm period automatic water samples and gravimetric filtration, discharge data
Laboratory procedures	NA	Sample grinding if necessary to separate < 2 mm fractions for detection on Gamma detectors	Sieving of source samples to particle size fraction (e.g. < 63 μm) most representative of sediment samples and analysis of various fingerprint properties including geochemistry and FRN	Gravimetric filtration to convert turbidity time series to suspended sediment concentrations, or to use routine (e.g. daily) water samples to provide information on suspended sediment concentrations
Data processing	Sum up all rill volumes and divide the volume by estimated catchment areas (or field area) to derive erosion rates for the field. Convert volumetric estimate (m^3) to mass (t)	Using selected conversion models to numerically account for inventory dynamics at annual temporal steps.	Statistical analyses for source discrimination, numerical modelling with uncertainty analyses for apportionment, integration with suspended sediment yield ranges and landscape retention multiplier	Quality assurance of discharge time series. Suspended sediment load estimation combining discharge and suspended sediment concentration time series
Potential outputs	Event(s)-based or sampling period erosion rate for a field	Medium-term gross and net water erosion rates for individual sampling locations and erosion rates for a field	Annual grassland erosion estimates	Annual grassland erosion estimates

(1982/83) on silty erodible soils in Somerset, Colborne and Staines (1985) found that of 200 randomly stratified selected fields, 58 eroded (29%). In eastern Scotland (Watson and Evans, 2007), over 13 years Watson monitored 5244 fields of which only 109 (2.1%) eroded (range 0 to 6.6% in any one year). In a nearby area, only 2.7% of a total of 1375 fields monitored over a 6 year period (range 0.1–6.1%) experienced erosion. It may be that not all rilled fields in a locality were/are recorded, but because of the intensity of observation when surveying localities it is unlikely many were/are missed. Hence, field-based assessments of rill erosion show that only a (usually small) part of the landscape erodes as a result of rilling in any one year, thereby precluding linear extrapolation of measured erosion rates across the entire surface of any landscape under scrutiny. Most field monitoring does not assess sheet erosion as it leaves practically no evidence that it occurred, i.e. no incision or deposition, because the amounts moved are so small (Evans, 2017). Where there was evidence of splashed sand grains and 'lines' of deposited organic matter estimates showed that surface lowering was of the order of 0.06 mm. If a field suffers sheet erosion of 0.06 mm per event ($= 0.05\ \mathrm{t\ ha^{-1}}$), that is equivalent to $100 \times 0.05 = 5\ \mathrm{t\ km^{-2}}$, and wash must occur a number of times a year for it is known that runoff leaving obvious evidence of erosion can take place, for example, in winter cereals on average 6 times a year (Evans, 2017). Although field observations suggest that more fields are experiencing wash each year due to widespread soil compaction (Evans et al., 2016; Evans, 2017), the rates associated with sheet wash are low and so amount to little even if extrapolated across entire agricultural landscapes.

Compared with the traditional volumetric field survey method, the ^{137}Cs-based approach clearly represents an indirect technique of erosion assessment. The requirement for collecting deep soil cores (> 70 cm in some cases) makes its application in stony soils difficult because many cores need to be taken before the required depth is attained. Depending on the local soil reference inventory (Sutherland, 1996), and the sample counting times available in the context of sample numbers and laboratory capacity, uncertainty associated with radionuclide detection can be as high as 15%. Assay counting times for reasonable detection accuracy limit the number of samples that can be run and thus the number of sites that can be investigated. Post-processing of field and laboratory (gamma assay) data to obtain field-based soil erosion rates requires good understanding of the key processes involved, e.g. tillage erosion, in order to select and parameterise the most appropriate conversion models. Work by Zhang et al. (2015) noted the lack of systematic sensitivity and uncertainty analyses for evaluating the outputs of the ^{137}Cs soil erosion conversion models. In response to those criticisms, Zhang et al. (2015) assessed three widely-used conversion models with the results suggesting that soil redistribution estimates are extremely sensitive to ^{137}Cs reference and target sample inventories, but less sensitive to factors such as bulk density, tillage depth and the incorporation of a particle size correction factor in the models. Similarly, uncertainty analysis showed that spatial variabilities of ^{137}Cs inventories in both reference and target location sites are the major components of the total uncertainty associated with soil erosion estimates, followed by the particle size correction factor used in the conversion models. Additional factors such as tillage depth or bulk density were shown to make minor contributions to total uncertainty. Widely used conversion models from the International Atomic Energy Agency (IAEA) only recognise two principal land use categories (cultivated land and permanent grassland) and assume steady state erosion throughout the medium-term (last ~50 years) modelling period. These assumptions make it difficult to account for land use change or intra-annual variations of erosion intensity. The representation of multiple soil redistribution processes, including tillage translocation and soil loss in conjunction with harvesting equipment and tyre trafficking across the soil surface, has the potential to make the estimated erosion rates more realistic, but a lack of site specific information on the temporal dynamics of these processes over decades clearly poses challenges for generating realistic rates for soil erosion by water alone. Without explicit separation in conversion models of the processes involved, it must be remembered that ^{137}Cs-based estimates reflect the sum of all soil redistribution processes. The potential dominance of some additional processes including tillage erosion, and drain flow losses weaken the causal link between any change in ^{137}Cs inventories and surface driven water erosion processes, thereby undermining a fundamental assumption of the technique. As the most significant influx of ^{137}Cs occurred over the period between 1954 and 1963, the inventory change for agricultural land is more related to erosion intensities during that time period. With natural radioactive decay and a lack of new influx (apart from Chernobyl in 1986), radioactivity levels decrease according to the half-life of the nuclide and the application of ^{137}Cs becomes more difficult as activities decline and counting errors consequently increase. Over the very short time period of high atmospheric fallout from nuclear weapons testing and nuclear accidents, there is significant evidence in the agricultural literature that ^{137}Cs accumulated on and in grazed grasslands and that this resulted in high activities in animal dung, urine and milk as well as appearing in several biogeochemical cycles (e.g. Stewart et al., 1965; Desmet et al., 1990; Assimakopoulos et al., 1993; Greger, 2004). Indeed it is only since March 2012 that the UK Food Standards Agency years has allowed Welsh hill farmers to put their meat into the human food chain > 25 years after Chernobyl fallout affected the region (BBC news accessed 25/10/16 http://www.bbc.co.uk/news/uk-wales-17472698). Redistribution of ^{137}Cs by grazing animals and loss in dung, urine and milk are also not accounted for in the majority of FRN modelled erosion estimates and again could in part explain the higher estimates of net erosion using the ^{137}Cs approach. It is highly unlikely that ^{137}Cs was originally distributed evenly over the landscape as surely some must have been transported downslope in runoff or adsorbed by sediment already in transport during the storm. Thus, on some slopes the initial deposition of ^{137}Cs would be less than on slopes where runoff did not take place; a possible process highlighted by Dalgleish and Foster (1996) but generally ignored in FRN models.

The traditional volumetric approach for assessing soil erosion by water is not without flaws as it is dependent upon the visual identification of features on the ground and the actual measurement of rill/gully volumes. Uncertainty associated with the identification of small rills and the measurement of their cross section areas means that not all rills will necessarily be mapped and measured and inter-rill/sheet erosion is ignored. The determination of catchment areas for the rill networks may be problematic. This approach also assumes that the majority of eroded soil is represented by rill geometries measured after the erosive events. This is a fair assumption if erosion is assessed at the end of the growing season for farmers tend only to eradicate rills and gullies during their next ploughing and tillage operations. However, sometimes once rills are well-developed, they can provide an efficient delivery pathway for any erosion (wash) on inter-rill areas. Wash can, in turn, be accelerated because of the presence of a rill network, i.e. soil loss could occur without further significant expansion of the rill network as it has sufficient transport capacity to move sediment out of the eroding field by increasing structural connectivity sensu Fryirs et al. (2007). It is likely that such losses are small (Evans, 2006, 2017; Evans et al., 2016) and will add little to the estimates of losses by rill erosion.

The approach used to estimate the grassland soil erosion rates clearly also has limitations and uncertainties. The estimates are for the < 63 μm (silt and clay) fraction only, whereas the ^{137}Cs-based approach uses the < 2 mm fraction. Sediment source fingerprinting as an approach assumes that all key sources have been sampled, that the fingerprint properties used to distinguish and apportion key sediment sources including eroding grassland, have not undergone significant transformation during mobilisation and delivery to, and through, the river channel system and that the limited number of source and target sediment samples collected from any catchment are representative of the tracer characteristics of those materials (cf. Foster and Lees, 2000;

Collins et al., 2017). The suspended sediment yield ranges applied to the grassland landscapes used in this work are based on a typology which extrapolates high and medium quality suspended sediment load data for over 100 catchments in the absence of a national intensive monitoring network (Walling et al., 2007; Cooper et al., 2008). Nonetheless, those high and medium quality empirical data are characterised by uncertainties on account of their limited temporal duration (only ≥ 2 years), unrepresentative spatial coverage largely driven by localised intensive research projects and substantial variations in the techniques and procedures used for data collection and processing (Walling et al., 2007). Additional uncertainties are also associated with the selection of generic multipliers for correcting the suspended sediment yield ranges to represent the impact of upstream long-term landscape sediment retention. Here, however, existing published work for the UK has suggested that long-term sediment storage on river floodplains can represent ~40–50% of the annual suspended sediment load delivered to the river channel system in large (800– > 3000 km^2) river basins (e.g. Walling et al., 1998). Given that the sub-catchments used to estimate grassland soil erosion rates were much smaller (mean area of 137 km^2; range 1 km^2 to > 1000 km^2), the multipliers were scaled down in recognition that the long-term sediment storage term is likely to be smaller in the sediment budgets of these landscapes. The comparison for grassland compares ^{137}Cs-based estimates reflecting the sum of all redistribution processes with data generated using monitoring techniques for the impacts of water erosion. Although some of the soil redistribution processes impacting arable land (e.g. root crop harvesting) will not be relevant here, vehicle and livestock trafficking are both observed to result in soil being redistributed out of gateways in grass fields independent of soil erosion.

The overall range in the gross erosion estimates for grassland (0.17–1.39 t ha^{-1} yr^{-1}) derived using the data integration approach described herein is consistent with that reported by the few studies working on grass at plot or field scales including, for example, data from the North Wyke Farm Platform in SW England (Orr et al., 2016). Table 7 presents the baseline soil loss estimates from the 15 sub-catchments (Fig. 6) for the period 1/10/2012–30/09/2013. These rates range from 0.06 t ha^{-1} yr^{-1} to 0.21 t ha^{-1} yr^{-1}. The magnitude of these grassland soil erosion rates for medium soils using the Cross Compliance textural classification scheme is more comparable to those estimated using the combination of sediment source tracing, suspended sediment yield and landscape retention factors (0.24–0.58 t ha^{-1} yr^{-1}) than those based on the ^{137}Cs (1.79–2.77 t ha^{-1} yr^{-1}) approach (Table 5). Furthermore, given that the NWFP adheres strictly to grazing best management practices, the soil loss rates could be expected to be lower than those estimates for other grassland landscapes across England and Wales where the implementation of such best practices will be more variable among the farms therein. Similarly, and accepting the problems of extrapolation between different spatial scales and the inherent limitations and uncertainties associated with different measurement techniques, the grassland erosion rates based on data integration (Table 5) are more consistent with those (< 0.08–0.23 t ha^{-1} yr^{-1}) reported at plot scale by Fullen (1992, 1998) and catchment scale (0.54–1.21 t ha^{-1} yr^{-1}) by Bilotta et al. (2010). The ^{137}Cs-based ranges far exceed these generated using more conventional procedures, especially in view of the fact that the catchment scale estimates reported by Bilotta et al. (2010) were not corrected for sediment source apportionment and thereby include inputs from sources other than eroding agricultural soils, including eroding channel banks for example.

Table 7
Baseline grassland soil erosion rates from the North Wyke Farm Platform for the period 1/10/2012–30/09/2013.

Sub-catchment (number relates to Fig. 6)	Erosion rate t ha^{-1} yr^{-1} [a]
1	0.10
2	0.10
3	0.15
4	0.11
5	0.16
6	0.06
7	0.21
8	0.12
9	0.08
10	0.08
11	0.07
12	0.09
13	0.13
14	0.17
15	0.16

[a] Using a suspended sediment concentration/turbidity rating relationship (Peukert et al., 2014).

The comparisons presented herein imply that lowland arable and grassland soil erosion rates estimated using ^{137}Cs are typically overestimates, but by varying and inconsistent amounts. Estimates of suspended sediment loads transported by rivers and of sediment inputs derived from sources other than farmland such as river channels, damaged road verges and urban areas (e.g. Collins et al., 2009, 2012, 2013; Evans, 2012; Zhang et al., 2014) are other sources of information which may help elucidate which of the approaches to estimating erosion generates the more realistic data. Estimates of soil erosion made using traditional field surveys, and those from sediment delivery to watercourses (net erosion, see above discussion of method for grassland erosion estimates), suggest that conventional survey-based assessments of soil erosion in lowland England can, as one would expect on the basis of current national scale understanding of sediment source apportionment (Collins and Anthony, 2008a, b; Collins et al., 2009; Zhang et al., 2014) explain a good part of the contemporary suspended sediment loads in rivers (Evans, 2006) and are hence of the right order of magnitude. Surface wash soil losses of 0.1–0.3 t ha^{-1} yr^{-1} (10–30 t km^{-2} yr^{-1}) close the 'gap' between field-based estimates of soil loss and measured suspended sediment loads in rivers still further.

We do not intend to repeat the detailed points raised by Parsons and Foster (2011) in questioning the validity of soil erosion rates generated using the ^{137}Cs-based approach and in their reply to Mabit et al. (2013) (Parsons and Foster, 2013). There, the focus was on the lack of testing of fundamental assumptions of the technique used for converting ^{137}Cs inventories into erosion rates whereas our focus here is on a comparison of traditional survey-based and FRN erosion rates. However, we would urge erosion researchers and modellers using FRNs to consult and use the public health, ecological and agricultural literature relating to the loss and/or redistribution of ^{137}Cs by processes other than physical soil erosion (see above).

Overall, it seems that models to predict erosion rate and risk, including those used to convert the distribution of ^{137}Cs contents in topsoils to erosion rates, do give estimates of erosion that are too high when compared with survey-based estimates. We acknowledge that survey-based estimates may not account for all soil particles transported, since for example, surface wash is difficult to estimate/account for, but in Britain is likely to move very small amounts of material and that which is fine-textured with little impact on erosion rate. Nevertheless, it seems to us that many models are an unsound basis for extrapolating results across a landscape using, for example, GIS, for such models give rainfall, soil and slope factors too much weighting (Evans, 2013; Evans and Boardman, 2016a, b) compared, for example, to land use. Thus, it is accepted (University of Exeter, 2008, pp. 13) that grassland (pasture) erodes less than arable land, but it is grassland areas that have the highest rates of erosion (Figs. 7 and 8).

5. Conclusions

A review and comparison has been presented of assessments of soil erosion rates on both arable and grassland across lowland agricultural

England and Wales using different approaches. The differing process representation of measurement and monitoring techniques must always be borne in mind with such comparisons. Rates estimated using ^{137}Cs conversion models are often, virtually consistently, higher or much higher than other survey-based estimates. The ^{137}Cs-based approach continues to be promoted vigorously to assess soil erosion without being adequately validated across scales; calibration/validation using other models or estimates based on FRN data is not sufficiently robust. Not forgetting the problems of assessing erosion in the field with visual surveys, it would be expected that ^{137}Cs-based and survey-based assessments should be in better agreement. It would appear, however, that the ^{137}Cs-based approach overstates the severity of the water-related erosion problem in England and Wales and may well do so elsewhere. The comparisons herein suggest that the technique needs to be validated more rigorously and conversion models re-calibrated on the basis of carefully designed field-based assessments and data collection for processes such as tillage erosion and soil loss in conjunction with crop harvesting. It is only in Britain that sufficient survey-based data has been assembled to enable the comparisons presented in this paper. Until the ^{137}Cs-based approach is refined, its results should be treated with great caution in the context of characterising soil erosion by water, despite its adoption and promotion by many international agencies (e.g. IAEA). This is especially true in areas where intensive mechanised tillage has been practised. ^{137}Cs-based estimates of soil erosion should be interpreted in the context of those generated using alternative independent methods (Golosov et al., 2017). Currently, ^{137}Cs-based estimates do not reflect soil erosion across lowland England and Wales by water alone, but instead, the sum of various additional processes, which are likely to include tillage translocation, soil loss associated with crop harvesting, ^{137}Cs fallout runoff independent of attachment to soil particles, wind erosion and ^{137}Cs assimilation pathways. There remains a need to incorporate temporal dynamics for key factors controlling erosion, when interpreting ^{137}Cs-based erosion estimates, given the pronounced sensitivity of measured ^{137}Cs inventories to controlling factors during the period of peak fallout rather than under present day conditions.

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